The environmental impact of a municipal solid waste incinerator: 15 years of monitoring

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Abstract

A Municipal Solid Waste Incinerator located in Rimini province (Italy) has been monitored for 15 years in order to assess its environmental impact. An integrated environmental monitoring system was designed and implemented over the years. Furthermore, the impact assessment was supported by other tools, such as life cycle analysis (LCA) and risk assessment. In order to fulfil new rules, over the years, the plant underwent several revamping processes. The environmental monitoring was activated in 1997 an involved the analysis of several matrices: soil, atmospheric deposition, vegetation and particulate airborne matter. Based on the obtained results, the monitoring evolved and the sampling sites, analites, matrices and/or sampling techniques were modified. LCA application to the plant was carried out both to investigate the contribution of the incinerator to different environmental categories and to evaluate the effect of the revamping process on plant impacts. In order to assess health effects connected to plant activity, risk assessment applied to air emissions was evaluated for the period 1997–2006. All the study results show that incineration plant emissions do not appreciably affect the contaminant load in the study area. Source apportionment techniques demonstrated that the main sources in the study area are vehicular traffic and regional contribution. LCA indicates quantitatively the lower environmental impact resulting from structural upgrade operations. Risk

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assessment shows that the risk concerning toxic and carcinogenic effects remains largely within the acceptability limits set by USEPA.
Keywords: municipal solid waste incinerator (MSWI), environmental monitoring, life cycle assessment, risk assessment, environmental impact, waste management, dispersion models.

1 Introduction

Today, urban waste management is of great interest in Europe, due to various factors: the high quantities of waste generated every year, the necessity to guarantee sanitary standards for human health, growing concerns about the social acceptability of certain waste treatments, and the high potential for material and energy recovery embodied in waste fractions Passarini *et al.* [1].

Waste incineration is a disposal method increasingly used in Municipal Solid Waste (MSW) management. Since 1995, Eurostat have been collecting European statistics on municipal waste. Since this year, in the Europe-27 incineration process for MSW treatment, grew from 14 to 22% in weight of the total MSW produced, which reached 251.604 tons in 2011 (Eurostat [2]).

Far back, there is awareness on environmental and health impact entailed by waste management. Already in 1975, with the first directive on waste (EU Directive 75/442/CEE [3]), the European Union stated that "*the essential objective of all provisions relating to waste disposal must be the protection of human health and the environment against harmful effects caused by the collection, transport, treatment, storage and tipping of waste*" and "*the recovery of waste should be encouraged*" and that "*provision should be made for a system of permits for undertakings which treat, store or tip waste*". As far as the specific impact of MSW incinerators is concerned, they were ruled since 1984, with the first directive on the combating of air pollution from industrial plants, which encompasses also MSW incinerators (EU Directive 360/1984/EC [4]). With this directive, the principle of the best available technologies for preventing atmospheric pollution was introduced. But the first law limits on the emissions of MSW incinerator and operating conditions indications were promulgated in 1989 (EU Directive 369/1989/EC [5]); (EU Directive 429/1989/EC [6]). Ever since, several directives followed [7–12]; they introduced always more strict rules on operating conditions and on emission limit values. Over the years, incinerator plants underwent thus several revamping processes, in order to fulfill new rules.

In Italy, incineration process is less used than in other European countries, however its use has increased since 1995, from 5 until 16% in 2011 (Eurostat [2]).

In the touristic town of Rimini (Italy), a medium-sized Municipal Solid Waste Incinerator (MSWI) (according to Italian standard) has been operating since 1976. In order to fulfill the new regulations, several revamping processes were carried out. In this work, strategies and tools used to assess its impact from 1997 until 2013, evolved accordingly the plant changing, were presented.

