UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

APR 05 2010

OFFICE OF CONGRESSIONAL AND
INTERGOVERNMENTAL RELATIONS

The Honorable Dennis J. Kucinich
Chairman
Subcommittee on Domestic Policy
Committee on Oversight and Government Reform
Washington, D.C. 20515

Dear Mr. Chairman:

Thank you for your letter dated January 15, 2010, to the United States Environmental Protection Agency (EPA), concerning dental fillings as a source of mercury pollution, and particularly, the atmospheric emissions of mercury related to dental use of mercury. I share your desire to reduce mercury in the environment and appreciate your continued interest in this particular aspect of the problem.

In this response letter, I would like to describe what the Agency is currently doing to address mercury in the environment and in particular what we are doing to address mercury emission factors that are of particular interest to you and the members of the Domestic Policy Subcommittee. We will keep you and the Subcommittee apprised of our efforts as we continue our work on updating these factors.

EPA is addressing the issue of mercury in the environment broadly

Addressing domestic and global mercury concerns is a top priority of the Agency. EPA recognizes that pollution from all sources of mercury is a serious concern to human health and the environment. Mercury is well-documented as a toxic, environmentally persistent substance that demonstrates the ability to bioaccumulate and to be atmospherically transported on a local, regional, and global scale. In addition, mercury can be environmentally transformed into methylmercury which biomagnifies and is highly toxic. As such, EPA is currently working to address mercury in various media, including atmospheric emissions, aquatic discharges, solid and hazardous wastes, and manufacture of mercury containing products.

EPA's Office of Prevention, Pesticides, and Toxic Substances published EPA's *Roadmap for Mercury* in July 2006, as a clear statement of EPA's commitment to address mercury in the environment for the public and its stakeholders. As summarized in EPA's *Roadmap for Mercury,* EPA has a long history of addressing mercury. Between 1970 and 1990, there was increasing scientific evidence of mercury pollution problems in the United

States. This evidence included serious health impacts from low-level exposure to mercury, increasing amounts of mercury pollution in air, water, and waste, increasing human exposure to mercury through fish consumption, and the enormous problem of local, regional, national and global transport of mercury pollution.

To address a number of these problems, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) was enacted in 1980, providing authority to clean up past waste. Following that, the Clean Air Act of 1990 provided authority to address mercury air emissions at key sources. In 1995, the Great Lakes Water Quality Guidance provided water quality standards to address mercury in that media. The 1996 Battery Act followed soon after, introducing mercury regulations in the area of products and manufacturing. Since that time, EPA has helped to shape and enforce mercury-pertinent provisions of environmental statutes, as well as spearhead domestic initiatives to study and reduce the harmful effects of mercury in environment.

In one of our most important efforts to address mercury, EPA's Office of Air and Radiation is developing a rule to control toxic air emissions including mercury from power plants. Since 2005, some power plants have been able to achieve 90 percent or greater reductions – both in demonstration projects and actual commercial operation. EPA is currently soliciting information from the electric utility industry that will provide actual operating experience at a number of facilities and provide the basis for requiring maximum achievable control technologies.

Because a major portion of mercury in the U.S. environment comes from sources outside the U.S., EPA has supported and led efforts to address mercury in multilateral treaties. Under the United Nations Economic Convention for Europe Convention for Long-Range Transboundary Air Pollution Protocol (LRTAP), EPA supported negotiations in 1998 of the LRTAP Heavy Metals Protocol which was subsequently ratified by the U.S. This Protocol includes limitations and deadlines for emissions for new and existing stationary sources as well as mandatory mercury concentrations limits for certain types of batteries. EPA also supported international efforts to address mercury reduction efforts in 2001-2002 by contributing scientific expertise, leadership, and funding to the United Nations Environment Program's (UNEP's) 2002 *Global Mercury* Assessment. Subsequently, the UNEP Governing Council in 2003 mandated the Global Mercury Program. In 2005, the UNEP Global Mercury Partnership was launched with significant support from the United States, and from EPA in particular. We provide a significant amount of the funding, lead two of the partnership areas, and support a wide range of projects across the Partnership as a whole. Since the 2009 decision by the UNEP Governing Council to negotiate a global mercury treaty by 2013, EPA has provided technical expertise and policy guidance throughout the process and we will contribute significantly as members of the U.S. negotiating team.

Currently, EPA is following through on the commitments made in the *Roapmap for Mercury*, including the use of regulatory authority under the Toxic Substances Control Act (TSCA) to reduce or eliminate the use of elemental mercury in certain products, including both existing and obsolete products. EPA also provides strong leadership to the UNEP

Global Mercury Partnership and technical expertise to the LRTAP Task Force for Heavy Metals. In the future, EPA looks forward to participating in the Intergovernmental Negotiating Committee that will begin the process of developing legally binding instrument to address global mercury issues. Those actions, as well as the convening of the Commodity Mercury Stakeholder Workgroup to inform the development and to support the implementation of the Mercury Export Ban of 2008, demonstrate a solid commitment to address mercury-related issues.

Notwithstanding these successes, the mercury challenge is multi-faceted and without simple solutions. We continue to provide global leadership through our ongoing efforts to address domestic and global mercury demand as a means to reduce risks to human health and the environment.

Assessing the amount of mercury pollution attributable to dental mercury

Based on National Emission Inventory data, EPA estimated unintentional air emissions of mercury from source categories related to waste combustion and incineration. At this time, there are limited atmospheric emissions estimates for mercury-containing product categories, including dental amalgam. For example, available 1990 and 2005 NEI data contain mercury emissions estimates for "Other (numerous very small sources)," but does not quantify emissions by specific mercury-containing items or sub-categories. In 2005, estimated emissions from the "Other (numerous very small sources)" category were 16.9 tons (16.5 percent) of 102.7 tons total U.S. anthropogenic mercury air emissions. Similar global estimates do not account for emissions specifically linked to dental amalgam. However, in 2005, the total estimate of global mercury emissions was 1,930 metric tons. Categories applicable to dental amalgam, "Cremation" (1 percent) and "Waste incineration and other" (7 percent) accounted for approximately 154 metric tons.

Dental amalgam use in the U.S. is gradually declining because the incidence of dental decay is decreasing and because improved substitute materials are now available for certain applications. In 2001 the total amount of dental mercury sold annually in the U.S. was 27.9 metric tons. Dental mercury use decreased slightly to 27.6 metric tons in 2003, but by 2007 had declined steeply to 14.9 metric tons, a decrease of 46 percent over three years. As of 2007, dental amalgam remained the second largest category of mercury use in all products (after switches and relays), constituting about 24 percent of mercury in all products sold in the U.S. that year. (Interstate Mercury Education and Reduction Clearinghouse fact sheet: *Mercury Use in Dental Amalgam*, January 2010)

Unlike other mercury-added products, there are no state restrictions on the sale or distribution of dental amalgam. At the national level, dental amalgam is regulated as a medical device by the Food and Drug Administration (FDA) under the Federal Food, Drug and Cosmetic Act. Increased consumer awareness of the presence of mercury in amalgam fillings may drive future declines in mercury amalgam use; however, the expense of non-mercury fillings can affect patients' preferences for dental restorative materials.

It should be noted that it is difficult to translate dental amalgam use and consumption data to estimates of mercury pollution in the environment.

Efforts to assess mercury releases from dental amalgam

As your letter suggests, mercury from dental amalgam is a source of controllable mercury released to the environment and likely will remain a significant concern into the future. Mercury from dental amalgam is released to the environment through three primary pathways: in wastewater, as solid waste, and through cremation of bodies containing dental amalgam.

Mercury Amalgam in Wastewater

The majority of dental mercury amalgam waste is currently discharged from dental offices to wastewater treatment systems where it usually settles out in sewage sludge that is then incinerated, heat treated, landfilled, and/or land applied as biosolids (also known as "sludge"). In 2008, EPA estimates there are approximately 160,000 dentists working in 120,000 dental offices that use or remove amalgam in the United States, almost all of which discharge their wastewater exclusively to municipal sewage treatment plants. Most dental offices currently use some type of basic filtration system to reduce the amount of mercury solids passing into the sewer system. However, the adoption of best management practices and the installation of amalgam separators, which generally have a removal efficiency of 95 percent, have been shown to reduce discharges even further. In October 2007, the American Dental Association adopted new Best Management Practices for Amalgam Waste that recommends the use of dental amalgam separators and the recycling of captured amalgam solid waste.

Mercury Amalgam in Solid Waste

Waste amalgam solids that are improperly disposed in medical waste ("red bag") containers will be either incinerated or autoclaved, thus causing volatilized mercury to escape into the environment. Mercury amalgam also accumulates on consumable dental supplies, such as cotton swabs and gauze, and these materials are usually disposed in the regular trash. In local areas where trash is incinerated, the mercury in this trash can be released via air emissions. To avoid such mercury air emissions, dental offices should properly dispose of captured amalgam solid waste by sending it to a dental waste recycler.

Mercury Emissions from Crematoria

Dental amalgam also contributes to mercury emissions through the cremation of bodies containing dental amalgam. A mercury flow worksheet developed for EPA Region 5 estimated that in the United States in 2005 almost 3,000 kilograms (6,613 lbs.) of mercury were released to the environment from crematoria. There remains a lack of good empirical data on the magnitude of mercury emissions from crematoria. At this time, no federal or state regulations restrict mercury emissions from crematoria.

EPA is working to update its estimates of pollution linked to dental amalgam

EPA plans to update mercury emission factors

EPA is developing a new emissions factors program designed to produce high quality emission factors by the end of next year. Once our new emissions factors development process is complete, any emissions sources (including wastewater sludge incineration and crematoria) that provide electronic source test plans to our Internet-based database will enable us to generate emissions factors for all pollutants, including mercury, using the most current data available.

EPA plans to revise regulations for sewage sludge incinerators

EPA plans to propose and finalize a rule setting new source performance standards and emission guidelines for sewage sludge incinerators by December 2010. This will include a standard for mercury emissions.

EPA recognizes the importance of revising and improving emissions data. Part of the sewage sludge incinerator (SSI) rule effort involves data collection and improving the accuracy of emissions data. EPA is currently using CAA Section 114 authority to collect information from active units. EPA's two part information collection request was sent on October 23, 2009, and EPA anticipates receiving data by March 31, 2010. As part of this information collection effort, EPA is collecting new and existing mercury data for the revised standard.

The Other Solid Waste Incinerator (OSWI) rule originally included new source performance standards and emission guidelines for 2 categories -- very small municipal waste combustion and institutional waste incineration units on December 16, 2005 (70 FR 74870). The rule was challenged by environmental groups in court and in response, the Agency plans to propose, accept notice and comment, and finalize new regulations in the near future. Both the SSI and OSWI rules will include a standard for mercury, eight other pollutants, and set an opacity standard.

EPA does not plan to regulate human crematoria at this time

At this time, however, EPA does not plan to regulate human crematoria. In the preamble to the final OSWI rule (70 FR 74870), EPA concluded that the human body is not solid waste. Since the law requires EPA to establish regulations for solid waste incineration units, EPA concluded that human crematories were not solid waste incinerators, and, therefore, it was not appropriate to regulate them under Clean Air Act Section 129. EPA also stated that if in the future we conclude that human crematories should be regulated, other Clean Air Act authorities or state authorities could be used.

EPA encourages dental amalgam capture and recycling

EPA's Office of Solid Waste and Emergency Response (OSWER) is currently partnering with Marquette University to finalize a teaching module to encourage dental amalgam capture and recycling, targeting student and practicing dentists. This effort has focused on addressing the capture of dental amalgam waste and dental amalgam recycling rather than emissions. Teaching dentists responsible dentistry will help prevent excess amalgam waste from entering the waste stream and ending up in sewage sludge. The teaching module is currently undergoing review, and will be posted to the EPA website upon final approval.

In December 2008, EPA's Office of Water signed a Memorandum of Understanding (MOU) with the American Dental Association (ADA) and the National Association of Clean Water Agencies (NACWA) to establish and monitor the effectiveness of a Voluntary Dental Amalgam Discharge Reduction Program. The purpose of the MOU is to have dental offices install and properly maintain amalgam separators, and recycle the collected amalgam waste. Early in 2009, the MOU parties agreed on a method for estimating the baseline and the data to be collected and analyzed. In June 2009, EPA received the baseline report, which included highlights of ADA's survey results on installation rates of separators across the country. We also expanded our coordination with stakeholders to include the Quicksilver Caucus, a coalition of State environmental associations who are concerned with mercury discharges, and also with the Mercury Policy Project, which is an NGO focused on reducing mercury from all sources. As all the parties continue to coordinate on next steps, we look forward to narrowing the performance goals and agreeing on best approaches to encourage installation of separators.

Again, thank you for your letter and your continued interest in this issue. If you have any questions, please contact me, or your staff may contact Tom Dickerson of my office at (202) 564-3638.

Sincerely,

Arvin Ganesan
Deputy Associate Administrator
for Congressional Affairs

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Atmospheric Chemistry and Physics

Mercury emission from crematories in Japan

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Abstract. Anthropogenic sources of mercury emissions have a significant impact on global pollution. Therefore, finding uncharacterised sources and assessing the emissions from these sources are important. However, limited data are available worldwide on mercury emissions from crematories. In Japan, 99.9% of dead bodies are cremated, which is the highest percentage in the world, and more than 1600 crematories are in operation. We thus focused on emissions from crematories in Japan. The number of targeted facilities was seven, and we used continuous emission monitoring to measure the mercury concentrations and investigate mercury behaviour. The total mercury concentrations in stack gases were a few $\mu g/Nm^3$ (normal cubic meters). Considering the time profile of mercury and its species in cremations, the findings confirmed that the mercury in stack gas originated from dental amalgam. The amount of mercury emissions was calculated using the total concentration and gas flow rate. Furthermore, the annual amount of mercury emission from crematories in Japan was estimated by using the total number of corpses. The emission amount was considerably lower than that estimated in the United Kingdom. From statistical analyses on population demographics and measurements, future total emissions from crematories were also predicted. As a result, the amount of mercury emitted by crematories will likely increase by 2.6-fold from 2007 to 2037.

1 Introduction

Given in its high volatility, mercury is emitted into the atmosphere from both anthropogenic and natural sources. Subsequently, it enters oceans, lakes, and rivers from the atmosphere directly or from deposits in surrounding basins, even when no specific source of the element is present (Fitzgerald et al., 1998). Some of the inorganic mercury in water is converted into organic mercury, which can be very toxic and is subject to biological accumulation. Consequently, the emission of mercury is of great concern. The United Nations Environment Programme (UNEP) is conducting studies with the goal of a worldwide reduction in mercury (UNEP Chemicals, 2002). These programmes require estimates of the national emissions of mercury from major sources in each country. A report by the Arctic Monitoring and Assessment Programme and UNEP Chemicals (2008), lists mercury emissions from crematories because dental amalgam contains a significant amount of mercury.

According to the Ministry of Health, Labour and Welfare (MHLW) of Japan (2008a), 99.9% of all bodies (1 108 334) were cremated in about 1600 facilities in 2007; this percentage is the highest in the world. With demographic changes, the number of deaths is increasing, and the number of cremations will also increase (MHLW, 2008b). For religious reasons, mercury emissions from crematories in Japan are not regulated by the Air Pollution Control Act or the Waste Management and Public Cleansing Act. However, examining mercury emissions from crematories is needed to determine their environmental impact and to take measures to reduce or monitor them if necessary. Anthropogenic sources of mercury emissions have a significant impact on global pollution. Therefore, finding uncharacterised sources and evaluating the emissions from them are important. However, only limited data on mercury emissions from crematories are available in the literature. According to the Department for Environment, Food and Rural Affairs (DEFRA) in the United Kingdom (2004), mercury emissions in 2020 will be 1.67 times those in 1995 and will peak in 2035. It will contribute 11–35% of the total mercury emissions in the United Kingdom in 2020. In Sweden, it was estimated to be the third highest contributor of all anthropogenic sources of mercury (Hogland, 1994). Emissions from crematories are also very likely to have a significant impact in Japan.

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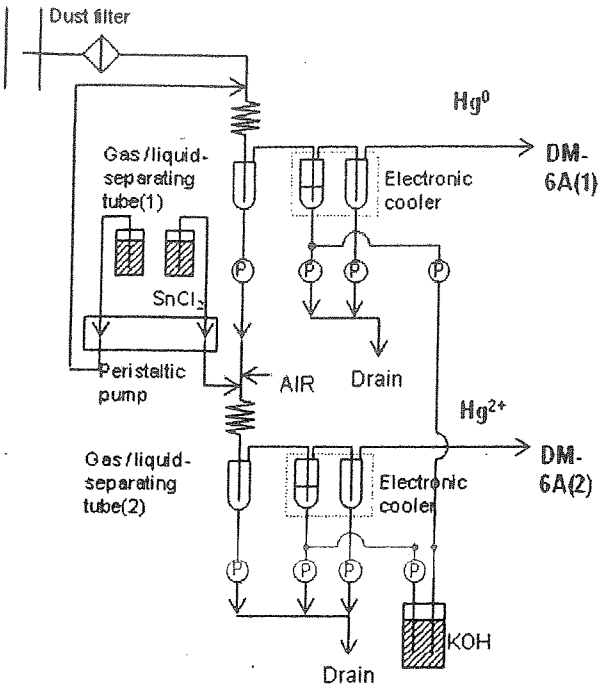

Fig. 1. The pretreatment section of the speciation CEM.

The purposes of this research were to measure actual emission levels, estimate emissions from crematories in Japan using measurement data and clarify the behaviour of mercury in crematory flue gas with the goal of predicting the environmental fate of the mercury in the surrounding area. Finally, future trends in emissions were estimated.

2 Materials and methods

2.1 Facilities

Table 1 shows the configuration of the seven crematories and the sampling conditions. In Japan, to prevent dioxin emissions from crematories, a guideline was implemented in 2000 requiring installation of air pollution control devices (APCDs) in newly constructed facilities. Although the removal efficiency of mercury by APCDs is beyond the scope of this research, APCDs do have a significant impact on the mercury concentration in final flue gas. Thus, we selected three crematories (Facilities No. 1, 2, and 3) that were constructed after 2000. In these particular facilities, bag filters were used as dust collectors and advanced APCDs had been installed. Additionally, Facility No. 7 also has a bag filter system, although it was constructed in 1998. Conversely, Facilities No. 5 and 6 were not equipped with even a dust collector. All crematories had a series of one secondary combustion chamber to one main combustion chamber, and in all cases, flue gases were cooled by air ejectors. Facility No. 1 used a heat exchanger for flue gas cooling. Natural gas and oil were used as auxiliary fuel in four (Facilities No. 1, 2, 4, and 7) and three (Facilities No. 3, 5, and 6) of the crematories, respectively.

The origin of the mercury is believed to be dental amalgam (Mills, 1990). Since there is a large difference in mercury emissions between a body with or without mercury amalgam, we conducted many measurements at two facilities (No. 6 and 7) to determine an accurate average mercury concentration. In other crematories, flue gas was sampled twice for each crematory. Since bodies are cremated individually, flue gas was sampled throughout a cremation, from ignition of the secondary burner to extinction of the main burner.

2.2 Mercury emission monitoring

Knowing the species of mercury in stack gas will contribute to a better understanding of the environmental fate of mercury. The mercury concentrations in stack gas were monitored continuously using a speciation mercury continuous emission monitor (CEM, MS1A+DM-6A; Nippon Instruments). This device was developed by Nippon Instruments and the Central Research Institute of the Electric Power Industry in Japan (Chua et al., 2003). The pretreatment section of the speciation CEM is shown in Fig. 1.

An aqueous solution of 1 mol/L potassium chloride and flue gases that had been passed through the dust filter is mixed with a gas/liquid contact coil to transfer oxidised mercury (Hg^{2+}) into the liquid phase. The gas flow rate is 0.5 L/min, and the gas and solution are separated in the gas/liquid separating tube. Elemental mercury (Hg^0) in the gas phase is washed with 1 mol/L potassium hydroxide to remove any acid gas, and the excess moisture in the gas is condensed with an electric cooler. Then, the gas including Hg^0 is fed into the first detection device, which is an atomic absorption mercury analyser (DM-6A(1)). The aqueous solution in the gas/liquid separating tube is directed into another gas/liquid contact coil and then mixed with 1% stannous chloride and 10% sulphuric acid. In this process, the Hg^{2+} in solution is reduced, liberating gaseous Hg^0, which passes through another gas/liquid separating tube before it is fed into the second detection device (DM-6A(2)) to quantify the Hg^{2+}. Here, the gas flow rate must be the identical to that in line DM-6A(1).

Given this speciation, CEM was developed based on the Ontario Hydro method, which is used to determine the elemental, oxidised, particle-bound and total mercury emissions from coal-fired stationary sources (ASTM D6784-02, 2008); it was compared with the Ontario Hydro method periodically and showed an excellent correlation for mercury concentrations ranging from 0 to 100 $\mu g/Nm^3$ in a municipal solid waste incinerator (Chua et al., 2003). The detection limit of this device is 0.1 $\mu g/Nm^3$.

Some flue gas obtained at Facility No. 7 was simultaneously sampled using an absorption method based on Japanese Industrial standard K0222. Flue gas was passed through

Table 1. The configurations and mercury concentration of seven crematories.

Facility No.	1		2		3		4		5		6	7
Dust collector	Bag filter		Bag filter		Bag filter		Electrostatic precipitator		–		–	Bag filter
Advanced APCD	Catalytic reactor		Catalytic reactor		Activated carbon filter		–		–		–	–
Flue gas cooling device	Heat exchanger +air ejecor		Air ejector		Air ejector		Air ejector		Air ejector		Air ejector	Air ejector
Ventilation	Induced		Induced		Induced		Induced		Induced		Induced	Induced
The number of secondary chambers connected to flue gas treatment line	2		2		2		3		2		1	2
Fuel	Natural gas		Natural gas		Kerosene		Natural gas		Kerosene		Kerosene	Natural gas
Experimental No.	1	2	1	2	1	2	1	2	1	2	44	33
Cremation time	58	57	66	59	48	68	64	45	71	66	84 (60–107)	57 (47–75)
Age	64	75	91	79	98	85	66	80	84	65	81 (52–99)	77 (29–101)
Sex	female	female	female	female	female	female	male	female	female	male	male(24), female(20)	male(22), female(11)
Hg concentraion ($\mu g/Nm^3$)	0.2	0.9	0.3	0.4	0.4	3	30.3	1.4	2.8	0.3	3.0 (0.2–82.7)	4.3 (0.8–25.2)
Hg^0 concentraion ($\mu g/Nm^3$)	0.1	0.5	0.2	0.1	0.3	2.7	30.1	1.3	2.7	0.2	2.6 (0.1–81.2)	2.2 (0.0–23.6)
Hg^{2+} concentraion ($\mu g/Nm^3$)	0.1	0.4	0.1	0.3	0.1	0.3	0.3	0.1	0.1	0.1	0.5 (0.1–2.7)	2.1 (0.7–8.1)

a glass filter and bubbled through a sulphuric acid solution with potassium permanganate ($KMnO_4$). Mercury absorbed in the solution was measured using a frameless reduction vapourisation atomic absorption mercury analyser (RA-2; Nippon Instruments). The sulphuric acid solution with $KMnO_4$ oxidises organic or inorganic mercury into bivalent mercury ions, and using stannous chloride as a reducing agent, bubbling changes ions into mercury vapour. This mercury vapour is then directed to an absorption cell and atomic absorption was measured at a wavelength of 253.7 nm to determine the quantity of mercury.

Carbon monoxide (CO), oxygen (O_2), carbon dioxide (CO_2), nitrogen oxide (NO_x), and sulphur dioxide (SO_2) concentrations were also monitored using continuous emission monitors (CGT-7000 for CO, NOA-7000 for O_2 and NO_x, SOA-7000 for SO_2; Shimadzu Co. Ltd.). In crematory No. 7, the HCl concentration was measured manually based on Japanese Industrial Standard K0107.

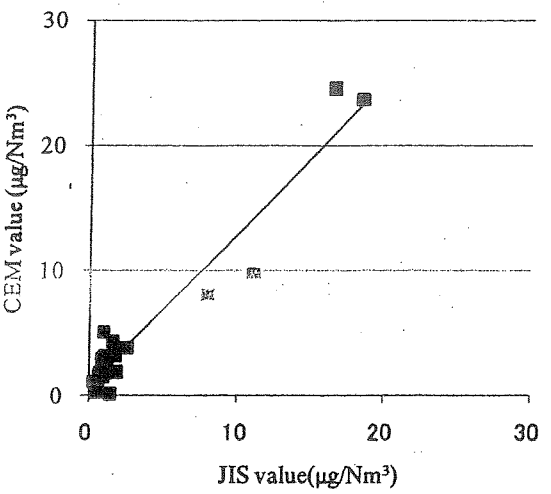

Fig. 2. The relationship between the JIS and CEM methods.

3 Results and discussion

3.1 Mercury concentration in crematory flue gas

First, we show the relationship between the JIS and CEM methods in Fig. 2. As a result, the regression equation is as follows:

$$CEM = 1.23(JIS) + 0.687 (R^2 = 0.93), \quad (1)$$

where CEM = CEM value ($\mu g/Nm^3$); JIS = JIS value ($\mu g/Nm^3$).

Although the CEM values were slightly higher than the JIS values, the correlation coefficient was considered to be sufficiently high to determine trends in mercury emissions.

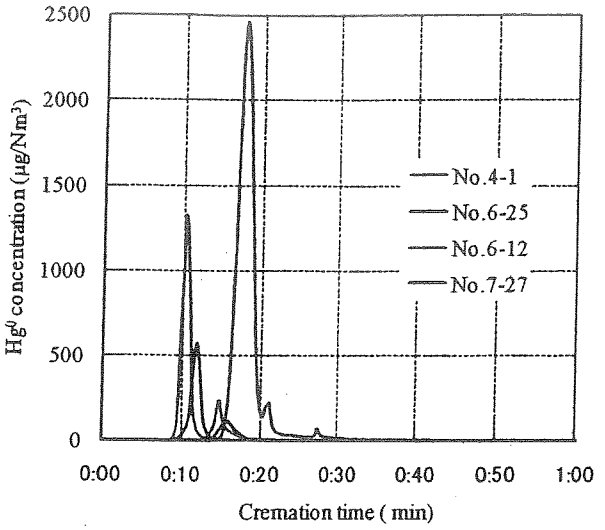

Fig. 3. Temporal changes in the elemental mercury (Hg^0) concentration in crematory flue gas.

When using the CEM value, we may need to be aware of some degree of overestimation.

The total averaged mercury concentration in stack gas was 3.6 μg/Nm^3, which consisted of Hg^0 (2.6 μg/Nm^3) and Hg^{2+} (1.1 μg/Nm^3). Hg concentrations ranged from 0.2 to 82.7 μg/Nm^3. When the concentration was normalised by 12% O_2 to compare the concentration to municipal solid waste incinerator flue gas, the total averaged mercury concentration was 17.8 μg/Nm^3, which was higher than that in stack gas of a municipal solid waste incinerator (Takaoka et al., 2002). This is because the O_2 concentration is so high (15.8–20.8%) that the concentration normalised by 12% O_2 becomes high. The mercury concentrations for the facilities are shown in Table 1. The average mercury concentration by facility ranged from 0.4 to 15.9 μg/Nm^3. This difference is not caused by the structure of each crematory, including APCDs, but depends on whether the body contains mercury amalgam.

Mercury concentrations in 22 crematory flue gas samples at three crematories, A, B, and C, were measured in the United Kingdom (Edwards, 2001). According to the reports, the average mercury concentrations normalised by 11% O_2 were 690, 880 and 430 μg/Nm^3 at crematory A, B and C, respectively. The mercury concentration in flue gas is influenced by the volume of exhaust gas per cremation. The average flue gas volume varied considerably by crematory; indeed, the range was 3250–14600 Nm^3/h. The average volume of exhaust gas in a cremation in Japan is 3–10 times larger than that in the United Kingdom because the flue gas was cooled by air dilution using an air ejector in Japan. However, this means that the mercury concentration in the UK exhaust gas is very high by comparison. Hogland (1994) reported the mercury concentration from a crematory in Lund, Sweden. Although the volume of exhaust gas was about 1/5–1/20 of that in Japan, the maximum concentration achieved was 60 000 μg/Nm^3, which is extremely high. From comparisons with data for other countries, we suggest that the average mercury quantity emitted in Japan is low.

The contribution of Hg^0 to the total mercury was 70%, which is relatively high. Hg^0 concentrations ranged from 0.1 to 81.2 μg/Nm^3, whereas Hg^{2+} concentrations ranged from 0.1 to 8.1 μg/Nm^3. A higher peak was observed only in Hg^0.

