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Emissions from the Burning of Vegetative Debris in Air Curtain Destructors

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ABSTRACT

Although air curtain destructors (ACDs) have been used for quite some time to dispose of vegetative debris, relatively little in-depth testing has been conducted to quantify emissions of pollutants other than carbon monoxide and particulate matter. As part of an effort to prepare for possible use of ACDs to dispose of the enormous volumes of debris generated by Hurricanes Katrina and Rita, the literature on ACD emissions was reviewed to identify potential environmental issues associated with ACD disposal of construction and demolition (C&D) debris. Although no data have been published on emissions from C&D debris combustion in an ACD, a few studies provided information on emissions from the combustion of vegetative debris. These studies are reviewed, and the results compared to studies of open burning of biomass. Combustion of vegetative debris in ACD units results in significantly lower emissions of particulate matter and carbon monoxide per unit mass of debris compared to open pile burning. The available data are not sufficient to make general estimates regarding emissions of organic or metal compounds. The highly transient nature of the ACD combustion process, a minimal degree of operational control, and significant variability in debris properties make prediction of ACD emissions impossible in general. Results of scoping tests conducted in preparation for possible in-depth emissions tests demonstrate the challenges associated with sampling ACD emissions, and highlight the transient nature of the process. The environmental impacts of widespread use of ACDs for disposal of vegetative debris and their potential use to reduce the volume of C&D debris in future disaster response

scenarios remain a considerable gap in understanding the risks associated with debris disposal options.

INTRODUCTION AND BACKGROUND

On August 29, 2005, Hurricane Katrina came ashore along the Gulf Coast, with the eye initially passing over Plaquemines Parish with 140 mph[1] wind speeds, then continuing north and hitting the Louisiana/Mississippi border with wind speeds still over 125 mph. Less than a month later, Hurricane Rita made landfall near the Texas/Louisiana border as a major hurricane with 120 mph wind speeds. Both of these storms produced major storm surges, which combined with the high winds to create enormous amounts of debris from destroyed structures, downed trees, and other vegetative debris. Hurricane Wilma struck the southern Gulf Coast of Florida a month later as a major hurricane, again leaving behind a trail of damage and substantial debris. It is clear that 2005 was a landmark year for hurricanes and tropical storms, but 2004 was also notable for the damage caused by four hurricanes to hit Florida. Although the 2006 hurricane season was relatively quiet for the U.S., it has been predicted that the number of hurricanes and tropical storms will continue to be above the long-term average for at least the next decade, and that the intensity of storms may also be increasing.[2-4] These predictions, in addition to the increases in building and population in coastal areas prone to hurricane damage,[5] emphasize the need to understand the implications of options for disposal of hurricane debris.

The debris left by each of these storms presented and continues to present tremendous challenges for disaster recovery efforts, first in the efforts to restore transportation, power, and communications, but ultimately in the need to dispose of massive volumes of solid waste material. Hurricanes often create debris measured in millions of cubic yards, and can overwhelm local waste management capabilities.[6] The volume of debris from Hurricane Katrina has been estimated to be on the order of 100 million cubic yards, or approximately 50 million tons, including vegetative and construction and demolition (C&D) debris, household hazardous wastes, white goods (including large appliances), and waste containers (including propane and fuel tanks).[7] This compares to an estimated

245 million tons per year of municipal solid waste generated in the U.S. in 2005, which translates to approximately 3.7 million tons per year in Louisiana, based on per capita waste generation estimates.[8] Clearly, the debris generated by Hurricanes Katrina and Rita represented a major and very sudden increase in the level of solid waste that required disposal.

One of the methods used to reduce this enormous volume of debris is to burn combustible material, either in large open piles or using air curtain incinerators, also called air curtain destructors (ACDs). More controlled combustion processes, such as found in municipal solid waste combustion systems, may not be suitable due to distance from the disaster site or because of design or regulatory limits on the properties of the waste feed. Even so, any combustion process, and particularly uncontrolled combustion without flue gas cleaning systems, generates potentially significant levels of pollutants that could be emitted into the air. The use of ACDs to reduce the volume of hurricane debris is therefore an approach that carries with it the potential for additional and possibly lasting environmental damage. To develop a comprehensive and protective plan for responding to future disasters, it is important to understand the capabilities and potential risks associated with debris burning and its alternatives, including landfilling, grinding, material reuse, and use as or conversion to alternative fuels.

The purpose of this paper is to discuss the results of pilot- and full-scale tests that have been previously reported, and to compare those results with results from open pile burning of debris and limited testing of emissions from an ACD conducted during the U.S. Environmental Protection Agency's (EPA) response to Hurricane Katrina.

AIR CURTAIN DESTRUCTOR

Air curtain destructors are generally used to dispose of vegetative debris, such as from large land clearing or forest management operations. These units operate by burning the combustible material in an enclosed space with an open top, over which a high velocity "curtain" of air is directed to reduce the escape of large particles and to improve air circulation into the burning debris. Figure 1 illustrates the general operation of an ACD.

In this schematic diagram, the air flow inside the firebox is depicted as flowing in a generally circular pattern, counterclockwise. The circulation in an actual unit is much more complex (as will be discussed below) but in general provides circulation of air into the combustion zone, and recirculates at least a portion of combustion byproducts back into the high temperature combustion region surrounding the debris. This combination of high air flow into the combustion zone and recirculation of the combustion products is designed to reduce visible particulate matter (PM) emissions and provide increased gas-phase residence times compared to open pile burning.

There are several types of ACD designs. The firebox can be a pit dug into the ground with a transportable blower and curtain air plenum positioned to blow the curtain air over and down into the pit.[9] These designs are common in applications such as destruction of forest clearing debris because they are relatively light and can be towed into remote areas with poor roads. A second type of ACD uses a refractory-lined firebox that is entirely above ground.[10] These are approximately the size of a large waste dumpster and incorporate the air curtain fan on the same skid as the firebox. A third design extends the side and back walls of the firebox to minimize the impact of wind and may also incorporate provisions for introducing combustion air (underfire air) into the firebox, underneath the debris to improve the airflow through the combustion zone. This type of unit cannot be transported as an integral unit and can require a week or more to set up and begin operations. In some cases, such as shown in Figure 2, these larger units have a more complex loading arrangement. Other variants on the design include misters or even secondary combustion chambers.

