Evaluation of Methods for the Physical Characterization of the Fine Particle Emissions from Two Residential Wood Combustion Appliances

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Abstract: The fine particle emissions from a U.S. certified non-catalytic wood stove and a zero clearance fireplace burning Ouercus rubra L. (northern red oak) and Pseudotsuga menziesii (Douglas fir) cordwood each at two different moisture levels were determined. Emission testing was performed using both timeintegrated and continuous instrumentation for total particle mass, particle number, particle size distribution, and fixed combustion gases using an atmospheric wind tunnel, full-flow laboratory dilution tunnel, and dilution stack sampler with a comparison made between the three dilution systems and two sampling filter types. The total mass emission factors (EFs) for all dilution systems and filter media are extremely variable ranging from < 1 to ~ 55 g kg⁻¹ of dry wood depending on the combination of appliance type, wood species and moisture content, filter medium, and dilution system. For Teflon filter sampling of stove emissions in the wind tunnel, the total mass EFs varied from ~ 2 to 8 g kg⁻¹ of dry fuel depending on wood type whereas the equivalent fireplace emissions burning wet oak averaged 11 g kg⁻¹. A substantial number of ultrafine particles in the accumulation size range were also observed during all tests as determined by an Electrical Low Pressure Impactor (ELPI) and Scanning Mobility Particle Sizer. The PM-2.5 (particles $\leq 2.5 \ \mu m$ in aerodynamic diameter) fractions determined from the ELPI electrometer data ranged from 93 to 98% (mass) depending on appliance type as reported previously by Hays et al. (Aerosol Science, 34, 1061, 2003).

Keywords: wood burning appliances; room heaters; fireplaces; particle emissions; dilution systems

1. Introduction

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The fine particles generated by residential wood combustion (RWC) appliances continues to be an environmental concern in many parts of the world (Luhar et al., 2006; Robinson et al., 2007; Sciare et al., 2008; Hellen et al., 2008). This is especially true in the U. S. where there is a renewed interest in the use of wood for home heating resulting from the high cost of natural gas and heating oil (Houck, 2006). In wood burning regions of the U. S., levels of PM-2.5 (i.e., particles $\leq 2.5 \,\mu$ m in classical aerodynamic diameter or 2.5 μ mA) above the National Ambient Air Quality Standard have been measured at ambient monitoring sites (U.S. EPA, 2004). A recent survey has also revealed that approximately 80% of the wood stoves currently in use in the U. S. are older types which have higher PM emissions than those meeting the current U. S. Environmental Protection Agency (U.S. EPA) New Source Performance Standard (U.S. EPA, 2000a; Houck and Keithley, 2004). In addition to creating PM-2.5 air quality problems, the particles created by RWC appliances also contain substantial quantities of polycyclic aromatic hydrocarbons (PAHs) and other hazardous air pollutants (HAPs) which have been implicated in adverse health effects (Purvis, et al., 2000; McCrillis, 2000; Fine, et al., 2001; Hedberg, et al., 2002; Gullett, et al., 2003; Hedman, et al., 2006; McDonald et al., 2006; Naeher et al, 2007).

Historically, the testing of wood stoves and fireplaces has generally involved the determination of total particle mass emissions using a full-flow laboratory dilution tunnel and filter sampling train similar to that published in U.S. EPA Reference Test Method 5G (U.S. EPA, 2000b). Other, more limited, evaluations of selected PAHs and other chemical constituents have also been made using either dilution stack samplers or traditional in-stack cascade impactors (Stiles, 1983; McCrillis, 2000; McDonald, 2000; Fine, et al., 2001; Tissari, et al., 2007). Also, in-home testing of U.S. EPA certified wood stoves and the subsequent evaluation of these stoves for long term durability has been conducted using automated stack sampling equipment (McCrillis and Jaasma, 1993; Champion and Jaasma, 1998). Finally, a number of investigators such as Lipsky and Robinson (2006) have found that dilution conditions can substantially effect the measurement of particulate emissions from RWC appliances.