The chemical form of mercury in flue gas is known to be influenced by the gas composition, especially the presence of halogen compounds (Takaoka, 2005). HCl was measured in Facility No. 7, and its concentration was found to range from 2 to 13 mg/Nm^3. Even when the HCl was removed by a bag filter (this facility did not use alkaline reagents for acid gas removal), the concentration was very low compared with that in the municipal solid waste incinerator. To check the validity of the mercury form from the viewpoint of thermodynamics, the stable form of mercury was calculated under the following conditions: 19.6% O_2, 10 ppm HCl and 4% H_2O using Fact sage 5.0. As a result, $HgCl_2$ was found to be stable at 200 °C and Hg^0 to be dominant at temperatures above 600 °C. Because Hg^0 was dominant in flue gas based on measurement results, there was a difference in the chemical form between the results of thermodynamics and measurements. It might be cleared by measurement in upstream flue gas before bag filter.

3.2 Temporal change in mercury concentration

Two patterns in changes of mercury concentrations were observed. The first pattern is shown in Fig. 3. In some samples, a large peak in Hg^0 appeared at 10–20 min; two samples exceeded 1000 μg/Nm^3. This period indicates the burning of the cephalic part of the corpse. Considering the behaviour of mercury in cremations, the findings confirmed that the mercury in stack gas originated from the mercury in dental amalgam. This behaviour was previously reported by Hogland (1994). During this period, peaks were observed in 36 samples.

In the other pattern, no distinct peak of Hg^0 was detected in the remaining 51 samples. In some samples, a small peak of Hg^{2+} was noted. Various internal organs of the human body contain mercury, and the mercury quantity in an adult is estimated to be more than 3.3 mg/body, apart from dental amalgam (The Chemical Society of Japan, 1977). Moreover, the mercury contents of liver (0.71 mg) and kidney (0.28 mg) are reportedly quite high. If these organs are burned in a short time, calculations indicate that a small peak (about 1–3 μg/Nm^3) may appear. From comparisons of the measurements and the above calculations, we can conclude that mercury in dental amalgam has a significant impact on mercury emissions from crematories.

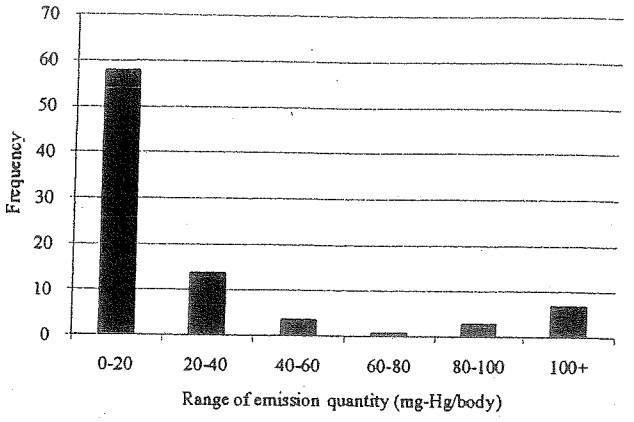

Fig. 4. The distribution of emission quantities calculated using measured data.

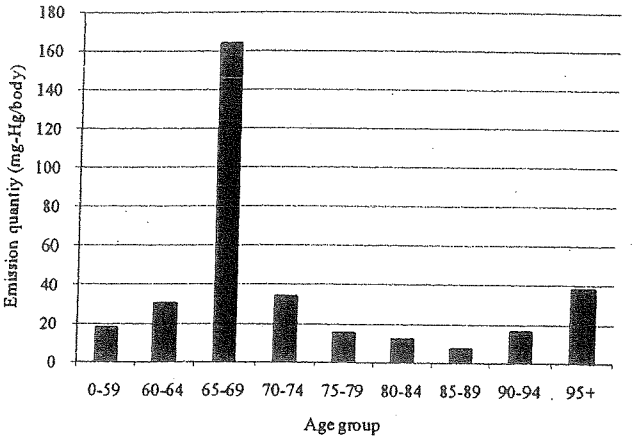

Fig. 5. The amount of mercury emission per cremation calculated using measured data by age group (number of samples by age group: 0–59:4, 60–64:6, 65–69:7, 70–74:9, 75–79:15, 80–84:17, 85–89:9, 90–94:12, 95+:8).

3.3 Mercury emission from crematories

Total emissions from all active crematories in Japan were estimated using the following equations:

$$\text{Total emissions (mg/year)} = \text{Emission quantity (mg/body)} \times \text{the number of cremations (bodies/year)} \quad (2)$$

$$\text{Emission quantity (mg/body)} = \text{Mercury concentration (mg/Nm}^3\text{)} \times \text{dry gas volume (Nm}^3\text{/h)} \times \text{cremation period (h)/the number of cremations (bodies)} \quad (3)$$

The amount of mercury emission was calculated to be 31.7 mg/cremation using the total concentration and gas flow rate. The standard deviation was 64 mg/body; this was so large because the mercury quantity per body has a large range from 0.7 to 362 mg/body. In the United Kingdom, 150 mg/four cremations was proposed as a regulatory criterion by DEFRA (2004). Taking this to be equivalent to 37.5 mg/body, the values in 15 samples in this research were over this level.

The distribution of emission quantities calculated using measured data is shown in Fig. 4. As the emission quantity increases, the frequency decreases. However, the frequency increased at over 80 mg/body. In the United Kingdom, in total, 54 cremations were tested at two separate locations (Rahill, 2008). From the results, 31 cremations were suspected to have been of bodies with no amalgam fillings. Average mercury release per cremation over 54 cremations was reported to be 240 mg/body. The United States Environmental Protection Agency (US EPA) also reported that the emission quantity was 456 mg/body from nine cremations (Rahill, 2008). Although 0.94 mg/body was reported in another US EPA document, this value would have been for bodies with no amalgam filling (US EPA, 1997). According to the UNEP tool kit for the identification and quantification of mercury releases (2005), the emission quantity in various countries ranged from 0.1 to 5.1 g/body. Comparing our results with these reported data, including corpses with amalgam fillings, the emission quantity obtained in this research is quite low. The use of amalgam has a close relationship with mercury emissions. The amount of mercury in amalgam in one filling was reported to be 600 mg by Mills (1990). We sought to examine the actual mercury content in one used amalgam filling. According to our ongoing research, the weight per filling removed by a dentist ranged from 56 to 231 mg. The mercury content in an amalgam filing ranged from 42.5 to 53.0%. Therefore, the average amount of mercury per filling was 51.6 mg. This is at least one of the reasons why emissions in Japan are low.

The amount of mercury emission per cremation calculated using measured data by age group is shown in Fig. 5. The maximum value was obtained in the age range 65–69 years. The second highest value was observed in the age ranges of 95+, 70–74, and 60–64 years. In addition, a difference in mercury emission was observed between males and females. This trend is consistent with a report from the United Kingdom (Edwards, 2001).

These mean values (31.7 mg/body) were multiplied by 1 169 174, the number of bodies cremated in 2007, which was calculated using the number of corpses (1 108 334 including dead bodies of uncertain age (MHLW, 2008b)) and the cremation rate (99.9%); total emissions were estimated to be 35.1 kg/year. Considering the number of bodies and the emission quantity by age group, the total estimated emissions decreased slightly to 32.5 kg/year. This constitutes less

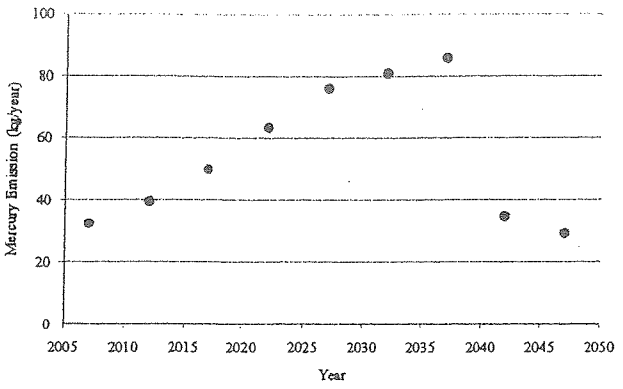

Fig. 6. Estimated future trends of total mercury emissions in Japan from crematories based on population demographic statistics and measured data.

than 0.01% of the total amount of mercury released into the atmosphere (21–28 tons/year) in Japan (Kida et al., 2008).

To estimate mercury emissions in the United Kingdom, 3 g/body was used as the emission quantity (DEFRA, 2004). This value is based on Mills' report (1990), which assumes that a dead body has five restored teeth with amalgam fillings containing 0.6 g mercury. Based on this assumption, the mercury emission was calculated to be 3300 kg/year in Japan. This procedure leads to an obvious overestimation. The mercury release to the air from crematories should be based on measurements.

Next, total future trends in emission were calculated using statistics on population demographics (NIPSSR, 2008) and the emission quantity obtained in this research. That is, the emission quantity was multiplied by the number of dead bodies by age group. Here, we assume that the emission quantity obtained in this research is a property of each group and shifted it to the emission quantity of the next age range as 5 years passed. We set the cremation rate to be 100%. As a result, estimated future trends of total mercury emissions from crematories based on the statistics of population demographics and measured data are shown in Fig. 6. The amount of mercury emissions from crematories is expected to increase to 86 kg/year, about 2.6-fold, between 2007 and 2037. The distinct distribution of emission quantities is expected to show a rapid decrease in 2042 because the highest emission quantity (164 mg/body) moved outside the age ranges. In fact, the distribution of emission quantities would then be expected to moderate and then decline because of the impact of dental care, such as the removal of amalgam fillings, the loss of teeth and loss of mercury in amalgam fillings (Skare, 1995) as the age group shifts. Although we should use measurement data to estimate current emissions, we can combine measurement data with demographic statistics on dental care or material flow data of mercury amalgam to estimate accurate future trends in mercury emissions from crematories.

4 Conclusions

In this study, to measure the actual emission level and estimate the emission from crematories in Japan using measurement data, the mercury concentration in crematory flue gas from mercury emissions was examined at seven facilities. Total averaged mercury concentration in stack gas was 3.6 $\mu g/Nm^3$, which consisted of Hg^0 at 2.6 $\mu g/Nm^3$ and Hg^{2+} at 1.1 $\mu g/Nm^3$. The mercury concentration ranged from 0.2 to 82.7 $\mu g/Nm^3$. At two facilities, we used continuous emission monitoring to measure mercury concentrations and to evaluate mercury behaviour. In some samples, a large peak of Hg^0 appeared at 10–20 min. Considering the behaviour of mercury in cremations, the findings confirmed that mercury in stack gas originated from dental amalgam. The amount of mercury emitted was calculated to be 31.7 mg/cremation using the total concentration and gas flow rate. The emission quantity obtained in this research is apparently quite low. Although the reason for this is unclear, the mercury amount per filling in the Japanese oral cavity may be smaller than that in other countries. Furthermore, the annual amount of mercury emission from crematories in Japan was estimated using the total number of corpses. The total emissions were estimated to be 35.1 kg/year. Total future trends in emissions were calculated using demographic statistics and the emission quantity obtained in this research. As a result, the amount of mercury emissions from crematories is expected to increase by 2.6-fold between 2007 and 2037.

One possible countermeasure would be to remove amalgam fillings before cremation, but this may be difficult to actualize for practical and religious reasons.

Acknowledgements. We are grateful for financial support in the form of a Grant-in-Aid for Waste Treatment Research and Health Sciences Research Grants in FY 2007 to 2009 from the Japanese government and for cooperation with the sampling and mercury analysis from each municipality, and Shoji Eguchi of Taiyo Chikuro Industries and Koji Tanida of Nippon Instruments.

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Feds propose first controls on mercury emissions

Wednesday, March 16, 2011
By Don Hopey, Pittsburgh Post-Gazette

The U.S. Environmental Protection Agency has proposed the first-ever national controls on mercury and other air pollution toxics from power plants. The health-based regulations are expected to prevent as many as 17,000 premature deaths and 11,000 heart attacks a year.

The standards, announced Wednesday in response to a court deadline, are designed to reduce emissions of mercury -- a potent neurotoxin -- arsenic, chromium, lead, nickel and acid gases from power plants by 91 percent, while providing the utility industry four years to comply.

There are now no national standards for mercury emissions and acid gases, half of which come from power plants. There are 17 states with mercury controls but Pennsylvania is not among them.

Two Pennsylvania coal-fired power plants, the Keystone power plant in Armstrong County and the Conemaugh power plant in Indiana County, are listed among the top 25 mercury emitters in the U.S., according to a report released today by the Environmental Defense Fund.

"Today's announcement is 20 years in the making, and is a significant milestone in the Clean Air Act's already unprecedented record of ensuring our children are protected from the damaging effects of toxic air pollution," said EPA Administrator Lisa Jackson at a news conference in Washington, D.C. "With the help of existing technologies, we will be able to take reasonable steps that will provide dramatic protections to our children and loved ones, preventing premature deaths, heart attacks, and asthma attacks."

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The proposed rule is open for public comment. A final rule is expected in November.

Coal-fired power plants are responsible for 99 percent of mercury emissions from the electric power industry. The toxic pollutants are known to cause neurological damage, according to the EPA, including lower IQ in children. The pollutants also cause environmental damage to rivers, lakes and streams and the fish that live in them. Many states, including Pennsylvania, have fish consumption advisories due to mercury pollution.

"This is historic. It would end the lethal loophole that permits coal-burning power plants to spew poisonous pollution into the air," said Frank O'Donnell, president of Clean Air Watch, an environmental organization focused on air quality. "Indeed, this is the single biggest step for public health protection that the EPA will take this year. Thousands of Americans will live longer and many millions will breathe easier as a result. Not only that, but fish will be safer to eat as toxic mercury is reduced from water bodies."

The EPA estimates that the proposed rule's public health and economic benefits, including the creation of an estimated 31,000 short-term construction jobs and 9,000 long-term maintenance and operational jobs, will greatly exceed the costs of implementation. Every dollar spent to install pollution controls will produce public health and economic business benefits of up to $13 dollars. That could total as much as $140 billion annually.

Ms. Jackson said the installation of toxics pollution controls at the 44 percent of the nation's coal-fired power plants that have no controls could lead to utility bill increases of from $3 to $4 a month for consumers. It might also cause utilities to close some of the nation's oldest and biggest polluting power plants and invest in new power plant construction instead.

More details in tomorrow's Pittsburgh Post-Gazette.

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Environmental mercury release, special education rates, and autism disorder: an ecological study of Texas

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Abstract

The association between environmentally released mercury, special education and autism rates in Texas was investigated using data from the Texas Education Department and the United States Environmental Protection Agency. A Poisson regression analysis adjusted for school district population size, economic and demographic factors was used. There was a significant increase in the rates of special education students and autism rates associated with increases in environmentally released mercury. On average, for each 1000 lb of environmentally released mercury, there was a 43% increase in the rate of special education services and a 61% increase in the rate of autism. The association between environmentally released mercury and special education rates were fully mediated by increased autism rates. This ecological study suggests the need for further research regarding the association between environmentally released mercury and developmental disorders such as autism. These results have implications for policy planning and cost analysis.

Keywords: Mercury; Special education; Autism; Environmental toxins; Ecological

Introduction

Exposure to a variety of environmental neurotoxicants is known to affect normal child development, resulting in a spectrum of adverse outcomes, ranging from severe mental retardation and developmental disability to more subtle changes in functioning, depending in part on the timing and dose of the chemical agent (Landrigan and Garg, 2002; Mendola et al., 2002; Rice and Barone, 2000).

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) section 104 (i), as amended by the Superfund Amendments and Reauthorization Act (SARA), requires the Agency for Toxic Substances and Disease Registry (ATSDR) and the Environmental Protection Agency (EPA) to prepare a list, in order of priority, of substances that are most commonly found at waste facilities on the National Priorities List (NPL) and which are determined to pose the most significant potential threat to human health due to their known or suspected toxicity and potential for human exposure. Accordingly, mercury is listed as the third-most frequently found (arsenic and lead are

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first and second) toxic substance in the United States (ATSDR, 2001).

Symptoms of nervous system disruption associated with chronic exposure to mercury has been known since the 19th century, when mercury was widely used in the felt industry which led to the expression of "hatter's disease" (Hu, 1998). Further epidemiological evidence of the neurotoxicity of mercury dates back to the 1950s, when it was ascertained that thousands of people in Minamata and Niigata Japan suffered various neurological impairments caused by consumption of mercury contaminated fish (Harada, 1978). However, the neurotoxicity of low-level mercury exposure has only recently been documented (NAS, 2000; EPA, 1997) and recent reports implicate mercury in the etiology of various developmental and learning disabilities (Ramirez et al., 2003; Grandjean et al., 2003) including autism (Bernard et al., 2001, 2002).

Recent evidence for mercury toxicity relevant to the biology of autism is compelling (Palomo et al., 2003; Aschner and Walker, 2002; Bernard et al., 2002; Vojdani et al., 2003) and Bradstreet et al. (2003) report that levels of urinary mercury after a 3-day treatment with an oral chelating agent, meso-2,3-dimercaptosuccinic acid (DMSA), in children with autistic spectrum disorders were three times those in a matched normal control sample.

Environmentally released mercury is a major source of mercury exposure. Mercury is released into the environment largely from fossil fuel (mainly coal) combustion by electrical utilities and from municipal and medical waste incinerators. This inorganic mercury becomes airborne and may be carried for miles before being deposited on soil or water. This inorganic form of mercury is then converted to a toxic form (methylmercury) by chemical reactions or by bacteria, which is absorbed by aquatic microorganisms that are eaten by fish, and in this manner accumulates up the aquatic food chain. Humans are primarily exposed through fish consumption (Myers et al., 2000) and transmission from mothers to infants is well documented in animal models (Newland et al., 1994) and human studies (Ramirez et al., 2000; Grandjean et al., 1995). Results from several studies show that maternal mercury exposure during pregnancy is associated with neuropsychological deficits in children and that this association is most evident in women with stable exposures throughout pregnancy (Ramirez et al., 2003; Grandjean et al., 2003).

Other than accidental poisoning at the population level, where developmental disabilities have been reported as the result of large mercury spills (Racz and Vandewater, 1982), there have been no published studies examining the risk of disability associated with mercury released into the environment within the current legal limits. The available information regarding exposure to toxic agents associated with developmental disorders is suggestive but inconclusive (Ostrowski et al., 2003). In a prior study, we report evidence for an association between environmentally released mercury and various developmental disorders, including autism, at the state level ($n = 50$) (unpublished manuscript). We considered the positive association between developmental disabilities and environmentally released mercury in that investigation as preliminary due to the relatively small number of large geological regions. In this study, we investigate the association between environmentally released mercury pollution and autism rates at the county ($n = 254$) and school district level ($n = 1184$) in Texas. The advantage of using county level data in this study allows an investigation using greater numbers of smaller geographic units in the analysis—this can potentially increase our power to detect an effect if in fact it present. Since Texas ranks 4th among states with the highest reported mercury releases (next to California, Oregon, and West Virginia) (USEPA-TRI, 2004), analysis of data from this state can be useful for further investigation of the association between environmental mercury release and developmental disorders. In this study, we investigate the association between total special education rates, autism, and environmental mercury release.

Methods

Data source and sample data regarding environmentally released mercury for each county were obtained from the United State Environmental Protection Agency Toxics Release Inventory (TRI) (USEPA-TRI, 2004). TRI collects information about chemical releases and waste management reported by major industrial facilities in the US. The TRI database was established by Section 313 of the Emergency Planning and Community Right-To-Know Act of 1986 (EPCRA). Under EPCRA, industrial facilities in specific sectors are required to report their environmental releases and waste management practices annually to the EPA. Facilities covered by this act must disclose their releases to air, water, and land of approximately 650 toxic chemicals, as well as the quantities of chemicals they recycle, treat, burn, or otherwise dispose of on-site and off-site. The current analysis uses reports of pollution that industrial facilities provided to TRI for the calendar year 2001. The total number of pounds of environmentally released mercury was obtained for each county.

Administrative data from the Texas Education Agency (TEA) from school years 2000–2001 were analyzed. Data and data description are available at the TEA website at http://198.214.99.202. In compliance with the Texas Education Code, the Public Education Information Management System (PEIMS) contains

data necessary for the legislature and the TEA to perform their legally authorized functions in overseeing public education. The database consists of student demographic, personnel, financial, and organizational information. Autism counts per school district were obtained by special request from the TEA. Data were from 1184 school districts in 254 counties in Texas. These districts represented approximately 4 million children enrolled in grades K through 12.

Diagnosis of autistic disorder was abstracted from the school record for each year of the study period. Diagnoses were made by qualified special education psychologists employed by the TEA or from psychologists or medical doctors outside the TEA system. While diagnoses were not standardized, there is considerable evidence that diagnoses of autistic disorder are made with good reliability and specificity in the field (Eisenmajer et al., 1996; Hill et al., 2001, Mahoney et al., 1998).

District population wealth was calculated as a school district's total taxable property value in 2001 as determined by the Comptroller's Property Tax Division (CPTD), divided by the total number of students in the district in 2000–2001. Property value was determined by the CPTD as part of its annual study, which attempts to present uniformly appraised property valuations statewide. The CPTD value is calculated by applying ratios created from uniform independent appraisals to the district's assessed valuations.

Racial composition was accounted for by the proportion of European-American children enrolled in schools within each district.

Total number of students was calculated as all enrolled students as of October 28, 2000 in grades kindergarten through twelve, who attended at least 1 day of school for that school year. Statewide, 6975 students, or 0.2% of all students, were enrolled but did not attend school.

Proportion of economically disadvantaged students was calculated as the percentage of students who were eligible for free meals under the National School Lunch and Child Nutrition Program, reduced-price meals under the National School Lunch and Child Nutrition Program, or other public assistance.

Total number of students enrolled in special education was calculated as the number of students receiving special education in each district.

Urbanicity. Eight separate demographic district regions were utilized in the analysis: (1) *Major urban* districts are the districts with the greatest membership in counties with populations of 650,000 or more, and more than 35% of the students are identified as economically disadvantaged. (2) *Other central city*—The major school districts in other large, but not major, Texas cities. Other central city districts are the largest districts in counties with populations between 100,000 and 650,000 and are not contiguous to any major urban districts. (3) *Major suburban* districts are contiguous to major urban districts. If the suburban district is not contiguous, it must have a student population that is at least 15% of the size of the district designated as major urban. (4) *Other central city suburban*—Other school districts in and around the other large, but not major, Texas cities. They are contiguous to other central city districts. If the suburban district is not contiguous, it must have a student population that is at least 15% of the size of the district designated as central city. (5) *Independent town*—The largest school districts in counties with populations of 25,000–100,000. (6) *Non-metro: fast growing* school districts that are not in any of the above categories and that exhibit a 5-year growth rate of at least 20%. These districts must have at least 300 students in membership. (7) *Non-metro: stable* school districts that are not in any of the above categories, yet have a number of students in membership that exceeds the state median. (8) *Rural* school districts that do not meet the criteria for placement into any of the above categories. These districts either have a growth rate less than 20% and the number of students in membership is between 300 and the state median, or the number of students in membership is less than 300.

In the analysis, the first two categories above were combined to form an "urban" dummy variable, categories three and four were combined to form a "suburban" dummy variable and categories five through seven formed an "other" category, with rural districts as the reference group.

Statistical methods. Since the 1184 school districts were nested within 254 counties, we modeled the data using a multilevel Poisson regression model to adjust estimates due to a potential county level clustering effect—which can bias estimated standard errors downward, thus leading to type I errors if not properly addressed (Barcikowski, 1981).

A multilevel Poisson regression model allowing for over-dispersion of the dependent variable was used in which the total number of children with autism and the number of special education students (excluding autism) was modeled separately as a function of the total pounds of environmentally released mercury. The model was adjusted for percent of the population of European-American descent, district population wealth, percent economically disadvantaged and urbanicity. Rates were offset by the total number of children served in a school district. For the model predicting autism rates, special education counts were included as a covariate in a subsequent model. For the model predicting special education rates, autism counts were also included as a covariate in a separate model. All models were estimated using MLwiN software with a log link function specified (Goldstein et al., 1998). The analysis yields adjusted relative rate estimates as a function of pounds of environmentally released mercury.

Results

Table 1 shows the descriptive statistics of the study variables. The standard deviation and the maximum and minimum values indicate considerable variation for all study variables. Table 2 shows the results of the regression model where autism rates were modeled as a function of pounds of mercury and sociodemographic covariates (model 1), plus adjustment for the number of special education students (excluding autism) (model 2).

Model 1 shows that for each 1000 lb of environmentally released mercury, the rate of autism increases by 61%. A small but significant rate increase is noted for districts with higher wealth, and a small but significant inverse association is observed for percentage of European American and economically disadvantaged students. A large effect is observed for community type. The highest rate increase is observed when comparing urban to rural school districts—relative to rural districts there is a 473% higher rate of autism. There is a 255%

Table 1
Descriptive statistics for study variables ($n = 1184$ school districts in 254 counties)

	Mean	SD	Minimum	Maximum
Autism count total	5.11	21.39	0	416
Total special education population count	414.12	1205.21	0	21,900
Pounds of environmental mercury release	203.99	522.84	0	2059
Total student population	3382.30	10908.99	6	209,916
Percent economically disadvantaged	47.28	21.70	0	100
Percent European American	58.33	29.71	0	100
District wealth	$189,080	$262,290	0	$4,276,736
Community type				
% Urban	4.1	—	—	—
% Suburban	13.2	—	—	—
% Rural	34.9	—	—	—
% Other	47.8	—	—	—

Table 2
Poisson regression estimates predicting relative rate of autism prevalence

	Estimate (SE)	Relative rate	Lower 95% CI	Upper 95% CI
Model 1: Predicting autism prevalence rates as a function of mercury release with demographic covariate adjustments				
Mercury (per 1000 pounds)	0.479 (0.041)	1.614	1.487	1.752
Percent European American	−0.023 (.001)	0.977	0.975	0.979
District wealth (per 100,000 dollars)	0.060 (0.010)	1.062	1.041	1.083
Percent economically disadvantaged	−0.029 (0.001)	0.971	0.969	0.973
Urban versus rural	1.553 (0.109)	4.726	3.800	5.877
Suburban versus rural	0.935 (0.108)	2.547	2.052	3.161
Other versus rural	0.027 (0.112)	1.027	0.821	1.285
Model 2: Predicting autism prevalence rates as a function of mercury with demographic and special education count adjustment				
Mercury (per 1000 pounds)	0.160 (0.031)	1.174	1.103	1.249
Percent European American	−0.019 (0.001)	0.981	0.979	0.983
District wealth (per 100,000 dollars)	0.010 (0.010)	1.010	0.990	1.030
Percent economically disadvantaged	−0.034 (0.001)	0.967	0.965	0.969
Urban versus rural	0.953 (0.078)	2.593	2.219	3.031
Suburban versus rural	0.808 (0.074)	2.243	1.935	2.601
Other versus rural	−0.356 (0.087)	0.700	0.589	0.834
Special education count (per 1000)	0.172 (0.005)	1.188	1.176	1.200

higher rate of autism in suburban relative to rural districts.

In model 2, after adjustment for the number of special education students, mercury remained a significant predictor of autism rates, indicating a 17% increase in autism rates for every 1000 lb of mercury released in the environment. The number of special education students was a significant predictor of autism rates as well. Wealth was no longer a significant predictor and the other covariates showed decreases relative to model 1, but remained significant.

Table 3 shows the regression estimates where special education rates (excluding autism counts) were modeled as a function of pounds of mercury and sociodemographic covariates (model 3), plus adjustment for the number of autistic students (model 4).

Model 3 shows that each 1000 lb of reported mercury release is associated with a 43% increase in the rate of special education students. Small but significant increases were associated with the percentage of European Americans, economically disadvantaged and district wealth. Community type was strongly associated with special education rates. All community-type categories show a much higher percentage of special education students relative to rural communities.

In model 4, after adjusting for total autism counts, the association between pounds of mercury and special education rates was no longer statistically significant—with the other covariates in the model remaining significant. This indicates that increased rates in autism account for the association between environmentally released mercury and the rate of special education students.

Discussion

To the best of our knowledge, this is one of the first investigations to report an ecological association between developmental disorders and environmentally released mercury.

The results of this study demonstrate that school district autism and special education rates are significantly associated with environmentally released mercury. This association was independent of the number of children served in the educational system for that district, district wealth, ethnic make-up, and community type. Further, these results indicate that the association between mercury release and school district special education rates was completely accounted for by increased rates of autism. This indicates that, in Texas, the increase in special education rates attributable to environmental mercury can be explained by increases in autism. The results of this study are consistent with our prior nation-wide study where an association between various developmental disabilities and environmentally released mercury was observed at the state level

Table 3
Poisson regression estimates predicting relative rate of special education prevalence

	Estimate (SE)	Relative rate	Lower 95% CI	Upper 95% CI
Model 3: Predicting special education prevalence rates as a function of mercury with demographic adjustments				
Mercury (per 1000 pounds)	0.360 (0.030)	1.433	1.350	1.522
Percent white	0.004 (0.001)	1.004	1.002	1.006
District wealth (per $100,000)	0.050 (0.010)	1.051	1.030	1.073
Percent economically disadvantaged	0.012 (0.001)	1.012	1.010	1.014
Urban versus rural	2.741 (0.104)	15.502	12.591	19.087
Suburban versus rural	2.110 (0.103)	8.248	6.713	10.135
Other versus rural	1.550 (0.110)	4.711	3.781	5.871
Model 4 Predicting special education prevalence rates as a function of mercury with demographic and autism count adjustments				
Mercury (per 1000 pounds)	−0.062 (0.032)	0.940	0.882	1.002
Percent white	0.008 (0.001)	1.008	1.006	1.010
District wealth (per $100,000)	0.030 (0.010)	1.030	1.010	1.051
Percent economically disadvantaged	0.014 (0.001)	1.014	1.012	1.016
Urban versus rural	2.240 (0.068)	9.393	8.199	10.762
Suburban versus rural	1.902 (0.066)	6.699	5.871	7.645
Other versus rural	1.174 (0.073)	3.235	2.795	3.743
Autism count (per 100)	0.689 (0.022)	1.992	1.906	2.081

(unpublished manuscript). However, the results of this report should be interpreted with caution for a number of reasons.

