

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

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

39

40 **INTRODUCTION AND BACKGROUND**

41 On August 29, 2005, Hurricane Katrina came ashore along the Gulf Coast, with the eye
42 initially passing over Plaquemines Parish with 140 mph¹ wind speeds, then continuing
43 north and hitting the Louisiana/Mississippi border with wind speeds still over 125 mph.
44 Less than a month later, Hurricane Rita made landfall near the Texas/Louisiana border as
45 a major hurricane with 120 mph wind speeds. Both of these storms produced major
46 storm surges, which combined with the high winds to create enormous amounts of debris
47 from destroyed structures, downed trees, and other vegetative debris. Hurricane Wilma
48 struck the southern Gulf Coast of Florida a month later as a major hurricane, again
49 leaving behind a trail of damage and substantial debris. It is clear that 2005 was a
50 landmark year for hurricanes and tropical storms, but 2004 was also notable for the
51 damage caused by four hurricanes to hit Florida. Although the 2006 hurricane season
52 was relatively quiet for the U.S., it has been predicted that the number of hurricanes and
53 tropical storms will continue to be above the long-term average for at least the next
54 decade, and that the intensity of storms may also be increasing.²⁻⁴ These predictions, in
55 addition to the increases in building and population in coastal areas prone to hurricane
56 damage,⁵ emphasize the need to understand the implications of options for disposal of
57 hurricane debris.

58

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

68 245 million tons per year of municipal solid waste generated in the U.S. in 2005, which
69 translates to approximately 3.7 million tons per year in Louisiana, based on per capita
70 waste generation estimates.⁸ Clearly, the debris generated by Hurricanes Katrina and
71 Rita represented a major and very sudden increase in the level of solid waste that required
72 disposal.

73

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

87

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

92

93 **AIR CURTAIN DESTRUCTOR**

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

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

107

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

122

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

130

131 **PREVIOUS WORK**

132 Published data on emissions from ACD units are scarce. There have been a number of
133 studies of open burning over the past decade that provide some basis for comparison. For
134 ACDs, however, the data are less available and tend to be less detailed. A brief
135 description of ACD operation and emissions published by the U.S. Forest Service¹¹ is an
136 example of the available documents that describe ACD operations and describe
137 emissions, although only qualitatively.

138

139

CO and PM

140 There are a few studies that have been done over the past 40 years on ACD emissions.
141 Data from three full-scale tests have been found, and two papers that evaluated pilot-scale
142 ACD systems are also in the literature. The earliest of the full-scale studies was
143 published in 1972 by Lambert, who described emissions from a pit-type ACD unit used
144 to burn forest-clearing debris.¹² The pit in which the debris was burned was 41 feet long,
145 8 feet wide, and 15 feet deep (12.5 m x 2.4 m x 4.6 m). Lambert reported emissions of
146 carbon monoxide (CO) and carbon dioxide (CO₂), Ringlemann smoke number, and
147 temperatures. CO and CO₂ were measured using a continuous emission monitor (CEMs),
148 PM measurements were taken using a high volume (“hi-vol”) ambient sampler, and
149 opacity was measured using the Ringelmann visual method (although no longer in
150 official use in the U.S., the Ringelmann method has been used since the 1880s, as
151 described in Ref. 13; a brief but more technical discussion of the method can be found in
152 Ref. 14). Temperatures were measured with thermocouples in the debris bed and with an
153 optical pyrometer. The average CO measured was 140 ppm, with CO₂ at 0.75%. PM
154 levels were reported as “too low to measure,” although opacities were reported to be at ½
155 Ringlemann smoke number (5%) for 95% of the operating time. During unit startup, the
156 opacity was reported to be 40% (Ringlemann 2), and when diesel fuel was introduced to
157 ignite the bed, Ringlemann numbers as high as 8 were noted.¹²

158

159 Temperatures were consistently found to be at least 1600 °F (920 °C), and increased with
160 time. Peak temperatures of over 2200 °F (1250 °C) were measured after 11 hours of

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

171

172 A more recent report describing emissions from ACDs was prepared by Fountainhead
173 Engineering in 2000.¹⁵ This study reports emissions from an above-ground ACD unit
174 (Air Burners Model S-127), and provides data on CO, CO₂, and PM emissions and
175 opacity sampled at the top of the ACD unit. Over four test runs, the average CO
176 concentration was measured at 54 ppm at CO₂ levels of 0.2%, suggesting a greater level
177 of dilution than in the Lambert study. PM concentrations were measured at 6600 µg/m³,
178 and emission rates were reported at 2.14 lb/hr (0.97 kg/hr). At this concentration, opacity
179 levels measured using EPA Method 9¹⁶ were found to range between 4% and 7.5%, with
180 an average of 5.4%.

