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Public health benefits of compliance with current E.U. emissions standards for municipal waste incinerators: A health risk assessment with the CalTox multimedia exposure model

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Abstract

The Angers municipal solid waste incineration plant, in operation since 1974, was upgraded in 2000 to comply with new European standards. This article discusses the risks associated with past and present emissions from the incinerator and its nearby furnace. Emissions of SO_2, HCl, particulate matter, lead, mercury, cadmium and dioxins were studied. We characterised the risks associated with exposure via inhalation and ingestion of locally grown products, before and after the upgrade. Emissions were estimated from regulatory measurements, and ambient air concentrations estimated with a Gaussian dispersion model. The CalTox multimedia model was used to calculate concentrations in the food chain. Food intake rates came from a nationwide survey. Inhalation exposure to respiratory irritants produced a hazard ratio less than 1 in all scenarios, except for SO_2 in the immediate neighbourhood of the incinerator, before the change in furnace fuel and in case of high-pressure weather conditions. The individual excess risk of cancer was less than 10^{-6} and the hazard ratios for metals were less than 1. Before compliance, the average dioxin exposure attributable to the incinerator accounted for roughly one quarter of the average total exposure from traffic and other combustion activities. Although the corresponding hazard ratio was less than 1, the individual lifetime excess risk, assuming no change in emissions, was $2 \cdot 10^{-4}$. After compliance, all hazard ratios and future individual lifetime excess risks appear minimal. These results are consistent with environmental data and other studies, but many uncertainties remain, such as intermedia transfer coefficients for dioxins. Nevertheless compliance has vastly reduced the probability of health effects.
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Keywords: Waste incineration; Health risk assessment; Multimedia model; Dioxin

1. Introduction

In recent decades in France, incineration has increasingly replaced landfills for the treatment of solid waste. Incinerator emissions are complex and depend on the type of waste, the design of the incinerator, combustion conditions and pollution control equipment. Some pollutants (e.g., acid gases, metals and various organic compounds, including dioxins) are associated with health hazards and have thus raised serious concerns in France and elsewhere.

Numerous health risk assessments have studied municipal waste incinerators, especially during their planning stages (NRC (National Research Council), 2000; Snary, 2002). Many dealt with only one kind of pollutant: metals (Sedman et al., 1994) or, most often, dioxins (Domingo, 2002; Ma, 2002; Nouwen and Cornelis, 2001; Schuhmacher et al., 2001; Yoshida et al., 2000), since

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they cause the most common incinerator-related risks. The studies by Nouwen (Nouwen and Cornelis, 2001) and Domingo (Domingo et al., 2002) looked at how decreasing dioxin emissions affected risk, but few authors examine changes due to compliance with new emissions standards.

The risk assessment described here was a response to the concern of the population of Angers, France, about risks due to past and current emissions, after the plant was renovated in 1999 to comply with European directives 89/369 and 89/429 for metals and gases and directive no. 2000/76/CE for dioxins. Because this municipal solid waste incinerator (MSWI) is located in the suburbs of a metropolitan area, not far from a market garden zone, its emissions could have led to significant exposure through ingestion. For these reasons, the study was designed to encompass an array of pollutants (metals, dioxins, acid gas and particulate matter), measured both before and after upgrading of the plant; inhalation and ingestion routes were both considered.

2. Material and methods

The Angers MSWI has been in operation since 1974. The plant capacity is about 100 000 tons per year. The smoke-stack, 60 m high, was equipped with electrofilters before compliance. The emission control devices currently used are fabric filters and a wet acid–gas absorber with lime slurry and activated carbon injection. A nearby furnace provides an additional power of 60 GW to the city heating system, as needed. Its stack is 36 m high. It has used very low sulphur content fuel (<1%) since 1985.

The Angers metropolitan area has a total population of 230 000; and the city, 160 000. The incinerator is surrounded by a densely populated zone and by an agricultural zone with market garden farming and horticulture (cf. Fig. 1). After assessment of the influence of the stacks' atmospheric emissions by a Gaussian dispersion model, the study zone was defined as the area where ambient air concentrations exceeded 10% of the maximum modelled concentration and consists of an area of a radius of about 3 km around the plant. This zone includes a population of approximately 100 000 inhabitants.

The absence of stagnant water–there is only a river–means that neither the public water supply nor fish can be significantly contaminated. The likelihood of exposure through direct contact is considered negligible (EPA (US Environmental Protection Agency), 1998). Hence, exposure takes the following routes: inhalation of contaminated air, ingestion of locally grown products, and ingestion of soil particles.