First, this is an ecological study that precludes interpretation at the individual level. We have used aggregate units in this analysis to investigate differential rates of autism as a function of pounds of mercury at the county level. While we properly addressed the potentially biasing effects of clustering (school districts nested within counties) by utilizing appropriate analytic methods (e.g. multilevel-analysis), individual data are required to make a better case for the observed associations and their interpretations. Nevertheless, ecological studies of this type are often an important first step in identifying subsequent areas of investigation.

Second, a causal association between environmentally released mercury and developmental disorders cannot be determined from this cross-sectional data. Data availability permitting, future studies could investigate this association by using longitudinal data where changes in mercury levels over time may be used as a predictor of the rate of change in developmental disorders over time.

Third, we should consider that school-based administrative autism data, such as these, are only a proxy for true community prevalence. However, these autism rates are most likely biased downward. For example, Yeargin-Allsopp et al. (2003) found that, in one metropolitan area, 18% of children who qualified for a diagnosis of autism according to their study criteria were receiving special education services but had not been categorized as having autism. The critical unknown issue is whether identification of children in the special education system is systematically biased in the same direction as reporting of environmental mercury release. For example, counties in which administrations are more aggressive regarding penalties for underreporting toxic release may also have educational policies that result in a greater number of children identified for special education services. Despite the limitations of these administrative data, as demonstrated, these data can be a useful component to preliminary epidemiological studies (Dales et al., 2001). By demonstrating an association between environmentally released mercury and developmental disorders, the results of this study provide a necessary first step in identifying plausible contributing factors of risk for developmental disabilities.

This line of research has implications for toxic substance regulation and prevention policies. The effects of differing state policies regarding toxic release of mercury on the incidence of developmental disorders should be investigated. For example, policies that have successfully limited exposures to lead have had direct effects on morbidity and have demonstrated reductions in health care costs related to lead exposure (Sargent et al., 1999, Galke et al., 2001; Brown, 2002). However, while federal efforts toward reducing mercury exposure through policy have been successful to some extent by signing bills into law, proportionally few have been enacted (Mercury Policy Project (MMP), 2004). Despite existing policy recommendations, debate concerning acceptable levels of safety still remains (Dourson et al., 2001; Kaiser, 2000), thus, limiting progress toward evaluating policies related to reducing exposure to mercury.

Conclusions

What is currently known about the low-level toxicity of mercury from behavioral toxicology and behavioral teratology studies are convincing enough to warrant further study. This study is among the first to demonstrate an association between environmentally released mercury at the county level and the rate of developmental disability. Given the limitations of this ecological association, future studies should investigate this association using other methodologies and samples. This line of research has important implications for public health policy and supports prior recommendations for reducing environmentally released mercury (Needleman, 1995; Landrigan et al., 1994).

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Characterizing the Emissions of Polychlorinated Dibenzo-*p*-dioxins and Dibenzofurans from Crematories and Their Impacts to the Surrounding Environment

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This study was set out to characterize PCDD/F emissions from crematories and assess their impacts on the surrounding environment. Two crematories located in southern Taiwan were investigated, including the one (C1) with no air pollution control device installed and the other (C2) installed with a bag filter. Results show the mean PCDD/F emissions (11% oxygen) from the stacks of C1 and C2 were 2.36 and 0.322 ng I-TEQ Nm^{-3}, respectively. The mean emission factors for C1 and C2 were 13.6 and 6.11 μg I-TEQ $body^{-1}$, respectively. The removal efficiency of the bag filter on PCDD/Fs was 55.1%. The estimated PCDD/F emission rate for all crematories in Taiwan was 0.838 g I-TEQ yr^{-1} accounting for 227% and 112% of the annual emissions from all medical waste incinerators and municipal waste incinerators, respectively. The above results indicate that PCDD/F emissions from crematories were quite significant. To assess the impact of PCDD/F emissions from a crematory to the surrounding environment, ambient air samples were collected from the downwind site of C1 with the maximum ground concentration. We found the estimated maximum ground concentration at the downwind site of C1 (= 0.521 pg I-TEQ Nm^{-3}) was much higher than that found at the background, rural area, residential area, urban area, and industrial area (= 0.006, 0.023, 0.052, 0.093, and 0.190 pg I-TEQ Nm^{-3}, respectively). The above results suggest that PCDD/F emissions from a crematory did significantly affect its surrounding environment. In conclusion, a proper control strategy should be taken immediately in order to eliminate PCDD/F emissions from crematories.

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Introduction

After polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) were discovered in the flue gases and fly ash of municipal waste incinerators in 1977 (*1*), PCDD/F emissions from various sources have become a serious issue in many countries, because of their toxicological effects and associated adverse health implications. PCDD/Fs are mainly formed during anthropogenic activities and are usually referred to as dioxins. Intensive studies have been conducted on various PCDD/F emission sources, including the waste combustion sources, chemical-industrial sources, and other thermal sources. Nevertheless, only a few studies were carried out on crematories (*2*−*7*).

In a study conducted by Fledler in Germany, PCDD/F toxic equivalent (i.e., TEQ) concentrations of ~8 ng TEQ Nm^{-3} were measured in the stack flue gases of crematories (*2*). Takeda et al. measured PCDD/F emissions from 17 crematories in Japan found PCDD/F concentrations and their corresponding TEQ concentrations in the stack flue gases were 4.9−1200 ng Nm^{-3} (12% oxygen) and 0.064−24 ng TEQ Nm^{-3}, respectively (*5*). In the UK, a laboratory study conducted by the Warren Spring Laboratory found the mean PCDD/F TEQ concentration of 46 ng TEQ Nm^{-3} (11% oxygen) for the cremation process (*7*). The wide range of PCDD/F concentrations arising from various crematories are believed to be due to their intrinsic differences in operation conditions, air pollution control devices, and involved incinerating materials (*5*).

In the U.S. (*8*), UK (*7*), and Japan (*5*), PCDD/F emission rates for the crematory source were found as ~9.1, 1−35, and 1.3−3.8 g TEQ yr^{-1}, respectively. In principle, the total PCDD/F emission from the crematory was relatively small as compared with that from the municipal waste incinerator. For example, a recent study conducted in Japan has indicated that the crematory emission accounted for only 0.13−0.29% of that emitted from municipal waste incinerators (*5*). But it should be noted that most crematories are equipped with a low stack and are situated in the proximity of the residential area. In particular, most of them do not adopt any air pollution control device to eliminate PCDD/F emissions from stacks. Based on these, it can be expected that PCDD/F emissions from a crematory might significantly affect its surrounding environment. Moreover, it should be noted that recently the cremation ratio has increased dramatically in Taiwan and many other countries. In Taiwan, the cremation ratio is expected to increase from 66.9% in 2000 to 85.0% in 2005. In the U.S., the cremation ratio has increased significantly from 15.2% in 1987 to 25.0% in 2000 and is expected to reach 37.0% in 2010 (*8*). In Japan, because of the encouragement of the governmental policy its current cremation ratio is as high as 99% (*4*). Based on these data, it is expected that crematories will play an important role on PCDD/F emissions not only in Taiwan area but also in many other countries.

Indeed, in addition to PCDD/Fs, PCBs, and PAHs, total suspended particles (TSP) and odor released from crematories might also cause serious problems to human health. However, the U.S. EPA has reported that there appears to be no "safe" level for dioxin exposure, and the levels of dioxin and dioxin-like chemicals found in the general U.S. population were "at or near levels associated with adverse health effects" (*9*). Subjected to both cost and manpower, only PCDD/F emissions from crematories were studied in this work. In this study, two crematories located in southern Taiwan with similar operation conditions were investigated. The congener

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TABLE 1. Basic Information for the Two Investigated Crematories of C1 and C2

crematory	C1	C2
operational temperature of the primary combustor	730 °C	750 °C
operational temperature of the secondary combustor	620 °C	650 °C
capacity	0.5 body h^{-1}	0.5 body h^{-1}
auxiliary fuel	diesel (40 L h^{-1})	diesel (40 L h^{-1})
air pollution control devices		bag filter
temperature of the stack flue gas	300 °C	200 °C
height of the stack	5 m	6 m

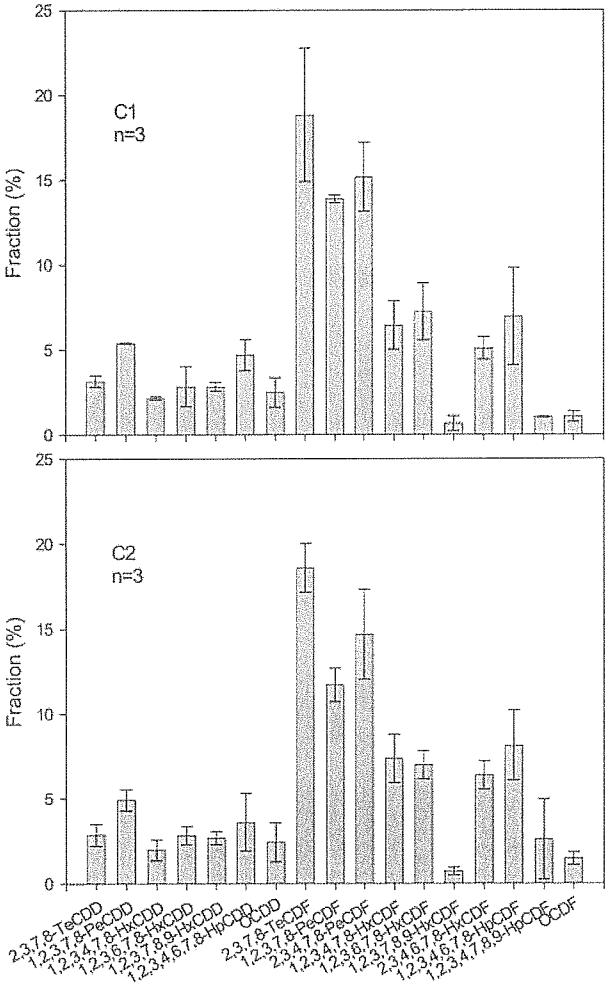

FIGURE 1. Congener profiles of seventeen 2,3,7,8 chlorinated substituted PCDD/Fs containing in the stack flue gases of C1 and C2.

profiles of PCDD/Fs in the stack flue gases of these two crematories were presented and compared. PCDD/F emission factors and emission rates for both crematories were determined. Furthermore, to assess the influence of PCDD/F emissions from a crematory process to the surrounding environment, the PCDD/F concentrations in the atmosphere of the background area, rural area, residential area, urban area, and industrial sites were measured and were compared with the results that were obtained from the vicinity of a crematory.

TABLE 2. Mean PCDD/Fs Emission Factors for the Two Investigated Crematories of C1 and C2

	C1 ($n = 3$)		C2 ($n = 3$)	
PCDD/Fs	mean g body^{-1}	RSD %	mean g body^{-1}	RSD %
2,3,7,8-TeCDD	2.26	74	0.969	61
1,2,3,7,8-PeCDD	4.01	80	1.70	46
1,2,3,4,7,8-HxCDD	1.59	78	0.738	31
1,2,3,6,7,8-HxCDD	1.76	48	0.894	53
1,2,3,7,8,9-HxCDD	2.19	87	0.964	52
1,2,3,4,6,7,8-HpCDD	3.78	93	0.977	82
OCDD	2.12	101	0.552	43
2,3,7,8-TeCDF	12.9	66	6.39	48
1,2,3,7,8-PeCDF	10.3	80	3.71	46
2,3,4,7,8-PeCDF	10.7	71	4.82	45
1,2,3,4,7,8-HxCDF	5.24	94	2.73	49
1,2,3,6,7,8-HxCDF	5.90	95	2.34	42
1,2,3,7,8,9-HxCDF	0.358	19	0.242	66
2,3,4,6,7,8-HxCDF	4.00	89	2.19	44
1,2,3,4,6,7,8-HpCDF	6.05	104	3.06	55
1,2,3,4,7,8,9-HpCDF	0.766	77	0.490	65
OCDF	0.698	58	0.495	60
total PCDD/Fs (μg body^{-1})	74.6	81	33.3	43
total I-TEQ (μg I-TEQ body^{-1})	13.6	75	6.11	45

Material and Methods

Investigating PCDD/F Emissions from the Cremation Process. Two crematories (denoted as C1 and C2) located in southern Taiwan were selected in this study. Both crematories had the same capacity (= 0.5 body h^{-1}) and were equipped with two combustion chambers (i.e., a primary and a secondary combustion chamber) operated under very similar combustion conditions (= 730 °C and 620 °C and 750 °C and 650 °C in the primary and secondary combustion chambers for C1 and C2, respectively). Both crematories used diesel as their auxiliary fuel with the same feeding rate specified at 40 L h^{-1}. C1 was not equipped with any air pollution control device, but C2 was facilitated with one bag filter with an automatic shaking control unit. Both crematories were equipped with a low stack (stack height = 5 and 6 m for C1 and C2, respectively). Basic information for these two crematories is described in more detail in Table 1.

Three PCDD/F samples were collected from the stack flue gas for each of the two selected crematories according to the U.S. EPA modified Method 23. The sampling train adopted in this study is comparable with that specified by the U.S. EPA Modified Method 5. Prior to sampling, XAD-2 resin was spiked with PCDD/F surrogate standards prelabeled with isotopes. The sampling time for each stack flue gas sample was ~2.5 h. To ensure the free contamination of the collected samples, one trip blank and one field blank were also taken when the field sampling was conducted.

Assessing PCDD/F Emissions from the Crematory to the Surrounding Environment. To assess the influence of PCDD/F emissions from a crematory on the surrounding environment, four ambient air samples were collected from two sampling sites at the vicinity of C1. The above two sampling sites were known with the maximum ground concentrations of C1 determined by using the Industrial Source Complex Short-Term Model (ISCST3). Yet, it is true that the accuracies regarding the use of ISCST3 on estimating maximum ground PCDD/F concentrations might be affected by the deposition of particle-phase PCDD/Fs and the decay of PCDD/Fs in the air due to photolysis (*10*). In this study, because C1 was known with a low stack and hence the duration of emitted PCDD/Fs transported from the stack to the ground level could be quite short. Based on this, we assumed both the deposition of particle-phase PCDD/Fs and photolysis of PCDD/Fs during the transportation period were

TABLE 3. PCDD/F Emission Factors for the Crematory Obtained from Different Studies

emission factors	reference	annotation
13.6 μg I-TEQ body^{-1}	this study	none of APCD
6.11 μg I-TEQ body^{-1}	this study	bag filter as its APCD
2.4–80 μg I-TEQ body^{-1}	(*7*)	
6 μg I-TEQ body^{-1}	(*3*)	
28 μg I-TEQ body^{-1}	(*10*)	derived from one crematory in Germany
70–80 μg I-TEQ body^{-1}	(*10*)	derived from two crematories in UK
0.5 μg I-TEQ body^{-1}	(*10*)	derived from one crematory in U.S.
9.2 μg I-TEQ body^{-1} (regarding ND as 0)		
11 μg I-TEQ body^{-1} (regarding ND as the half value of the detection limit)	(*4*)	derived from 10 crematories in Japan
3.97 μg I-TEQ body^{-1} (arithmetical mean)	(*5*)	derived from 17 crematories in Japan
1.83 μg I-TEQ body^{-1} (geometric mean)		

negligible. In this study, the dispersion parameters (such as atmospheric stability and mixing height) adopted in ISCST3 were determined based on the hourly meteorological data. According to the data obtained from the local weather bureau, we found that the prevailing winds were NW and N with their average wind speeds of 4.2 m s^{-1} and 2.4 m s^{-1}, respectively. Based on this, two sampling sites situated at the downwind sites of C1 with distances 80 m (SE) and 65 m (E) away from the stack were then determined in this study. For comparisons, five sampling sites were also selected for collecting ambient air samples during the same time. The first sampling site, the Keng-Ting National Park ($n = 2$), was situated at the southern end of Taiwan. This site was selected because it was far away from all possible pollution sources and hence its PCDD/F concentration could be regarded as the background level. The second sampling site was located at the Taitung county ($n = 4$), the least industrialized area in Taiwan, and hence was thought to be representative for the rural area. The other thee sampling sites were selected from the residential area ($n = 2$), urban area ($n = 4$), and industrial area ($n = 4$) of the same city as where C1 was located (i.e., the Kauhsiung city, the most industrialized area in Taiwan).

Each ambient air sample was collected using a PS-1 sampler (Graseby Andersen, GA) according to the revised EPA Reference Method T09A. The sampling flow rate was specified at ~0.225 m^3 min^{-1}. Each sample was collected continuously on three consecutive days (sampling volume = ~972 m^3). The PS-1 sampler was equipped with a quartz-fiber filter for sampling particle-phase PCDD/Fs and followed by a glass cartridge for sampling gas-phase PCDD/Fs, respectively. A known amount of surrogate standard was spiked to the glass cartridge in the laboratory prior to the field sampling being conducted.

Sample Analysis. Analyses of stack flue gas and ambient air samples followed the U.S. EPA modified method 23 and EPA Reference Method T09A, respectively. All chemical analyses were carried out by the Super Micro Mass Research and Technology Center in Cheng Shiu Institute of Technology—the only accredited laboratory in Taiwan for PCDD/F analyses. Each collected sample was spiked with a known amount of the internal standard. After being extracted for 24 h, the extract was concentrated, treated with concentrated sulfuric acid, and then followed by a series of sample cleanup and fractionation procedures. The eluate was concentrated to ~1 mL, then transferred to a vial, and then further concentrated to nearly dryness by using a nitrogen stream. Prior to PCDD/F analysis, the standard solution was added to the sample to ensure the recovery during the analysis process.

Two high-resolution gas chomatographs/high-resolution mass spectrometers (HGC/HMS) were used for PCDD/Fs analyses (one for analyzing stack flue gas samples and the other for ambient air samples). The HGC (Hewlett-Packard 6970 Series gas, CA) was equipped with a DB-5 fused silica capillary column (L = 60 m, ID = 0.25 mm, film thickness = 0.25 μm) (J&W Scientific, CA) and with a splitless injection. The oven temperature program was set according to the following: begin at 150 °C (held for 1 min), then increase at 30 °C min^{-1} to 220 °C (held for 12 min), then increase at 1.5 °C min^{-1} to 240 °C (held for 5 min), and finally increase at 1.5 °C min^{-1} to 310 °C (held for 20 min). Helium was used as the carrier gas. The HMS (Micromass Autospec Ultima, Manchester, UK) mass spectrometer was equipped with a positive electron impact (EI+) source. The analyzer mode of the selected ion monitoring (SIM) was used with resolving power at 10 000. The electron energy and source temperature were specified at 35 eV and 250 °C, respectively.

Results and Discussion

Characteristics of PCDD/F Emissions from Crematories. The congener profiles of the 2,3,7,8-substituted PCDD/Fs were selected as the signatures of the crematory emissions. Each selected congener was normalized by reference to the total weight of all 2,3,7,8-congeners. Figure 1 shows the congener profiles of the seventeen 2,3,7,8 chlorinated substituted PCDD/Fs (mean±SD) detected from the stack flue gases of C1 and C2. The top three congeners for both crematories were 2,3,7,8-TeCDF, 2,3,4,7,8-PeCDF, and 1,2,3,7,8-PeCDF. The above results were quite similar to the congener profiles obtained from 10 crematories in Japan (*4*). Indeed, the involved incinerating materials (such as the weight of the dead body, sex, type of coffin, and other accompanied funeral materials, etc.) in crematories of the above-mentioned study might be different for us. However, the above-mentioned study has concluded that the incinerating materials might have a very limited effect on the congener profiles (*4*). The results obtained from this study further support the plausibility of the above inference.

PCDD/F Emissions from Crematories. Table 2 shows the mean emission factors for C1 (no air pollution control device was installed) and C2 (equipped with a bag filter) were 74.6 and 33.3 μg body^{-1} (in terms of total PCDD/F emissions) and 13.6 and 6.11 μg I-TEQ body^{-1} (in terms of total I-TEQ emissions), respectively. It is known that both C1 and C2 had quite comparable operation conditions (see Table 1). Therefore, it was assumed that both crematories might result in similar PCDD/F emissions during the cremation process. Based on this, the removal efficiency of the bag filter could be determined according to the following equation

$$\text{removal efficiency } (\eta;\ \%) = (A - B)/A \times 100\%$$

where A and B were the mean emission factors of C1 and C2, respectively. Based on this, it can be found that the removal efficiencies of the bag filter on the total PCDD/F emission and the total PCDD/F I-TEQ emission were 55.4% (= (74.6−33.3)/74.6) and 55.1% (= (13.6−6.11)/13.6), respectively. The

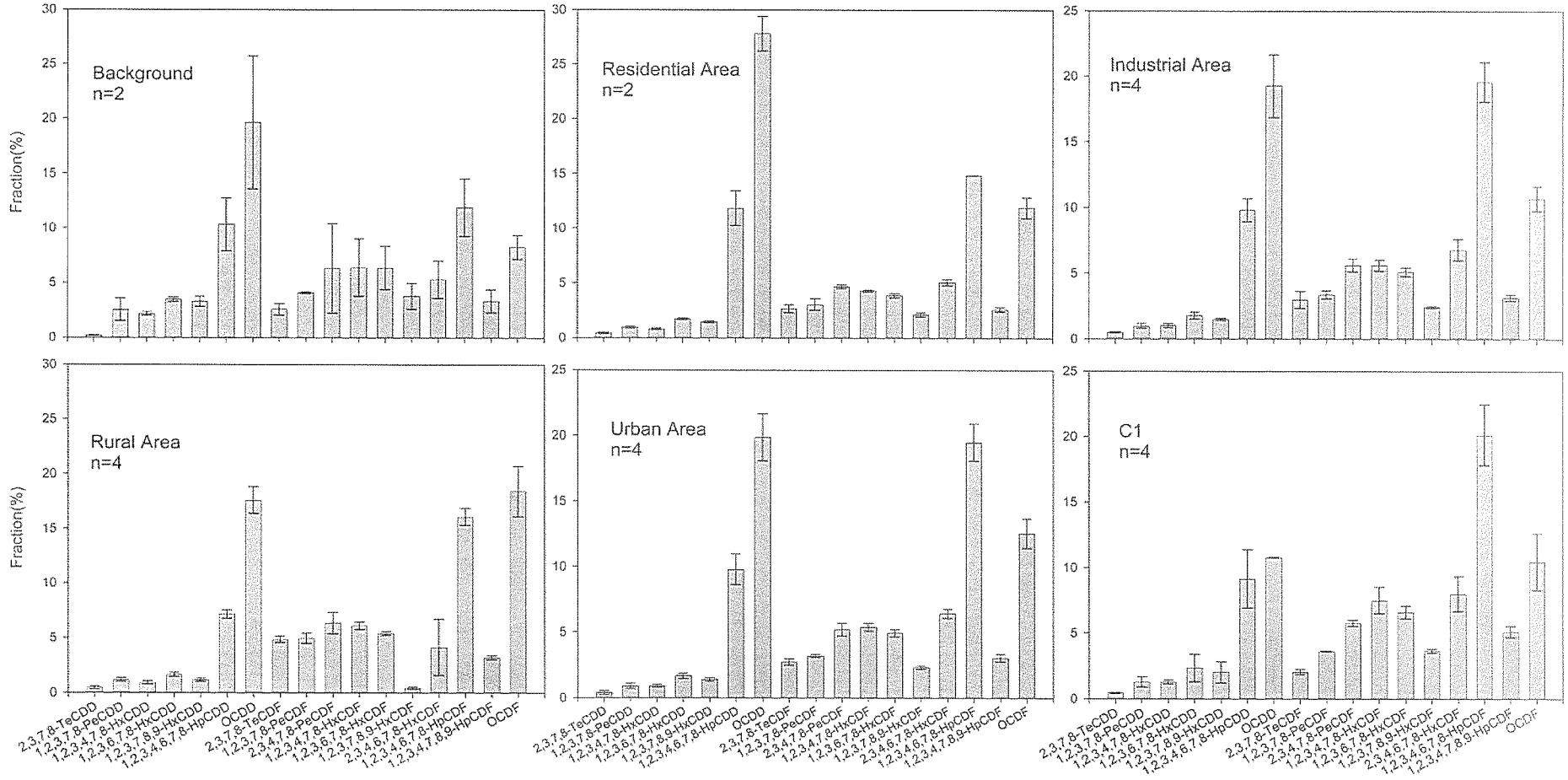

FIGURE 2. Congener profiles of seventeen 2,3,7,8 chlorinated substituted PCDD/Fs in ambient air.

TABLE 4. Mean PCDD/F Concentrations Found in Ambient Air of the Background, Rural Area, Residential Area, Urban Area, Industrial Area, and the Vicinity of C1

	background ($n = 2$)		rural area ($n = 4$)		residential area ($n = 2$)		urban area ($n = 4$)		industrial area ($n = 4$)		C1 ($n = 4$)	
PCDD/Fs	mean pg Nm^{-3}	RSD %	mean pg Nm^{-3}	RSD %	mean pg Nm^{-3}	RSD %	mean pg Nm^{-3}	RSD %	mean pg Nm^{-3}	RSD %	mean pg Nm^{-3}	RSD %
2,3,7,8-TeCDD	0.000	47	0.002	38	0.004	11	0.005	20	0.013	36	0.029	26
1,2,3,7,8-PeCDD	0.002	4	0.004	13	0.008	3	0.013	24	0.027	40	0.088	62
1,2,3,4,7,8-HxCDD	0.002	29	0.003	27	0.007	2	0.013	30	0.027	42	0.085	46
1,2,3,6,7,8-HxCDD	0.002	31	0.005	24	0.015	4	0.024	28	0.049	43	0.166	75
1,2,3,7,8,9-HxCDD	0.002	24	0.004	26	0.012	3	0.020	27	0.041	41	0.142	71
1,2,3,4,6,7,8-HpCDD	0.007	14	0.023	19	0.101	23	0.142	29	0.269	45	0.626	58
OCDD	0.013	7	0.057	19	0.237	16	0.283	18	0.516	41	0.708	36
2,3,7,8-TeCDF	0.002	56	0.016	17	0.022	4	0.039	10	0.076	18	0.136	45
1,2,3,7,8-PeCDF	0.003	39	0.016	16	0.026	8	0.045	16	0.088	31	0.237	35
2,3,4,7,8-PeCDF	0.005	91	0.020	20	0.040	6	0.074	18	0.150	36	0.381	40
1,2,3,4,7,8-HxCDF	0.005	73	0.020	13	0.036	7	0.077	19	0.151	40	0.480	23
1,2,3,6,7,8-HxCDF	0.005	65	0.017	14	0.033	5	0.070	18	0.138	41	0.428	29
1,2,3,7,8,9-HxCDF	0.003	6	0.001	35	0.018	2	0.033	14	0.067	45	0.241	39
2,3,4,6,7,8-HxCDF	0.004	66	0.014	67	0.043	4	0.092	20	0.188	48	0.540	51
1,2,3,4,6,7,8-HpCDF	0.009	57	0.051	11	0.126	10	0.279	20	0.538	45	1.30	25
1,2,3,4,7,8,9-HpCDF	0.002	5	0.010	12	0.022	18	0.043	21	0.085	43	0.330	27
OCDF	0.006	24	0.058	5	0.100	2	0.178	12	0.288	41	0.660	15
total PCDD/Fs	0.070	37	0.320	14	0.849	10	1.43	17	2.71	41	6.57	36
PCDDs	0.028	13	0.097	19	0.383	16	0.500	21	0.941	41	1.84	51
PCDFs	0.042	53	0.223	13	0.466	5	0.930	16	1.77	41	4.73	30
PCDDs/PCDFs ratio	0.742	42	0.432	10	0.819	11	0.536	9	0.535	8	0.377	23
total I-TEQ (pg I-TEQ/ Nm^3)	0.006	58	0.023	17	0.050	4	0.093	16	0.190	38	0.521	41

above results were similar to the results that obtained by Giugliano et al. (*11*). In their study, they measured PCDD/F concentrations at both inlet and outlet of the fabric filter of a municipal solid waste incinerator. Although the removal efficiency on total particulates was as high as >99.9%, removal efficiencies on total PCDD/F emissions and total PCDD/F I-TEQ emissions were ~45% and ~64%, respectively. Based on this, it is concluded that the removal efficiency of the bag filter on PCDD/F emissions was inadequate.

