Exposure and Risk from Ambient Particle-bound Pollution in an Airshed Dominated by Residential Wood Combustion and Mobile Sources

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A major field study was conducted in Boise, Idaho, during the heating season of 1986 to 1987 as part of the Integrated Air Cancer Project. Filter samples were systematically collected in residences and in the ambient air across the community to characterize the particle-bound pollutants. The extractable organic matter (EOM) from the filter samples was apportioned to its source of origin, either residential wood combustion (RWC) or mobile sources (MS). Two composite samples, with apportioned contributions from RWC and MS, were prepared from the Boise ambient samples and tested for tumor-initiation potency. A comparative potency lung cancer risk estimate has been made based on the two ambient composite samples from this airshed. In addition, a microenvironmental exposure model was developed from the Boise data and from national survey data to estimate the exposure to EOM from RWC and MS. In this paper, the microenvironmental model is extrapolated to provide an estimate of the average annual exposure and dose in Boise to EOM from RWC and MS. The annual model considers actual pollutant levels in Boise, historical changes in RWC usage and meteorological dilution factors and the likely activities in the various microenvironmental zones and their resultant inhalation rates. Combined with the lifetime risk estimates, the average annual dose suggests a risk of about 4 × 10⁻⁴ based on the composite ambient samples. Despite the fact that RWC accounts for 73% of the EOM on an annual average basis, it accounts for only about 20% of the estimated lifetime risk.

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Key words: exposure, dose, mobile sources, lifetime risk, residential wood combustion, wood smoke

Introduction

The Integrated Air Cancer Project (IACP) is a long-term U.S. Environmental Protection Agency (EPA) research project to improve understanding of the atmospheric burden and origins of carcinogenic pollutants (1,2). The goals of the research program are to identify the principal carcinogens in the air to which humans are exposed, to determine which emission sources are the major contributors to the atmospheric burden of carcinogens, and to improve the scientific capability for estimating both human exposure and the resultant comparative human cancer risk arising from exposure to air pollution from specific emission sources. The IACP research strategy has focused on products of incomplete combustion (PICs), especially those from motor vehicles and residential heating. PICs constitute a large fraction of the atmospheric burden of pollutants on a national basis, and motor vehicles and residential heating are major, ubiquitous emission sources of PICs in populated areas.

The first residential heating source to be studied was residential wood combustion (RWC). RWC was selected because it represented one of the largest sources of PICs—polycyclic aromatic hydrocarbons—and mutagenicity on a national basis (2). It was chosen because it was under review for regulatory action; and because the high mass loadings associated with wood smoke would enable sufficient mass to be collected during the field study to conduct the desired chemical and biological analyses. Emissions from mobile sources (MS) were also included in the study since they are ubiquitous in populated areas and since mobile source emissions are estimated to comprise the single largest source of air toxin risk from airborne toxic pollutants (2).

During the heating season of 1986 to 1987, a major sampling program was carried out in Boise, Idaho. The Boise field program included both ambient and indoor residential sampling programs, both of which have been detailed elsewhere (3,4). The Boise study also included surveys of the Boise population to characterize the use of RWC across the community and to identify other possible sources of exposure (1). Two survey questionnaires were used: the first was a general survey dealing with home heating and motor vehicle usage, and the second dealt with RWC in more detail and was administered only to respondents who burned wood. To address the goal of improving assessments of human exposure and risk from airborne carcinogens, the IACP included an effort to characterize exposure levels in indoor residential locations.

The data from the residential and ambient sampling programs in Boise were used, together with daily activity diaries completed by the participants in the residential sampling effort, to develop an exposure model for
estimating wintertime exposures during the field study (5). Indoor sampling data are especially critical to the exposure estimates, since people normally spend about two-thirds of their time in their residences. The IACP strategy also included efforts to improve the risk estimates of particle-bound pollutants from RWC and MS. The extractable organic matter (EOM) bound to ambient particles was apportioned to RWC and MS (6) using tracer compounds indicative of each source type. Two large composite samples, with known contributions from RWC and MS, were prepared from the apportioned samples and used in rodent skin tumorigenicity studies to determine the tumor initiating potency of the samples. The data from the animal tumorigenicity studies were employed in estimating the comparative lifetime cancer risk from the particle-bound ambient pollutants found in Boise air (7).