Virtually no data currently exist, however, on the properties of RWC emissions as the plume naturally cools and dilutes in the atmosphere. Only limited measurements have been conducted, none of which can be used for the development of quantitative emission factors (Rau, 1989; Bighouse and Houck, 1995; Gysels and Van Grieken, 1999). Such data are needed in order to make a valid comparison between the characteristics (e.g., size distribution, chemical composition, etc.) of RWC emissions determined in the laboratory vs. those actually existing in the

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atmosphere (Donahue et al., 2009). This information is also needed for air quality source apportionment studies conducted as part of the development of PM-2.5 State Implementation Plans.

This paper describes an emission characterization program conducted by the U.S. EPA's National Risk Management Research Laboratory (NRMRL) of a U. S. certified non-catalytic wood stove and a zero clearance fireplace installed in a calorimeter chamber burning *Quercus rubra L*. (northern red oak) and *Pseudotsuga menziesii* (Douglas fir) cordwood at two different moisture levels. Emissions were sampled using a combination of techniques including an atmospheric wind tunnel to represent natural cooling and dilution as well as two different laboratory dilution systems. The objectives of the study were to determine the physical and chemical characteristics of the PM emissions from each appliance under "near-real-world" conditions and compare the plume data collected in the wind tunnel to laboratory dilution methods. This paper addresses only the physical characterization of the RWC aerosol from the two appliances.

2. Test Facility

The NRMRL RWC laboratory consisted of a calorimeter chamber, atmospheric wind tunnel (or, in some tests, full flow laboratory dilution tunnel), dilution stack sampler, and associated samplers and on-line analyzers. A diagram of the facility is shown in Fig. 1 with a brief description provided below.

Each appliance tested was installed in a 14.5 m³ calorimeter chamber (see Supporting Information) constructed of conventional building materials. Ventilation of the calorimeter chamber was provided by two centrifugal blowers operating in a "push/pull" configuration to achieve a nominal air flow rate of ~ 57 actual m³ min⁻¹. The "pull" side blower was regulated by a pressure transducer to achieve a neutral differential pressure (i.e., < 1 Pa) between the inside and outside of the chamber thus allowing proper operation of the appliance. The chamber was calibrated over three power levels utilizing a 30 kW bank of resistance heaters with the current supplied to the heaters determined by a clamp-on ammeter.

An atmospheric wind tunnel (see Supporting Information) was constructed especially for this study and installed on the roof of the laboratory. The purpose of the tunnel was to emulate, to the extent possible, the near-field plume cooling and dilution processes occurring during actual in-home operation. The ~ 16.2 m long wind tunnel consisted of three major sections: a 3.1 m square "working section" where the plume was initially cooled and

diluted; a ~ 9.4 m transition section; and a 3.7 m measurement section (Fig. 1). Unconditioned ambient air at a rate of ~ 590 actual m³ min⁻¹ was pulled through the tunnel by a large axial fan to produce an equivalent wind velocity of ~ 1.0 m s⁻¹ through the working section and a 600:1 nominal dilution ratio (DR). A honeycomb flow straightener was also installed on the tunnel inlet to reduce wind effects and promote uniform flow distribution. Finally, two small axial fans were installed near the stack entrance point near the bottom of the working section to help mix the plume with the ambient dilution air and allow proper sample collection downstream.

A suite of on-line particle analyzers and manual sampling trains was used in the wind tunnel measurement section as described in Table 1. The on-line analyzers were connected to probes located inside a 15.2 cm diameter, high volume sampling manifold which isokinetically extracted 6.4 m³ min⁻¹ of air from the tunnel. The location of the manifold inlet and air sampling rate were selected using the results of both velocity profiles and tracer gas (CO) sampling conducted in the tunnel. The manifold and analyzers were located in a heated and insulated enclosure attached to the side of the tunnel to maintain suitable operating temperature. Also, three different time-integrated sampling trains were positioned inside the measurement section to determine total particle mass emissions and chemical composition as described below.