2 Methodological approach

The MSWI studied in this research is nowadays authorized to burn 150,000 tons per year of urban, hospital, and cemetery solid waste. Its environmental impact has been studied since 1997. In this year, important changes to plant were undertaken, i.e. continuous measurement of stack emissions and energy recovery were introduced, in order to fulfill 89/369/CEE and 89/429/CEE directives (Herambiente [13]). The plant characteristics and a summary of the other structural operations carried out in the last years to revamp the plant are reported in Passarini *et al.* [1] and Vassura *et al.* [14]. In order to thoroughly outline the situation on plant environmental impact, several aspects were considered. First of all, the contribution of the incinerator to different environmental categories was investigated. This step was achieved by the application of Life Cycle Assessment (LCA) (Morselli *et al.* [15]). This methodology was applied also to assess the evolution of environmental impacts of the plant, to offer an idea of how structural upgrade measures may affect the total environmental performance (Passarini *et al.* [1]). Secondly, mathematical dispersion models were used to determine the area mainly affected by plant emission fallout. Thanks to pollutant deposition and dispersion maps, it was possible to design a representative environmental monitoring sampling network. Furthermore, dispersion models are also one of the tools required to perform risk assessment (Gunatilaka *et al.* [16]). During the first years ,contaminant distribution was obtained applying the mathematical models EPA models (ISC, RAM, COMPLEX) and a model (DIMULA) by Italian Agency for Energy and Environment (ENEA) (Morselli *et al.* [17]), and therein cited literature. Particularly, Industrial Source Complex version 3 (ISC3) was extensively used in the first stages of the study and for Risk Assessment (USEPA [18]). Since 2005, this model was substituted by Calpuff model for the purposes of environmental monitoring. This last has several advantages compared to ISC3. Particularly, it is a three-dimensional model rather than considering level land and can thus consider the complex land orography. This is a very important characteristic for the application to this case study, since the incinerator is located in a valley (Venturini *et al.* [19]). Moreover, Calpuff is a non-steady-state model. For this reason, it allows to reproduce atmospheric dispersion of pollutants emitted in non-steady and not-homogeneous conditions, such as calm wind condition and recirculation phenomenon (Scire *et al.* [20]).

3 Life cycle assessment

An evaluation of environmental impacts occurring along the whole incinerator life cycle was carried out in Morselli *et al.* [15]: a cradle-to-gate approach was applied to identify and quantify direct and indirect emissions resulting from construction and demolition of the incineration plant, pre-treatments of waste input, combustion process, energy generation, gas cleaning, transport and disposal of residual waste (e.g. bottom and fly ash), and wastewater treatment. Several impact categories were selected to convert emission factors (referring to

the year 2001) to environmental burdens, including air acidification, eutrophication, depletion of non-renewable resources, global warming, aquatic, terrestrial, and sediment ecotoxicity, human toxicity, photochemical oxidant formation, and ozone depletion. Energy recovery implied the larger benefits thanks to electricity and heat recovery, expressed as avoided impacts from conventional sources generation. Excluding those environmental gains, direct emissions from combustion phase determined the major contributions to air acidification and eutrophication, global warming, terrestrial ecotoxicity, and human toxicity. Bottom and fly ash accounted mostly for aquatic and sediment ecotoxicity, whilst pre-treatments and gas cleaning generated the worst scores for photochemical oxidant formation and ozone depletion.

Construction and demolition of the plant had lower impacts, but since structural upgrade measures may lead to the strongest effects in improving incinerator performance, a further study dealt with the evolution of environmental impacts from the plant after revamping and maintenance operations (Passarini *et al.* [1]). In the work, LCA was applied to model and compare six plant configurations, each describing the main structural upgrades interventions occurred over the years 1996–2011. As expected, the results showed improving environmental impacts after the implementation of new procedures and systems for pollutants abatement and control, mainly gas treatment technologies (Figure 1).

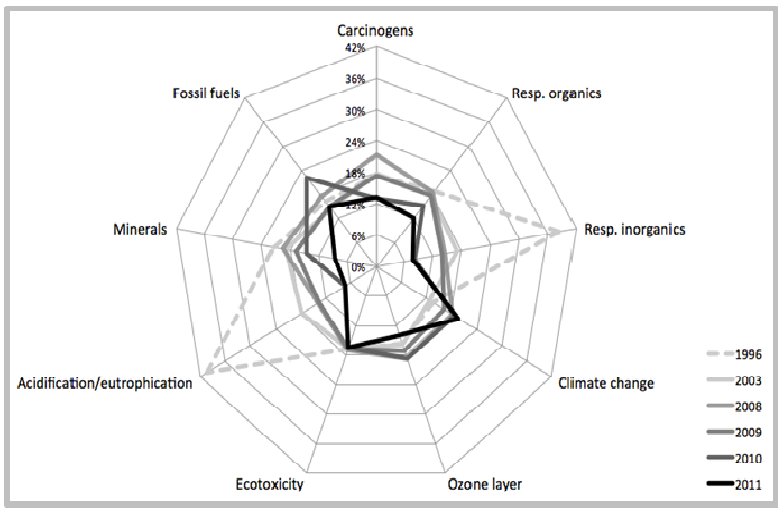

Figure 1: Radar chart showing the percentage contribution to each impact category over the years.

Analysis of process contribution revealed, anyway, the need to enhance the management of bottom ash, for instance to use that solid residue as a filler or similar recycling/reusing options, in order to decrease risks from its current landfilling.