This paper combines estimates of exposure and risk, extrapolates the data for annual average conditions, and calculates the estimated risk for a population exposed to ambient particulate pollution like that measured in Boise in 1986 to 1987.

Background

Boise was selected as the field study site from a potential list of more than 30 towns and cities for several reasons: a) RWC was known to be a significant contributor to the high aerosol particle loadings which normally occurred in Boise during the fall and winter. b) The airshed appeared to be relatively simple, with no large background or confounding emission sources. c) There were numerous sampling sites available in the Boise area which seemed promising for the objectives of this study. d) The terrain and meteorology seemed appropriate for extrapolation to other locations. e) The local government and environmental agencies expressed strong support for the project. Boise is the capital city of Idaho, with a population in 1980 of slightly more than 100,000 people. The city is a center of state and local government functions and is home to a variety of corporate headquarters. There are no large or heavy industrial sources. The urbanized section of the city is located along the Boise river, which flows through the city from the southeast toward the northwest. The valley floor is approximately 2700 feet above sea level. The area is bordered on the north and east by mountains which rise to an elevation of more than 7000 feet. To the south and west, the land rises in a series of steps, called benches, until a broad plain is reached at 450 feet above the valley floor. Meteorologically, the wind flow during the sampling period should be dominated by up-valley flow during the day and down-valley flow at night.

The ambient sampling in Boise was conducted at three primary sites and four auxiliary sites from November 1986 through February 1987. Sampling periods were 12 hr, with changeover times at 7 A.M. and 7 P.M. There were 13 sampling periods scheduled per week, and one period was dedicated to calibration, maintenance, etc.

The residential sampling each week involved a matched pair of homes. Over the study, 10 pairs of homes were sampled. One of the homes in each pair used a wood stove, a fireplace insert, or a fireplace. The other home did not burn wood. Sampling was conducted in 12-hr periods identical to those at the ambient sampling sites. Sampling began each Saturday morning and was completed at 7 A.M. Wednesday. When samples were collected at the residences, identical samples were also taken at two of the primary ambient sampling sites, one in a residential area impacted by RWC, and one near a roadway. The homes selected for the residential sampling were matched for age, size, etc.

With a sample size of only 20 homes, the residential sampling cannot be considered population-based. The homes used in the residential sampling study were typically volunteered by individuals who had heard about the project, and they usually were located in middle-class communities. A statistical survey of the Boise area found that wood burning was common in 1986 to 1987. Sixty-two percent of all homes in the Boise area had a wood burning appliance, and 40% of all homes used them. Twenty-three percent of all homes burned wood daily, and another 4% burned wood almost every day. The fuels primarily used in central heating systems throughout Boise are natural gas (53%), electricity (31%), wood (2%), coal (1%), all other fuels, (12%). Only 2% of homes reported the use of kerosene heaters. Residential sampling was limited to houses with no confounding combustion sources. This condition was met by two-thirds of all the homes. Specifically, none of the sampled homes had residents who smoked or used kerosene heaters. Just as Boise was chosen because there were no confounding ambient sources, houses without smokers and kerosene heaters were picked to avoid the presence of confounding indoor sources of particles and extractable organic matter.

All participants, 12 years of age or older, from each of the 20 sampled homes were asked to maintain activity diaries for the 4-day period during which their home was being monitored. No special activity questionnaire was administered: rather each respondent was provided with a logbook in which they were asked, for each day, to record a description, location, and duration for each activity lasting five minutes or more. A sample logsheet was provided, illustrating the level of detail desired of the entries. Forty-seven respondents prepared activity logbooks; unfortunately, four of the diaries were not utilized in the exposure model because of internal inconsistencies and large periods of unallocated time.

Methods

Exposure Model Calculations

Exposure is defined as "the contact at one or more boundaries (e.g., mouth and skin) between a human and a contaminant(s) at a specific concentration(s) for a period of time" (8). Mathematically, exposure is the integral, over a period of time, of the time-varying concentrations to which a human is exposed. Exposure, therefore, can be expressed as "the product of the concentration of pollutant to which one is exposed in a particular setting times some specific time period" (9). Exposure can be estimated by splitting the day into the series of distinctive periods that each person spends in a particular microenvironment. Then the product of concentration times time spent in each microenvironment is summed. A microenvironment represents a location or activity that is distinctive in terms of the exposure under investigation. For the full urban population, both the concentrations and the times spent in each microenvironment are expected to be a distribution of values, and the resultant exposures should also be a distribution. Average conditions (e.g., concentrations of pollutants, times spent in each microenvironment, etc.) may be used to represent an average population exposure, but such an approximation does not represent the significant individual-to-individual variability that exists.