181

182 The final full scale measurements were reported by Trespalacios describing operation of
183 an ACD burning vegetative hurricane debris in Toa Baja, Puerto Rico.¹⁷ This study
184 measured pollutants on the perimeter of the ACD operation site rather than taking
185 samples from the outlet of the unit. The unit used in this operation was similar to that
186 used in the Fountainhead tests. Concentrations of CO and PM were measured at points
187 50 (15 m) and 100 feet (30 m) upwind and downwind from the ACD. More detailed data
188 on organic and metal emissions were also collected.

189

190 Average CO concentrations 50 ft (15 m) downwind from the ACD were 9.3 ppm, and
191 average PM concentrations at the same location were 570 µg/m³. The average values are

192 for three measurements taken over 10 s intervals shortly after loading “dirty debris,” a
193 wet mixture of soil and vegetative debris. It is very possible that the steady state
194 averages would be lower than those reported, but additional measurements were not
195 reported.

196

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

204

205 There are three pilot-scale studies reported in the literature that are also relevant. The
206 first of these was published in 1968, and is very consistent with the later full scale results
207 noted above. Burckle et al. burned cordwood, municipal solid waste (MSW), and tires in
208 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
209 m) in size. When burning a 318 lb (144 kg) charge of wood with the air curtain fan
210 operating at 420 scfm ($11.9 \text{ m}^3/\text{min}$), CO was measured at 100-1000 ppm over the
211 course of the test. The tests were conducted for a single fuel charge, and measurements
212 were initiated after combustion had stabilized and continued until the fuel charge burned
213 out. CO₂ concentrations while burning wood at these conditions ranged from 0.1% to
214 1.75%. These values are consistent with the range of concentrations from full scale units.

215

216 PM was measured at 0.53 grains/dscf ($1.2 \text{ g}/\text{m}^3$), corrected to 12% CO₂, and an emission
217 factor of 12.7 lb/ton of fuel was calculated for this test. This concentration value seems
218 quite high, but if the values are corrected to 0.2% CO₂ (the CO₂ concentrations reported
219 in the Fountainhead study), the concentrations are much closer to the results reported for
220 the full scale units. The correction to 12% CO₂ reflects a comparison with enclosed
221 combustion systems such as boilers or incinerators, and may not be as appropriate a
222 comparison as a lower CO₂ concentration likely to be measured in an open burning

223 situation. At 0.2% CO₂, PM concentrations would be approximately 20,000 μg/m³,
224 which is roughly three times the value reported in the Fountainhead report. Although this
225 is considerably higher than the 6600 μg/m³ value from the Fountainhead report, it is
226 reasonably close considering the difference in scale. Differences in fuel and scale make
227 it difficult to conclude whether the two results are in fact comparable or if the similarity
228 is simply coincidental. In either case, the Burckle paper provides valuable information
229 on the transient nature of the process.

230

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

242

243 The second pilot-scale study is less directly applicable, but does provide some additional
244 insight into the ACD combustion process. Linak et al. burned 1 lb (454 gm) charges of
245 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)
246 pilot-scale unit and made detailed measurements of the organic compounds emitted from
247 the process.¹⁹ They also measured CO and PM and compared the results with and
248 without the use of a simulated air curtain. The peak CO dropped slightly from 42 ppm
249 without the air curtain to 37 ppm with the air curtain on when burning new sheet plastic.
250 With the same fuel, average as-measured CO concentrations dropped from 29 ppm with
251 the air curtain off to 23 ppm when the air curtain was on. It is unclear whether this
252 reduction was due to improved performance or to dilution of the CO by the curtain air.

