

**TECHNICAL SESSION IV**

***Research & Emerging  
Technologies***



## **Dioxin and Trace Metal Emissions From Combustion of Carbonized RDF Slurry Fuels**

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## INTRODUCTION

In 1994, the U.S. generated approximately 209 million tons of Municipal Solid Waste (MSW), with 61% landfilled, 24% recycled, and 15% processed through Municipal Waste Combustion (MWC).<sup>1</sup> In order to divert a larger portion of this generated MSW from landfills, MWC will have to play a growing role in MSW disposal. However, recently promulgated New Source Performance Standards (NSPS) for MWC will add an additional financial burden, through mandated emission reductions and air pollution control technologies, to an already financially pressured MWC marketplace.<sup>2</sup>

In the past, Refuse Derived Fuel (RDF), a solid fuel produced from MSW, has been fired in industrial and coal boilers as an alternative means of MWC. While lower sulfur dioxide (SO<sub>2</sub>) emissions provided the impetus, firing RDF in industrial and coal boilers frequently suffered from several disadvantages including increased solids handling, increased excess air requirements, increased air emissions, increased slag formation in the boiler, and higher fly ash resistivity.<sup>3</sup>

Bituminous and higher rank coals can generally be slurried in water to form pumpable fuels of satisfactory solids loading, and hence heating value.<sup>4</sup> These slurry fuels, sometimes termed Coal-Water-Mixtures (CWM) or Coal-Water-Fuels (CWF), are commercially fired in pulverized coal and oil boilers.<sup>5</sup> However, low rank coals and solid fuels like MSW and RDF do not slurry as well. Supported by the U.S. Environmental Protection Agency (EPA), U.S. Department of Energy (DOE), and U.S. National Science Foundation (NSF), EnerTech has developed a process called SlurryCarb<sup>™</sup>, which greatly improves the solids loading, heating value, chlorine content, and uniformity of MSW and RDF slurry fuels. It is anticipated that the enhanced carbonized slurry fuels can be effectively fired in pulverized coal or oil boilers without the previously cited disadvantages. This paper summarizes the latest emissions and combustion tests with the carbonized RDF slurry fuel. EnerTech completed this program in conjunction with the American Plastics Council (APC - Washington, DC) and the Energy and Environmental Research Corporation (EER - Irvine, CA).

With EnerTech's SlurryCarb<sup>™</sup> process (see Figure 1), a pumpable slurry of RDF is continuously pressurized with a pump to between 1200 and 2500 psi. The RDF slurry is pressurized above the saturated steam curve to maintain a liquid state when the slurry is heated to approximately 480 to 660°F. Slurry pressure and temperature then are maintained for less than 30 minutes in plug-flow reactors. At this temperature and pressure, oxygen functional groups in the molecular structure of the RDF are split off as carbon dioxide gas. This evolved carbon dioxide gas comprises a significant weight percentage of the feed RDF, but only a minimal percentage of the heating value. While the mass of total solids is reduced approximately 20-70%, the carbonized product still contains approximately 95-98% of the energy content of the feed RDF. The carbonized RDF particles are dramatically reduced in size and can be concentrated to a higher solids loading, and hence specific heating value, comparable to CWM and CWF. In addition, chlorinated organics in the RDF are decomposed to hydrochloric acid, neutralized with alkalis, and removed as chloride salts (along with a majority of other slag forming compounds), producing an essentially chlorine free carbonized RDF slurry fuel.

## RESEARCH PERFORMED

The primary objective of this program was to determine the emissions of dioxins/furans, trace metals, and other flue gas pollutants from combustion of the carbonized RDF slurry fuel in a pilot scale pulverized coal boiler simulator. The combustion tests were completed in EER's 800,000 Btu/hr pilot scale Boiler Simulation Facility (BSF). Flue gases were sampled after an electrostatic precipitator, according to EPA

protocols, for dioxin/furans and trace metal emissions. Continuous Emission Monitors (CEMs) were utilized to determine carbon monoxide (CO), nitrogen oxides (NO<sub>x</sub>), SO<sub>2</sub>, and hydrochloric acid (HCl) emissions.

### **Carbonized RDF Slurry Fuel Preparation**

RDF pellets from Thief River Falls, Minnesota have been used as the source material throughout EnerTech's process development program. This source of RDF was chosen due to its proximity to EnerTech's research facilities, and heating value and ultimate analysis similarity to RDF produced with wet based resource recovery operations (see table 1). The Thief River Falls Material Recovery Facility (MRF) receives mainly MSW, some commercial office waste, and limited agricultural solid waste. The incoming waste is shredded and hand picked to remove glass bottles, aluminum cans, and PET and HDPE plastic bottles. The remaining waste is trommeled, air classified, subjected to magnetic separation, and finally pelletized. The RDF for EnerTech's development program was purchased as pellets and then shredded by EnerTech in a Nelmor granulator to a minus 1/16-inch particle size. It is roughly estimated that the RDF consisted of approximately 60-70 wt.% paper, 15-20 wt.% plastics, and the balance "other" (dry basis).

The carbonized RDF slurry fuel for the combustion tests was produced using the University of North Dakota's Energy & Environmental Research Center's continuous 7.5 ton/day (wet basis) hot-water-drying pilot plant. A detailed description of this facility can be found in previous papers by EnerTech.<sup>6, 7, 8</sup> In January 1996, a series of experiments was completed with this pilot plant in which the plastic content of the RDF was varied by adding the appropriate amount of virgin plastic materials. The objective of this program was to assess the effect of plastics on carbonized slurry fuel properties. Using a factorial design-of-experiment methodology, RDF total wt.% plastic, RDF plastic composition, reactor temperature, and RDF chlorine content were varied across eight distinct test conditions, and samples of each product stream were analyzed for relevant properties. The carbonized RDF from these eight tests was blended together and used as the carbonized RDF slurry fuel for the combustion tests described in this paper.

### **Fuel Delivery and Combustion Facility Description**

Combustion tests with the carbonized RDF slurry fuel were completed in EER's pilot scale 800,000 Btu/hr BSF. The BSF (please see Figure 2) is a pulverized coal boiler simulator and was designed to provide an accurate simulation of the flue gas temperatures and residence times in a full scale utility boiler. The BSF consists of a burner, vertically down-fired radiant furnace, horizontal simulated convective pass, and electrostatic precipitator. A Moyno progressive cavity pump was used to pump the carbonized RDF slurry fuel to a variable swirl diffusion burner with an axial fuel injector. Compressed air was used to atomize the slurry fuel. In the burner, primary air was injected axially, while the secondary air stream was injected radially through the swirl vanes to give controlled fuel/air mixing. The cylindrical furnace section is constructed of eight modular refractory-lined sections with an ID of 22 inches and an overall furnace height of 22 feet.

The convective section of the BSF also is refractory lined, and consists of three horizontal passes, with each pass having a length of 18 feet and square dimensions of 8 inches by 8 inches. The convective section contains air cooled tube bundles to simulate the reheater, superheater, and economizer sections of a full scale boiler. A tubular electrostatic precipitator is located at the end of the convective pass for particulate control. Numerous coals, oils, and other fossil fuels previously have been tested in the BSF by EER, leading to the development of an extensive database of test procedures and results.

## CEMs and Flue Gas Sampling Systems

Flue gas emissions from combustion of the carbonized RDF slurry fuel were determined through extractive manual sampling and a Continuous Emission Monitor (CEM) system. CEM components included a water cooled sample probe, sample conditioning system (to remove water and particulate), and gas analyzers. CEM sampling was performed in the convective pass upstream of the electrostatic precipitator (ESP), at a gas temperature of 600°F. Species analyzed, detection principles, and limits of precision are as follows:

- Oxygen (O<sub>2</sub>): paramagnetism, 0.1 vol.% precision
- NO<sub>x</sub>: chemiluminescence, 1 ppm precision
- CO: nondispersive infrared spectroscopy, 1 ppm precision
- Carbon Dioxide (CO<sub>2</sub>): nondispersive infrared spectroscopy, 0.1 vol.% precision
- SO<sub>2</sub>: nondispersive ultraviolet spectroscopy, 1 ppm precision
- HCl: gas filter infrared spectroscopy, 1 ppm precision (Servomex Model 2510 HCl analyzer)

Polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) were determined from combustion of the carbonized RDF slurry fuel through extractive sampling downstream of the ESP by EPA Method 23. The facility was operated such that the ESP inlet temperature was 320°F and the outlet temperature was 280°F where sampling was performed (temperature drop across the ESP was due to radiant and convective heat losses). With EPA Method 23, a flue gas sample is extracted isokinetically from the stack through a heated probe and filter, and directed into an impinger train. The sample trains were recovered immediately after sampling and the samples were sent to an analytical laboratory the next day for extraction and analysis. Analytical work was performed by Alta Analytical Laboratory, Inc. using high-resolution gas chromatography and mass spectrometry (DB5 column only since 2378-TCDD concentration was below 10 pg per sample and DB225 column was not needed). Quality assurance procedures included leak-checks, strict precleaning of apparatus, and charging and recovery of impinger modules under controlled conditions.

Coincident with PCDD/PCDF sampling, emissions of trace metals from combustion of the carbonized RDF slurry fuel were determined using EPA Method 29. Again, a flue gas sample was extracted isokinetically from the stack through a heated probe and filter, and directed into an impinger train. Metal analyses were performed by Research Triangle Institute. Mercury analysis was performed by cold vapor atomic absorption spectroscopy. All other metals were analyzed by inductively coupled plasma mass spectrometry. Rigorous quality assurance procedures again were applied, including leak-checks and strict precleaning of apparatus. Determination of mercury emissions included analysis of the KMnO<sub>4</sub> impinger. This procedure has been updated to prevent cross-contamination by using an empty impinger between the HNO<sub>3</sub> and KMnO<sub>4</sub> impingers. A lab blank also was run which did not reveal any interference or contamination.

## RESEARCH RESULTS

### Fuel Properties

Table 1 summarizes the fuel and rheological characteristics of the carbonized RDF slurry fuel used for the combustion test. As discussed, the carbonized RDF slurry fuel employed in this study was derived from a densified RDF spiked with additional plastics. When compared to CWFs or CMWs, the heating value and solids loading of the carbonized RDF slurry fuel were comparable. In addition, SlurryCarb™ has significantly reduced the chloride, mercury, and cadmium content of the carbonized RDF slurry fuel from the Thief River Falls RDF. In order to complete the combustion tests with the worst case scenario

slurry fuel, the carbonized RDF was not washed to reduce chloride content, and the slurry fuel still contained significant chloride (approximately 700 ppm, dry basis). Washing of a small sample of the carbonized RDF reduced the chloride content to below 100 ppm (dry basis).

Slurry viscosity was determined by shearing a slurry sample in the annular space between a rotating cylinder and a stationary cup. The torque necessary to rotate the cylinder at a given speed was measured by a torsion spring. In general, slurry fuels are nonnewtonian fluids. With nonnewtonian fluids, the viscosity is dependent upon the shear rate and as a result, the rotating speed of the cylinder.

### Emissions

The baseline firing rate of the carbonized RDF slurry fuel during the combustion test in EER's BSF was approximately 750,000 Btu/hr (approximately 130 lbs/hr, wet or slurry basis). Approximately 20% excess air was provided based upon the ultimate analysis of the slurry fuel. The total combustion air flow rate was 150 scfm. Primary air flow rate was 15 scfm or approximately 10% of the total combustion air. Atomization air also was approximately 15 scfm, with the balance of combustion air as secondary air (approximately 120 scfm). Tertiary air was not used. Combustion air was preheated to 200°F. Flue gas flow rate from the ESP was approximately 145 dry scfm.

EPA Method 23 was used to determine PCDD/PCDF emissions from combustion of the carbonized RDF slurry fuel. Two sampling periods were completed, and the analytical results were speciated into total PCDD/PCDF, five congeners (tetra- through octa-), and nine 2,3,7,8 substituted individual isomers. Table 2 summarizes the results of the two PCDD/PCDF sampling periods and calculated averages. Sampling times and analytical methods were established to provide a level of precision more than an order of magnitude greater than current regulatory NSPS for MWC. Each sampling period lasted approximately 2 hours. As can be seen from Table 2, total PCDD/PCDF concentration averaged 0.38 ng/dscm (7% O<sub>2</sub>, dry basis) for the two sampling periods, which is 97% lower than the NSPS for MWC of 13 ng/dscm.

Figure 3 and Figure 4 depict the PCDD and PCDF congeners distribution. Reasonably good agreement was obtained between the total PCDD/PCDF concentration for the sampling periods, although the levels of individual isomers varied between each sampling period. Most species were present in measurable quantities, including all congener groups. The less chlorinated species were more abundant for both the PCDD and PCDF. A few isomers were below detection limits. Most notably, 2378 TCDD concentration was below the detection limit of 0.0013 ng/dscm (7% O<sub>2</sub>, dry basis) for both runs.

Table 3 and Figure 5 summarize the 13 trace metal emissions characterized from the two sampling periods and calculated averages. Agreement between the two sampling periods was fairly good. For both sampling periods, arsenic and thallium were below the detection limits of 0.0005 and 0.0008 mg/dscm (7% O<sub>2</sub>, dry basis), respectively, while beryllium was below the detection limit for the first run only. As can be seen from Table 3 and Figure 5, mercury emissions averaged 0.0066 mg/dscm (7% O<sub>2</sub>, dry basis) for the two sampling periods, which is approximately 92% lower than the NSPS for MWC of 0.08 mg/dscm. Cadmium emissions averaged 0.0017 mg/dscm (7% O<sub>2</sub>, dry basis), which is 83% lower than the NSPS for MWC of 0.01 mg/dscm. Lead emissions averaged 0.0346 mg/dscm (7% O<sub>2</sub>, dry basis) for the two sampling periods, which is over 65% lower than the NSPS for MWC of 0.1 mg/dscm.

Table 4 includes the average emissions of NO<sub>x</sub>, CO, SO<sub>2</sub>, and HCl across both sampling periods (approximately 4 hours) and compares them to the recently promulgated NSPS for MWC. CO emissions averaged 59 ppmv (7% O<sub>2</sub>, dry basis) with a standard deviation of 22 ppmv. These CO emissions were

comparable to the NSPS for MWC of 50-150 ppmv (7% O<sub>2</sub>, dry basis), but were significantly higher than previous reported combustion experiments by EnerTech with the carbonized RDF slurry fuel.<sup>8</sup> The higher CO emissions likely were due to lower flame temperatures caused by the higher water content of the slurry fuel, and poorer atomization. Please note that CO emissions from the carbonized RDF slurry fuel were achieved with excess air levels of approximately 20%.

NO<sub>x</sub> emissions averaged 132 ppmv (7% O<sub>2</sub>, dry basis) with a standard deviation of 16 ppmv. These NO<sub>x</sub> emissions were approximately 12% lower than the NSPS for MWC, without any catalytic or non-catalytic reduction. The NSPS for MWC of 150 ppmv (7% O<sub>2</sub>, dry basis) is based upon selective non-catalytic reduction NO<sub>x</sub> control.

HCl emissions from combustion of the carbonized RDF slurry fuel averaged 6 ppmv (7% O<sub>2</sub>, dry basis) with a standard deviation of 2 ppmv. Compared to the NSPS for MWC, HCl emissions were 75% lower, without any acid gas scrubbing. Please note that the NSPS for MWC is based upon acid scrubbing for HCl emission control. Based upon the carbonized RDF chloride content, the theoretical HCl emission concentration was 40 ppmv (7% O<sub>2</sub>, dry basis) assuming complete conversion of chloride to HCl. This indicates that approximately 85% of the fuel chloride was not converted to HCl or was removed from the flue gas through capture by alkaline ash components.

SO<sub>2</sub> emissions averaged 71 ppmv (7% O<sub>2</sub>, dry basis) with a standard deviation of 8 ppmv, or approximately 0.13 lbs SO<sub>2</sub>/10<sup>6</sup> Btu. The theoretical emission concentration was 80 ppmv (7% O<sub>2</sub>, dry basis), based on the carbonized RDF slurry fuel sulfur content and assuming complete conversion of fuel sulfur to SO<sub>2</sub>. This indicates that approximately 12% of the fuel sulfur was captured by ash mineral matter or not converted to SO<sub>2</sub>. SO<sub>2</sub> emissions were higher than the NSPS for MWC of 30 ppmv, and likely would require approximately 60% reduction through sorbent injection to comply with this standard. However, when compared to coal, the carbonized RDF slurry fuel would be considered a premium ultra-low sulfur fuel, even under Phase II of the Clean Air Act Amendments of 1990 (CAAA). The CAAA Phase II SO<sub>2</sub> emission limit for coal boilers will be approximately 1.2 lbs/10<sup>6</sup> Btu. Under the CAAA, up to 30% of the coal boiler's input could be the carbonized RDF, and SO<sub>2</sub> emissions would be regulated under the 1.2 lbs/10<sup>6</sup> Btu standard.

Emissions of CO, NO<sub>x</sub>, and PCDD/PCDF are not only dependent on fuel composition but also burner design and furnace operating conditions, and comparison to full scale boiler performance requires application of scaling parameters on a boiler specific basis. The CO and NO<sub>x</sub> emissions from combustion of the carbonized RDF slurry fuel were near the low end range of CO and NO<sub>x</sub> emissions measured for several dozen coals at EER's BSF and should be similar to commercial scale performance of pulverized coal boilers. In general, SO<sub>2</sub> and HCl from EER's BSF are indicative of commercial scale performance and scaling factors are not needed. Insufficient data exists for PCDD/PCDF emissions from coal combustion in EER's BSF to make any inferences about necessary scaling factors and commercial scale performance.

The filter on the EPA Method 23 train was used to measure fly ash loading of the combustion flue gases. During the two sampling periods, the average particulate matter (PM) loading of the exit flue gases from the ESP was 31.2 mg/dscm (7% O<sub>2</sub>, dry basis). Compared to the NSPS for MWC, the PM emissions would have to be reduced approximately 50% to comply with the standard of 15 mg/dscm. However, this is easily accomplished through a higher collection efficiency with the ESP. Based upon ash content of the carbonized RDF slurry fuel, approximately 99.6% of the PM was removed by dropout in the BSF and by the ESP.



A high volume sampler with a full page filter was used to obtain an ash sample upstream of the ESP for carbon burnout analysis. Through laboratory analysis, carbon content of the fly ash was determined to be 0.46 wt.% (dry basis). This corresponds to 99.92% carbon burnout, which is extremely good. This carbon conversion also is similar to or slightly better than carbon burnout levels typically obtained with pulverized coal firing.

## CONCLUSIONS

The carbonized RDF slurry fuel combusted exceptionally well, under conditions which were typical of a modern pulverized coal or oil boiler. Overall, the emissions from combustion of the carbonized RDF slurry fuel ranged significantly lower to comparable to the recently promulgated NSPS for MWC, especially total PCDD/PCDF emissions. Total PCDD/PCDF emissions were approximately 0.38 ng/dscm (7% O<sub>2</sub>, dry basis) or  $236 \times 10^{-6}$  lbs/10<sup>12</sup> Btu, which were 97% lower than the promulgated NSPS.

As a comparison, dioxin and furan emissions recently have been characterized from a number of coal boilers. The emissions from nine of these coal boilers have been summarized in a recent report to the DOE.<sup>9</sup> In general, the reported concentrations for individual isomers and total PCDDs/PCDFs were at the detection limit values or invalidated by a blank detection at similar concentrations. As an example, the total PCDD/PCDF emissions were determined using EPA Method 23 across three days of stack sampling from Cooperative Power Association's 1,100 MW Coal Creek Station Unit No.1 in Underwood, North Dakota, firing 100% lignite coal from the Falkirk mine. Fourteen out of seventeen of the individual isomers were below detection limits for all three sampling periods. An emission factor then was estimated as a summation of half the detection limit plus the concentrations all isomers detected. Based upon this calculation procedure, the total PCDD/PCDF emissions from the Coal Creek Station were estimated to be approximately  $38 \times 10^{-6}$  lbs/10<sup>12</sup> Btu.<sup>10</sup> These coal-fired dioxin/furan emissions were the same order of magnitude for dioxin/furan emissions from combustion of the carbonized RDF slurry fuel. However, the variety of coal types and boiler configurations is immense and emission data of total PCDD/PCDF limited, so that any conclusions or comparisons should be interpreted cautiously.

The emissions of several trace metals from the nine coal boilers also were summarized in the previously cited report to the DOE.<sup>9</sup> Unlike PCDD/PCDF emissions from coal boilers, the majority of individual trace metal emissions were higher than the detection limits and could be quantified. Table 5 compares the range of trace metal emissions from the nine coal boilers to the carbonized RDF slurry trace metal emissions. For the majority of the characterized metals, the carbonized RDF slurry fuel emissions were near the low to middle range of the coal boiler emissions, except for antimony and lead which were near or over the high range. It is noteworthy that the coal boiler trace metal emissions generally vary by one to four orders of magnitude, and again any conclusions or comparisons should be interpreted cautiously.

The results presented in this paper and previously cited papers by EnerTech formed the basis for the design and construction of a 20 ton/day (as received MSW basis) demonstration facility of wet resource recovery and SlurryCarb<sup>TM</sup>. The demonstration facility was designed and constructed in Japan by a consortium of Japanese companies and began operation in February 1997. Future technical papers will discuss this facility.

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**TABLES**

	<b>Reno, Nv. Wet RR RDF<sup>a,b</sup></b>	<b>Thief River Falls, Mn. RDF</b>	<b>Carbonized RDF Slurry Fuel</b>
<b>Ultimate, wt.%, dry</b>			
Carbon	52.3	45.7	68.4
Hydrogen	7.9	5.6	7.7
Nitrogen	0.9	0.9	0.4
Sulfur	0.3	0.2	0.1
Oxygen (difference)	32.0	39.2	11.4
Ash	6.6	8.4	12.0
<b>Chlorine, µg/g, dry</b>	4,100	3,000	700
<b>Mercury, µg/g, dry</b>	N.D.	0.30	0.09
<b>Cadmium, µg/g, dry</b>	N.D.	1.28	0.50
<b>Lead, µg/g, dry</b>	N.D.	29.8	82.0
<b>Rheology (Sheared)</b>			
Viscosity, cP @ 100 Hz↓	500	500	340
Solids Load., wt.%	11.2	9.1	43.9
<b>Higher Heating Value</b>			
dry, Btu/lb	9,560	7,995	13,350
Slurry, Btu/lb	1,070	730	5,865

a) Wet RR = Wet based resource recovery process

b) Reno, Nevada MSW was processed with Biomass Systems, Inc. (San Leandro, Ca.) wet RR pilot plant

**Table 1 - Fuel Properties of Raw RDF and Carbonized RDF Slurry Fuel.**

	Sampling Period 1 (ng/dscm) <sup>a</sup>	Sampling Period 2 (ng/dscm) <sup>a</sup>	Average (ng/dscm) <sub>a</sub>
<b>Total TCDD</b>	<b>0.0838</b>	<b>0.0167</b>	<b>0.0502</b>
2378 TCDD	<0.0013	<0.0013	<0.0013
<b>Total PeCDD</b>	<b>0.0603</b>	<b>0.0116</b>	<b>0.0360</b>
12378 PeCDD	<0.0017	<0.0017	<0.0017
<b>Total HxCDD</b>	<b>0.0540</b>	<b>0.0273</b>	<b>0.0407</b>
123478 HxCDD	0.0024	<0.0014	0.0018
123678 HxCDD	0.0041	0.0029	0.0035
123789 HxCDD	0.0031	0.0022	0.0027
<b>Total HpCDD</b>	<b>0.0238</b>	<b>0.0233</b>	<b>0.0236</b>
1234678 HpCDD	0.0119	0.0127	0.0123
<b>OCDD</b>	<b>0.0343</b>	<b>0.0360</b>	<b>0.0352</b>
<b>Total PCDD</b>	<b>0.2562</b>	<b>0.1149</b>	<b>0.1857</b>
<b>Total TCDF</b>	<b>0.1050</b>	<b>0.0558</b>	<b>0.0804</b>
2378 TCDF	0.0044	0.0024	0.0034
<b>Total PeCDF</b>	<b>0.0365</b>	<b>0.0385</b>	<b>0.0375</b>
12378 PeCDF	0.0034	0.0044	0.0039
23478 PeCDF	0.0033	0.0039	0.0036
<b>Total HxCDF</b>	<b>0.0217</b>	<b>0.0314</b>	<b>0.0266</b>
123478 HxCDF	0.0050	0.0066	0.0058
123678 HxCDF	0.0028	0.0047	0.0037
123789 HxCDF	<0.0019	0.0031	0.0025
234678 HxCDF	0.0026	0.0034	0.0030
<b>Total HpCDF</b>	<b>0.0182</b>	<b>0.0355</b>	<b>0.0269</b>
1234678 HpCDF	0.0084	0.0223	0.0153
1234789 HpCDF	0.0029	0.0061	0.0045
<b>OCDF</b>	<b>0.0112</b>	<b>0.0344</b>	<b>0.0228</b>
<b>Total PCDF</b>	<b>0.1926</b>	<b>0.1956</b>	<b>0.1942</b>
<b>Total PCDD/PCDF</b>	<b>0.4488</b>	<b>0.3105</b>	<b>0.3799</b>
<b>TEQ<sup>b,c</sup></b>	<b>0.0057</b>	<b>0.0063</b>	<b>0.0061</b>

a) All emissions corrected to 7% O<sub>2</sub>, dry basis

b) TEQ = Toxicity Equivalency of 2,3,7,8-TCDD (EPA Toxic Equivalent Factors - 1989)

c) Totals include half the detection limits for those compounds that were below detection limits

**Table 2 - Summary of PCDD/PCDF Emissions From Combustion of the Carbonized RDF Slurry Fuel.**

	Sampling Period 1 (mg/dscm) <sup>a</sup>	Sampling Period 2 (mg/dscm) <sup>a</sup>	Average (mg/dscm) <sub>a</sub>
Antimony (Sb)	0.0176	0.0304	0.0240
Arsenic (As)	<0.0005	<0.0005	0.0002
Barium (Ba)	0.0076	0.0120	0.0098
Beryllium (Be)	<0.0001	0.0001	0.0001
Cadmium (Cd)	0.0013	0.0021	0.0017
Total Chromium (Cr)	0.0041	0.0086	0.0064
Copper (Cu)	0.0162	0.0322	0.0242
Lead (Pb)	0.0250	0.0443	0.0346
Manganese (Mn)	0.0233	0.0241	0.0237
Mercury (Hg)	0.0060	0.0071	0.0066
Nickel (Ni)	0.0029	0.0045	0.0037
Selenium (Se)	0.0001	0.0001	0.0001
Thallium (Tl)	<0.0008	<0.0008	0.0004

a) All emissions corrected to 7% O<sub>2</sub>, dry basis

**Table 3 - Summary of Trace Metal Emissions From Combustion of the Carbonized RDF Slurry Fuel.**

	EnerTech		Municipal Waste Combustion	
	Carb. RDF Emissions <sup>a</sup>	Control Technolog y	1995 NSPS <sup>a,b</sup>	Required Control Technology
Carbon Dioxide (CO <sub>2</sub> ), vol. %	11.1	-----	-----	
Carbon Monoxide (CO), ppmv	59	None	50-150	Good Combustion Practice
Nitrogen Oxides (NO <sub>x</sub> ), ppmv	132	None	150	SCR or SNCR
Sulfur Dioxide (SO <sub>2</sub> ), ppmv	71	None	30	Acid Scrubber
Hydrochloric Acid (HCl), ppmv	6	None	25	Acid Scrubber
Total PCDD/PCDF, ng/dscm	0.38	None	13	Activated Carbon Injection
TEQ <sup>c</sup> , ng/dscm	0.0061	None	0.20 <sup>d</sup>	Activated Carbon Injection
Mercury (Hg), mg/dscm	0.0066	None	0.08	Activated Carbon Injection
Cadmium (Cd), mg/dscm	0.0017	ESP	0.01	Baghouse or ESP
Lead (Pb), mg/dscm	0.0346	ESP	0.1	Baghouse or ESP
Particulate Matter (PM), mg/dscm	31.2	ESP	15	Baghouse or ESP

a) All emissions corrected to 7% O<sub>2</sub>, dry basis

b) NSPS = New Source Performance Standards, published December 19, 1995 in the *Federal Register*

c) TEQ = Toxicity Equivalency of 2,3,7,8-TCDD (EPA Toxic Equivalent Factors - 1989)

d) Dioxins are not regulated by TEQ under the NSPS, and TEQ is presented for reference only

**Table 4 - Summary of Emissions From Combustion of the Carbonized RDF Slurry Fuel and Comparison to NSPS For MWC.**

	Carbonized RDF Slurry Fuel		Range From Coal Boilers <sup>9</sup>	
	Average (mg/dscm) <sup>a</sup>	Average (lb/10 <sup>12</sup> Btu)	Low (lb/10 <sup>12</sup> Btu)	High (lb/10 <sup>12</sup> Btu)
Antimony (Sb)	0.0240	20.22	0.04	2.4
Arsenic (As)	0.0002	0.22	0.14	3.5
Barium (Ba)	0.0098	8.25	N.A.	N.A.
Beryllium (Be)	0.0001	0.07	0.04	1.7
Cadmium (Cd)	0.0017	1.43	0.03	3.2
Total Chromium (Cr)	0.0064	5.37	0.13	51
Copper (Cu)	0.0242	20.39	N.A.	N.A.
Lead (Pb)	0.0346	29.17	0.53	29
Manganese (Mn)	0.0237	19.95	2.60	30
Mercury (Hg)	0.0066	5.53	0.44	22
Nickel (Ni)	0.0037	3.13	0.55	40.1
Selenium (Se)	0.0001	0.08	0.02	193
Thallium (Tl)	0.0004	0.34	N.A.	N.A.

a) All emissions corrected to 7% O<sub>2</sub>, dry basis

Table 5 - Comparison of Carbonized RDF Slurry Fuel and Coal Boiler Trace Metal Emissions.