The exposure model developed from the Boise data considered five microenvironments, or zones (5). In addition, the model allowed for consideration of the location within the zone, the activity being conducted, and any sources known to exist in the zone. This ZLAS analysis allowed for individual variability in exposures. In extrapolating to annual conditions, however, average conditions have often been assumed for convenience: such an assumption does not imply that large variabilities in exposures do not exist.

Microenvironmental Zones in the Exposure Model

The 43 daily activity diaries from the Boise study were used to estimate the time spent
by the Boise population in five zones, or microenvironments: indoors at home (I); outdoors (O); in-transit (T); at the workplace (W); and at other indoor locations like commercial stores, churches, post office, etc. (X). Table 1 shows the percentage of time spent in each of the five zones, as determined from the Boise winter-time diary data. While the relatively small sample size, 43, and the exclusion of children under 12 years of age suggest uncertainties in the validity of the diary-based exposure model, the Boise data seem, nonetheless, understandable and consistent with other, more extensive, surveys. Table 1 also shows the percentage of time for each zone as determined from a national survey for year-round activity patterns (10). The time allocations for Boise seem reasonable compared with the national survey data. One might expect the time spent indoors during the winter in Boise to be greater than the national annual average, and, conversely, for the time spent outdoors to be less. Boise is a modestly sized city with an estimated population in the mid-1980s of about 108,000 in the center city and around 190,000 in the metropolitan statistical area (11). Commute times, T, in Boise would reasonably be expected to be less than the transit times in a national average.

The actual contact between humans and airborne pollutants by inhalation, however, is more than simply the product of concentration and time. In moving from exposure to dose via inhalation, one must eventually consider the quantity of air inhaled (i.e., the inhalation rate). Lioy (8) describes the potential dose, D, as:

\[ D = \int_{t_1}^{t_2} C(t) f(x) \, dt \]  

where \( f(x) \) is termed the "contact rate" (i.e., for exposure by inhalation, it is the inhalation or breathing rate). Since exposure has the units of concentration \( \times \) time, then the potential dose, which is the product of exposure and the contact rate, has units of mass (e.g., (\( \mu g/m^3 \) \( \times \) days \( \times \) m\(^3\) day\(^{-1}\)) = \( \mu g \)).

Clearly, an individual's inhalation rate is a function of activity level, and the activity level varies from microenvironment to microenvironment. One is more likely to be sedentary or to sleep indoors, and to exercise more vigorously outdoors. In estimating the potential dose for Boise, a contact rate, \( f(x) \), was calculated for each zone in the model by combining the Boise activity data with physiological estimates of inhalation rates. The daily activity diaries (5) of the Boise participants were used to identify the nature and duration of activities that occurred in each microenvironmental zone. The Boise activity data were then combined with ventilation rates from physiological experiments (12,13) to estimate an average inhalation rate for each activity listed by the Boise participants in each of the zones. An average inhalation rate was then calculated for the Boise participants for each zone. Table 1 also shows the average inhalation rate used in this paper as the contact rate dosimetry factor in estimating the potential dose in each microenvironment. Given the obvious correlation of activities and zones, the use of an average for each zone is statistically more appropriate than using a single average inhalation rate. However, the validity of the estimates of average inhalation rate in each zone is limited both by the applicability of the results from the 43 diaries and the appropriateness and accuracy of the physiological estimates used. Using the Boise data in Table 1 for both the inhalation rate and the time in each microenvironment, one calculates a daily inhalation rate of 18.5 m\(^3\) day\(^{-1}\). This value is only about 7% less than the commonly used value of 20 m\(^3\) day\(^{-1}\). If one uses the national activity data in Table 1, a daily inhalation volume of 19.1 m\(^3\) is calculated, which is only 4.3% below the commonly used value. Clearly the assumptions about inhalation rate and zonal activities yield reasonable estimates on a daily average.

