

ULTRAFINE PARTICLES FROM WTE AND OTHER COMBUSTION SOURCES

Lital Yinon¹ and Nickolas J. Themelis², and V. Faye McNeill³

and 1. Earth Environmental Engineering; 2. Earth Engineering Center, 3. Chemical Engineering, Columbia University

Corresponding author: N.J. Themelis (njt1@columbia.edu)

ABSTRACT

The size of combustion generated particles ranges from a few nanometers up to 1 micron, whereas the size of naturally occurring PM such as pollens, plant fragments, and sea salt is generally larger than 1 micron. Particles generated by photochemical processes in the atmosphere are generally smaller than 1 micron. Ultrafine particles (UFP), also called "nanoparticles", are <0.1 micron and in recent yearshave attracted attention due to potential adverse health effects associated with them. The contribution of UFP to the total PM mass is very small. However, they dominate the total number of particles in urban aerosols. Their sources are both mobile and stationary combustion sources and also gas-to-particle conversions.

In coal and waste combustion systems, UFP are hypothesized to be generated mainly by nucleation of metal vapors. Coal naturally contains a vast range of inorganic elements among which are heavy metals. Sources of heavy metals in MSW include batteries, electronic devices, light bulbs, house dust and paint chips, food containers, used motor oils, plastics, yard wastes and some papers. The input of these metals into WTE facilities can be controlled by better source-separation of metal-containing materials. In 2007 almost 50% of the approximately 4.16 billion MWh generated in the United States was produced by coal power plants whereas only 0.3% was generated by the WTE industry. A preliminary study has shown that in terms of contribution to UHF in the

atmosphere, MSW combustion has a minor effect in comparison to coal-fired power plants in the U.S. This paper will report on the results of this investigation

Introduction

Particulate matter (PM) refers to solid or liquid particles suspended in the air. The atmosphere contains considerable concentrations of PM, particles that range from a few nanometers up to 100 microns in diameter. The sources of PM are both anthropogenic and natural. PM can be emitted either directly from industrial activities, transportation, power plants and natural sources, or indirectly through conversion of gaseous precursors such as ammonia and sulfur oxides. PM is categorized according to its aerodynamic diameter: PM₁₀, also called "coarse PM", refers to articles with diameter of 10 μm or less, PM_{2.5}, also called "fine PM", refers to particles with diameter of 2.5 μm or less, and PM_{0.1}, also called "ultrafine particles" (UFP), have diameters of 0.1 μm or less.

The size of particles generated by combustion ranges from a few nanometers up to 1 μm, whereas the size of naturally occurring PM, such as pollens, plant fragments, and sea salt is generally larger than 1 μm. Particles generated by photochemical processes in the atmosphere are generally smaller than 1 μm (Seinfeld and Pandis). The number concentration of atmospheric PM varies spatially and temporally, with typical number concentrations being on the order of thousands of particles per

cm³ (Costabile et al, 2009).

In recent years, increasing attention has been given to ultrafine particles (UFP) due to the potential adverse health effects associated with them. UFP can stay suspended in air for several days and be transported over thousands of kilometers (Hinds, 1999). Comparing with larger particles, UFP have a much larger surface area to volume ratio and can carry large amounts of adsorbed pollutants (Oberdorster, 2001). UFP are ubiquitous in the atmosphere and dominate the total number of particles in urban PM (Lightly et al, 2000; Seinfeld and Pandis). However, their contribution to the total mass of PM in air is very small. For example, the mass of a 10 μm particle is equivalent to the mass of one billion 10-nm particles.

UFP are formed by combustion processes and also through gas-to-particle conversion. In the urban environment, a major fraction of UFP is related to road traffic and emissions from stationary combustion sources (Young and Keeler, 2004; Wehner et al, 2008). Heavy duty diesel vehicles are in fact one of the greatest contributors to urban UFP (EPA), emitting up to two orders of magnitude more UFP, as compared to gasoline trucks (Whener et al, 2009), with a typical UFP emission rate of 10¹⁵ particles per kilometer of travel (Morawska et al, 2008). Legislation regarding emissions and ambient concentrations of PM is based on particle mass concentration, to which UFP are a minor contribution.