Table 3 shows total PCDD/F I-TEQ emission factors of crematories reported by other research. It can be found that the emission factors found in this study were quite comparable with that found by Takeda et al. (*4*). However, the wide range of PCDD/F emission factors found in other studies indicating that to generalize a universal emission factor for the cremation process might be not possible at this stage. Obviously, this could be due to the intrinsic differences in the types of combustion chamber, the operating conditions, and the types of air pollution control devices among various crematories.

Currently, a total of 32 crematories have been established in the Taiwan area. All crematories are facilitated with a low stack. Half of them were equipped with no air pollution control device, and the rest of them were only equipped with either a bag filter or a cyclone. According to statistical data provided by the Ministry of the Interior, there were ~85 000 cremations (cremation ratio = 66.9%) in 2000. By directly adopting the emission factors of C1 and C2 (assuming C1 and C2 are representative to those crematories installed without and with air pollution control devices, respectively), this study yielded the total PCDD/F I-TEQ emission rate for all crematories was ~0.838 g I-TEQ yr^{-1}. Comparing the above results with other emission sources, we found the total emission from crematories accounted for ~227% and 112% of the emissions from medical waste incinerators (= 0.369 g I-TEQ yr^{-1}) (*12*) and municipal waste incinerators (= 0.750 g I-TEQ yr^{-1}) (*13*), respectively. Unlike what was found in Japan (i.e., crematory emissions accounted for only 0.13–0.29% of I-TEQ yr^{-1} of that emitted from municipal waste incinerators) (*5*), crematories in Taiwan did play a much more important role in PCDD/F emissions.

Significance of PCDD/F Emissions from Crematories on the Surrounding Environment. Figure 2 shows the congener profiles of PCDD/Fs of the background, rural area, residential area, urban area, industrial area, and the vicinity of C1, respectively. All six categories show that the most abundant congeners in the atmosphere were 1,2,3,4,-6,7,8- HpCDD, OCDD, 1,2,3,4,6,7,8- HpCDF, and OCDF, which were consistent with those found in other studies (*14*–*17*). Table 4 lists the mean PCDD/F concentrations for the above six categories as 0.070, 0.320, 0.849, 1.43, 2.71, and 6.57 pg Nm^{-3}, respectively, and the corresponding I-TEQ concentrations were 0.006, 0.023, 0.052, 0.093, 0.190, and 0.521 pg I-TEQ Nm^{-3}, respectively. In principle, the results obtained from this study (except for the concentration of the vicinity of C1) are similar to that found in Germany (e.g., rural area = 0.025–0.070 pg I-TEQ Nm^{-3}; urban area = 0.070–0.350 pg I-TEQ Nm^{-3}) (*18*). In this study, the mean I-TEQ concentration in the vicinity of C1 was ~86.8, 22.6, 10.0, 5.6, and 2.7 times higher than that of the background, rural area, residential area, urban area, and industrial area, respectively. The high I-TEG concentration found in the vicinity of C1 might be because the involved crematory (i.e., C1) had a low stack and was installed with no air pollution control devices.

Yet, it is true that PCDD/F emissions obtained from this study were on a time-weighted-average basis. It did not provide real-time variations on PCDD/F emissions. However, in this study we did find the contents of N_2, O_2, and CO_2 in stack flue gases of C1 and C2 during the sampling period were quite stable (N_2, O_2, and CO_2 concentrations = ~80.6%, 14.6%, 4.80% for C1 and = ~80.2%, 15.8%, and 3.90% for C2, respectively). At this stage, whether PCDD/F emissions were also as stable as the above compounds warrants the need for further investigation. Nevertheless, the results obtained from this study do indicate that the impact of PCDD/F emissions from crematories to the surrounding environment was quite significant. Therefore, it is concluded that a proper control strategy should be taken immediately in order to eliminate PCDD/F emissions from crematory sources.

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Review article

Toxic emissions from crematories: A review

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ABSTRACT

In recent years, the cremation ratio of cadavers has increased dramatically in many countries. Crematories have been identified as sources of various environmental pollutants, being polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs), and mercury those raising most concern. In contrast to other incineration processes for which the number of studies on their toxic emissions is considerable, references related to PCDD/F and mercury emissions from crematories and their health risks are very limited. In this paper, the scientific information concerning these issues, using the databases PubMed, Scopus and Scirus, is reviewed. Results show that in comparison with PCDD/F emissions from other sources, those corresponding to crematories are significantly lower, while those of mercury should not be underrated.

Contents

1. Introduction: incinerators, crematories and toxic emissions

Nowadays, there are more than 1000 crematories in Europe (United Kingdom: 250, France: 125, Spain: 132, Sweden: 68, etc) being the percentage of cremations approximately 37% (ICS, 2006). In 2006, the total number of cremations in Europe was more than 1,500,000 (ECN, 2008). In turn, the countries with the highest number of crematories are China and Japan, with 1549 and 1500, respectively (data from 2006) (ICS, 2006). The pollutants emitted by the combustion of organic matter with presence of other trace elements are: combustion gases (NO_x, CO, SO_2, PM....), heavy metals, and polychlorinated dibenzo-*p*-dioxins and dibenzofurans (PCDD/Fs), among other persistent organic pollutants. Heavy metals and PCDD/Fs, stand out because of their toxicity and capacity for bioaccumulation, which means potential risks for human health. Because of their toxicological properties, together with their persistence capacity, PCDD/Fs were listed by the Stockholm Convention on Persistent Organic Pollutants of 2001 as one of the "dirty dozen" pollutants whose levels should be significantly reduced. With regard to heavy metals, although most elements may be removed from crematory emissions through particulate control devices (EDI, 2006), as the concentrations of mercury may be considerable in human bodies due to the use of dental amalgam fillings, special attention should be paid to this toxic metal.

Environmental policies are becoming more and more stringent with respect to the emission limits of potentially toxic pollutants. However, monitoring surveys are important in order to ensure the proper working of cleaning systems, to control the environmental levels, to assess environmental exposure, to evaluate health risks associated with different pollutant sources, and to identify the relative importance emission sources into the atmosphere in order to adopt

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the necessary measures to protect the environment and the human health. In that context, ambient air monitoring is an essential issue to estimate pollutant emissions such as PCDD/Fs and mercury.

In humans, most PCDD/F and heavy metal body burden comes from the ingestion of contaminants (Parzefall, 2002; Llobet et al., 2008). Some physiologically based pharmacokinetic models have been applied to predict the PCDD/F levels in human tissues (including blood) on the basis of the ingestion of PCDD/Fs through food and human milk. These models are useful not only to investigate past, present, and future trends, but also to help in human health risk assessment due to PCDD/F intake. Using one of these models, Aylward and Hays (2002) reported that absorbed intake levels of 2,3,7,8-TCDD decreased from 1972 to 2002 by more than 95%. Notwithsatnding, and taking into account that food contamination is a direct consequence of the bioaccumulation of pollutants through the food chain, it is important to assess the contribution of the different activities to the environmental concentrations.

In contrast to incinerators, only a few studies have been published on PCDD/F emissions from crematories (Hutzinger and Fiedler, 1993; Takeda et al., 2000, 2001; Luthardt et al., 2002; Wang et al., 2003). Although human cremation is an increasing practice, the number of studies regarding the potential risks derived from crematory emissions is very scarce in relation to the most dangerous compounds (PCDD/Fs and mercury), being even non-existent for other compounds such as NO_x, CO, SO_2, PAHs, etc. In this context, further research on crematories is necessary. In the following sections the information currently available regarding this issue is presented and discussed.

1.1. Incinerators

In recent years, incineration has become one of the most widely used alternatives for waste management. This process is considered by regulators as a strategic option for waste reduction and disposal (Richter and Johnke, 2004; Kollikkathara et al., 2009). In comparison with other waste treatments, incineration presents advantages such as volume reduction, energy recovery, and elimination of pathogen agents (Kuo et al., 2008). However, the public opinion of most developed countries is frequently concerned about the installation of municipal, hazardous, and medical waste incinerators (Domingo, 2002; Singh and Prakash, 2007). Among the pollutants emitted by waste incinerators, PCDD/Fs have generated a lot of controversies (Schuhmacher and Domingo, 2006), mainly because they are among the most toxic environmental compounds (Kogevinas, 2001; Steenland et al., 2004; Mandal, 2005). Although PCDD/Fs, usually referred to as dioxins, are generally produced in many combustion processes (Kulkarni et al., 2008; Zheng et al., 2008; Shen et al., 2009), until a few years ago, incinerators were catalogued as one of the most important sources of toxic emissions, not only PCDD/Fs but also heavy metals (Shibamoto et al., 2007; Zheng et al., 2008). Therefore, incineration has received prolonged special attention, and the concern raised has had significant implications in current regulatory practices (Franchini et al., 2004; Lonati et al., 2007; Kim et al., 2008). Intensive studies have been conducted on various PCDD/F emission sources, including the waste combustion sources, chemical-industrial sources, and other thermal sources.

The installation of modern cleaning technologies to comply with the maximum emission level of PCDD/Fs, established by the European Directive in 0.1 ng I-TEQ/Nm^3 has substantially minimized the environmental impact of incinerators (Glorennec et al., 2005). Although incinerators have traditionally been pointed out as important air emitters of PCDD/Fs (Quass et al., 2004; Kim et al., 2008; Wang et al., 2009), there are many other industrial (cement kilns and power plants) and diffuse (vehicle emissions, domestic coal/wood combustion and natural fires) sources also emitting these pollutants (Fuster et al., 2001).

A number of recent studies have demonstrated that emissions of toxic pollutants from modern municipal solid waste incinerators (MSWIs) have a relatively low environmental impact in comparison with other alternatives of waste disposal or different industrial activities (Domingo, 2002; Schuhmacher and Domingo, 2006; Kao et al., 2007). Although human exposure to PCDD/Fs mainly occurs via food consumption, and more specifically through the ingestion of fatty foodstuffs (Domingo and Bocio, 2007; Llobet et al., 2008), environmental exposure to PCDD/Fs must not be neglected. Among the different pathways of direct exposure to these pollutants, inhalation seems to be the most important route (Nadal et al., 2004).

1.2. Crematories

Although crematories of human beings are also combustors, from a legal/regulatory point of view, these facilities are not considered as incinerators. A human crematory contains one or more combustion units known as cremators, used solely for the cremation of human bodies within appropriate containers. With respect to the potential PCDD/F emissions from crematories, it must be noted that these compounds are formed during combustion processes when chlorinated products such as plastic are burned. In crematories, these plastics may be present as prosthetics or as part of the container. The body also contains a percentage of chlorine, and thus cremation produces PCDD/Fs. Moreover, when waste wood is burnt, the level of PCDD/Fs in the flue gas emissions has been reported to be significantly lower than that derived from other sources (Lavric et al., 2004). Even non-treated wood contains small amounts of chlorine. It means that PCDD/F emissions might be only minimized, but not eliminated (Salthammer et al., 1995). PCDD/Fs are created on particles of soot that enable the hazardous chemical to travel from the incineration site. These particles will eventually settle out onto land (Suzuki, 2007). Contaminated grass enables PCDD/Fs to enter the food chain and it will ultimately be consumed by humans and stored in body fat.

Mercury is another environmental pollutant usually emitted during incineration (Llobet et al., 2002; Ferré-Huguet et al., 2007; Muenhor et al., 2009). In crematories, mercury enters the process because it is present in the body being cremated. Although mercury is only the thirty-sixth most abundant element in the body (at 6 mg for the average body), there is a source of mercury that means serious concern. Fillings made with dental amalgam contain more than 0.5 g of mercury. This metal will leak from these fillings because of mercury's low vapor pressure and add to the mercury levels already present in the body. The intense temperatures of cremation cause the mercury present in the fillings to volatilize, and added to the mercury present in the body may give place to a release of relatively large amount of this toxic metal. Studies have found as much as 200 $\mu g/m^3$ of mercury during the cremation process of a body with dental amalgam fillings (DEFRA, 2003).

Cremators are usually made of high-grade steel plate and lined inside with heavy refractory tile or brick. Most cremators have a variety of automatic controls and use gas for heating the cremator. As a result of the Clean Air Act of 1990, the US EPA first classified crematories as medical waste incinerators, and later as OSW ("Other Solid Waste") incinerators. After an intensive, costly and aggressive testing project in 1999 on working crematories that covered most types of emissions, including particulate matter, carbon monoxide, and mercury, done jointly with the Cremation Association of North America and reviewing information presented, the US EPA decided not to regulate human or animal crematories. As a result of the US Cremation Association's meeting with the US EPA in November 1991, it became known that the original regulations proposed for crematories were based on no actual test data. This inspired the US Cremation Association to have substantial testing performed to increase everyone's knowledge base. This testing was completed in 1999 and the data became US EPA's foundational information in their national emissions inventory (CANA, 2009).

Among the concerns raised by crematories, there does not appear to be any risk to the environment or the operator under normal conditions, when cremating someone who has been treated with radiation therapy. Generally, radioactive implants are removed prior to the cremation. Cremation of radio-nuclides, or radioactive "seeds," that might remain in a body does not pose a problem due to the rather small number of cremations that occur annually and their relatively short half-life. With respect to the possibility that any element in human remains that would be harmful to the environment as a result of cremation it seems the response is no. Human remains consist of 85% moisture, which vaporizes during the cremation process, 10% combustible solids which release approximately 1000 BTUs (British Termal Unit) per pound and transfer from a solid to vapor state, and 5% non-combustible solids which absorb heat and energy from the cremation process and remain as solids (bone fragments and ash materials) when cremation is completed. The 5% non-combustible solids are usually returned to the family (DEFRA, 2003; CANA, 2009).

In fact, a crematory furnace consists of a refractory chamber in which the mortal remains to be cremated are placed. The coffin is forwarded into the furnace by a conveyor fixture. In the so-called "cold type" furnaces, the coffin is placed inside at a temperature of about 300 °C. In the "warm types", the temperature is 800 °C or higher. The cremation is carried out in furnaces which are fired using oil or natural gas. There are also some which run on electricity. In more modern installations the remains are transported to a post burning chamber located beneath where they are post-fired (minimum temperature 850 °C). The cremation time is about 1.2–1.5 h in the warm types and lasts for 2–2.5 h in the cold types. The average volume of waste gases is approximately 1200 Nm^3/h for gas and oil fired furnaces and approximately 600 Nm^3/h for electrically heated furnaces. The waste gases from the muffle are transported via the post-firing chamber and the recuperator and are subsequently purified by cyclones and fabric or electro filters (EDI, 2006).

In Europe, crematoria are certainly not of high relevance for the total emission of PCDD/Fs. The European Dioxin Air Emission Inventory, whose results were published by Quass et al. (2004), reported a 1985 upper estimate of 28 g I-TEQ/year, and a 2005 estimation of 13 (minimum) and 22 (maximum) g I-TEQ/year from cremation (incineration of corpses). The emissions corresponding to the total of sources considered (g I-TEQ/year) were 13,690 for 1985, and 1963–3752 for 2005. Taking this into account, up to now, the data from crematoria suggest that in most cases these installations may be disregarded. However, from the local view crematoria without or low quality flue gas cleaning might have adverse environmental impact. Therefore some spot-check measurements might be necessary to assess the possible emissions and confirm the currently available data, especially in those countries which did not provide any data. In a PCDD/F emission inventory for the Tarragona Province (Catalonia, Spain) that we performed in 1999, a total of 2.24 g I-TEQ/year was found, with a contribution for crematoria of only 0.00029 g I-TEQ/year (Fuster et al., 2001).

On the other hand, for years, The Cremation Association of North America (CANA) has witnessed the concern surrounding cremating human remains, and the corresponding release of primarily two emissions: particulate matter (PM) and mercury. PM can be defined as solid particles suspended in a gas as a byproduct of all combustion processes, including cremations. PM emissions are released into the environment in many ways, including through residential and commercial fuel-based heating, cars, trucks, restaurant grills and fireplaces. None of these sources of PM have any emission controls to reduce, monitor or limit PM emissions, while crematories have emission controls as part of their design to limit the amount of PM entering the atmosphere (CANA, 2009). On the other hand, mercury is derived from the use of silver amalgam in dental fillings that is released into the environment during the cremation process.

2. PCDD/F emissions from crematories

In recent decades, PCDD/F emissions from refuse incinerators became a serious problem in a number of developed countries. Numerous studies focused on estimating the quantity of PCDD/Fs emitted from municipal and industrial waste incinerators have been published. In contrast, in those countries with a notable ratio of cremation of human bodies, until recently emissions of PCDD/Fs from crematories were, in fact, unknown. For example, in Japan, where a 98.8% of dead bodies were cremated in 1997 (the highest percentage in the world), and with 1607 crematories in operation at that time, only a few studies have been carried out on PCDD/F emissions from crematories.

The reports about PCDD/F emissions from crematories in the world are really limited (Hutzinger and Fiedler, 1993; Federal States Pollution Control Committee, 1994; Eguchi et al., 1996; Fiedler, 2006). In Japan, Eguchi et al. (1996) reported the concentration of PCDD/Fs from a crematory to be 0.14–2.56 ng TEQ/Nm^3. This was less than the concentration of PCDD/Fs from crematories in Germany, 8 ng TEQ/Nm^3 found by Hutzinger and Fiedler (1993). In 1994, a working group of a subcommittee of the German Federal State Pollution Control Committee reported the levels of PCDD/Fs for 13 crematories from Germany. It was found that the concentration of PCDD/Fs from those crematories was 0.1–14.4 ng TEQ/Nm^3, and almost all of them were more than 1 ng TEQ/Nm^3 (The Working Group of Subcommittee, 1993).

Since in Japan, about 99% of dead bodies were cremated in a considerable number of crematories, it seemed necessary to investigate crematories of various types to estimate the quantity of PCDD/Fs emitted from these facilities. Takeda et al. (2000) measured the concentrations of PCDD/Fs in emission gases from 10 Japanese crematories. The relationship between PCDD/Fs and several factors such as structure, equipment, and operational state of the crematory were assessed. Furthermore, emission of PCDD/Fs from all Japanese crematories was estimated. The most relevant results were the following: 1) total concentration of PCDD/Fs from a crematory was 2.2–290 ng/Nm^3, and TEQ concentration was 0.0099–6.5 ng TEQ/Nm^3, 2) the concentration of PCDFs was higher than that of PCDDs, especially tetrachlorodibenzo-*p*-furans (T4CDFs), being 2,3,7,8-T4CDF detected in almost all samples, 3) for a homologue pattern of PCDFs, the concentration of T4CDFs was high, while that of the higher chlorinated compounds was low. For that of PCDDs, two patterns were identified: (a) a mountain shape pattern with peaks of hexachlorodibenzo-*p*-dioxins (H6CDDs), which was similar to the typical pattern of waste incinerators, and (b) a pattern the same as the PCDFs pattern, with decreasing concentrations when increasing the degree of chlorination, 4) the total concentration of PCDD/Fs from crematories whose dust concentration was less than 50 mg/Nm^3 tended to be low, 5) the total concentration of PCDD/Fs was highest in the first 20 min from the start, 6) it was found that sex and age of dead body did not affect the concentration of PCDD/Fs, 7) the existence of a dust collector, temperature of the secondary combustion chamber, and the number of main combustion chambers connected to a secondary combustion chamber affected the concentration of PCDD/Fs, and 8) the total amount of PCDD/Fs emitted from crematories in Japan was estimated to be 8.9 g TEQ/yr. Takeda et al. (2000) concluded remarking that the number of data was not enough to grasp the state of PCDD/F emissions from crematories in Japan.

In a subsequent study of the same research group, PCDD/Fs and co-planar PCB concentrations in flue gases from 17 Japanese crematories were measured in fly ashes and bottom ashes (mainly bone) from several crematories to assess the state of PCDD/F emissions from those facilities (Takeda et al., 2001). The effects of several factors were discussed to prevent PCDD/F emissions from crematories. Total concentration (normalized by 12% O_2) of PCDD/Fs ranged from 4.9 to 1200 ng/Nm^3, and TEQ concentration ranged from 0.064 to 24 ng TEQ/Nm^3. According to the results of that study, these measures for existing crematories were

recommended in order to reduce PCDD/F emissions: 1) keeping the temperature at 800 °C in main/secondary chambers during a whole cremation, and 2) lowering the temperature in the dust collector. For newly installed crematories, Takeda et al. (2001) suggested the following measures to prevent PCDD/F emissions, including the measures for existing ones: 1) connecting one secondary chamber to one main chamber, 2) installing the high efficiency dust collector and reducing dust concentration to less than 0.01 g/Nm^3, and 3) installing the sampling point for monitoring of PCDD/Fs.

In Germany, Luthard et al. (2002) analyzed the concentrations of PCDD/Fs and PCBs, and estimated total TEQ, in flue gas samples from eight different sources (two municipal waste incinerators (MWI), one hazardous waste incinerator (HWI), two sintering plants, one cement kiln, and two crematories). The highest TEQ values were found at crematory No. 2, the sintering plants, and at the MWI with older technology. TEQ emissions below the 0.1 ng/Nm^3 limit were found at the modern MWI, the HWI, and at the cement kiln.

In a previous study also conducted in Germany by Hutzinger and Fiedler (1993), PCDD/F concentrations of about 8 ng TEQ/Nm^3 were detected in the stack flue gases of crematories, while in the UK, a study conducted by the Warren Spring Laboratory found a mean PCDD/F concentration of 46 ng TEQ/Nm^3 (11% oxygen) for the cremation process (Eduljee and Dyke, 1996). In the USA, PCDD/F emission rate (expressed as TEQ) for the crematory source was found about 9.1 g TEQ per year (US EPA, 2000), which was in the range of that found in the UK: 1–35 g TEQ per year (Eduljee and Dyke, 1996), but higher than that reported in Japan: 1.3–3.8 g TEQ per year by Takeda et al. (2001). The wide range of PCDD/F concentrations arising from various crematories was believed to be due to their intrinsic differences in operation conditions, air pollution control devices, and involved incinerating materials (Takeda et al., 2001).

In general, total PCDD/F emissions from crematories are relatively small compared with those from MWIs. For example, in the study conducted in Japan by Takeda et al. (2001) the crematory emission accounted for only 0.13–0.29% of that emitted from MWIs. However, it should be noted that most assessed crematories were equipped with a low stack, and were situated in the proximity of residential areas. In particular, most of them did not adopt any air pollution control device to eliminate PCDD/F emissions from stacks. Based on these, it could be expected that PCDD/F emissions from a crematory might significantly affect its surrounding environment.

Recently, the cremation ratio has increased dramatically in many countries (Santarsiero et al., 2005; ECN, 2008). In Taiwan, the cremation ratio was expected to increase from 66.9% in 2000 to 85.0% in 2005 (Wang et al., 2003). In the USA, the cremation ratio also increased significantly from 15.2% in 1987 to 25.0% in 2000, and was expected to reach 37.0% in 2010 (US EPA, 2001). Based on these data, it is expected that crematories will play an important role on PCDD/F emissions not only in countries such as Japan or Taiwan, but also in many other countries. In addition to PCDD/Fs, PCBs and PAHs, as well as total suspended particles (TSP) from crematories might also be a cause of problems to human health.

Wang et al. (2003) characterized PCDD/F emissions from Taiwanese crematories and assessed their impacts on the surrounding environment. Two crematories (C) located in southern Taiwan were investigated, including C1 with no air pollution control device installed, and C2 with a bag filter. The mean PCDD/F emissions (11% oxygen) from the stacks of Cl and C2 were 2.36 and 0.322 ng I-TEQ/Nm^3, respectively, while mean emission factors for C1 and C2 were 13.6 and 6.11 μg I-TEQ/body, respectively. The removal efficiency of the bag filter on PCDD/Fs was 55.1%. The estimated PCDD/F emission rate for all crematories in Taiwan was 0.838 g I-TEQ/year. In an emission inventory of PCDD/Fs in Taiwan, a total of 67.25 g I-TEQ of PCDD/Fs released annually was estimated (Chen, 2004). Secondary copper smelting accounted for more than 39% of the total PCDD/F emissions, being higher than those from all waste incinerators combined (23.7%). Based on that inventory, PCDD/F emissions from crematories would be relevant, as they were 227% and 22.4% of the annual emissions from all medical waste incinerators and MWIs, respectively. To assess the impact of PCDD/F emissions from a crematory to the surrounding environment, ambient air samples were collected from the downwind site of C1 with the maximum ground concentration (Wang et al., 2003). The estimated maximum ground concentration at the downwind site of C1 (0.521 pg I-TEQ/Nm^3) was much higher than that found at the background, rural area, residential area, urban area, and industrial area (0.006, 0.023, 0.052, 0.093, and 0.190 pg I-TEQ/Nm^3, respectively). The authors indicated that the high I-TEQ concentration found in the vicinity of C1 might be due to the fact that the involved crematory had a low stack, being installed with no air pollution control devices. It was concluded that PCDD/F emissions from a crematory did significantly affect its surrounding environment, and therefore, a proper control strategy was essential in order to eliminate PCDD/F emissions from crematories. A summary of the most relevant results corresponding to some of the above studies is shown in Table 1.

PCDD/F emissions from well-maintained cremators were measured (Edwards, 2001) and found to be much lower than previous measurements made in the early 1990s. The average emission was 61 ng I-TEQ per cremation, giving a UK total PCDD/F emission from crematoria of 0.027 g I-TEQ, which meant 0.008% of the UK total emission of 325 g I-TEQ. From measurements made in the early 1990s, about 5% of UK emissions to air of PCDD/Fs were attributed to crematoria. Recent emission levels were similar to PCDD/F emission limits in Waste Incineration Directive 2000/76/EC. Although that Directive does not apply to crematoria, its emission limits indicate what good exhaust gas treatment can achieve.

3. Mercury emissions from crematories

In addition to PCDD/F emissions from crematories, another environmental aspect that has received particular attention is the release of mercury. This element is liberated both because dental amalgams that are unstable at cremation temperatures (650–700 °C), and because of the free mercury metal is highly volatile (Nieschmidt and Kim, 1997). In Switzerland, Rivola et al. (1990) estimated that mercury contamination due to cremation varied in 1988 between 45.8

Table 1
A summary of data concerning air emissions of dioxins and furans (PCDD/Fs) from crematories in different countries.

Country	Emissions of PCDD/Fs	Remarks	Reference
Germany	8 ng TEQ/Nm^3	–	Hutzinger and Fiedler (1993)
Germany	0.1–14.4 ng TEQ/Nm^3	In almost all the 13 crematories assessed PCDD/F emissions were higher than 1 ng TEQ/Nm^3	Federal States Pollution Control Committee (1994)
United Kingdom	46 ng TEQ/Nm^3	1–35 g TEQ/year	Eduljee and Dyke (1996)
Japan	0.0099–6.5 ng TEQ/Nm^3	Total concentrations in 10 crematories: 2.2–290 ng/Nm^3	Takeda et al. (2000)
Japan	0.064–24 ng TEQ/Nm^3	Total concentrations in 17 crematories: 4.9–1200 ng/Nm^3	Takeda et al. (2001)
Germany	0.24 (Crem. 1) and 3.71 (Crem. 2) ng TEQ/Nm^3	Six industrial plants and two crematories were assessed. Among the 8 facilities, the highest TEQ values were found at Crem. 2	Luthardt et al. (2002)
Taiwan	2.36 (Crem. 1) and 0.322 (Crem. 2) ng TEQ/Nm^3	The mean emission factors for Crem. 1 and Crem. 2 were 13.6 and 6.1 μg I-TEQ/body, respectively	Wang et al. (2003)

and 79.0 kg, based on both data from the sample analyzed and the fact that 55.5% of Swiss funerals were cremations, the average age of death was 73, and that 70% of the people of that age retained some of their teeth. According to these authors, mercury contamination by cremation comprised 0.61–1.53% of the total mercury contamination produced by all waste incineration methods in that country. Also in Switzerland, Matter-Grütter et al. (1995) determined the amount of mercury released at two crematoria. A total of 60 mercury "output" calculations were carried out by the Swiss Material Testing Institute. The amount of mercury initially present ("input") in the dentitions of 54 deceased persons was assessed from their post-mortem dental radiographs and by clinical examination. The correlation between the "input" and the "output" was 0.93, irrespectively of the age at death. However, the "input" was calculated to be 1.8 times higher than the "output" for the deceased people with amalgam restorations. In a blind study, the difference was 1.3 times. The main source of mercury was undoubtedly the amalgam restorations. The amount of mercury recorded during the cremation of 88% of the deceased people without amalgam restorations was under the accepted level of 200 μg/m^3. However, in three cases, the amount of mercury was slightly higher than 200 μg/m^3. In contrast, the amount of mercury recorded during the cremation of only 18% of the deceased people with amalgam restorations was less than the accepted level of 200 μg/m^3. The amount of mercury contamination during cremation as a result of amalgam fillings was so low that no additional preventive measures were required at those crematoria.