Concentrations of Pollutants

Apportionment of the Boise field study data indicated that, on average, the extractable organic matter (EOM) on the fine ambient particles in Boise came primarily (>89%) from RWC and mobile sources (MS) (6). The remaining 11% of the EOM may have come from a different source, or it may also have derived from RWC and MS, with the 11% residual representing the combined uncertainty in the measurements and the apportionment model. Nonetheless, for a first approximation, one may assume that all of the EOM in the winter-time Boise airshed is attributable to RWC and MS.

![Boise Source Factors](image)

**Figure 1.** Comparison of percent utilization of wood-burning appliances in Boise and normal heating degree-days.
The factors that determine the concentrations of pollutants in the ambient air are: the magnitude of the emissions (i.e., the source strength); and the volume of air into which the emissions are mixed. The source strengths for MS and RWC are dependent on vehicle usage and the amount of wood burned. Figure 1 shows the profile for both the use of wood burning appliances in Boise (from the survey) and normal heating-degree days (14). Both values are normalized, so that the largest value represents 100%. It is clear that use of wood burning appliances in Boise parallels the need for heat with slight offsets in fall and spring. Since the normal heating degree-days are a 30-year running average of monthly data, they were used as a factor to adjust the RWC source strength for extrapolation throughout a normal year. The figure also shows the monthly variation of vehicle miles traveled (VMT) in urban areas in 1987 (15). The urban VMT are given as a percent of the annual total, and they range from a low of 7.0% in February to a high of 9.3% in August. A normalization factor for the MS strength was calculated from the VMT data and used to adjust the MS contribution throughout the year.

The ambient concentrations that result from RWC and MS emissions are inversely related to the volume of air into which they mix. The mixing volume is primarily a function of two meteorological parameters, the mixing depth or inversion height, and the wind speed (16). The mixing volume is often smallest during the winter months, which leads to the highest concentrations. Holzworth tabulated seasonally averaged normal mixing heights for Boise and a variety of other U.S. cities (16). The daytime and nighttime mixing heights were averaged and combined with normal wind speed data (14) to provide an approximate mixing volume for each month. The inverse of the product of wind speed times inversion height was normalized and used as a factor to adjust both the RWC and MS strength term throughout the year.

Figure 2 shows a map of the metropolitan area of Boise, with the sampling sites identified. Meteorologically, the wind flow during the sampling period is dominated by an up-valley flow (from the northwest to the southeast) during the day and a down-valley flow at night (from the southeast to the northwest). The sampling sites are oriented primarily along the direction of the anticipated wind flow. The fine mass was nearly uniformly distributed across the inhabited Boise area (3). Only the measurements at Camel Back Park (CBP), which

A similar mathematical relationship also exists between the tracers of RWC and MS, and both the fine mass (particles less than 2.5 μm aerodynamic diameter) and the PM-10 mass observed during the Boise study. RWC and MS account for >88% of the average PM-10 mass. For RWC, 65% of the PM-10 mass is extractable (EOM), while for MS, only 25% of the PM-10 mass was extractable. Not only may the emissions from vehicular traffic be harder to extract, but some portion of the PM-10 mass attributable to MS may be insoluble dust or other particles that are introduced into the air by mechanical action.
The penetration factors for homes without wood burning appliances clustered around 0.5, while factors from homes with wood stoves or fireplaces clustered around 0.7 for RWC and 0.5 for MS. Measurements of the air exchange rates, which are related to the penetration factor, also suggested a mean penetration factor of about 0.5 for homes (18). Table 2 shows the penetration factors assigned to each zone. Penetration factors into homes were assumed to vary seasonally, with a minimum of 0.5 in winter, increasing to 0.7 in spring and fall, and peaking at 0.9 in summer. The increases in spring, fall, and summer are assumed to reflect that homes are more likely to be open during the better weather. The outdoor zone and the transit zone were assumed to have good air exchange, and a penetration factor of 1.0 was assumed. It was assumed that throughout the year, workplace locations and places like shopping malls, stores, office buildings, etc., maintained greater and more consistent ventilation rates than homes in wintertime. The W and X zones were assigned penetration factors of 0.7.

| Zones     | Penetration factor | Relative loadings |
|-----------|--------------------|------------------|
| Indoor    | 0.5, 0.7, 0.9      | 1.10             |
| Outdoors  | 1.0                | 1.10             |
| In-transit| 1.0                | 0.89             |
| Workplace | 0.7                | 1.00             |
| X (other) | 0.7                | 0.89             |

*The penetration factor varied seasonally, from 0.5 in winter, to 0.7 for spring and fall, to 0.9 in summer.*

Mountain View School is located midway between Winstead Park and the Fairgrounds. All available data from both sites were averaged to provide monthly loadings representing data from 5 to 6 years.