FIGURES

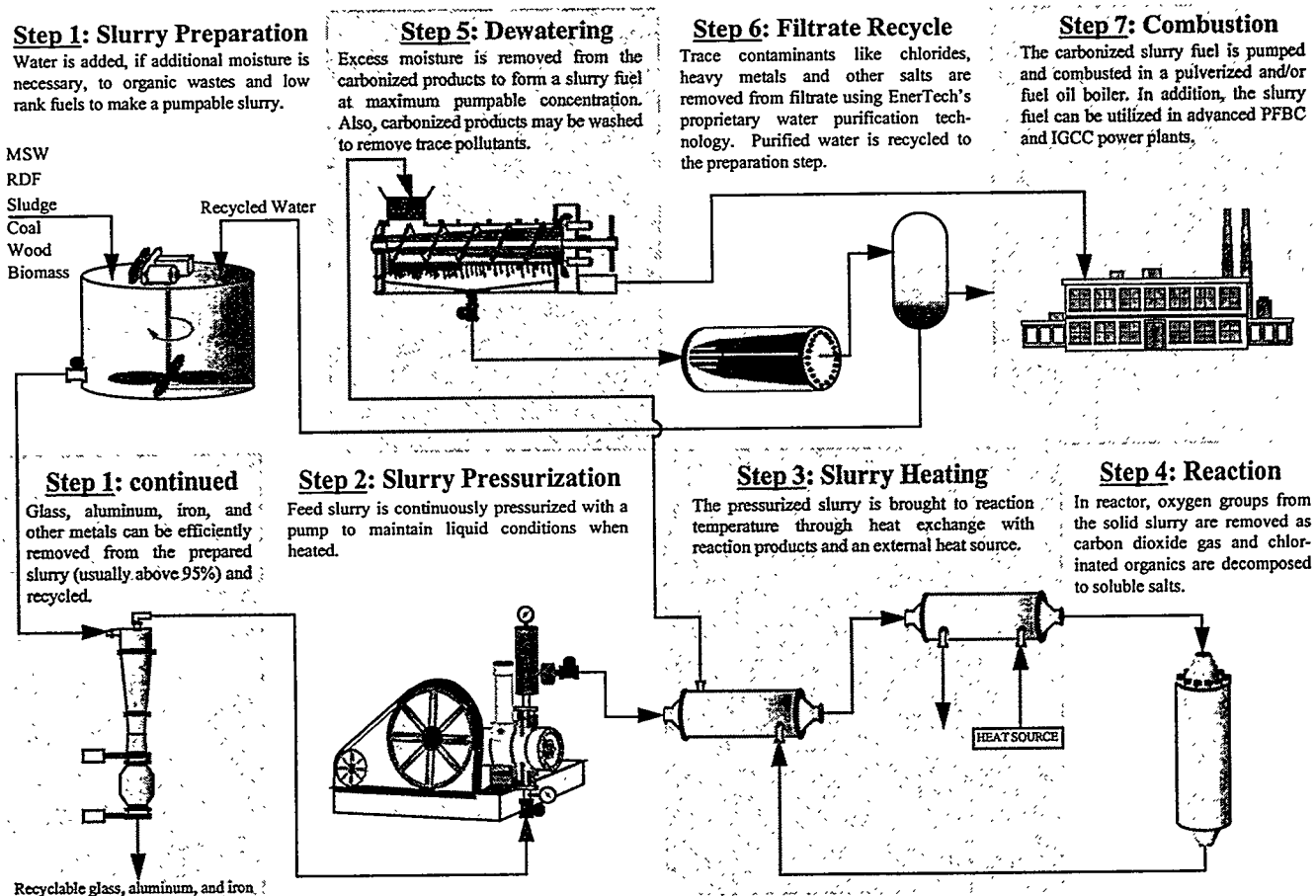


Figure 1 - Simplified Flow Diagram of EnerTech's SlurryCarb™ Process.

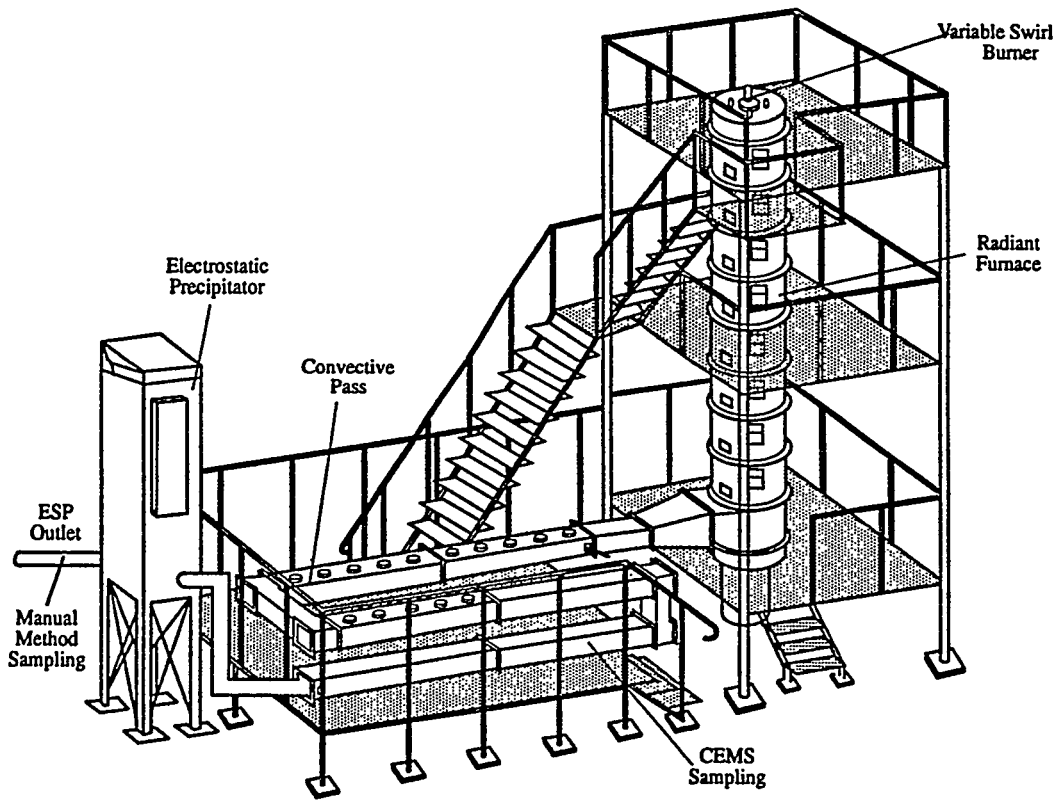


Figure 2 - Simplified Schematic of EER's 800,000 Btu/hr Boiler Simulation Facility (BSF).

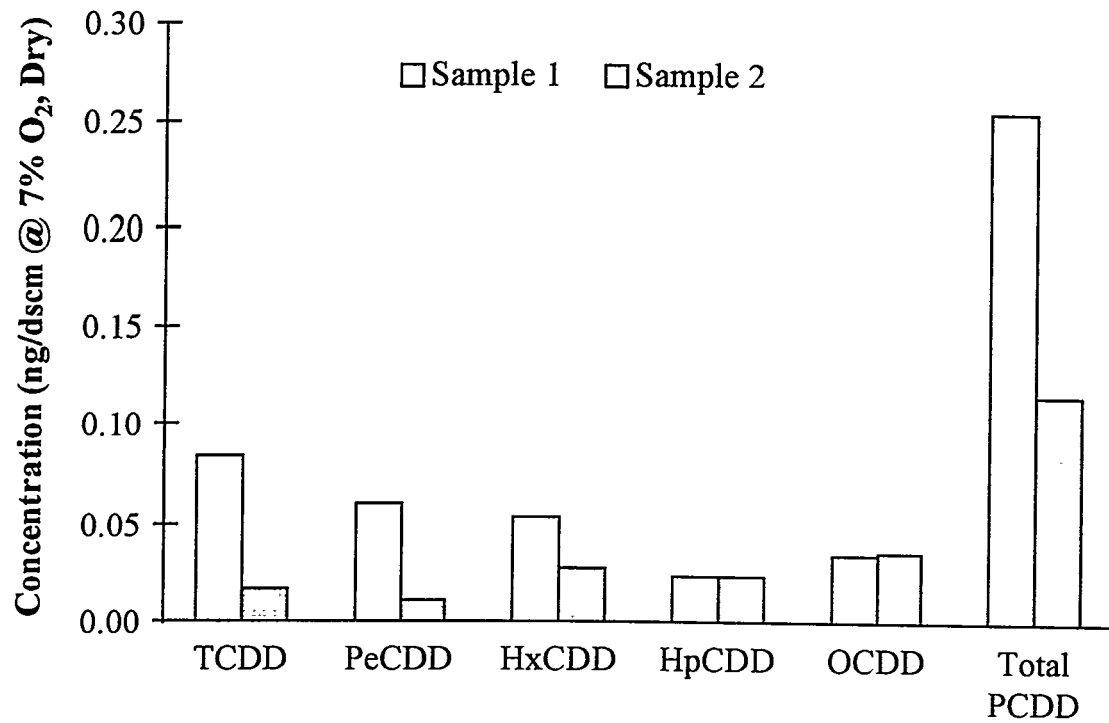


Figure 3 - Distribution of PCDD Congeners From Combustion of the Carbonized RDF Slurry Fuel.

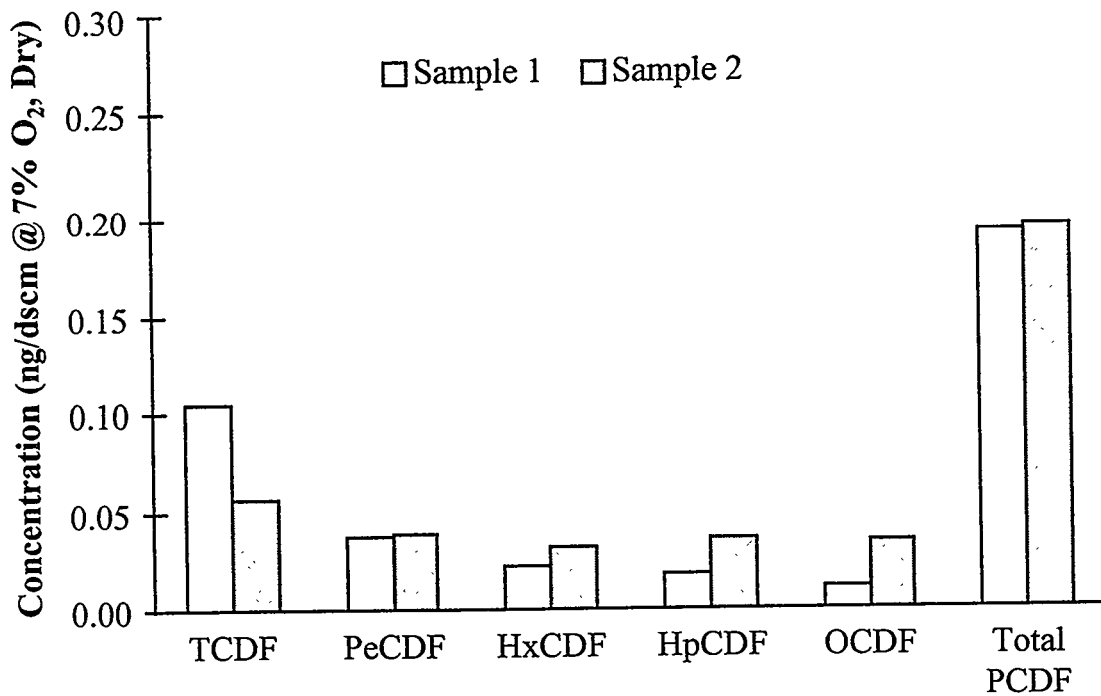


Figure 4 - Distribution of PCDF Congeners From Combustion of the Carbonized RDF Slurry Fuel.

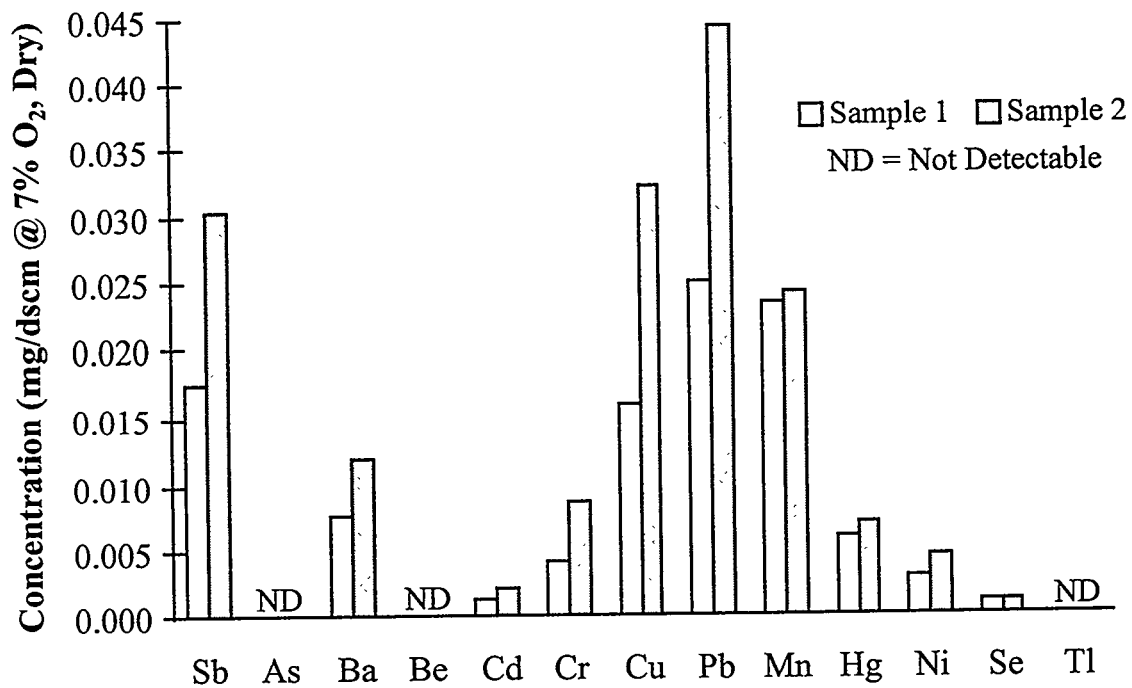


Figure 5 - Summary of Trace Metal Emissions From Combustion of the Carbonized RDF Slurry Fuel.



**Prevention of PCDD/PCDF Formation by Coal Co-Firing**

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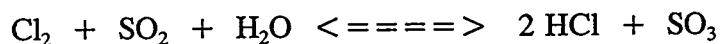
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## INTRODUCTION

Formation of polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) has been observed in waste combustors. However, PCDD and PCDF yields from coal combustion are relatively insignificant. Previous bench- and pilot-scale research<sup>1,2</sup> has shown that the presence of sulfur dioxide (SO<sub>2</sub>) can inhibit PCDD and PCDF formation, and suggested co-firing high-sulfur coal with refuse-derived fuel (RDF) to reduce the emissions. This work describes research in a large scale combustor which shows that coal/RDF co-firing can significantly lower PCDD and PCDF formation.

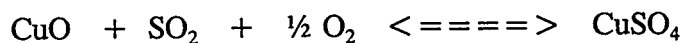
PCDD and PCDF formation mechanisms have been reviewed in detail by Addink and Olie.<sup>3</sup> Measured yields can be affected by the combustion quality, levels of chlorine and catalyst species, residual carbon on the fly ash, and process parameters such as temperature and residence time. Two candidate mechanisms have been established for the inhibition of PCDD and PCDF formation by sulfur species:

1) Gas-phase reaction where SO<sub>2</sub> converts chlorine (Cl<sub>2</sub>) to hydrogen chloride (HCl), a less-likely chlorinating agent<sup>4,2</sup>:



Our recent work<sup>2</sup> appears to confirm this mechanism as a possibility.

2) Poisoning of copper catalysts by SO<sub>2</sub><sup>1</sup>:



It has been shown that copper sulfate (CuSO<sub>4</sub>) is a less active catalyst for the production of Cl<sub>2</sub> through the Deacon process, as well as for the biaryl synthesis step of PCDD and PCDF formation.<sup>1</sup>

Lindbauer et al.<sup>5</sup> have reported that co-firing coal in a municipal solid waste (MSW) incinerator leads to appreciably lower PCDD and PCDF levels. Recently, Ogawa et al.<sup>6</sup> compared the effect of adding pure SO<sub>2</sub> versus generating SO<sub>2</sub> through coal addition, and found the latter to be more effective. Thus, there may be other benefits with coal co-firing than simply being a source of SO<sub>2</sub>.

In this work, experiments were conducted in a large, pilot-scale unit which is a state-of-the-art facility with fuel handling and combustion release rates representative of large field units. The effect of coal co-firing was studied for two types of municipal-waste-based solid fuels. Tests were conducted over a range of process variables (e.g., calcium hydroxide [Ca(OH)<sub>2</sub>] injection, HCl concentration, flue gas temperature, quench, and residence time) so that the results could have implications for a wide variety of waste combustors.

## EXPERIMENTAL FACILITY AND PROCEDURES

The newly constructed, EPA Multi-Fuel Combustor (MFC) facility was used for conducting tests for this project. The MFC is rated at 0.6 MW (2x10<sup>6</sup> Btu/hr) thermal output and is capable of burning a wide variety of solid fuels including MSW, RDF, biomass, and coal. The modular design of the

facility provides flexibility for studies on various pollutant emissions and control and for solid fuels with unknown firing and handling characteristics.

A schematic and a layout view of the MFC facility are shown in Figures 1 and 2, respectively. The MFC consists of a waste feeding system, a coal feeder, a lower combustion chamber containing a stoker, a radiant section, a convective flue gas passage, a baghouse, and a scrubber system. In addition, there is a separate fuel preparation system for shredding, screening, and mixing of the fuel. A large loading hopper conveys the processed fuel to the fuel silo.

As indicated in Figure 2, there are several ports for flue gas sampling and temperature measurements along the convective section and the duct. The convective section is equipped with cooling coils with high-pressure water circulation. The cooling coil originally present in location B was removed for this project, to accommodate PCDD/PCDF sampling. A typical temperature drop across the convective section is from 600 to 150°C which includes the PCDD/PCDF formation temperature "window" (200 to 500°C). Residence time/quench across this window is known to be an important parameter, and it can be varied in our tests by changing the temperature set point of the cooling water. From the convective section, flue gas was sampled through continuous emission monitors (CEMs) for recording the oxygen (O<sub>2</sub>), carbon dioxide (CO<sub>2</sub>), carbon monoxide (CO), HCl, and SO<sub>2</sub> concentrations.

The test program involved two major phases: In Phase I, the waste-derived fuel used was commercial RDF from municipal waste. The as-received RDF did contain some larger pieces and had to be run through the shredder in order to avoid clogging of the feed system. With this shredded fuel (simply referred to as "RDF" in this paper), the feed and burn could not be controlled sufficiently and the temperatures were lower than that observed in typical waste combustor operation. As a result, the flue gas O<sub>2</sub> and CO levels were higher than desired. Therefore, only a few tests were run with the RDF. To alleviate this problem, a commercial, densified refuse-derived fuel (dRDF), again from municipal waste, was acquired for subsequent testing under Phase II. The combustor operation was much improved with more uniform fuel feed and lower CO levels. In both phases, the coal used was an Illinois #6 coal, donated by Monterey Coal Company, Carlinville, IL, for this project. The coal was ground and classified to an average size of about 1 mm. The coal was fed using a screw feeder and entered the burner at the same location as the RDF/dRDF. Analyses of the dRDF and coal are shown in Table 1.

For Phase I, tests included separate RDF and coal baseline tests and a RDF/coal co-fired test. Under Phase II (dRDF), the experiments were structured to provide information on all two-factor interactions among the parameters of coal feed, dRDF feed rate, quench, sorbent injection, and HCl addition. During each test, the fuel feed rate was adjusted, whenever necessary, to maintain a constant flue gas temperature. Nearly the same firing rate was maintained between tests, again, by matching the flue gas temperature reading. For low fuel feed runs, natural gas was co-fired to compensate for the decrease in heat release and maintain similar temperature ranges between runs. Quench was varied by varying the temperature set point of the cooling water used in the convective section of the furnace. For runs with HCl addition, there was about a 100 ppm increase in HCl concentration. The sorbent used was a commercial hydrated lime, fed as a slurry at a Ca/Cl ratio of about 1.5. The sorbent injection temperature was approximately 700°C. Under Phase II, a total of 30 tests, including blanks, were run for various parameter settings.

For most of the runs, flue gas was sampled from two locations (A and B in Figure 2) for PCDD/PCDF according to EPA Method 23.<sup>7</sup> For selected runs, dioxin samples were drawn from an additional sampling port, just before the convective section (location D). Before and after running Method 23 trains, a velocity traverse of the duct was carried out to measure the flue gas flow rate. The samples were analyzed in EPA's in-house Organics Support Laboratory (OSL), via procedures described elsewhere.<sup>2</sup> Recently, the OSL has expanded its capability to include quantification of mono-tri PCDD and PCDF congener class mass as well. Thus, the results obtained are levels of each mono-octa PCDD and PCDF congener class in the sample. However, since mono-tri PCDD and PCDF are not subject to regulation, most of the following discussions are based on the tetra-octa PCDD/PCDF yields.

## RESULTS AND DISCUSSION

### Phase I tests with the RDF

The run conditions and sampling data are provided in Table 2. For the trial RDF burn (run 1), the temperatures were lower, resulting in high CO formation. Runs 2 and 4 involved better combustion conditions, although there were CO excursions exceeding 2000 ppm. Run 3 was a natural gas blank between runs 2 and 4, with no RDF or coal feed. Run 5 was a coal baseline and run 6 the coal/RDF co-fired test. Results from the above tests are shown in Figure 3 as tetra-octa total PCDD+PCDF yield.

Substantial levels of PCDD and PCDF are measured at both sampling locations (A, B) for the RDF combustion. In fact, the difference in yields between the two locations is small. Thus, much of the formation takes place in-flight, within seconds from the burner. It is clear that an effective control technology should discourage this formation, before the flue gas and particles reach the particulate control device. Combustion quality appears to play an important role in PCDD and PCDF formation: run 1 (high CO and lower temperature) yields are much higher than those from run 2 or run 4. Given the variations in the run conditions and the nature of PCDD/PCDF formation, the difference between the two RDF baseline run (runs 2 and 4) yields is small, rendering credibility to the data obtained. Also, as with field units, yields of PCDF are higher than those of PCDD in these tests. However, the CO levels in our RDF baseline tests are higher than in typical municipal waste combustors (MWCs).

Contrary to the RDF, no PCDD or PCDF was detected from coal burning (run 5). Even the natural gas blank test (run 3) yielded PCDD/PCDF, although at much lower levels compared to RDF. Although the system was blown with a high flow of air prior to each test, some particle deposition on the system walls from earlier testing is inevitable, and could cause measurable yields even with natural gas. However, the coal baseline test showed no such residual or hysteresis effect. It appears that flue gas from coal burning, presumably SO<sub>2</sub>, strongly inhibits formation.

The addition of coal to RDF in the co-fired test provided better burn conditions -- increased temperatures and lower CO. In this test (run 6), the resulting sulfur-to-chlorine (S/Cl) ratio was about 1.5. Under these conditions, the PCDD and PCDF yields decreased sharply from the RDF baseline levels. Past work at the EPA facilities<sup>2</sup> had suggested a S/Cl ratio of above 1.2 for effective PCDD/PCDF emission control. Thus, Phase I tests indicate that co-firing RDF with high-sulfur coal is an effective option for MWCs to inhibit PCDD and PCDF formation.

## Phase II tests with the dRDF

For each test, temperature and flue gas composition data were stored in the data acquisition system. Run averages of these data were calculated for the duration of PCDD/PCDF sampling so that these values correspond to the measured yields. With the total flue gas flow rate known from the velocity traverse data, the average flue gas temperature profile data were used to determine the flue gas residence times at various locations in the MFC. In the temperature profiles, the residence time variable,  $t_R$ , is set to zero at 650°C. The choice of 650°C stems from the fact that most of PCDD/PCDF formation is known to take place below this temperature. In this project, for three runs, PCDD/PCDF levels were measured for the high-temperature (~ 600°C) dioxin sampling port (Port D), and the yields were relatively small.

Ranges of measured/estimated parameter values for Phase II tests are given in Table 3. The gas composition data show low CO levels from dRDF burning, indicating good combustion quality. In general, yields from dRDF combustion were lower compared to RDF combustion. The possible reasons are that, with the former, the HCl concentration in the flue gas is lower and the combustion quality is better (lower CO).

For one of the dRDF baseline tests (without coal), PCDD/PCDF levels were measured at three different sampling ports and the results are shown in Figure 4. The figure shows the total PCDD and PCDF yields for tetra-octa as well as for mono-octa congener classes. The PCDD/PCDF levels measured at the high-temperature port are relatively low, and the majority of the formation takes place between Ports D and B, in less than 0.5 sec. An examination of the temperature profile, shown in the same figure, indicates that in this region, the temperature decreases sharply. A more gradual temperature decline would result in larger residence times in this temperature window, possibly allowing more PCDD and PCDF formation. After Port B, the change in PCDD and PCDF levels is small. The figure also shows that the yields of mono-tetra PCDD and PCDF are significant, and it is possible that under different process conditions some of them might shift to the toxic, higher chlorinated congeners.

An average analysis of all the experimental data has been performed. The data are divided into three groups, dRDF alone, dRDF with coal co-firing, and dRDF with sorbent injection, although within a group other parameters such as HCl concentration and quench may vary. For each group, average PCDD and PCDF yield is computed for each congener class. Results are plotted for mono-octa PCDD and PCDF congener classes in Figure 5. Both coal co-firing and sorbent injection decrease PCDD and PCDF formation significantly; the congener class pattern is similar to that of the base dRDF case and, therefore, the reduction is not congener-class-specific. The total tetra-octa PCDD and PCDF yields for the three groups are plotted in Figure 6. The results clearly indicate that coal/dRDF co-firing as well as calcium-based sorbent injection reduces PCDD and PCDF formation.