3. UFP in Coal, Biomass, and Waste Combustion Systems

In coal, biomass and waste combustion systems, ultrafine particles are assumed to form mainly by homogeneous nucleation of metal vapors (in either chloride or oxide form) followed by growth through coagulation and condensation of other vaporized materials (Chang et al, 2000; Carbone et al, 2008; Linak et al, 2007). Nucleation occurs in a supersaturated vapor phase. The saturation ratio of a solute A in air at a given temperature, is defined as:

$$S = \frac{P_A}{P_A^*}$$

where P_A is the partial pressure of A and P_A^{*} the saturation vapor pressure. S= 1 for saturated vapor, S<1 for sub-saturated vapor, and S>1 for supersaturated vapor. At saturation, there is no

nucleation of a stable phase of A from the gas phase; for nucleation to occur S has to be greater than 1. The aerodynamic radius of the smallest stable homogeneous nucleated particle is defined as the “critical size”. Below the critical size particles will spontaneously decompose and return to the vapor phase. According to the classical nucleation theory (Seinfeld and Pandis), the free energy of formation of the critical size cluster is expressed by:

$$\Delta G^* = \frac{16\pi v_1^2 \sigma^3}{3(kT \ln S)^2}$$

where v₁ is the volume of the condensable molecule in the condensed phase, σ the surface tension, k the Boltzmann constant and T the temperature. The critical size, r*, is derived from Kelvin’s equation:

$$r^* = \frac{2\sigma v_1}{kT \ln S}$$

Increasing the saturation ratio, S, decreases both the free energy barrier and the critical radius. Since σ, the surface tension of the metal in the liquid state (Table 1) is very high for metals and considering the cubic term in the free energy equation, a very high saturation ratio is required in order to overcome the free energy barrier. For these extreme supersaturation ratios, r* has to be very small. Supersaturation conditions are reached when the vapor is cooled or/and when the partial pressure of the metal vapor increases beyond saturation. In general, homogeneous nucleation of metal vapor is enhanced by high cooling rates and low system pressure (Singh et al, 2002).

Table 1: Surface tension values of selected heavy metals in liquid state (Gale and Totemeier, 2004)

Metal T	emperature (K)	Surface Tension (milliNewton/meter)
Ag	1233.7	966
Al	933	914
Cd	594	570
Cr	2148	1700
Cu	1356	1303
Fe	1809	1872
Hg	234.13	498
Mn	1514	1090
Ni	1727	1778
Pb	600	458
Zn	692	782

The classical nucleation theory has been criticized as being inadequate to predict the critical particle size because it assigns surface tension bulk properties to clusters of particles (Lightly et al, 2000; Martens et al, 1987). However, more recent models and experimental work have shown that homogeneous nucleation of metal vapor does result in stable particles with diameter of less than 10 nm (Bahadur and McClurg, 2004; Lummen and Kraska, 2005; Carbone et al, 2008). Finney and Fink (2008) hypothesized that for transition metals, the critical size of the nucleus depends on the strength of the metal-metal bonds. Since the heat of vaporization of a metal, ΔH_{vap} (Table 2), can be used as a measure of the metal-metal bond strength and since the critical size strongly influences the final particle size, there should be a correlation between ΔH_{vap} of a metal and the size of the metal nanoparticle. More specifically, for a metal with high value of ΔH_{vap} , nucleation should be favored over growth, thus resulting in smaller particles. This hypothesis is consistent with literature data regarding the preparation of transition metals nanoparticles (Finney and Fink, 2008; Wegner et al, 2001; Li and Truhlar, 2008; Bondi et al, 2009; Schladt et al, 2009; Narayana and El-Sayed, 2005).