The ambient loadings from RWC and MS throughout the year were estimated as the product of three terms: a source term, representing the maximum possible loading from each source type; a normalized factor (0 ≤ f ≤ 1), representing the anticipated dilution by the monthly average mixing volume; and a second normalized factor, representing the monthly change in usage of the source. For the RWC source, the latter factor varied according to normal heating degree-days; it varied according with the monthly changes in urban vehicle miles travelled for MS. The source terms were adjusted until RWC and MS accounted for the same fraction of the PM-10 mass that was observed for December and January during the Boise field study. Data from December and January were used to adjust the source terms, because the inversion height estimates were seasonal averages, and the winter months (December through February) were much more stable and consistent than the rapidly changing months of fall (September through November). Figure 4 shows the monthly average loadings of PM-10 in Boise (as reported in AIRS for 1986–1991), together with the predicted loadings of RWC and MS. The RWC contribution shows a strong variation with season and the need for heating, while the MS contribution varies primarily as the inverse of the estimated mixing value. Contributions in December and January without a known origin may represent either an unidentified source of PM-10, or be indicative of uncertainty in the apportionment methodology. However, these other sources clearly have a significant impact in the spring, summer, and fall. In Boise’s emissions inventory, nearly 90% of the summertime total suspended particulate (TSP) is attributed to
fugitive dust \((19)\). Presumably, this is mostly wind-blown dust from the arid, semi-desert regions around Boise. Nonetheless while TSP and PM-10 are not identical, it is not surprising that particulate loading during the spring through the fall has a significant unknown contribution. The presence of other sources during the year does not reduce the impact of MS and RWC calculated in this paper. It simply means that additional sources, and additional risks if any, must be considered if the complete exposure and risk is to be estimated. The monthly loadings shown in Figure 4 were averaged seasonally and used to estimate dose.

An average dose was estimated on a quarterly basis by multiplying the seasonal average PM-10 concentrations for each of the five zones in the model. The doses of EOM from RWC and MS in each zone were calculated as:

$$D_{\text{EOM}} = C_s \times f_{\text{zone}} \times f_{\text{penetration}} \times f_{\text{EOM}} \times R_{\text{inhalation}} \times f_{\text{time}}$$ \[2\]

where \(C_s\) is the seasonal average concentration of PM-10 associated with RWC or MS; \(f_{\text{zone}}\) is the relative ratio of the RWC or MS loading for each zone (Table 2); \(f_{\text{penetration}}\) is the assumed penetration factor (Table 2); \(f_{\text{EOM}}\) is the fraction of the PM-10 loading that is extractable organic material (for RWC the factor is 0.65, and for MS the factor is 0.25). The factor \(f_{\text{inhalation}}\) is the inhalation rate (\(\text{m}^3 \text{day}^{-1}\)) assumed for the average level of activity in each zone (Table 1); and \(f_{\text{time}}\) is the time spent in each zone, based upon the daily diaries in Boise and upon national surveys (Table 1).

The dose rate, in units of \(\mu\text{g EOM/ day}\), from RWC or MS is summed over all zones to yield a seasonal average dose rate. The doses from both RWC and MS are highest in the winter and lowest in the summer. Figure 5 shows the estimated average daily potential dose rate of EOM (in \(\mu\text{g EOM/day}\)) attributable to RWC and MS as a function of season. The estimate is derived from the exposure model using the factors that have been detailed above. The annual average dose rate is estimated to be around 68 \(\mu\text{g EOM/day}\) for RWC and about 26 \(\mu\text{g EOM/day}\) for MS. Together, RWC and MS are estimated to provide an average dose rate in excess of 90 \(\mu\text{g EOM/day}\). The annual estimate of EOM dose suggests an average exposure concentration of 4.7 ± 1.0 \(\mu\text{g EOM/m}^3\), 3.4 ± 0.9 \(\mu\text{g EOM/m}^3\) from RWC and 1.3 ± 0.3 \(\mu\text{g EOM/m}^3\) from MS. (The uncertainty in the exposure calculation was estimated by allowing each of the factors in Equation 2 to vary randomly and normally, based on the values and distributions measured in the Boise data, and repeatedly calculating the estimated dose. The standard deviation of the resulting estimates were about 20 to 25% of the mean.) Considering only the extractable organic material from RWC and MS, RWC accounts for about 72% of the annual dose of EOM, while MS accounts for about 28%.