In addition to the atmospheric wind tunnel, a dilution stack sampler was also used in all tests. This sampler is a modification of the NRMRL wood stove dilution sampler (WSDS) which provides a nominal 30:1 dilution of the raw stack gas (Merrill and Harris, 1987). For this particular study, however, a supplemental liquid nitrogen-cooled system was added to reduce the dilution air temperature to 0 $^{\circ}$ C with residual moisture, gas-phase organics, and PM removed by a combination silica gel/activated carbon trap and final 0.01 µm PM filter. A slipstream of sample air from the WSDS was also directed through a 106 L residence chamber similar to the commonly used dilution sampler developed by Hildemann et al. (1989). The residence time in the chamber depended on the number and flow rate of the samplers used but generally ranged from 149 to 224 s as compared to ~ 7 s in the wind tunnel.

In six of the 28 tests conducted a full-flow laboratory dilution tunnel, described elsewhere (U.S. EPA, 2000b; Purvis et al., 2000; McCrillis, 2000), was used in place of the atmospheric wind tunnel. However, no on-line particle analyzers were operated during the limited laboratory dilution tunnel tests.

Finally, a suite of on-line analyzers was used to monitor CO and CO_2 (Rosemont Model 880), NO_x (Thermo Electron Model 10), total hydrocarbons (THC; Thermo Electron Model 42), and O_2 (Rosemont Model 755) in the exhaust stack. The gas sample for the above analyzers was extracted from the flue at a point 2.4 m above the scale

platform using a stainless steel 'button-hook" probe pointed downstream to the direction of gas flow. Carbon monoxide was also monitored in either the wind tunnel or dilution tunnel to determine the dilution ratio for each test. In addition, a Delmhorst RDX-1 electrical capacitance moisture meter was also used to determine the average moisture content of each wood charge.

3. Experimental Protocol

3.1 Appliance and Calorimeter Operation.

A special set of operating procedures was prepared for each appliance based on recommendations provided by an expert panel convened for this purpose. These procedures incorporated a number of existing methods including: U.S. EPA Methods 28 and 5G, the State of Washington (SoW) fireplace method, and the operating procedures for the WSDS (Merrill and Harris, 1987; State of Washington, 1995; U.S. EPA, 2000b; U.S. EPA, 2000c). These procedures were included in the Quality Assurance Project Plan for the study and are briefly described below.

For the wood stove (Quadrafire Model 3100) experiments, each test run consisted of three burn phases performed in series: kindling $(7.8 \pm 0.9 \text{ dry kg hr}^{-1})$, high burn rate $(5.2 \pm 0.9 \text{ dry kg hr}^{-1})$, and low burn rate $(1.5 \pm 0.4 \text{ dry kg hr}^{-1})$. Time-integrated emission samples (Table 1) were collected during all burn phases to produce a composite sample representative of the entire test. Phase-specific emissions were also determined using the data obtained from the Tapered Element Oscillating Microbalance (TEOM) and other continuous instruments described previously.

In the kindling phase, a single fuel charge was prepared consisting of: loosely waded black-print newspaper, ~ 1 kg of split wood arranged in a "tic-tac-toe" configuration, and two kindling logs weighing ~ 1.2 kg for a total charge weight of ~ 3.1 kg. Nominal dimensions of the split wood and kindling logs were 2.5 cm x 2.5 cm x 36 cm and 7.6 cm x 5 cm x 36 cm, respectively. All fuel was allowed to equilibrate to the temperature of the test facility prior to being introduced into the appliance.

After placing the kindling inside the combustion chamber, the newspaper was lit, the charging door allowed to remain open ~ 2.5 cm until such time that the fire was well established (or no longer than 5 min after starting the

fire), and the time-integrated samplers started. At the end of 11 min, the main fuel charge was added to the stove to start the high burn phase.