Table 2: ΔH_{vap} values of selected heavy metals at boiling point. (Gale and Totemeier, 2004)

Metal	ΔH_{vap} (kJ/mol)
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Ag	257.8
Al	290.9
Cd	99.6
Cr	342.1
Cu	304.8
Fe	340.4
Hg	61.1
Mn	231.1
Ni	374.3
Pb	178.8
Zn	114.3

Following combustion, as the gaseous products travel through the flue gas system, the heavy metals are subject to homogeneous nucleation and condensation onto fly ash particles (Hester and Harrison, 2002); these reactions are followed by surface growth.

The final particle size depends on the residence time of the flue gas in the combustion system. Longer residence times provide more opportunity for particles to coagulate and grow (Liank and Wendt, 1993).

UFP emission from coal power plants

Combustion of coal in thermoelectric power plants is a major source for direct emission of UFP. In 2007, almost 50% of the approximately 4.16 billion MWh generated in the United States, corresponding to 1.047 billion tons of coal, were produced by coal-fired power plants (DOE).

Coal naturally contains metals in various concentrations. Table 3 shows the concentration ranges of some of the trace elements found in coal. Li et al (2009) investigated the particle size distribution produced by burning coal in a pilot scale pulverized coal combustor, where a single-wire tubular electrostatic precipitator (ESP) was used to collect fly ash particles. The particle size distribution at the ESP inlet showed a high number concentration of particles with the mode around 75nm and number concentration of about 10^8 particles/cm³. At the ESP outlet, the number concentration of particles larger than 70 nm decreased about three orders of magnitude. However, the reduction in the number concentration of particles smaller than 70 nm decreased significantly at particle sizes below 70 nm. Wang et al (2008) observed a number concentration of 6×10^8 particles/cm³ for particles

in the size range of 5.6-560nm in a coal-fired power plant in China.

Table 3: Concentrations of some trace elements in coal (Ruth, 1998)

Element	Concentration (ppm)	range
Zn	10-400	
Pb	5-55	
Cu	8-30	
Ni	3-30	
Cd	<0.1-7	
Hg	0.1-0.2	

Analysis of fly-ash samples from coal burned in laboratory combustors showed that the chemical composition of the ultrafine fraction is significantly different than the composition of the fine and coarse fractions. It was observed that the ultrafine fraction contained higher concentrations of trace elements such as Cu, Pb, Cr, Mn, Zn, As, and Se. In some cases, the metal concentration in the UFP was 50 times higher than in the fine and coarse particles (Seames and Wendt, 2000; sui et al, 2007; Wang et al, 2008). The enrichment of metals in UFP has also been observed in stack gas samples taken from coal-fired power plants (Yoo et al, 2005). These observations are consistent with the hypothesis that metal vapors nucleate to form UFP. Table 4 summarizes findings from various research studies regarding metal enrichment in UFP from coal combustion.

Table 4: Enrichment of metals in UFP-summary of various studies

Study UFP	Enriched
Seames and Wendt, 2000	As, Cd, Se

Yoo et al, 2005	Cd, Cr, Mg, Mn, Ni, Pb
Sui et al, 2007	Ca, Cr, Cu, Fe, Mg, Mn, Pb, Zn
Wang et al, 2008	As, Cd, S, Sb, Se, Pb, and to a lesser extent: Zn, Cr, Ni, Cu, V, Co, Cr, Ni, Mg, Mn

UFP emission from biomass-fired power plants

Biomass combustion is used extensively to generate heat and power from organic-matter derived fuels, such as agricultural and wood residues (Wiinikka et al, 2006). The energy from biomass combustion used for heat and electricity generation and for transportation (biofuels) accounts for approximately 1.4% of the energy use worldwide (Goldemberg and Johansson, 2004). Utilization of biomass within the European Union has increased significantly over the last decades and has almost tripled in comparison to its use in 1999 (Khan et al, 2008). In 2007, 1.3% of the electric power in the United States was generated by biomass combustion (DOE).

