# APCA NOTE-BOOK

# Estimates of Air Pollution from Backyard Burning

**S. A. Edgerton, M. A. K. Khalil and R. A. Rasmussen**
Department of Environmental Science
Oregon Graduate Center
Beaverton, Oregon

In many areas of the U.S. and particularly in Oregon, residential woodburning has been identified as a major source of fine particle pollution in the wintertime. In this paper, evidence is presented that open burning of twigs, branches, leaves, orchard prunings, and other domestic yard debris, which is called backyard burning, may on the average contribute as much to the daily fine particle pollution during the spring as woodburning does during the winter. Whether backyard burning causes significant air pollution has been a controversial issue in the Portland, Oregon, area, especially over the past few years. Efforts were made in 1980 to legislate a total ban on backyard burning. In 1981 when the ban was scheduled to take effect, a new state statute was legislated which required that before a ban could be placed on residential backyard burning, the Environmental Quality Commission would have to establish the necessity of the ban in meeting air quality standards and show the availability of alternate methods for the disposal of debris at acceptable costs.

The feasibility of using a gaseous tracer model will be demonstrated here to estimate the contribution of backyard burning to air pollution in the Portland area. In the past receptor modeling relying on chemical mass balances has been inadequate for accurately determining the contribution of backyard burning or woodburning to urban air pollution primarily because of the great variability in the patterns of elemental tracers emitted from these sources. Two of the important species characteristic of woodburning, organic and elemental carbon, have many sources other than vegetative burning, and show variabilities in emissions from wood combustion of 16% and 81% respectively.[1] The other important elemental species in the emission profile, S, Cl, and K, all show variabilities of over 100% in source emissions from woodburning.[1] Recently a gaseous tracer has been identified which can be used to make almost real time estimates of fine particle pollution from sources of biomass burning such as backyard burning, residential woodburning,[2] and field burning. This tracer is methylchloride ($CH_3Cl$), and it can be used to determine the pollution from biomass burning more accurately than the elemental tracers used in the past. The uniqueness of this tracer for pollution from woodburning arises from three important characteristics. First, $CH_3Cl$ can be measured over short times, providing estimates of particulate pollution from combustion sources over time scales of several minutes compared to 8–24 hours for elemental tracer methods. Second, there are no significant local sources of $CH_3Cl$ other than biomass combustion, thus making $CH_3Cl$ a unique tracer of woodburning. Finally, our recent studies show that the source emissions of this tracer are far less variable than elemental tracers.

The major source of $CH_3Cl$ in the background atmosphere is oceanic, and its global concentration and seasonal cycle have been well characterized.[3,4] Our study using $CH_3Cl$ as a tracer of residental woodburning in the suburban Portland area showed source contributions to fine particles in agreement with estimates from other receptor modeling studies when its short-term contributions were extrapolated to 24-h averages.[2,5,6] Because of the relatively high background concentration of $CH_3Cl$, the propagation of errors in the calculation of the contribution of this source to the ambient aerosol shows that the uncertainties may be as high as 40–50%.[2] However, these uncertainties in estimating the woodburning contribution are no greater than those in the chemical mass balance. Hence, at present it is expected that the gaseous tracer model using $CH_3Cl$ as a tracer for woodburning is at least as accurate as the chemical mass balance methods, and has the advantage of resolving the contribution of woodburning to ambient pollution over short time periods.

The gaseous tracer model developed previously[2] and $CH_3Cl$ data collected over the past year are utilized to estimate the fine particle pollution from the three categories of biomass burning mentioned above. It is shown how backyard burning might significantly contribute to fine particle pollution in the Portland, Oregon, area during the springtime.

## Model

The simplified gaseous tracer model is similar to a one-component chemical element balance model[7,8] using gaseous instead of elemental tracers. Basically the concentration of the aerosol $C_a$ is related to the excess concentration of the gas over the background ($\Delta C_g$) by the emission ratio $\alpha$:

$$C_a = \alpha \Delta C_g \quad (1)$$

$$\alpha = Q_a/Q_g \quad (2)$$

$$\Delta C_g = C_g - C_{g0} \quad (3)$$

$Q_a$ is the emission of fine aerosol from the source of interest and $Q_g$ the emission of a tracer gas, in this case $CH_3Cl$. The estimated values of $\alpha$ are given in Table I for the various categories of biomass burning. $C_g$ is the measured concentration of $CH_3Cl$ on a given day at the suburban site, and $C_{g0}$ is the concentration at the background site.