Experimental results have been analyzed statistically for Port A and Port B data separately. The model results for both cases are similar and the model  $R^2$  for the data for each port is 0.76. Among the five exogenous variables (labeled COAL, RDF, QUENCH, SORBENT, and HCL), four 2-factor interactions involving all five variables were found to be significant: COAL-HCL, RDF-QUENCH, QUENCH-SORBENT, and SORBENT-HCL. Thus, model predictions for coal co-firing are available. Model predictions are shown in Figure 7 for Port A data, and similar results are obtained (but not shown) for Port B data. The model predicts that an increase in HCl concentration increases

the PCDD and PCDF yield for dRDF combustion. More importantly, at both low and high HCl levels, the model indicates that coal co-firing reduces PCDD and PCDF formation significantly. More detailed modeling efforts are in progress, and the results will be available later.

## **CONCLUSIONS AND RECOMMENDATIONS**

For commercial dRDF combusted in our 0.6 MW stoker combustor, a majority of the PCDD and PCDF formation took place between 600 and 300°C within about 0.5 sec.

Based on Phase I tests, improving the combustion quality appears to lower PCDD and PCDF yields.

Co-firing Illinois #6 coal with both the RDF and dRDF reduced PCDD and PCDF formation substantially. This reduction appears to be uniform across the entire congener range. Injection of  $\text{Ca}(\text{OH})_2$  injection is nearly as effective as coal co-firing.

Co-firing coal with waste-derived fuel is a candidate technology for reducing PCDD and PCDF emissions in commercial waste burning facilities. Further development on a full-scale facility is needed to demonstrate field applicability.

## **ACKNOWLEDGMENTS**

This work was also co-sponsored by Illinois Clean Coal Institute (ICCI) (Project Manager: Ken Ho) and the National Renewable Energy Laboratory (NREL) (Project Monitor: Phil Shepherd). Logistical support from Richard Valentine (U.S. EPA/APPCD) is greatly appreciated. Suh Lee, Joey Valenti, Russell Logan, Scott Moore, John Foley, Dennis Tabor, and Ann Preston (Acurex Environmental Corporation), and Jeff Ryan (U.S. EPA/APPCD) provided extensive technical, sampling, and analytical assistance.

## **DISCLAIMER STATEMENT**

This work was sponsored in part by the Illinois Department of Energy and Natural Resources (IDENR) through its Coal Development Board and Illinois Clean Coal Institute (ICCI). However, any opinions, findings, conclusions, or recommendations expressed herein do not necessarily reflect the view of IDENR and the ICCI.

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6. H. Ogawa, N. Orita, M. Horaguchi, T. Suzuki, M. Okada and S. Yasuda, "Dioxin Reduction by Sulfur Component Addition," *Chemosphere*, 32, 151 (1996).
7. Method 23, *Title 40 Code of Federal Regulations*, Part 60, Appendix A, U.S. Government Printing Office, Washington, DC (1991).



Table 1. Analyses of the densified refuse-derived fuel (dRDF) and Illinois #6 coal

	dRDF	coal
<b>Proximate analysis (%)</b>		
Moisture	7.83	14.96
Ash	8.71	8.32
Volatile matter	70.62	33.51
Fixed Carbon	12.84	43.21
<b>Ultimate analysis (%)</b>		
Moisture	7.83	14.96
Carbon	41.70	60.25
Hydrogen	5.58	4.22
Nitrogen	0.80	1.09
Sulfur	0.09	3.36
Ash	8.71	8.32
Oxygen (by difference)	35.29	7.80
Chlorine	0.14	0.09
Calcium	1.35	n.m.
Copper	0.002	n.m.
kJ/kg (Btu/lb)	16,408 (7,059)	25,257 (10,866)

n.m. not measured

Table 2. Multi-Fuel Combustor test conditions for Phase I tests with the RDF

Run No.	fuel	run type	flue gas conditions					Port B (conv. section)		Port A (pre-baghouse)	
			O <sub>2</sub> %	CO <sub>2</sub> %	CO ppm	HCl ppm	SO <sub>2</sub> ppm	Ts °C	D+F <sup>1</sup>	Ts °C	D+F <sup>1</sup>
1	RDF	trial burn	16.5	3.6	1345	n.m.	n.m.	250	1253	134	1340
2	RDF	baseline	14.8	5.3	201	n.m.	n.m.	302	716	153	798
3	natural gas	blank	13.7	2.7	3	n.m.	n.m.	260	46	117	95
4	RDF	baseline	12.8	6.2	690	n.m.	n.m.	239	497	124	630
5	coal	baseline	17.8	2.3	86	n.m.	n.m.	285	n.d.	151	n.d.
6	RDF & coal	co-fired	17.1	3.9	41	321	492	341	180	180	26

<sup>1</sup>tetra-octa PCDD+PCDF (ng/dscm)

All concentrations are dry and not corrected to 7% O<sub>2</sub>

Ts sampling temperature

n.m. not measured

n.d. none detected

Table 3. Ranges of test conditions and results for the Phase II MFC tests

Parameter	parameter range	
	dRDF runs	coal/dRDF runs
O <sub>2</sub> (%)	11.5 - 14.4	9.3 - 16.9
CO <sub>2</sub> (%)	4.5 - 8.1	4.8 - 10.0
CO (ppm)	0 - 28	2 - 97
HCl (ppm)	11 - 224	21 - 193
SO <sub>2</sub> (ppm)	3 - 17	136 - 427
<b>Port A</b>		
t <sub>R</sub> (sec)	1.90 - 3.06	0.43 - 1.58
T <sub>s</sub> (°C)	151 - 192	146 - 196
tetra-octa PCDD + PCDF (ng/dscm)	1 - 1369 <sup>1</sup>	4 - 57
<b>Port B</b>		
t <sub>R</sub> (sec)	1.66 - 3.69	0.37 - 1.94
T <sub>s</sub> (°C)	296 - 380	299 - 370
tetra-octa PCDD + PCDF (ng/dscm)	9 - 1910 <sup>1</sup>	2 - 104

<sup>1</sup>not including sorbent injection runs

All concentrations are dry, not corrected to 7% O<sub>2</sub>.

Residence time t<sub>R</sub> based on t=0 at 650°C.

T<sub>s</sub>=Sampling port temperature.

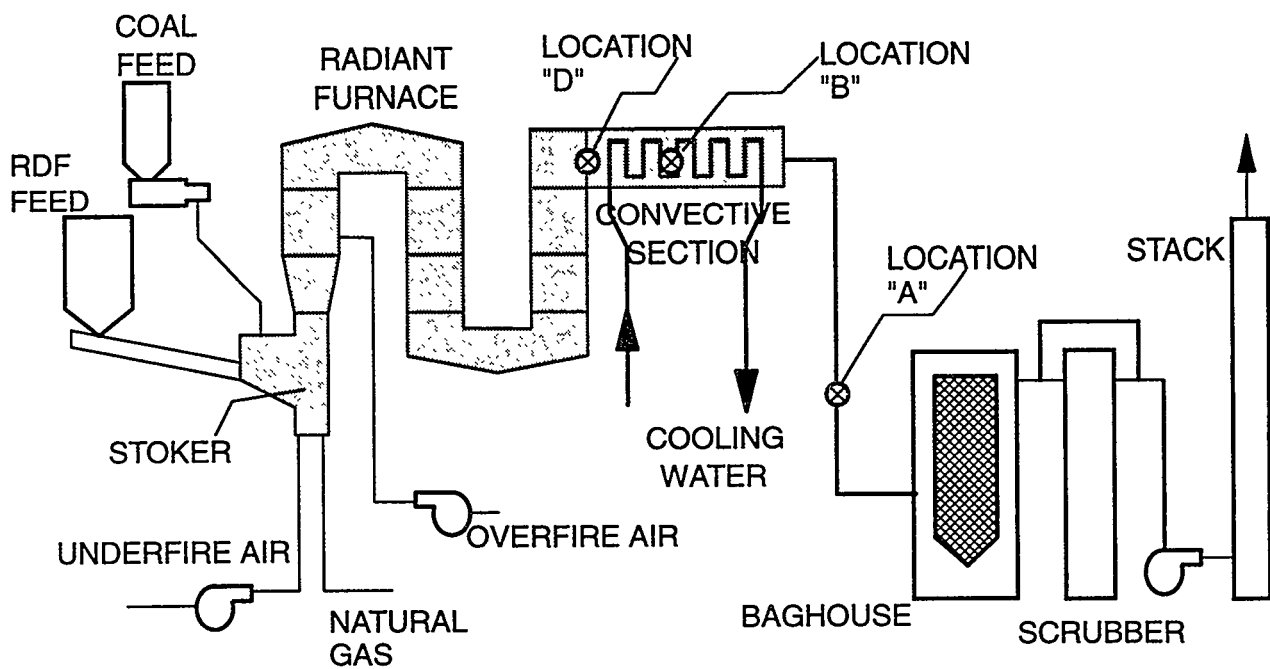


Figure 1. Schematic of the Multi-Fuel Combustor (MFC)

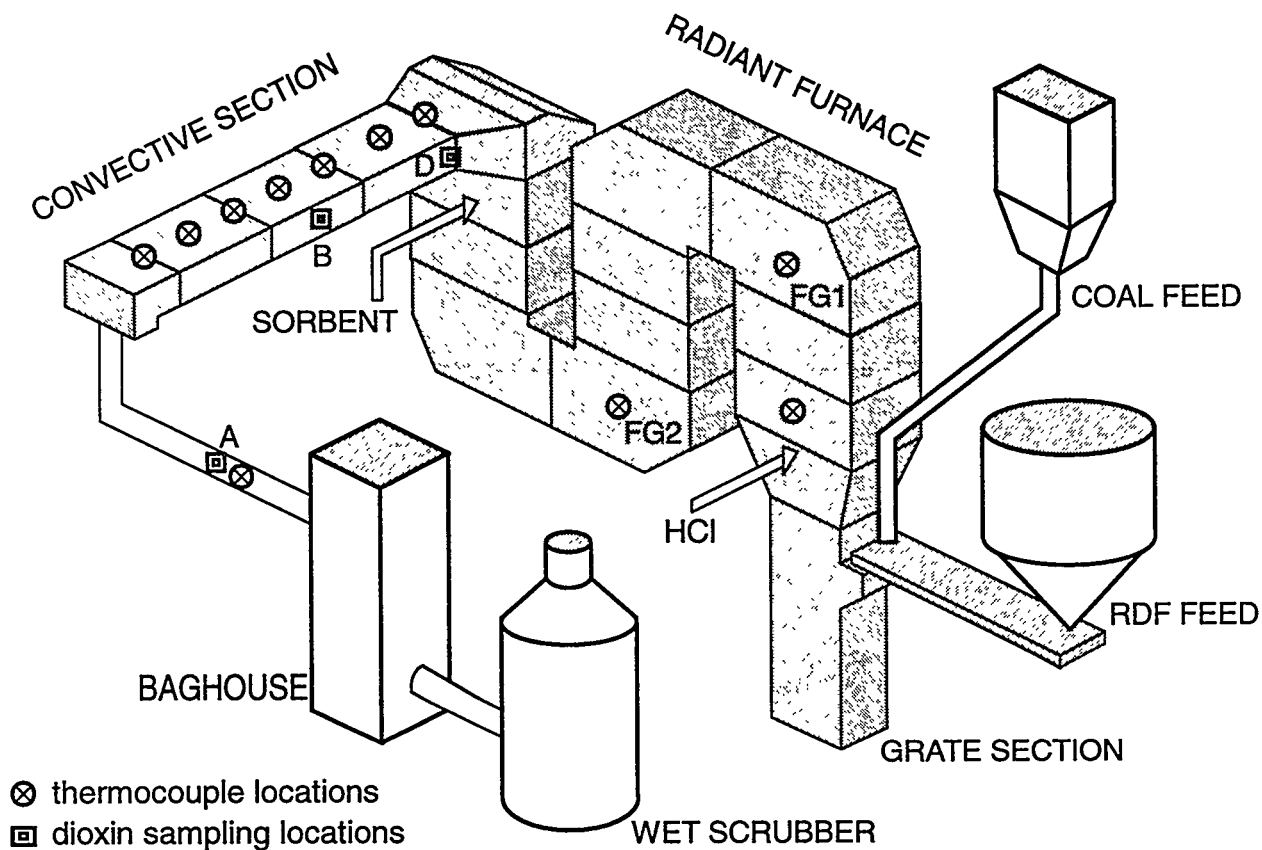


Figure 2. A view of the Multi-Fuel Combustor

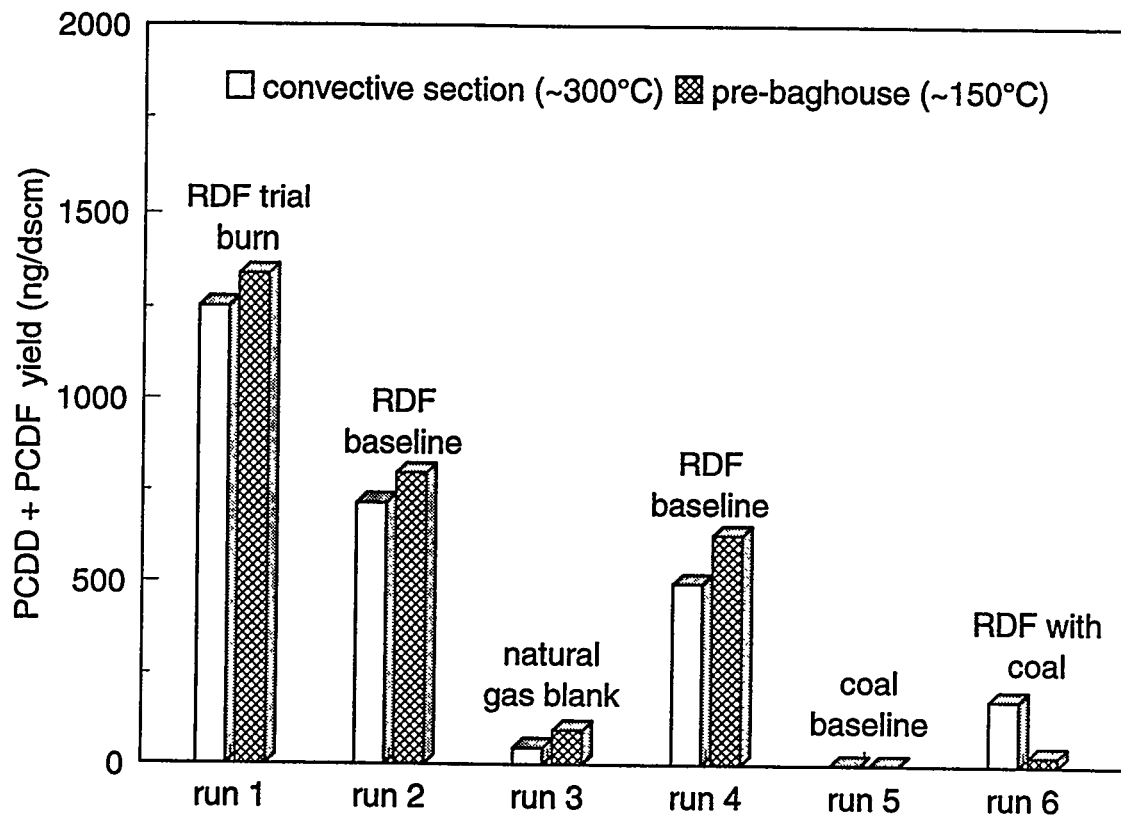
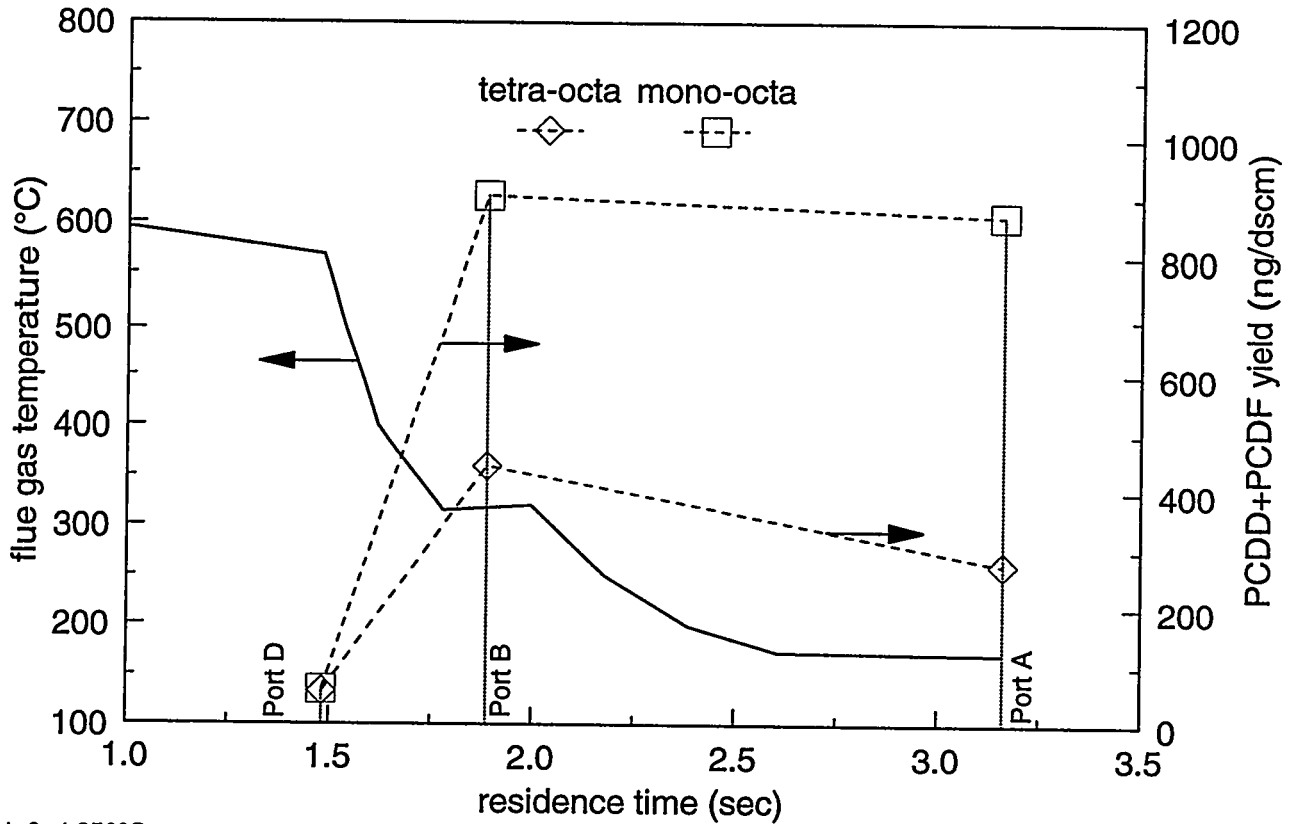


Figure 3. Total PCDD and PCDF (tetra-octa) yields from Phase I RDF tests



t=0 at 650°C

Figure 4. PCDD and PCDF yields from dRDF combustion measured at the three sampling ports

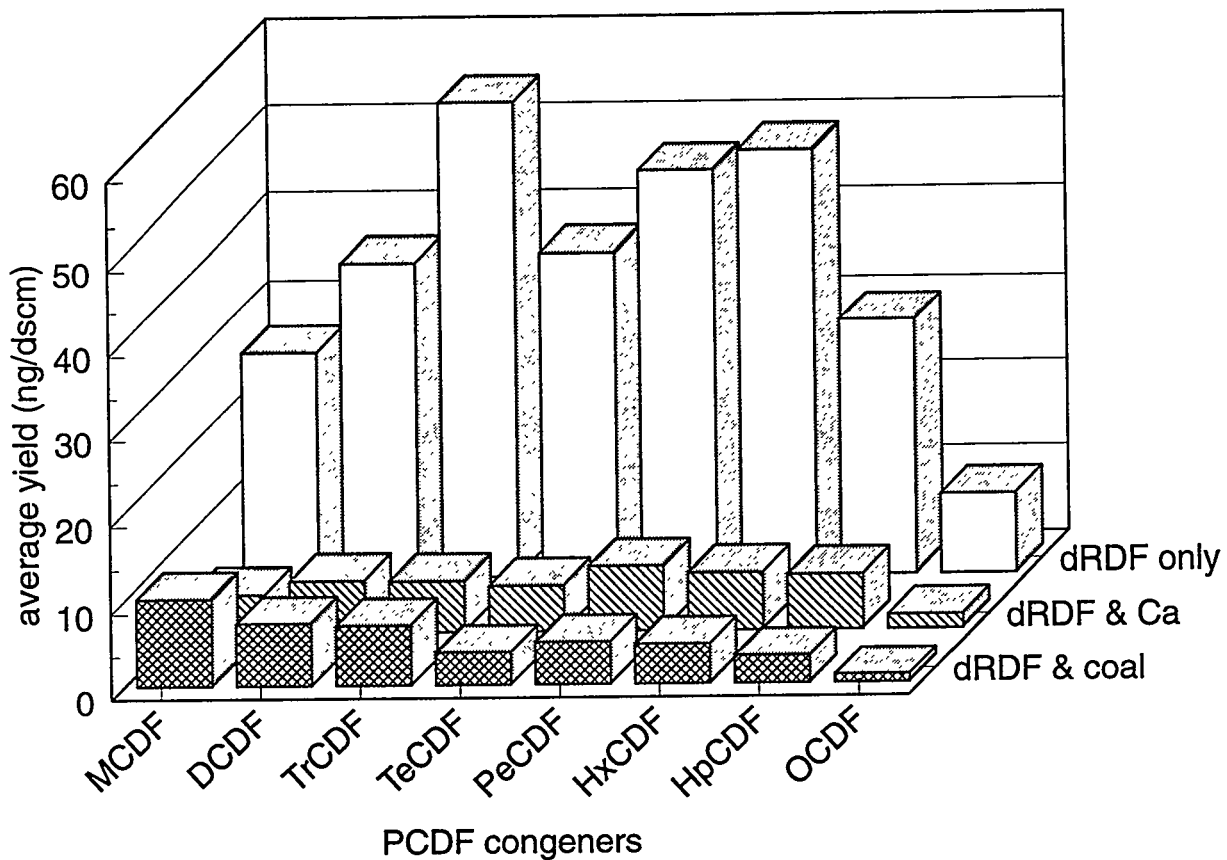
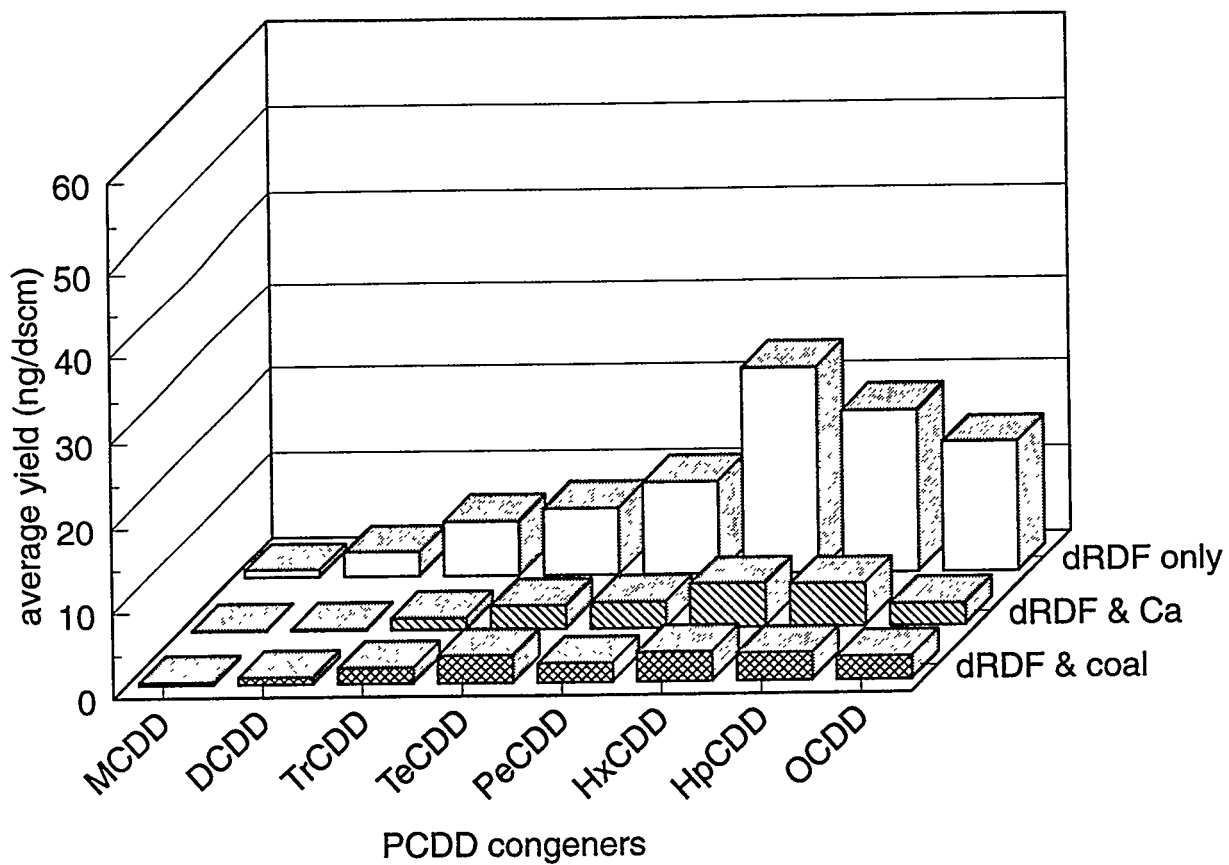


Figure 5. Effects of coal co-firing and sorbent injection on PCDD and PCDF congener yield.

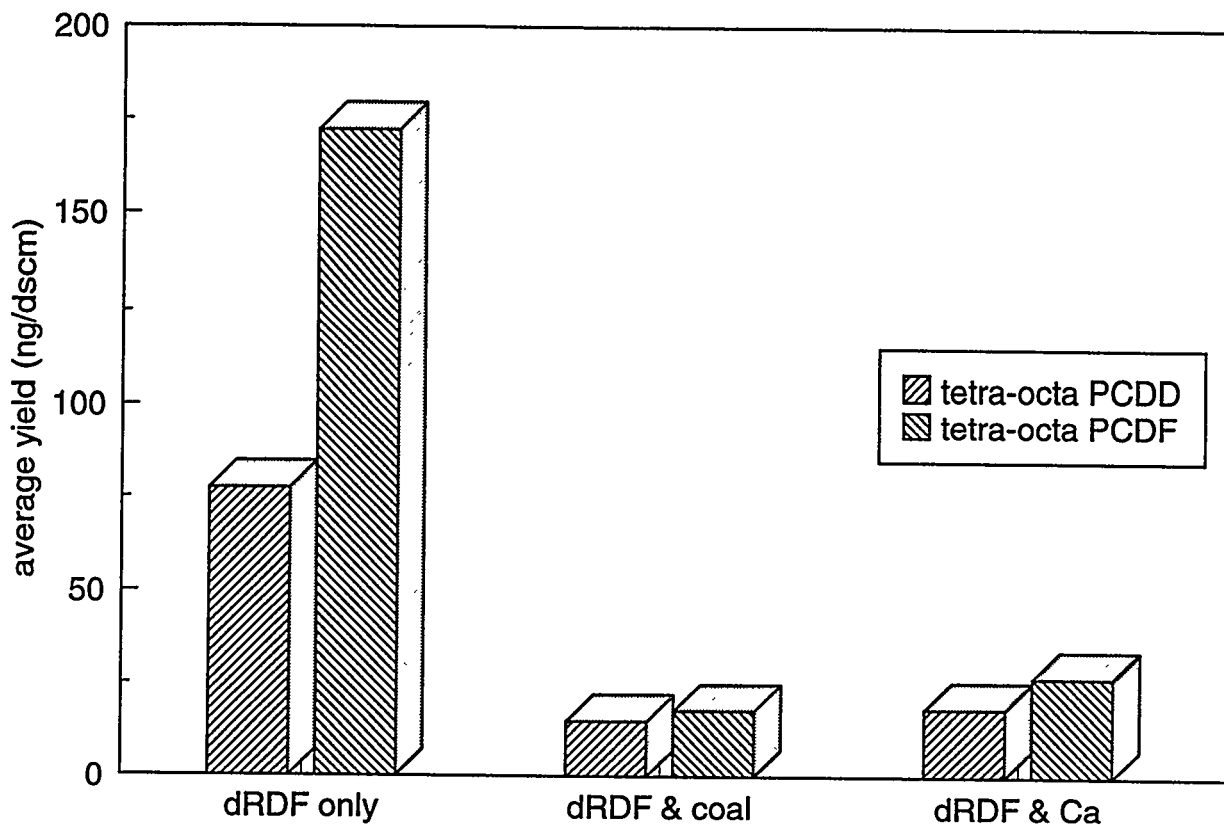


Figure 6. Effects of coal co-firing and sorbent injection on total PCDD and PCDF yield.