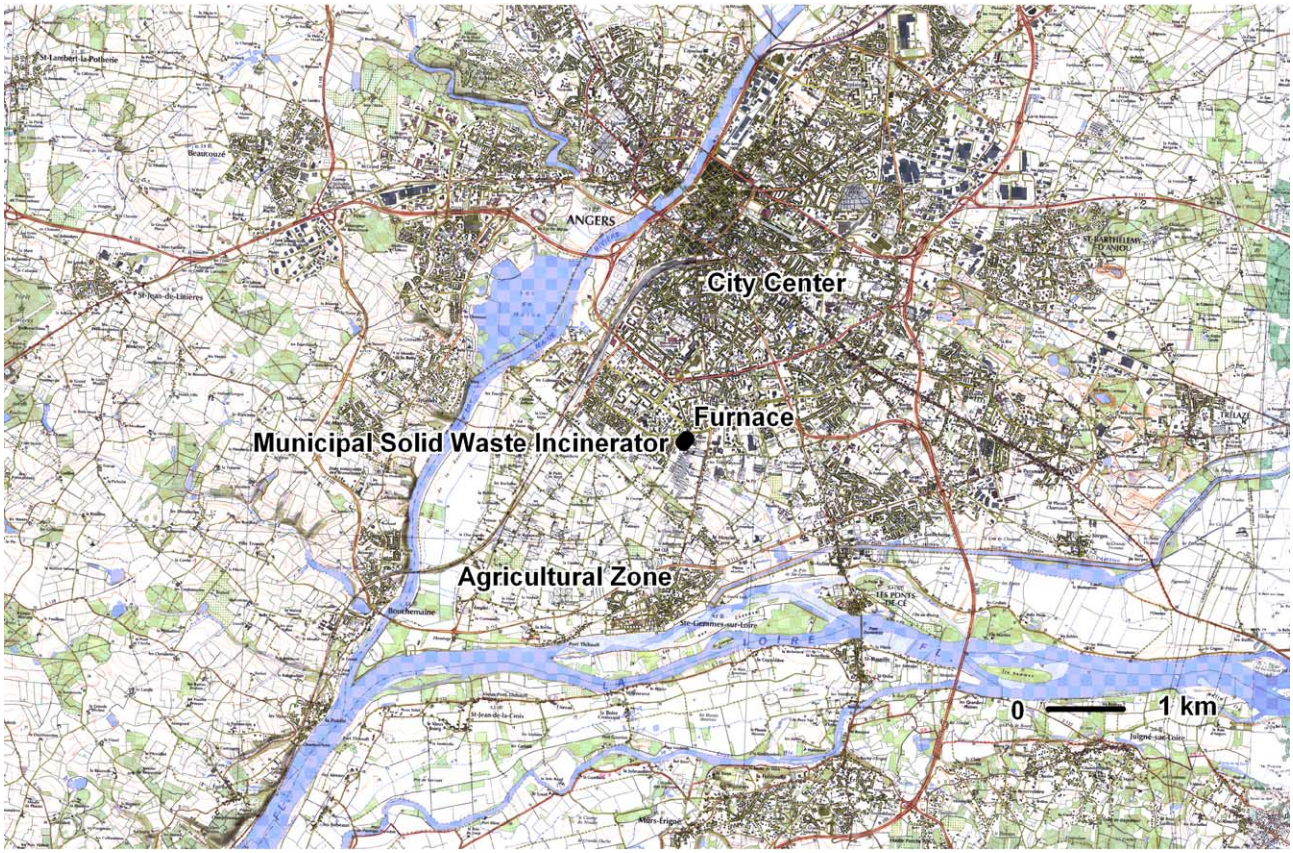


Fig. 1.

Of the many pollutants that have been measured at the incinerator stack, we selected several compounds according to the following criteria:

- quantity emitted (a criterion that supported the study of sulphur dioxide, hydrogen chlorine, particulate matter, cadmium, lead, mercury and chromium),
- specificity of incinerator emissions relative to urban background (cadmium and mercury),
- known health effects and available human toxicity values,
- social concern (dioxins and lead).

We chose 7 pollutants: sulphur dioxide (SO_2), hydrogen chlorine (HCl), particulate matter (PM), cadmium (Cd), lead (Pb), mercury (Hg) and dioxins. Chromium was not selected because its species at emission are unknown; other studies indicate that its toxic forms account for only a small fraction of total chromium emissions (Hallenbeck et al., 1993).

We studied the exposure routes and effects of these compounds: irritation of the respiratory system (inhaled sulphur dioxide, hydrogen chlorine and particulate matter); neuro-behavioural effects (inhaled and ingested lead, inhaled mercury); lung cancer (inhaled cadmium); renal effects (ingested cadmium); all types of cancer (ingested dioxins) and developmental risks (ingested dioxins).

We selected the corresponding human toxicity reference values according to the World Health Organisation (WHO), the United States Environmental Protection Agency (EPA) or the French High Council of Public Hygiene, as reported in Table 1. There is still no international agreement about whether a threshold exists for dioxins; the literature includes two different human toxicity values (EPA (Environmental protection agency), 2001a; WHO (World Health Organisation), 1998) (the EPA values are still in draft version and are not official), and we performed risk calculations with both. The toxicity values for SO_2 and particles are based on guidelines for ambient air concentrations from WHO (WHO (World Health Organisation), 2000) and the French High Council of Public Hygiene (Conseil Supérieur d'Hygiène Publique de France, 1996).

Incinerator emissions were estimated from measurements conducted for regulatory purposes, available from 1990 and after. Because measurements were taken rarely (once a year in general), the emission rate was estimated as the median value; for the sensitivity analysis, a confidence interval was calculated from the arithmetic mean (Student method). Only 4 dioxin values were available before renovation; hence, the range was set to the minimum and maximum observed values. For particles, the PM10 fraction (for which risks can be derived from epidemiological data) was estimated at 62% of the total particulate matter (EPA (Environmental Protection Agency), 1992). The estimated emission rates are indicated in Table 2.

The furnace sulphur dioxide emission rate was estimated at 38 g/s from 1975 through 1985, and 6 g/s after 1985. This estimation is based on fuel consumption data and an SO_2 emission factor of 78 kg/ton before 1985 and 20 kg/ton thereafter, when the city switched to a fuel much lower in sulphur content, <1% (Angers municipal services, personal communication).