84-79

**Table I.** Fine particle contribution from biomass burning at suburban site estimated with gaseous tracer model over all sample days and emission ratio $\alpha$ for three categories of biomass burning.

| Biomass burn category | No. of sample days | Fine particle concentration[a] $\mu g/m^3$ | Emission ratio $\alpha$[b] (Reference) |
|---|---|---|---|
| Woodburning | 39 | 52 ± 10 | 0.60 (2,3,10,11) |
| Backyard burning | 7 | 61 ± 6 | 0.66 (3,12) |
| Field burning | 11 | 5 ± 2 | 0.14 (3,13) |

[a] Values are average concentrations during peak pollution periods in the evening.

[b] $\alpha$ = (mass of fine particles emitted/s)/(mass of $CH_3Cl$ emitted/s). Actual values of $\alpha$ are not yet available. Hence $\alpha$ is calculated as: $\alpha = (Q_a/Q_{CO_2})(Q_{CO_2}/Q_{CH_3Cl})$. The emission factors $Q$ are found in the listed references.

## Experimental Data and Results

Measurements of $CH_3Cl$ were taken at a rural background site (Oregon Graduate Center) with some low density housing nearby in Beaverton, Oregon, which is about 16 km west of Portland, on 142 days throughout the year from September 1982 to September 1983. These samples were collected in the morning when any accumulated pollution from the previous evening is expected to have been dispersed. Over the same time, 65 days were randomly selected for evening $CH_3Cl$ measurements at a much higher density suburban site in nearby Hillsboro, Oregon. The two sites are only about 8 km apart. The monthly means of $CH_3Cl$ at the background site and the monthly means at the suburban site are shown in Figure 1. The seasonal cycle found in the morning at the background site closely resembles the cycle of $CH_3Cl$ measured in clean air at Cape Meares, Oregon, over the past five or six years, lending support to its use as a background for the suburban site. At the background site all of the standard errors in the monthly means fall below 18 pptv or less than 3% of the measured values, which range from 560 to 645 pptv. This represents the uncertainty with which the background concentration may be determined for use in the model. At the suburban location the standard error was never greater than 4% of the monthly mean. At that site three or more samples were collected within a 10-minute period to determine the precision of the grab sample technique over the short sampling period, assuring that the air is well mixed. The standard deviation found on each sample day was always less than 3% of the measured average value. The equipment for sampling and analysis of $CH_3Cl$ by GC-ECD has been developed and used for clean air measurements over the past decade.[3,9]

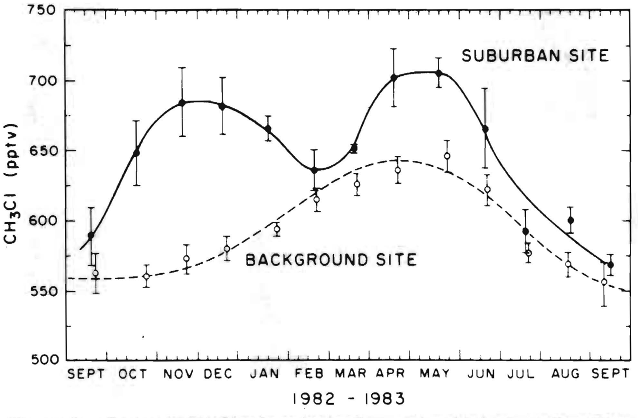

**Figure 1.** Seasonal $CH_3Cl$ values for background and suburban sites. Each data point represents a monthly average concentration for the background and for the suburban site. Error bars represent standard error of the mean. The curves are hand drawn to provide a qualitative suggestion of the cycles. Note the double humped suburban cycle, one hump for woodburning and one for backyard burning.

Monthly averages of particulate contribution from biomass burning as estimated using the model are shown in Table II and Figure 2. The model (Eq. 1–3) is applied individually to each sample day for each category of biomass burning from Table I. The backyard burning season lasts from March 1 to June 15 and from October 1 to December 15. Samples collected during the spring are labeled as mostly woodburning or mostly backyard burning according to the degree day data from that sample day. Each degree day represents the departure of the mean temperature for the day one degree below 65°F. Days with fewer than five degree days are labeled backyard burning days since on these days the temperature often reaches into the mid-70s and not much wood heating is expected. Temperature records from actual spring sample days show that on these days the temperature did not go below 70°F until 6:00 or 7:00 P.M. Days with greater than 10 degree days are labeled woodburning days. None of the sample days fell within the range of 5–10 degree days. Because the November-December sample days were cool, the wood and backyard burning sources cannot be separated at this time. Therefore, sample days where backyard burning was prohibited by the Department of Environmental Quality (DEQ) are labeled "woodburning days" and other days are labeled "unresolved backyard and woodburning." The field burning season is from July to the end of September. The majority of field burning occurs in a valley more than 100 km south of Portland, and its effect in the Portland area is expected to be minimal. Days of excess $CH_3Cl$ occurring in July, August, and September may therefore be attributed mostly to field burning.