For all these designs, the operation when burning vegetative debris is fundamentally the same. The initial charge of debris is loaded into the unit and ignited, usually with diesel fuel or kerosene. Once the debris has ignited, the blower is started and additional debris is loaded into the unit as needed to maintain combustion. The ignition process can generate a temporary puff of black smoke as the diesel fuel ignites, and smoke typically increases for a brief period as subsequent loads of debris are loaded. Generally, no auxiliary fuel is used to maintain good combustion within the unit.

PREVIOUS WORK

Published data on emissions from ACD units are scarce. There have been a number of studies of open burning over the past decade that provide some basis for comparison. For ACDs, however, the data are less available and tend to be less detailed. A brief description of ACD operation and emissions published by the U.S. Forest Service[11] is an example of the available documents that describe ACD operations and describe emissions, although only qualitatively.

CO and PM

There are a few studies that have been done over the past 40 years on ACD emissions. Data from three full-scale tests have been found, and two papers that evaluated pilot-scale ACD systems are also in the literature. The earliest of the full-scale studies was published in 1972 by Lambert, who described emissions from a pit-type ACD unit used to burn forest-clearing debris.[12] The pit in which the debris was burned was 41 feet long, 8 feet wide, and 15 feet deep (12.5 m x 2.4 m x 4.6 m). Lambert reported emissions of carbon monoxide (CO) and carbon dioxide (CO_2), Ringlemann smoke number, and temperatures. CO and CO_2 were measured using a continuous emission monitor (CEMs), PM measurements were taken using a high volume ("hi-vol") ambient sampler, and opacity was measured using the Ringelmann visual method (although no longer in official use in the U.S., the Ringelmann method has been used since the 1880s, as described in Ref. 13; a brief but more technical discussion of the method can be found in Ref. 14). Temperatures were measured with thermocouples in the debris bed and with an optical pyrometer. The average CO measured was 140 ppm, with CO_2 at 0.75%. PM levels were reported as "too low to measure," although opacities were reported to be at ½ Ringlemann smoke number (5%) for 95% of the operating time. During unit startup, the opacity was reported to be 40% (Ringlemann 2), and when diesel fuel was introduced to ignite the bed, Ringlemann numbers as high as 8 were noted.[12]

Temperatures were consistently found to be at least 1600 °F (920 °C), and increased with time. Peak temperatures of over 2200 °F (1250 °C) were measured after 11 hours of operation in one test, and as high as 2500 °F (1420 °C) in a separate test. Average steady state temperatures were measured at 1950 °F (1120 °C), dropping to 1450 °F (840 °C) within an hour after the blower was turned off. Lambert reported that overloading of the unit by piling logs 3 feet above the air curtain did not visibly change opacity, and also noted the presence of a blue flame along the length of the air plenum that was visible during night operations. He attributed this flame to the combustion of volatile products that were forced up the air plenum side wall into an area with adequate oxygen to allow combustion to be completed. Lambert also described burning railroad ties in the unit, which produced heavy black smoke below the air curtain, with the pit surface "uniformly covered with a sheet of bright orange flames all along the air curtain."

A more recent report describing emissions from ACDs was prepared by Fountainhead Engineering in 2000.[15] This study reports emissions from an above-ground ACD unit (Air Burners Model S-127), and provides data on CO, CO_2, and PM emissions and opacity sampled at the top of the ACD unit. Over four test runs, the average CO concentration was measured at 54 ppm at CO_2 levels of 0.2%, suggesting a greater level of dilution than in the Lambert study. PM concentrations were measured at 6600 $\mu g/m^3$, and emission rates were reported at 2.14 lb/hr (0.97 kg/hr). At this concentration, opacity levels measured using EPA Method 9[16] were found to range between 4% and 7.5%, with an average of 5.4%.

The final full scale measurements were reported by Trespalacios describing operation of an ACD burning vegetative hurricane debris in Toa Baja, Puerto Rico.[17] This study measured pollutants on the perimeter of the ACD operation site rather than taking samples from the outlet of the unit. The unit used in this operation was similar to that used in the Fountainhead tests. Concentrations of CO and PM were measured at points 50 (15 m) and 100 feet (30 m) upwind and downwind from the ACD. More detailed data on organic and metal emissions were also collected.

Average CO concentrations 50 ft (15 m) downwind from the ACD were 9.3 ppm, and average PM concentrations at the same location were 570 $\mu g/m^3$. The average values are for three measurements taken over 10 s intervals shortly after loading “dirty debris,” a wet mixture of soil and vegetative debris. It is very possible that the steady state averages would be lower than those reported, but additional measurements were not reported.

Assuming a factor of 10 dilution at 50 ft (93 ppm CO at the ACD face), the corresponding PM concentrations would be 5700 μg/m^3, which is consistent with the Fountainhead data. It is unclear what the actual dilution factor is, but the relationship between CO and PM in the Trespalacios study appears to be of the same order of magnitude as the Fountainhead data. A factor of 6 dilution would result in CO of 56 ppm (vs. 54 ppm for the Fountainhead measurements) and PM of 3400 μg/m^3, yielding a PM:CO ratio roughly half that of the Fountainhead unit.

There are three pilot-scale studies reported in the literature that are also relevant. The first of these was published in 1968, and is very consistent with the later full scale results noted above. Burckle et al. burned cordwood, municipal solid waste (MSW), and tires in a pilot-scale ACD.[18] The unit was 3 ft wide x 3 ft long x 4 ft deep (0.9 m x 0.9 m x 1.2 m) in size. When burning a 318 lb (144 kg) charge of wood with the air curtain fan operating at 420 scfm (11.9 sm^3/min), CO was measured at 100-1000 ppm over the course of the test. The tests were conducted for a single fuel charge, and measurements were initiated after combustion had stabilized and continued until the fuel charge burned out. CO_2 concentrations while burning wood at these conditions ranged from 0.1% to 1.75%. These values are consistent with the range of concentrations from full scale units.