In Japan, Yoshida et al. (1994) measured the amount of mercury released at three crematories. The concentration of atmospheric mercury at those three facilities ranged from 4.3 to 19.7 ng/m^3. This rank was nearly identical to the levels found in the control (university campus) area, being also similar to the general levels of atmospheric mercury in the country. The amount of mercury released from one of the crematories was subsequently estimated using official published statistical data in Japan and calculated as follows: sigma[(age specific number of dead that were cremated) × (the number of restored teeth by age category) × (mercury content per amalgam filling (0.6 g))] × (prevalence rate of restoration with amalgam). The amount of mercury released from this crematory was estimated to be approximately 9.4 kg per year, or a daily release of 26 g into the ambient air, which indicated that mercury released by cremation was similar to that from other man-made sources.

A number of reports have been published giving estimates on the amount of mercury released into the atmosphere by crematoria and the concentration of soil mercury found around crematoria in the USA and England (Mills, 1990; Kunzler and Andree, 1991; Basu and Wilson, 1991; Burton, 1991; Hogland, 1994). As in other countries, in New Zealand, a high percentage of deaths are followed by cremation and this figure is expected to rise in the future. This increasing use of cremation as the method of corpse disposal, coupled with the fact that each amalgam restoration is approximately 50% mercury, implied that a significant amount of mercury was being emitted into the environment every year. In that country, Nieschmidt and Kim (1997) using cremation data available from the International Cremation Statistics (ICS, 1992) and the calculations of Burton (1991), estimated emissions about 22.8 kg of mercury per year, and that global annual mercury emissions would total 6962 kg. Globally, atmospheric mercury emissions from crematories of this magnitude would account for about 0.8% of total anthropogenic mercury emissions (based on the estimates of Nriagu and Pacyna, 1988). Recently, Santarsiero et al. (2006) reported some preliminary results concerning mercury and total particulate matter emissions during three cremation processes in Italy. A mercury concentration ranging from 0.005 to 0.300 mg/m^3 and a mercury emission factor ranging from 0.036 to 2.140 g/corpse cremated were obtained. The total particulate matter concentration range was from 1.0 to 2.4 mg/m^3.

It must be noted that mercury (as well as other pollutants) emissions from crematoria are not covered by the European Union regulations. Currently, matters of crematories are the responsibility of local Authorities. However, mercury emissions have been the subject of the OSPAR Recommendation. In fact, the OSPAR document, namely OSPAR Recommendation 2003/4 on Controlling the Dispersal of Mercury from Crematoria (OSPAR, 2003) in the OSPAR Convention Area identified crematoria as producing a significant source of mercury in the environment and listed various options, in terms of the best available technologies (BAT) to reduce and control mercury emissions. The reports on emissions made by parties involved with this recommendation, will provide an indication of the effectiveness and if further action is needed. Future trends in mercury emission are difficult to predict since they are strongly affected by the following variables: the number of cremations per year, the number of amalgam fillings and the related mercury, and the content present at cremation. Therefore, for the assessment of current and future mercury emission factors the following must be taken into account: the amalgam fillings and related dentistry practices used in the past, those currently used and those to be used in the near future, and the distribution of dental amalgams within the population (Santarsiero et al., 2006).

Recently, in the 9th International Conference on Mercury as a Global Pollutant (ICMGP) held in Guiyang, China, Reindl (2009) concluded that there were significant uncertainties in North American data, as few studies existed concerning mercury emissions from crematories. North American demographics may be different than European for restoration sizes, composition and number. An increase in emissions for the next several decades can be expected followed by a decrease. Reindl (2009) recommended collecting information on the amount of mercury released per cremation, mass balance, air, ash, deposited on crematorium surfaces and speciation of air emissions, which is essential for regulators.

The AEAT study for PCDD/Fs above cited (Edwards, 2001), included also measurements of mercury emissions 18 crematories in the UK. Most mercury in bodies is in dental amalgam fillings, and as the number of fillings varies from person to person, a wide variation of mercury emissions could be expected. The measurements showed 6 crematories with very little mercury and a considerable variation in emissions from the other crematories. The average emission across all the crematories was 0.9 g of mercury per crematory, an amount that was less than the calculated emission factor of 3 g per crematory that was used to estimate UK mercury emissions from crematoria, and reported in the National Atmospheric Emission Inventory (DEFRA, 2003). The calculations gave a range of mercury emissions from crematories as 5.3%–15.7% of UK mercury emissions to air in 2000. Crematoria emissions in the UK are expected to increase from 0.4–1.34 tonnes in 2000 to 0.68–2.2 tonnes in 2020 unless gas cleaning of exhausts is introduced (DEFRA, 2003).

4. Occupational and environmental health effects from crematories

Information on occupational exposure to individuals working in crematories is particularly scarce. To the best of our knowledge, only a study in the UK has examined this potential exposure (Maloney et al., 1998). By measuring the levels of mercury in hair, it was concluded that exposure to mercury vapor by workers in crematoria was rather low compared with others who were occupationally exposed to this element. Of the 97 crematoria workers assessed, 3% had concentrations higher than 6 μg/g, generally considered as a tolerable concentration for mercury in hair. According to their results, the authors considered that there was sufficient evidence to warrant emission monitoring and control in crematoria workers.

On the other hand, and related with environmental exposure to emissions around crematories, Dummer et al. (2003) investigated the risk of stillbirth, neonatal death, and lethal congenital anomaly among babies of mothers living close to both incinerators and crematories in Cumbria, north west England, 1956–1993. A significant increase was noted during this period on the risk of stillbirth closer to crematories. The risk of

anencephalus was also significantly increased during the same period. Although most (92%) cases of anencephalus were stillborn, the significantly increased risk of stillbirth remained after exclusion of anencephalus cases from the analysis. From 1972 onwards there was an increased risk of all other congenital anomalies, excluding neural tube defects and heart defects, with increasing proximity to crematoriums, which was significant for the period 1983–1993. These findings remained significant after exclusion of the most influential births. It was concluded that threw was an increased risk of lethal congenital anomaly (specifically spina bifida and heart defects) in relation to proximity to incinerators, and an increased risk of stillbirth and anencephalus in relation to proximity to crematoriums (Dummer et al., 2003). However, a causal effect from the statistical associations could not be inferred, since as it was noted there are a number of factors (medial advances, clinical characteristics, etc.), that may also affect the results, which is one of the main drawbacks or limitations of all geographical epidemiological studies. On the other hand, since incinerators and crematories are sources of harmful substances, it is worth to keep investigating their potential effects on public health.

5. Conclusions

An exhaustive revision on the scientific literature by means of the databases PubMed (http://www.ncbi.nlm.nih.gov/pubmed/), Scopus (http://info.scopus.com/) and Scirus (http://www.scirus.com/) using crematories/crematoria, toxic emissions, dioxins and furans, mercury, and health risks as searching terms, has detected only a scarce number of published studies. With respect to organic emissions from crematories, PCDD/Fs have been the most investigated. In comparison with PCDD/F emissions from waste incinerators (municipal, hazardous and medical), those corresponding to crematories are significantly lower. However, if the facilities are installed near nuclei of population, any specific filter is used, and the high of the stack is relatively low, human health risks should not be discarded, and a monitoring program should be carried out. In relation to mercury, which is a highly volatile element, to date the most important source has been dental amalgams containing mercury, which are unstable at cremation temperatures. In recent decades, serious efforts have been made in most developed countries in order to reduce the levels of environmental mercury. In this sense, if mercury emissions from crematories are not properly controlled, these facilities could mean a relatively important source of atmospheric pollution. Another relevant aspect of toxic emissions of crematories is the noted lack of general regulations of these emissions. For example, neither the European Union nor the US EPA has established specific recommendations for crematories.

In summary, because of cultural and other reasons, in recent years the cremation ratio has considerably increased in many countries, a trend that is expected will continue in the near future. Therefore, we think that crematories must also be among the facilities whose emissions should be specifically regulated and monitored.

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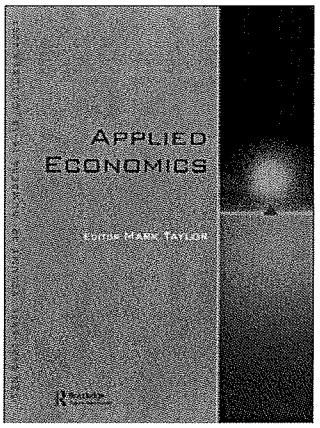

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Directional heterogeneity of environmental disamenities: the impact of crematory operations on adjacent residential values

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Directional heterogeneity of environmental disamenities: the impact of crematory operations on adjacent residential values

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A hedonic study of residential house sales in Rawlins, Wyoming, was conducted to estimate the impact of an environmental shock from a new point source upon adjacent residential property values. We use a unique data base of house sale prices and associated house attributes, including structural and neighbourhood characteristics and geographic distances and directions from the source of the shock, atmospheric emissions from a new crematory. Our data spans 27 months of house sales: 7 months before, and 20 months after the startup of crematory operations. Results indicate that proximity, measured both in terms of direction and distance from the crematory, imparts a statistically significant negative impact on average house sale prices – an increase of 0.3 to 3.6% of average sale price for every one-tenth mile increase up to one-half mile in distance away from the crematory, but depending on direction from the crematory. This distance benefit increases somewhat with calendar time only for houses located west of the crematory.

I. Introduction

Residential property values depend both on physical and locational attributes. Attributes include structural, neighbourhood and environmental characteristics, all of which may impact the selling price of a property. Indeed, housing markets are one of the few places where environmental amenities are traded in formal markets along with physical amenities. As such, for decades, economists have used hedonic property value techniques to measure monetary equivalents of a variety of environmental quality changes that affect consumers' welfare via their purchase and consumption of the good 'housing.' Recent examples include air quality (Kiel and McClain, 1995; McMillen and Thorsnes, 2003), aesthetic views (Bourassa *et al.*, 2004) and proximity to other amenities or disamenities such as proximity to natural areas (Thorsnes, 2002) or landfills (Ketkar, 1992).

Hedonic property value studies are useful if they provide empirical evidence that selling prices of a heterogeneous market good reflect alternative levels of amenities (good or bad). Given the sometimes

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elusive nature of environment-related benefits and costs, such information is particularly useful as it provides evidence that people are willing to pay more for higher levels of environmental quality.

When proximity to an environmental externality affects value, both direction and distance may matter. For example, many point sources of pollution produce either noticeable odours or airborne pollutants so that prevailing winds (or lack of air movement) create directional heterogeneity in distance effects. We demonstrate that if directional effects are present but ignored, one might observe no proximity impact on house value even though impacts are indeed present but are not the same in all directions. To date, published hedonic property value studies that employ distance measures pay little attention to direction. While some of these studies account for spatial trends (e.g. Gillen *et al.*, 2001), such as spatial autocorrelation in model error terms, these studies do not specifically address distance profiles as a function of direction. Herriges *et al.* (2005) and Cameron (2006) are the only studies we are aware of which empirically examine direction and distance impacts of an environmental disamenity using a hedonic property value model. But these last two studies disregard the potential impact of spatial autocorrelation and heteroskedasticity upon their reported results. Here we account for direction and distance impacts and test for and make appropriate corrections for spatial autocorrelation and spatial heteroskedasticity.

The following section explains our approach to assessing the impacts on residential property values of proximity to the shock of exposures to atmospheric emissions from a crematory of whose start-up operations adjacent property owners had never been informed.[1] Section III describes our data and model specification, and our results and value estimates are reported in Section IV. Section V concludes.

II. Hedonic Model and Pre-Testing

This section outlines a basic hedonic model to assess the marginal impact on house sale prices of proximity to a point source, environmental disamenity shock, holding constant all other attributes important to these values. The literature has identified several empirical issues that must be addressed in order to optimize both statistical efficiency and precision of estimates using hedonic techniques. The most common and addressable issues include choice of functional form, bias due to omission of relevant explanatory variables and definition of the extent of the market to be examined (sampling).

Following Rosen (1974), this study uses a first-stage hedonic model, in which the hedonic price function is estimated using a sample of prices and characteristics of observed 'transacted' properties,

$$\text{SALEPRICE}_i = \alpha + \sum_j \beta_j D_{ij} + \sum_k \beta_k H_{ik} + e_i \quad (1)$$

where SALEPRICE_i denotes nominal selling price of house i ($i = 1, \ldots, N$), which is a function of two sets of observed variables, D and H. The j variables in D describe the house in terms of its date of sale, and distance and direction from the environmental disamenity. The k variables in H describe the house in terms of its general structure (living area, number of bathrooms, etc) and its accessibility to public facilities. α is a constant term. Expression (1) defines the hedonic price function as a locus of equilibrium points. If the property attributes observed are independent of any not observed, Bajari and Benkard (2005) show this implies the existence of a hedonic price function even if the housing market is imperfectly competitive and lacks a continuum of types.

While choice of functional form for (1) is somewhat arbitrary for the researcher, we choose a double-log specification based upon a number of preliminary regressions (not reported) and statistical testing of goodness-of-fit. The specifications reported in Table 2 emerge as clearly best in terms of statistical fit. These results are consistent with Cropper *et al.* (1988) who show that the double-log form usually performs best relative to linear, semi-log inverse semi-log, and other quadratic forms for first-stage hedonic models, both in terms of model flexibility and ability to measure marginal prices in the presence of misspecifications. Also, functional forms that are too general may not prove robust to small misspecifications (Cassel and Mendelsohn, 1985).

The hedonic technique is especially useful for determining values of general reductions in 'receptor effects,' i.e. a single disamenity 'bundle' comprising several effects such as noise, foul odours, or bothersome visual effects. When these general receptor effects dominate, identification and valuation of specific environmental impacts, which include exposures to specific contaminants, can be problematic unless detailed information is available

[1] Thus the housing market could not anticipate the likely effects of crematory operations.

on all individual effects in the disamenity bundle (see e.g. Palmquist *et al.*, 1997). Since individual effects within the bundle are plausibly correlated, omission of any one or a subset of effects from the hedonic regression will bias the predicted impacts of remaining disamenities accounted for in the regression. However, if assessment of general effects is the focus, windfall losses to receptors will equal to the total decline in predicted property values (Polinsky and Shavell, 1976; Palmquist, 1991). These losses are often expressed in terms of proximity to the disamenity source. Losses due to proximity to an environmental disamenity are larger if the proximity-related decline in property value also includes a slowing of appreciation rates (Mitchell and Carson, 1986). Our goal is to assess the decline in predicted residential property values associated with proximity to a newly installed crematory, of whose planned installation and start-up adjacent property owners were unaware. Proximity (distance and direction) is assumed to capture general receptor effects associated with living near the environmental shock from the atmospheric emissions of the new crematory operation. To assess the proximity-related change in predicted values fully, we also assess the value impact of emitter effects on house price appreciation rates.

While assessment of general proximity effects greatly simplifies model specification and data requirements, other potential estimation problems linger. For instance, if an environmental disamenity affects a large area, and/or there are multiple sources of changed emissions, hedonic price functions can shift, implying that the total predicted change in aggregate property values serves only as an upper bound for the true change in value (Bartik, 1988). That is, marginal changes in property values as measured by the slope of a hedonic price function need not equal that aggregate change in value which is determined by general equilibrium adjustments involving induced relocations and changes in population and housing supply. We limit our analysis to marginal changes since the externality we consider is localized relative to the size of the housing market.

Sample selection bias represents another potential estimation issue because, say, more expensive homes might more likely be offered for sale when confronted by a disamenity shock. We believe this issue to be insignificant for this study since residences in the neighbourhood subjected to the shock are very nearly all middling in their attributes and residents. Also, Jud and Seaks (1994) conclude that ignoring the sample selection issue leads to an average error of only 1% in housing price change estimates.

More importantly, since unobserved or omitted variables in hedonic regressions are often locationally correlated, 'spatial autocorrelation' is frequent in hedonic regressions. Though spatial autocorrelation does not bias ordinary least squares coefficient estimates and thus benefit measures (Leggett and Bockstael, 2000; Kim *et al.*, 2003; Neill *et al.*, 2007), estimates can be inefficient, which leads to biased SEs and inaccurate hypothesis tests.[2] We conducted a series of Kelejian and Robinson (1992) tests to check for any significant presence of spatial autocorrelation in the data of our case study. These tests failed to confirm spatial autocorrelation in all our Table 2 model specifications. However, White (1980) tests failed to reject spatial heteroskedasticity in these specifications. Therefore, the results presented in Table 2 discussed below, use White's (1980) heteroskedasticity consistent covariance matrix to address potentially biased SEs in our ordinary-least-squares (OLS) estimates.

Finally, heterogeneity in distance effects with respect to direction from an environmental disamenity can potentially obscure what might otherwise be a clear price–distance relationship. With directional diffusion of airborne pollutants, one would naturally expect prevailing winds to exacerbate effects for some neighbourhoods while virtually eliminating effects from others, even where distance to the upwind area from the pollution source is considerably less. Also, direction-specific geographic features such as hills and forests can enhance or counter the impact of prevailing winds. If distance and direction are correlated, omission of direction from the hedonic model will result in omitted variable bias of the coefficient estimate for distance. Their direction of drift plausibly affects the impact of mobile pollutants on property values. Surprisingly, almost all published hedonic property value studies that employ distance-to-source as their proximity measure do not include information on orientation of a property to the pollution source. Palmquist *et al.* (1997), Gillen *et al.* (2001), Herriges *et al.* (2005) and Cameron (2006), are the sole exceptions we have been able to identify. But the first two, while acknowledging 'importance' of direction, do not formally consider its effects in

[2] Even if spatial correlation were present, an assumption that any spillovers among neighbouring sites are strictly pecuniary would permit the coefficient on the pollution variable in an OLS hedonic price regression to be interpreted as the complete marginal effect of pollution on house value (Small and Steimetz, 2006). Strictly pecuniary effects imply that the value of neighbouring sites affects the sale price of a particular site but does not affect the amenities of that site.

their empirical framework. This leaves only the latter two studies that explicitly account for distance with directional heterogeneity by combining distance and direction (in the form of upwind and downwind siting for Herriges *et al.*, and of polar coordinates for Cameron) into the hedonic property value model. Our data lack sufficient detail on direction to implement the Cameron (2006) framework. However, we know the location of each sample property within one of eight possible 45° regions (N, NE, E, SE, etc). This enables us to establish a reasonable estimate of the combined influence of distance and direction effects by introducing dummy variables for direction to account for directional interactions in our hedonic OLS regressions.

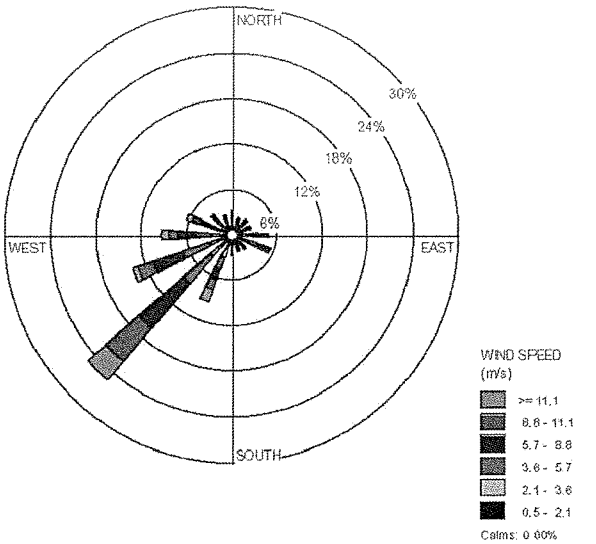

Fig. 1. Rawlins, Wyoming wind rose

III. Data

Our data consists of all 372 single family home transactions in the city of Rawlins, Wyoming, between January 2004 and March 2006. These sales are dispersed throughout the Rawlins city limits. Rawlins, population 8538 in 2000, and 8633 in 2004, is located in Carbon County in South Central Wyoming. Only one settlement with more than 1000 people lies within 100 miles, and that one settlement is nearly 40 miles distant. Rawlins covers approximately 7 square miles and has a population (housing) density of 1153 (521) per square mile. Thus the community's small population and its geographical isolation make treating it as a unified housing market a reasonable assumption. After deletion of 29 properties with missing attribute data, our total sample consists of 343 transactions.

Figure 1 presents a wind rose compiled for the geographical center of Rawlins (NEPA, 2006). The length of each 'spoke' around the circle is the annual frequency the wind blows from a particular direction. These spokes are further broken down into discrete frequency categories indicating the percentage of time the wind blows within a certain speed range from the indicated direction. Each concentric circle represents a different annual frequency, emanating from zero at the center to the highest annual frequency at the outer circle.

Figure 1 shows the Rawlins wind blowing primarily from the southwest; the longest spoke indicates that 25% of all hourly winds emanate from the direct southwest, and roughly 12% of the time from the west and west/southwest. The highest recorded wind velocities are also from the southwest (greater than 11.1 m/s). The upper right-hand quadrant of the rose indicates that wind rarely blows from the northeast or south/southeast, however, roughly 12 % of all hourly winds do blow from the east and east/southeast, albeit at low velocities (0.5 to 2.1 m/s).

The subject crematory is surrounded by residential developments to its north, west and southwest, with commercial development to its southeast. The landscape around the crematory and adjacent residential area has some notable attributes. In particular, a ridge (approximately 200–300 feet in elevation) embraces the residential area to the southwest, west and north of the crematory, forming a continuous, inverted 'J' around this area. The diameter of this area is approximately 0.9 miles. No residential development is located directly west, northwest and north of the J-shaped ridge, nor directly northeast of the crematory.[3]

In March 2004, the Rawlins City Planner issued a building permit to the subject mortuary to install a 40 ton, two-chamber, natural gas-fired Millennium II crematory in a vehicle storage garage adjacent to the mortuary building. Controversy remains as to whether this Planner was authorized to issue a permit for this expanded, nonconforming use of an

[3] This may seem to contradict our data, which indicates (in Table 1) a good deal of housing sales activity in the region northeast of the crematory. However, these homes are located further (about 1 mile on average) northeast of the crematory; open fields, a cemetery and school athletic fields occupy much of the nonresidential area directly northeast of the crematory.

Table 1. Variables, definitions and descriptive Statistics ($N = 343$)

Variable	Definition	Mean	SD
T	Number of months after 31 December 2003 the house was sold.	14.6122	7.0846
AGE	Age of house in years as of 2006.	51.1166	26.3047
SQFT	Square footage of house that is aboveground.	1293.8430	553.4636
SQFTBSMT	Square footage of basement if house has a basement.	666.6531	480.0317
BEDROOMS	Number of bedrooms.	3.2507	1.0123
BATHS	Number of bathrooms.	1.8674	0.6165
FINBSMT	House has a finished basement; 1 = yes; 0 = no.	0.2945	0.4565
TOWNHOUSE	1 if townhouse; 0 otherwise.	0.0583	0.2347
ATTACH2	House has attached 2-car garage; 1 = yes; 0 = no.	0.2653	0.4421
ATTACH3	House has attached 3-car garage; 1 = yes; 0 = no.	0.0058	0.0763
ATTACH4	House has attached 4-car garage; 1 = yes; 0 = no.	0.0058	0.0763
DETACH2	House has detached 2-car garage; 1 = yes; 0 = no.	0.2157	0.4119
DETACH3	House has detached 2-car garage; 1 = yes; 0 = no.	0.0146	0.120
DETACH4	House has detached 2-car garage; 1 = yes; 0 = no.	0.0087	0.0932
DISTANCE	Distance (in tenths of 1 mile) house is located away from crematory.	10.4924	7.8914
DOWNTOWN_MINUTES	Travel distance (in minutes by car) house is located away from the Rawlins downtown area.	2.417	1.763
SALEPRICE	Sale price of house in thousands of nominal dollars.	99.2185	49.8136
North	House is north of the crematory; 1 = yes; 0 = no.	0.0321	0.1764
South	House is south of the crematory; 1 = yes; 0 = no.	0.0262	0.1601
East	House is east of the crematory; 1 = yes; 0 = no.	0.035	0.184
West	House is west of the crematory; 1 = yes; 0 = no.	0.0496	0.2174
Northeast	House is northeast of the crematory; 1 = yes; 0 = no.	0.4227	0.2174
Northwest	House is northwest of the crematory; 1 = yes; 0 = no.	0.1662	0.3728
Southeast	House is southeast of the crematory; 1 = yes; 0 = no.	0.1808	0.3854
Southwest	House is southwest of the crematory; 1 = yes; 0 = no.	0.0875	0.2829

existing funeral home facility in an area zoned for residences since the 1970s. None of the residents in adjacent neighbourhoods were ever notified of plans for the crematory. Cremation operations began in August 2004. Soon after, citizens began complaining to City and State authorities about the crematory with its glaring, all-night illumination, noise and – most notable – noxious odor, which permeated residents' houses, making them feel ill and 'devaluing' (Morton, 2005) their properties. Starting in October 2004, and continuing through the time interval of our data set, the local daily newspaper updated the community on the status of the issue and printed numerous letters from citizens giving their views. In January 2005, the Wyoming Department of Environmental Quality ordered an emissions test and determined that the crematory had emissions comparable to its state permit request with several notable exceptions: annual ambient cadmium and dioxin/furan concentrations at the crematory property boundary exceeded National (and Wyoming) Air Quality Standards, by approximately 205 and 2200%, respectively (URS, 2006). Hydrogen chloride concentrations at this boundary exceeded the one-hour US Environmental Protection Agency's 'remediation goal' by 797%, with sulfur oxide, nitrogen oxide, chromium, and mercury concentrations being from 43 to 74% of the Agency's National Standard or remediation goal. Cadmium, chromium, dioxin/furans, hydrogen chloride and mercury are toxins for which any positive concentration may have human health impacts (Wexler, 2005).

No other new or substantially changed sources of (dis)amenities appeared in Rawlins residential neighbourhoods during our data time interval. Our data includes information on a variety of housing and neighbourhood characteristics typically used in the hedonic property valuation literature to explain variation in single family house selling prices. These data also contain variables describing direction from and distance to the crematory. Our data is deficient in its lack of information on lot size. This omission may detract somewhat from the explained sum of squares of our regressions; however, since our data contains detailed information on the number of attached and detached garages, following Boxall *et al.* (2005), we assume lot size to be captured at least in part by the presence (as well as extent) of transportation-related or other (e.g. maintenance- or recreation-related) vehicle storage structures beyond the livable area of the

Table 2. Parameter estimates with ln(SALEPRICE) as the dependent variable

	Specification			
	1		2	
Variable	Coefficient	*t*-Value	Coefficient	*t*-Value
CONSTANT	0.71441*	1.811	0.67640*	1.698
T	0.01874**	3.999	0.01975**	4.368
Ln(AGE)	−0.16206**	− 7.958	− 0.15841**	− 7.916
Ln(SQFT)	0.46158**	8.337	0.46870**	8.534
Ln(SQFTBSMT)	0.04026**	5.802	0.03971**	5.925
Ln(BEDROOMS)	0.35161**	4.950	0.33811**	4.912
Ln(BATHS)	0.09943	1.482	0.10991*	1.665
Ln(DOWNTOWN MINUTES)	0.11815**	2.278	0.03896	0.666
FINBSMT	0.06881*	1.929	0.06107*	1.759
TOWNHOUSE	−0.21317**	−3.526	−0.18906**	−3.068
ATTACH2	0.26031**	6.394	0.25692**	6.643
ATTACH3	0.31041*	1.658	0.33981**	2.326
ATTACH4	0.24196*	1.734	0.21615**	2.149
DETACH2	0.13376**	2.750	0.14521**	3.124
DETACH3	0.43269**	5.336	0.41962**	4.594
DETACH4	0.35251*	1.654	0.44466**	1.986
ln(DISTANCE)	0.06320**	2.060	0.08960**	2.820
$[\ln(\text{DISTANCE})]^2$	−0.01873**	−2.458	−0.01803**	−1.970
ln(DISTANCE) × *T*	−0.00375*	−1.883	−0.00897**	−3.126
ln(DISTANCE) × North × *T*			0.00818**	3.327
ln(DISTANCE) × South × *T*			0.00403	0.810
ln(DISTANCE) × East × *T*			−0.00345	−0.697
ln(DISTANCE) × West × *T*			0.01480**	3.651
ln(DISTANCE) × Northeast × *T*			0.00582**	2.941
ln(DISTANCE) × Northwest × *T*			0.00289	1.158
ln (DISTANCE) × Southwest × T			0.00771**	3.402
χ^2 (White's homoscedasticity test)	91.13	104.96		
Adjusted R^2	0.7143	0.7326		
F-statistic	48.51	38.48		
Number of observations	343	343		

Notes: * Significant at less than 10%; ** Significant at less than 5%.

house – as indicated by number of attached and/or detached garage spaces.[4]

As for other plausible but unobserved influences upon residential sale prices, we assume them to be independent of the influences we do observe, thus implying the existence of a hedonic price function (Bajari and Benkard, 2005). Distance to schools is a prominent observed influence in numerous hedonic price studies. We lack house-by-house data on it. In the Rawlins case, however, nearly all residences are within walking distance of an elementary school.[5] Variable definitions and descriptive statistics are presented in Table 1.