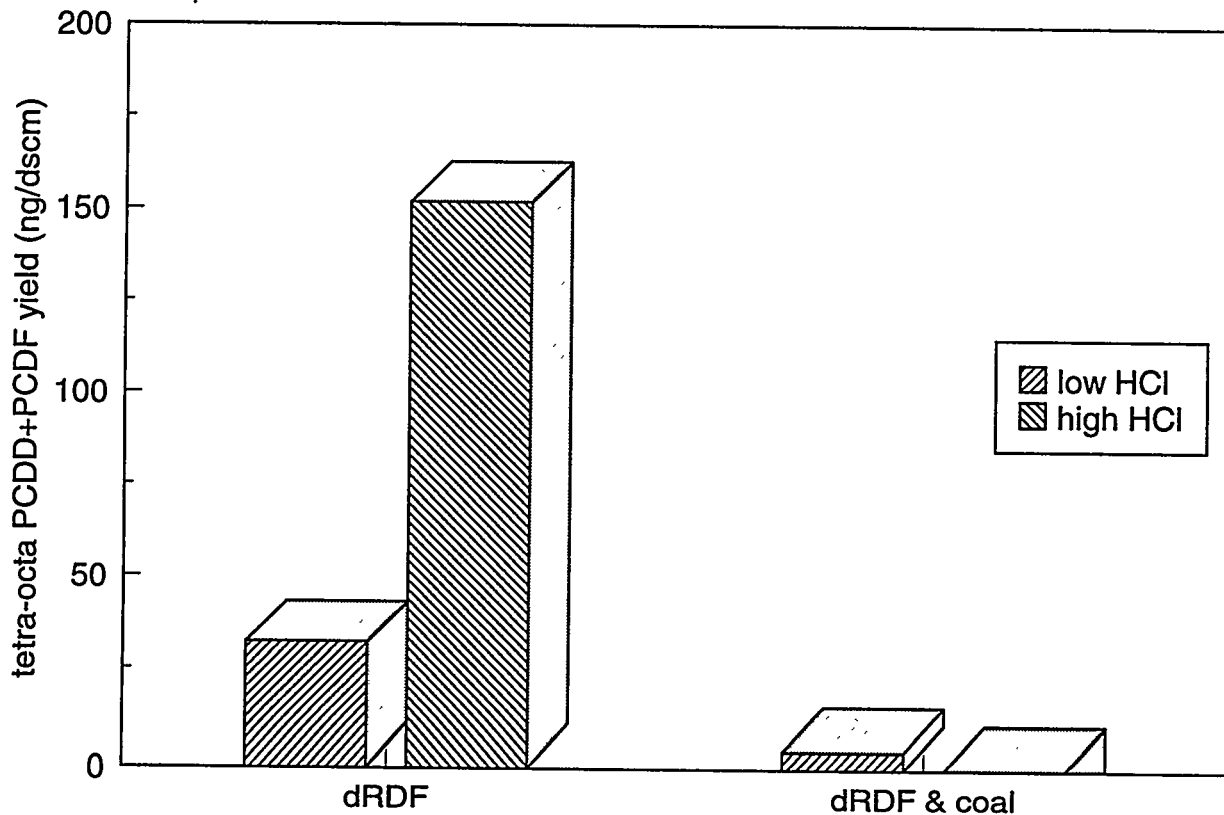


Figure 7. Effect of coal co-firing on PCDD+PCDF yield from the model for Port A





**An Experimental Study on Mercury Sorption by  
Activated Carbons and Calcium Hydroxide**

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## INTRODUCTION

Title III of the 1990 Clean Air Act Amendments (CAAA) requires the U.S. Environmental Protection Agency (EPA) to submit a study on 189 hazardous air pollutants (HAPs) from industrial sources. This study will include an emission and risk (to public health) assessment of the HAPs. Among the 189 HAPs, mercury has drawn special attention due to its increased levels in the environment and well-documented food chain transport and bioaccumulation.<sup>1,2</sup>

Mercury, present in hazardous/municipal wastes and in coal<sup>3</sup>, is readily volatilized during combustion and incineration processes.<sup>4</sup> Mercury is the most volatile among various trace metals, and major portions of it can pass through existing particulate control devices.<sup>4</sup> A sorbent that can react with mercury can effectively shift this metal from the vapor phase to the particulate (sorbent) phase, facilitating its removal. Mercury control processes which use adsorption on dry sorbents do not pose the problem of treatment and stabilization of the waste liquid stream and, therefore, seem very attractive for coal combustors and hazardous/municipal waste incinerators. The need to develop mercury control technologies and the attractive features of adsorption processes on dry sorbents led researchers to focus their efforts on the evaluation of the adsorption kinetics and sorbent capacity of many different solid sorbents. Past research has identified two different classes of sorbents to be effective in mercury removal: activated carbons and calcium-based sorbents.<sup>5-7</sup>

An important scientific issue that needs to be addressed is the chemical form in which mercury is released from combustion systems. Previous investigations conducted in EPA laboratories<sup>6,7</sup> have indicated that mercury control strategies are dependent upon the type of mercury species that exist in the coal/waste combustion flue gases. These studies have shown the relative ease of controlling oxidized mercury (specifically mercuric chloride,  $\text{HgCl}_2$ ) as opposed to elemental mercury ( $\text{Hg}^0$ ). Hall et al.<sup>8</sup> showed that, in a simulated municipal waste combustor (MWC), flue gas mercury is mainly found as  $\text{HgCl}_2$ . They postulated that  $\text{HgCl}_2$  is the most favorable mercury species due to relatively high hydrogen chloride (HCl) concentration in MWCs. On the other hand,  $\text{Hg}^0$  is the prevailing form of mercury in emissions from coal combustion processes.<sup>9</sup> However, recent pilot-scale coal combustion test results have indicated that combustion of certain types of coal (Blacksville, a bituminous coal from the Pittsburgh No. 8 seam) can lead to a flue gas mercury species profile dominated by oxidized mercury (most probably  $\text{HgCl}_2$ ).<sup>10</sup> Research in this area is on-going to determine the conditions that favor formation of oxidized mercury in coal combustion processes.

The presence of both  $\text{Hg}^0$  and  $\text{HgCl}_2$  in combustion flue gases justifies a comprehensive research study on the adsorption of both these species by solid sorbents. As mentioned, preliminary investigations found activated carbons to be efficient  $\text{Hg}^0$  sorbents,<sup>6</sup> and calcium-based compounds to be effective  $\text{HgCl}_2$  sorbents.<sup>7</sup> The activated carbon studies<sup>6</sup> were focused on determining the effect of temperature (23 vs. 140°C) on the adsorption of  $\text{Hg}^0$  on two different kinds of thermally activated carbons (FGD and PC-100, Norit Americas Inc.) in a fixed-bed reactor system. It was concluded that surface area and temperature have a strong effect on the  $\text{Hg}^0$  adsorption capacity of the activated carbons. PC-100, with twice the specific surface area compared to that of FGD, captured about four times the amount of  $\text{Hg}^0$  captured by FGD. Both thermally activated carbons exhibited almost double the adsorption capacity at 23°C compared to that at 140°C. This study reports on the continuation of that investigation<sup>6</sup> with emphasis on obtaining  $\text{Hg}^0$  breakthrough sorption curves. Systematic tests were conducted in which the

effects of temperature, inlet  $\text{Hg}^0$  concentration, and the presence or absence of sulfur dioxide ( $\text{SO}_2$ ) were investigated. The adsorption of  $\text{Hg}^0$  was studied by varying the temperature in the range of 60-140 °C and the  $\text{Hg}^0$  concentration in the gas stream in the range of 8-40 ppb (65-327  $\mu\text{g}/\text{dscm}$ ). The selected concentration range was higher than that observed in coal-fired units (1-20  $\mu\text{g}/\text{dscm}$ ).<sup>10</sup>  $\text{SO}_2$  is believed to be an important flue gas species influencing the capture of mercury compounds by activated carbons and calcium-based sorbents.<sup>7</sup> The effect of this species was studied at a concentration of 1000 ppm (2.6  $\text{g}/\text{dscm}$ ).

A previous investigation<sup>7</sup> illustrated the relative ease of removal of  $\text{HgCl}_2$  as compared to  $\text{Hg}^0$ . It was concluded that calcium-based sorbents --calcium hydroxide [ $\text{Ca}(\text{OH})_2$ ] and mixtures of  $\text{Ca}(\text{OH})_2$  and fly ash-- can potentially be very cost effective in adsorbing  $\text{HgCl}_2$  from simulated flue gases. A comprehensive, systematic study on the control of  $\text{HgCl}_2$  by activated carbons has been conducted by Karatza et al.<sup>11</sup>, concluding that at a temperature of 150 °C and gas-phase  $\text{HgCl}_2$  concentration of 1000  $\mu\text{g}/\text{dscm}$  a thermally activated carbon can capture a mass of  $\text{HgCl}_2$  greater than 1% of its weight. The investigation presented here reports on the continuation of a previous study<sup>7</sup>; it focuses on a systematic study on the effect of temperature (60-140 °C),  $\text{HgCl}_2$  concentration [11-73 ppb (122-809  $\mu\text{g}/\text{dscm}$ )], and  $\text{SO}_2$  (1000 ppm) on the  $\text{HgCl}_2$  capture by  $\text{Ca}(\text{OH})_2$ .

In summary, the scope of the present work was to study adsorption of  $\text{Hg}^0$  by thermally activated carbons and  $\text{HgCl}_2$  by  $\text{Ca}(\text{OH})_2$ . While the focus of the previous investigations<sup>6,7</sup> was mainly on a screening type of comparison between these sorbents, this work investigates the fundamental and quantitative dependence of the adsorption capacity of sorbents and mercury removal on sorption parameters such as temperature, gas-phase mercury species concentration, and flue gas species ( $\text{SO}_2$ ). The experiments were performed in a laboratory-scale, fixed-bed apparatus, in which the inlet gas containing nitrogen,  $\text{Hg}^0$  or  $\text{HgCl}_2$ , and, on occasions,  $\text{SO}_2$  were contacted with sorbents.

## EXPERIMENTAL PROCEDURE

A schematic of the experimental apparatus used to study the capture of  $\text{Hg}^0$  and  $\text{HgCl}_2$  is shown in Figure 1. Pure  $\text{HgCl}_2$  powder contained in a diffusion vial was the source of  $\text{HgCl}_2$  vapor, and pure  $\text{Hg}^0$  liquid in a permeation tube (VICI Metronics Inc.) was the source of  $\text{Hg}^0$  vapor. The concentration of  $\text{HgCl}_2$  or  $\text{Hg}^0$  in the gas stream was controlled by adjusting the water bath temperature. A detailed description of the calibration of the mercury generation system and its quality control checks has been reported elsewhere.<sup>6,7</sup> The mercury vapor generated was carried into a manifold by a nitrogen stream where it mixed with  $\text{SO}_2$  (in the  $\text{SO}_2$  studies only) at a constant total system gas flow rate of 350  $\text{cm}^3/\text{min}$  (at standard temperature and pressure). A three-way valve placed before the manifold (Figure 1) diverted the mercury-laden nitrogen stream away from the manifold when desired. The first three-way valve placed after the manifold was used to direct flow to or away from the fixed-bed reactor. The sorbent to be studied (approximately 0.02 g diluted with 2 g inert glass beads; bed length of approximately 2 cm) was placed in the reactor which was maintained at the desired bed temperature by a temperature controller. A furnace kept at 850 °C was placed downstream of the reactor to convert any oxidized mercury ( $\text{Hg}^{++}$ , as in  $\text{HgCl}_2$ ) to  $\text{Hg}^0$ . According to thermodynamic predictions, the only mercury species that exists at this temperature is  $\text{Hg}^0$ .<sup>12</sup> The presence of the furnace enabled detection of non-adsorbed  $\text{HgCl}_2$  as  $\text{Hg}^0$  by the on-line ultraviolet (UV)  $\text{Hg}^0$  analyzer, thus providing actual, continuous  $\text{Hg}^0$  or  $\text{HgCl}_2$  capture data by the fixed-bed of sorbent. The UV  $\text{Hg}^0$  analyzer used in this study

responded to SO<sub>2</sub> as well as Hg<sup>0</sup>. For instance, a gas stream consisting of 1000 ppm SO<sub>2</sub> and 15 ppb Hg<sup>0</sup> produced a SO<sub>2</sub>/Hg<sup>0</sup> signal ratio of 8/18. Contributions from SO<sub>2</sub> were corrected for by placing a SO<sub>2</sub> analyzer (UV) on-line downstream of the Hg<sup>0</sup> analyzer. The SO<sub>2</sub> analyzer was incapable of responding to mercury in the concentration range used in this study. By subtracting the SO<sub>2</sub> signal measured by the SO<sub>2</sub> analyzer from the total response of the mercury analyzer, the mercury concentration was obtained.

In each test, the fixed-bed was exposed to the mercury-laden flue gas for 24 hours or until 100% breakthrough (saturation) was achieved (whichever came first). During this period the exit concentration of mercury was continuously monitored. The percent removal of Hg<sup>0</sup> or HgCl<sub>2</sub> at any time was obtained as follows: percent removal = 100\*[(mercury)<sub>in</sub>-(mercury)<sub>out</sub>]/(mercury)<sub>in</sub>. The amount of mercury species uptake at any time (q, weight mercury species/weight sorbent) was determined by integrating (in time) and evaluating the area under the removal curves. Selected experiments conducted during this test program were run in duplicates and indicated a variation of ±10% in the experimental results.

## SORBENTS

The capture of Hg<sup>0</sup> was investigated using two different thermally activated carbons: FGD and PC-100. The capture of HgCl<sub>2</sub> was investigated using Ca(OH)<sub>2</sub>. FGD, manufactured by Norit Americas Inc., is a lignite-coal-based activated carbon with a bulk density of 0.51 g/cm<sup>3</sup>, ignition temperature of 450°C, and particle size greater than 400 mesh (less than 30 μm). Information about the total surface area and average pore size diameter of FGD (and other sorbents) was obtained by using a Micromeritics ASAP 2600 based on nitrogen adsorption/desorption with the Brunauer-Emmett-Teller (BET) method. FGD has a total surface area of 500 m<sup>2</sup>/g and an average pore diameter of 3.2 nm. PC-100, manufactured by Norit Americas Inc., is a bituminous-coal-based activated carbon with a bulk density of 0.52 g/cm<sup>3</sup>, ignition temperature of 450°C, and a particle size greater than 400 mesh (less than 30 μm). PC-100 has a total surface area of 900 m<sup>2</sup>/g and an average pore diameter of 1.8 nm. It has nearly twice the surface area of FGD; most of this surface area is due to its smaller pores than those of FGD as indicated by PC-100's smaller average pore diameter. The Ca(OH)<sub>2</sub> used in the HgCl<sub>2</sub> removal studies was a reagent grade Ca(OH)<sub>2</sub> (Sigma Inc.) containing 97.6% Ca(OH)<sub>2</sub> and 1.8% calcium carbonate (CaCO<sub>3</sub>). It has a total surface area of 13 m<sup>2</sup>/g and an average pore diameter of 33.4 nm.

## EXPERIMENTAL RESULTS AND DISCUSSION

### HgCl<sub>2</sub> Capture by Ca(OH)<sub>2</sub>

The effects of temperature and inlet concentration on the HgCl<sub>2</sub> uptake by Ca(OH)<sub>2</sub>, obtained under the baseline conditions (nitrogen and HgCl<sub>2</sub> only), are shown in Figure 2. During the 24 hours of exposure, HgCl<sub>2</sub> uptake (q) increased linearly as time of exposure progressed; Ca(OH)<sub>2</sub> did not reach its saturation capacity under the baseline conditions. This observation may indicate an abundance of HgCl<sub>2</sub> adsorption sites on Ca(OH)<sub>2</sub>. Considering the acidic nature of gaseous HgCl<sub>2</sub>, these sites can be hypothesized to be alkaline. Note that, for the sake of comparison to the Hg<sup>0</sup> data, the results were truncated to show the first 5 hours of exposure; further uptakes up to 24 hours can be obtained by extending the lines.

The percent removals [constant during the exposure period; measured (HgCl<sub>2</sub>)<sub>out</sub> was constant] of incoming HgCl<sub>2</sub> by the fixed-bed of Ca(OH)<sub>2</sub> as a function of temperature and inlet concentration are shown in Figure 3. Increasing the HgCl<sub>2</sub> inlet concentration and decreasing the bed temperature

increased the percent removals. The bed temperature had a strong effect on the percent removals at the lowest concentration (11 ppb) tested, and the inlet concentration influenced percent removals more strongly at the highest temperature (140°C) tested.

In adsorption processes, a rise in partial pressure and a fall in temperature result in an increase in the amount of the material adsorbed (adsorbate). The temperature and the concentration (partial pressure) trends observed here suggest that the process is an adsorption (physical and/or chemical) controlled process and that the rate of  $\text{HgCl}_2$  capture is determined by how fast molecules in the vicinity of the active sites are being adsorbed. This indicates that the capture process is not controlled by an external (boundary layer) mass transfer mechanism; boundary layer mass transfer is fast and the  $\text{HgCl}_2$  concentration in the vicinity of the surface is very close to the bulk concentration. Moreover, the observed effect of temperature on the amount captured (or percent removed) does not support an external mass transfer hypothesis, since the mass transfer rate is proportional to the 1.5 power of absolute temperature.

The effect of  $\text{SO}_2$  on  $\text{HgCl}_2$  capture by  $\text{Ca}(\text{OH})_2$  was studied by adding 1000 ppm  $\text{SO}_2$  to the nitrogen/ $\text{HgCl}_2$  stream, while keeping the total flow through the fixed-bed constant. The results are shown in Figure 4. This figure shows the effect of  $\text{SO}_2$  at a constant inlet  $\text{HgCl}_2$  concentration of 38 ppb for the three studied temperatures. At the low temperature (60°C),  $\text{SO}_2$  had no effect. However, increasing the temperature drastically reduced the sorption capacity of  $\text{Ca}(\text{OH})_2$  in the presence of  $\text{SO}_2$ . It appears that  $\text{Ca}(\text{OH})_2$  and  $\text{SO}_2$  may react at higher temperatures, leading to the destruction of the available (alkaline) sites needed for  $\text{HgCl}_2$  capture. In the presence of  $\text{SO}_2$  and at 100°C, saturation capacity [0.7 mg  $\text{HgCl}_2$ /g  $\text{Ca}(\text{OH})_2$ ] was achieved after 5 hours of exposure; at 140°C, saturation capacity [0.1 mg  $\text{HgCl}_2$ /g  $\text{Ca}(\text{OH})_2$ ] was achieved after only 1 hour of exposure. Note that in the absence of  $\text{SO}_2$ , saturation capacities were never achieved during the 24 hours of exposure.

### **$\text{Hg}^0$ Capture by Activated Carbons**

The effects of temperature and inlet concentration on the  $\text{Hg}^0$  uptake by activated carbon, FGD, under the baseline conditions, are shown in Figure 5. Unlike  $\text{HgCl}_2$  capture by  $\text{Ca}(\text{OH})_2$ , during a typical 5-hour run the sorbent (FGD) reached its saturation capacity, indicating that the available active sites for capturing  $\text{Hg}^0$  in the activated carbon are limited. The comparison between  $\text{Hg}^0$  and  $\text{HgCl}_2$  uptake results (Figures 2 and 5) suggests that it is more difficult to control  $\text{Hg}^0$  emissions. It should be noted that in a previous investigation<sup>7</sup>, the baseline tests conducted using  $\text{Ca}(\text{OH})_2$  showed that no  $\text{Hg}^0$  was captured, indicating the ineffectiveness of  $\text{Ca}(\text{OH})_2$  (lack of active sites) in capturing  $\text{Hg}^0$ .

From Figure 5 it is apparent that the amount of  $\text{Hg}^0$  captured is influenced more by the bed temperature than by the inlet concentration (note the ordinate scales). Increasing the temperature reduced the saturation capacity and the time needed to reach the saturation capacity. Increasing the inlet  $\text{Hg}^0$  concentration increased the saturation capacity, and the initial rate of  $\text{Hg}^0$  uptake (slope of tangent at the origin). Similar to the  $\text{HgCl}_2$  case, the temperature and concentration trends suggest that the process is active site adsorption controlled.

The same set of tests were performed using PC-100 as the activated carbon sorbent. Results for PC-100 in comparison to FGD are presented in Figure 6 (bed temperature of 100°C). PC-100 also consistently exhibited higher saturation capacity than did FGD at temperatures of 60° and 140°C (not presented

here). It should be noted that PC-100 exhibited the same temperature and concentration trends as those of FGD: higher  $\text{Hg}^0$  captures at lower temperatures and higher inlet concentrations.

The effect of  $\text{SO}_2$  on  $\text{Hg}^0$  capture by FGD was studied by adding 1000 ppm  $\text{SO}_2$  to the nitrogen/ $\text{Hg}^0$  (40 ppb) stream while keeping the total gas flow through the fixed-bed constant. The highest inlet  $\text{Hg}^0$  concentration was chosen to minimize the interference of  $\text{SO}_2$  in the  $\text{Hg}^0$  analyzer discussed previously. Figure 7 shows the effect of  $\text{SO}_2$  on  $\text{Hg}^0$  uptake at a constant inlet  $\text{Hg}^0$  concentration of 40 ppb for the three studied temperatures. At the lowest temperature tested ( $60^\circ\text{C}$ ),  $\text{SO}_2$  had essentially no effect on the  $\text{Hg}^0$  uptake by the FGD; however, increasing the temperature increased the  $\text{Hg}^0$  capture by the FGD. It appears that at higher temperatures,  $\text{SO}_2$  might have reacted with the FGD and created active sulfur (S) sites, which are considered to be very active in capturing  $\text{Hg}^0$ .<sup>6</sup> The activity of FGD in the presence of  $\text{SO}_2$  at higher temperatures resembled that of sulfur-impregnated (chemically impregnated) activated carbon studied by Krishnan et al.<sup>6</sup> Based on these observations, one may consider injecting this activated carbon upstream of the acid gas removal devices of the full-scale combustion systems in order to take advantage of the enhancement effect of  $\text{SO}_2$  on  $\text{Hg}^0$  capture.

Unlike FGD, the presence of  $\text{SO}_2$  did not have any effect on  $\text{Hg}^0$  capture by PC-100. This may be due to the inability of PC-100 to form sulfur-activated sites. One may relate this observation to the differences in chemical structure and composition of these two thermally activated carbons. FGD and PC-100 are lignite-coal-based and bituminous-coal-based activated carbons, respectively. Future efforts will be directed toward a comprehensive understanding of the chemical nature (active sites) of these activated carbons and its influence on  $\text{Hg}^0$  capture. These future studies will also help explain the observed higher activities of PC-100 as compared to FGD, since this may not be just a total surface area effect, as indicated by  $\text{SO}_2$  studies.

## CONCLUSION

The effects of bed temperature, mercury concentration, and the presence of  $\text{SO}_2$  on the capture of  $\text{Hg}^0$  and  $\text{HgCl}_2$ , the mercury species identified in coal/waste combustion flue gases, by thermally activated carbons (FGD and PC-100, Norit Americas Inc.) and a calcium-based sorbent [ $\text{Ca}(\text{OH})_2$ ] were examined in a fixed-bed, bench-scale system. Sorption studies indicated an abundance of  $\text{HgCl}_2$  adsorption sites in  $\text{Ca}(\text{OH})_2$ . Considering the acidic nature of gaseous  $\text{HgCl}_2$ , these sites can be assumed to be alkaline. Increasing the  $\text{HgCl}_2$  inlet concentration increased its removal, and increasing the bed temperature decreased this removal. The observed temperature and concentration trends suggest that the process is adsorption controlled and that the rate of  $\text{HgCl}_2$  capture is determined by how fast molecules in the vicinity of the active sites are being adsorbed. The presence of  $\text{SO}_2$  drastically reduced the sorption capacity of  $\text{Ca}(\text{OH})_2$  at a bed temperature range of  $100\text{--}140^\circ\text{C}$ . Sorption studies further indicated that the available active sites for capturing  $\text{Hg}^0$  in the activated carbons are very limited, suggesting that it is more difficult to control  $\text{Hg}^0$  emissions. Increasing the inlet  $\text{Hg}^0$  concentration and decreasing the bed temperature increased the saturation capacities of the activated carbons, the time needed to reach this capacity, and the initial rate of  $\text{Hg}^0$  uptake. PC-100, with twice the total surface area as that of FGD, consistently exhibited higher saturation capacity than that for FGD. The presence of  $\text{SO}_2$  drastically increased the  $\text{Hg}^0$  capture capabilities of FGD and had no influence on  $\text{Hg}^0$  capture by PC-100. These observations may be due to the differences in chemical structure and composition of these two thermally activated carbons.

Based on these observations, the optimum region for control of HgCl<sub>2</sub> by calcium-based sorbents may be considered to be downstream from acid gas (SO<sub>2</sub>) removal devices. One may also consider injecting activated carbon (FGD) upstream of the acid gas removal devices of the full-scale combustion systems in order to take advantage of the enhancement effect of higher SO<sub>2</sub> levels on Hg<sup>0</sup> capture.

## ACKNOWLEDGMENTS

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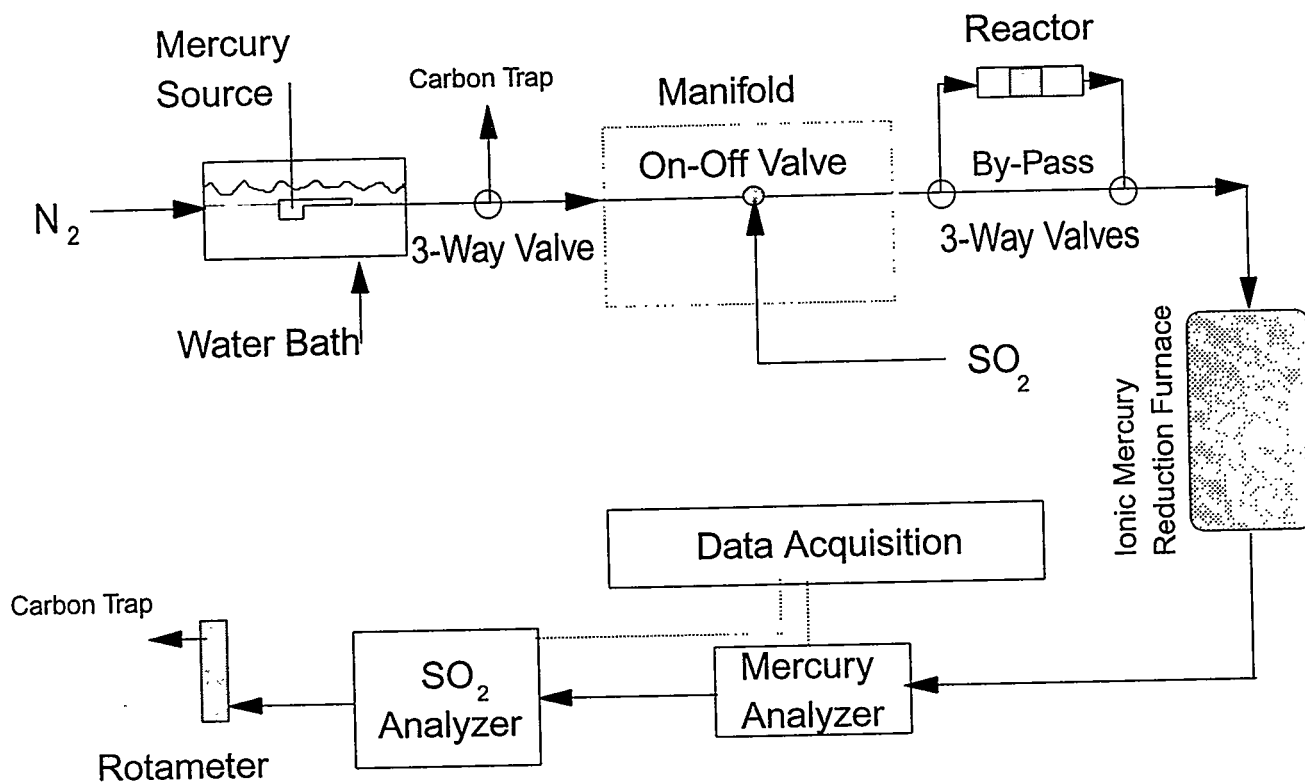


Figure 1. Schematic of the bench-scale, fixed-bed reactor setup.