PM was measured at 0.53 grains/dscf (1.2 g/m^3), corrected to 12% CO_2, and an emission factor of 12.7 lb/ton of fuel was calculated for this test. This concentration value seems quite high, but if the values are corrected to 0.2% CO_2 (the CO_2 concentrations reported in the Fountainhead study), the concentrations are much closer to the results reported for the full scale units. The correction to 12% CO_2 reflects a comparison with enclosed combustion systems such as boilers or incinerators, and may not be as appropriate a comparison as a lower CO_2 concentration likely to be measured in an open burning situation. At 0.2% CO_2, PM concentrations would be approximately 20,000 $\mu g/m^3$, which is roughly three times the value reported in the Fountainhead report. Although this is considerably higher than the 6600 $\mu g/m^3$ value from the Fountainhead report, it is reasonably close considering the difference in scale. Differences in fuel and scale make it difficult to conclude whether the two results are in fact comparable or if the similarity is simply coincidental. In either case, the Burckle paper provides valuable information on the transient nature of the process.

An interesting aspect of the Burckle paper is the finding that PM emissions from wood were relatively insensitive to air flow rate. For municipal solid waste (MSW) and tires, the PM emissions increased linearly (as measured by lb/ton of fuel burned) with increasing air flow. Burckle et al. attributed this to the higher ash content of both MSW and tires compared to wood. They do not explain why the PM emissions increase with air flow, but do suggest that other fuels such as sawdust may exhibit higher PM emissions as smaller particles could be entrained in the air/gas flows and carried out of the unit. This may have significant implications for burning of C&D debris, which will have significantly higher levels of incombustibles and will also likely have higher levels of dust and debris fragments that can be stirred up by handling and loading activities and the air curtain and possible underfire air flows.

The second pilot-scale study is less directly applicable, but does provide some additional insight into the ACD combustion process. Linak et al. burned 1 lb (454 gm) charges of black polyethylene agricultural sheet plastic in a 1 ft x 1 ft x 1 ft (0.3 m x 0.3 m x 0.3 m) pilot-scale unit and made detailed measurements of the organic compounds emitted from the process.[19] They also measured CO and PM and compared the results with and without the use of a simulated air curtain. The peak CO dropped slightly from 42 ppm without the air curtain to 37 ppm with the air curtain on when burning new sheet plastic. With the same fuel, average as-measured CO concentrations dropped from 29 ppm with the air curtain off to 23 ppm when the air curtain was on. It is unclear whether this reduction was due to improved performance or to dilution of the CO by the curtain air.

Interestingly, PM concentrations increased when using the simulated air curtain. For the new sheet plastic, PM concentrations were measured at 4730 μg/m^3 when the air curtain was on and at 3560 μg/m^3 when it was off. CO_2 levels during these tests were reported to "vary minimally" in the range of 0.3-0.6%. Although the fuel used in these tests was significantly different, the reported PM concentrations were very similar to those reported in the full scale tests. It is unclear whether this is due to similarities in the combustion process or coincidental.

The third pilot-scale study was reported by Lutes and Kariher, and focused on the open burning of land clearing debris.[20] Samples of woody debris from Tennessee and Florida were burned in a 36 in (91 cm) long x 18 in (46 cm) wide x 16 in (41 cm) deep pilot-scale ACD. The unit was tested with curtain air on and off. Results for CO and $PM_{2.5}$ showed only minor changes in concentration, but significant reductions in mass of emissions per unit mass of fuel burned. Average CO concentrations were reported at 34 ppm without the blower and 37 ppm with the blower (as measured conditions), and average $PM_{2.5}$ concentrations were measured at 24,600 μg/m^3 without the blower and 40,400 μg/m^3 with the blower. On the basis of emissions per unit mass of fuel, CO fell from 20 g/kg without the blower to 12 g/kg with the blower. Similarly, $PM_{2.5}$ emissions fell from 12 g/kg without the blower to 10 g/kg with the blower. The higher concentrations of CO and PM but lower emissions per unit mass of fuel are a consequence of the more rapid consumption of fuel when the blower was used. For comparable fuel charges, the rate of fuel consumption when the blower was used was as much as two times faster than when the blower was not used.[20] This leads to higher average pollutant concentrations, but over a shorter period of time. For full-scale units, the dilution of the exhaust gases by ambient air entrained into the exhaust plume may also lead to significant differences in reported exhaust concentrations. The dilution will depend upon where the sample is collected – both the location across the opening of the ACD and how high above the ACD exit the sample is collected.

Organics, Metals, and Other Emissions

The Lambert study did not measure emissions other than CO_2, CO, and PM.[12] The Fountainhead study did take SO_2 and NO_2 measurements, but concentrations were sporadic. SO_2 was found on only one of four runs (at 1 ppm) and NO_2 measured inconsistently at 1-4 ppm.[15]

More extensive measurements of trace compounds were taken during the Toa Baja study. Several metals and organics were measured upwind and downwind of the ACD in an effort to quantify emissions from the combustion process. Six metals were detected in the samples: aluminum (Al), cadmium (Cd), chromium (Cr), iron (Fe), lead (Pb), and potassium (K). Al and Fe were detected more consistently, but were also detected at higher levels upwind than downwind, on average. Of the remaining metals, only Pb was detected in more than one of the 8 samples collected, and then in only two.[17] It is probable that the higher Al and Fe concentrations are the consequence of the use of "dirty fuel" (wet vegetation combined with soil). The high upwind values further suggest these metals are the result of soil-borne elements, and also make it questionable whether the downwind samples were emitted from the ACD as opposed to being from fugitive dust. Because emissions of metals are very strongly dependent upon the composition of the fuel, the applicability of these results to other units is limited to a recognition that ACD combustion conditions appear to be adequate to result in the formation of metal-containing particles.

Of the volatile organic compounds detected, benzene, toluene, chloromethane, and formaldehyde were detected in each downwind sample at concentrations higher than the upwind sample. Besides these compounds, only p-xylene and propionaldehyde were detected in more than two (of eight) samples. The downwind propionaldehyde concentrations were each lower than the upwind concentration, suggesting that the source was not the combustion process. p-xylene was detected at levels above the upwind concentration in at least one sample during each test, indicating that the source was the ACD.