Because the zone is flat, we used a Gaussian plume dispersion model to calculate the ambient air concentrations attributable to the incinerator—and to the furnace for SO_2. We used POL'ER software, a risk assessment model developed for point sources for the French Institute of Public Health Surveillance by the public health laboratory of Grenoble University (Balducci, 2000). We calculated concentrations in a grid, with cells defined by their distance (500-m steps) and position (20° steps) from the source. Deposition speed of pollutants was set at the following values: $8 \cdot 10^{-3}$ m/s for PM (Mc Mahon and Denison, 1979), $5 \cdot 10^{-3}$ for metals and SO_2 (Sehmel, 1980), $2 \cdot 10^{-3}$ for dioxins (SFSP (Société Française de

Table 1
Human toxicity values used for health risk assessment of the Angers incinerator

Pollutants	Source	Human toxicity value for chronic (and acute[a]) exposure ($\mu g/m^3$)	IUR inhalation unit risk $(\mu g/m^3)^{-1}$	Reference dose for chronic oral exposure (RfD) (mg/kg day)	Oral unit risk (OUR) $(\mu g/kg\ day)^{-1}$
HCl	EPA (2001b)	20			
SO_2	WHO (2000)	50 (125)			
PM	CSHPF[b] (1996)	30 (80)			
Hg	EPA (2001b)	0.3			
Pb	WHO (2000)	0.5			
Pb	JECFA (1999)			$3.5 \cdot 10^{-3}$	
Cd	EPA (2001b)		$1.8 \cdot 10^{-3}$		
Cd	JECFA (1999)			$1 \cdot 10^{-3}$	
Dioxins (I-TEQ)	EPA (2001a)				$1 \cdot 10^{+3}$
Dioxins (I-TEQ)	WHO (1998)			$1 \cdot 10^{-9}$	

[a] [For a 24 h exposure].
[b] French high council for public hygiene.

Table 2
Emissions from the Angers incinerator, before and after upgrading of the plant

Compound	Incinerator's median emission rate (g/s) (average 95% CI). 1975–1999	Incinerator's median emission rate (g/s) (average 95% CI). 2000–2001
SO_2	1.9 (1.2–6.9)	$7.8 \cdot 10^{-2}$ ($3.2 \cdot 10^{-2}$–$1.4 \cdot 10^{-1}$)
HCl	7.51 (4.2–8.6)	$6.3 \cdot 10^{-2}$ ($4.0 \cdot 10^{-2}$–$7.8 \cdot 10^{-2}$)
PM	1.17 (1.0–2.5)	$7.3 \cdot 10^{-2}$ (0–$2.9 \cdot 10^{-1}$)
Pb	$1.0 \cdot 10^{-2}$ ($8.2 \cdot 10^{-3}$–$1.8 \cdot 10^{-2}$)	$3.9 \cdot 10^{-4}$ ($6.2 \cdot 10^{-5}$–$2.0 \cdot 10^{-3}$)
Cd	$1.6 \cdot 10^{-3}$ ($1.2 \cdot 10^{-3}$–$3.7 \cdot 10^{-3}$)	$5.6 \cdot 10^{-4}$ ($1.7 \cdot 10^{-4}$–$1.2 \cdot 10^{-3}$)
Hg	$8.5 \cdot 10^{-4}$ ($5.9 \cdot 10^{-4}$–$2.2 \cdot 10^{-3}$)	$7.4 \cdot 10^{-4}$ (0–$2.3 \cdot 10^{-3}$)
Dioxins (I-TEQ)	$1.5 \cdot 10^{-7}$ ($1.3 \cdot 10^{-7}$–$1.6 \cdot 10^{-7}$) (min–max)	$2.2 \cdot 10^{-10}$ ($1.6 \cdot 10^{-10}$–$4.5 \cdot 10^{-10}$) (min–max)

Santé Publique), 1999) and 0 for HCl (no data, 0 chosen as a default value). Meteorology data came from the Angers monitoring station of Meteo-France (the French national weather service). Two situations were considered: (1) typical high-pressure weather conditions, to study the acute exposures relevant for respiratory irritants (SO_2 and PM); and (2) average conditions over the past 30 years, to study chronic exposures and accumulation of pollutants in the food chain.

To assess exposure through inhalation, we calculated ambient air concentrations for each grid cell and assumed an outdoor/indoor ratio equal to one. Exposures and risks were considered for both normal and unfavourable conditions of dispersion and calculated for 24-h and chronic exposure periods. To study exposure through ingestion, we calculated the concentrations of pollutants in the food chain with the widely used CalTox (v2.3) multimedia software, developed by the U.S. EPA (EPA (Environmental Protection Agency), 1997). The model combines a transport and transformation model of pollutants in the environment and a multimedia multiple pathway exposure model. This model was chosen because of its relevance, in terms of source term, exposure pathways, and types of environments and chemicals. CalTox input parameters were chosen as follows. The source term was the ambient air concentrations in the market garden farming zone. Chemical parameters were CalTox default parameters except for dioxin, for which we had more recent literature (EPA (Environmental Protection Agency), 1998) available. For environmental fate, all dioxins were assumed to behave like OCDD (octochlorodibenzodioxin), the congener emitted in the largest quantity, and the OCDD/TCDD ratio was assumed to range from 4 to 20 before compliance and from 60 to 250 afterwards. Local data from government meteorological and agriculture offices were used to complete the site-specific meteorological and soil parameters. Ingestion of local products was assumed to equal nationwide consumption of home-grown products: 28% for fruits and vegetables, 10% for meat, 13% for eggs, and 4% for milk (Bertrand, 1993). All model parameters are presented in Appendix A.