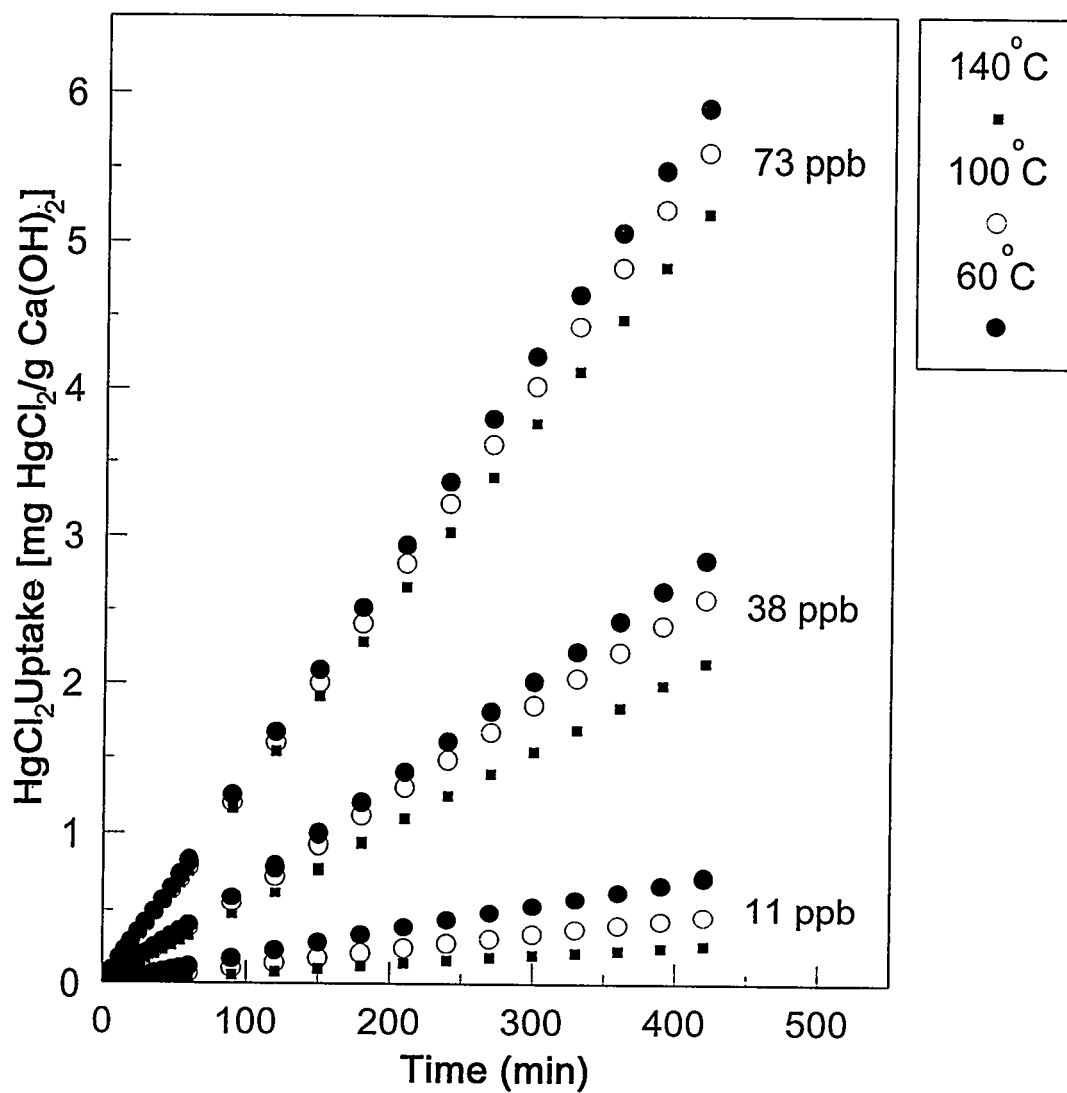


Figure 2. Effects of bed temperature and inlet HgCl<sub>2</sub> concentration on HgCl<sub>2</sub> uptake by Ca(OH)<sub>2</sub> (baseline conditions).

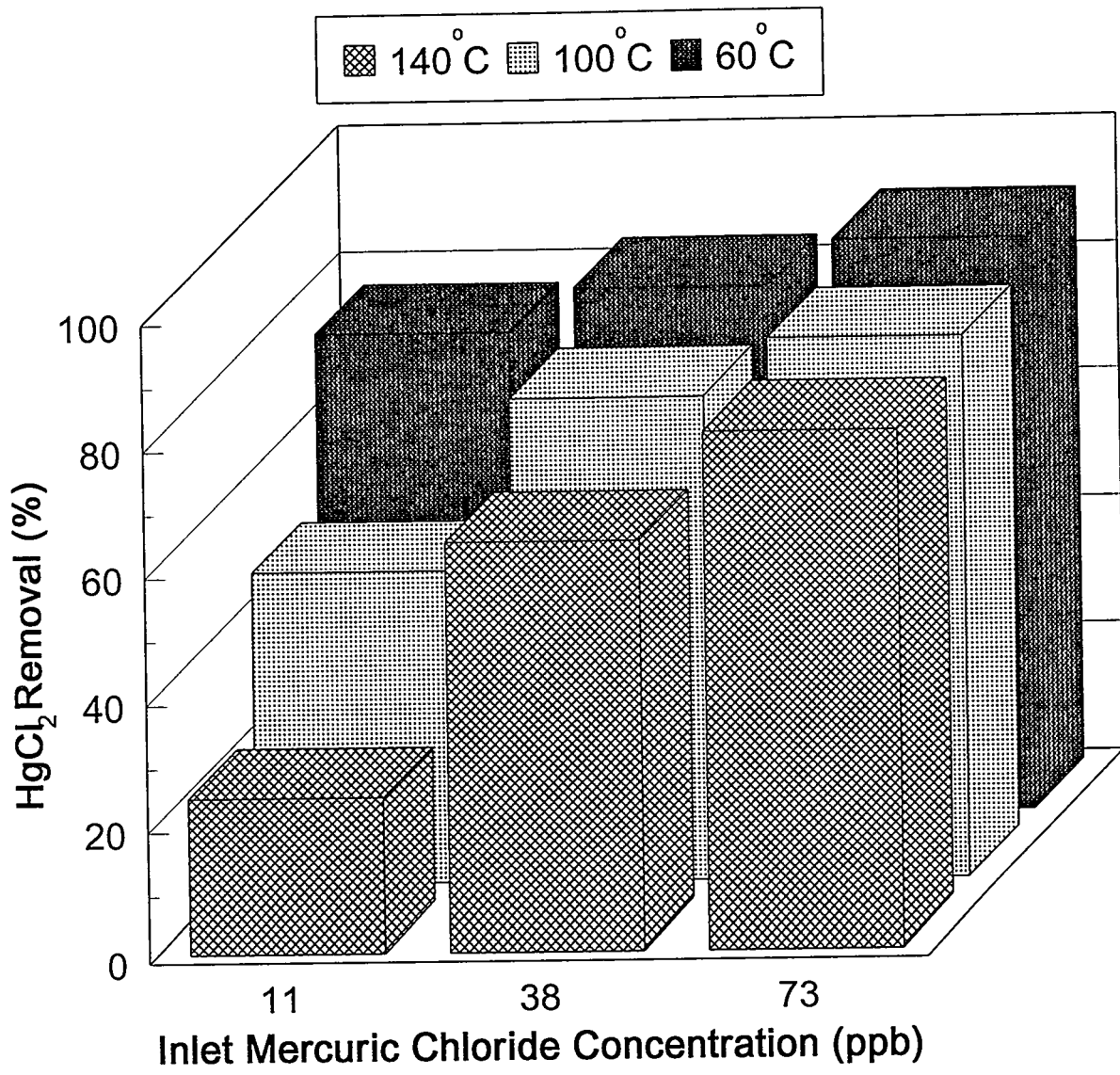


Figure 3. Effects of bed temperature and inlet HgCl<sub>2</sub> concentration on HgCl<sub>2</sub> removal by Ca(OH)<sub>2</sub> (baseline conditions).

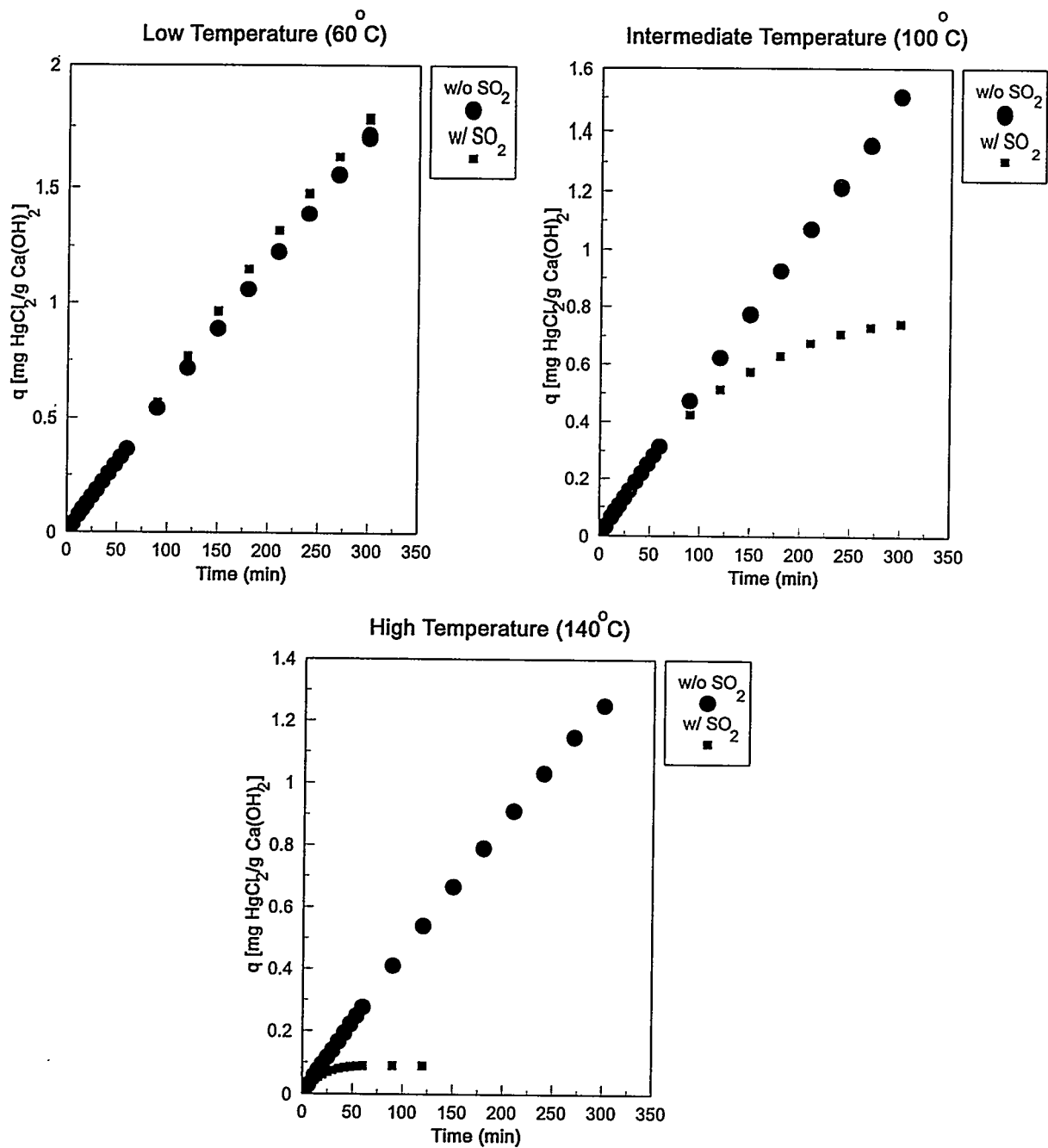


Figure 4. Effect of  $\text{SO}_2$  (1000 ppm) on  $\text{HgCl}_2$  uptake by  $\text{Ca(OH)}_2$  (inlet  $\text{HgCl}_2$  concentration of 38 ppb).

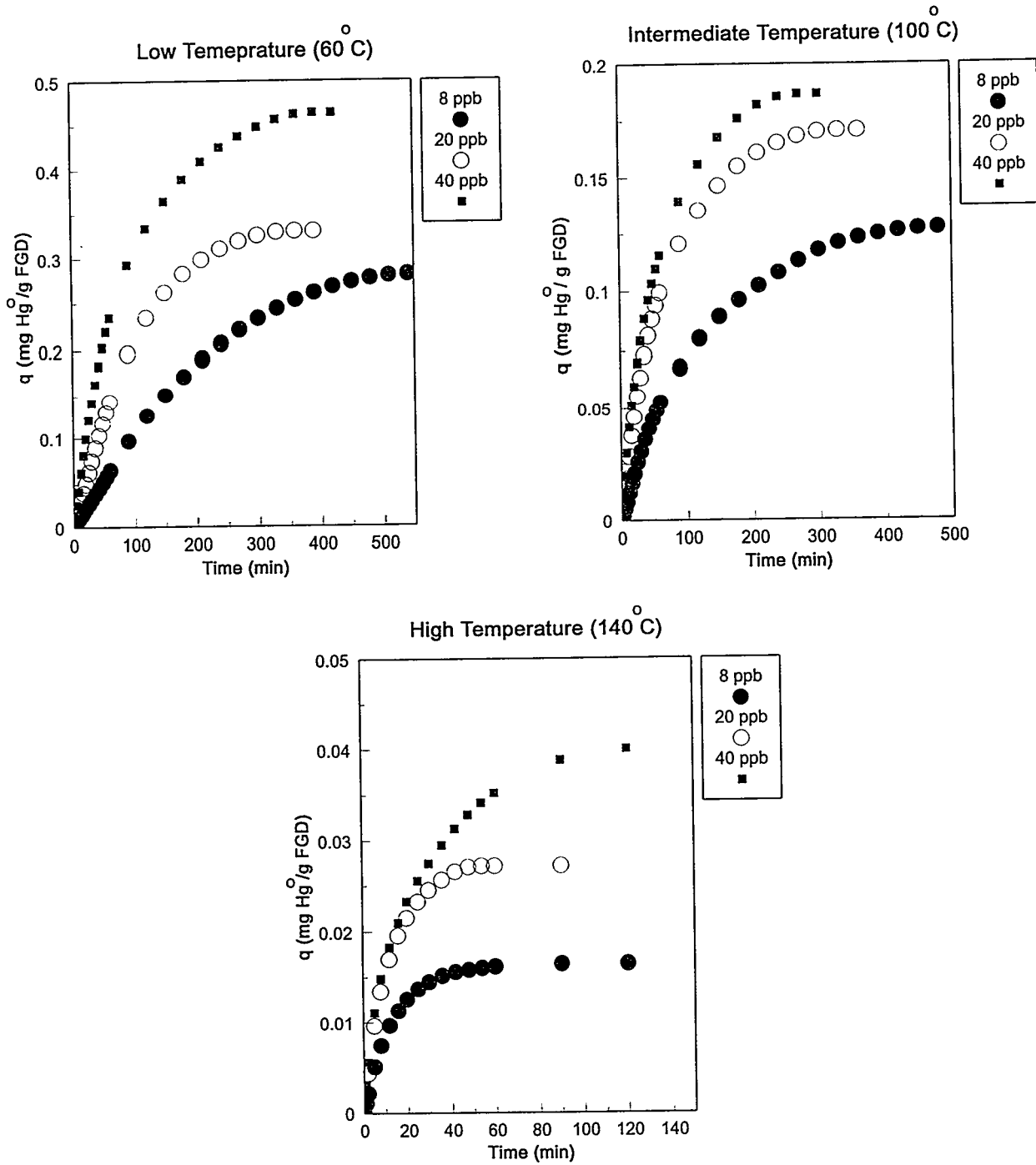


Figure 5. Effects of bed temperature and inlet  $Hg^0$  concentration on  $Hg^0$  uptake by activated carbon, FGD (baseline conditions).

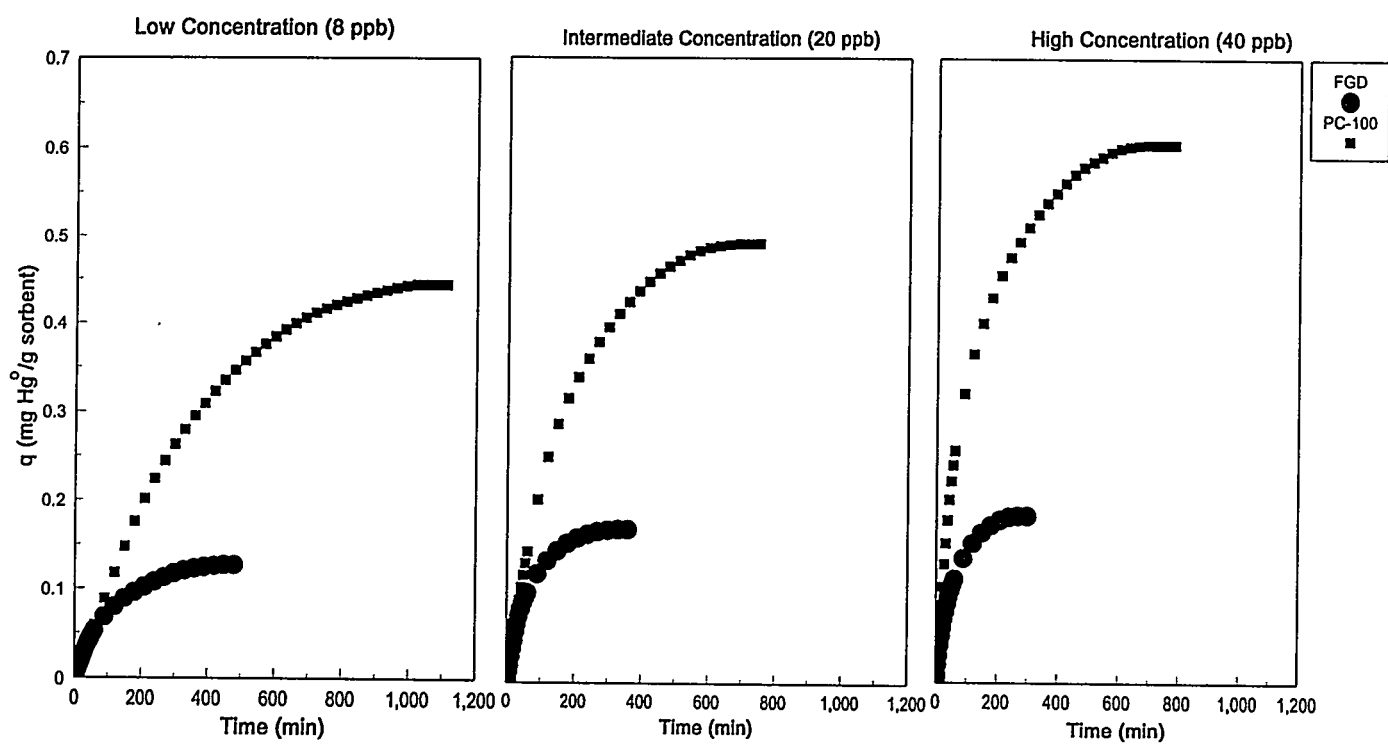


Figure 6. Uptake of  $Hg^0$  by activated carbons, FGD and PC-100 (bed temperature of  $100^\circ C$ , baseline conditions).

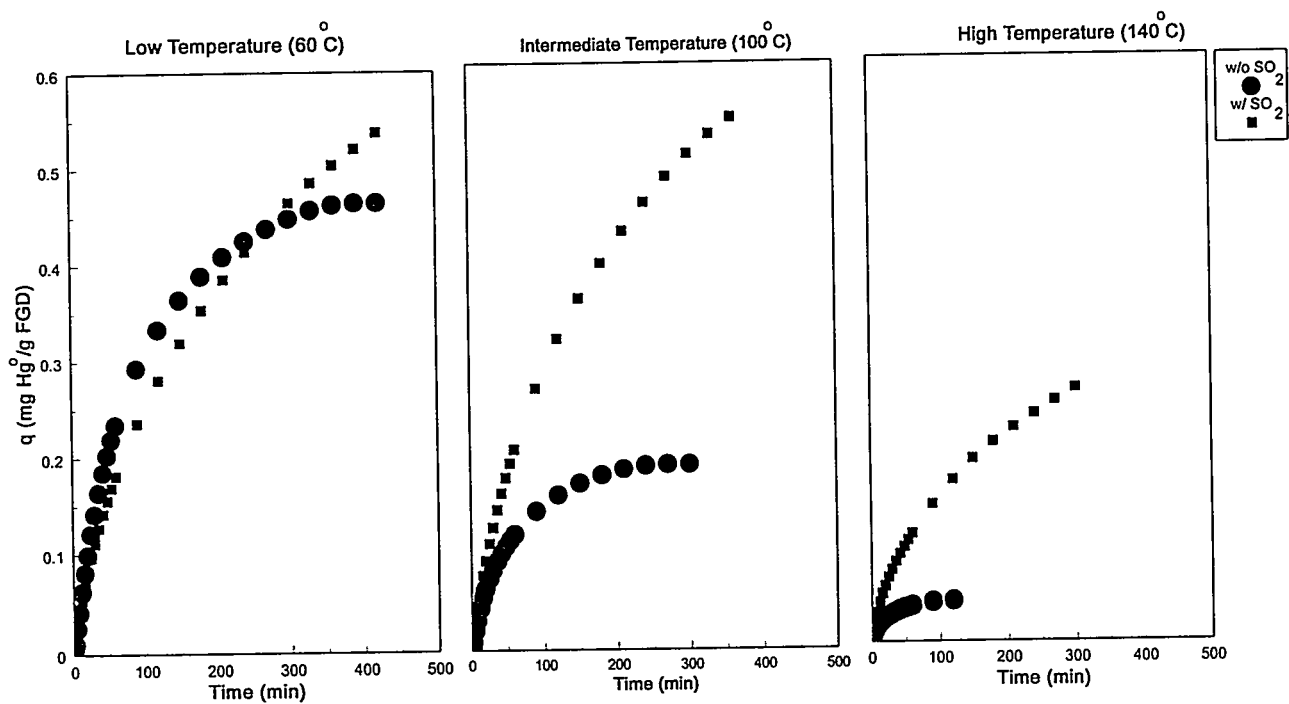


Figure 7. Effect of SO<sub>2</sub> (1000 ppm) on Hg<sup>0</sup> uptake by activated carbon, FGD (inlet Hg<sup>0</sup> concentration of 40 ppb).

Characterization of PIC Emissions from Combustion of Pentachlorophenol-Treated Wood Wastes

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## INTRODUCTION

The use of waste wood as fuel for producing energy is a promising supplement to fossil fuels for many regions of the country. In addition to recovering energy and conserving landfill space, burning waste wood fuels also mitigates global warming created by fossil fuel combustion. However, the environmental consequences resulting from emissions generated by combustion of waste wood which contains paints, resins, or preservatives are not well understood. The combustion of waste wood treated with chemicals may produce potentially hazardous products of incomplete combustion (PIC) emissions such as dioxins.<sup>1,2</sup>

Characterization of PIC emissions from the combustion of waste wood previously treated with pentachlorophenol is reported in this study. Utility poles and crossbars are typically treated with a preservative such as pentachlorophenol in order to prolong their service life. They are disposed of by landfilling after being taken out of service. Burning such wood waste in boilers for steam generation becomes an increasingly attractive waste management alternative as it contains substantial energy values and reduces landfilling costs. Pilot-scale combustion tests were conducted under well controlled conditions in a 0.58 MW (2 million Btu/hr) combustor to compare PIC emissions from burning untreated wood and pentachlorophenol-treated wood. Sampling and analyses for a wide variety of PICs, including volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs), and dioxins and furans, were performed to assess the effect of pentachlorophenol preservative present in wood on PIC emissions.

## EXPERIMENTAL

### Test Facility

The combustion testing was conducted using a multifuel combustor (MFC) installed in the Environmental Protection Agency's research facility located at Research Triangle Park, North Carolina. The MFC, shown in Figure 1, is a pilot-scale stoker combustor with 0.58 MW (2 million Btu/hr) maximum thermal output which is sufficient for simulating the full range of conditions that might be encountered in practical systems. The MFC is capable of burning a wide variety of solid fuels including municipal solid waste, refuse derived fuel, biomass fuel, and coal singly or in combination. The combustor is constructed with modular sections, which provides maximum flexibility for modifying the combustor for research purposes. Access ports are installed throughout the entire combustor for sampling and visual observation.

The combustor was operated in a spreader stoker firing mode. Wood fuel was stored in a large roll-off container located near the building which houses the MFC. A tote bin filled with the wood fuel was transported to the MFC facility and the fuel was sequentially dumped into a loading hopper installed outside the facility building, using a forklift. An electric drive conveyor belt, installed under the loading hopper and penetrating through the facility building wall, moved the fuel to a vibrating active



storage silo installed inside the building. The silo is designed for a 2 hour turnover time using an average fuel density of 80 kg/m<sup>3</sup> (5 lb/ft<sup>3</sup>) and a feed rate of 163 kg/hr (360 lb/hr). The silo is large enough to dampen out effects of spurious loading yet small enough to fit inside the facility building. A vibratory silo is designed to improve mixing and reduce wall sticking. The fuel was distributed from the silo onto an airtight horizontal vibratory feeder conveyor connecting the silo to the transition section of the spreader stoker. The mass flow of the conveyor was calibrated for control of the fuel feed rate. The fuel traveled horizontally along the conveyor and dropped through the transition section into the feed injection port of the spreader stoker. The vibratory transition section is designed with an inlet air plenum area and adjustable inlet angle plate to facilitate spreading fuel onto the 61 by 30 cm (2 by 1ft) stoker grate where a burning fuel bed is formed.

### **Wood Fuel**

Two different wood fuels, untreated wood and treated wood, were obtained from a plant where wood poles are treated with pentachlorophenol. The untreated wood was pine poles that are typically treated in the plant. The treated wood was a mixture of recently treated poles, mostly small pieces cut from the end to meet customers' length specifications and aged treated poles taken out of service. Both treated and untreated poles were ground into chips less than 7.6 cm (3 in.) in diameter. The two fuels were characterized by proximate, ultimate, and chlorine analyses: results are shown in Table 1. It can be seen that the treated wood fuel is drier with higher heating value as compared to the untreated wood. The chlorine content of the treated wood fuel is 10 times higher than that of the untreated wood.

### **Combustion Test Parameters**

The only parameter investigated was the difference in emissions when burning treated vs. untreated wood fuels under similar combustion conditions; no attempt was made to conduct parametric tests for this study. In general, the operating conditions of the test were considered optimal when the fuel was burning at the designed heat release rate with nominally 160 % excess air and a low level (< 100 ppm) of carbon monoxide (CO) emissions in the flue gas.