4 Environmental monitoring

Since 1997, the study of the environmental impact of the plant is evaluated through the application of an integrated environmental monitoring system (Morselli *et al*. [15, 17, 21–24]; Vassura *et al.* [14, 25]; Venturini *et al.* [19, 26, 27]).

Over the years, on the basis of the obtained results, the monitoring underwent several revisions. Table 1 summarizes environmental monitoring network development.

Some important steps in monitoring network development are listed below:

In 2000, the sampling network was revised, on the basis of the obtained results and of the application of more precise and specific diffusion models. One sampling site was located in an area of minimum plant emission deposition, in order to have information on the pollution background of the study area. Since during the previous stage of the monitoring polychlorinated dibenzodioxin/polychlorinated dibenzofuran (PCDD/PCDF) and polychlorinated biphenyl (PCB) concentration was always below instrumental LoQ and Polycyclic Aromatic Hydrocarbons (PAH) concentration was low, only heavy metals (HM) were determined in soil, vegetation and moss-bags.

In 2004, the sampling network was further revised. New simulation models, more detailed and updated, were used. Particularly, ISC3 dispersion model was substituted by Calpuff model. The deposition model results can be found in Vassura *et al.* [14]. Beside the new dispersion model, other revisions were applied. Specifically, bulk samplers for the collection of atmospheric deposition, cheaper and more adaptable than wet and dry sampler, were tested and more HM (Al, As, Ca, Cd, Co, Cr, Cu, Fe, Hg, K, Mg, Na, Ni, Sb, Sn, Ti, V, Zn) were determined, in order to identify the best environmental indicators to study purpose. In fact, the results obtained up to now revealed an important contribution of other emission sources. Therefore, the identification of other source indicators could provide useful information on their impact on the study area. Differently to previous years, soil samples were collected annually only at two sites, while vegetation and moss-bag sample analysis stopped. In fact, vegetation sample heterogeneity did not allow to verify analysis reproducibility. Moreover, pollutants were mainly deposited on the surface rather than in the inner part of the sample. For this reason, they were easily removed by washing procedure.

In 2005, wet and dry samplers were substituted by bulk samplers. Comparison between these kind of samplers revealed consistent results; furthermore, bulk sampler error was lower than wet and dry. PCDD/PCDF are highly characteristic of incineration plant. Therefore, organic compounds determination in soil samples was reintroduced. They were determined at the Interuniversity Consortium ‘‘Chemistry for the Environment’’ (INCA), a laboratory equipped with more sensitive analytical techniques, i.e. high resolution capillary column gas chromatography (HRGC)/high resolution mass spectrometry (HRMS), which allowed to reduce the LoQ compared to 1997–1999 analysis. On the basis of the results obtained up to 2008, since 2009

Table 1: Environmental monitoring networks.

	Matrices	Sampling sites	Determined analytes	Sampling frequency	Period
1997-1999	Soil	15 (empirical choice)[a]	Al, Cd, Cr, Pb, Ni, Hg, Cu, Zn, Mn; PCDD/PCDF, PCB	Annually	1997-1999
	Vegetation (seasonal and evergreen)	12 (empirical choice)[a]	Al, Cd, Cr, Pb, Ni, Hg, Cu, Zn, Mn; PAH	Six-monthly	1997-1999
	Atmospheric deposition (wet and dry sampler)	3 (based on ISC3 model results)[b]	Al, Cd, Cr, Pb, Ni, Hg, Cu, Zn, Mn	Monthly	1999
2000-2003	Soil	6 (based on ISC3 model results)[c]	Al, Cd, Cr, Pb, Ni, Hg, Cu, Zn, Mn	Annually	2000-2003
	Vegetation (seasonal and evergreen)	Same as soil	Same as soil	Six-monthly	2000
	Moss-bags	Same as soil	Same as soil	Monthly	2002-2003
	Atmospheric deposition (wet and dry sampler)	3 (based on ISC3 model results)[c]	Al, Cd, Cr, Pb, Ni, Hg, Cu, Zn, Mn; PCDD/F, PCB, HCB, PAH	Twice a year (organic contaminants); monthly (inorganic contaminants)	2000-2001
2005-2010	Soil	4 (based on Calpuff model results)[d]	Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn; PCDD/F, PCB, HCB, PAH	Annually	2005-2008 inorganic contamin
	Atmospheric deposition (bulk sampler)	Same as soil [d]	Al, As, Cd, Cr, Cu, Fe, Mn, Ni, Pb, Zn; Ca^{2+}, K^{+}, Mg^{2+}, Na^{+}, NH_4^{+}, Br^{-}, Cl^{-}, F^{-}, NO_2^{-}, NO_3^{-}, SO_4^{2-}; pH, conductivity; PCDD/F, PCB, HCB, PAH	Same as 2000-2003	2005-2010; site 5 from 2009
2011-2013	Soil	5 (based on Calpuff model results) (Figure 3)	PCDD/F, PCB, HCB and PAH	Annually	2011-2013
	Atmospheric deposition (bulk sampler)	Same as soil	Same as 2005-2010	Same as 2005-2010	2011-2013;

[a] In Morselli *et al.* [22], [b] in Morselli *et al.* [17], [c] in Morselli *et al.* [21] [d] in Venturini *et al.* [19].