253

254 Interestingly, PM concentrations increased when using the simulated air curtain. For the
255 new sheet plastic, PM concentrations were measured at 4730 $\mu\text{g}/\text{m}^3$ when the air curtain
256 was on and at 3560 $\mu\text{g}/\text{m}^3$ when it was off. CO_2 levels during these tests were reported to
257 “vary minimally” in the range of 0.3-0.6%. Although the fuel used in these tests was
258 significantly different, the reported PM concentrations were very similar to those reported
259 in the full scale tests. It is unclear whether this is due to similarities in the combustion
260 process or coincidental.

261

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

282

283

Organics, Metals, and Other Emissions

284 The Lambert study did not measure emissions other than CO₂, CO, and PM.¹² The
285 Fountainhead study did take SO₂ and NO₂ measurements, but concentrations were
286 sporadic. SO₂ was found on only one of four runs (at 1 ppm) and NO₂ measured
287 inconsistently at 1-4 ppm.¹⁵
288
289 More extensive measurements of trace compounds were taken during the Toa Baja study.
290 Several metals and organics were measured upwind and downwind of the ACD in an
291 effort to quantify emissions from the combustion process. Six metals were detected in
292 the samples: aluminum (Al), cadmium (Cd), chromium (Cr), iron (Fe), lead (Pb), and
293 potassium (K). Al and Fe were detected more consistently, but were also detected at
294 higher levels upwind than downwind, on average. Of the remaining metals, only Pb was
295 detected in more than one of the 8 samples collected, and then in only two.¹⁷ It is
296 probable that the higher Al and Fe concentrations are the consequence of the use of “dirty
297 fuel” (wet vegetation combined with soil). The high upwind values further suggest these
298 metals are the result of soil-borne elements, and also make it questionable whether the
299 downwind samples were emitted from the ACD as opposed to being from fugitive dust.
300 Because emissions of metals are very strongly dependent upon the composition of the
301 fuel, the applicability of these results to other units is limited to a recognition that ACD
302 combustion conditions appear to be adequate to result in the formation of metal-
303 containing particles.
304
305 Of the volatile organic compounds detected, benzene, toluene, chloromethane, and
306 formaldehyde were detected in each downwind sample at concentrations higher than the
307 upwind sample. Besides these compounds, only p-xylene and propionaldehyde were
308 detected in more than two (of eight) samples. The downwind propionaldehyde
309 concentrations were each lower than the upwind concentration, suggesting that the source
310 was not the combustion process. p-xylene was detected at levels above the upwind
311 concentration in at least one sample during each test, indicating that the source was the
312 ACD.
313

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

320

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

331

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

342

343

Comparison of ACD and Open Burning Emissions

344 For PM, the general range of open biomass burning emissions tends to be on the order of
345 10 g/kg of fuel. Gerstle and Kemnitz reported total PM emissions of 6-12 g/kg for
346 landscape refuse and 7-9 g/kg for municipal refuse.²¹ For different types of biomass,
347 Andreae and Merlet reported PM_{2.5} emissions between 4 and 13 g/kg and total PM
348 emissions up to 18 g/kg,²² Lemieux et al. reported values for total PM emissions ranging
349 between 10 and 19 g/kg,²³ and Dennis et al. used emissions of 8-19 g/kg of total PM in
350 their estimates of total emissions from open burning.²⁴ These values are significantly
351 higher than the emissions per unit mass of fuel calculated in the Fountainhead test, which
352 were at 0.05 g/kg. It is very likely that even poorly operated systems will exhibit
353 significantly lower PM emission levels when they are able to increase the high-
354 temperature residence time of the pyrolyzed organics that form most of the fine PM. For
355 instance, an early dedicated vegetative debris burner was reported to have experienced
356 “excessive smoke” due to overcharging the unit during a test. Even at this overload
357 condition, the reported PM emissions were 1.1 g/kg, roughly an order of magnitude lower
358 than uncontrolled open burning.²⁵ There are likely to be some differences in emissions
359 due to the different biomass types and burning conditions, which ranged from prescribed
360 burning of savanna to burning of residential vegetative debris in small piles. Even so, the
361 PM emission factors for open burning are relatively consistent given the wide variety of
362 materials and conditions.