Concentrations of polychorinated dibenzo-d-dioxins (PCDDs) and polychorinated dibenzofurans (PCDFs) were also measured upwind and downwind of the ACD unit. Toxic equivalent (TEQ) concentrations of PCDDs and PCDFs were not detected in the upwind samples, but were detected in four of six downwind samples, indicating the formation of PCDDs/Fs in the combustion process. No other semivolatile organics were detected in the Toa Baja samples.[17]

The pilot-scale study by Burckle et al. measured total hydrocarbons, carbonyls (reported as formaldehyde), and carboxyls (reported as acetic acid). A continuous hydrocarbon (HC) monitor showed a large initial spike in HC concentrations, which then decreased to a minimum and gradually increased again as the fuel charge burned out. The spike occurred within 10 min of loading, and the second peak was reported at about 60 min after loading. The reported data suggest that increasing curtain air flow can, in at least some cases, result in increased HC concentrations for wood, tires, and MSW (see Figure 3). In general, HC emissions (on the basis of mass of emissions per mass of fuel) from wood were lower than HC emissions from tires, which were in turn lower than the HC emissions from MSW.[18]

The pilot scale work reported by Linak et al. noted that, "The use of forced air slightly reduced the time necessary to burn each charge, but it did not affect the types or concentrations of PICs [products of incomplete combustion] emitted." [19] The study identified 37 volatile and semivolatile organic compounds in the collected samples, as well as 18 polycyclic aromatic hydrocarbons (PAHs). Linak et al. noted that emissions of benzene, toluene, ethyl benzene, and 1-hexane increased slightly when the forced air was operating and suggested that this may be due to quenching of the flame by the cooler forced air. With the exception of this difference, there was little reported change in the measured emissions of organic compounds due to changes in fuel type or operating condition.

Comparison of ACD and Open Burning Emissions

For PM, the general range of open biomass burning emissions tends to be on the order of 10 g/kg of fuel. Gerstle and Kemnitz reported total PM emissions of 6-12 g/kg for landscape refuse and 7-9 g/kg for municipal refuse.[21] For different types of biomass, Andreae and Merlet reported $PM_{2.5}$ emissions between 4 and 13 g/kg and total PM emissions up to 18 g/kg,[22] Lemieux et al. reported values for total PM emissions ranging between 10 and 19 g/kg, [23] and Dennis et al. used emissions of 8-19 g/kg of total PM in their estimates of total emissions from open burning.[24] These values are significantly higher than the emissions per unit mass of fuel calculated in the Fountainhead test, which were at 0.05 g/kg. It is very likely that even poorly operated systems will exhibit significantly lower PM emission levels when they are able to increase the high-temperature residence time of the pyrolyzed organics that form most of the fine PM. For instance, an early dedicated vegetative debris burner was reported to have experienced "excessive smoke" due to overcharging the unit during a test. Even at this overload condition, the reported PM emissions were 1.1 g/kg, roughly an order of magnitude lower than uncontrolled open burning.[25] There are likely to be some differences in emissions due to the different biomass types and burning conditions, which ranged from prescribed burning of savanna to burning of residential vegetative debris in small piles. Even so, the PM emission factors for open burning are relatively consistent given the wide variety of materials and conditions.

Interestingly, the pilot-scale results were similar to the reported uncontrolled open burning emission rates. It is unclear why the pilot-scale ACD studies were unable to similarly duplicate the reduced PM concentrations indicated by the full-scale ACD tests, but the complexities involved in achieving simultaneous similarity in combustion, heat transfer, and air flow may be a significant barrier to effective pilot-scale evaluations of ACD emissions, at least without using a computational fluid dynamics (CFD) approach to the design of the experimental apparatus. The hypothesis noted above that a higher residence time may play a significant role in reducing emissions of organics and unburned fuel could well be one of the reasons why pilot-scale units have not been able to effectively simulate the emission rates measured during tests of full-scale units. Scaling an ACD system to simulate physical conditions may well result in combustion product residence times that are below characteristic reaction times needed to achieve higher burnout levels.

Reported emissions of CO from open biomass burning varied significantly. Andreae and Merlet reported values of 65-110 g/kg of CO, Lemieux et al. reported a range of 16-110 g/kg, and Gerstle and Kemnitz reported 25-40 g/kg of CO emissions.[21-23] Hays et al. reported CO concentrations of 4-2000 ppm for the several types of biomass tested, with similar ranges over 2-3 orders of magnitude noted for each type.[26] These values compare to approximately 0.5 g/kg of CO from an ACD burning woody debris as reported in the Fountainhead study.[27] (The full report notes a CO emission rate of approximately 20 lb/hr, compared to a PM emission rate of 2.1 lb/hr for the same runs. The calculated PM mass emission factor of 0.054 g/kg of fuel is provided in a separate letter to the Georgia Department of Natural Resources. The 0.054 g/kg PM emission factor would result in a CO mass emission factor of approximately 0.5 g/kg of fuel, given the same fuel feed rate during the test runs.) The reduction in CO is not as substantial as that shown for PM, but is still significant. It is difficult to directly compare the Lambert and Toa Baja results because of a lack of fuel feed data, but they seem to be of the same order of magnitude as the Fountainhead study (assuming an order of 10 dilution in the Toa Baja results).

There are a number of studies that report trace organic emissions from different types of open burning. In many of these studies, the lists of organic compounds are quite extensive. Rather than evaluate each compound, we will simply note that: (a) existing data on trace organic emissions from full-scale ACDs are almost nonexistent; (b) studies of open burning have consistently found considerable trace organic compound emissions; and (c) even well-controlled industrial combustion sources exhibit some level of trace organic emissions. Thus, one would expect to measure some trace organics in ACD emissions.