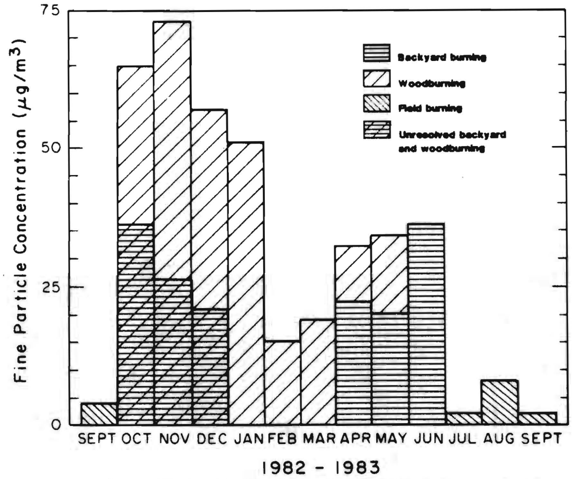

**Figure 2.** Estimates of fine particle concentrations during evening hours of peak pollution on days when there is backyard burning, woodburning, and/or field burning. The total monthly contribution to particulate pollution from these sources is expected to be much lower than the peak values reported in the figure. Measurements of $CH_3Cl$ were taken at a suburban site in Hillsboro, Oregon, and the fine particle concentrations from these sources were estimated by using a gaseous tracer model.

The suburban samples were collected during the evening hours between 8:00 and 10:00 P.M. in order to estimate the winter wood heating effects. These hours represent the time of day when the mixing height generally reaches its lowest daily value and atmospheric dispersion of pollutants is often at a minimum. In the wintertime low level inversions frequently form during the early evening hours, and the concentrations of pollutants from residential woodburning is usually at its highest value. During the spring and fall backyard burning seasons DEQ requests that all fires be out two

**Table II.** Monthly estimates of fine particle concentrations from biomass burning using gaseous tracer model. $CH_3Cl$ is used as the tracer.

| Month (No. of sample days) | Source[a] (No. of days attributed to source) | Average contribution from each source | Average particulate concentration from biomass burning µg/m$^3$ | Range µg/m$^3$ |
|---|---|---|---|---|
| September (7) | FB (5) | 6 | 4 | 0–11 |
| | NC (2) | | | |
| October (7) | BYB (2) | 77 | 65 | 65–92 |
| | WB (5) | 60 | | 11–131 |
| November (11) | WB (3) | 108 | 73 | 66–166 |
| | BYB & WB (8) | 60 | | 13–122 |
| December (4) | WB (2) | 72 | 57 | 34–100 |
| | BYB & WB (2) | 42 | | |
| January (8) | WB (7) | | 51 | 27–104 |
| | NC (1) | | | |
| February (3) | WB (2) | | 15 | 10–35 |
| | NC (1) | | | |
| March (5) | WB (5) | | 19 | 6–39 |
| April (4) | BYB (1) | 67 | 32 | 67 |
| | WB (3) | 32 | | 9–47 |
| May (4) | BYB (2) | 47 | 34 | 46–48 |
| | WB (2) | 32 | | 9–47 |
| June (3) | BYB (2) | 54 | 36 | 42–65 |
| | NC (1) | | | |
| July (5) | FB (3) | 3 | 2 | 1–4 |
| | NC (2) | | | |
| August (2) | FB (2) | | 8 | 8–9 |
| September (2) | FB (1) | 4 | 2 | 0–4 |
| | NC (1) | | | |

[a] FB = field burning; BYB = backyard burning; WB = woodburning; NC = no contribution.

hours before sunset. However, the decreased dispersion during the early evening often causes the day's accumulated pollutants from the burning of yard debris to remain concentrated in the atmosphere for several hours after the fires have been extinguished. Box model estimates of the dispersion of pollutants on days where the vegetative burning source is attributed to backyard burning show that from 30% to 70% of the pollutants would remain in the atmosphere at the 9:00 P.M. sample time.