For the high burn condition, a main fuel charge of 160 ± 16 kg m⁻³ of usable firebox volume was added to the stove's combustion chamber within 1 min of completing the kindling phase. This charge was comprised of fuel pieces with a maximum diameter of 8 to 13 cm sized using a previously developed template (Bighouse et al., 1994). After adding the fuel, the charging door was closed and the dampers opened to their maximum setting. No adjustments were made to the fuel until at least half the charge was consumed.

Within 1 min of completing the high burn phase, a supplemental fuel charge of 160 ± 16 kg m⁻³ of usable firebox volume was added to the stove. For this charge, the maximum diameter of the fuel was 13 to 20 cm as determined using the same template described above. After closing the charging door, the dampers were closed to their minimum setting and the fuel allowed to burn until such time that the flue gas temperature (as measured at 2.4 m above the flue collar) dropped to 75° C above ambient at which time the time-integrated samplers were shut off and the test completed.

For the zero-clearance fireplace (FMI Model 42E), a similar fueling protocol was employed except that only the kindling $(19 \pm 11 \text{ dry kg hr}^{-1})$ and high burn phases $(2.2 \pm 1.2 \text{ kg hr}^{-1})$; three charges in series) were evaluated with the damper fully open. Adjustments to the fuel were only made during the kindling charge. In addition, the test was considered complete when the flue gas temperature dropped to 25° C above the ambient temperature as measured 2.4 m above the flue collar.

3.2 Test Fuels

Two different wood species were tested: North Carolina red oak, and Douglas fir cordwood imported from the Pacific Northwest. Each fuel was tested at two moisture contents in an attempt to represent a nominal "wet" and "dry" condition. For the red oak, the fuel moisture content (dry basis) of the wet wood was \sim 30% whereas the dry fuel was \sim 11%. In the case of the Douglas fir, the moisture contents were \sim 22% and \sim 12%, respectively, for the wet and dry conditions.

3.3 Time-Integrated Sampling and Analysis

Different time-integrated sampling trains were used depending on the particular test being conducted. For the "non-speciated" runs (i.e., those where chemical composition was not determined) modified Method 5G (MM5G) and Teflon filter sampling trains were used in the WSDS and either the wind tunnel or dilution tunnel, as applicable. In the MM5G train, a standard 47-mm fiberglass filter followed by a silica gel trap was used to determine PM mass concentration similar to the published U.S. EPA method (U.S. EPA, 2000b). A second sampling train consisting of a 47 mm Teflon (Teflo[®]) filter with a silica gel backup tube was also used to provide an independent measure of total PM mass concentration. For the "speciated" tests, a 47-mm pre-fired quartz filter followed by two polyurethane foam (PUF) plugs was used to determine PM elemental/organic carbon content as well as semi-volatile organic compounds in both the particle- and gas-phase. This train replaced the MM5G fiberglass system in the speciated runs.

Fiberglass filter samples from the MM5G sampling train were analyzed per the published U.S. EPA method and weighed with a Denver Instruments analytical balance weighing to the nearest 0.1 mg. However, the 47-mm Teflon filters were equilibrated for at least 24 h at constant temperature of 22 °C \pm 1 °C and relative humidity of 35% \pm 1% in a special designed weigh room both before and after sampling. A Sartorious MC-5 microbalance was used to weigh the Teflon filters to \pm 3 µg. During weighing, the balance was checked at frequent intervals with standard (Class S) weights to ensure accuracy. Before sampling, the filters remained in the same controlled environment until a second analyst conducted a 100% audit as a precision check. After sampling, the same procedure was used with 10% of the filters reweighed for quality control purposes. Analysis of the filters and other media for elemental and organic carbon, semi-volatile organics, water-soluble ions, and organic acids will be described in a companion paper.

4. Data Reduction and Analysis Procedures

A total of 28 test runs were conducted in the winter/early spring which consisted of 24 stove tests and four fireplace tests. A breakdown of these experiments by fuel type, moisture content, and dilution system is provided in the Supporting Information. Of the 28 tests, one speciated run was performed for each operating condition defined by the type of appliance, fuel moisture content, wood species, and dilution system (e.g., stove-wet oak-wind tunnel).