Similarly, the presence of PCDDs and PCDFs indicated in the Toa Baja study should be expected, as these compounds have also been found in studies of uncontrolled open burning. Gullett et al. showed that PCDDs and PCDFs from the simulated open burning of forest biomass were not solely from the volatilization of condensed material, but were also formed in the combustion process.[28] This pilot-scale work verified earlier measurements of actual forest and biomass fires that showed these events emit PCDDs and PCDFs.[29, 30]

CONTEMPORARY SCOPING MEASUREMENTS OF ACD OPERATION

In the aftermath of Hurricanes Katrina and Rita, there was interest in using ACDs to dispose of a portion of the enormous volume of debris left in the storms' wakes. Given the age of many homes in the affected areas, it was expected that a considerable number of homes would likely contain asbestos in one or more products and forms. The majority of asbestos was expected to be in chrysotile form, which can be thermally transformed into a non-hazardous forsterite form at temperatures above 800° C (1470° F).[31, 32] With the highly transient nature of ACD operation and the need to maintain temperatures above 800° C, the question was raised regarding the potential for ACDs to be used as a means to achieve the thermal conversion of chrysotile to forsterite under actual operating conditions. In late October 2005, researchers from EPA's Office of Research and Development (ORD) conducted a limited number of simple scoping tests on a full-scale ACD that was being used to demonstrate its ability to burn vegetative debris in the New Orleans area. The purpose of these scoping tests was to provide preliminary information on possible disposal options, evaluate ACD operating characteristics, and determine the most effective approaches to sampling for pollutant emissions during more in-depth testing.

Operation and Measurements

The ACD used was an Air Burners, LLC model S327, burning only dry vegetative debris. Loading of the unit occurred from the air plenum side. The unit was situated so that the air curtain was blowing in the same direction as the prevailing wind, which was reported at 10-15 mph with gusts of 25 mph.

Gas temperature and velocity and concentrations of several gases were measured approximately 6-15 in above the top of the ACD wall. A rough traverse of the area over the top of the ACD was made to identify any variations in gas concentrations, temperatures, or velocities. Portable continuous emission monitors were used to measure concentrations of CO_2, CO, oxygen (O_2), oxides of nitrogen (NO_x), and sulfur dioxide (SO_2). Temperature measurements were taken using thermocouples and with a Series OS523-2 Omegascope infrared thermometer. Five type K thermocouples were installed in the ACD prior to loading with debris to measure wall and combustion bed temperatures at approximately the midpoint of the unit's length, at 7, 32, and 71 in (18, 81, and 180 cm, respectively) from the ACD bottom on the blower side and 32 and 61 in (81 and 150 cm, respectively) on the side opposite the blower. A K-type thermocouple was also used at the tip of the sampling probe to measure gas temperatures. Thermocouple signals were recorded using a hand-held thermocouple readout and entered onto manual data sheets. The optical pyrometer was used to take temperatures across the surface of the burning debris bed, and the results were recorded manually.

Observations and Results

During "steady state" operation, the opacity of the plume was near zero, and the location of the plume had to be determined using an infrared video detector. When additional debris was loaded into the unit after it had reached steady state operation, the opacity increased to a readily visible level, which lasted for less than a minute following the introduction of the debris charge. The formation of a visible plume did not occur consistently after each charge of debris. Transient plumes were observed in similar operations of an ACD with extended back and side walls operating in a different location, but also burning dry vegetative debris.

The averages of five gas concentration measurements for CO, CO_2, NO, NO_2, and SO_2 are shown in Table 2. The measurements were taken over the span of 46 minutes in different locations, with each measurement lasting less than a minute. The gas concentrations generally showed relatively high variability, which is not surprising for the low number of measurements taken. Concentrations of CO and CO_2 were higher than those reported previously, but the low number of measurements and possible differences in measurement methods make it difficult to draw meaningful conclusions in comparison to previous work. Concentrations of NO, NO_2, and SO_2 showed similarly high variability, with the low number of measurements again being of concern. Unfortunately, mass feed rates for the fuel were not measured during these scoping tests, so it is not possible to estimate the emission factors for these pollutants.

There was considerable variation in temperatures across the unit, measured from the blower side to the loading side. The unit is typically loaded from the side opposite the blower. However, for the Phase 1 tests, the unit was loaded from the blower side to allow greater access for gas and temperature measurements. For this discussion, we will refer to the blower side and the loading side (the side opposite the blower) as this terminology more accurately reflects typical operating practice. The higher temperatures were noted along the blower side. Blower side wall temperatures ranged from 670° to 1030° C, with the highest temperatures nearest the combustion bed. Wall temperatures were at a minimum near the midpoint of the 8 ft (2.4 m) wall height, with temperatures increasing again to 930° C approximately 3 ft (0.9 m) from the top. On the blower side, the wall temperatures were 750° C at a point about 3 ft (0.9 m) above the combustion bed and 600° C at a point about 4 ft (1.2 m) from the top. Combustion bed temperatures measured with the optical pyrometer ranged from 1020° C near the blower side to 740° C near the loading side. The average temperature (average of all locations and all times) was 920° C.

Unlike the simple circular pattern suggested in Figure 2, the flow of exhaust gases is quite complex in the unit. Velocity measurements suggest that the vast majority of exhaust flow is occurring in a relatively narrow area along the length of the unit on the side opposite the blower (see Figure 5). Measurements of 15 fps in this narrow area were close to the estimated temperature-adjusted flow velocity based on the ACD fan output. This distribution pattern is not thought to significantly impact the level or composition of emissions from the ACD. However, this finding is important relative to designing approaches to sampling emissions from ACDs. Sampling procedures should take into account the significant variability in gas velocities across the top of the ACD to ensure that the gas sampling locations selected include the area(s) of highest emissions outflow.

Following the completion of these tests, plans were completed for more detailed testing of emissions from the combustion of C&D and vegetative debris from ACDs. However, as the recovery effort progressed, several factors led to the potential for using ACDs to be significantly reduced. The concerns over emissions raised by previous work (noted above) and by an external review of debris disposal using ACDs;[33] more available landfill space than expected in the immediate aftermath of the storm; and a significantly longer lead time for making decisions regarding the demolition of severely damaged buildings all resulted in a decision not to use ACDs as a disposal option at that time. Therefore, there was a reduced need to conduct the more detailed tests during the initial period of storm recovery. However, there is still considerable interest in conducting more detailed studies of air emissions from ACDs[34] and it is possible that such tests will be conducted in the future.