Variables used to measure D_{ij} include distance in tenths of a mile from the crematory (DISTANCE), and directional dummy variables indicating which of the 45° regions (from the crematory as point of origin), N, S, E, W, NE, SE, SW, NW, contains the sample house. To account for revisions in people's expectations about the Rawlins residential property market, a time trend variable, *T*, measures the number of months after 31 December, 2003 each

[4] Our data indicates a higher correlation between multiple vehicle storage structures and distance away from the downtown area, implying larger lot sizes are most prevalent among residences located at the outer edge of the Rawlins city limits, well beyond the areas plausibly affected by crematory emissions.

[5] Adding covariates to a hedonic price function to avoid omitted variable bias has a cost. If the added covariate is imperfectly measured in the sense that it does not correspond exactly to that feature which the market actually values, measurement error will increase. As more covariates are added, the measurement error bias will increase, thus increasing the noise-to-signal ratio. Atkinson and Crocker (1987) and Graves *et al.* (1988) use the Bayesian diagnostics of Leamer (1978) to demonstrate that measurement error bias appears to be a more serious problem in hedonic price studies than does omitted variable bias.

house was sold. Thus our sample includes properties sold as much as 7 months before and up to 20 months after the environmental shock to the crematory's residential neighbours from its August 2004 start-up. The average T for our sample is 14.61 months; our sample contains a few houses that have sold more than once over our 27-month sampling period. Variables used to measure H_{ik} include house age in years (AGE), square feet of living space both above-ground (SQFT) and below ground (SQFTBSMT), number of bedrooms (BEDROOMS) and bathrooms (BATHS), whether the house has a finished basement (FINBSMT), whether the house is a townhouse (TOWNHOUSE), travel time (by car) in number of minutes from house to downtown Rawlins (DOWNTOWN_MINUTES), and categorical covariates indicating whether or not the house has each of several numbers of attached or detached garage spaces (ATTACHED, DETACHED). ATTACHED_1 and DETACHED_1 are the excluded Table 2 categories, implying that the valuation impacts of the coefficients for the included categories are relative to the valuation impacts of these exclusions.

IV. Results

Table 2 reports OLS estimates of two specifications of the hedonic property value equation. Examination of the covariates in Table 2 indicates that, for both specifications, nearly all estimated coefficients have the correct signs, are statistically significant, and have very similar and plausible magnitudes across specifications when transformed to dollar values. For example, an additional square foot of living space (above ground) is worth roughly \$36 in the average house. An additional bedroom is worth slightly over \$10 300, while a finished basement contributes about \$6250 to the price of an average home.[6] These estimates are very close to the values found in other studies (see e.g. Palmquist *et al.*, 1997; Boxall *et al.*, 2005). Reported at the bottom of the Table 2 are White's (1980) chi-square test statistics of the null hypothesis of homoskedasticity, which clearly reject the null hypothesis at less than the 1% level. Asymptotic SEs used to calculate all Table 2 t-statistics are from White's (1980) heteroskedasticity-consistent covariance matrix. Finally, the reasonably high adjusted R-squared and F-values reported at the bottom of Table 2 indicate that the regressions, as specified, both have adequate fit, and explain a substantial portion of the total variation in observed home sale prices.

Turning to distance effects, specification 1 gives model parameter estimates accounting for time of sale and for distance from the crematory, but with no direction-specific terms. The predicted distance benefit as derived from specification 1 is:

$$\frac{\partial \ln(\text{SALEPRICE})}{\partial \ln(\text{DISTANCE})} = \underset{(2.060)}{0.0632} - \underset{(-1.883)}{0.00375}T - \underset{(-2.458)}{0.0375}\ln(DISTANCE) \quad (2)$$

Accounting for direction-specific heterogeneity, the predicted distance benefit derived from specification 2 is:

$$\frac{\partial \ln(\text{SALEPRICE})}{\partial \ln(\text{DISTANCE})} = \underset{(2.820)}{0.0896} + [\beta_j(\text{direction}_j) - \underset{(-3.126)}{0.00897}]T - \underset{(-1.970)}{0.036}\ln(\text{DISTANCE}) \quad (3)$$

The first term in expression (3) accounts for any nondirection-specific and time-invariant distance benefit. The bracketed terms in (3) account for direction- and nondirection-specific distance benefits, both time varying (in expression (2) all direction-specific benefit terms are assumed zero). The final right-hand-side term in (3) accounts for the distance benefit which is also distance-specific but nondirection-specific and time invariant. A series of F-tests confirms Table 2 specifications 1 and 2 as the clear best-fit benefit hedonic specifications for the Rawlins data. We summarize these tests as follows. First, we introduced and tested for the statistical significance of distance-specific distance coefficients which were time varying and/or direction-specific [we likewise tested in specification 1 for the significance of a time varying, distance-specific coefficient for expression (2)]. All these coefficients were individually and jointly nonsignificant. Second, we introduced and tested for the significance of direction-specific coefficients which were nondistance-specific and time invariant (i.e. direction-specific differences applying to the first term in expression (3)). These coefficients were likewise individually and jointly nonsignificant. Finally, though not applicable to the distance benefit expressions in (2) and (3), we also tested for any direction-specific differences associated with

[6] Interpretation of dummy variable coefficients in Table 2 requires a slight correction. For example, the correct marginal impact on SALEPRICE of the coefficient for FINBSMT is $\exp(\beta_{FB}) - 1$, where β_{FB} is the coefficient estimate for FINBSMT reported in Table 2 (Halvorsen and Palmquist, 1981).

the time coefficient, T; these tests (for specifications 1 and 2) confirmed a single coefficient estimate for T common to all Rawlins regions as most appropriate.[7]

Table 2 specifications 1 and 2 clearly demonstrate that failure to account for directional heterogeneity in Rawlins leads to omission of some important and possibly misleading benefit assessment information. The direction-specific terms in specification 2 are highly significant jointly as well as nearly all individually significant. Table 3 sheds some light on the benefit assessment implications of omission of directional heterogeneity for the Rawlins crematory example. Columns 2–4 of Table 3 provide a breakdown of mean values of DISTANCE, T, and SALEPRICE for all ($N=343$) Rawlins homes sold between 31 December 2003 and 28 March 2006, along with various subsample means of Rawlins homes sold within a specified proximity (distance and direction) to the crematory. As one works down the columns of Table 3, DISTANCE to the crematory declines from a maximum radius of 0.5 to 0.1 miles. Column 5 gives the mean benefit for successive one-tenth mile DISTANCE increases away from the crematory conditional upon direction from the crematory; and column 6 expresses this mean benefit as a percentage of mean SALEPRICE for the particular subsample of homes in question. For example, the subsample of 43 homes located north of and within a distance of 0.2 to 0.3 miles from the crematory would gain an average of $5006.59 if they were to lie within 0.3 to 0.4 miles. The first row of the topmost block in Table 3 provides the mean nondirection-specific DISTANCE benefit for the entire Rawlins sample; the next three rows show the mean distance benefit for all Rawlins homes located North, West and Southwest of the crematory. Each block below this first block presents similar calculations for sample homes within a given distance from the crematory. Benefit expression (2) is used to calculate all nondirection-specific (All Directions) estimates; expression (3) is used to calculate the direction-specific estimates appearing in the last three rows of each block. Absent an accounting of directional heterogeneity in the sample, the average Rawlins home SALEPRICE benefit associated with a 1-tenth mile DISTANCE increase away from the crematory for the period of 31 December 2003 to 28 March 2006, is −$754.08. With directional heterogeneity accounted for in the sample ($N=343$), Rawlins homes located North, West and Southwest of the crematory reveal a mean DISTANCE benefit of $534.51, $3 659.76 and $243.58. Column 6 shows these estimates amount to roughly 0.5, 3.6 and 0.3% of average SALEPRICE for homes in these directions.

As one moves down Table 3, estimates based on benefit expression (2) clearly demonstrate that a 'classical concentric circles' approach to DISTANCE in a hedonic assessment of the Rawlins data–accounting for distance to but not direction from the environmental disamenity–severely understates the assessed benefit associated with home location further away from the disamenity. At the bottom of Table 3, benefit expression (2) finally reveals a positive mean DISTANCE benefit associated for homes lying within a one-tenth mile radius of the crematory. This benefit amounts to $3657.88, or 4.89% of mean SALEPRICE as calculated from the seven sample homes sold in this area. However, expression (2) says that homes located anywhere up to 0.4 miles outside this radius suffer from not being closer to the crematory and its emissions.

Table 3 estimates based on benefit expression (3) reveal a much larger positive and increasing hedonic benefit function with distance for homes North, West, or Southwest of the crematory. Columns 4 and 5 in the table show that homes North and West of the crematory exhibit the highest benefit, ranging from 2% of mean SALEPRICE for homes within the 0.4 to 0.5 mile DISTANCE radius to over 30% of mean SALEPRICE (roughly $19 400 to $27 700) for homes within a 0.1 mile radius. The DISTANCE benefit increases slightly with time (approximately 0.0058% per month) for homes located West of the crematory, but does not appear to increase with time for homes located North or Southwest of the crematory. Homes Southwest of the crematory exhibit more modest benefit increases of 0.5 to 4.7% of mean SALEPRICE (roughly $490 to $4400) as DISTANCE declines from maxima of 0.5 to 0.1 miles. These estimates would be consistent with the Rawlins wind rose data given in Fig. 1 (e.g. prevailing winds sometimes blow from the east), if the 'J-shaped' ridge causes Southwesterly winds to swirl in North and then in West or Southwesterly directions, or if the ridge

[7] Our Table 2 coefficient estimates of 0.018–0.019 for T are not an estimate of the average monthly appreciation rate for Rawlins houses over the time span of our data. This estimate captures an 'embodied' figure, reflecting both Rawlins-specific appreciation and the discount rate; the two cannot be separated (Kiel and McClain, 1995b).

Table 3. Direction- and nondirection-specific benefit estimates for Rawlins, WY

Region	Mean DISTANCE (in tenths of 1 mile) from the crematory	MeanT (number of months after 31.December 2003 house was sold)	Mean nominal SALEPRICE	Mean benefit for one-tenth mile DISTANCE increase away from the crematory	Mean benefit as percent of mean nominal SALEPRICE
Full sample estimates ($N=343$)					
Estimates using benefit expression (2)					
All directions	10.4924	14.6122	99,218	−754.08	−0.76
Estimates using benefit expression (3)					
North	4.5462	15.4615	$104 831	$534.51	0.51
West	3.8667	16.8667	101 627	3 659.76	3,6
Southwest	4.6933	15.9333	83 420	243.58	0.29
Subsample homes located within 0.4 to 0.5 miles of crematory ($N=94$)					
Estimates using benefit expression (2)					
All directions	3.1043	16.2766	93 770	−1 217.83	−1.3
Estimates using benefit expression (3)					
North	2.3444	15.4444	$95 422	$1 914.41	2.0
West	2.1538[a]	18.4615[a]	104 262[a]	8 210.55[a]	7.87[a]
Southwest	3.7526	17.3158	91 758	489.06	0.53
Subsample homes located within 0.3 to 0.4 miles of crematory ($N=72$)					
Estimates using benefit expression (2)					
All directions	2.65	16.431	92 384	−1 218.85	−1.32
Estimates using benefit expression (3)					
North	2.125	14.75	93,600	$2 251.10	2.4
West	2.1538	18.4615	104 262	8 210.55	7.87
Southwest	2.8111	18.2222	108 111	1 124.89	1.04
Subsample homes located within 0.2 to 0.3 miles of crematory ($N=43$)					
Estimates using benefit expression (2)					
All directions	2.0302	16.6512	92 241	−1 172.07	−1.27
Estimates using benefit expression (3)					
North	1.120	13.0	74 260	$5 006.59	6.74
West	1.9364	19.1818	102 491	9 402.24	9.17
Southwest	2.220	18.40	110 200	1 862.57	1.69
Subsample homes located within 0.1 to 0.2 miles of crematory ($N=18$)					
Estimates using benefit expression (2)					
All directions	1.2333	16.6111	93 797	−528.96	−0.56
Estimates using benefit expression (3)					
North	0.675	15.250	57 200	$7 796.74	13.63
West	1.30	19.40	108 180	16 081.95	14.87
Southwest	1.40	9.0	93 000	4 388.07	4.72
Subsample homes located within 0.1 miles of crematory ($N=7$)					
Estimates using benefit expression (2)					
All directions	0.5714	15.0	74 814	3 657.88	4.89
Estimates using benefit expression (3)					
North	0.3333	13.0	54 367	$19 434.39	35.75
West	0.7	24.0	80 000	27 698.32	34.62
Southwest[b]	–	–	–	–	–

Notes: [a]No West-region homes with 0.5 miles > DISTANCE > 0.4 miles.
[b]No Southwest-region homes with DISTANCE < 0.1 miles.

inhibits air movements so as to increase odour for homes located West and Southwest of the crematory. The fact of the matter is that nearly all of the complaints about crematory emissions issue from these three directions.

V. Conclusions

The lack of studies involving direction as well as distance to a pollution source is startling, particularly in light of the widespread application of the

hedonic technique to assessing damages associated with airborne and other mobile pollutants. This article takes advantage of a unique data set to evaluate the impact of a direction-sensitive environmental shock on residential property values in a small, isolated Wyoming community. The regressions included structure, neighbourhood and location variables. Results reveal that control for directional heterogeneity increases the estimated impact of distance from the source of the shock upon residential property values; this impact appears strongest for sample houses North, West and Southwest of the source. Failure to control for directional heterogeneity results in the implausible conclusion that distance undifferentiated by direction from the point emission source has a positive impact on selling price for houses located very close (within 0.1 miles) to the disamenity source, while houses located two to five times farther away experienced reduced sale prices.

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RESEARCH REPORT

Adverse pregnancy outcomes around incinerators and crematoriums in Cumbria, north west England, 1956–93

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Study objective: To investigate the risk of stillbirth, neonatal death, and lethal congenital anomaly among babies of mothers living close to incinerators and crematoriums in Cumbria, north west England, 1956–93.

Design: Retrospective cohort study. Logistic regression was used to investigate the risk of each outcome in relation to proximity at birth to incinerators and crematoriums, adjusting for social class, year of birth, birth order, and multiple births. Continuous odds ratios for trend with proximity to sites were estimated.

Setting: All 3234 stillbirths, 2663 neonatal deaths, and 1569 lethal congenital anomalies among the 244 758 births to mothers living in Cumbria, 1956–1993.

Main results: After adjustment for social class, year of birth, birth order, and multiple births, there was an increased risk of lethal congenital anomaly, in particular spina bifida (odds ratio 1.17, 95% CI: 1.07 to 1.28) and heart defects (odds ratio 1.12, 95% CI: 1.03 to 1.22) around incinerators and an increased risk of stillbirth (odds ratio 1.04, 95% CI: 1.01 to 1.07) and anencephalus (odds ratio 1.05, 95% CI: 1.00 to 1.10) around crematoriums.

Conclusions: The authors cannot infer a causal effect from the statistical associations reported in this study. However, as there are few published studies with which to compare our results, the risk of spina bifida, heart defects, stillbirth, and anencephalus in relation to proximity to incinerators and crematoriums should be investigated further, in particular because of the increased use of incineration as a method of waste disposal.

The incineration of domestic and industrial waste releases dioxins and other chemicals into the environment.[1 2] Crematoriums have been identified as sources of atmospheric mercury.[3 4] Such pollutants, many of which act as endocrine disruptors, are hazardous to human health.[5–8] However, very little is known about the public health impact of low dose, long term environmental exposure to these chemicals.[6 9 10] Epidemiological studies have identified an increased risk of congenital anomaly and low birth weight in children born close to landfill sites, which are potential sources of this complex family of chemical pollutants.[11–17] Higher levels of environmental pollutants—including dioxins, lead, and cadmium—have been found in the blood of children living near to waste incinerators in Belgium.[7] Reduced testicular volume and delayed sexual maturity among children living in areas with high exposure were also reported,[7] linking exposure to endocrine disruptors to components of the testicular dysgenesis syndrome.[18] Despite concern over the health effects of emissions from incinerators[6] and crematoriums,[4] there is little information concerning pregnancy outcomes for mothers living in their vicinity. Identification of possible health effects of incinerators is important given the growth of incineration as a method of waste disposal[19] and its widespread use for the disposal of animal carcasses during the 2001 outbreak of foot and mouth disease in the UK.[20]

This study investigated the risk of stillbirth, neonatal death, and lethal congenital anomaly among the offspring of mothers living close to incinerators and crematoriums in Cumbria, north west England, between 1956 and 1993.

METHODS

The Cumbrian Births Database

The study area was the county currently defined as Cumbria.[21] The Cumbrian Births Database has been described in detail elsewhere.[22 23] In summary, birth registration details of all 241 524 live births and 3234 stillbirths born to mothers usually resident in the study area, from the opening of the first crematorium in 1956 to 1993, were supplied from the Office for National Statistics and entered onto a computer database.[22 23] During this period a stillbirth was defined as a fetal death occurring after 28 weeks gestation (from 1 October 1992 fetal deaths occurring after 24 weeks gestation were included, consistent with current legal definitions).[24 25] Death registrations for the cohort, including those that occurred outside Cumbria, were supplied by the Office for National Statistics from the National Health Service Central Register (NHSCR), which was the primary source of ascertainment of deaths. NHSCR routinely records deaths of all residents of England and Wales who have ever registered with a general practitioner. However, hospital records within Cumbria and in regional referral centres outside Cumbria were searched to ascertain unregistered stillbirths and infant deaths.[26] All causes of stillbirth and death were coded to ICD-9. Causes of death and stillbirth were confirmed, where possible, through examination by a consultant neonatologist of details obtained from medical and/or postmortem records (the cause of about 50% of deaths were confirmed in this way). Thus, when postmortem and/or clinical records were available, causes of stillbirth and death were validated from a number of sources and derived using a more robust method than relying on death or stillbirth certificates. Neonatal death was defined as death within the first four weeks of life.

Several outcome groups were considered: stillbirth, neonatal death, stillbirth plus neonatal death, lethal congenital anomaly (overall and by cause category). Deaths from congenital anomaly (ICD740–759) were grouped by cause, using a standard classification of infant deaths,[27] into the following hierarchical and mutually exclusive categories: all neural tube defects (ICD740–742), congenital heart defects

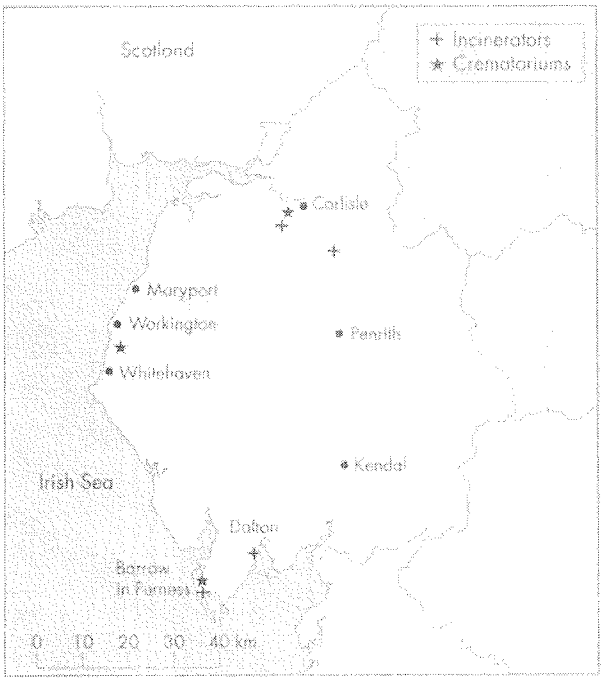

Figure 1 Location of incinerators and crematoriums in Cumbria, 1956–93.

(ICD745–747), other congenital anomalies. Neural tube defects were subdivided into: anencephalus (ICD740), spina bifida (ICD741), other central nervous system anomalies (ICD742). All other lethal congenital anomalies were grouped because of the small numbers within each ICD-9 cause category.

The mother's address on the child's birth certificate was postcoded and hence grid referenced.[28] The father's occupation, as recorded on the birth certificate, was assigned a social class.[29] Algorithms based on parents' names were used to assign birth order and identify multiple births.[23]

Geographical data (see table 1)

The grid references and dates of operation of incinerators in Cumbria were ascertained from Environment Agency records. No incinerators operated before 1977, and four operated between 1977–93. The locations of all crematoriums were ascertained from specialist digests and the dates of operation were obtained. During the period 1956–1993, three crematoriums operated. Details were captured in the geographical information system Arc/Info.[30] Mercury represents the main pollutant from crematoriums.[3,4] By contrast, emissions from incinerators incorporate a more complex mixture of dioxins, furans, particulates (such as chloride and sodium), heavy metals (including lead and chromium), and volatile organic compounds (such as chloroform).[1,31] Because of the differences in emissions, incinerators and crematoriums were analysed separately. Three of the four incinerators in Cumbria all dealt with materials defined as difficult by the Environment Agency,[32] the other (incinerator 1) processed only inert and biodegradable material.[32] Because of the historical nature of this study no detailed emissions data were available. Details of the material dealt with at each incinerator are presented in table 1. The location of all incinerators and crematoriums in Cumbria, 1956–93 is show in figure 1.

Analysis methods

A measure of exposure of each birth to incinerators and crematoriums was computed using the distance function $1/(D+0.1)^2$ where D was the distance in km from the site and the measure was summed over all sites that were in operation at the time of birth.

Stillbirth and neonatal death rates fell substantially over the study period.[33] The cause of stillbirth was recorded on the stillbirth registration only from 1961 onwards. Hence the analysis in relation to proximity to crematoriums was stratified by time period: 1956–60, 1961–71, 1972–82, 1983–93. As incinerators in Cumbria were in operation only between 1977 and 1993, this analysis was not stratified by time period.

Multivariate logistic regression[34] was used to model how the risk of each outcome varied in relation to proximity to incinerators and crematoriums, adjusting for the known demographic risk factors—year of birth, social class, birth order, and multiple births—using offsets from an analysis of the effects of demographic risk factors without the exposure function. Year of birth was modelled using both quadratic and linear terms. Social class, birth order, and multiple births were treated as categorical variables (social classes I, II, IIIn, IIIm, IV, V, armed forces, and unknown, father not recorded on the births certificate; birth order 1, 2, 3, and ≥4; multiple births, yes/no). A sensitivity analysis was carried out repeating the logistic regression, but excluding births with the greatest influence, as measured by Pregibon's influence statistic.[34] For incinerators the analysis was repeated for the period before any incinerators were open, 1956–76. Because multiple births may not be considered independent events, robust estimates of variance were used and significance assessed from the corresponding p value.[35]

RESULTS

Incinerators (see table 2)

The risk of stillbirth and neonatal death was not significantly increased closer to incinerators. However, the risk of lethal congenital anomaly was significantly higher ($p<0.01$). This significantly increased risk was restricted to heart defects and neural tube defects, specifically spina bifida. Sensitivity analysis demonstrated that these results remained significant when the most influential births were excluded. Replication of the

Table 1 Incinerators and crematoriums in Cumbria in operation during the study period, 1956–93

Site	Location	Material*	Time period of operation
Incinerator 1	Barrow in Furness	Inert, biodegradable	1977–1992
Incinerator 2	Ulverston	Hazardous, flammable, chemicals	1978–1994
Incinerator 3	Dalston	Filter material, treatment sludges, biodegradable	1979–1991
Incinerator 4	Arnculfswaite	Biodegradable, putrescible	1991–present
Crematorium 1	Carlisle	–	1956–present
Crematorium 2	Barrow in Furness	–	1963–present
Crematorium 3	Distington	–	1974–present

*Information from the Shipley Digest and Environment Agency records.

Table 2 Continuous odds ratios (OR)† for risk of stillbirth, neonatal death, and lethal congenital anomaly in relation to proximity to incinerators, 1956–76 (before incinerators opening)‡ and 1977–93, adjusted for social class, birth order, year of birth, and multiple births

Outcome	Number of cases	OR	95% CI	p
1956–76 (before incinerators opening)				
Stillbirth + neonatal death	4715	0.97	0.93 to 1.01	
Stillbirth	2622	1.00	0.96 to 1.03	
Neonatal death	2093	0.92	0.84 to 1.00	
Lethal congenital anomaly‡	1583	0.94	0.86 to 1.02	
All neural tube defects‡	602	0.95	0.85 to 1.06	
Anencephalus‡	262	0.96	0.82 to 1.13	
Spina bifida‡	244	0.86	0.67 to 1.10	
Other CNS anomaly‡	96	1.02	0.97 to 1.08	
Heart defects‡	247	1.01	0.91 to 1.12	
All other anomalies‡	300	0.94	0.81 to 1.09	
1977–93				
Stillbirth + neonatal death	1182	1.03	0.93 to 1.13	
Stillbirth	612	1.04	0.90 to 1.19	
Neonatal death	570	1.02	0.90 to 1.14	
Lethal congenital anomaly	417	1.10	1.03 to 1.19	**
All neural tube defects	132	1.13	1.04 to 1.23	**
Anencephalus	33	1.08	0.99 to 1.18	
Spina bifida	60	1.17	1.07 to 1.28	**
Other CNS anomaly	39	0.73	0.34 to 1.56	
Heart defects	104	1.12	1.03 to 1.22	**
All other anomalies	181	0.90	0.67 to 1.22	

*p<0.05, **p<0.01. †These ORs are continuous, for example, the odds of lethal congenital anomaly at a distance, D, from an incinerator compared with the odds at 3 km from incinerators is $1.10^{(3-D)}$. Hence the odds ratio comparing risk at a distance of 0.5 km compared with that at 3 km (or further) is about 1.3. ‡Before incinerators opening lethal congenital anomalies were analysed only for the time period 1961–76.

analysis, using the location of incinerators for the time period *before* they were open, showed no increased risk for any outcome (table 2).

Crematoriums (see table 3)

During 1956–93 there was a significantly increased risk ($p<0.01$) of stillbirth closer to crematoriums, reflecting a consistently increased risk from 1961 onwards. The risk of anencephalus was also significantly increased during this period ($p<0.05$), due to a significantly increased risk in 1961–71. Although most (92%) cases of anencephalus were stillborn, the significantly increased risk of stillbirth remained after exclusion of anencephalus cases from the analysis. From 1972 onwards there was an increased risk of all other congenital anomalies, excluding neural tube defects and heart defects, with increasing proximity to crematoriums, which was significant ($p<0.01$) for the period 1983–1993. These findings remained significant after exclusion of the most influential births.

DISCUSSION

Summary

We found a significantly increased risk of lethal congenital anomaly (specifically spina bifida and heart defects) in relation to proximity to incinerators, but not of stillbirth or neonatal death. In contrast with Elliott *et al*,[12] who found an increased risk for certain congenital anomalies in areas where landfill sites were later opened, we found no increased risk for any outcome in areas where incinerators were subsequently opened. Hence, there was no evidence that these increased risks might be attributable to features of the environment where incinerators were located.

Around crematoriums, there was a consistently increased risk of stillbirth from 1961 onwards. There was also a significantly increased risk of anencephalus during 1961–1971, when case ascertainment was highest because this time period largely pre-dated antenatal screening for this outcome. In the later two time periods there were very few cases of anencephalus in term pregnancies and hence statistical power to detect an effect was greatly reduced. There was a significantly increased risk of all other lethal congenital anomalies around crematoriums from 1983 onwards. This increased risk was not observed in earlier time periods despite a greater number of cases, suggesting either that a small association was obscured in earlier time periods by cases due to causes that were eliminated or reduced during 1983–93, or that the significant association in 1983–93 was a chance finding.

The significant statistical associations are different for incinerators and crematoriums. While we cannot infer a causal effect from these statistical associations, the inconsistency may be attributable to the different pollutants emitted by crematoriums and incinerators,[1 4 5 31] or it may reflect confounding with other unmeasured risk factors, or it may be a chance finding. In addition, the time periods of operation of incinerators and crematoriums were different (1977–93 and 1956–93 respectively). Hence, while we observed a significantly increased risk of anencephalus with proximity to crematoriums during 1961–1971, we did not in the later time periods for either crematoriums or incinerators. It is unlikely that any association between proximity to incinerators or crematoriums and the risk of anencephalus would be detectable in later time periods when the number of cases was low because of prenatal screening and therapeutic termination.