363

364 Interestingly, the pilot-scale results were similar to the reported uncontrolled open
365 burning emission rates. It is unclear why the pilot-scale ACD studies were unable to
366 similarly duplicate the reduced PM concentrations indicated by the full-scale ACD tests,
367 but the complexities involved in achieving simultaneous similarity in combustion, heat
368 transfer, and air flow may be a significant barrier to effective pilot-scale evaluations of
369 ACD emissions, at least without using a computational fluid dynamics (CFD) approach
370 to the design of the experimental apparatus. The hypothesis noted above that a higher
371 residence time may play a significant role in reducing emissions of organics and
372 unburned fuel could well be one of the reasons why pilot-scale units have not been able
373 to effectively simulate the emission rates measured during tests of full-scale units.

374 Scaling an ACD system to simulate physical conditions may well result in combustion

375 product residence times that are below characteristic reaction times needed to achieve
376 higher burnout levels.

377

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

393

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

402

403 Similarly, the presence of PCDDs and PCDFs indicated in the Toa Baja study should be
404 expected, as these compounds have also been found in studies of uncontrolled open
405 burning. Gullett et al. showed that PCDDs and PCDFs from the simulated open burning

406 of forest biomass were not solely from the volatilization of condensed material, but were
407 also formed in the combustion process.²⁸ This pilot-scale work verified earlier
408 measurements of actual forest and biomass fires that showed these events emit PCDDs
409 and PCDFs.^{29, 30}

410

411 **CONTEMPORARY SCOPING MEASUREMENTS OF ACD OPERATION**

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

428

429 **Operation and Measurements**

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

434

435 Gas temperature and velocity and concentrations of several gases were measured
436 approximately 6-15 in above the top of the ACD wall. A rough traverse of the area over

437 the top of the ACD was made to identify any variations in gas concentrations,
438 temperatures, or velocities. Portable continuous emission monitors were used to measure
439 concentrations of CO₂, CO, oxygen (O₂), oxides of nitrogen (NO_x), and sulfur dioxide
440 (SO₂). Temperature measurements were taken using thermocouples and with a Series
441 OS523-2 Omegascope infrared thermometer. Five type K thermocouples were installed
442 in the ACD prior to loading with debris to measure wall and combustion bed
443 temperatures at approximately the midpoint of the unit's length, at 7, 32, and 71 in (18,
444 81, and 180 cm, respectively) from the ACD bottom on the blower side and 32 and 61 in
445 (81 and 150 cm, respectively) on the side opposite the blower. A K-type thermocouple
446 was also used at the tip of the sampling probe to measure gas temperatures.
447 Thermocouple signals were recorded using a hand-held thermocouple readout and
448 entered onto manual data sheets. The optical pyrometer was used to take temperatures
449 across the surface of the burning debris bed, and the results were recorded manually.

450

451

Observations and Results

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

460

461 The averages of five gas concentration measurements for CO, CO₂, NO, NO₂, and SO₂
462 are shown in Table 2. The measurements were taken over the span of 46 minutes in
463 different locations, with each measurement lasting less than a minute. The gas
464 concentrations generally showed relatively high variability, which is not surprising for
465 the low number of measurements taken. Concentrations of CO and CO₂ were higher than
466 those reported previously, but the low number of measurements and possible differences
467 in measurement methods make it difficult to draw meaningful conclusions in comparison

468 to previous work. Concentrations of NO, NO₂, and SO₂ showed similarly high
469 variability, with the low number of measurements again being of concern. Unfortunately,
470 mass feed rates for the fuel were not measured during these scoping tests, so it is not
471 possible to estimate the emission factors for these pollutants.

472

473 There was considerable variation in temperatures across the unit, measured from the
474 blower side to the loading side. The unit is typically loaded from the side opposite the
475 blower. However, for the Phase 1 tests, the unit was loaded from the blower side to allow
476 greater access for gas and temperature measurements. For this discussion, we will refer
477 to the blower side and the loading side (the side opposite the blower) as this terminology
478 more accurately reflects typical operating practice. The higher temperatures were noted
479 along the blower side. Blower side wall temperatures ranged from 670° to 1030° C, with
480 the highest temperatures nearest the combustion bed. Wall temperatures were at a
481 minimum near the midpoint of the 8 ft (2.4 m) wall height, with temperatures increasing
482 again to 930° C approximately 3 ft (0.9 m) from the top. On the blower side, the wall
483 temperatures were 750° C at a point about 3 ft (0.9 m) above the combustion bed and
484 600° C at a point about 4 ft (1.2 m) from the top. Combustion bed temperatures
485 measured with the optical pyrometer ranged from 1020° C near the blower side to 740° C
486 near the loading side. The average temperature (average of all locations and all times)
487 was 920° C.