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Time-integrated particle mass emission rates and factors were determined using the results of the gravimetric filter analyses and the applicable stack and dilution system operating parameters recorded during each test using the general calculation procedures outlined in U.S. EPA Method 5G (U.S. EPA, 2000b). The wind tunnel TEOM data were likewise reduced from the raw data files to obtain both test average and burn phase specific total particle mass emission rates and factors. A similar calculation scheme was also used to determine the emissions on a particle number basis using the wind tunnel ELPI data.

The average differential number particle size distribution (PSD) determined from the ELPI or SMPS was obtained by averaging all the dN/dlogDp data in the same size bins for the entire test. The resulting PSD was then smoothed against the bin sizes using the "supsmooth" function provided in the MathCAD (Mathsoft, Inc.) software package. Similar PSD data on a differential mass basis developed from the gravimetric analysis of ELPI aluminum foil collection substrates were presented earlier by Hays et al. (2003).

5. Experimental Results

5.1 Total Particulate Emissions

The total mass emission factors (EFs) and emission rates (ERs) on a test average basis and EFs by burn phase are summarized in Fig. 2. Looking at the EFs in Fig. 2a for the wind tunnel, it can be seen that the stove particle emissions ranged from 2.4 to 7.7 g kg⁻¹ dry fuel. Also, the stove factors for oak were generally higher than fir for the same relative fuel moisture with the burning of wet fuel producing almost twice the PM emissions. The wind tunnel results also show the fireplace to have the highest overall particle EF (11 g kg⁻¹) of the two appliances burning wet oak. For the laboratory dilution tunnel EFs shown in Fig. 2a, the stove emissions measured for wet oak were significantly [relative percent difference (RPD) = 45%] greater than those determined in the wind tunnel whereas for the stove burning dry fir, the two tunnel systems produced essentially equivalent (RPD = 2.6 %) results. There is no apparent reason for the differences observed between the two tunnel systems for wet oak except that the emission factors for fir were generally more reproducible for all scenarios evaluated in the study.

The average total mass EFs by burn phase are shown in Fig. 2b as determined from the TEOM data corrected to an equivalent Teflon filter value obtained through linear regression analysis. Looking at the wind tunnel results in

Fig. 2b, the kindling EFs are generally similar across all appliances and wood types with the burning of fir in the stove being slightly higher. For the high burn and low burn operating conditions in the stove, wet fuel produced ~ 1.2 to 2.2 times higher PM emissions than dry fuel depending on wood species. Also, the highest particle EFs for each appliance-fuel combination were measured during the low burn phase with its more limited airflow available for combustion as reported in the literature (McCrillis, 2000; McDonald et al., 2000; Gras et al., 2002; Jordan and Seen, 2005). Finally, for the fireplace, the total mass emissions were 2.6 times higher while burning wet oak as compared to the stove burning the same fuel at high burn.

With regard to the dilution tunnel data in Fig 2b, the dilution tunnel EFs were either higher or lower than those generated in the wind tunnel for the stove burning wet oak depending on burn phase whereas for fir they were always a factor of ~ 1.7 higher during both high and low burn. Note, however, that only very limited data are available for the dilution tunnel and caution should be exercised when making any definitive comparison to the wind tunnel results.

The wind tunnel EFs shown in Fig. 2 are within the range of published values (McDonald, et al., 2000; Gras et al., 2002; Zhang and Morawska, 2002) but generally lower than both the current U.S. EPA guidance (U.S. EPA, 1996) and prior testing of the same appliance conducted by McCrillis (2000) using a dilution tunnel. The EFs for the fireplace are also consistent with most prior work (Hildemann et al., 1991; Rogge et al., 1998; Broderick et al., 2005) but again slightly lower than U.S. EPA guidance (U.S. EPA, 1996) and previous measurements on the same appliance as reported by Purvis et al. (2000).