DISCUSSION AND CONCLUSIONS

When properly operated, both anecdotal evidence and comparison with measurements from simple open burning indicate that ACDs burn vegetative debris in such a way that emissions of PM are reduced, probably significantly, compared to open burning. Concentrations of PM as indicated by opacity measurements are lower for ACDs, which produce plumes with very low opacity for the majority of operating time, and generate visible plumes only during start up and immediately after loading. These transient "puffs" of emissions are likely to be accompanied by increased emissions of organic compounds as well as PM, based on experience with transient events in rotary kiln incinerators and with biomass combustion.[35-37]

The lower PM and CO emissions are consistent with the improved combustion conditions that are present with ACDs as compared to open burning – better air flow, containment of heat around the combustion zone, and more controlled introduction of debris. These improved conditions would suggest that emissions of organic compounds are also lower for ACDs than for open burning, but adequate data are not yet available to draw such a conclusion. The existing data do show a significant potential for emissions of toxic organic compounds. The indications of PCDD/PCDF emissions during the Toa Baja tests, for instance, suggest that ACD combustion of chlorine-containing material could lead to the formation and emission of chlorinated organics.

The questions about emissions from C&D debris remain open. Under normal conditions, C&D debris can be maintained separately from vegetative debris. However, these types of debris are intermingled during disasters and separating them during recovery would require time and resources that are more effectively used for other response needs. Therefore we are left with a need to understand how emissions may differ when burning C&D as opposed to vegetative debris, or (more likely in a practical situation) a mixture of the two. Differences in composition and heat content make a direct extrapolation from existing data from vegetative debris combustion unrealistic. Higher concentrations of relatively inert inorganic compounds, particularly metals, would be expected in C&D debris; whether those compounds are emitted into the ambient atmosphere or are retained in the bottom ash remains unknown. The likely presence of chlorine and other halogens in C&D debris may also have a significant impact on the types of compounds that are formed in the combustion process and possibly emitted into the air. Higher concentrations of sulfur are also likely in C&D debris than in vegetative material, which can also significantly impact the high-temperature chemistry within the ACD firebox. Our current understanding of the behavior of these compounds in combustion environments is largely shaped by studies of either open burning or enclosed and controlled combustion of municipal solid waste, neither of which can be directly applied to the current problem. In short, the combustion of C&D debris in ACDs is a new problem that has not been addressed by previous research.

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FIGURE CAPTIONS

Figure 1. Schematic of air curtain destructor operation.

Figure 2. Photographs of different air curtain destructor designs.

Figure 3. Emission factors vs. curtain air flows for carbonyls (top), carboxyls (center), and total hydrocarbons (bottom) reported by Burckle et al.[18]

Figure 4. Air curtain bed temperatures as measured by optical pyrometry.

Figure 5. Velocity profile across top of air curtain destructor. The velocity peaks near the side wall opposite the air curtain plenum.

Tables

Table 1. Summary of CO and PM concentrations reported in the literature.

Report	**Full scale**			**Pilot scale**		
	Lambert[12] [a]	**Fountainhead[15]**	**Toa Baja[17] [b]**	**Burckle et al.[18]**	**Linak et al.[19]**	**Lutes and Kariher[20]**
Fuel	Wood	Wood	Wood/Soil (wet)	Cord wood	Polyethylene plastic	Woody Debris
CO (ppm)	140	54 (~ 0.5 g/kg)	9.3	100-1000	23	37
PM opacity and concen-tration	5% opacity	6600 $\mu g/m^3$ 5.4% opacity	568 $\mu g/m^3$	20000 $\mu g/m^3$ [c]	4730 $\mu g/m^3$	40000 $\mu g/m^3$
PM emission factor	NA[d]	0.054 g/kg	NA	5.0 g/kg	NA	10 g/kg
CO_2	0.75%	0.2%	NA	0.1-1.75%	0.3-0.6%	0.05%

a. Pit-type unit
b. Ambient measurements taken 50 feet downwind of the ACD unit
c. Corrected to 0.2% CO_2. Reported values were 0.53 grains/dscm (1.2 g/m^3) at 12% CO_2.
d. Not available

Table 2. Concentrations of gases measured at top of air curtain destructor.

	Average	**Range**
O_2 (%)	18.0	16.2 – 19.5
CO (ppm)	237	319 - 183
CO_2 (%)	2.5	1.2 – 4.1
NO (ppm)	75.0	11 – 100
NO_2 (ppm)	4.0	0 – 10
SO_2 (ppm)	4.6	2 – 8

Figures

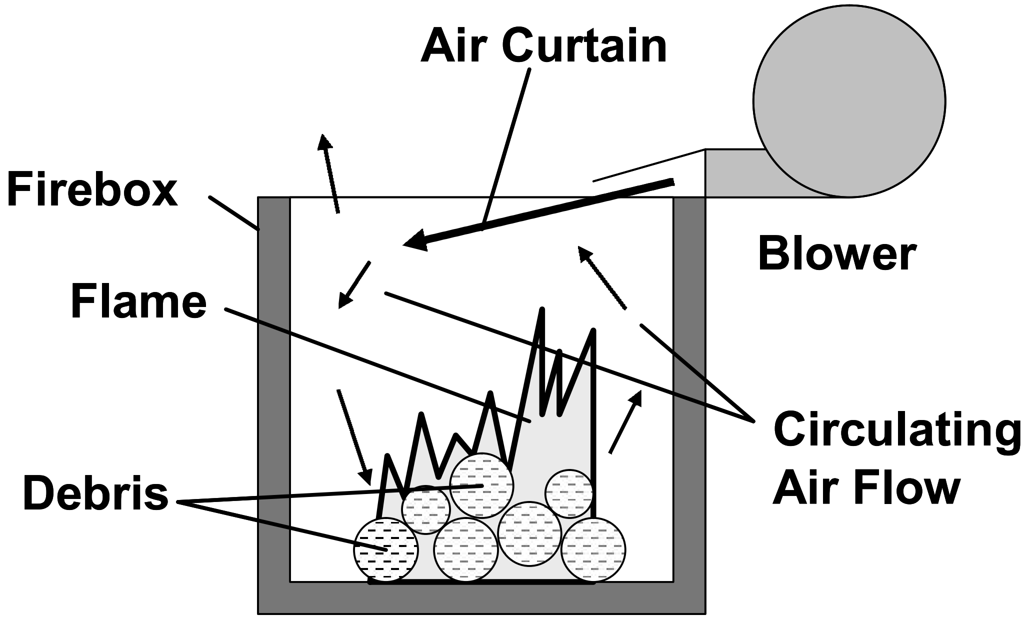

Figure 1. Schematic diagram of air curtain destructor operation.