Strengths and weaknesses of the study

Our study covered 38 years, allowing us to investigate a potential environmental hazard with a large cohort of 244 758 births. Changes in medical practices over time may have affected the results. Medical advances, such as improved antenatal care, which allows more fetuses to be carried to at least 28 weeks, and improved gestational dating, may have increased the number of deaths classified as stillbirths. However, other advances, such as better fetal monitoring and

Table 3 Continuous odds ratios (OR)‡ for risk of stillbirth, neonatal death, and lethal congenital anomaly in relation to proximity to crematoriums, 1956–93, by time period, adjusted for social class, birth order, year of birth, and multiple births

Outcome	Number of cases	OR	95% CI	p
1956–60				
Stillbirth + neonatal death	1508	0.95	0.74 to 1.22	
Stillbirth	887	0.85	0.60 to 1.20	
Neonatal death	621	1.06	0.77 to 1.52	
1961–71				
Stillbirth + neonatal death	2559	1.10	1.01 to 1.20	*†
Stillbirth	1413	1.19	1.09 to 1.31	**
Neonatal death	1146	0.93	0.75 to 1.15	
Congenital anomaly	906	1.10	0.95 to 1.27	
All neural tube defects	493	1.12	0.94 to 1.33	
Anencephalus	219	1.23	1.01 to 1.50	*
Spina bifida	1968	1.06	0.79 to 1.42	
Other central nervous system	76	0.65	0.33 to 1.26	
Heart defects	177	1.21	0.91 to 1.62	
All other anomalies	236	0.95	0.66 to 1.38	
1972–82				
Stillbirth + neonatal death	1212	0.98	0.87 to 1.09	
Stillbirth	602	1.04	0.93 to 1.16	
Neonatal death	610	0.89	0.72 to 1.11	
Congenital anomaly	452	0.80	0.59 to 1.09	
All neural tube defects	200	0.68	0.39 to 1.16	
Anencephalus	69	0.35	0.11 to 1.18	
Spina bifida	88	0.71	0.28 to 1.78	
Other central nervous system	43	0.97	0.64 to 1.47	
Heart defects	125	0.58	0.26 to 1.27	
All other anomalies	137	1.04	0.84 to 1.29	
1983–93				
Stillbirth + neonatal death	618	0.99	0.87 to 1.23	
Stillbirth	332	1.01	0.97 to 1.05	
Neonatal death	286	0.84	0.60 to 1.17	
Congenital anomaly	201	1.02	0.99 to 1.05	
All neural tube defects	41	0.76	0.37 to 1.56	
Anencephalus	7	0.65	0.13 to 3.19	
Spina bifida	18	1.02	0.97 to 1.08	
Other central nervous system	16	0.08	0.00 to 5.62	
Heart defects	49	0.50	0.15 to 1.62	
All other anomalies	111	1.03	1.01 to 1.06	**
1956–93				
Stillbirth + neonatal death	5897	1.02	0.99 to 1.05	
Stillbirth	3234	1.04	1.01 to 1.07	**
Neonatal death	2663	0.91	0.78 to 1.04	
Congenital anomaly (1961–93)	1569	1.02	0.96 to 1.08	
All neural tube defects	734	1.00	0.87 to 1.16	
Anencephalus	295	1.05	1.00 to 1.10	*
Spina bifida	304	0.99	0.77 to 1.27	
Other central nervous system	135	0.70	0.43 to 1.14	
Heart defects	351	1.00	0.77 to 1.31	
All other anomalies	484	1.04	1.00 to 1.07	*

*p≤0.05, **p≤0.01. †Became non-significant when most influential births were excluded. ‡These ORs are continuous, for example the odds of anencephalus at a distance, D, from crematoriums compared with the odds at 3 km from crematoriums in 1961–71 is $1.23^{(3-D)}$. Hence the odds ratio comparing risk at a distance of 0.5 km compared with that at 3 km in 1961–71 is about 1.77.

improved resuscitation, may have decreased the number of stillbirths either by shifting potential stillbirths into the category of neonatal deaths or by preventing infant death. The introduction of antenatal screening and elective termination reduced the number of stillbirths and deaths attributable to congenital anomalies in recent years.[16] Thus the clinical characteristics of the cases in the 1950s and 1960s may be intrinsically different from those in the 1990s. However, all analyses were adjusted for year of birth, such that the risk of stillbirth, lethal congenital anomaly or neonatal death to mothers living close to incinerators or crematoriums was, in effect, compared with that of other mothers giving birth around the same time. Hence, the objectives of our study were not affected by changes in the nature of cases over time.

Because the Cumbrian Births Database recorded all birth registrations in Cumbria during the study period by date of birth and postcode of mother's residence, we had precise data on the population at risk and the location of each birth. Consequently we were able to estimate exposure and risk within a continuous model unconstrained by the availability of population statistics from other sources and we did not have to restrict our analysis to traditional geographical areal units. However, a limitation of our study was the unavailability of data on pregnancies less than 28 weeks gestation (24 weeks since 1 October 1992), which will affect the population at risk because some serious congenital anomalies might not continue to this stage of maturity, either through spontaneous abortion or termination. The inability to include such cases in

Key points

- Incinerators and crematoriums are sources of harmful chemicals (including dioxins), although little is known about the effects of long term low dose exposure
- We investigated the effects of proximity to incinerators and crematoriums on stillbirth, neonatal death, and lethal congenital anomaly
- We used precise details of the population at risk and the distance of each birth from all sites
- We found an increased risk of spina bifida and heart defects in relation to proximity to incinerators and an increased risk of stillbirth, anencephalus, and other congenital anomalies in relation to proximity to crematoriums

Recommendations

- Further work is needed to establish whether this statistical association is causal or not

our study is likely to have resulted in a conservative estimate of the effects of proximity to sources of pollution. A further limitation of our study was the exclusion of non-lethal congenital anomalies, although we were rigorous in our ascertainment of deaths, stillbirths, and lethal congenital anomalies.[26] Because no data were available for non-lethal congenital anomalies these cases could not be excluded from the live birth control group. However, as the control group comprised all live births that survived over 28 days non-lethal congenital anomalies would have comprised a very low proportion of the comparison group.

We were able to incorporate exposures of each birth to putative pollution from several sites. In addition, we had demographic information for each birth and hence were able to take account of individual risk factors, such as social class, which we have shown previously to be a better predictor of stillbirth rates than community based deprivation measures such as the Townsend score.[23] Grid references for incinerators and crematoriums were supplied to an accuracy of 100 metres.

We assumed that the mother's residence during pregnancy was the same as that recorded on the birth registration. Hence migration of mothers during pregnancy may have resulted in misclassification of exposure, which would have tended to obscure any association between risk of adverse pregnancy outcome and proximity to crematoriums or incinerators.

A further limitation was that, as actual pollution levels around each site were unknown and would be impossible to ascertain retrospectively over such a long time period, we relied on a function of distance as a surrogate for potential exposure. The form of the exposure function, $1/(D+0.1)^2$, assumed that exposure increased rapidly with proximity to the sites. A potential mechanism for absorption of toxic pollutants from incinerators or crematoriums by pregnant women might involve direct inhalation of pollutants or contact through food, soil, or water contamination. We assume higher pollution levels closer to the point source and thus the distance function is a reasonable surrogate indicator that has been used in many similar studies investigating health risks around pollution sources.[11–13 15] Although we could not consider any changes in pollution levels over time all analyses were adjusted for year of birth, so the risk of adverse pregnancy outcomes for mothers living close to crematoriums/incinerators was compared with that of mothers giving birth in the same year.

The facilities in Barrow in Furness and Dalton in Furness are located near to industrial sites defined as hazardous by the Environment Agency. Hence, there is some potential for confounding between proximity to incinerators/crematoriums and proximity to hazardous industrial sites.

There is potential for confounding between distance from incinerators and crematoriums and unmeasured risk factors, such as diet, lifestyle, or occupational exposures. However, we adjusted for individual social class, which is likely to be related to such lifestyle factors. Hence, this study can only identify a potential statistical association between exposure to incinerators or crematoriums (modelled by a function of distance) and adverse pregnancy outcomes. We cannot establish the biological plausibility of these findings given the lack of detailed emissions data. Further studies are now required using actual pollution levels around crematoriums and incinerators to investigate the biological plausibility of our findings.

We undertook a large number of comparisons and hence it is possible that some of the significant results may be chance findings, arising through multiple significance testing. However, our results of raised risk of stillbirth, congenital heart defects, and neural tube defects were generally consistent between time periods and sensitivity analysis showed they were robust, which lessens the probability of them being chance findings. As with all geographically based studies, there was potential for confounding with lifestyle and sociodemographic risk factors that were not included in the analysis, although we were able to adjust for individual level socioeconomic status, which has not been possible in many other studies.

Comparison with other studies

Although several studies have considered pregnancy outcomes for mothers living close to hazardous waste and municipal landfill sites,[11–15 17 37 38] there is a paucity of epidemiological data concerning pregnancy outcomes around incinerators and crematoriums with which to compare our study. Our finding of an increased risk of lethal congenital anomalies, in particular neural tube defects and congenital heart defects, in babies born close to incinerators is consistent with the results of some studies of congenital anomalies around hazardous waste and municipal landfill sites[11–15] but not with others.[37 38] Nevertheless, our findings need to be interpreted cautiously, as both the pollutants and exposure pathways associated with these sources differ. While incinerators are sources of a range of chemicals, including some also emitted by hazardous waste and municipal landfill sites, they also emit dioxins, heavy metals, and particulates.[31] Furthermore, the exposure pathways from incinerators and landfill sites are different[12 13 31]: exposure of humans to landfill pollution results from water supply contamination, groundwater run off, and atmospheric contamination from landfill gases,[12 13] whereas pollutants from incinerators are primarily dispersed atmospherically.

Although incinerators and crematoriums in Cumbria were located in urban areas, there were so few in operation that only 10% of the Cumbrian birth cohort were born within 2 km of an incinerator or crematorium, in contrast with the finding by Elliott *et al*[12] that 80% of the population in England and Wales live within 2 km of a landfill site.

Conclusions

We found an increased risk of lethal congenital anomaly (specifically spina bifida and heart defects) in relation to proximity to incinerators and an increased risk of stillbirth and anencephalus in relation to proximity to crematoriums. In view of the scarcity of published data and our use of a distance function to represent potential exposure it is difficult to assess whether these statistical associations reflect a causal effect. Further investigations using actual pollution levels and high quality data, including lethal and non-lethal outcomes in term pregnancies and elective terminations, are required. Sufficient investment must be made in national registration systems to ensure these issues can be investigated adequately. The UK system for registration of congenital anomalies is known to be incomplete and this severely restricts its credibility.[39]

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Conflicts of interest: none.

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PostScript

LETTERS

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Adverse pregnancy outcomes around incinerators and crematoriums

Dummer and colleagues, research provides an important step towards establishing the evidence base around the adverse pregnancy outcomes associated with incinerators and crematoriums.[1] Investigating the possible adverse health effects from environmental hazards is a public health challenge that demands the use of systematic and reproducible research methods. We have some concerns regarding the study described by Dummer and colleagues.

The study focuses on selected "fatal" pregnancy outcomes. One key concern is that this excludes miscarriages, abortions, and non-lethal congenital anomalies. Excluding these outcomes may misrepresent and underestimate any possible association between the exposures under consideration and "pregnancy outcomes". Indeed, it is possible that the "fatal" pregnancy outcomes considered by the researchers may actually be inappropriate in relation to the chemicals released from incinerators and crematoriums. Studies on exposure to lead, for example, indicate that the most likely pregnancy related outcomes associated with high exposure are low birth weight, spontaneous abortion, and length of gestation.[2]

The measurement of exposure in this study is also unclear. The reason for this is that distance from an incinerator or crematorium is used as a surrogate measure of exposure and the pathway for the absorption of "toxic pollutants" is considered to be direct inhalation of pollutants or contact through food, soil or water contamination.

Actual pollution levels at the sites are not provided and there is no indication of whether these sites are located in isolation or located in close proximity to other industrial processes. Under these circumstances is difficult to see how the surrogate measure of distance could be used as a proxy for exposure to emissions from incinerators and crematoriums.

We question the robustness of any study conducted over such a long period. The margins of error, based around outcomes that will vary considerably with the diagnostic and coding changes over this period, are likely to be great despite the adjustments for year of birth. In addition, as the authors note the stillbirth and neonatal death rates fell substantially over this period, for reasons other than the environmental exposure in question. However, the authors fail to consider whether or not this large fall in numbers may have influenced the ability to detect an effect in the later study periods when all the incinerators and crematoriums were open. Additionally there was no mention of whether any study size calculations were made.

The authors stratify the analysis by time period to correspond to the opening of incinerators and crematoriums over this study period and then draw conclusions based around comparing the results between each study period. As most of the confidence intervals for the "significant" odds ratios in the periods after the installation of incinerators and crematoriums overlap with the earlier periods, care should be taken in drawing conclusions regarding these "significant" odds ratios. We also question the robustness of conclusions drawn from studies with multiple significance testing, which increase the chances of a significant result arising by chance. We also question whether other key confounders should have been accounted for in the analyses.

A further point of relevance to this study is that tighter legislation on incinerator emissions means that emissions have reduced dramatically since the time of the study and further still in the past decade. For example between 1992 and 1998, most incinerator emissions reduced by a factor of over 95%.[3] However, the health effects of incinerators and crematoriums are still issues of major concern to the public.

Unfortunately this report was misrepresented in the press as evidence for a link.[4] The authors, of course, cannot be held responsible for press misquoting the findings, but it is important that research conclusions are very clearly stated and links to other types of waste disposal not used indiscriminately.

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Authors' reply

Duffell *et al* have challenged the design and interpretation of the findings of our study of stillbirths and lethal congenital anomalies in the vicinity of crematoriums and incinerators in Cumbria.

While we share some of their concerns about the limitations of the study, they clearly have little appreciation of the complexity and difficulty of performing epidemiological studies of this nature.

We agree that it would be highly desirable to have included miscarriage, abortion, and non-lethal congenital anomalies in our study, but construction of such a dataset for the time period of the study would be impossible: there are no sources for this information—as we stated in our paper. We also agree that exclusion of these adverse pregnancy outcomes could have led us to underestimate any association with proximity of the site of interest—the fact that we did have significant findings is therefore surprising.

Duffell *et al* note that our choice of adverse outcomes may be "inappropriate" for the chemicals associated with incinerators and crematoriums, noting that studies on lead exposure indicate that possible adverse pregnancy outcomes include spontaneous abortion and low birth weight. However, as we state in the study, while the main pollutant from crematoriums is mercury, emissions from incinerators incorporate a complex mixture of dioxins, furans, particulates, heavy metals (including lead and chromium), and volatile organic compounds. However, there is little information concerning the effects on pregnancy outcomes of living near to sources of this varied range of pollutants.

Our surrogate measure of exposure was proximity to incinerators and crematoriums; we were unable to use actual pollutant or exposure measurements because these do not exist for the time scale of our study. While we acknowledge the limitations of such a surrogate measure, many previous studies have also relied on distance from pollution sites as an indicator of exposure because of a lack of detailed historical environmental monitoring data. Duffell *et al* indicate that we do not consider if any of the sites included in our study are located close to other sources of industrial pollution. However, we clearly note that the facilities in Barrow in Furness and Dalton in Furness are located close to industrial sites defined as hazardous by the Environment Agency. Hence we acknowledge that there is potential for confounding between our proximity measure and proximity to hazardous industrial facilities.

The fact that the confidence intervals on pre-operation and post-operation odds ratios overlap is not relevant to the interpretation of the odds ratios post-operation being significantly different from one.

We are well aware of the problems of multiple significance testing—which is why we were cautious in the interpretation of our findings and why we looked for consistency between time periods. The effects of our carefully modelling (using both linear and quadratic terms), within broad time strata, of the effect of year of birth is that pregancies

with adverse outcomes were compared with all other pregancies in the same year, thus minimising the possible bias due to diagnostic and coding changes over time.

Duffell *et al* also question whether key confounders may have been an explanation for our significant findings. While there is always the possibility of residual confounding, we did control carefully for both socioeconomic factors and the changing underlying risk of the outcomes we considered. It is not immediately apparent what potential confounders could be associated with both adverse pregnancy outcome and proximity to the incinerator and crematorium sites that are not related to either social class or time period. Nevertheless the fact remains that our results, as we acknowledged in the paper, may be chance findings.

The investigation of the health consequences of potentially hazardous installations in the environment is difficult, especially historically; when emissions tended in general to be higher but environmental monitoring data are at best scant and often nonexistent.

Information on health outcome is also difficult to assemble and it is of particular concern that access to such information at a population level is likely to get more rather than less restrictive. This is an important issue and one with which we should all be concerned: it is neither in the public interest, nor the patient interest.

The "full" study of all relevant health outcomes related to *actual* exposures are very seldom achievable in any context and studies such as ours have an important part to play, though we fully support the concerns of Duffell *et al* that they are often wrongly reported by the media. It is our concern that authors may not fully understand their responsibility in ensuring that any press release from the journal is a fair and accurate rather than sensationised account of their findings and the extent to which they may be involved directly with the media in the period immediately after publication. In this case, as Duffell *et al* observed, our findings were misrepresented in the media to a certain extent despite our greatest efforts at every stage for this not to happen, though often sensationalised headlines were followed by more balanced text.

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BOOK REVIEW

Community health advocacy

S Loue, L S Lloyd, D O'Shea. (Pp 171; price not stated). Kluwer Academic/Plenum, 2003. ISBN 0-306-47390-9.

The concept of health advocacy is nowadays considered one of the main tasks of public health. Health professionals try to solve health problems with effective strategies. One of them is, specifically, community health advocacy. To act on health problems in the global context, participation and communitarian perspective is a choice. People need advocacy and empowerment to achieve health outcomes more than isolated health interventions assistance. Although the role of health services is important to promote and secure health, there are other issues that are important to guarantee people's health. Community, organisations, and institutions must advocate for health with the necessary tools and in the context of each society.

"Community health advocacy" is a reflection on the role of different issues in relation with an empowering view of public health. To conceptualise the subject, the authors define terms as community, needs, action, grassroots, or empowerment. They try to explain the actions of all social agents that are (or can be) involved in health advocacy. They also emphasise how to integrate values and ethics in health advocacy objectives.

The different chapters provide an approach of what strategies and efforts of advocacy are required. At the end of each chapter, a specific experience of health advocacy is included related to the issue discussed. Finally, there is a section with discussion questions to investigate the subject more deeply.

In chapters 1 and 2, the authors define the relation between community and public health. They also explain community characteristics and interests from the point of view of sociology, anthropology, or psychology. They question what community needs assessments are, and what are the causes that make a particular group of persons unable to achieve a better quality of life. Assessment is a process to provide people with the possibility to identify their own needs. This is a preliminary step for health advocacy.

In chapter 3 the authors investigate ways of organising the community. They define the grassroots approach as a "bottom up" strategy, and show how people can drive health programmes themselves to achieve health goals. This focus is a tool for social change, empowering people in their own decisions about health.

Chapter 4 explains the objective of building coalitions. To improve people's health by advocacy grassroots organisations can organise themselves into coalitions created at different levels. To secure society's health rights it is necessary to build networks to approach the different political situations. Advocacy efforts require relationships to be built to provide people with opportunities for collective problem solving.

In chapters 5 and 6, the authors review the role of legislative advocacy as a mechanism of health advocacy. This is a formal process that makes the rules that must be accepted by the people. If the legal system is on the side of community needs, it will be one of the most important strategies to improve their health. Also, the administrative agencies have an important role in the rule making process. Organisations and individuals can act as advocates in influencing the process of legislative building, or trying to change rules that do not contribute to the health improvement of the community.

Chapter 7 explains the importance of the state court system and its capacity to be a framework for civil mobilisation.

In chapter 8, the authors explain the influence of the media as an advocate for health. An effective use of media can have an impact in society drawing attention to different health problems, the community action to solve it, or the role of politics. The information about health through media is influenced by different powers. For this reason, people must use the media as a support to advance in the solutions for health problems.

Chapters 9 and 10 are related with the evaluation of the advocacy action, and the conflicts that can result in the relation between advocacy and ethics. The authors emphasise the importance of evaluating the advocacy actions, to improve other activities in the future. People learn through their successes and failures. This is a formative perspective of monitoring and evaluation. Finally, health advocacy must act to secure health rights supporting people in *building their own history*. The main ethics consideration related to health advocacy is to respect people's problems and solutions and their point of view.

In my opinion, this book is an excellent tool to be used in different areas. For example, it could be useful as a support in the planning of community projects through social and political institutions. Different bodies have in this book an opportunity to learn of other organisational experiences. Finally, this book is a great guide for teaching public health, and also in the process of research in health promotion and health advocacy.

D Gil

ROSTAD MORTUARY CREMATORY AIR QUALITY IMPACT ANALYSIS

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1 Introduction and Summary of Results

This report provides an analysis of the air quality impacts of air pollutants from the incineration of human bodies in a CMS Millennium II cremator located at the Rostard Mortuary in Rawlins, Wyoming. A crematory is a source of criteria pollutants and toxic air contaminants. These pollutants are listed in Table 2. These pollutants are emitted into the atmosphere through an exhaust stack during the cremation process, which may have an impact on the nearby properties in the neighborhood. Since the Rostad Mortuary Crematory is surrounded by a residential area and is located within 600 meters of the nearest schools (Mountain view school and St. Joseph school), it is important to examine the potential impact of the emissions from the crematory. In order to predict the impact of the emissions, an air quality impact analysis was performed using the current version of the EPA-approved Industrial Source Complex Short Term, Version 3 air dispersion model (ISCST3).

The emissions from the crematory stack were categorized into 20 species and modeled as a point source. The major emissions from crematories are nitrogen oxides (NOx), carbon monoxide (CO), sulfur dioxide (SOx), particulate matter (PM10), mercury, hydrogen fluoride (HF), hydrogen chloride (HCl), non-methane volatile organic compounds (NMVOCs), other heavy metals, and some persistent organic pollutants (POPs) including dioxins and furans, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), etc. The detailed emission rates and stack information are described in Section 3.

Default options of ISCST3 were used along with building downwash information and one year (1994) of hourly meteorological data. Fine nested grid receptors were used to predict the impact more accurately including 25-meter spaced fine grid receptors for the area within the vicinity of the crematory.

The modeling results are shown in Table 3 and indicate that the criteria pollutants (NOx, SO, SOx, and PM10) and most of the toxic pollutants except cadmium and dioxin/furan do not exceed the National Ambient Air Quality Standards (NAAQS) and Wyoming Ambient Air Quality Standards (WAAQS) or the preliminary remediation goal (PRG) levels (URS, 2006). However, the annual cadmium and dioxin/furan concentrations exceed the PRG significantly, by 205% and 2200%, respecuvely.

The modeling results show that the maximum impact is predicted to occur on the immediate vicinity of the crematory as shown in Figures 5 through 9.

2 Modeling Analysis Design

The air quality dispersion model and related options used to determine ambient impacts are described in this section, along with the receptor network employed.

2.1 Model Selection

The air quality impacts in the areas surrounding the Rostad Mortuary Crematory were determined with the latest version of the EPA ISCST3 model, which is commonly used for

regulatory analyses involving the prediction of impacts (concentration or deposition) within 50 kilometers (km) of a source. The ability of the ISCST3 model to accommodate varying source types and terrain makes it an appropriate selection for this analysis.

2.2 Model Input Defaults/Options

The ISCST3 model was run with regulatory default options. These options are:

- Stack tip downwash (except for Schulman Scire downwash);
- Buoyancy induced dispersion (except for Schulman Scire downwash);
- No gradual plume rise (except for building downwash);
- Calms processing routines;
- Upper bound concentration estimates for sources influenced by building downwash from super squat buildings;
- Default wind profile exponents; and
- Default vertical potential temperature gradients.

As noted above, the model was executed with the calms processing invoked. Periods of calm winds are identified in the meteorological data processing by defining a threshold wind speed. If the recorded wind speed is less than the threshold wind speed, then the wind speed for that given hour is reset (and wind direction adjusted) to reflect a period of calm wind. The option for modeling elevated terrain was also selected.

2.3 Rural/Urban Classification

The ISCST3 model includes rural and urban algorithm options. These options affect the wind speed profile, dispersion rates, and mixing-height formula used in calculating ground-level pollutant concentrations. A protocol was developed by the EPA to classify an area as either rural or urban for dispersion modeling purposes. The classification is based on average heat flux, land use, or population density within a 3-km radius of the modeled facility, with land use being the most definitive criterion (USEPA, 2003). The urban/rural classification scheme based on land use is as follows:

> *The land use within the total area, A0, circumscribed by a 3-km circle about the source, is classified using the meteorological land use-typing scheme proposed by Auer (1978). The classification scheme requires that more than 50% of the area, A0, be from the following land use types in order to be considered urban for dispersion modeling purposes: heavy industrial; light-moderate industrial; commercial; single-family compact residential; and multi-family compact residential. Otherwise, the use of rural dispersion coefficients is appropriate.*

Since most of area is the medium low density of residential and rural area as shown in the aerial photomap, the rural land use classification was used in this analysis.

2.4 Receptor Network

A receptor grid, or network, defines the locations of predicted air pollutant concentrations that are used to assess compliance with the relevant standards or guidelines. The following receptor network was used for this analysis:

- 25-m spaced receptors along the facility fence line and out to 100 m from the fenceline;
- 50-m spaced receptors from beyond 100 m to 250 m from the fence line;
- 100-m spaced receptors from beyond 250 m to 1000 m from the fence line; and
- 250-m spaced receptors from beyond 1000 m to 2000 km from the fence line.

This network is composed of Cartesian (X, Y) receptors with Universal Transverse Mercator (UTM) coordinates. For consistency with the terrain elevation data, the modeling was conducted using the North American Datum of 1927 (NAD27). Receptors were modeled with terrain elevations for each point interpolated from US Geological Survey (USGS) 7.5-minute Digital Elevation Model (DEM) data.

The near field and far field receptor networks are illustrated in Figure 1. Detailed locations of buildings, the stack, and the fenceline are shown in Figure 2 and Figure 3.