We characterised the risks for individuals who lived in Angers and the surrounding area for two time periods of residence: from 1974 to 1999 and from 2000 to 2030. For carcinogenic compounds, individual excess risks were calculated for corresponding exposure durations by multiplying exposure, averaged over exposure duration, by the corresponding (inhalation or oral) unit risk. For so-called 'threshold effects', exposures were compared with the reference concentration (RfC) or dose (RfD), that is, we calculated a hazard ratio (exposure divided by the corresponding reference concentration or dose). Using census data to estimate population in each cell, we characterised the risks for each cell and the population-averaged risk over all cells.

A simple sensitivity analysis was also performed: high and low values were set as input parameters and the results compared with those obtained with the 'best estimate' input parameter.

3. Results

We present separately the findings for before and after compliance with EU standards, considering first inhalation (for normal and unfavourable dispersion conditions), then ingestion.

Table 3 summarises the ambient air concentrations and associated noncarcinogenic inhalation risks before compliance with EU standards. Carcinogenic effects were due to cadmium inhalation, with mean and maximum exposure concentrations of respectively $6.5 \cdot 10^{-4}$ and $2 \cdot 10^{-3}$ $\mu g/m^3$. The corresponding individual risks were $4 \cdot 10^{-7}$ and

Table 3
Ambient concentrations and inhalation hazard ratios from the Angers incinerator and its nearby furnace[a] 1975–1999

Pollutant	Average ambient concentrations ($\mu g/m^3$)	Maximum ambient concentrations ($\mu g/m^3$)	Population averaged hazard ratio	Maximum hazard ratio
SO_2	9	50	0.1	0.9
HCl	3	9	0.1	0.5
PM10	0.5	1	$1 \cdot 10^{-2}$	$5 \cdot 10^{-2}$
Pb	$4 \cdot 10^{-3}$	$1 \cdot 10^{-2}$	$7 \cdot 10^{-3}$	$3 \cdot 10^{-2}$
Hg	$4 \cdot 10^{-4}$	$1 \cdot 10^{-3}$	$1 \cdot 10^{-3}$	$4 \cdot 10^{-3}$
During unfavourable dispersion conditions				
SO_2	$1.4 \cdot 10^{+1}$	$3.4 \cdot 10^{+2}$	$8 \cdot 10^{-3}$	2.7

[a] Before using very low sulphur content fuel.

Table 4
Exposure duration averaged daily dose, by exposure medium, attributable to the Angers incinerator, 1975–1999

Exposure medium	Dioxin (I-TEQ)		Cadmium		Lead	
	(pg/(kg day))	%	(μg/(kg day))	%	(μg/(kg day))	%
Exposed fruits and vegetables	$4 \cdot 10^{-1}$	80	$1 \cdot 10^{-2}$	60	$7 \cdot 10^{-2}$	73
Unexposed fruits and vegetables	$9 \cdot 10^{-6}$	0	$2 \cdot 10^{-5}$	0	$2 \cdot 10^{-5}$	0
Meat	$9 \cdot 10^{-2}$	16	$6 \cdot 10^{-3}$	35	$2 \cdot 10^{-2}$	21
Milk	$1 \cdot 10^{-2}$	2	$2 \cdot 10^{-5}$	0	$3 \cdot 10^{-4}$	0
Eggs	$7 \cdot 10^{-5}$	0	$2 \cdot 10^{-5}$	0	$8 \cdot 10^{-5}$	0
Soil	$1 \cdot 10^{-2}$	2	$7 \cdot 10^{-4}$	4	$5 \cdot 10^{-3}$	5
Total ingestion	$5 \cdot 10^{-1}$	100	$2 \cdot 10^{-2}$	100	$9 \cdot 10^{-2}$	100

$1 \cdot 10^{-6}$ for a 25-year exposure. For ingestion risks, we first calculated the ambient air concentrations in the market garden farming zone (in μg/m^3) as $6.9 \cdot 10^{-8}$ for dioxins, $7.4 \cdot 10^{-4}$ for cadmium and $4.8 \cdot 10^{-4}$ for lead. The associated doses, calculated with CalTox, are summarised in Table 4. The hazard ratio for dioxins (following the threshold value set by WHO) was $5 \cdot 10^{-1}$, and the excess risk (EPA approach) for a 25-year exposure $2 \cdot 10^{-4}$. The hazard ratio was $2 \cdot 10^{-2}$ for cadmium and $3 \cdot 10^{-2}$ for lead.

Table 5 summarises the ambient air concentrations and associated noncarcinogenic inhalation risks after compliance with the new EU standards. Mean and maximum exposure concentrations for cadmium were respectively $2.3 \cdot 10^{-4}$ and $6.9 \cdot 10^{-4}$ μg/m^3, and the corresponding individual cancer risks $2 \cdot 10^{-7}$ and $5 \cdot 10^{-7}$. The ambient air dioxin concentrations in the market garden farming zone were reduced (in μg/m^3) to $1 \cdot 10^{-10}$; figures for cadmium and lead were $2.6 \cdot 10^{-4}$ and $1.8 \cdot 10^{-4}$. The associated doses were $8 \cdot 10^{-4}$ pg/kg day, $6 \cdot 10^{-3}$ and $4 \cdot 10^{-3}$ μg/kg day. The hazard ratio for dioxins (WHO approach) was $1 \cdot 10^{-3}$, and the excess risk (EPA approach) $3 \cdot 10^{-7}$ for 30 years exposure. The hazard ratio for cadmium was $6 \cdot 10^{-3}$ and for lead $1 \cdot 10^{-3}$.

4. Discussion

The risk assessment of the Angers MSWI identified two situations of potential concern: the inhalation of SO_2 during episodes of little atmospheric dispersion, and the ingestion of local products, which can result in exposure exceeding the current French mean by 25%. We discuss here the uncertainties and variability of these results, compare them with other studies, and finally interpret the risk estimates.