Finally, total particle number EFs and ERs from the wind tunnel ELPI measurements are provided in Fig. 3 in a format similar to that of Fig. 2. As shown in Fig. 3, the particle number EFs show similar trends to those observed for the mass EFs determined from the Teflon filter data. These results do indicate, however, that large numbers of particles are emitted per mass of wood burned.

5.2 Methods Comparison

The test average total mass EFs for all dilution systems and filter media are shown in Fig. 4. As can be seen from this figure, the average EFs presented are extremely variable ranging from < 1 to ~ 55 g kg⁻¹ of dry wood

depending on the combination of appliance type, wood species and moisture content, filter medium, and dilution system. The average EFs themselves are also highly variable as indicated by the large error bars.

Looking at the Teflon total mass EFs for the three dilution systems in Fig. 4a, a number of observations can be made. First, for oak burned in the stove, the dilution sampler measurements produced substantially higher EFs than the wind tunnel, whereas the stove burning fir was fairly comparable. Also, the dilution sampler measured greater stove emissions for dry oak as compared to wet oak, which is inconsistent with that found in the wind tunnel where burning wet fuel always exhibited higher emissions. In the case of the fireplace, the average EF measured in the dilution sampler was approximately half of that determined in the wind tunnel. Generally higher dilution sampler results could be attributed to a lower DR and dilution air temperature (see Lipsky and Robinson, 2006) but no explanation can be offered for the opposite effect as was seen for the fireplace. Finally, comparing the dilution tunnel to the dilution sampler for the stove in Fig. 4a, somewhat (i.e., factor of 1.4 to 1.9) smaller particle EFs were measured by the dilution stack sampler for the wood species tested. These results cannot be explained by differences in dilution conditions, however.

Additional differences were also observed between the filter media used in the various dilution systems. In the dilution/wind tunnel tests, except for the stove burning wet oak measured in the wind tunnel, the Teflon filter EFs were higher than those generated from the fiberglass filter data. For the dilution sampler, just the opposite was true. Also, the fiberglass EFs measured in the dilution sampler for the stove burning oak were not only higher than the Teflon filters values but also unrealistically (a factor of 4 to 23) greater than those determined in the wind tunnel for the same filter medium and as reported in the published literature (McCrillis, 2000; McDonald et al., 2000; Zhang and Morawska, 2002).

To further investigate differences in the results from the two filter types, individual test results for the wind and dilution tunnels were plotted by filter medium against the equivalent dilution sampler data and simple linear regression analyses conducted. These analyses showed a good correlation ($r^2 > 0.9$) between the dilution tunnel and dilution sampler for both filter types whereas little or no correlation was found between the wind tunnel and dilution sampler. The most apparent reason for this lack of correlation are inherent differences (e.g., residence time, dilution air temperature, etc.) associated with natural vs. artificial dilution.

5.3 Particle Size Results

As stated above, test average PSDs were developed from both the ELPI and SMPS data. Example PSDs and size statistics for selected tests representing different wood types burned in the stove and oak in the fireplace are shown in Fig. 5. As shown in Fig. 5 for the stove, the PSDs for dry oak (Test 10) and wet fir (Test 17) exhibit two modes, a minor mode centered at \sim 15 and a major mode at 40 nm. In the case of the dry fir stove test (Test 14), only a single accumulation mode at \sim 30 nm was found. However, the PSD for Test 14 is slightly broader than that for Tests 10 and 17 indicating more particles in the larger size ranges as reflected by the size statistics. Finally, in the case of the fireplace PSD shown in Fig. 5 (Test 21), a single accumulation mode centered at \sim 70 nm was observed.