The amount of fuel burned was not measured continuously during a test. It was estimated by weighing the fuel before it was dumped into the loading hopper of the MFC and measuring the time required for its complete consumption to calculate an average feed rate. The average feed rate was used to estimate the amount of fuel burned during a timed emissions sampling period. The MFC is equipped with continuous emission monitors (CEMs) -- for measuring oxygen (O<sub>2</sub>), CO, carbon dioxide (CO<sub>2</sub>), nitrogen oxide (NO), sulfur dioxide (SO<sub>2</sub>), total hydrocarbon (THC), and hydrogen chloride (HCl) -- and equipment for continuously monitoring process temperatures and pressures.

### **Sampling and Analytical Procedures**

The emissions sampling point for the present study was selected at the duct which connects the convective section of the furnace to the baghouse. The selection of a sampling point prior to any gas cleaning device was made to ensure that any difference in flue gas pollutant concentrations generated by combustion of treated and untreated wood fuels would be observed. Emission measurements made after the flue gas cleaning device would be expected to mask this difference and would be reflective of the effectiveness of the devices (i.e., baghouse and caustic scrubber) used.

The sampling program of the present study focused primarily on organic emissions, particularly those which are considered potentially hazardous, from the combustion of wood fuels. Sampling and analytical protocols used were based on existing EPA methods for measuring organic constituents in flue gases. Samples for measuring volatile organic PIC emissions were taken using EPA SW Method 0030, Volatile Organic Sampling Train (VOST), and the volatile PICs in the VOST samples were identified and quantified using SW Methods 5040 and 8240.<sup>3,5</sup> Samples for measuring SVOC emissions were obtained using SW Method 0010, and SVOCs in the samples were analyzed using SW Method 8270B.<sup>6,7</sup> Method 23 was used for measuring dioxin and furan emissions.<sup>8</sup>

## RESULTS AND DISCUSSION

### Test Conditions

Six combustion tests, three tests each for both the untreated and the treated wood fuels, were performed under similar combustor operating conditions. The untreated wood tests were performed first followed by the treated wood tests to avoid cross contamination. Triplicate runs were made to determine experimental repeatability. Since it was not practical to measure fuel feed rate continuously, the tests were performed under constant excess air level (nominally 160% excess air) and a constant combustion gas temperature (850° C) measured by a thermocouple located 152 cm (60 in) above the grate. The estimated heat release rates for all the tests are shown in Table 2 and the average CEMs-measured emission rates are shown in Table 3. The heat release rates are calculated based on the average fuel feed rate estimated by the amount of fuel consumed during the test period and the heating value of the fuel measured experimentally. It can be seen from Table 2 that the calculated heat release rates and the measured flue gas flow rates for Untreated Tests 2 and 3 are lower than those of the other tests. The lower flue gas flows for Untreated Tests 2 and 3 are consistent with their low fuel feed rates. The heat release rate was set at 80% of the maximum designed output of the MFC for the present study. The fuel feed rate was adjusted manually throughout all tests in order to maintain a constant temperature of 850° C above the stoker grate and keep CO emissions below 100 ppm, which were considered optimum conditions for the tests. The combustion gas temperature measured may not have been always truly representative. The tremendous turbulence generated above a burning hot fuel bed when the cold underfire air passing through the grate and the cold overfire air injected into the furnace above the grate may create a non-uniform gas temperature profile. Observations through installed viewing ports indicated that turbulent flow passed along the entire furnace during the tests. The heating values measured from the two fuel samples may not be truly representative due to the nonhomogeneous nature of the wood fuels. These two uncertainties may have contributed to the lower estimated heat release rates for the two tests.

It was difficult to produce exactly identical combustion conditions for all the tests, mainly due to the fact that the two wood fuels are not homogeneous. The treated wood fuel contains aged wood, which is drier, and has 30% higher heating value than the untreated wood fuel. Uneven fuel feed rates resulting from the fairly frequent manual adjustment of the fuel feeder could have caused fluctuations in air-to-fuel ratio and momentary excursions of CO emissions. CEM data for CO and CO<sub>2</sub> and visual observations of fuel flow into the stoker suggest that small excursions occurred during the tests. The nonhomogeneous nature of the fuels coupled with the practical difficulties to burn such fuels under well

defined conditions make the accurate estimation of heat release rates during tests impossible. The high CO emissions, shown in Table 3, calculated after the untreated wood tests, are another indication that combustion conditions were difficult to control when burning the untreated wood, which has low heating value and high moisture content.

### **Pentachlorophenol Destruction Efficiency**

The concentration of pentachlorophenol in the treated wood fuel was analyzed to be 4,100 mg/kg (ppmw). For the six tests, pentachlorophenol was not detected in any of the semivolatile organic samples. The estimated pentachlorophenol destruction efficiency (DRE) for each test has been conservatively calculated based on the pentachlorophenol analysis practical quantitation limit (PQL). DRE results are presented in Table 4. The PQL is based on the lowest calibration concentration and does not include factors such as percent recovery or matrix effects. It is evident from these results that burning of the pentachlorophenol-treated waste wood in a stoker combustor destroys the pentachlorophenol efficiently with a DRE higher than that required for hazardous waste destruction (99.99%).

### **Volatile Organic Compound (VOC) Emissions**

Analyses were performed for 50 VOCs in each of the test samples. The VOC emission results presented in Table 5 are the compounds found in concentrations higher than the analytical detection limit in at least one test. Higher levels of benzene were found in the untreated wood test samples, while bromomethane was found at higher levels in the treated wood test samples. Chloromethane, 1,3-butadiene, iodomethane, acetone, chloroform, and 1,2-dichloroethane were found in both untreated and treated wood test samples. The trace levels of chlorinated VOC emissions found in the untreated wood tests may possibly be caused by small amounts of naturally occurring chlorine in untreated woods. More chlorinated VOCs were found in the treated wood tests, probably as a result of higher chlorine content in the pentachlorophenol-treated wood.

### **SVOC Emissions**

Of the 87 semivolatile organic compounds for which analyses were performed in the SVOC samples, only the following five targeted compounds were found to be above the analytical detection limits: phenol, acetophenone, naphthalene, 2-nitrophenol, and phenanthrene. The total SVOC emission rate for all six tests is calculated by adding all the quantifiable SVOC emissions in the test, and results are summarized in Table 6. The typically low SVOC emissions for both the untreated and treated wood tests are an indication that good combustion conditions were achieved during the tests. The slightly lower SVOC emissions for the treated wood tests compared to those for the untreated wood tests also suggest that the combustion of the drier treated wood fuel produced lower PICs. More moisture released during the combustion of the "green" untreated wood fuel may lower the localized combustion zone temperature and cause more PIC formation.

### **Dioxin/Furan (PCDD/PCDF) Emissions**

The dioxin/furan samples measured by Method 23 were taken at the exit of the convective heat exchanger and before the baghouse. The temperature of the flue gas at the sampling point was between 150 and 160° C. It has been shown that the dioxin and furan formation "temperature window" in flue

gases is between 200 and 450° C.<sup>9</sup> It is reasonable to expect that the formation of dioxins and furans has already been completed before the sampling location.

As shown in Table 7, total PCDD/PCDF emissions from the untreated wood tests [using the International Toxic Equivalency (I-TEQ) method of accounting for different toxicities of PCDD/PCDF congeners] averaged 0.274 ng I-TEQ/dscm and from treated wood tests 1.190 ng I-TEQ/dscm. These emission rates are prior to any control devices. Particulate control devices could potentially remove part of this material from the gas stream, depending on the fly ash collection efficiency, PCDD/PCDF formation, and solid-to-vapor-phase partition with the device. Since the temperature at the sampling point is below 160° C, it is likely that the rates of formation and desorption of fly-ash-associated PCDD/PCDF in a subsequent particulate collection device would be low.<sup>10</sup> The PCDF emissions are significantly greater than PCDD emissions for both the untreated and treated wood tests, indicating de novo synthesis formation reactions rather than condensation reactions.<sup>11</sup> The PCDD/PCDF congener distributions for the treated and untreated tests also differ. In the untreated wood tests, the distribution is peaked at the the lower-chlorinated tetra-PCDD and di-PCDF congeners; in the treated wood tests, the distribution is shifted to the higher-chlorinated hexa-PCDD and penta-PCDF congeners. The PCDD/PCDF emission rates measured from both the untreated and treated wood tests are considered low when compared to the stack emissions from commercial municipal waste combustors, which range from 0.01 to 400 ng I-TEQ/dscm.<sup>12</sup> The low PCDD/PCDF emissions measured from the untreated wood tests are similar to those from burning natural wood, which range from 0.066 to 0.214 ng I-TEQ/dscm.<sup>1</sup> The PCDD/PCDF emissions from the treated wood tests are higher than those (0.0359 ng I-TEQ/dscm) from a waste-to-energy plant burning a mixture of clean wood and pentachlorophenol-treated waste wood.<sup>13</sup>

The higher PCDD/PCDF emissions from the treated wood tests compared to those from the untreated wood tests are consistent with their higher chlorinated VOC emissions. Approximately 200 ppm of HCl was measured in emissions from the treated wood tests, while HCl was not detected in the untreated wood tests. The PCDD/PCDF samples were collected into front half and back half of the sampling train and analyzed separately, which gives an indication of the PCDD/PCDF associated with the particulate material and as gaseous emissions. Table 8 shows that the percentage of PCDD/PCDF in the front half catch is greater in the treated wood tests than in the untreated wood tests. The treated wood tests had a higher flue gas flow rate (see Table 2) as a result of the higher heating value of the treated wood burned, which would enhance particulate load at the sampling point. This information suggests that particulate carryover may affect the total amount of PCDD/PCDF, especially in the treated wood samples where a larger percentage is in the front half catch. Combustor operating conditions in addition to the wood treatment may also contribute to higher measured PCDD/PCDF emissions for the treated wood tests.

## CONCLUSIONS

This study was conducted to identify potential air emissions problems associated with the combustion of waste wood, primarily utility poles treated with pentachlorophenol preservative, as a fuel for energy production in a boiler. The emphasis of the study was placed on the characterization of the PIC emissions from combustion of pentachlorophenol-treated waste wood. The methodology used was a comparative test of emissions prior to the air pollution control device of a pilot-scale combustor burning

untreated wood and wood treated with pentachlorophenol preservative as a fuel. The tests showed that combustion of pentachlorophenol-treated wood is an effective method of destroying the pentachlorophenol contained in the wood, with destruction efficiencies higher than 99.99 %.

Differences in VOC, SVOC, and PCDD/PCDF emissions from the combustion of untreated and treated wood fuels have been noted. The data do not enable identification of the exact cause of these differences in emissions. These differences are possibly caused by the significantly different moisture content, heating value, and chlorine content of the two fuels. The difference in flue gas flow rate required for the combustion of these two fuels with different combustion characteristics (moisture content and heating value) may also cause the differences in PCDD/PCDF emissions. These emissions are representative only of this facility and configuration and may not be quantitatively comparable to other combustors.

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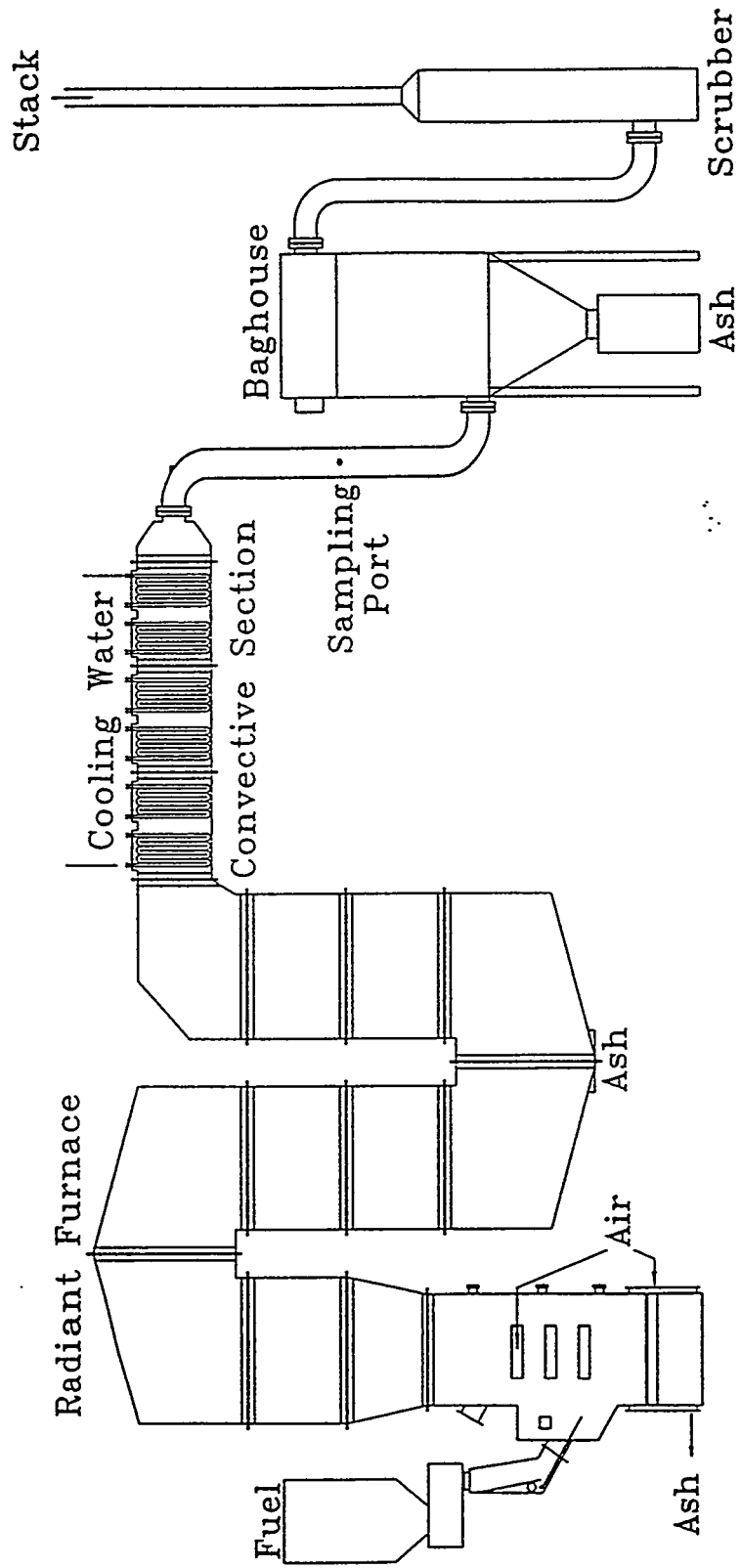


Figure 1. Schematic of the multifuel combustor.

Table 1. Characteristics of wood fuels.

Constituent	Content <sup>a</sup>	
	Untreated Wood	Treated Wood
<u>Proximate Analysis</u>		
Moisture, %	33.44	21.46
Ash, %	0.25	0.57
Volatile Matter, %	57.92	66.28
Fixed Carbon, %	8.39	11.69
Heating Value, kJ/g (Btu/lb)	12.953 (5,569)	16.833 (7,237)
<u>Ultimate Analysis</u>		
Carbon, %	33.83	43.24
Hydrogen, %	4.20	4.92
Nitrogen, %	0.28	0.28
Sulfur, %	0.01	0.05
Oxygen, %	27.99	29.48
<u>Chlorine Analysis</u>		
Pentachlorophenol, ppm	6.7	4,100
2,4-dichlorophenol, ppm	2.4	3.6
Chlorine, %	0.02	0.20

a: Measured on an as-received basis

Table 2. Combustor operating conditions.

Test	Averaged Fuel Feed Rate, kg/hr (lb/hr)	Combustor Thermal Output, MW (million Btu/hr )	Flue Gas Flow, dscm
Untreated 1	134.4 (296.4)	0.48 (1.65)	19.4
Untreated 2	117.1 (258.1)	0.42 (1.44)	16.6
Untreated 3	114.1 (251.6)	0.41 (1.40)	17.9
Treated 1	100.6 (221.8)	0.47 (1.61)	25.0
Treated 2	102.3 (225.6)	0.48 (1.63)	21.0
Treated 3	102.5 (225.9)	0.48 (1.63)	25.8



Table 3. Summary of flue gas temperature and CEM data.

Location	Temperature, °C					
	Untreated Test 1	Untreated Test 2	Untreated Test 3	Treated Test 1	Treated Test 2	Treated Test 3
Stoker Section Exit	849	801	827	869	921	870
Sampling Port	152	153	153	155	160	160

Constituent	Concentration, %					
	Untreated Test 1	Untreated Test 2	Untreated Test 3	Treated Test 1	Treated Test 2	Treated Test 3
O <sub>2</sub>	12.5	13.2	12.9	13.1	13.3	13.3
CO <sub>2</sub>	7.5	7.2	7.1	6.7	6.5	6.5
H <sub>2</sub> O	11.0	9.8	9.8	8.7	8.7	9.0
CO <sup>a</sup>	203	249	456	66	21	20
NO <sup>a</sup>	103	133	83	295	178	210
THC <sup>a</sup>	77	56	2	4	3	3
HCl <sup>a</sup>	<1	<1	<1	195	194	183
SO <sub>2</sub> <sup>a</sup>	10	11	13	<1	<1	<1

a: in ppm @ 7% O<sub>2</sub>

Table 4. Destruction efficiency for pentachlorophenol.

Test	DRE, %
Treated Test 1	>99.9974
Treated Test 2	>99.9976
Treated Test 3	>99.9975

Table 5. VOC emission rates.

VOCs	Emission Rate, $\mu\text{g}/\text{dscm}$ @ 7% O <sub>2</sub>					
	Untreated Test 1	Untreated Test 2	Untreated Test 3	Treated Test 1	Treated Test 2	Treated Test 3
Chloromethane	53.3	28.5	16.9	55.8	96.7	51.5
1,3-Butadiene	0.6	0.2	1.2	0.1	0.1	0.1
Bromomethane	1.7	1.4	2.1	15.4	33.9	70.5
Iodomethane	0.8	2.4	5.7	1.7	5.1	24.1
Acetone	11.6	7.2	7.5	4.5	11.7	14.3
Chloroform	0.9	0.3	0.3	2.6	1.8	3.9
1,2-Dichloroethane	1.3	5.2	0.1	1.6	0.7	2.9
Benzene	25.4	16.2	27.9	2.1	2.6	3.0

Table 6. SVOC emission rates.

Test	Total Emission Rate, $\mu\text{g}/\text{dscm}$ @ 7% O <sub>2</sub>
Untreated Test 1	10.1
Untreated Test 2	11.4
Untreated Test 3	11.7
Treated Test 1	7.0
Treated Test 2	7.7
Treated Test 3	5.9

Table 7. Total PCDD/PCDF emission rates.

Test	Emission Rate <sup>a</sup>			
	Total PCDD, ng/dscm	Total PCDF, ng/dscm	Total PCDD/PCDF, ng/dscm	Total PCDD/PCDF, ng I-TEQ/dscm
Untreated Test 1	8.4	22.3	30.7	0.5
Untreated Test 2	2.6	17.2	19.8	0.2
Untreated Test 3	1.0	11.1	12.1	0.1
Treated Test 1	15.0	42.2	57.2	1.3
Treated Test 2	20.9	56.5	77.4	1.2
Treated Test 3	18.7	45.0	63.7	1.1

a: @ 7% O<sub>2</sub>

Table 8. Distribution of PCDD/PCDF in sampling train.

Test	% of PCDD I-TEQ in Front Half	% of PCDF I-TEQ in Front Half
Untreated Test 1	31.6	24.1
Untreated Test 2	53.0	40.1
Untreated Test 3	49.0	33.3
Treated Test 1	75.8	63.4
Treated Test 2	70.4	59.4
Treated Test 3	68.0	59.7



**TECHNICAL SESSION IV**

***Retrofits & Advanced  
Technologies***



**Retrofitting A Spray Dryer ESP-Equipped Facility**

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## **INTRODUCTION**

During the development of the December 19, 1995 United States Environmental Protection Agency's (EPA) Emission Guidelines for Municipal Waste Combustors, much consideration was given to electrostatic precipitator (ESP) equipped facilities. The cost for replacement of well performing ESPs is prohibitive, especially if the facility is already equipped with a spray dryer absorber (SDA). EPA proposed and promulgated Emission Guidelines which were designed to allow facilities with modern well performing ESPs to comply without replacement of existing equipment.

The Essex County Resource Recovery Facility is one example of a facility equipped with a modern, well performing SDA/ESP combination. Due to actions taken by the State of New Jersey Department of Environmental Protection (DEP), the Essex County Facility has already been retrofitted and is in position to comply with the December, 1995 Emission Guidelines. This paper provides further discussion of the actions taken to retrofit the Essex County Facility and presents limited resultant emission data.

## **FACILITY DESCRIPTION**

The Essex County Resource Recovery Facility is located in Newark, New Jersey and is owned and operated by American Ref-Fuel Company. Refer to Figure 1 for a site layout of the retrofitted facility. The facility processes approximately 2,500 tons of municipal solid waste per day. Operation began in November, 1990 with completion of performance and compliance testing in early 1991.

The facility is comprised of three independent mass-burn processing trains and two steam turbine generators. Each of the three processing trains is equipped with a Deutsche Babcock Anlagen (DBA) Roller Grate System and a four-pass boiler. The DBA Roller Grate System was designed and developed in Europe over 50 years ago and is exclusively licensed to American Ref-Fuel Company in North America.

Municipal solid waste is received on the tipping floor, where it is inspected with any unacceptable waste removed before being deposited into the 11,000 ton capacity refuse bunker. In the bunker, the refuse is mixed and usually inspected by one of two overhead refuse crane operators. The waste is stored in the bunker before being charged to one of the three boiler feed hoppers. Maintaining tight control over the waste before incineration and feeding a consistent mixture of refuse aids in good combustion and controlling emissions within state and federal requirements.

Refuse is charged to the furnace by a ram feeder located at the bottom of the feed chute. Waste slowly tumbles down the roller grates, passing through drying, combustion, and burnout zones before the remaining ash residue falls into the water filled quench tank of one of two ash extractors. Wetting the ash minimizes the potential for fugitive dust and facilitates safe handling. Ferrous metal is recovered from the facility residue stream.

Combustion is closely monitored by a Control Room Operator and is controlled by a computerized combustion system which adjusts the combustion parameters to maintain a constant boiler load and control emissions. Combustion air is provided by an under grate primary air fan and a secondary air fan.

Superheated steam is generated in the pendant superheater sections and tube bundles. An economizer section warms feedwater in the fourth pass resulting in a flue gas exit temperature of about 450-520 degrees F. Much of the ash carried with the flue gas from the furnace drops out as the gases turn between the second and third boiler pass. Superheated steam is collected in a common header from the



three boilers and is routed to two 35 megawatt turbine generators. Approximately 70 megawatts of electricity are produced with 12 percent used for in-house purposes. The steam is condensed in dual air-cooled condensers.

### **Facility Emission Control Equipment**

The flue gas passes from the furnace section through the four pass boiler. After leaving the boiler and economizer, the flue gas is split and travels through one of two precyclones and spray dryer absorbers (SDA). Part of the remaining fly ash is removed in the precyclones. Lime slurry is injected in the dual fluid nozzle SDAs to facilitate removal of acid gases. Slurry injection rate is controlled by the SDA sulfur dioxide (SO<sub>2</sub>) level. More lime slurry is injected as the SDA inlet SO<sub>2</sub> increases. In order to maintain a constant outlet temperature of 300 degrees F, additional temperature control water is atomized in the SDA's.

From the dual SDA's the flue gas passes through an electrostatic precipitator (ESP), induced draft fan and is discharged through a three flue stack. The ESP has three fields which remove remaining particulate and spent lime and salts of reaction from the SDAs. The ESPs have a duct spacing of about 12 inches and a design specific collection area of 575 ft per 1000 acfm. Gas velocity is slightly greater than two feet per second and the aspect ratio is less than 1. The total projected collecting area is almost 100,000 ft.

The fly ash and spent lime are collected and treated with a WES-PHIX® phosphoric acid conditioner and water in a pugmill before being combined with the bottom ash. The combined ash stream is stored in an enclosed bunker. A clam shell overhead crane is used to load trucks within the building. A tarping station completes the ash handling area and allows for controlled fugitive ash emissions.

The facility is equipped with a continuous emission monitoring system and continuous opacity monitoring system. These systems provide the facility feedback on SO<sub>2</sub> concentrations and reduction, carbon monoxide (CO), nitrogen oxide (NO<sub>x</sub>) and opacity levels. This information allows rapid adjustment of the facility operation by the combustion control system and Control Room Operator to maintain in compliance.

### **FACILITY HISTORY**

American Ref-Fuel Company of Essex County obtained an Air Permit to Construct a Municipal Solid Waste Resource Recovery Facility in December, 1985 from the New Jersey Department of Environmental Protection (NJDEP). The original air pollution control equipment included good combustion control, and a dual SDA/ESP. The dual SDA/ESP was selected based upon the recommendations of the project consulting engineer. At the time this selection was made, SDAs were not in common use. Thus, dual 60 percent capacity SDA's were selected to allow for additional redundancy and reliability. In addition, limited long term performance had been demonstrated for fabric filters (FF) and ESP's were the particulate collection device of choice for most existing waste-to-energy facilities and utility boilers.

## **NO<sub>x</sub> CONTROL REQUIREMENTS**

The facility permit, as issued, contained numerous emission requirements. For NO<sub>x</sub>, two requirements were provided: (1) 95 pounds per hour (pph) per unit averaged over three successive test runs using USEPA Method 7; and (2) a one-hour average of not-to-exceed 300 parts per million on a dry basis corrected to seven percent oxygen (ppmdv @ 7% O<sub>2</sub>). The 95 pph was derived by the use of an emission factor of three pounds of NO<sub>x</sub> generated per ton of MSW. Prior to completion of the facility it was confirmed through testing at American Ref-Fuel's Hempstead, New York facility that the commonly used 3 lbs/ton emission factor was in error, based upon incomplete and inaccurate information available at that time, and was not achievable.

A BACT analysis concluded that combustion controls alone were not adequate to bring the facility into compliance with the 95 lb/hr requirement and that Selective Non-Catalytic Reduction (SNCR) was the appropriate add-on control system to be employed by the facility. The evaluation process resulted in the selection of the Thermal deNO<sub>x</sub> aqueous ammonia technology offered by Exxon Corporation. After reviewing the process designed by Exxon, American Ref-Fuel purchased the technology agreement and modified the system to best fit the facility.

## **NO<sub>x</sub> CONTROL RETROFIT**

When it was determined that an SNCR system was required, the plant was too close to its startup date to effectively implement a redesign of the boiler to allow for an integrated NO<sub>x</sub> control system. A retrofit strategy was developed by American Ref-Fuel corporate engineering to implement NO<sub>x</sub> control according to a schedule imposed by the New Jersey DEP.

Technical grade aqueous ammonia (27.4 wt %) is utilized in the Essex thermal deNO<sub>x</sub> process. Aqueous ammonia is stored onsite at atmospheric pressure in a 20,000 gallon tank and pumped to three identical vaporizer skids located in the boiler house. Steam is used to evaporate the aqueous ammonia. The ammonia vapor is mixed with steam prior to being injected into the furnace. This carrier steam aids the dispersion of the relatively small quantity of ammonia.

The ammonia vapor/steam mixture can be injected into the boiler through one of three available sets of injection nozzles located on the side walls of the upper first pass of the waterwall boiler. At each level nine nozzles are available on each side of the boiler. The nozzles vary in size and are arranged in a pattern of alternating relatively large and small nozzles at each level.

Because this project was an application of relatively new technology being applied in a retrofit manner on boilers with existing unique emission control technology, as much flexibility as possible was designed into the system to allow for variations in operation such as boiler load and fouling.

Several challenges occurred for this project. First, no provisions were made during the original boiler design for penetration at the required locations. Locating the storage tank, vaporizers and other required equipment was a challenge on the congested site. Routing of feed lines, electrical power supplies, and controls needed to conform to the existing layout. Installation was completed while the facility was operating with tie-in occurring during scheduled boiler outages.