**Comparative Potency Calculations**

The comparative potency method for human cancer risk assessment of complex mixtures of polycyclic organic matter (POM) is a method for estimating human cancer risk when there are no human cancer data for the specific POM mixture being assessed, but there are data for a similar POM mixture \((20)\). In applying this method to the POM in Boise \((7)\), the relative tumor potency of ambient aerosol EOM in rodents was compared to the tumor potency of related POM emissions for which there are human data. The ratio of the tumor potencies in rodents for the two similar POM samples is assumed to be equal to the ratio of the potential of the samples to produce human cancers. That fundamental underlying assumption, the constant relative
potency hypothesis, and its validation, are described elsewhere (21-23). The most recent applications of this method to POM sources, including the ambient air in Boise are described by Lewtas (20,24).

The complex POM emissions to which the POM from Boise are compared are from coke ovens, roofing coal tar, and cigarette smoke. These emissions have published cancer unit risk numbers that have been estimated using the human lung cancer data from epidemiological studies of humans exposed to these emissions. The cancer unit risk is expressed as the individual lifetime excess lung cancer risk from continuous exposure to 1 μg EOM/m³ of inhaled air.

The mouse skin tumor-initiation potency is highly correlated with the human cancer unit risks for related POM emission sources as shown in Figure 6 for coke oven, roofing coal tar and cigarette smoke. The cancer unit risk for these three emissions was based on low-dose extrapolations from human epidemiological studies (21) and range from 3.3 x 10⁻⁴ lifetime risk/μg EOM/m³ to 2.2 x 10⁻⁶ lifetime risk/μg EOM/m³ for cigarette smoke. Diesel emissions was the first POM source for which the cancer unit risk was estimated using the comparative potency method (21-22). In Figure 6, the Diesel point's Y axis is based on extrapolation from animal inhalation data of 0.7 x 10⁻⁶ lifetime risk/μg EOM/m³ (25).

**Results**

Two composite ambient samples were prepared from the Boise ambient samples collected at Elm Grove Park and at the Fire station. The contribution of RWC and MS to EOM in each sample was determined by receptor modeling methods using tracer species (7). The ratio of EOM from RWC to that from MS was determined and used to order the listing of the available filter sets. Two composite samples were made, starting from different ends of the list in order to maximize the differences in contribution between RWC and MS. Filters were added to the composites until a total predicted extractable mass of 2.5 g was obtained. Those composites, designated WSC (for Wood Smoke Composite) and WSMSC (for Wood Smoke-Mobile Source Composite), were tested for tumor-initiation potency.

To estimate the human lung cancer unit risk using the comparative potency method, the tumor potency of the EOM from the ambient air particulate matter was determined in dose-response studies in the Sencar mouse skin tumor-initiation assay. The slope (papillomas/mouse/mg) for each Boise composite sample is shown in the Table 3.

The predicted composition of the EOM in the wood smoke dominated composite, WSC, is 78% from RWC, 11% from MS, and 11% residual. The residual component in the composite sample arises from two sources: extractable material from the filters, and material not apportioned to RWC or MS by the apportionment model. Extraction of unexposed filters yields a small quantity of organic material. This blank remains in the composite sample and contributes to the residual. This material is not highly mutagenic (26), and therefore, is not expected to contribute substantially to the tumor-initiation potency. As mentioned above, the second source of the residual EOM derives from the fact that the multiple linear regression method used to estimate the contributions of RWC and MS contains a small positive intercept. The presence of the constant intercept in the regression equation means that there is a small fraction of EOM in every sample that is associated with the constant. In the composite samples, this contribution adds a few percentage points to the residual component.