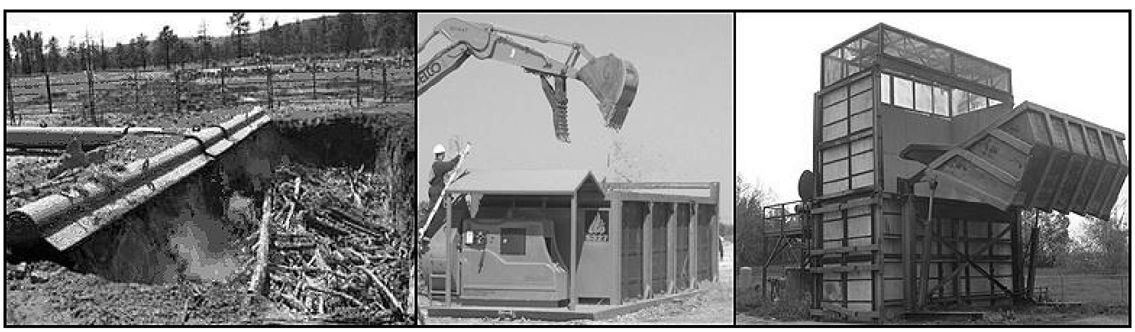

Figure 2. Photographs of different air curtain destructor designs.

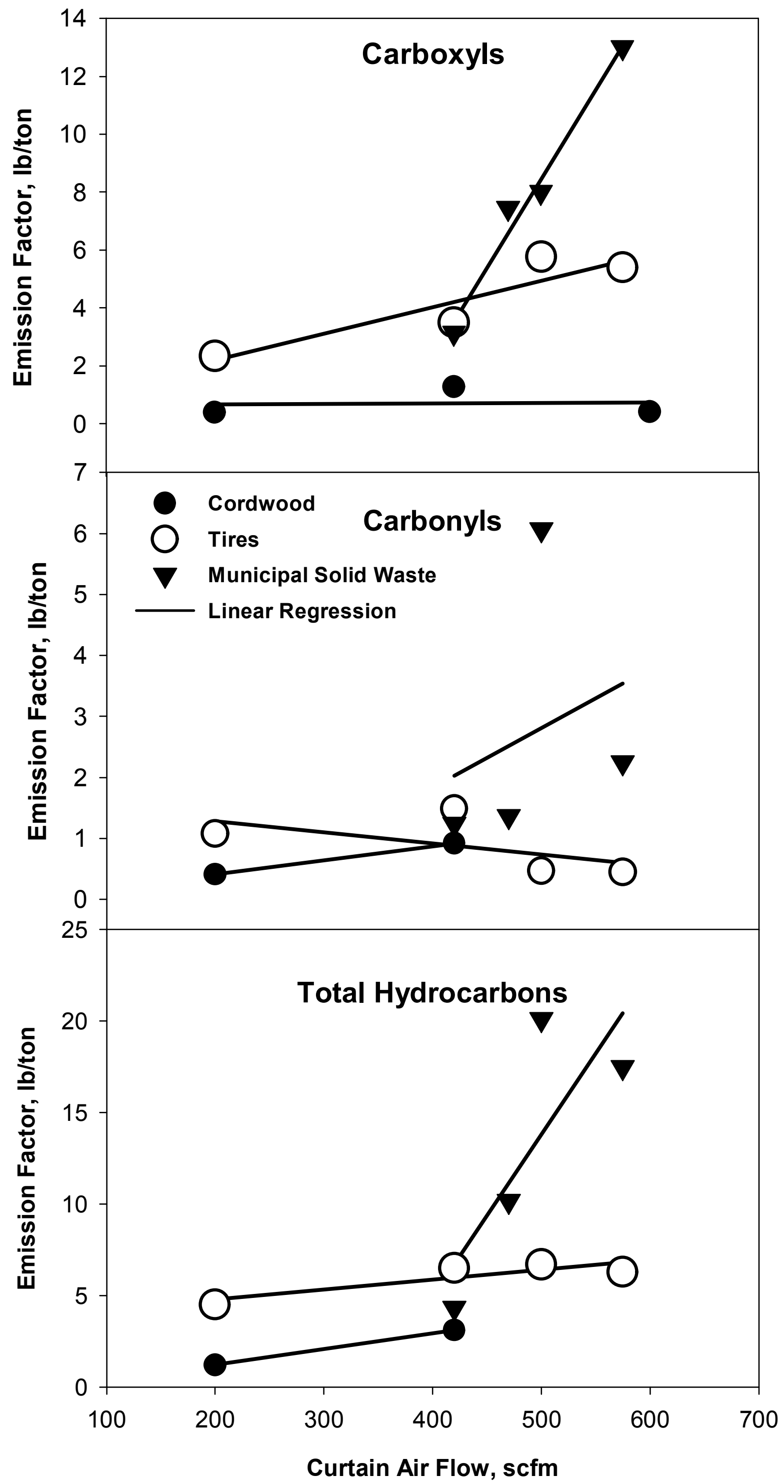

Figure 3. Emission factors vs. curtain air flows for carbonyls (top), carboxyls (center), and total hydrocarbons (bottom) reported by Burckle et al.[18]