HM determination in soil was stopped. In fact, a significant accumulation of these contaminants in the superficial A-horizons is not observed. In Table 2 the time span required to observe a significant accumulation of HM in superficial soils (15 cm), by considering mean soil concentration and deposition fluxes, is reported. These results indicate that it will not be possible to appreciate, from a year to another, significant increases in soil contamination for HM. Since 2009, a 5th site, located in the city center of the town of Rimini, was added.

Table 2: Atmospheric flux deposition (mg/m2day), concentration (mg/kg ds) and accumulation time span (years) in superficial soil for the main HM.

	Soil (mg/kg ds)	**Deposition (ug/m^{2}day)**	**Time span to double the present value (years)**	**Time span to increase of 10% (years)**
As	5	8	340	30
Cd	1	1	590	60
Cr	66	4	8200	820
Cu	26	14	1000	100
Mn	560	20	15000	1500
Ni	44	4	6400	640
Pb	47	27	950	100
Zn	77	64	640	60

From 2011 to 2013, the sampling network was further modified. Two of the five sites were changed. Specifically site 4, located in an area of minimum plant emission deposition, was moved from a suburban area to a rural background area. The deposition maps considered up to now refer to dry deposition. Wet deposition maps are quite different since a concentration gradient with a maximum centered in the plant can be observed (Figure 2). For this reason, a sampling site (1) was added between site 2 and the incinerator (Figure 3), in order to assess how the distance from the plant affects atmospheric deposition.
In addition to the above mentioned monitoring network, since 2006, atmospheric particulate matter (PM) was also collected, mainly at site 3 (Figure 3). The analysis of this matrix was introduced because it provides data concerning shorter time span (1 day); therefore, it is possible to gain short-term variation on atmospheric pollution. Several analytes (HM, PAH, OC/EC, soluble ions) were studied. During some sampling campaigns, samplers were coupled with a wind sensor, which allows the turning on and off of the instrument depending on wind direction and speed. With this approach, it was possible to compare the contaminant burden of PM samples (PM_{10} or $PM_{2.5}$) collected downwind or upwind of the incinerator (Vassura *et al.* [25]; Venturini *et al.* [26, 27]).

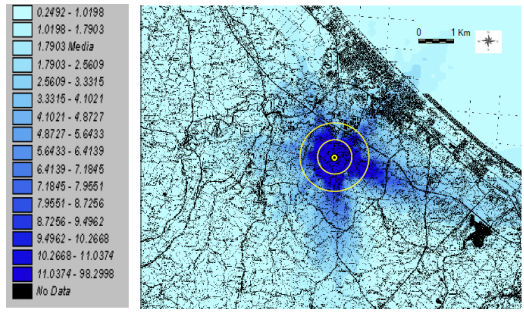

Figure 2: Wet deposition map (ISC3 model).

Figure 3: Studied area and monitoring sites in 2011-2013 (from © 2014 Google Images © 2014 DigitalGlobe-modified).

To sum up, considering the total sampling period, it was concluded that pollutants determined in all the analyzed environmental matrices are evenly distributed in the study area. Nevertheless, temporal variability is quite high. For all the determined components, the pollutant load in the study area is low, similar to what is found in other suburban and rural area. The only appreciable differences refer to city centre site, which shows higher deposition fluxes for several HM and organic compounds and an higher PAHs concentration in soil samples. When the plant was shut in 2008 for a revamping process, atmospheric deposition fluxes were not significantly lower than in other years. Moreover, PCB deposition flows (Figure 4) show an higher concentration in 2008 than in previous and following years. In order to quantitatively assess the contribution of the incinerator to local pollution and to yield information about the other sources affecting the area, Positive Matrix Factorization (PMF) analysis, a new approach in factor analysis, was applied (Venturini *et al.* [19, 27]). These studies confirm that incinerator contribution to the total pollutant load is negligible and that the main source in the study area is vehicular traffic and regional contribution.