488

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

499

500 Following the completion of these tests, plans were completed for more detailed testing
501 of emissions from the combustion of C&D and vegetative debris from ACDs. However,
502 as the recovery effort progressed, several factors led to the potential for using ACDs to be
503 significantly reduced. The concerns over emissions raised by previous work (noted
504 above) and by an external review of debris disposal using ACDs;³³ more available
505 landfill space than expected in the immediate aftermath of the storm; and a significantly
506 longer lead time for making decisions regarding the demolition of severely damaged
507 buildings all resulted in a decision not to use ACDs as a disposal option at that time.
508 Therefore, there was a reduced need to conduct the more detailed tests during the initial
509 period of storm recovery. However, there is still considerable interest in conducting
510 more detailed studies of air emissions from ACDs³⁴ and it is possible that such tests will
511 be conducted in the future.

512

513 **DISCUSSION AND CONCLUSIONS**

514 When properly operated, both anecdotal evidence and comparison with measurements
515 from simple open burning indicate that ACDs burn vegetative debris in such a way that
516 emissions of PM are reduced, probably significantly, compared to open burning.

517 Concentrations of PM as indicated by opacity measurements are lower for ACDs, which
518 produce plumes with very low opacity for the majority of operating time, and generate
519 visible plumes only during start up and immediately after loading. These transient
520 “puffs” of emissions are likely to be accompanied by increased emissions of organic
521 compounds as well as PM, based on experience with transient events in rotary kiln
522 incinerators and with biomass combustion.³⁵⁻³⁷

523

524 The lower PM and CO emissions are consistent with the improved combustion conditions
525 that are present with ACDs as compared to open burning – better air flow, containment of
526 heat around the combustion zone, and more controlled introduction of debris. These
527 improved conditions would suggest that emissions of organic compounds are also lower
528 for ACDs than for open burning, but adequate data are not yet available to draw such a
529 conclusion. The existing data do show a significant potential for emissions of toxic

530 organic compounds. The indications of PCDD/PCDF emissions during the Toa Baja
531 tests, for instance, suggest that ACD combustion of chlorine-containing material could
532 lead to the formation and emission of chlorinated organics.

533

534 The questions about emissions from C&D debris remain open. Under normal conditions,
535 C&D debris can be maintained separately from vegetative debris. However, these types
536 of debris are intermingled during disasters and separating them during recovery would
537 require time and resources that are more effectively used for other response needs.

538 Therefore we are left with a need to understand how emissions may differ when burning
539 C&D as opposed to vegetative debris, or (more likely in a practical situation) a mixture of
540 the two. Differences in composition and heat content make a direct extrapolation from
541 existing data from vegetative debris combustion unrealistic. Higher concentrations of
542 relatively inert inorganic compounds, particularly metals, would be expected in C&D
543 debris; whether those compounds are emitted into the ambient atmosphere or are retained
544 in the bottom ash remains unknown. The likely presence of chlorine and other halogens
545 in C&D debris may also have a significant impact on the types of compounds that are
546 formed in the combustion process and possibly emitted into the air. Higher
547 concentrations of sulfur are also likely in C&D debris than in vegetative material, which
548 can also significantly impact the high-temperature chemistry within the ACD firebox.

549 Our current understanding of the behavior of these compounds in combustion
550 environments is largely shaped by studies of either open burning or enclosed and
551 controlled combustion of municipal solid waste, neither of which can be directly applied
552 to the current problem. In short, the combustion of C&D debris in ACDs is a new
553 problem that has not been addressed by previous research.