There are uncertainties because it is impossible to assess the risks associated with all the pollutants contained in MSWI emissions. Only a few are measured for surveillance purposes, and assessment is limited to the compounds measured for which we have sufficient data on health effects. The U. S. National Research Council (NRC (National Research Council), 2000) has concluded that the most relevant emissions are those of particulate matter, lead, mercury and dioxins. All were considered here.

Human toxicity values depend on the current state of knowledge. Hu and Shy (Hu and Shy, 2001) recently reviewed epidemiologic studies of health effects related to waste incineration and concluded that no effects on respiratory symptoms or pulmonary function have been observed and that the findings for cancer and reproductive outcomes are inconsistent. In other words, the estimated risk is subject to substantial uncertainty, especially that of dioxins, for which the dose–response relation is still debated in the scientific and regulatory arenas. The human toxicity value chosen for particles also raises questions, for they are a mixture of compounds. We chose not to use the relative risk estimates derived by WHO (WHO (World Health Organisation), 2000) from cohort studies because their applicability for particles of industrial origin is unclear. Another work (SFSP (Société Française de Santé Publique), 1999), however, did use the relative risk estimates for life expectancy related to urban particles (Brunekreef, 1997) and observed no significant loss of life expectancy associated with stack emissions. Moreover the contribution of MSW incineration to ambient air PM concentrations is miniscule compared with traffic sources (0.5 versus the 20–30 μg/m^3 typically observed in French cities (Ministère de l'écologie et du développement durable, 2002). These uncertainties are thus not expected to affect risk estimates greatly.

Time series of pollutant emissions show the quantities measured to vary widely. This may be due to natural fluctuations but also to sampling–or measurement–errors. A sensitivity analysis (data not shown) of the particle emissions with a 95% Confidence Interval (CI) boundary on the mean did not find that the mean emission rate estimates influenced risk substantially. Moreover the emission rate estimated for the Angers incinerator is quite similar (within a factor of 2) to the national

Table 5
Air exposure concentrations and inhalation hazard ratios from the Angers incinerator and its nearby furnace 2000–2001

Pollutant	Average exposure concentration (μg/m^3)	Maximum exposure concentration (μg/m^3)	Population averaged hazard ratio	Maximum hazard ratio
SO_2	$1.6 \cdot 10^{-2}$	$2.9 \cdot 10^{-1}$	$7.9 \cdot 10^{-6}$	$2.4 \cdot 10^{-3}$
HCl	$1.3 \cdot 10^{-2}$	$2.5 \cdot 10^{-1}$	$4.3 \cdot 10^{-5}$	$1.2 \cdot 10^{-2}$
PM10	$1.5 \cdot 10^{-2}$	$2.7 \cdot 10^{-1}$	$1.2 \cdot 10^{-5}$	$3.4 \cdot 10^{-3}$
Pb	$7.9 \cdot 10^{-5}$	$1.5 \cdot 10^{-3}$	$1.0 \cdot 10^{-5}$	$2.9 \cdot 10^{-3}$
Hg	$1.5 \cdot 10^{-4}$	$2.8 \cdot 10^{-3}$	$4.8 \cdot 10^{-5}$	$1.4 \cdot 10^{-2}$
During unfavourable dispersion conditions				
SO_2	2.4	$5.6 \cdot 10^{+1}$	$1 \cdot 10^{-3}$	$5 \cdot 10^{-1}$

estimates (SFSP (Société Française de Santé Publique), 1999). The concentrations modelled from furnace fuel consumption data are close to the few measurements available.

The Gaussian type model used to assess atmospheric concentrations is adequate for nonreactive gases and fine particles in plains-type areas, which the Angers region is. SO_2 and HCl are reactive gases, however, and their concentrations and exposures are thus overestimated. Nevertheless, the concentrations we estimated are quite similar, within an order of magnitude, to the attributable concentrations reported near old incinerators in France (SFSP (Société Française de Santé Publique), 1999) and the United States (NRC (National Research Council), 2000).

Inhalation exposure was assumed to be equal to ambient air concentrations around homes. While this assumption depends heavily upon the compound, the house ventilation characteristics and the season, the indoor/out door concentration rate ranges from 0.25 to 1 (Monn et al., 1987). This also tends to overestimate exposures. Boudet (Boudet et al., 1999) found, in a place similar to our study area, especially in that land where use regulations do not create a sharp partition between work and residential areas, that estimated exposures with time–activity weighted exposure were quite similar to those without this weighting.