Biomass contains various trace elements such as Zn, Cd, Cu, Cr, Pb, and Hg, which vary considerably, depending on the biomass type and source (Demibras, 2005). Biomass combustion leads to relatively high PM emissions and the particle size distribution varies for different fuels and combustor systems (Nussbaumer, 2003). Combustion of wood in different laboratory scale combustion systems showed that the majority of the particles emitted were in the size range of 30 to 300 nm with a mode of 80 nm (Wieser and Gaegauf, 2000). The particle number concentration varied widely from 10^7 - 10^{10} particles/cm³. Chemical analysis of particles emitted from wood combustion showed that metals such as Zn and Mg were concentrated in the UFP (Johansson et al, 2003; Tissari et al, 2008). The number concentration of particles in the size range of 17 to 300 nm emitted from boilers operating on three different biofuels (sawdust, wood pallets, and forest residues) ranged from 6.3×10^7 to 7.7×10^7 particles/cm³ with a particle size mode ranging between approximately 70-100 nm (Weirzbicka et al, 2005). The concentration of Zn, Cd, Pb and Cr in the particles depended on the type of fuel, but was generally significantly higher in the smaller

particles.

UFP emission from WTE plants

Incineration of waste is used throughout the world for treatment and disposal of MSW. In the U.S. there are 87 WTE facilities that process a total of approximately 29 million U.S. tons of municipal solid waste (MSW) (Themelis, 2005). A typical calorific value for MSW is about 9000 kJ/kg (Hester and Harrison, 2002). In 2007, WTE accounted for 0.3% of the electric power produced in the United States (DOE).

MSW contains variable amounts of metals. Sources of heavy metals in MSW include batteries, electronic devices, light bulbs, house dust and paint chips, food containers, used motor oils, plastics, yard wastes and some papers (Hasselriis and Licata, 1995).

In a WTE facility in Germany, Ultrafine particles were detected in the flue gas at various gas temperatures. Samples were collected from the flue gas at 700°C (prior to boiler), 300°C (after the boiler), and from the stack at 80°C (Maguhn et al, 2003). At the 700°C sampling point, a bimodal size distribution was observed with the one maximum at about 80 nm and the other at 40 nm. At the 300°C sampling point, the observed size distribution showed a shift of the main maximum to a larger particle size of about 140 nm, probably due to coagulation of particles upon travel through the boiler section. The particle size distribution of the stack samples showed a significant reduction in the total number of particles in comparison with the 300°C, due to particulate control (ESP and fabric filters). The measured particle size distribution ranged from approximately 1 µm to 17 nm with a broad maximum at about 40-70 nm and a UFP number concentration of about 10^5 particles/cm³.

In a very recent study, the particle size distribution observed in the stack flue gas of a WTE plant in Italy (Buonanno et al, 2009) was somewhat similar to that observed by Maguhn et al, with a peak at around 80nm. The number concentration varied among the different samples, from 2×10^5 to 1×10^5 particles/cm³, and 65% of the total number of particles were in the ultrafine range. The flue gas cleaning methods used in this facility were selective non catalytic reduction of NO_x (SNCR), ESP, spray absorber system and fabric filters.

UFP emissions from WTE are challenging to

characterize due to the variability of MSW fed to the combustion unit. Cernuschi et al (2009) observed a significant variability in the mean number concentration of UFP in the stack gas of three Italian WTE plants ranging from 4×10^3 to 7×10^4 particles/cm³.

Chemical composition of WTE UFP

In the above studies, information about the waste composition, particularly the metal content, was not provided and the UFP were not analyzed for their chemical composition. However, since UFP generated from both coal and biomass combustion were found to be enriched in metals, it is expected to find the same trend in UFP generated by MSW combustion. In a laboratory experiment, Tsukada et al (2008) showed that the particle size distribution in a modeled WTE exhaust gas, generated by sublimation of PbCl₂ and CdCl₂, ranged from 3 nm to 300 nm and most of the particles were of diameter less than 100 nm. This finding is in agreement with the concept of UFP formation from metal vapor (in this case in chloride form) generated by waste combustion.