554

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658 FIGURE CAPTIONS

659 Figure 1. Schematic of air curtain destructor operation.

660 Figure 2. Photographs of different air curtain destructor designs.

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

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

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

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669 Tables
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 671

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 concentration	5% opacity	6600 µg/m ³ 5.4% opacity	568 µg/m ³	20000 µg/m ^{3(c)}	4730 µg/m ³	40000 µg/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%

672 a. Pit-type unit
 673 b. Ambient measurements taken 50 feet downwind of the ACD unit
 674 c. Corrected to 0.2% CO₂. Reported values were 0.53 grains/dscm (1.2 g/m³) at 12% CO₂.
 675 d. Not available

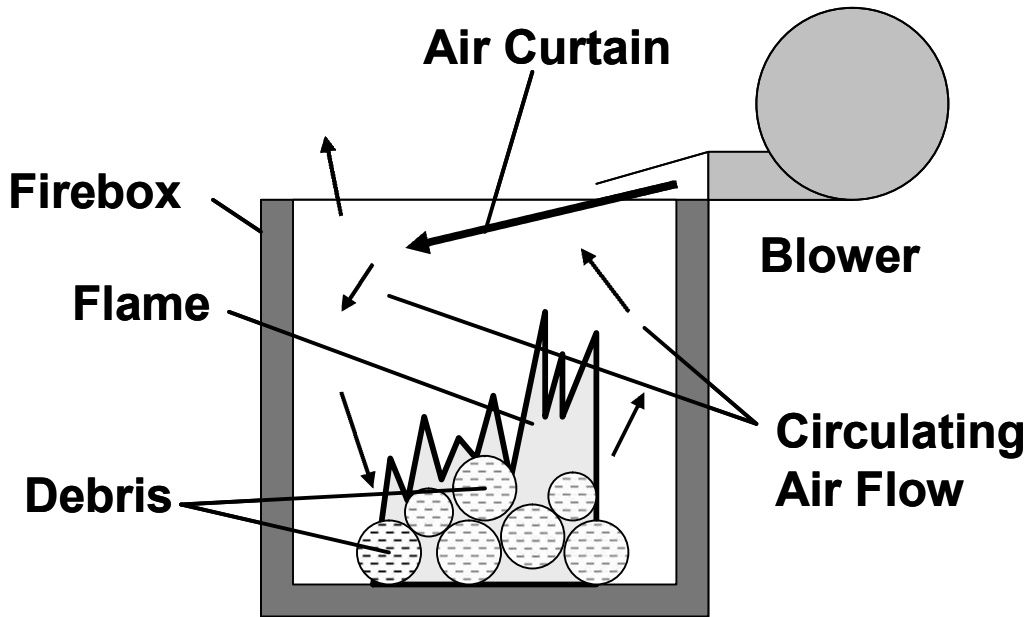
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Table 2. Concentrations of gases measured at top of air curtain destructor.