CalTox, like other multimedia exposure models, has not been fully validated, and use of its equations can therefore produce uncertain estimates. For instance, some chemical parameters are based on sparse data. Computation default values tend to overestimate exposures. An example is the absence of consideration by CalTox of cooking procedures, such as whether fruits and vegetables are washed or peeled; deposed dioxins and metals are in a particulate form that can be removed with good cooking habits. All dioxins were assumed to behave like OCDD (octochlorodibenzodioxin), the congener emitted in greatest quantity. In an alternative model, 20% (that is, the upper bound of likely values) of emitted dioxins were considered to be TCDD: the resulting ingested dose was lower in mass (TCDD has lower bio-transfer rates) but of same order of I-TEQ magnitude, because of its higher toxicity. To assess uncertainty as well as variability, we conducted a sensitivity analysis for dioxin, the most critical pollutant. The parameters included are those mentioned by EPA (EPA (Environmental Protection Agency), 2001a) as sensitive: source intensity, transfer parameters and exposure (intake, time of exposure). This analysis showed that the greatest variability (proportional to the input value) was due to exposure factors (time of residence, local food consumption) and that the main uncertainty was due to intermedia transfer parameters. We could not quantify the uncertainty because of the lack of alternative data for rain-splash and partitioning (air particle) factors. An error of one order of magnitude in input value would result an error of a similar order of magnitude for average daily dose but we cannot establish if this 10-fold error factor in input value is realistic. In addition, population variability has been examined through a worst case scenario: a person living close to the incinerator who consumes home-grown foodstuffs at the upper boundary of the nationwide survey. This subject would receive a dose 4.5 times greater than the mean. For dioxins, we compared the media concentrations calculated (ambient air, soil, vegetables, milk) with values observed in various French and European surveys (INSERM (Institut National de la Santé et de la Recherche Médicale), 2000); the calculated concentrations were, as can be seen in Table 6, within the range of observed values. The milk concentrations were also compatible with the rare data available for local cattle. The principal exposure pathways modelled (vegetable contamination, vegetable and soil ingestion by cattle) are consistent with those from other work (INSERM (Institut National de la Santé et de la Recherche Médicale), 2000; NRC (National Research Council), 2000; Schuhmacher et al., 2001). Similarly, the exposure levels we calculated also agree with those reported in other health risk assessments (Domingo et al., 2002; Yoshida et al., 2000) or measured in humans (Fierens et al., 2003). It is worth noting that the fruit and vegetable consumption here is important for the dose. The reason is that vegetables are the foodstuff most often locally produced and consumed; it is this aspect that is specific to MSWI dioxin exposure, which therefore should not be compared with typical total dioxin exposure, which mainly occurs through animal product consumption.

Interpreting dioxin exposure and risks before renovation of the plant is difficult in view of the many uncertainties, including whether dioxin toxicity involves a threshold level. The model for the pre-renovation period yields ambient air concentrations attributable to the incinerator of about the same order of magnitude as urban background levels due to sources such as traffic and residential combustion activities. The excess exposure from ingestion was approximately 25% of the current (INSERM (Institut National de la Santé et de la

Table 6
Estimated concentrations near the Angers MSWI (1975–2000) compared with concentrations elsewhere in France and Europe near MSWIs

	Angers MSWI before compliance	Observed values (INSERM, 2000)
Emission concentration (ng/Nm^3)	9–16	0.1–100 (France)
Air concentrations (fg I-TEQ/m^3)	65	1–800 (United Kingdom and Germany)
Soil concentrations	27 pg/g of surface soil	13–252 pg/g of soil (in Netherlands)
Vegetables pg/g dry weight	3	1–50 (France)
Milk pg/g fat matter	2	0.5; 30 (France)

Recherche Médicale), 2000; InVS et al., 2000) French mean exposure levels for adults and 5% for children. Hence, assessing the importance of exposure due to the incinerator requires assessing the consequences of current exposure from other sources, which is beyond the scope of this study. After the incinerator was upgraded to meet EU standards, the individual excess risk (over a 70-year period) from dioxins was $8 \cdot 10^{-7}$—a negligible level of risk. As frequently observed around *modern* incinerators, total dioxin exposure is mainly due to other sources (Domingo, 2002). Our results concerning dioxins are consistent with other risk assessments in France (Boudet et al., 1999; SFSP (Société Française de Santé Publique), 1999) and elsewhere (Domingo et al., 2002; NRC (National Research Council), 2000); these assessments find low risks for modern incinerators but not for old ones.

Our results suggest that the environmental concentrations of metals attributable to the incinerator, both before and after its upgrading, are low compared with background values measured in urban areas and with guidelines for vegetable protection (WHO (World Health Organisation), 2000). This is partly due to the emission rate but also to the height of the stack (60 m) and consequently to the dilution of the pollutants emitted. The corresponding hazard ratios and excess risks are low. Safety margins appear sufficient to conclude with confidence that the probability of health effects is minimal.

For respiratory irritants, the short-term hazard ratio for SO_2 was close to or greater than 1, depending on meteorological conditions, but it is now <0.5. This pollutant comes mainly from the furnace. These figures indicate that irritation of the respiratory system among the general population or susceptible subjects might well have occurred before 1985, when the furnace operated with higher sulphur-content fuel. These health effects might have occurred among those living near the furnace and most likely during high-pressure weather conditions. Because furnace operation was irregular it is difficult to evaluate the number of "at risk" days. After year 2000, considering the general overestimation of inhalation concentrations, the estimated hazard ratios seem to be low enough to assume that no adverse effects should occur.

Overall, we conclude that compliance of this municipal waste incinerator with current emissions standards substantially reduced emissions and exposures. For the inhalation pathway, reductions in risks and emissions are proportional, that is, one or two orders of magnitude, depending on the pollutant, as seen in Table 2. The decrease is substantial for acid gases, less so for metals because their attributable concentrations were low compared with other sources, especially traffic. But the major benefit has been the removal of huge quantity of dioxins from emissions with modern air pollution control systems. The emission rate has been divided by more than 500 and exposures due to incinerator have reached acceptable values that make a major difference to public health.