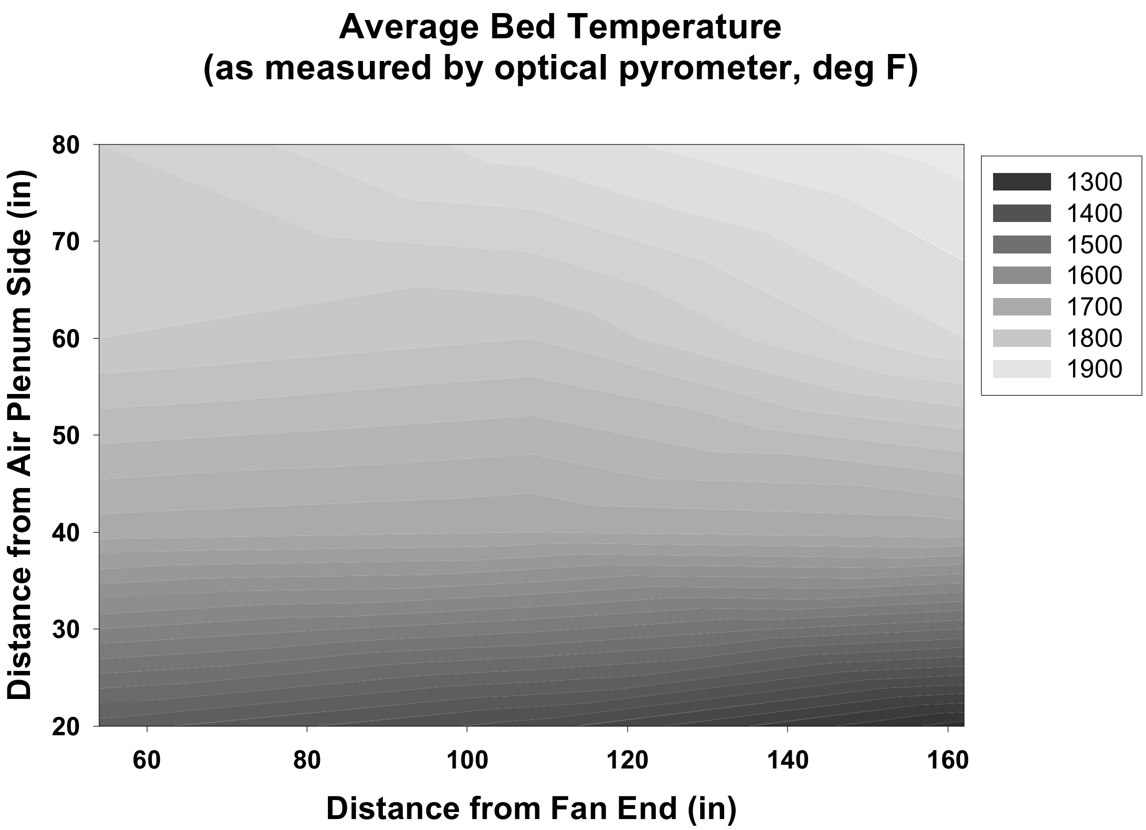

Figure 4. Air curtain bed temperatures as measured by optical pyrometry. Temperatures are lower near the curtain air plenum.

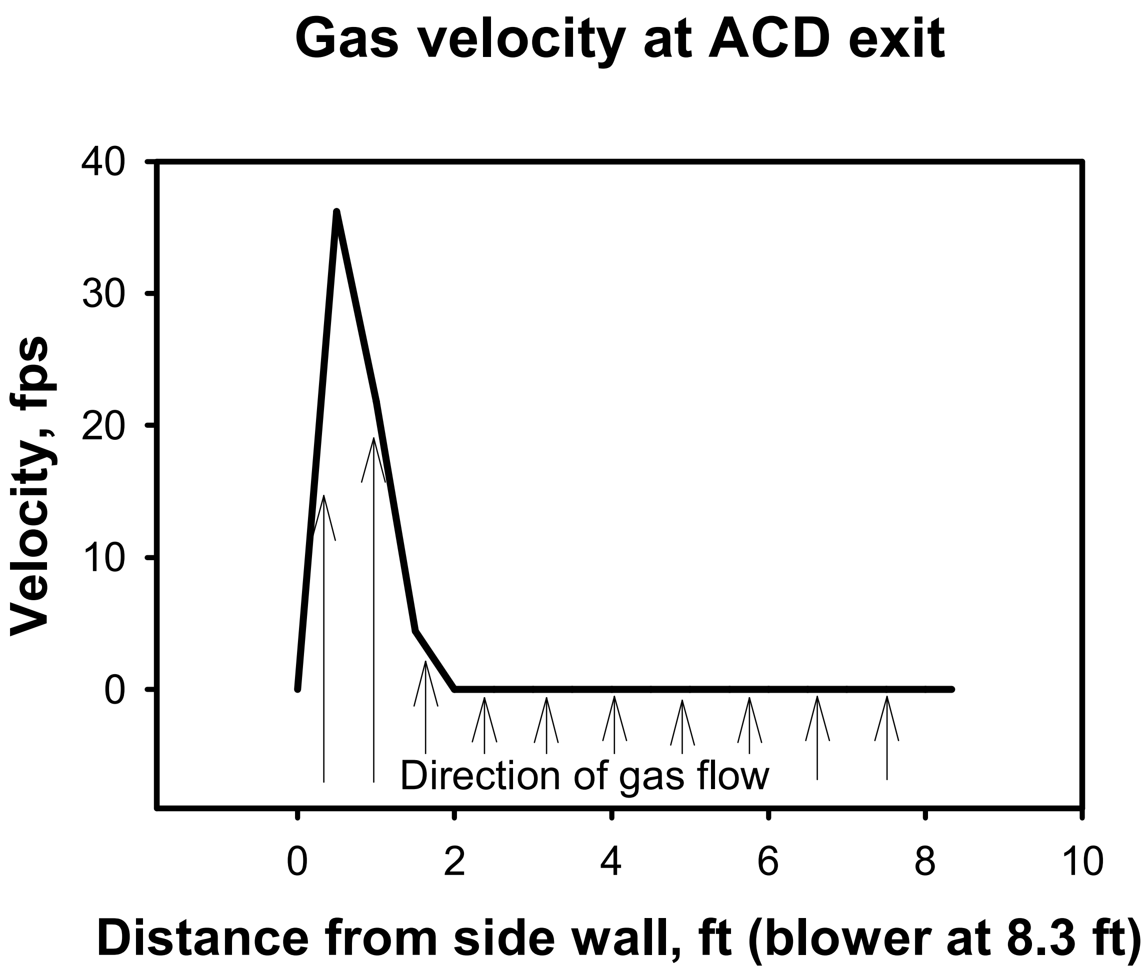

Figure 5. Velocity profile across top of air curtain destructor. The velocity peaks near the side wall opposite the air curtain plenum.