	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

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681 Figures
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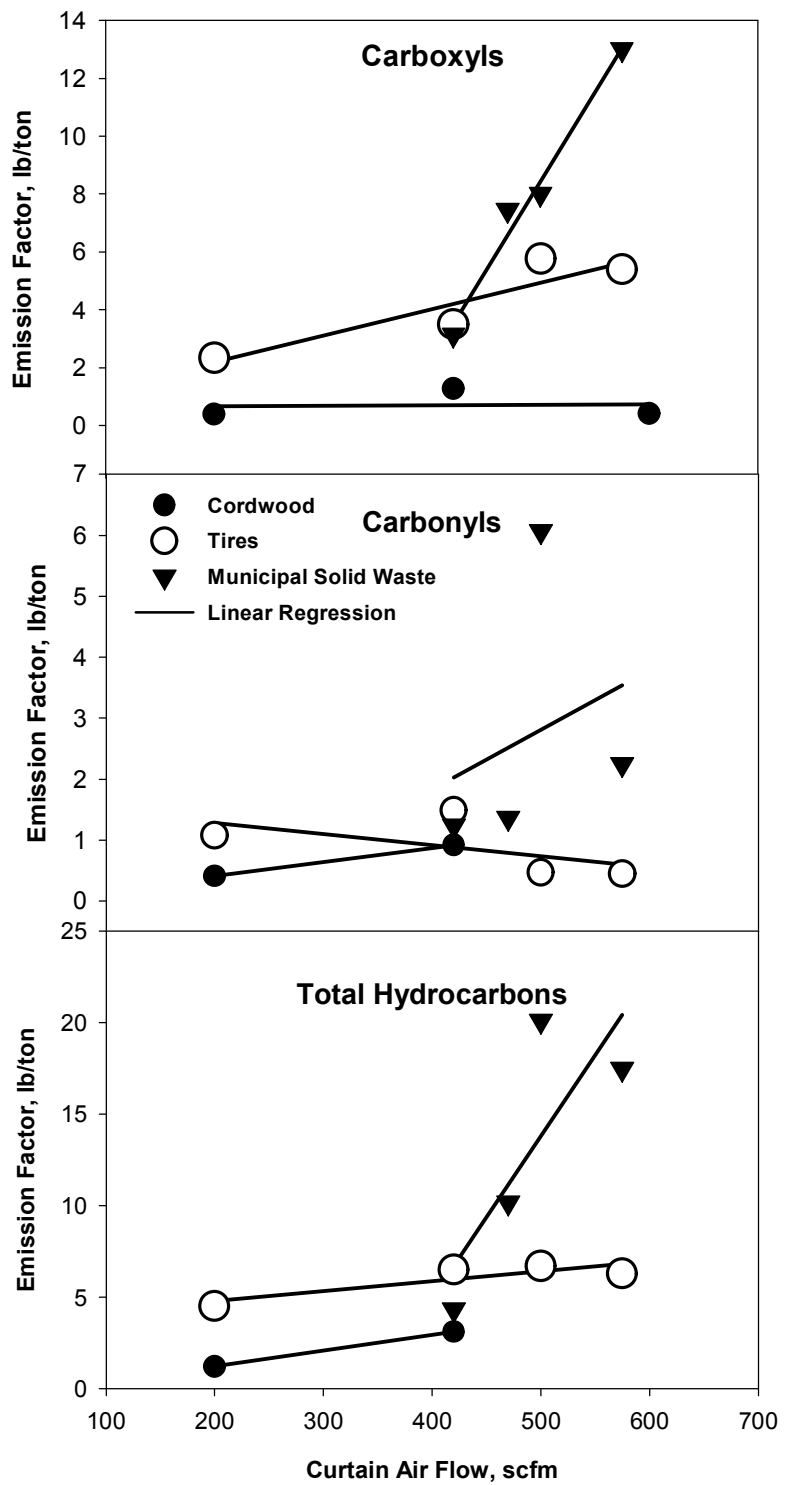
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Figure 1. Schematic diagram of air curtain destructor operation.



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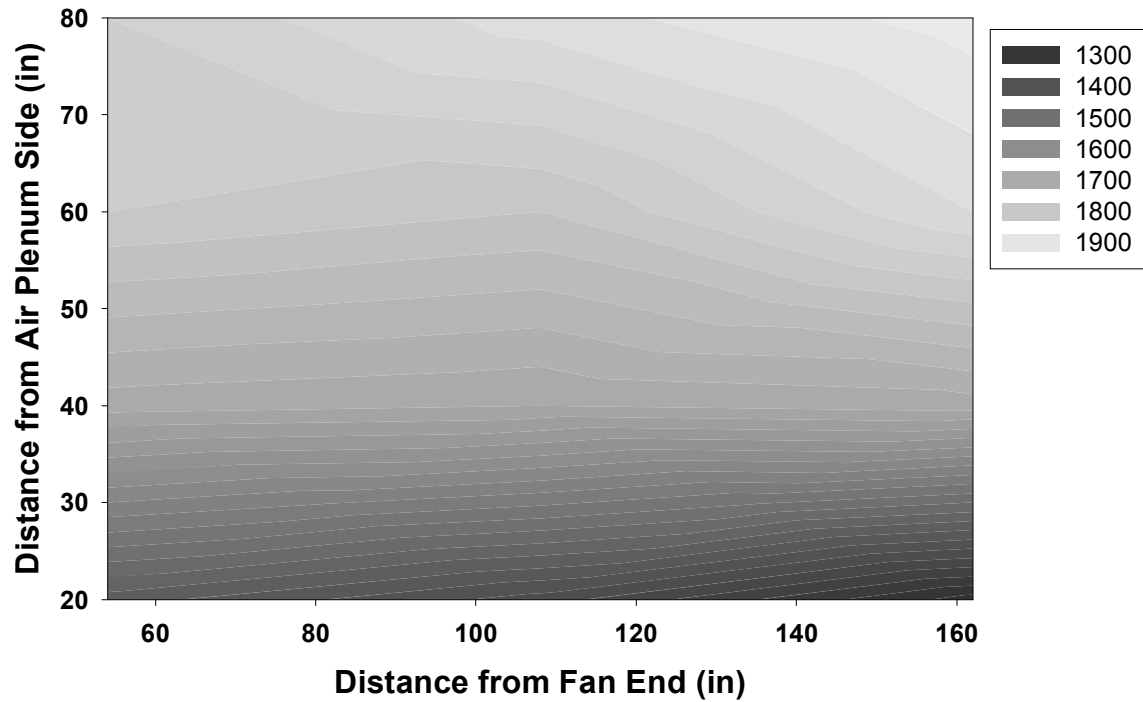
Figure 2. Photographs of different air curtain destructor designs.