Comparison of UFP number concentration in gas from combustion of different fuels

Table 5 provides a comparison of the number of ultrafine (or near ultrafine) particles emitted by combustion of several types of fuels as observed in different research studies. Of course, the number concentration of the particles measured in the stack is likely to be affected by the air pollution control methods used. It is therefore difficult to relate the observed number concentration solely to the fuel composition and combustion process. However, in a laboratory-scale test of fluidized bed combustion of different solid fuels, the number concentration of UFP generated by the combustion of refuse derived fuel (RDF; shredded and pre-sorted MSW) was about half than from biomass and about 14 times lower than from bituminous coal combustion (Urciuolo et al, 2008). This observation is interesting since refuse derived fuel should contain a smaller amount of metals than as-received MSW.

Table 5: UFP emission from combustion of different fuels*

<i>Paper (System)</i>	<i>Fuel used</i>	<i>Air pollution control (APC) system</i>	<i>UFP number concentration in stack gas (particles/cm³)</i>
Wang et al, 2008 (Industrial power plant)	Coal	ESP	6×10^8 ($5.6 < d_p < 560$ nm)
Wierzbicka et al, 2005 (district heating units)	Sawdust, Pellets, Forest residues	Multicyclones	Sawdust: 6.7×10^7 Pellets: 6.3×10^7 Forest residues: 7.7×10^7 ($17 < d_p < 300$ nm)
Urciuolo et al, 2008 (Experimental fluid bed reactor)	Bituminous coal, Biomass (pine seed shells), Commercial pre-dried granulated sludge (GC), Refuse derived fuel (RDF)	No Air Pollution Control system	Coal: 1.1×10^7 Biomass: 1.49×10^6 GC: 1.06×10^7 RDF: 7.52×10^5
Buonanno et al, 2009 (Industrial WTE facility)	As-received MSW	ESP, spray absorber, fabric filter, selective non-catalytic reduction	$\sim 1 \times 10^5$
Cernuschi et al, 2009 (3 industrial WTE facilities)	As-received MSW	Plant 1: ESP, dry absorption system, fabric filter, selective non-catalytic reduction. Plant 2: dry absorption system, fabric filter, selective non-catalytic reduction. Plant 3: dry absorption system, fabric filter, selective non-catalytic reduction, wet absorption system, quencher.	Plant 1: 3×10^3 - 1.7×10^4 Plant 2: 4×10^3 - 7×10^3 Plant 3: 2.45×10^4 - 7×10^4

* Data are not normalized

Comparison of UFP emissions from coal-fired and WTE plants in the U.S.

As shown in Table 5, on the basis of studies carried out on different fuels the overall number concentration of UFP emitted from coal-fired power plants in the U.S. is at least one order of magnitude higher than that observed in WTE power plants. Based on the quantities of fuel combusted annually in the U.S. and the corresponding volume of stack gas generated and assuming that the average UFP number concentration in coal-fired power plants is 10^7 particles/cm³ and in WTE facilities 10^6

particles/cm³, it is possible to compare the respective contribution of these two types of power plants to UFP emission in the United States (Table 6). However, it should be noted that the number concentration of UFP emission from WTE plants may be considerably lower, as demonstrated by Cernuschi et al (2009).

Table 6: Comparison of total UFP emitted from coal-fired and WTE power plants in the United States assuming comparable number concentration of UFP in the stack gas. (Data for biomass-fired power plants were not available).

Fuel U.S.	tons/year	Stack gas per fuel combusted (dscm/ton) ¹	Total stack gas (dscm/year)	Total UFP (particles/year)
Coal	1.05×10^9	10000	1.05×10^{13}	1.05×10^{26}
MSW	2.90×10^7	5000	1.45×10^{11}	1.45×10^{23}

¹Themelis, 2005

It can be seen that the volume of gas emitted from U.S. coal-fired power plants is so much greater than from WTEs that even if the same number concentration is assumed to exist in these two types of plants, the amount of UFP emissions from coal combustion is at least fifty times greater than that from WTE facilities.