## Discussion

The peak contributions of the various biomass burning sources to the fine particle concentrations as estimated by the gaseous tracer model are summarized in a histogram in Figure 2. While the source contributions overlap in many of the months, the month of January clearly shows the effects of woodburning, and the month of June shows the effects of backyard burning alone. All sample days in June where there was a contribution to fine particles occur before June 15, the end of the backyard burning season. The 39 days isolated as woodburning days show a contribution of 52 ± 10 µg/m$^3$ to the fine aerosol concentration on winter evenings, and the seven days isolated as entirely backyard burning days show a contribution of 61 ± 6 µg/m$^3$ to the fine aerosol on spring and fall evenings. The mean values reported above are derived from Eq. (1–3) by averaging the calculated values for $C_a$ from each day; the standard error values are 90% confidence limits of the mean. This illustrates that on the average the practice of backyard burning in the Pacific Northwest may contribute as much to the maximum fine particle pollution in the springtime or fall as woodburning does in the winter. However, the woodburning season is much longer and the maximum pollutant concentrations from woodburning are often two or three times those from backyard burning. Preliminary estimates suggest that backyard burning could contribute no more than one-third as much particulate pollution as woodburning on an annual basis.

The maximum estimated contribution from woodburning during the winter of 1982–1983 was 166 µg/m$^3$, and this occurred on November 14. But 18% of the evenings when there was woodburning showed a contribution of greater than 100 µg/m$^3$ during the evening hours, while none of the estimated concentrations from the evenings with backyard burning showed a concentration that high. The maximum particle concentration on an evening when there was backyard burning was 92 µg/m$^3$, and this occurred on October 24.

The gaseous tracer model applied to the seasonal $CH_3Cl$ measurements suggests that backyard burning may be a major source of fine particle pollution in the springtime and whenever the practice is allowed.

## Acknowledgments

We thank Dr. J. A. Cooper of NEA Laboratories and Mr. John Kowalczyk of the Oregon Department of Environmental Quality for helpful suggestions, and Ms. Patricia Quinn for some of the laboratory analyses. This work was supported in part by a grant from the U.S. EPA (No. R810091-01-0) and also by Biospherics Research Corporation and the Andarz Company.

## References

1. J. Core, "Combustion Process and Emission Characteristics," presented at the Workshop on Residential Wood and Coal Combustion, Portland, OR, November 1983.
2. M. A. K. Khalil, S. A. Edgerton, R. A. Rasmussen, "A gaseous tracer model for air pollution from residential wood burning," *Environ. Sci. Technol.* **17:** 555 (1983).
3. R. A. Rasmussen, L. E. Rasmussen, M. A. K. Khalil, R. W. Dalluge, "Concentration distribution of methyl chloride in the atmosphere," *J. Geophys. Res.* **85:** 7350 (1980).
4. M. A. K. Khalil, R. A. Rasmussen, "Atmospheric methyl chloride ($CH_3Cl$)," *Chemosphere* **10:** 1019 (1981).
5. J. A. Cooper, L. A. Currie, G. A. Klouda, "Assessment of contemporary carbon combustion source contributions to urban air particulate levels using carbon-14 measurements," *Environ. Sci. Technol.* **15:** 1045 (1981).
6. J. F. Kowalczyk, W. T. Greene, "New Techniques for Identifying Ambient Air Impacts from Residential Wood Heating," in *Residential Solid Fuels: Environmental Impacts and Solutions*, J. A. Cooper, and D. Malek, eds., Oregon Graduate Center, Beaverton, OR, 1981.

7. S. K. Friedlander, *Smoke, Dust and Haze*, J. Wiley, New York, 1977, chapters 5, 11.
8. S. K. Friedlander "Chemical element balance and identification of pollution sources," *Environ. Sci. Technol.* **7:** 235 (1973).
9. R. A. Rasmussen, M. A. K. Khalil, "Atmospheric Halocarbons: Measurements and Analyses of Selected Trace Gases," in *Proceedings of the NATO Advanced Study Institute on Atmospheric Ozone: Its Variation and Human Influence*, A. C. Aikin, ed., U.S. Department of Transportation, Report FAA-EE-80-20, Washington, DC, 1980, pp. 209–231.
10. D. G. DeAngelis, D. S. Ruffin, J. A. Peters, R. B. Reznik, "Source Assessment: Residential Combustion of Wood," U.S. Environmental Protection Agency, Contract No. 68-02-1874, Task 23, Washington, DC, 1981.
11. J. D. Cook, J. H. Himel, R. A. Moyer, "Impact of Forestry Burning Upon Air Quality," GEOMET Report No. EF664, GEOMET, Gaithersburg, MD, 1978.
12. R. W. Gerstle, D. A. Kemnitz, "Atmospheric emissions from open burning," *JAPCA* **17:** 324 (1967).
13. R. W. Boubel, E. F. Darley, E. A. Schuck, "Emissions from burning grass stubble and straw," *JAPCA* **19:** 497 (1969).