The EOM represented by the constant term in the regression may arise from the contribution of an unaccounted third source, or it may simply represent the combined uncertainties of the many ambient measurements, and of the model in representing physical reality. The published error bounds for the constant term (5,6) indicate that a value of zero can not be ruled out with confidence. Indeed, a zero intercept is likely. In reviewing the Boise data for this paper, the authors reexamined the complete data set, rejecting data from observational periods when comments by the field operators indicated a questionable sample. Of the 343 acceptable sample sets, 74 had EOM and tracer data to permit a regression calculation. Analysis of this refined data set yielded coefficients for the tracers almost identical to the published (5) values, but the intercept was calculated as -0.1 ± 1.6. Clearly, a zero intercept is possible. A zero intercept would indicate that RWC and MS are the only sources of EOM present. Any fraction of the residual mass that actually comes from RWC and MS, but is not assigned to those sources because of uncertainties in the measurements and models, would be expected to contribute proportionately to the RWC and MS contributions to the tumor-initiation potency.

The relative tumor-initiation potency is used in the comparative potency method to determine the lung cancer risk estimates as reported by Lewtas et al. (7,27). The results of the cancer unit risk estimation for the two ambient aerosol samples are shown in Table 4 together with estimates for

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**Table 3.** Composition and tumor-initiation potency of Boise ambient composite samples.

| Sample, %RWC, %MS | Tumor-initiation potency, papillomas/mouse/mg* |
|-------------------|-----------------------------------------------|
| WSC, 78/11/11     | 0.085, 0.0711-0.125                           |
| WSM, 51/33/16     | 0.215, 0.174-0.261                            |

Abbreviations: RWC, residential wood combustion; MS, mobile sources; WSC, wood smoke composite; WSMSC, wood smoke-mobile source composite.  
* Maximum likelihood estimate (lower bound-upper bound).
Table 4. Estimated lifetime risks from various sample sources.

| Emission source                      | EOM unit risk, Lifetime risk/µg EOM/m³ |
|--------------------------------------|----------------------------------------|
| Diesel vehicle: Nissan               | 3.60 x 10⁻⁴                          |
| Diesel vehicle: Oldsmobile           | 0.95 x 10⁻⁴                          |
| Diesel vehicle: VW Rabbit            | 0.27 x 10⁻⁴                          |
| Diesel vehicle: Mercedes             | 0.95 x 10⁻⁴                          |
| Gasoline vehicle: catalyst, unleaded | 0.42 x 10⁻⁴                          |
| Gasoline vehicle: noncatalyst, loaded| 1.10 x 10⁻⁴                          |
| Airtight woodstove: hardwood (oak)   | 0.05 x 10⁻⁴                          |
| Airtight woodstove: softwood mixture | 0.27 x 10⁻⁴                          |
| WSC, 78% WS / 11% MS / 11% residual  | 0.57 x 10⁻⁴                          |
| WSMSC, 51% WS / 33% MS / 16% residual| 1.28 x 10⁻⁴                          |

Abbreviations: EOM, extractable organic matter; WSC, wood smoke composite; WS, wood smoke; MS, mobile source; WSMSC, wood smoke-mobile source composite.

Figure 7. Lifetime risks estimated for Boise ambient composites and various source samples.

wood stove and automotive source samples and other comparative POM sources.

The animal tumor potency and estimated human cancer risk from exposure to the EOM from particles directly emitted from mobile sources (e.g., diesel and gasoline vehicles) is greater than that for EOM from wood stove emissions. A similar difference was observed in the tumor potency of an ambient sample dominated by woodsmoke (WSC: 78% RWC/11% MS) compared to one containing significantly more automotive emissions (WSMSC: 51% RWC/33% MS). The increase in motor vehicle-related content is sufficient to produce a significantly different (2.3-fold) tumor potency for these ambient air particulate samples.

Discussion

This study presents the first direct quantitative estimate of the tumor potency and human cancer risk from the organic matter adsorbed to ambient aerosols in an urban area. This airshed contained EOM essentially only from woodsmoke and motor vehicle emissions. The tumor potency of the organic matter is significantly higher for the sample with the greater motor vehicle emissions content. These data are consistent with the higher tumor potency measured for mobile source emissions when compared to emissions from wood stoves. Source apportionment of the mutagenic activity of extractable organic matter associated with fine particles in the Boise airshed during this period demonstrated that, although the average concentration of EOM from woodsmoke was greater than from mobile sources (14 µg EOM/m³ to 6 µg EOM/m³), the contribution of mobile sources to the mutagenicity in the airshed was greater, due to the estimated 3.6-fold higher mutagenic potency of the mobile source emissions in Boise (28). A similar result is also observed for the estimated cancer risk based on relative animal tumor potency (not on the relative mutagenicities).