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

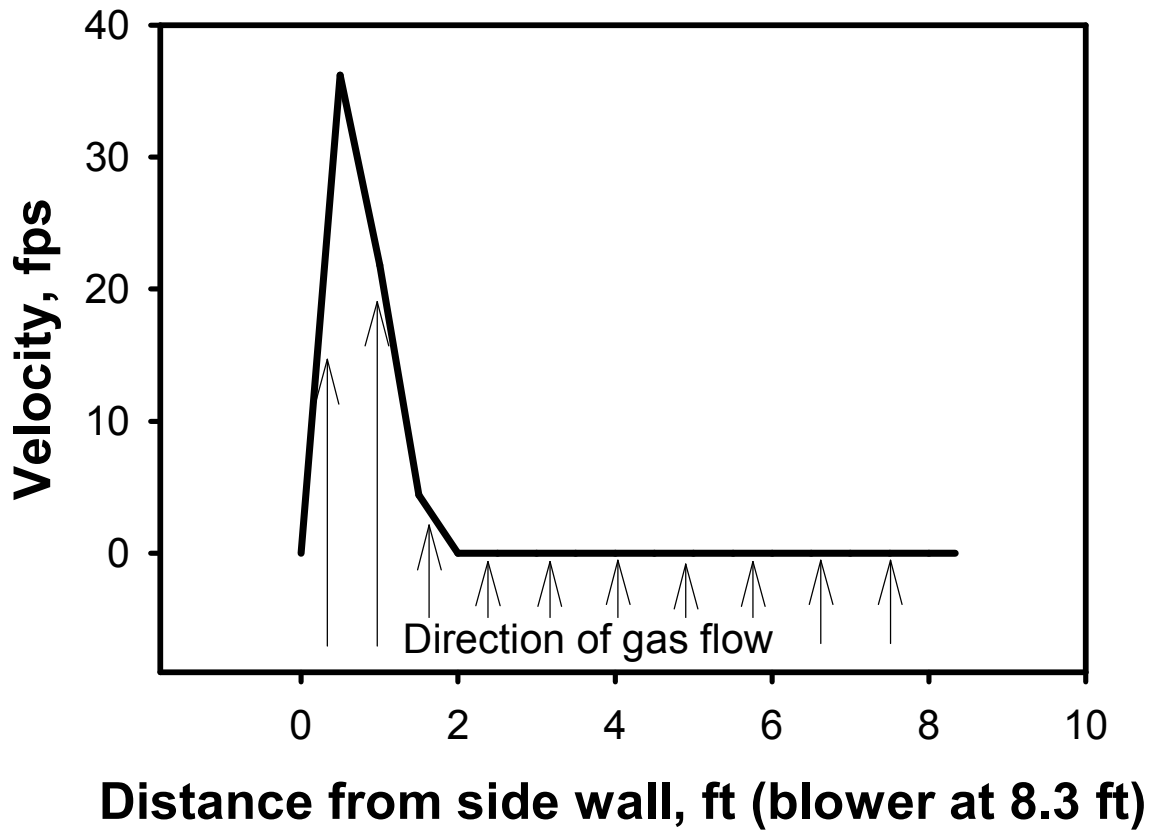
**Average Bed Temperature
(as measured by optical pyrometer, deg F)**



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

Gas velocity at ACD exit



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