Drs. Edgerton, Khalil and Rasmussen are with the Department of Environmental Science, Oregon Graduate Center, 19600 N.W. Walker Road, Beaverton, OR 97006. This note manuscript was peer reviewed.

---

# Backfitting Electrostatic Precipitators: Gas Velocity and Reliability Considerations

**E. M. Frisch and N. W. Frisch**
N. W. Frisch Associates, Inc.
Kingston, New Jersey

This note discusses the backfitting of an existing electrostatic precipitator with a new unit, either in series or parallel. Three cases are discussed: 1) situations in which the modified Deutsch equation is applicable, 2) situations in which gas velocity effects are significant and must be considered, and 3) realistic instances in which performance reliability of the existing unit is an important consideration in the overall system design.

Over the years as emissions standards have become more stringent and as switches to poorer fuel (lower sulfur, higher ash, etc.) have occurred, there has been a need for larger precipitators in a given existing installation. In general, backfitting via series or parallel ESP units are approaches that are considered. This analysis examines these options for the three cases enumerated above.

## Analysis

Consider the existing precipitator to have a collecting electrode (CE) area of $A_0$ ft$^2$ and to perform at an efficiency $E_0$ with the total system gas rate $Q_0$ acfm. The CE area of the backfit ESP is $A_1$ ft$^2$. The backfitted system efficiency is designated by $\overline{E}$. The split of gas to the precipitators in the parallel backfit case is given by the fraction $f$ to the existing ESP. Hence, the efficiency levels of each of the ESPs in the parallel backfit case are designated as $E_f$ and $E_{1-f}$. Subscripts S and P refer to series and parallel backfit situations. Thus, $\overline{E}_P$ denotes the overall operating efficiency of the parallel backfit ESP system. In all cases isothermal operation is assumed. Figure 1 shows the designated quantities.

## Modified Deutsch Equation Applicable

A useful performance equation is one due to Matts and Öhnfeldt,[1] namely

$$E = 1 - e^{-[w_k A/Q]^m} \tag{1}$$

where $E$ is the collection efficiency (fractional);
$w_k$ is the performance rate parameter usually referred to as the modified Deutsch migration velocity, ft/s;
$A$ is the collecting electrode area, ft$^2$; and
$Q$ is the volumetric gas flow rate, acfs.

The exponent $m$ in Eq. 1 is taken as constant for a given situation, generally falling in the range of 0.4 to 0.8 in fly ash situations.

In this analysis the value of $w_k$ is designated as $w_{k_0}$ for the existing ESP and $w_{k_1}$ for the backfit ESP. It is important to note that $w_{k_1}$ is based on a possibly hypothetical situation in which the backfit unit operates with the same particulate feed (size distribution, resistivity, loading-wise, etc.) as the existing unit. This approach is a convenient one as it makes it practical to reference $w_{k_1}$ to $w_{k_0}$ and also avoids the use of a $w_k$ value which, although specific for the performance of the series backfit ESP, would be obscure and also most sensitive to the efficiency of the existing ESP.

*Series Backfit.* Here the efficiency of the existing unit operating with a volumetric flow of $Q_0$ is simply

$$\overline{E}_S = 1 - e^{-[(w_{k0}A_0 + w_{k1}A_1)/Q_0]^m} \tag{2}$$

*Parallel Backfit.* Here the efficiency of the existing unit operating with a volumetric flow of $Q_0$ is

$$\overline{E}_f = 1 - e^{-[w_{k0}A_0/(fQ_0)]^m} \tag{3a}$$

and for the backfit unit the efficiency is

$$E_{1-f} = 1 - e^{-[w_{k1}A_1/(1-f)Q_0]^m} \tag{3b}$$

Assuming uniform gas density in the system, a material balance for particulate yields

$$1 - \overline{E}_P = f(1 - E_f) + (1 - f)(1 - E_{1-f}) \tag{4}$$

Hence for the parallel backfit case the overall loss is

$$1 - \overline{E}_P = fe^{-[w_{k0}A_0/fQ_0]^m} + (1 - f)e^{-[w_{k1}A_1/(1-f)Q_0]^m} \tag{5}$$