The particle size data by mass determined by the wind tunnel ELPI were reported previously (Hays et al., 2003). Using the ELPI electrometer data, the overall average PM-2.5 mass fraction was determined to be \sim 93% for the stove and \sim 98% for the fireplace for all fuels and burn conditions. This compares to studies by Purvis et al. (2000) and McCrillis (2000) which used an Anderson Mark III 8-stage, in-stack cascade impactor to determine PSDs for fireplaces and stoves (including those tested in the current work) using similar oak fuel and laboratory dilution tunnel system described above. In the Purvis (2000) study, the PM-2.5 fraction of the emissions from two conventional zero clearance fireplaces was found to range from 75 to 82% whereas McCrillis (2000) measured \sim 70% PM-2.5 at high burn and > 90% at low burn for the same Quadrafire stove evaluated in the current study. The differences in the PM-2.5 fraction can be attributed to variations in both the instrumentation and dilution system (i.e., wind tunnel vs. dilution tunnel) used to collect the data.

6. Discussion

As can be seen from the experimental data, substantial differences in EFs were produced by the different dilution systems and filter media evaluated which can be attributed to variations in dilution conditions, residence time, particle losses, and the high variability in aerosol characteristics typical of batch operated RWC appliances. All of these factors make a comparison of the various dilution methods more difficult especially when evaluating the laboratory dilution tunnel where the data were so limited. However, the total mass EFs from the dilution tunnel and dilution sampler appear to at least correlate with each other for most experimental conditions. This is not the case, however, when comparing the dilution sampler to the wind tunnel. Either higher or lower EFs were produced by the

dilution sampler depending on test parameters. The biggest differences between the three systems were observed during the burning of oak as compared to fir which produced generally similar EFs. Also, the test results obtained with Teflon filters appear to be more consistent than was the case for the fiberglass filters. This is especially important since the laboratory dilution tunnel and fiberglass sampling train used in the study is the accepted method for appliance certification in the U. S.

Based on the above test results, the use of Teflon filter sampling in the wind tunnel appears to be the most reliable of the techniques evaluated for the determination of total particle mass emissions from the two appliances and two fuels tested. The experimental results for the wind tunnel are at least internally consistent, within the same general range as prior studies, and most closely reflect "real world" dilution conditions. The wind tunnel is, however, relatively expensive and requires a large amount of roof area which may or may not be available at some facilities.

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Table 1 Instrumentation used for physical characterization measurements^a

Test Parameter	Sampling Rate	Wind Tunnel	Dilution Tunnel ^b	Modified WSDS ^c
Particle mass concentration	Continuous	R&P Model 1400a TEOM	N/A	R&P Model 1400a TEOM
	Time-integrated	MM5G (fiberglass filter; non-speciated runs only) ^d	MM5G (fiberglass filter; non-speciated runs only) ^d	MM5G (fiberglass filter; non-speciated runs only) ^d
	Time-integrated	Teflon filter sampler	Teflon filter sampler	Teflon filter sampler
Particle number concentration	Continuous	Dekati ELPI	N/A	N/A
Particle size distribution	Continuous	TSI Model 3934 SMPS	N/A	N/A
	Continuous and time-integrated	Dekati ELPI	N/A	N/A
Carbon monoxide	Continuous	Draeger 190 CO Datalogger	Draeger 190 CO Datalogger	N/A

^a TEOM = tapered element oscillating microbalance; SMPS = scanning mobility particle sizer; ELPI = electrical low pressure impactor; SMPS = Scanning Mobility Particle Sizer; and MM5G = modified U.S. EPA Reference Method 5G sampling train. N/A = instrument not available. Note that CO, CO₂, NO_x, and THC are also continuously monitored in the stack. Particle losses in each dilution system were not determined.

^b Modified U.S. EPA Method 5G dilution tunnel per 40 CFR Part 60, Appendix A. The dilution tunnel was used in place of atmospheric wind tunnel in some tests.

^c Wood stove dilution sampler equipped with a cyogenic cooling/conditioning system and residence chamber.
^d Non-speciated refers to those tests with no chemical characterization performed.

Figure 1 Click here to download high resolution image



^{**} Used in non-speciated runs only







Figure 5 Click here to download high resolution image



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