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Appendix A

Table 1A
CalTox input values for dioxins

	CalTox symbol	Used value	Source	Comment
Molecular weight (g/mol)	MW	322	HHRAP[a]	TCDD MW for results presentation
Octanol–water partition coefficient	Kow	$3.89 \cdot 10^{7}$	HHRAP	
Melting point (K)	Tm	598.1	HHRAP	
Vapor pressure in (Pa)	VP	$1.09 \cdot 10^{-10}$	HHRAP	
Solubility in mol/m^3	S	$7.4 \cdot 10^{-8}$	HHRAP	
Henry's law constant (Pa-m^3/mol)	**H-**	0.68	EPA, 2000[b]	
Diffusion coefficient in pure air (m^2/day)	Dair	$9.16 \cdot 10^{-02}$	HHRAP	
Diffusion coefficient; pure water (m^2/day)	Dwater	$3.19 \cdot 10^{-6}$	HHRAP	
Organic carbon partition coefficient Koc	**Koc-**	$2.4 \cdot 10^{7}$	HHRAP	
Partition coefficient in ground/root soil layer	**Kd_s-**	$5.4 \cdot 10^{5}$	HHRAP	Soil organic content 2.25%
Partition coefficient in vadose-zone soil layer	**Kd_v-**	$2.69 \cdot 10^{4}$	CalTox[c]	No importance in our case
Partition coefficient in aquifer layer	**Kd_q-**	$5.38 \cdot 10^{4}$	CalTox	No importance in our case
Partition coeffic. in surface wtr sediments	**Kd_d-**	$1.72 \cdot 10^{5}$	CalTox	No importance in our case
Prtn cff. plnt(abv-grd)/sl (kg[s]/kg[pFM])	**Kps-**	$2.06 \cdot 10^{-4}$	HHRAP	Water content 87%
Biotrnsfr fctr, plant/air (m^3[a]/kg[pFM])	**Kpa-**	$2.58 \cdot 10^{5}$	HHRAP	Water content 87%; air density 1.19
Biotransfer factor; cattle-diet/milk (day/L)	**Bk-**	$1 \cdot 10^{-3}$	HHRAP	Density 1
Biotransfer factor; cattle-diet/meat (day/L)	**Bt-**	$5.43 \cdot 10^{-3}$	HHRAP	Density 1
Biotransfer factor; hen-diet/eggs (day/L)	**Be-**	$4.51 \cdot 10^{-2}$	HHRAP	
Biotrnsfr fctr; brst mlk/mthr intake (day/kg)	**Bbmk-**	$9.24 \cdot 10^{-01}$	CalTox	
Bioconcentration factor; fish/water	**BCF-**	$1.86 \cdot 10^{4}$	CalTox	No importance in our case
Skin permeability coefficient; cm/h	**Kp_w-**	$1.03 \cdot 10^{-2}$	CalTox	
Fraction dermal uptake from soil	**dfct_sl-**	1.00	CalTox	
Reaction half-life in air (day)	Thalf_a	$3.00 \cdot 10^{+1}$	CalTox	
Reaction half-life in surface soil (day)	Thalf_g	$2.3 \cdot 10^{3}$	HHRAP	
Reaction half-life in root-zone soil (day)	Thalf_s	$6.66 \cdot 10^{+3}$	CalTox	
Reaction half-life in vadose-zone soil (day)	Thalf_v	$6.66 \cdot 10^{+3}$	CalTox	
Reaction half-life in ground water (day)	Thalf_q	$1.51 \cdot 10^{+3}$	CalTox	
Reaction half-life in surface water (day)	Thalf_w	$4.29 \cdot 10^{+2}$	CalTox	
Reaction half-life in sediments (day)	Thalf_d	$2.06 \cdot 10^{+3}$	CalTox	

[a] Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (HHRAP: http://www.epa.gov/epaoswer/hazwaste/combust/riskvol.htm#volume1).
[b] EPA dioxin reassessment.
[c] CalTox default parameter.

Table 2A
Site specific CalTox input parameters of interest

Designation and unit	CalTox symbol	Used value
Annual average precipitation (m/day)	rain	$1.83 \cdot 10^{-3}$
Flux; surface water into landscape (m/day)	inflow	0,00
Land surface runoff (m/day)	runoff	$1.40 \cdot 10^{-4}$
Atmospheric dust load (kg/m^3)	rhob_a	$3.8 \cdot 10^{-8}$
Deposition velocity of air particles (m/day)	v_d	$6.90 \cdot 10^{+2}$
Plant dry mass inventory (kg[DM]/m^2)	bio_inv	2.80
Plant dry-mass fraction	bio_dm	$2.20 \cdot 10^{-1}$
Plant fresh-mass density kg/m^3	rho_p	$8.30 \cdot 10^{+2}$
Ground-water recharge (m/day)	recharge	$5.00 \cdot 10^{-4}$
Evaporation of water from surface wtr (m/day)	evaporate	$4.38 \cdot 10^{-6}$
Thickness of the ground soil layer (m)	d_g	$1.00 \cdot 10^{-2}$
Soil particle density (kg/m^3)	rhos_s	$1.15 \cdot 10^{+3}$
Water content in surface soil (vol fraction)	beta_g	$1.31 \cdot 10^{-1}$
Air content in the surface soil (vol frctn)	alpha_g	$5.3 \cdot 10^{-1}$
Erosion of surface soil (kg/m^2 day)	erosion_g	$3.00 \cdot 10^{-4}$
Thickness of the root-zone soil (m)	d_s	$6.5 \cdot 10^{-1}$
Water content of root-zone soil (vol. frctn.)	beta_s	$1.25 \cdot 10^{-1}$
Air content of root-zone soil (vol. frctn.)	alpha_s	$5.3 \cdot 10^{-1}$
Thickness of the vadose-zone soil (m)	d_v	1
Water content; vadose-zone soil (vol. frctn.)	beta_v	$2.80 \cdot 10^{-1}$

Table 2A (*continued*)