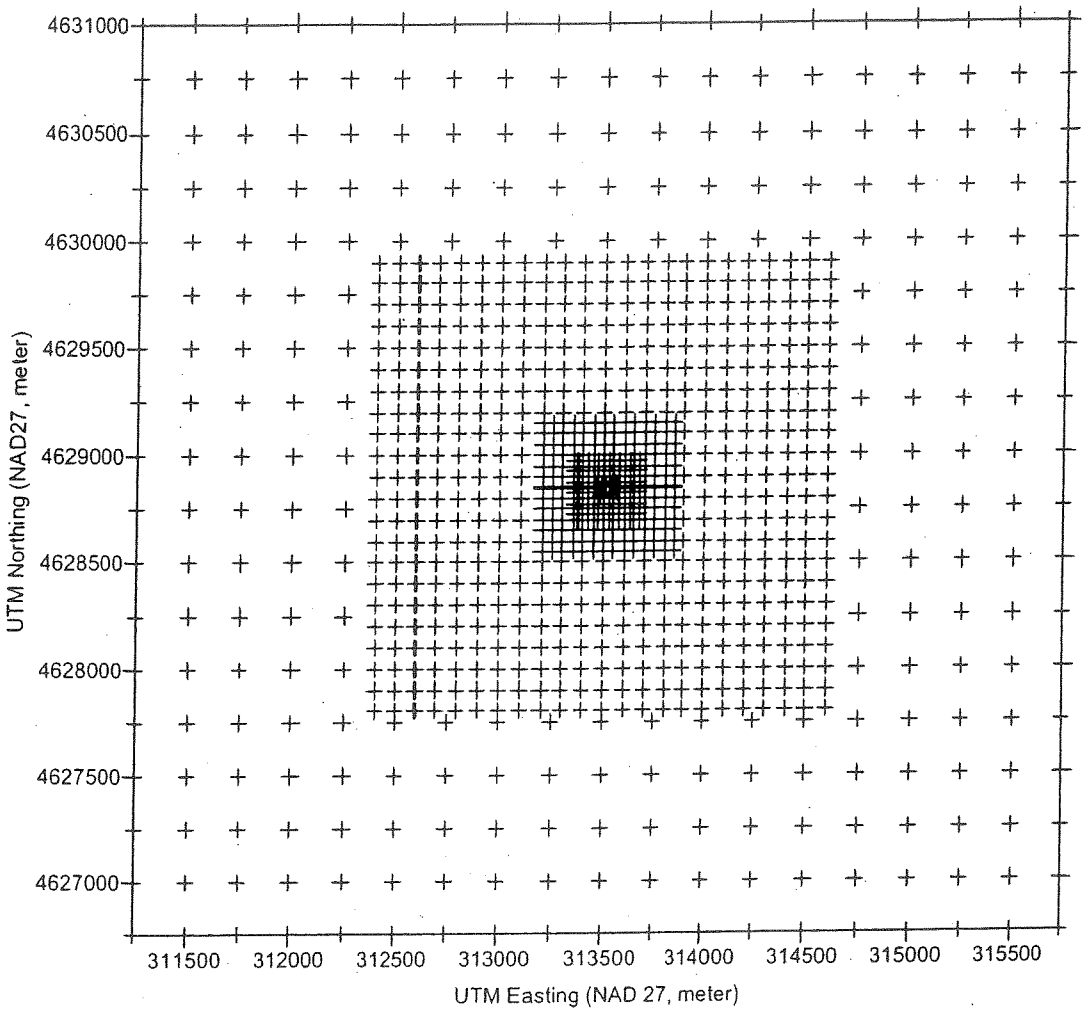

Figure 1. Near Field and Far Field Receptor Networks

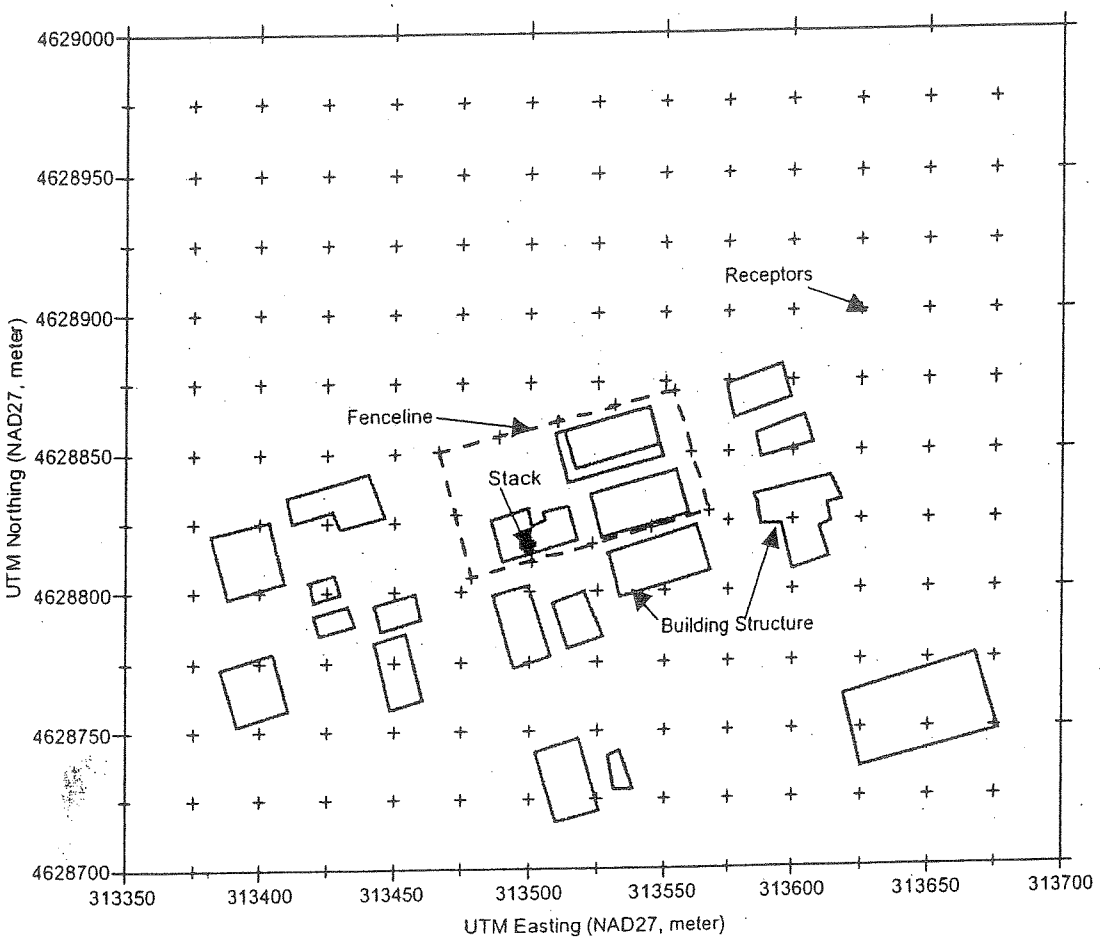

Figure 2. Location of Buildings, Stack, Property Fenceline, and Receptors

Figure 3. Location of Buildings, Stack, Property Fenceline, and Receptors on the Aerial Photomap

3 Source Data

3.1 Pollutants Emissions

The major emissions from crematories are nitrogen oxides (NOx), carbon monoxide (CO), sulfur dioxide (SOx), particulate matter (PM10), mercury, hydrogen fluoride (HF), hydrogen chloride (HCl), non-methane volatile organic compounds (NMVOCs), other heavy metals, and some persistent organic pollutants (POPs) including dioxins and furans, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), etc. The emission rates depend on the design of the crematory, combustion temperature, gas retention time, duct design, duct temperature, and any control devices (Environment Canada, 1999).

Particulates such as dust, soot, ash and other unburned particles originate from the cremation container, human remains, and other contents of the container. Carbon based organic particulates should be removed in the secondary combustion chamber and through proper adjustment and operation of the cremation equipment. Carbon monoxide results from the incomplete combustion of containers, human remains, fuel, and other contents.

Carbon monoxide may be minimized through proper adjustment and operation of the cremation equipment (Environment Canada, 1999).

Sulfur dioxide is produced from the combustion of fuel, containers, and contents. The sulfur content of natural gas and human remains is relatively lower than other fuels. Nitrogen oxides are formed by high temperature combustion processes through the reaction of the nitrogen in air with oxygen (Environment Canada, 1999).

Mercury emissions originate from the dental fillings that may contain 5 to 10 grams of mercury depending on the numbers and types used. Mercury may be removed through the use of selenium salt in the cremation chamber or scrubbers. Hydrogen fluoride and hydrogen chloride results from the combustion of plastics contained in the container and from stomach contents. NMVOCs are produced from incomplete or inefficient combustion of hydrocarbons contained in the fuel, human remains, and casket. Dioxins and furans result from the combustion of wood cellulose and chlorinated plastics in the correct temperature range (Environment Canada, 1999).

3.2 Emission Source Description

Crematory emissions exhaust from a stack, and thus it is modeled as a point source. The stack parameters are shown in Table 1.

Table 1. Source Stack Parameters (Rostad Mortuary, 2006)

Source Name	Stack Height		Stack Gas Exit Temperature		Stack Gas Exit Velocity		Stack Diameter		UTM Coordinates (NAD27) Easting/Northing	
	(ft)	(m)	(°F)	(K)	(ft/s)	(m/s)	(ft)	(m)	(m)	(m)
Stack1	15	4.572	1413	1040	16.08	4.902	2.03	0.619	313499	4628817

Emission rate for various pollutants from crematory operations are described on the basis of 100 lbs of body per hour. The emissions associated with the fuel combustion during the cremation were also included, and the fuel type was assumed to be natural gas. All emission rates are shown in Table 2, and the maximum emission rate was chosen conservatively as a model input emission rate among various references.

Table 2. Source Emission Rates

Reference	Established Emission Factors Note1	Stack Test Data Note 2	Keaton's Mortuary Note 3	US EPA Note 4	US EPA 1996 Note 5	CANA, 1993 Note 6	Canada 1996	US EPA/CANA Woodlawn Crematory Test 1999 Note 7	Max Emission Rate	
Pollutant	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Emission Rate (lb/hr)	Max (lb/hr)	Max (g/sec)
PM10	2.6619E-05	0.276	0.171233		3.902E-05	3.445E-01		0.283	3.445E-01	**4.340E-02**
NOX	0.323809524		0.32911		4.746E-01	9.791E-02		0.43	4.746E-01	**5.980E-02**
SOX	0.057142857		0.05		8.374E-02	7.003E-01		0.148	7.003E-01	**8.824E-02**
CO	4.74286E-10	0.064	0.150616		2.163E-01	3.263E-01		0.0099	3.263E-01	**4.111E-02**
VOC	0.147619048								1.476E-01	**1.860E-02**
ARSENIC					1.689E-08				1.689E-08	**2.128E-09**
CADMIUM					4.780E-09			0.000161	1.609E-04	**2.028E-05**
LEAD					2.862E-08			0.00097	9.700E-04	**1.222E-04**
CHROMIUM					1.298E-08				1.298E-08	**1.635E-09**
MERCURY					1.438E-06			0.000573	5.732E-04	**7.222E-05**
NICKEL					1.654E-08				1.654E-08	**2.084E-09**
COPPER					1.186E-08				1.186E-08	**1.495E-09**
COBALT					2.512E-09				2.512E-09	**3.165E-10**
DIOXIN				8.14335E-13	3.574E-11			8.12E-08	8.12E-08	**1.023E-08**
FURAN				8.14335E-13	4.669E-11			8.12E-08	8.12E-08	**1.023E-08**
HCL						2.446E-02	7.077E-02	0.39	0.39	**4.914E-02**
HF					2.882E-07				2.882E-07	**3.631E-08**
Fluoranthene					9.072E-11				9.072E-11	**1.143E-11**
Benzo[a]-pyrene					1.591E-11				1.591E-11	**2.004E-12**
Benzo[a]-anthracene					5.812E-12				5.812E-12	**7.323E-13**

Note:

1) According to NCDENR-DAQ, these emission factors were developed from testing performed by the California Air Resources Board. The test results are published in the October 29, 1992 report entitled "Evaluation Test on Two Propane Fired Crematories at Camellia Memorial Lawn Cemetery". A full copy of the report is generally unavailable. However, the report is based on testing of 2 propane-fired crematories (max. capacity 1.4 MMBTU/hr each). The minimum process rate (hr/case) for the similar crematory (see Note 2 below) is used to convert from lb/case to lb/hr.
2) The emission rates were obtained from "Air Compliance Test Report" for a crematory incinerator at Haisley-Hobbs Funeral Home in Fort Pierce, FL (attached). The average emission rates have been converted to lb/hr for presentation and comparison.
3) Engineering Evaluation Report Keaton's Mortuary Plant Number 14968, Application Number 6416. Emission Factors for PM10, SO2, NOx, CO, and POC are taken from AP-42, 1/95, Table 1.4-1, and 1.4-3. Based on the Emission factors from AP-42, Pathological Waste Combustion was added. Operating schedule: 16hrs/day, 7days/wk, 52wks/yr
4) Source of Dioxins and Furans in Australia Air Emissions May 2002; Page 38 Chapter 6.1.9. Crematoria
5) Emission Factors are for 1 55 to 70 kg body, about 65kg on average, No emission control devices were present in the creation of the emission factors. US-EPA emission factors include a 2 kg cardboard and 1kg wood container.
6) CANA emission factors averaged from test data in report for cardboard, cloth covered and finished wood containers.
7) Maximum outlet emission rate was used among the average values of 3 sets of 3 various runs.

4 Building Downwash Analysis

Stack exhaust has the potential to be influenced by building wakes, which in effect "wash down" the plume, causing increased ground-level concentrations. The EPA Building Profile Input Program (BPIP), adapted for use with ISCST3, was used to determine the direction-dependent building input parameters (USEPA, version 04274, 2004). This program was also used to calculate the Good Engineering Practice (GEP) stack height for each source location. The crematory stack height was not greater than the calculated GEP stack height; therefore, building downwash was considered.

Figure 2 shows the structures (purple solid line) that could influence the stack gas exhaust and are included in the downwash analysis. These structures were input to the BPIP program. Inspection of the BPIP output showed that some of these structures were of sufficient stature to cause some downwash of the stack exhaust.

5 Meteorological Data

ISCST3 model-ready meteorological data were obtained from the State of Wyoming Department of Environmental Quality (WDEQ). The meteorological data were collected near Rawlins, Wyoming and summarized into hourly averaged values for 1994. The anemometer height used to collect the wind data was 10 meters. According to the State Department of Environmental Quality, the average background NO_2 concentration for this area is 10 micrograms per cubic meter (g/m^3) (State of Wyoming, 2006).

A wind rose for the 1994 data set is shown in Figure 4. The prevailing wind directions are from the Southwest.

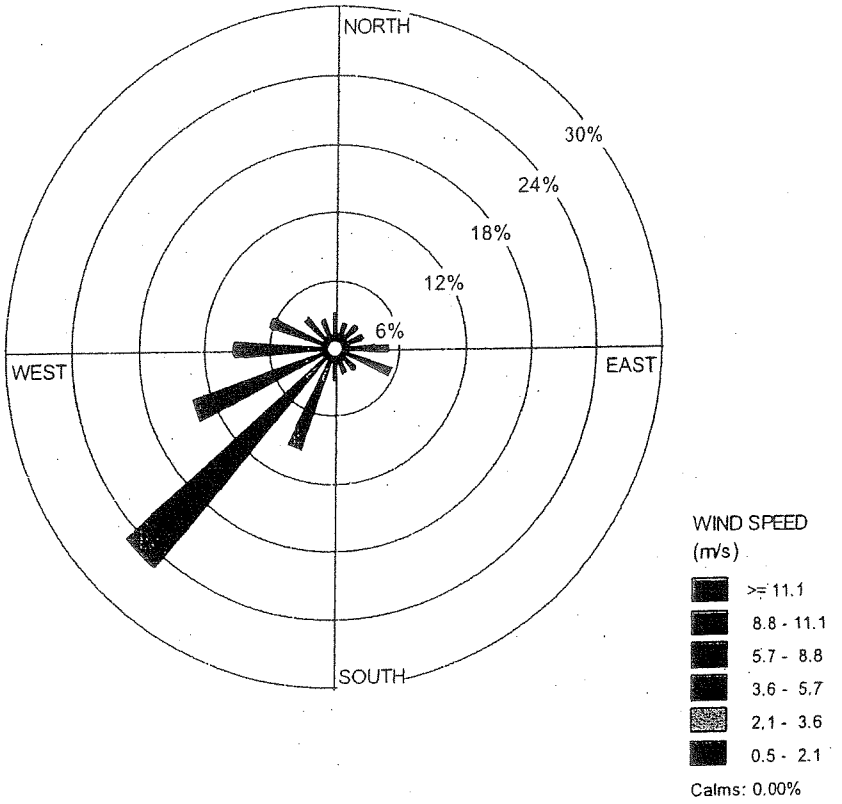

Figure 4. Rawlins Wind Rose

6 Modeling Results

The dispersion modeling was conducted to assess if all the estimated pollutant emissions listed in Table 2 from the crematory source would result in local concentrations that exceed the NAAQS and Wyoming AAQS. PM10, NOx, SOx, and CO were modeled for the various averaging periods compatible with the averaging period of the NAAQS and Wyoming AAQS. There is not a Wyoming AAQS for 1-hour NOx and 1-hour SOx, so the corresponding California AAQS were used for comparison. All other toxics and chemicals were modeled on 1-hour and annual averaged periods. The maximum impact of each pollutant is shown in Table 3 with the NAAQS and Wyoming AAQS.

The maximum 1-hour averaged impact of all the pollutants was predicted to occur to the east of the crematory stack, on the south central portion of the fenceline. Figure 5 shows the 1-hour averaged emission concentration isopleth for mercury. Figure 6 shows the 1-hour averaged emission concentration isopleth for NOx, and the maximum impact points

are indicated as a red star symbol on both figures. The trend of the isopleths for 1-hour averaged concentrations of all pollutants is similar to what is shown in Figures 5 and 6.

The maximum 3-hour averaged impact of SOx occurred north of the crematory stack, at the northwest portion of the fenceline on W. Walnut St. Figure 8 shows the 3-hour averaged emission concentration isopleth for SOx and the maximum impact points are indicated as a red star symbol.

The maximum annual averaged impact of all the pollutants occurred northeast of the crematory stack, at the northeast portion of the fenceline on W. Walnut St. Figure 7 shows the annual averaged emission concentration isopleth for NOx, Figure 9 shows the annual averaged emission concentration isopleth for cadmium, and the maximum impact points are indicated as a red star symbol on both figures. The trend of the isopleths for the annual averaged concentrations of all pollutants is similar to what is shown in Figures 7 and 9. The maximum 24-hour averaged impact of PM10 and SOx also occurred at the same location that the maximum annual averaged impact occurred.

For the toxic pollutants such as arsenic, cadmium, chromium, mercury, nickel, dioxin/furan and HCl, the annual averaged concentrations were predicted and compared with the annual preliminary remediation goal (PRG) for ambient air for EPA Region 9.

The 1-hour averaged concentrations of arsenic, chromium, nickel, and dioxin/furans were predicted by the model, but the annual averaged concentrations of these pollutants were not quantified because they were less than the model's minimum calculation limit of 0.00001 (0.1E-05). Even though the annual averaged concentration is shown as 0.00000 in the model output, this value is not actually zero. Therefore, the annual averaged concentrations of these pollutants were extrapolated from the 1-hour averaged concentrations. The ratio of the other pollutants' 1-hour average to their annual average was found to be 30. By applying this ratio to the 1-hour averaged concentration of arsenic, chromium, nickel, and dioxin/furans, the annual concentrations were found to be 3.33E-07 g/m^3, 3.33E-07 g/m^3, 3.33E-07 g/m^3, 1.0E-06 g/m^3 rather than 0.00000 g/m^3 respectively.

Annual Cadmium and Dioxin/Furan concentrations exceed the PRG significantly by 205% and 2200%, respectively.

Table 3. Comparison of Maximum Predicted Impact of crematory emissions with the NAAQS, Wyoming AAQS, and Annual Preliminary Remediation Goal of EPA Region 9

Pollutant	Avg. Period	Primary NAAQS	Secondary NAAQS	Wyoming AAQS	Annual Preliminary Remediation Goal (EPA Region 9)	Model Predicted Impact	Percent of Model Predicted Impact to Standard
Unit	(g/m³)	(g/m³)	(g/m³)	(g/m³)	(g/m³)	(g/m³)	(%)
PM10	24-hour	150	150	150		16.07	10.71%
	Annual	50	50	50		4.84	9.68%
NOX	1-hour	NA	NA	470 (California AAQS)		203.70	43.34%
	Annual	100	100	100		6.67	6.67%
SOX	1-hour	NA	NA	655 (California AAQS)		300.56	45.89%
	3-hour	NA	1300	1300		138.81	10.68%
	24-hour	365	NA	260		30.67	11.80%
	Annual	80	NA	60		9.85	16.42%
CO	1-hour	40000	NA	40000		140.05	0.35%
	8-hour	10000	NA	10000		35.73	0.36%
VOC	1-hour					63.36	
ARSENIC	Annual				0.00045	3.33E-07 (Estimated Value from Annual)	0.07%
	1-hour				NA	0.00001	*2.22% (Annual Standard Applied)*
CADIMUM	Annual				0.0011	0.00226	*205%*
	1-hour				NA	0.06908	*6280% (Annual Standard Applied)*
LEAD	Annual				NA	0.01364	
	1-hour				NA	0.41624	
CHROMIUM	Annual				0.000023	3.33E-7 (Estimated Value from Annual)	1.44%
	1-hour				NA	0.00001	*43.48% (Annual Standard Applied)*
MERCURY	Annual				0.31	0.00806	2.6%
	1-hour				NA	0.246	*79.35% (Annual Standard Applied)*
NICKEL	Annual				0.008	3.33E-7 (Estimated Value from Annual)	0.004%
	1-hour				NA	0.00001	*0.13% (Annual Standard Applied)*
COPPER	Annual				NA	0.00001	
HCL	Annual				21	5.48468	26.12%
	1-hour				NA	167.38	*797.05% (Annual Standard Applied)*
HF	Annual				NA	0.00012	
Dioxin/Furan	Annual				14 4.5E-08	1.0E-06 (Estimated Value from Annual)	2200%
	1-hour				NA	0.00003	66667% *(Annual Standard Applied)*

Figure 5. Isopleth of 1-hour averaged Mercury Impact (Max: 0.246 g/m^3)

Figure 6. Isopleth of 1-hour averaged NOx Impact (Max: 203.7 g/m^3)

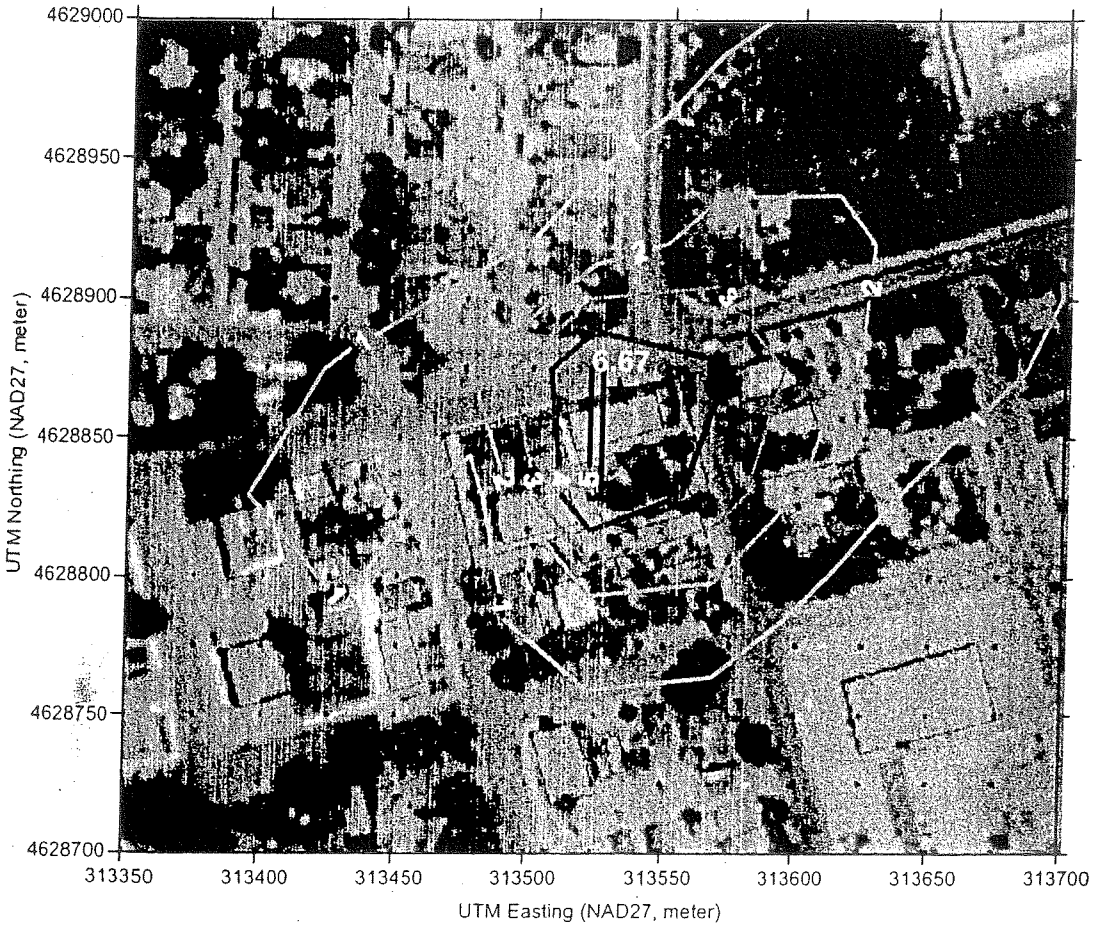

Figure 7. Isopleth of Annual Averaged NOx Impact (Max: 6.67 g/m^3)

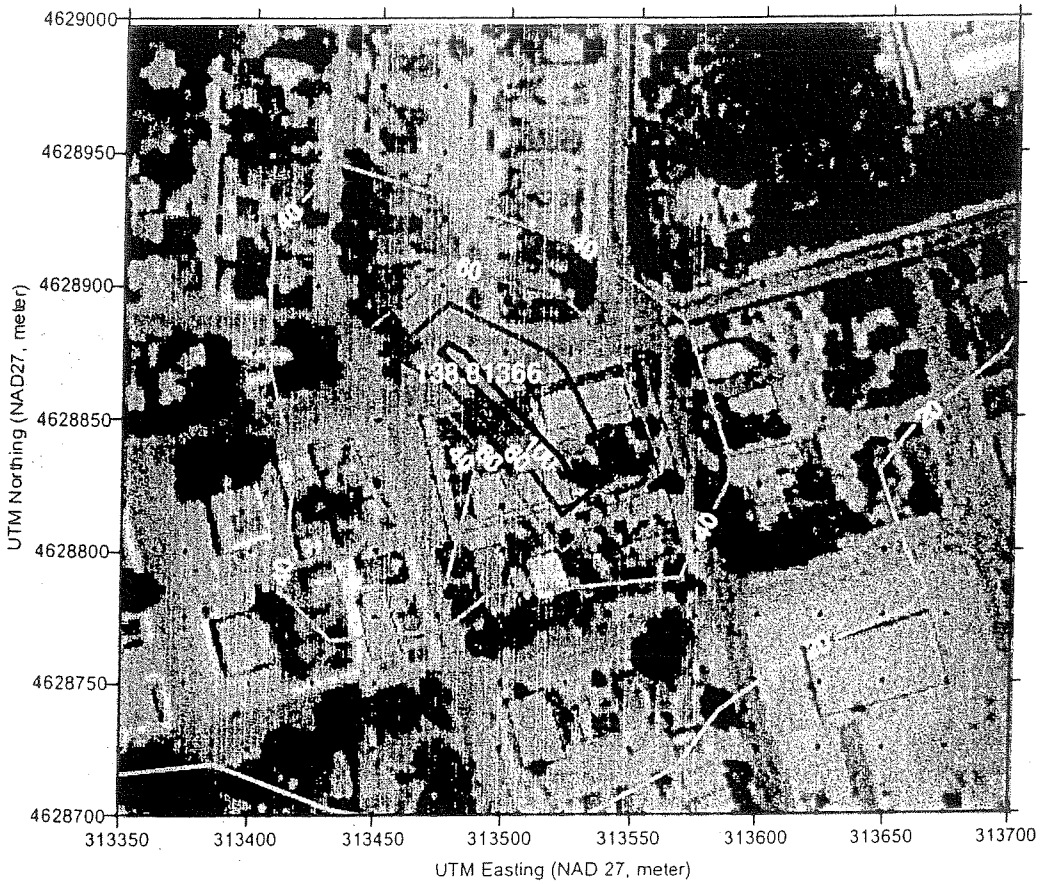

Figure 8. Isopleth of 3-hour Averaged SOx Impact (Max: 138.81 g/m^3)

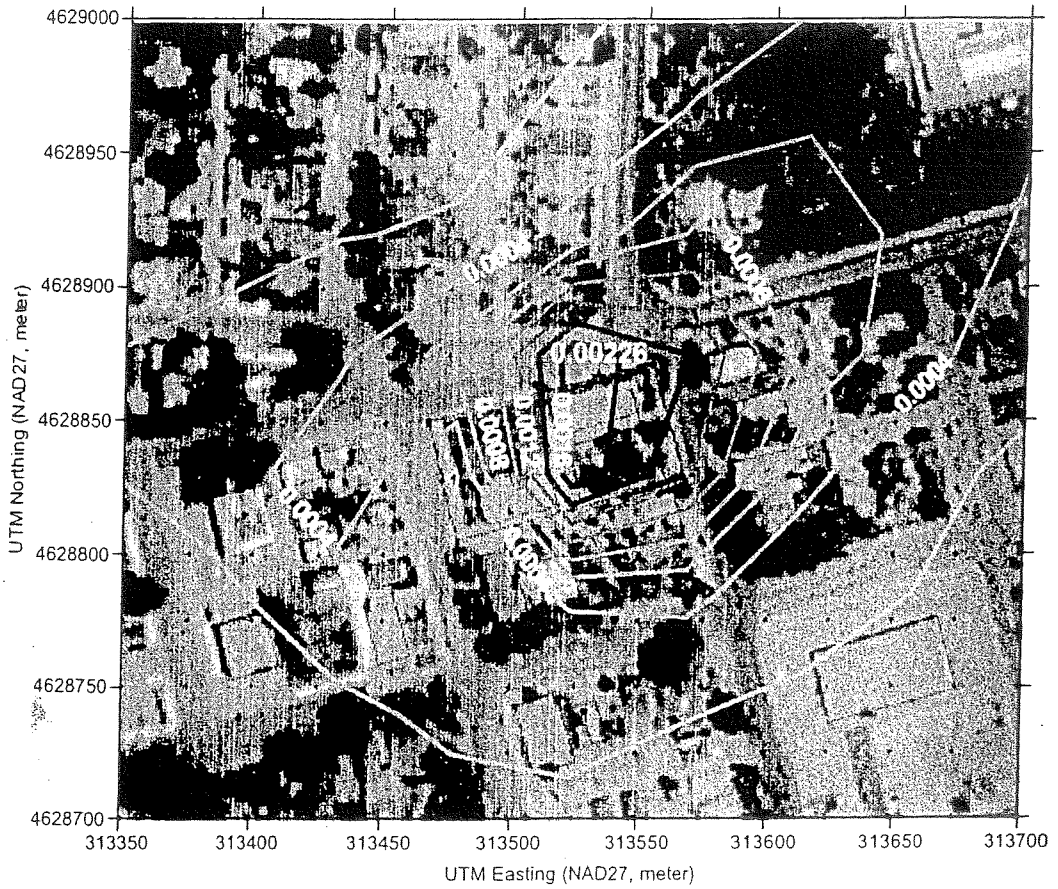

Figure 9. Isopleth of Annual Averaged Cadmium Impact (Max: 0.00226 g/m^3)

7 References

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