## **NO<sub>x</sub> CONTROL SYSTEM OPERATION**

There were no comparable systems to use to predict performance, thus questions remained regarding the potential system capability. Testing flexibility was hampered by a requirement to remain within permit limits unless a specific variance was granted by NJDEP. American Ref-Fuel wanted to maintain its clear stack and thus did not want any ammonia slip reacting with HCl to form a visible ammonia chloride (NH<sub>4</sub>Cl) plume. Concerns were also raised and addressed concerning public and employee safety due to a potential aqueous ammonia spill. Assurances had to be made that proper programs were in place to prevent spills and control them if necessary.

Several modifications were made to the initial design. Adjustments were made to the control logics, carrier steam supply and the nozzles used for injection. Only two problems were identified during startup. The first was a metallurgy compatibility concern which was quickly addressed. Secondly, the system vaporizers were and are an ongoing concern due to significant fouling. About once a month each system must be taken off-line for cleaning. This cleaning must be carefully prepared for to allow the system to return to service and maintain compliance under NJDEP's permit requirement of a three hour rolling average.

The thermal deNO<sub>x</sub> system is controlled from the plant's distributed control system (DCS). The function of the control system is to maintain the target outlet NO<sub>x</sub> concentration by regulating the aqueous ammonia flow rate. Changing of levels or zones is completed manually. Levels are changed infrequently as the boiler conditions change.

The SNCR system was installed in late 1994 and has worked well since. Compliance with the proposed MACT emission limit and averaging time is achievable. The system is routinely achieving an NJDEP requirement of 174 ppm<sub>dv</sub> @ 7% O<sub>2</sub> on a 3-hour rolling average and 95 lb/hr. The outstanding concern of achieving the NJDEP NO<sub>x</sub> emission requirement on a 3-hour rolling average basis during a required outage remains as a special challenge for the facility.

## **MERCURY CONTROL REQUIREMENTS**

In addition to the SNCR system, the facility has also been retrofitted with an activated carbon injection (CI) system. The original permit requirement for mercury emissions was 0.053 lb/hr. This value was derived from early test data at other waste-to-energy facilities which later proved to be lower than typical emission rates. Although the Essex County Facility was burdened with an artificially low mercury emission requirement, the facility was fortunate to have had mercury emissions among the lowest of any waste-to-energy facility in New Jersey before activated carbon systems were installed. While mercury had always been an emission of concern under the existing permit, aggressive measures taken by the NJDEP and the facility were successful in minimizing mercury in the waste stream and allow the facility as initially equipped to comply with its permit limits.

The Essex Facility and American Ref-Fuel worked with several efforts to remove mercury from the waste stream. Recycling programs and information were communicated to the various regional hospitals and pharmacies. These sources were assumed to be the largest consumers and dispensers of mercury batteries and hearing aid batteries. American Ref-Fuel also actively participated on a New Jersey DEP task force commissioned to address mercury emissions and establish control requirements.

The revised New Jersey mercury emission requirements for all facilities are presently 65 µg/dscm corrected to 7% O<sub>2</sub> or 80 percent reduction. The concentration requirement is based upon an annual

average of four quarters where as the reduction requirement must be demonstrated every quarter. A further requirement will lower the allowable concentration to 28  $\mu\text{g}/\text{dscm}$  or 80 percent removal in the year 2000. The new mercury regulations also required that the Essex County Facility install a system to control mercury.

## **MERCURY CONTROL RETROFIT**

Various mercury control technologies were reviewed by all facility operators and the New Jersey DEP to determine which system granted the greatest potential for success. Any system would also need to be capable of increasing control at a future date. It was determined that this objective could best be achieved with a dry activated carbon system and the regulations therefore required that this type of system must be installed at all New Jersey facilities. Norit Americas Inc. was selected by the Essex County Facility as the vendor and design and construction proceeded as expeditiously as possible.

The Norit technology employed utilizes a dry carbon of lignite origin and injects the carbon into the flue gas ahead of the system precyclone and spray dryer absorber. The carbon is handled dry and is pneumatically conveyed to the injection points. Safety concerns of carbon powder were also considered and addressed with the facility design by installation of controls, blowout panels, fugitive dust control and operating procedures.

To provide some additional flexibility, two potential injection points were provided, upstream of the spray dryer atomizer and in the spray dryer vessel. The injection points are shown in the unit cross section in Figure 2. These points were suggested due to concerns about dropping out a significant quantity of carbon in the precyclone and for ease of installation. Early testing determined that the longer residence time and improved distribution of injecting upstream of the precyclones more than compensated for any loss of carbon in the precyclones.

Construction proceeded on a fast track to achieve the required completion date of before December 31, 1995. Several retrofit related modifications were made during this time slowing construction completion. Partially because all the facilities in New Jersey were installing systems, certain critical vendor capabilities were also in short supply.

Difficulties associated with most retrofit projects, including routing of carbon lines, power lines and controls, limited siting options and existing foundations also were incurred and addressed. As previously noted, the Essex site is very congested and locating a carbon silo and blower building was not easy. Penetrations to the existing units only involved penetrating through plate steel, however, access to the chosen injection locations is very restrictive. Work proceeded while the facility was on line.

## **MERCURY CONTROL SYSTEM OPERATION**

Experience with carbon injection on SDA/ESP equipped facilities was very limited, consisting largely of the Warren County EPA carbon injection study. No long term reliable data was available for any similar systems. Concerns were raised over possible reduction in the particulate control performance of the ESPs. Carbon can demonstrate the ability to rapidly lose an electrical charge and thus lose its attraction with an ESP and "skip" through the ESP fields without being captured.

The activated carbon injection system installation was completed in late December, 1995 and the system has been in operation since. The ability of the facility to achieve the December, 1995 Emission Guideline requirements has been demonstrated. The NJDEP emission requirements of 65  $\mu\text{g}/\text{dscm}$  corrected to 7%

O<sub>2</sub> or 80 percent reduction, has also been demonstrated to be achievable. Accurate measurement of carbon injection rates are completed by measuring usage on a shift basis and verifying this value on a plant-wide basis by using the truck scales and silo measurements on a monthly basis.

## **FACILITY PERFORMANCE**

Limited emissions data is available for the retrofitted Essex County Facility. The facility has demonstrated exceptional emission results for all testing completed since the retrofits were completed. The emission results are compared to the Emission Guidelines promulgated on December 19, 1995 in Table 1 and show favorable results for all parameters impacted by the retrofit. An added benefit of the mercury control system is the excellent dioxin control achieved. If significant revisions are made to the Emission Guidelines in the process of any legal proceeding or if the interpretation or application of the Guidelines by NJDEP were to significantly revise the Guidelines for the State of New Jersey, meeting the revised requirements may be substantially more difficult. For example, if the NO<sub>x</sub> emission requirement is applied on a 3-hour rolling average as opposed to a 24-hour block basis, even though the concentration requirement is not adjusted, the emission requirement is substantially more difficult.

## **CONCLUSION**

The Essex County Resource Recovery Facility has been retrofitted and has the emission control configuration of Good Combustion Control/SDA/ESP/SNCR/CI. This configuration is capable of achieving the emission performance requirements of the 1995 EPA Emission Guidelines. This project demonstrates that the emission requirements of the Guidelines as promulgated can be achieved by some facilities equipped with well performing ESPs.

## **REFERENCES**

1. Pohlot, P.D., Armellino, K.E., Zapf, S., et al.; *Retrofit Thermal DeNO<sub>x</sub> Installation and Operation at a Resource Recovery Facility Equipped with a Spray Dryer/ESP*; Proceedings of 1995 Air and Waste Management Association, Solid Waste Management: Thermal Treatment and Waste-to-Energy Technologies, Washington, D.C., V1P-53; 1995.

Table 1. Essex County Emission Comparison of Selected Parameters Post-Retrofit.

Parameter	Facility Average Corrected to 7% O <sub>2</sub>			December 19, 1995 Emission Guideline
	Unit 1	Unit 2	Unit 3	
Particulate, gr/dscf	0.0025	0.0010	0.0010	0.012
NO <sub>x</sub> , ppm <sub>dv</sub>	less than 180	less than 180	less than 180	200
Mercury, µg/dscm	27	22	27	80 or
percent reduction	81%	85%	80%	85%
Total Dioxins <sup>①</sup> , ng/dscm	3.0	1.5	2.0	60

① Total dioxins is reported as tetra-octa polychlorinated dibenzo-p-dioxin and tetra-octa polychlorinated dibenzo furans reported on a total basis.

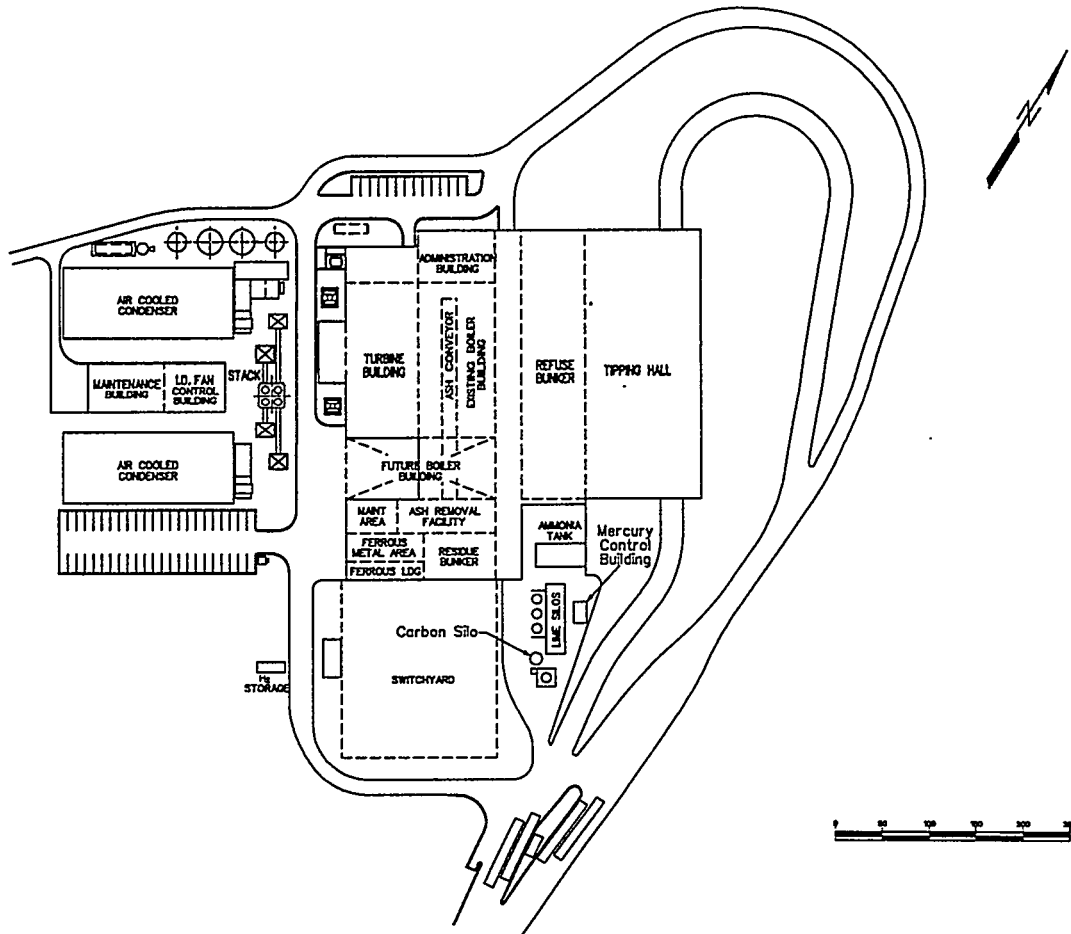


Figure 1 Essex County Resource Recovery Facility Site

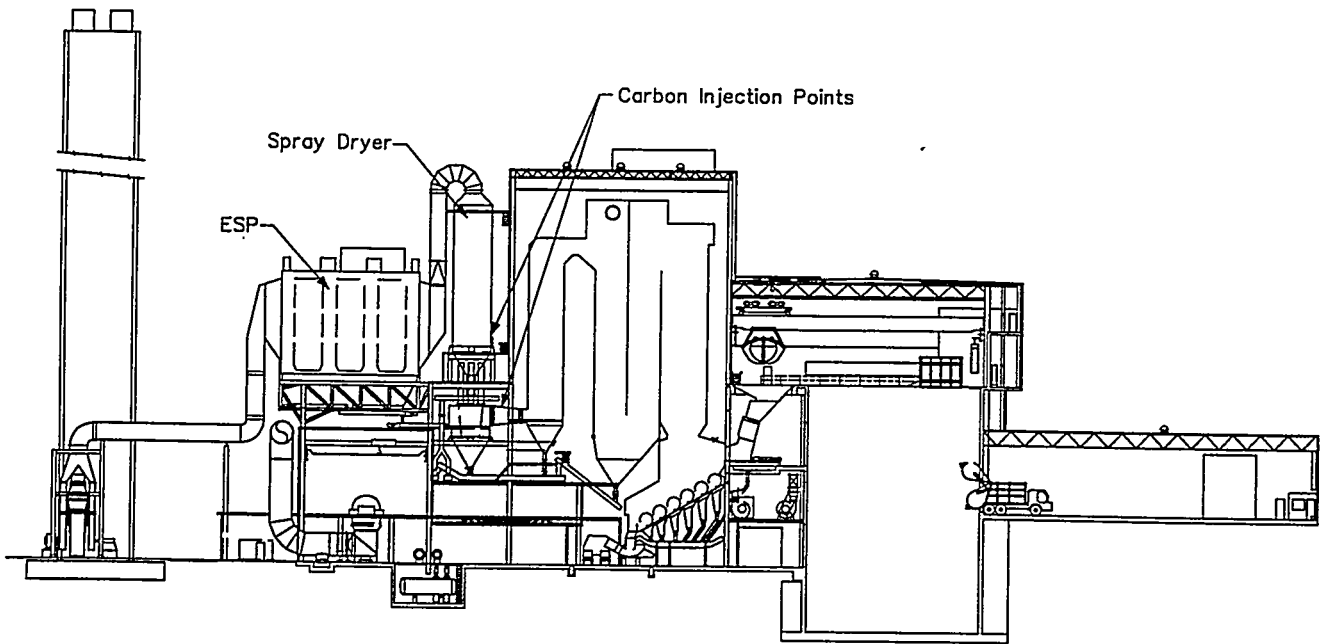


Figure 2. Essex County Resource Recovery Facility Cross-Section showing Carbon Injection Location





**Combined Heat Recovery and Dry Scrubbing for MWCs  
To Meet the New EPA Guidelines**

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## INTRODUCTION

Both the UK and US Municipal Waste Combuster (MWC) markets have undergone upgraded regulatory control. In the UK, the government's Integrated Pollution Control (IPC) regime, enforced by the 1990 Environmental Protection Act (EPA) Standard IPR5/3 moved control of emissions of MWCs from local councils to the government Environmental Authority (EA). Existing MWCs had until December 1, 1996 to complete environmental upgrades. Simultaneously, the European Community (EC) was finalizing more stringent legislation to take place in the year 2001.

In the US, the 1990 Clean Air Act amendments required the Environmental Protection Agency (EPA) to issue emission guidelines for new and existing facilities. Existing facilities are likely to have only until the end of 1999 to complete upgrades.

In the UK, retrofits had considerable incentive to maximize energy recovery because of government incentives for energy production from renewable resources. Four facilities with a total capacity of 900,000 tpy opted for energy recovery and air pollution control. Our company in the UK received contracts, which included energy recovery equipment as part of Air Pollution Control (APC) contracts at:

1. Coventry and Solihull Waste Disposal Company(CSWDC), originally with a capacity of 175,000 tpy but upgraded to 260,000 tpy.
2. London Waste Company, Edmonton, with a capacity of 690,000 tpy.

In North America, our company had received contracts from Kvaerner EnviroPower AB, for APC systems on four new Refuse Derived Fuel (RDF) fluid bed boilers that incorporated low outlet temperature economizers as part of the original boiler equipment. The Fayetteville, North Carolina facility was designed for 200,000 tpy.

What all these facilities have in common is low economizer outlet temperatures of 285°F coupled with our Total Dry Scrubbing Systems. MWC or RDF facilities using conventional spray dryer/fabric filter combinations have to have economizer gas outlet temperatures about 430°F to allow for evaporation of the lime slurry in the spray dryer without the likelihood of wall build up or moisture carry over. Since the Totally Dry Scrubbing System can operate with economizer gas outlet temperatures about 285°F, the added energy available for sale from adding low outlet temperature economizer heat recovery can be considerable.(Figure 1). This paper focuses on Procedair's new plant and retrofit experience using "Dry" Venturi Reactor/Fabric Filter combination with this lower inlet temperature operating conditions.

## SYSTEM COMPONENTS

The principal components in designing a high heat recovery/dry scrubber system are:

1. Heat recovery economizer designed for low outlet temperature (less than 290°F).
2. Dry venturi reactor for hydrated lime and powdered activated carbon injection and primary gas reaction with the boiler flue gases.
3. Fabric filter dust collector for particulate collection and secondary gas reaction.
4. Material handling for recycle of partially reacted and unreacted reagent and waste disposal.

## Heat Recovery Economizer

Gas temperature has a major effect on the performance of any dry-scrubbing system, particularly the SO<sub>2</sub> removal efficiencies as shown in (Figure 2). The designs utilized for the enhanced heat recovery for the Coventry and Edmonton retrofits utilized finned tubes to provide heat recovery in minimal space. Figure 3 shows typical designs of a single and double finned tube for economizers. Finned tube economizers have been used in Europe for this application for some time as shown in (Table 1). The Uppsala economizer is installed after the primary electrostatic precipitators and therefore sees relatively little dust (about .02 gr/dscf). The SYSAV economizer is located after a primary cyclone separator, and therefore, sees only about 1.0 gr/dscf dust load. The economizers installed on the Coventry and Edmonton retrofits have no pre-cleaners ahead of them. They therefore see the total flyash load. They are also installed before the acid gases are removed from the gas.

The Coventry economizers have three banks of finned tubes. The first two banks have carbon steel fins and the third or outlet bank has cast iron fins. The Edmonton retrofit used a first-stage bare tube section followed two carbon steel finned tube sections. Design details from the Coventry economizer is shown in Table 2. Both systems were integrated into the existing boiler heat cycle. The economizer subcontractors undertook extensive studies of the existing economizers and boiler steam drum designs to insure the added steam production could be handled by the existing boiler design. Both systems feature water recirculation systems to control the exit flue gas temperatures between 275°F and 285°F. The first of the two economizers for London Waste was put on line in June, 1995 with subsequent units being added about every three months. The three economizers at Coventry were put on line between twelve to fifteen months ago. To date, no cleaning other than normal soot blowing has been required for any of the units at Edmonton or Coventry. The units have been inspected every boiler outage (between eight to ten weeks). However, there is no sign of dust build up or corrosion.

## Dry Venturi Reactor

Our vertical dry venturi reactor shown in Figure 4 has been developed over more than 20 years. The gases pass through a convergent zone into the reaction throat where hydrated lime and powdered activated carbon are injected into the flue gas. The reactor tower is basically a vertical double annulus, which provides for both the homogenous mixing of totally dry reagents with the flue gas constituents and also insures the necessary residence time (contact time) for the reagent in the gas stream. This use of a vertical arrangement insures that there is minimal stratification with resultant uneven or poor neutralization. The injection of the reagents into the gas stream through a venturi insures complete distribution of the reagent in this turbulent flow section. Pressure drop across the Venturi Reactor is only 2" H<sub>2</sub>O, compared to 10" H<sub>2</sub>O for conventional fluid bed reactors and 2 ½" H<sub>2</sub>O for spray dryer type absorbers.

## Fabric Filter Dust Collector

Both the Coventry and Edmonton installations are equipped with our TGT pulse jet fabric filter units for the final cleaning stage. This is a modern state-of-the-art pulse jet fabric filter utilizes low pressure compressed air (25 to 45 psig), injected by means of our proprietary Integrated Action Piston (IAP) valve for high-volume low-pressure cleaning. The IAP valve replaces the conventional diaphragm valve and overcomes many of the disadvantages of this type of valve. Due to its large open section and piston operation, the pressure drop across the IAP valve is extremely low compared to a conventional diaphragm valve, with significant energy savings. The principal advantage, however, of the IAP valve is

that unlike the diaphragm valve, which both snaps open and snaps shut rapidly, it permits a rapid opening with a slow controlled closing sequence. The snapping shut of the diaphragm valve causes the fabric to slap back onto the cage creating bag to cage wear, shock wave problems and thus dust carry through. With the IAP valve, the bag is rapidly inflated for maximum cleaning efficiency, but is deflated slowly. Reducing the pressure within the bag progressively permits it to regain its normal filtration position in a slow and controlled manner and to obtain for the fabric what is termed as a "soft landing." The use of soft landing reduces the wear of the fabric caused by the fibers rubbing against each other and also the shock of the fabric returning to the support cage. This translates into longer bag life and lower emission levels. Prior to the development of the IAP valve, the only method by which filter bag could be cleaned in a similar manner was to isolate individual cells from the main gas flow and to clean the bags in that cell of line. This however, frequently requires the use of an additional compartment to allow for off-line cleaning. This method of cleaning not only does away with the need for the additional compartments, but also enables longer filter bags (greater than 16 ft) to be cleaned by the pulse jet method. At both Coventry and Edmonton, the bags are 23 ft long. By using long bags, and on-line cleaning, the footprint of the filter unit is minimized, which can be a very important consideration when retrofitting, especially in cases like Coventry, where the majority of equipment had to be installed inside the existing glass-walled building.

The TGT dust collector makes use of a proprietary baffled inlet (Figure 5), that provides the following benefits:

1. Directs the gases upwards and across the bags reducing greatly the interstitial ("can velocity") velocity.
2. Deflects a large part of the dust directly into the hopper to reduce the dust load on the bags.
3. Allows units to operate with on-line cleaning at high filtering velocities of 4 fpm, which again reduces the equipment footprint:

### **Powdered Activated Carbon Injection**

Increasingly, emphasis is being given by the Regulatory authorities to the emission of mercury and Dioxin/furans (PCDD/PCDF). Considerable work has been done on the removal of mercury and PCDD/PCDF from flue gas streams. With the Dry System, the Powdered Activated Carbon (PAC) is injected separately into the Venturi reactor. Procedair has considerable experience in the removal of polyhydrocarbons using dry injection of powdered coke or carbon products in to venturi reactors. This is derived from more than fifteen years of supplying APC equipment to control the hydrocarbon emissions from green anode plants in the primary aluminum industry.

### **Material Handling for the Recycling of the Partially Reacted Reagent**

The fabric filter separates the flyash/acid salts and unused lime. These dry solid particles are constantly evacuated from the baghouse hoppers and, in the Procedair Dry System, are in the main recycled. Without recycling, it is a fact that even with a good reactor tower design and secondary neutralization in the filter cake in the bag house, there will be over 60% of the lime unreacted and going to waste. This has a major impact on operational costs due to the need for high stoichiometric ratios to achieve regulatory emission control levels. By using reactor product recirculation, there is a five-fold benefit created by the "apparent stoichiometric ratio" at the Venturi Reactor (Figure 6).

1. Unreacted lime will be afforded further opportunity to be used, lowering fresh stoichiometric ratios and operating costs.
2. Recirculation, at rates of between 30 and 50 times the fresh lime feed rate, means the mass of reagent within the gas stream increases the probability of contact by pollutant molecules with reagent, thus increasing removal efficiency.
3. The multiple handling and passage of the particulates due to recirculation causes a particle size reduction, partially through attrition, and partially due to reacted surface material breaking away exposing unreacted material.
4. The continuous mass of material recirculating throughout the process acts as a buffer dampening peak pollution production levels and compensates for any reduction or interruption in fresh reagent flow.
5. The high recirculation rate also provides for recirculation of the activated carbon. Without recirculation, activated carbon would have to work on a once-through basis. Recycling reduces the usage of activated carbon at the same time minimizing PCDD/PCDF and mercury emissions.

These aspects are demonstrated in Figure 7, which is a comparison of systems with and without recirculation. Consistent higher removal efficiencies are obtained at lower fresh lime stoichiometric ratios with recirculation. Higher predictability is gained with recirculation. With the commercial necessity to guarantee consistently low emission levels at minimized stoichiometric ratios, these advantages are very important.

## **OPERATIONAL FACILITIES**

Dry scrubbing of MWC five gases has gone through three phases which are shown on Table 3. The first phase is dry-scrubbing from earlier installations, at temperatures of 400-450°F. Here emission levels were not as strictly regulated and the control of SO<sub>2</sub> frequently not required. The second is where the dry system is modified by the addition of an evaporative cooling tower between the boiler and the dry-scrubbing system. This permitted capture of SO<sub>2</sub> and consistent operation with the higher efficiencies required by earlier updates of regulations. The two reasons for this improvement over a dry-system are:

1. Inlet temperature to the scrubbing system can be very precisely controlled and for optimum performance lowered to 285°F without increasing corrosion risks in heat recovery equipment.
2. The relative humidity of the flue gas increased by the water which contributes to improved scrubbing performance.

The third, our present generation operating systems utilizing low outlet temperature economizers with totally dry-scrubbing systems to achieve the latest regulatory emission control levels.

### **BCH Energy, Fayetteville, NC.**

The BCH Energy facility comprised two Kvaerner bubbling fluid bed combustors burning RDF. Each combustor was rated at 300 tpd. Economizers, provided for the boilers by Kvaerner, cool the gases

below 290°F. It was felt that the use of the bubbling fluid bed would lead to higher flyash production due to the carryover of bed sand particles with the RDF flyash. For this reason the APC equipment starts with a cyclonic precleaner to reduce the dust load to the APC system. The design conditions for the BCH Energy project are shown in Table 4. The scope of supply for the equipment for the APC equipment is shown on Table 5. Results of the emission tests performed on the two boilers in 1996. Shown in Table 6, indicates that the APC equipment performance was better than that required by the permit.

### **Coventry and Solihull Waste Disposal Company (CSWDC)**

At the time of tendering, the CSWDC facility comprised three mass burn water wall incinerators utilizing Martin grate technology with a unit capacity of 320 tpd of MSW. The original operating permit for the facility was based on any two furnaces operating at one time; therefore, the existing air pollution control equipment consisted of only two lines of electrostatic precipitators for flyash removal through a manifold system at the outlet of the three boilers.

The CSWDC facility added an electrical generating steam turbine in 1991 to take advantage of the revenue available from electricity generation. The retrofit contract called for three sets of air pollution control equipment to allow the facility to operate all three at a time. This increased both the waste throughput revenue and the electrical generation revenue.

During the bidding process, extensive visits were made to other facilities in Europe and North America by the Coventry operational personnel. After talking directly to the operational staff at these facilities, two opinions were formed.

1. Despite having demonstrated its ability to meet the required emission standards, the semi-dry process (spray-dryer with lime slurry with fabric filter) in its various forms had the considerable disadvantages of requiring the preparation and handling of lime slurry and its atomization into the flue gas stream in a spray dryer.
2. The totally dry-scrubbing technology was seen to offer the potential to meet the new required standards with fewer potential operational difficulties and maintenance requirements. It was realized also that this system would allow the recovery of additional energy using an extended economizer rather than wasting energy by spraying water into the spray dryers.

The scope of work for the CWSDC retrofit is shown in Table 7 and the design criteria for the CWSDC system in Table 8. The Coventry facility has been tested and the test results are shown in Table 9.