Factors Affecting Metal Partitioning in WTE

Since ultrafine particles are associated with metals, it is first crucial to review the factors influencing the partitioning of metals between bottom ash, fly ash and flue gas. This section treats metal partitioning in WTE alone since unlike in coal and biomass fired power plants the fuel properties can be controlled to some degree. The factors that affect metal vaporization can be divided into three main categories: metal properties, waste properties, and combustion conditions.

Metal properties

Heavy metal partitioning in waste combustion systems is related to the physiochemical properties of the metal. For

example, volatile heavy metals such as Cd and Hg, are likely to be found in the flue gas while other metals with low vapor pressure and high boiling points such as Fe and Cu are usually trapped in the bottom ash (Hester and Harrison, 2002). Toledo et al (2005) found that among the tested heavy metals, Ni and Cd exhibited the most extreme behavior. Ni showed the lowest and Cd the greatest partitioning into the gas phase; this can be explained by the high and relatively lower boiling points of these two metals.

Waste properties

Metal partitioning in MSW is also affected by other factors, namely chlorine and alkaline content as well as moisture in the waste.

Cl content

Heavy metals are known to react with chlorine during the incineration process (Corella et al, 2000). The presence of chlorine compounds in the waste enhances formation of metal chlorides, which are more volatile than the elemental metals (Pavageau et al, 2004, Poole et al, 2007). Increased chlorine content in the waste is predicted to favor volatilization of Co, Cu, Fe, Mn, Ni, Pb, and Zn (Poole et al, 2008).

Controlled lab experiments showed that addition of chlorine significantly increased the transfer of different heavy metals into the gas phase and reduced the temperature at which the transfer occurred (Belevi and Langmeier (2), 2000). For example, at 700°C 70% of Zn was transferred to the vapor phase. Addition of chlorine increased the transfer of Zn into the vapor phase to 94%. The gaseous chlorides of Cd and Pb were found to be the dominant species for these metals in the temperature range of ~600°K to 1400°K. Above this temperature the dominant form is elemental gas for Cd and oxide for Pb.

The gaseous chloride of Hg was found to be the major Hg species below 900°K, while above this temperature elemental gaseous Hg dominated (Abanades et al, 2002). This result is in agreement with other research studies that reported that at high temperatures increased chlorine content seems to have a negligible effect on Cd and Hg, both of which are observed to exist mostly in the elemental gaseous form (Durlak et al, 1997). The effect of chlorine is of course dependent on its availability in the waste. PVC and C₂Cl₂ compounds are effective sources of chlorine ions as they are readily decompose (Wang et al, 1998). The weight percent of Cl in MSW ranges between 0.2-0.6 while in coal it ranges between 0.005-0.6.

Moisture

The effect of waste moisture is interesting. On one hand, increased waste moisture results in increased HCl production, which in turn reduces the amount of available Cl that can react with metals (Durlak et al, 1997). On the other hand, waste moisture tends to reduce the flame temperature due to the latent heat of evaporation resulting in a less efficient process (Yang et al, 2004) and in conditions that are less favorable for HCl formation. It has been observed that when the temperature is kept constant, increasing waste moisture shifts lead from PbCl₂(g) to PbO(l) due to reduction in available chlorine. When temperature is allowed to change, increasing moisture content results in lower temperature, which in turn shifts the lead from the oxide form to the gaseous chloride form (Durlak et al, 1997). Similar results were observed in a different experiment by Wang et al (1998): They observed that heavy metals tend to form metal chlorides in the fly ash in the presence of chlorine; and to form metal oxides in the bottom ash in the absence of chlorine. The effect of moisture should be more pronounced in

MSW combustion than in coal combustion since the weight percent of moisture in MSW is 25 comparing to 5 in coal (Ruth, 1998).