The composition of the annual estimate of exposure to EOM (considering only EOM from RWC and MS) was 72% RWC and 28% MS. This ratio is intermediate between the two composite samples from Boise that were used to estimate the lifetime risk from exposure to the ambient particulate pollutants. Excluding the residual mass from filter blanks and from uncertainties in the apportionment model, the composite samples were 87:13 RWC:MS and 61:39 RWC:MS. Interpolating between the risk values, one estimates the risk for a 72:28 mixture of RWC and MS to be 0.99 x 10⁻⁴ lifetime risk/µg EOM/m³. At the estimated annual exposure concentration of 4.7 µg EOM/m³, an individual lifetime risk of 4.7 x 10⁻⁴ is calculated.

Figure 7 shows the cancer unit risk estimates for a variety of samples as a function of the percentage of RWC in the sample. The plot considers only EOM from RWC or MS. The data at 100% RWC (or alternatively, 0% MS) represent risk estimates for wood stove source samples. The points at 0% RWC (or 100% MS) are the results of samples from a variety of gasoline-fueled or diesel vehicles. The Boise ambient composite samples are also plotted, assuming that all of tumor-causing EOM derives only from RWC or MS contributions. The asterisk in the figure represents the 72:28 RWC:MS mixture estimated to represent the annual average composition of exposure.

If one extrapolates beyond the Boise composite data, the cancer unit risk estimates in Table 4 for wood stoves seem consistent with the perceived 100% RWC intercept. Using the risk for the softwood mixture (0.27 x 10⁻⁴ lifetime risk/µg EOM/m³) with an estimated annual exposure concentration for RWC of 3.4 µg EOM/m³, one estimates that the wood smoke component contributes 0.92 x 10⁻⁴ to the total risk. The RWC source component, therefore, appears to account for only about 20% of the risk in the ambient sample. The remaining 80% of the risk is associated either with the mobile source component or, perhaps, is caused by transformation of the complex RWC and MS mixture. Even though Boise was selected for the IACP field study because of its significant RWC contribution to the ambient pollution, it appears that the RWC pollution may be responsible for only a small fraction of the total risk because of exposure to the particulate-bound pollutants present in Boise air.

This analysis focuses only on the exposure to, and risk from, extractable organic materi-
such as intrinsic to any study that uses measured data as the basis for estimating lifetime cancer risk. Other assumptions, especially those used in the exposure model regarding activity patterns, inhalation rates, infiltration factors, etc., are based upon small sample sets (i.e., 20 homes and 43 participants) in Boise. The uncertainties involved in extrapolating data from such small samples are hard to quantify. Because of the small size of the sample set, the authors have attempted to specify in detail the factors used in the exposure model and have compared them with national data, when available. (The Boise data seem to be consistent with the national data.) Finally, the major conclusions—that the estimated lifetime risk based upon the ambient samples from Boise is $>1 \times 10^{-6}$ and that RWC contributes the bulk of mass to the EOM but accounts for a much smaller fraction of the risk—are insensitive to many of the assumptions in the exposure model.

This study suggests that the POM associated with ambient particle bound pollution is a major source of cancer risk and that mobile source emissions make a larger contribution to this risk than wood smoke. An independent analysis of cancer risks across the entire United States due to outdoor exposures to airborne toxic pollutants has recently been reported by the U.S. EPA's Office of Air Quality Planning and Standards (29). This study did not involve collecting new exposure or risk data, but analysed information derived from a series of published studies and reports to derive a list of cancer unit risk estimates and estimates of the national exposures. This study concludes that PICs comprise a major fraction (35%) of the identified risk from toxic air pollutants and of this fraction, mobile sources contribute substantially more to the risk than does wood smoke. Although the conclusions of this report are generally consistent with the findings of this study, the report acknowledges the large uncertainties in the cancer unit risk factors used for complex mixtures of PICs and the need for more research to establish improved estimates of cancer risk from PICs and POM. This study reports for the first time an alternative approach to estimating the cancer risk form ambient particle-bound pollution in an airshed dominated by residential wood combustion and mobile sources using exposure data collected in the field and ambient samples of POM evaluated in animal tumor studies.

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