### **London Waste Limited**

The London Waste Limited, Edmonton facility was originally owned and operated by five London Boroughs. These formed a joint venture with SITA (GB) Limited, to form a company called London Waste Limited (LWL). The facility consists of five, 450 tpd mass burn grates and boilers, giving a

capacity of 690,000 tones/year. Each boiler was equipped with an economizer for heat recovery and electrostatic precipitator for dust removal. At the LWL facility, a decision was made during the bidding stage that the upgraded facility would be capable of meeting the more stringent European Community (EC) legislation for 2001. This meant that the outlet HCl emission had to be guaranteed at 10 mg/Nm<sup>3</sup> (6 ppm<sub>dv</sub>) against a high potential inlet condition of 2,000 mg/Nm<sup>3</sup> (1230 ppm<sub>dv</sub>). At the time, it was thought necessary to combine the precise temperature control and added humidification from a conditioned dry system with the heat recovery/dry system in order to have no risks on the guarantees. As a result, the LWL facility decided to upgrade the performance of the economizer but only to reduce the flue gas outlet temperature down to 330°F and install an evaporative cooling tower, to further cool the gases to 285°F. This condition would insure compliance with the UK standard IPR 5/3. To meet the more stringent EC standards, the economizer outlet temperature could be raised to 365°F allowing for increased humidification of the flue gas. It should be noted that CWSDC decided to take the maximum heat recovery during the early stages of the operation so increasing the energy income until 2001. Space, however, has been left in the facility building to allow the retrofitting of evaporative conditioning towers to provide conditioned dry systems should CWSDC need this in order to meet the EC 2001 regulations.

The design conditions and scope of supply for Edmonton are shown in Tables 10-11. The facility has undergone performance tests and the results are shown in Table 12. These show that the units can practically meet the more stringent EC 2001 when operated at the lower economizer outlet temperature.

## CONCLUSIONS

The use of a Heat Recovery/Dry-Scrubbing System for retrofit MSW installations provides the following:

1. Increased energy recovery from an existing facility is possible within the existing boiler design parameters. The finned tube economizers can reduce the gas outlet temperatures to below 285°F and control the outlet temperature at this level when combined with a temperature control system.
2. The capital costs for the Heat Recovery/Dry-Scrubbing System is competitive with conventional spray-dryer/fabric filter combinations. The facility gains the extra benefit of the added revenue from increased energy production.
3. The Heat Recovery/Dry Scrubbing System can provide performance levels that exceed those currently envisaged by the most recent EPA regulations for MWCs.
4. The footprint for the Heat Recovery/Dry-Scrubbing System is generally smaller than that required for the spray dryer/fabric filter combination.
5. The total gas volume to be handled by the dry scrubber is reduced by heat recovery instead of humidification, resulting in less fan horsepower. The pressure drop for the enlarged economizer/venturi reactor is approximately the same as for a spray dryer.
6. The use of powdered activated carbon (PAC) within the dry-scrubbing system is highly effective for the removal of mercury and PCDD/PCDF compounds. This removal is achieved at low consumption rates for PAC due to the recycling of the PAC with the other reagent compounds.
7. Finned tube economizers can operate with dirty gas and low outlet temperatures for extended periods without plugging and corrosion.

The heat recovery/dry scrubbing system has particular application to those facilities selling electricity or steam. The additional steam raised can be turned into useful energy, generally within the capability of the

existing boiler/turbine combination. This extra energy will supply the parasitic power needed for the APC equipment, and will allow additional export of power for revenue purposes.

### **Acknowledgements**

The authors wish to Acknowledgments the assistance given by CSWDC and LWL personnel in providing information for this paper. They also wish to thank Kvaerner and BCH Energy for their help with test data.



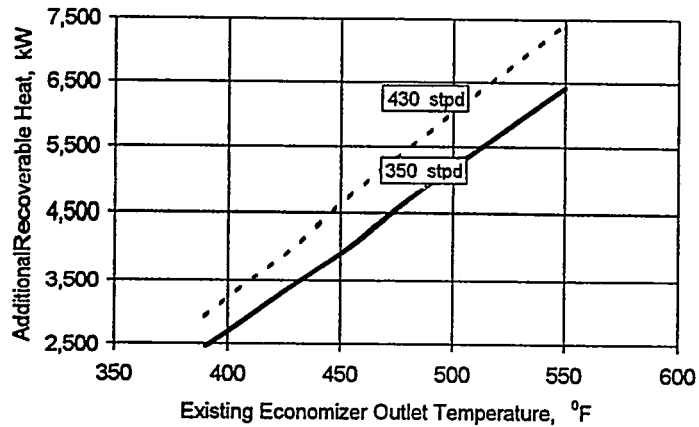


Figure 1. Additional Heat Recovery from Municipal Waste Boilers assuming 290 °F from New Economizer Section

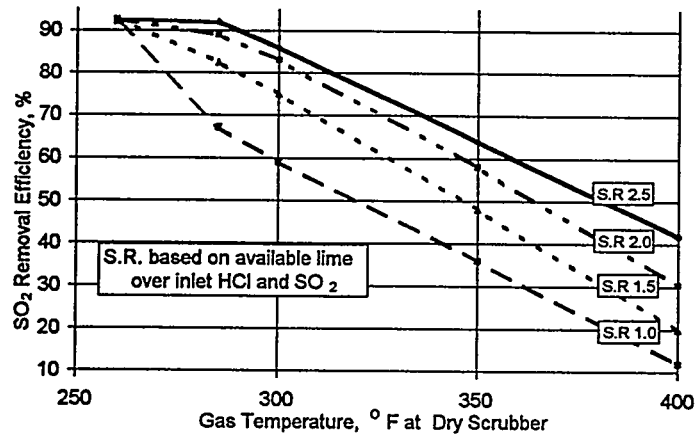


Figure 2. SO<sub>2</sub> Removal Efficiency by Dry Scrubbing at Various Gas Temperatures and Stoichiometric Ratios (S.R.)

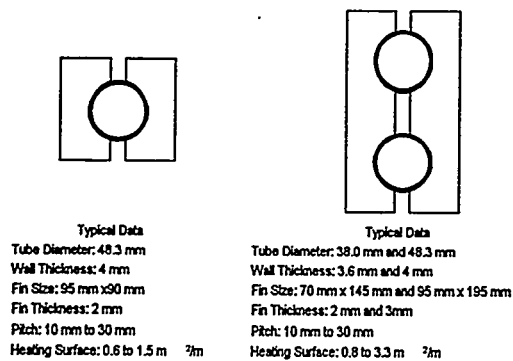


Figure 3. Typical Specifications for Single and Double Finned Tubes for Economizers for MSW Boilers

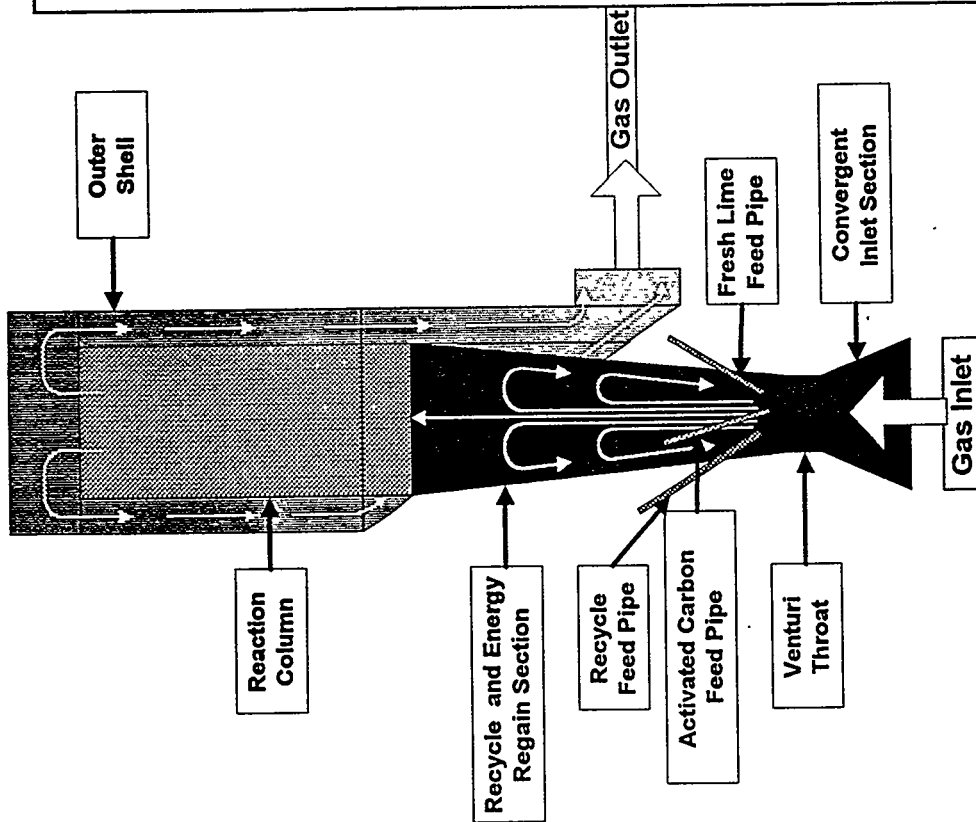


Figure 4. Dry Vertical Venturi Reactor

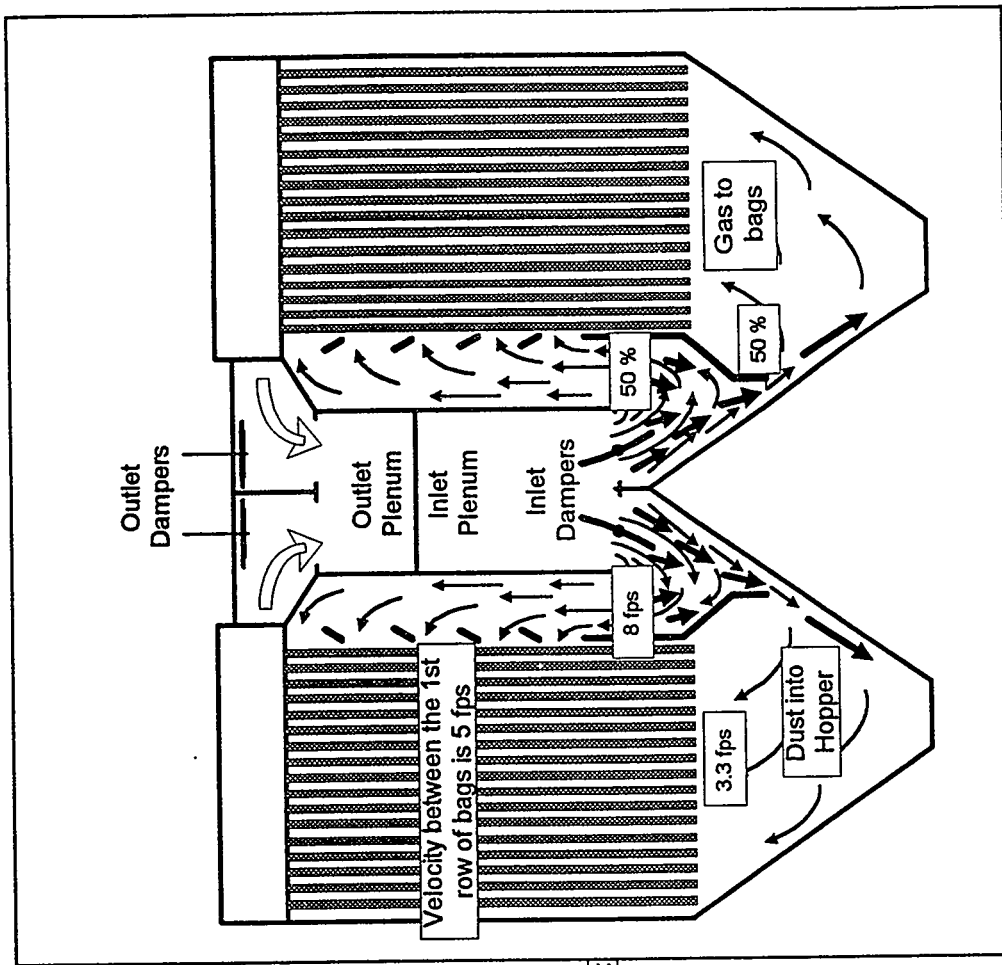


Figure 5. Baffled Inlet Design for TGT low pressure jet filters

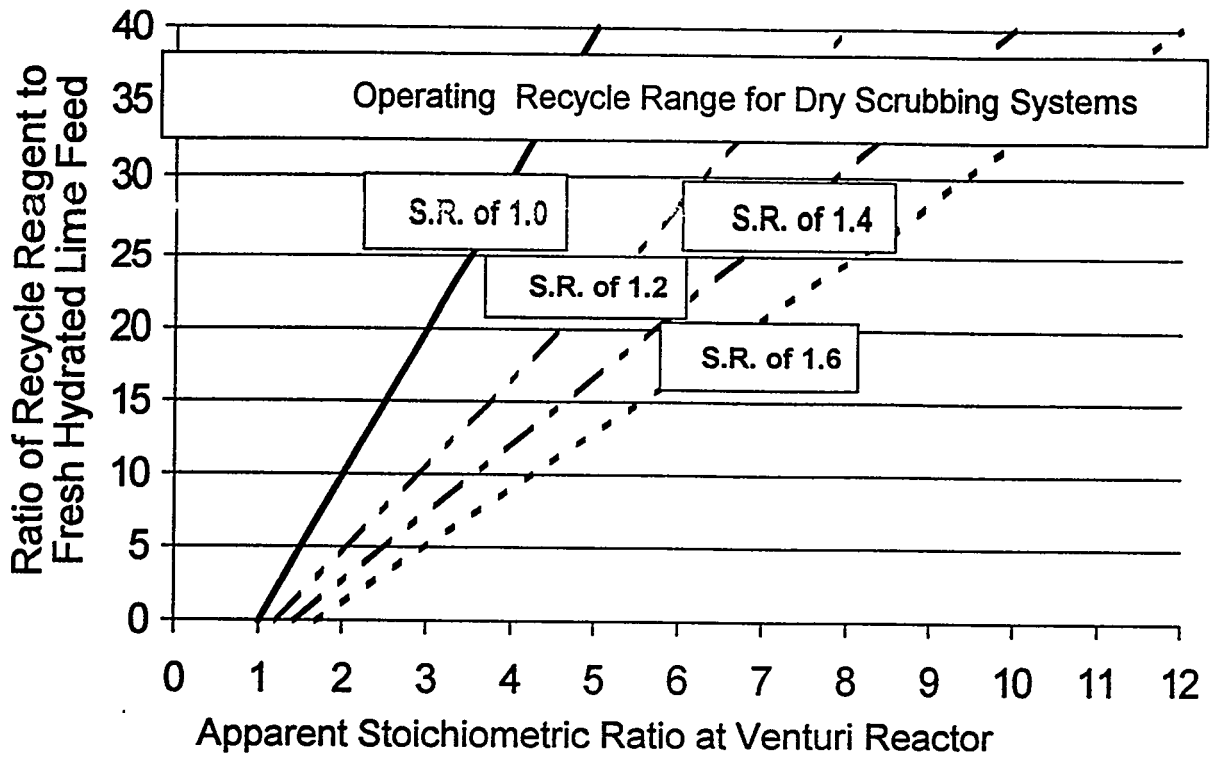


Figure 6. Effect of Recycle Rate on Apparent Stoichiometric Ratio at Venturi Reactor

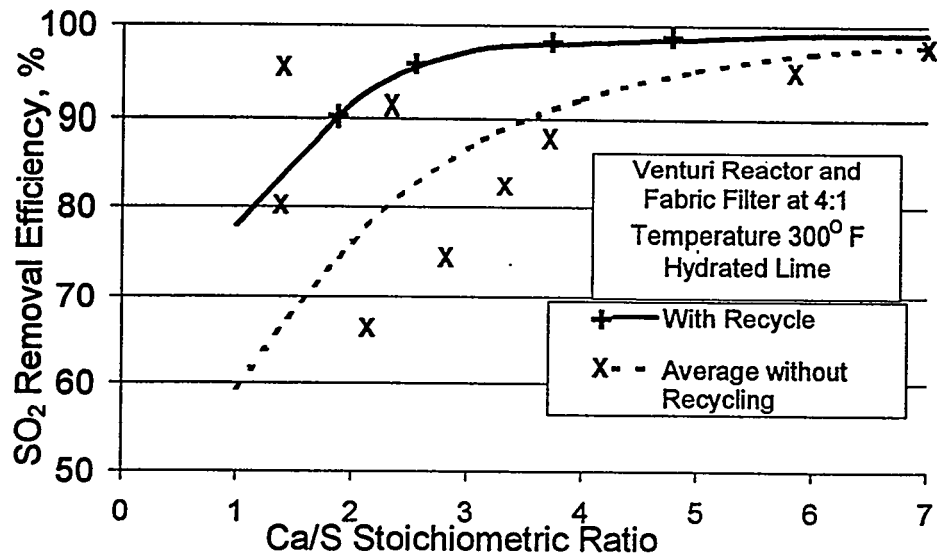


Figure 7. Pilot Plant Dry Scrubber System SO<sub>2</sub> Removal Capabilities

Flue Gas Flow: 45,000 SCFM

Feed Water Flow: 119,000 lb/hr  
(includes 20,000 lb/hr recirculation)

Feed Water / Hotwell  
Temperature: 194 ° F

Delivery Date	SYSAV	Uppsala	London Waste	Coventry
No. Units	1994 One	1991 One	1995 Five	1996 Three
Dust Load	0.8740	0.0218	1	4.0
Gas Flow	97,347	236,789	120,950	92,628
Inlet Temperature	572	482	550	626
Outlet Temperature	354	284	331	284
Water Flow	22,046	855,385	110,340	146,826
Inlet Temperature	275	248	248	266
Outlet Temperature	333	284	370	388
Tube Type	Double	Double	Bare & Double	Double
Material	Carbon Steel	Carbon Steel	Carbon Steel	Carbon Steel
Tube Diameter	1.50	1.50	1.50	1.50
Tube Length	8.5	20	19	15
No. Finned Tubes	630	660	1584	1584
Fin Pitch	0.8	0.8	1.0	0.9
Cleaning Method	Shot Cleaning	Shot Cleaning	Steam Blower	Steam Blower

Table 1. Typical Finned Tube Economizers

Gas Inlet, ° F	Gas Outlet, ° F	Water Inlet, ° F	Water Outlet, ° F	LMTD, ° F	Heat Transfer, MW	Boiler Cleanliness
Economizer outlet gas temperature controlled to 284 ° F						
536	284	257	365	109	3.6	Clean
626	284	257	342	126	5.0	Average
698	284	257	374	136	6.0	Fouled
Economizer outlet gas temperature controlled to 428 ° F						
536	428	257	302	232	1.6	Clean
626	428	257	342	255	3.0	Average
698	428	257	374	271	4.0	Fouled

Table 2. Coventry and Solihull Waste Disposal Company Economizer Heat Transfer Data

Dry Process	Year	Size	Gas Volume- acfm
Roselare, France	1987	2 X 100 tpd	2 X 30,400 at 482 ° F
Edegem, Belgium	1987	2 X 50 tpd	2 X 15,800 at 482 ° F
U. I. O. M., France	1988	2 X 100 tpd, Von Roll	2 X 22,300 at 464 ° F
<b>Conditioned Dry Process</b>			
Community of Quebec, Canada	1988	4 X 250 tpd, Von Roll	4 X 105,000 at 480 ° F
City of Windham, USA	1989	1 X 200 tpd	1 X 71,700 at 518 ° F
Peel Resource Recovery, Canada	1992	4 X 100 tpd, Consumat	2 X 103,400 at 500 ° F
<b>Heat Recovery /Dry Process</b>			
Kvaerner EnviroPower, USA	1994	2 X 300 tpd, Fluid Bed	2 X 63,500 at 290 ° F
CSWDC, UK	1995	3 X 320 tpd, Martin	3 X 64,210 at 285 ° F
LWL, UK	1995	5 X 450 tpd	4 X 110,000 at 285 ° F
Kvaerner EnviroPower, USA	1997	1 X 600 tpd, Fluid Bed	1 X 95,000 at 290 ° F

Table 3. Progression of Dry Scrubbing from Dry-Dry at High Temperature to Conditioned Dry to Heat Recovery and Total Dry

● Gas Flow	62,400 ACFM	● One (1) Hydrated Lime Handling System
● Temperature	293° F	● with Redundant Feeders
● Gas Analysis	H <sub>2</sub> O 13.1 to 17.3	● Procedair Activated Carbon Storage and Metered Feed System
● % by vol	CO <sub>2</sub> 11.0	● Two (2) Procedair Duplex Cyclones
	O <sub>2</sub> 6.1	● Two (2) Procedair Dry Venturi Reactors
	N <sub>2</sub> 69.7	● Two (2) Procedair Pulse jet Baghouses,
● SO <sub>2</sub> Concentration	240 ppm <sub>dv</sub>	● Reagent Recycling and Dust Handling System
● at 7% O <sub>2</sub>		● Controls and Instrumentation
● HCl Concentration	761 ppm <sub>dv</sub>	
● at 7% O <sub>2</sub>		
● HF Concentration	35 ppm <sub>dv</sub>	
● at 7% O <sub>2</sub>		
● Dust Concentration	4.5	
● gr/dscf at 7% O <sub>2</sub>		

Table 4. BCH Energy Design Inlet Conditions

Table 5. Equipment Supplied to Kvaerner for BCH Energy, Fayetteville, NC.

Stack Emission		Heavy Metal	mg/dscm @ 7% O <sub>2</sub>	Unit No 1	Unit No 2
Particulate	< 0.002 gr/dscf	Arsenic	<0.0003	<0.0004	<0.0004
SO <sub>2</sub>	< 0.25 ppm <sub>dv</sub>	Beryllium	ND < 0.0001	ND < 0.0001	ND < 0.0001
HCl	< 1.5 ppm <sub>dv</sub>	Cadmium	<0.001	<0.0004	<0.0004
Opacity	0	Chromium 6	ND < 0.0001	<0.0001	<0.0001
Mercury	< 1.0 µg/dscm	Lead	<0.015	<0.002	<0.002
PCDD/PCDF	< 3.0 ng/dscm	Mercury	<0.001	<0.001	<.002

Table 6. BCH Energy, Stack Emissions  
Averaged from Performance Test Data

- One (1) Hydrated Lime Handling System
- Three (3) Vertical Counterflow Economizers
- Three (3) Proceadair Venturi Reactors
- Three (3) TGT Low Pressure Pulse Baghouses
- Recycling, Waste Handling and Ash Silos
- Controls and Instrumentation and CEMs
- Activated Carbon Injection System
- Three (3) new Induced Draft Fans and Motors
- Three (3) pair of oil-fired Auxilliary Burners

Table 7. APC Equipment supplied and installed under the Contract with CSWDC

- Gas Flow 94,000 ACFM
- Temperature 630 °F
- Gas Analysis
- % by vol H<sub>2</sub>O 11.0  
CO<sub>2</sub> 11.0  
O<sub>2</sub> 10.0  
N<sub>2</sub> 68.0
- SO<sub>2</sub> Concentration 195 ppm<sub>dv</sub> at 7% O<sub>2</sub>
- HCl Concentration 950 ppm<sub>dv</sub> at 7% O<sub>2</sub>
- Dust Concentration 4.0 gr/dscf at 7% O<sub>2</sub>

Table 8. Unit Design Inlet Conditions for CSWDC



	Inlet Levels		Outlet Levels	
	g/Nm <sup>3</sup> @ 11% O <sub>2</sub>	ppmdv @ 7% O <sub>2</sub>	g/Nm <sup>3</sup> @ 11% O <sub>2</sub>	ppmdv @ 7% O <sub>2</sub>
Dioxins TEQ			0.025	
HCl	423	365	4.9	4.2
HF	1.8	2.8	0.1	0.16
SO <sub>2</sub>	165.6	81.3	0.5	0.25
	g/Nm <sup>3</sup> @ 11% O <sub>2</sub>	gr/dscf @ 7% O <sub>2</sub>	mg/Nm <sup>3</sup> @ 11% O <sub>2</sub>	gr/dscf @ 7% O <sub>2</sub>
Particulate	2.46	1.40	3.2	0.002
Total Metals	0.021	0.012	1.4	0.0008

Tin	mg/dscm @ 7% O <sub>2</sub> .07	Chromium	mg/dscm @ 7% O <sub>2</sub> .07
Cadmium	.01	Copper	.11
Lead	.07	Mercury	.81
Nickel	1.6	Arsenic	.01

Period- 1996	HCl			SO <sub>2</sub>			Hydrated Lime		Gas	
	Inlet	Outlet	Efficiency	Inlet	Outlet	Efficiency	lb/hr	S.R.	Volume	Temperature
	ppmdv	ppmdv	%	ppmdv	ppmdv	%				
Aug 7 to 10	551	15	97.35	120	14	88.55	242	1.92	62,536	284
Aug 18 to 21	537	12	97.84	131	12	91.23	271	2.04	46,986	280
Aug 23 to 26	492	14	97.17	128	20	84.66	239	2.10	55,579	286
Aug 31 to Sep 2	501	10	98.05	147	10	93.34	235	1.61	44,714	271

Table 9, CSWDC Emission Summary Line 1  
Independent Performance Tests performed 7/8/96

● Gas Flow	208,000 ACFM
● Temperature	930 °F
● Gas Analysis	
● % by vol	H <sub>2</sub> O 10.0
	CO <sub>2</sub> 9.0
	O <sub>2</sub> 9.4
	N <sub>2</sub> 71.6
● SO <sub>2</sub> Concentration	180 ppmdv at 7% O <sub>2</sub>
● HCl Concentration	1720 ppmdv at 7% O <sub>2</sub>
● Dust Concentration	0.4 gr/dscf at 7% O <sub>2</sub>

Table 10. London Waste Limited, Edmonton  
Design Inlet Conditions

- One (1) Hydrated Lime Handling System
- Five (5) Economizers
- Four (4) Evaporative Conditioning Towers
- Four (4) Proceair Venturi Reactors
- Four (4) TGT Low Pressure Pulse Baghouses
- Recycling, Waste Handling and Ash Silos
- Controls and Instrumentation and CEMs
- Activated Carbon Injection System
- New Induced Draft Fans and Motors

Table 11. APC Equipment supplied and installed under  
the Contract with LWL

Test to IPR 5/3			HCl Emissions			SO <sub>2</sub> Emissions			Hydrated Lime	
Date	Test	Start	Limit	Actual	% of Limit	Limit	Actual	% of Limit	Actual	Actual
			ppmdv	ppmdv	%	ppmdv	ppmdv	%	lb/hr	S.R.
2/9/97	1	11:00	25.9	8.5	33.0	147.3	6.0	4.1	420.6	1.73
	2	12:00	25.9	5.8	22.3	147.3	3.3	2.3	419.1	1.86
	3	13:00	25.9	3.5	13.3	147.3	3.1	2.1	416.7	1.52
	4	14:00	25.9	7.8	30.0	147.3	6.8	4.6	412.9	1.29
2/11/97	1	10:00	25.9	3.9	15.0	147.3	2.7	1.8	447.8	1.47
	2	11:00	25.9	3.7	14.3	147.3	4.5	3.0	445.5	1.44
	3	12:00	25.9	7.0	27.0	147.3	4.2	2.9	448.4	1.57
	4	13:00	25.9	6.8	26.3	147.3	4.9	3.3	446.4	1.81
2/11/97	1	14:00	25.9	7.9	30.7	147.3	6.1	4.1	451.1	1.85
	2	15:00	25.9	7.2	28.0	147.3	4.3	2.9	443.3	1.75
	3	16:00	25.9	11.2	43.3	147.3	11.8	8.0	442.7	1.41
	4	17:00	25.9	13.7	53.0	147.3	12.0	8.1	446.7	1.56
2/14/97	1	13:00	25.9	6.7	26.0	147.3	11.2	7.6	361.8	2.52
	2	14:00	25.9	5.1	19.7	147.3	5.4	3.6	358.9	2.03
	3	15:00	25.9	2.6	10.0	147.3	4.4	3.0	358.5	1.8
	4	16:00	25.9	3.2	12.3	147.3	3.1	2.1	360.2	3.12
2/15/97	1	12:00	25.9	4.7	18.0	147.3	6.9	4.7	438.7	1.69
	2	13:00	25.9	4.4	17.0	147.3	6.4	4.4	433.4	1.78
	3	14:00	25.9	4.9	19.0	147.3	7.8	5.3	436.3	1.89
	4	15:00	25.9	4.3	16.7	147.3	7.2	4.9	435.4	1.7
Mean			25.9	6.1	23.8	147.3	6.1	4.1	421.2	1.79

Table 12. Performance Test Results to IPR 5/3 for London Waste Limited, Edmonton