Designation and unit	CalTox symbol	Used value
Air content of vadose-zone soil (vol. frctn.)	alpha_v	$1.70 \cdot 10^{-1}$
Thickness of the aquifer layer (m)	d_q	$1.00 \cdot 10^{-3}$
Solid material density in aquifer (kg/m^3)	rhos_q	$2.65 \cdot 10^{+3}$
Porosity of the aquifer zone	beta_q	$2.00 \cdot 10^{-1}$
Fraction of land area in surface water	f_arw	$4.70 \cdot 10^{-2}$
Average depth of surface waters (m)	d_w	$5.00 \cdot 10^{+0}$
Suspended sediment in surface wtr (kg/m^3)	rhob_w	$8.80 \cdot 10^{-2}$
Suspended sdmnt deposition (kg/m^2/day)	deposit	$1.05 \cdot 10^{+1}$
Thickness of the sediment layer (m)	d_d	$5.00 \cdot 10^{-2}$
Solid material density in sediment (kg/m^3)	rhos_d	$2.65 \cdot 10^{+3}$
Porosity of the sediment zone	beta_d	$2.00 \cdot 10^{-1}$
Sediment burial rate (m/day)	bury_d	$1.00 \cdot 10^{-6}$
Ambient environmental temperature (K)	Temp	$2.89 \cdot 10^{+2}$
Surface water current in m/day	current_w	$1.00 \cdot 10^{+5}$
Organic carbon fraction in upper soil zone	foc_s	$2.25 \cdot 10^{-2}$
Organic carbon fraction in vadose zone	foc_v	$5 \cdot 10^{-3}$
Organic carbon fraction in aquifer zone	foc_q	$1.00 \cdot 10^{-2}$
Organic carbon fraction in sediments	foc_d	$3.20 \cdot 10^{-2}$
Boundary layer thickness in air above soil (m)	del_ag	$5.00 \cdot 10^{-3}$
Yearly average wind speed (m/day)	v_w	$2.94 \cdot 10^{+5}$

Table 2A (*continued*)

Designation and unit	CalTox symbol	Used value
Darcy velocity (m/day)	v_darc	$1.00 \cdot 10^{-1}$
Water dispersion coeff. (m^2/day)	D_T	$5.00 \cdot 10^{-2}$
Human Exposure Factors (Residential)		
Body weight (kg)	BW	$6.00 \cdot 10^{+1}$
Surface area (m^2/kg)	SAb	$1.3 \cdot 10^{-3}$
Fluid Intake (L/kg day)	Ifl	$2.20 \cdot 10^{-2}$
Fruit and vegetable intake (kg/kg day)	Ifv	$4.8 \cdot 10^{-3}$
Grain intake (kg/kg day)	Ig	$2.3 \cdot 10^{-3}$
Milk intake (kg/kg day)	Imk	$4.9 \cdot 10^{-3}$
Meat intake (kg/kg day)	Imt	$3.00 \cdot 10^{-3}$
Egg intake (kg/kg day)	Iegg	$3.00 \cdot 10^{-4}$
Fish intake (kg/kg day)	Ifsh	$3.00 \cdot 10^{-4}$
Soil ingestion (kg/kg day)	Isl	$8.06 \cdot 10^{-7}$
Breast milk ingestion by infants (kg/kg day)	Ibm	$1.10 \cdot 10^{-1}$
Inhalation by cattle (m^3/day)	Inc	$1.04 \cdot 10^{+2}$
Inhalation by hens (m^3/day)	Inh	2.20
Ingestion of pasture, dairy cattle (kg[FM]/day)	Ivdc	$1.1 \cdot 10^{+2}$
Ingestion of pasture, beef cattle (kg[FM]/day)	Ivbc	$1.00 \cdot 10^{+2}$
Ingestion of pasture by hens (kg[FM]/day)	Ivh	$1.20 \cdot 10^{-1}$
Ingestion of water by dairy cattle (L/day)	Iwdc	$5.00 \cdot 10^{+1}$
Ingestion of water by beef cattle (L/day)	Iwbc	$5.00 \cdot 10^{+1}$
Ingestion of water by hens (L/day)	Iwh	$8.40 \cdot 10^{-2}$
Ingestion of soil by cattle (kg/day)	Isc	$6.5 \cdot 10^{-1}$
Ingestion of soil by hens (kg/day)	Ish	$1.30 \cdot 10^{-5}$
Fraction of water needs from ground water	fw_gw	0.1
Fraction of water needs from surface water	fw_sw	0.9
Frctn irrgtn wtr contamnnts trnsfrd to soil	f_ir	$2.50 \cdot 10^{-1}$
Frac exposed fruits and veg	fabv_grd_v	$9.00 \cdot 10^{-1}$
Fraction of fruits and vegetables local	flocal_v	$2.80 \cdot 10^{-1}$
Fraction of grains local	flocal_g	$1.00 \cdot 10^{-3}$
Fraction of milk local	flocal_mk	$4.00 \cdot 10^{-2}$
Fraction of meat local	flocal_mt	$1.00 \cdot 10^{-1}$
Fraction of eggs local	flocal_egg	$1.30 \cdot 10^{-1}$
Fraction of fish local	flocal_fsh	$4.00 \cdot 10^{-2}$
Plant–air prttn fctr, particles, m^3/kg[FM]	Kpa_part	$3.30 \cdot 10^{+3}$
Rainsplash (mg/kg[plnt FM])/(mg/kg[soil])	rainsplash	$3.40 \cdot 10^{-3}$
Room ventilation rate, house (m^3/h)	Vrhouse	$7.50 \cdot 10^{+2}$
Indoor dust load (kg/m^3)	dust_in	$3.00 \cdot 10^{-08}$
Exposure frequency to soil on skin (day/year)	Efsl	52
Soil adherence to skin (mg/cm^2)	Slsk	$5.00 \cdot 10^{-1}$

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