Alkaline content

The effect of alkaline metals in the waste (mainly Na and K) somewhat resembles the effect of moisture. In both cases, speciation is governed by the affinity of hydrogen, in the case of moisture, or alkaline metal to chlorine. Na and K have a stronger affinity to chlorine than heavy metals (Wang et al, 1998). This again results in less available chlorine and a consequent reduction in heavy metal volatilization. Durlak et al (1997) reported that increased Na content in the waste significantly reduced the percentage of lead in the fly ash and increased the percentage of lead in the bottom ash.

Lee and Wu (2005) developed a model that predicts higher inter-coagulation rates with the following conditions: high number concentration of coarse mode, a large mean size difference between fine and coarse modes, and a narrow geometric standard deviation of the fine mode. They argue that for practical application of the model, monodisperse sorbent particles with mean diameter close to 1µm and a number concentration of more than 10⁷/cm³, would be the most effective. It is very likely that the activated carbon particles injected in WTE plants to capture metal and organic molecules also reduce the number concentration of emitted UFP by scavenging some of the particles. However, more research is required in order to determine the effect of activated carbon particles in WTE on nuclei scavenging.

Conclusions

Research studies have demonstrated that exposure to UFP can cause adverse health effects, some of which are severe. Power plants are a significant anthropogenic source of UFP and current particulate control methods are not very efficient in controlling these emissions. In the United States, the overall contribution of WTE and biomass to UFP emission is very small relative to coal power plants, commensurate with the lower scale of energy production from MSW and biomass. Assuming 10⁶ and 10⁷ particles/cm³ of stack gas generated by WTE and coal-fired power plants respectively, it was estimated that the total number of UFP emitted annually by coal-fired power plants is nearly three orders of magnitude higher than the emissions of WTE facilities. Although UFP emissions from WTE

plants are not well characterized as yet, the few research studies conducted show a significant variability ranging from a few thousands to about 10^6 particles/cm³. Thus, the overall actual contribution of WTE to UFP emission might be even lower than calculated based on the above assumptions.

Waste combustion is an environmentally conscious process when it is done properly. Countries such as Switzerland, Norway and Sweden, that were ranked as the top three nations in the Environmental Performance Index (2008), combust almost all of their non-recyclable waste. These countries have rigorous recycling policies, which play an important role in the environmental impacts of waste combustion. The primary responsibility in separating metals from waste should be on the consumers. Thus, public awareness and legislation in the United States are necessary. Recovery of metals at the source, such as small batteries, fluorescent lights, etc. should become customary. Improvement in the particulate control methods, to address UFP emissions, is needed in all stationary combustion sources. A few possibilities exist but more research in that area is required.

Suggestions for future research

This study indicated that further research is needed in order to better characterize UFP emission from coal-fired, biomass-fired and WTE power plants, both in terms of chemical composition and number concentration. The initial plan in this thesis was to also include field tests on the analysis of samples of stack gas from a coal-fired and a WTE power plant for total number concentration and elemental composition. Although this part was not carried out eventually, we have designed an experimental setup that may be implemented in the future, as described below.

The collection and separation of particles into size bins is to be done using an impactor. The most widely used instruments for this purpose are: Microanalysis Particle Sampler (MPS), Micro-Orifice Uniform-Deposit Impactor for nanoparticles (nano-MOUDI), and Dekati Low Pressure Impactor (DLPI). Both MPS and nano-MOUDI separate particles into three size bins and gravimetric analysis of the samples would give a rough size distribution. DLPI has 13 stages and thus would be a better fit for isolating UFP. The particle size distribution can be done using Scanning Mobility Particle Sizer (SMPS). We have not decided whether or not to

include sample dilution, which is used to simulate the evolution of the particle population after emission from the stack. The departmental laboratories do not have the necessary equipment for dilution. However, it can be purchased from the DLPI manufacturer or borrowed from our collaborators at Clarkson University.

Elemental composition analysis can be done using proton induced x-ray emission (PIXE) or using x-ray fluorescence (XRF). Both methods are sensitive even to very low concentrations so sampling time may be short. In addition, Samples are analyzed directly from the substrate (either Nylon or Teflon) so no preparation is required.

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