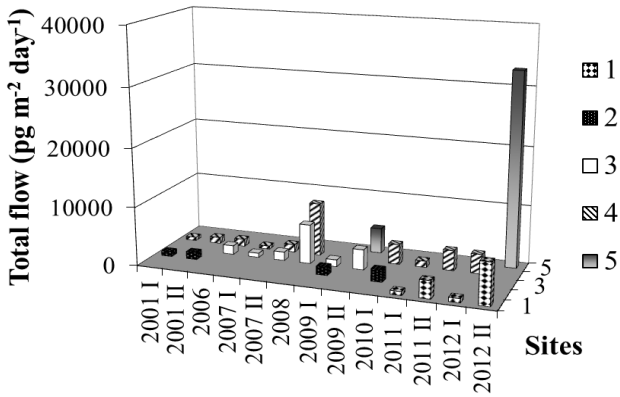

Figure 4: Total PCB flows during the sampling.

5 Risk assessment

In 2009–2010 a risk assessment was performed on an area of about 11km^2 centered on the plant, in order to evaluate the impact on human health due to HM and organic pollutants emitted by the incinerator (Morselli *et al.* [28]). In particular both categories of children and adults were considered as sensitive receptors and the following pollutants, showing toxic and/or carcinogenic activities, were taken into account: As, Cd, Cr, Ni, Pb, benzo[a]pyrene and PCDD/PCDF (toxic and carcinogenic effects) Hg, Mn and Zn (toxic effects).

Considering the characteristics of both the area and receptors, beside the air inhalation direct exposure pathway, three indirect pathways were studied: absorption through the food chain, skin absorption via dermal contact and accidental soil ingestion. Pollutant transport and dispersion were evaluated by the Gaussian ISC3 calculation code USEPA [18] and the assessment was performed by applying the short-term ISC3model (ISCST3). In order to give a conservative estimate of risk, contaminant concentrations in ambient air and depositions were predicted by considering, for each pollutant, the highest flow measured from 1997 to 2007 and assuming all the chemical is in its most toxic form.

In general, as regards the individual hazard associated with the exposure to toxic chemicals, the risk (expressed as Hazard Quotient Risk) has always been found at least 3 orders of magnitude lower than the reference value (1; USEPA [29]). In particular, child receptors result significantly affected by indirect exposure pathways, while adult receptors are mainly interested by direct pathways (Figure 5a). For carcinogenic chemicals, the estimated individual risk (expressed as Individual Cancer Risk) is, on average, more than 2 orders of magnitude lower than the acceptability limit (10^{-6}; USEPA [29]), with a predominant contribution of direct exposure pathway (Figure 5b). The risk remains within the acceptable limits (10^{-5}; ISPRA [30]), even if the combination of all the studied pollutants is taken into account.

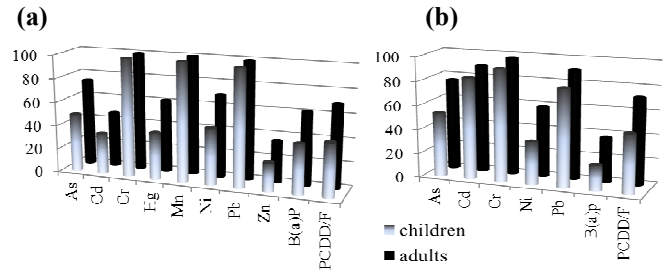

Figure 5: Percent contribution of direct exposure pathways to the total risk for children and adults, due to toxic (a) and carcinogenic (b) effects.

6 Conclusion

The assessment of the environmental impact of an industrial plant should consider several tools that provide complementary data. In this work, an example of how LCA, environmental monitoring and risk analysis can together provide a complete overview of the environmental impact of an incinerator is reported.

Impact category indicators and the evaluation of emissions from the incinerator carried out through an LCA approach, allowed a comprehensive analysis of the overall plant's performance on the environment, and lead to identify priorities to improve combustion efficiency and pollutants reduction.

As far as environmental monitoring is concerned, all the analyzed environmental matrices do not show significant differences between the sites located in the area affected by plant emission fallout and the control site. Incineration plant emissions do not entail changes in atmospheric concentrations that appreciably affect the contaminant load in the studied environmental matrices. When the plant was shut down, atmospheric deposition fluxes were not significantly lower than in other years. The incinerator's relative contribution to the total pollutant load seems to be negligible compared to a higher background concentration, which could be ascribed to the nearby urban area.

As regards Risk Analysis, even considering the worst but realistic conditions, the risk associated with toxic and carcinogenic effects is widely within the limits, both for each single pollutant and for the sum of their potential effects.

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