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FY 1983 Annual Report

BUILDING VENTILATION AND INDOOR AIR QUALITY
Excerpt from
Energy Efficient Building Program Chapter

July 1984

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BUILDING VENTILATION & INDOOR AIR QUALITY

Applied Science Division
Lawrence Berkeley Laboratory
University of California
Berkeley, California 94720

Excerpt from the Energy Efficient Buildings Program Chapter
of the Annual Report, FY 1983

July 1984

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BUILDING VENTILATION & INDOOR AIR QUALITY

ANNUAL REPORT FY 1983

important considerations. In particular, the residents frequently complained of their inability to open and close the windows, and of their dissatisfaction with the heat pumps. Operation of the thermostat (a rather complicated affair with three control levers and two status lights) appears crucial to the whole pattern of energy use, and we will continue to study its use and misuse.

Planned Activities for FY 1984

We hope to monitor the energy use of another multi-unit building, although no new work is planned at this time for the Oakland apartment project. (An occupant survey may be undertaken in Oakland in the summer of FY 1984 to study behavioral effects in some detail if sufficient manpower and funding are available.) Work at the housing project for the elderly will continue with winter monitoring of indoor temperatures, heat pump performance, and energy use. We will continue to monitor weather conditions and analyze energy use by individual units. Changes in mechanical systems may be carried out, to analyze their impacts on energy use and comfort. The results will be written up as guidelines for the project architects, as operation and maintenance manuals for the Housing Authority, and as an illustrated booklet for the residents.

Building Ventilation and Indoor Air Quality*

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Lawrence Berkeley Laboratory (LBL) carries out a broad program of work on indoor air quality in the context of its research on energy-efficient buildings and associated environmental effects. Efforts of the Building Ventilation and Indoor Air Quality Program include characterization of the emission of various pollutant classes from their respective sources, study of the effectiveness of ventilation in removing pollutants from indoor atmospheres, and examination of the nature and importance of chemical and physical reactions that can affect the occurrence and amount of airborne pollutants. The program has groups devoted specifically to three major pollutant classes: combustion products, arising from indoor appliances and other sources; radon and its daughters, arising from materials that contain radium, a naturally occurring radionuclide; and formaldehyde and other organics, arising from a variety of materials and furnishings. In addition, other groups study techniques for controlling airborne concentrations, develop devices for monitoring pollutants in laboratories and buildings, and design or carry out field surveys of indoor air quality. Significant effort is also devoted to assessment of the health effects of indoor pollutant exposures.

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The core of these efforts consists of research on the behavior of indoor air pollutants, specifically the influence on airborne concentrations of three classes of factors: source strengths or emission rates, ventilation rates or effectiveness, and reaction and transformation mechanisms. The work carried out by the individual groups is primarily experimental, consisting of fundamental—and often quite sophisticated—experiments on pollutant emission, transport, and removal, complemented by significant efforts advancing the theoretical framework within which results may be considered.

POLLUTANT EMISSIONS FROM COMBUSTION APPLIANCES

Researchers in the Building Ventilation and Indoor Air Quality (BVIAQ) Program and elsewhere have demonstrated that operating unvented combustion appliances increases indoor pollutant concentrations: higher levels of carbon dioxide (CO_2), carbon monoxide (CO), nitrogen dioxide (NO_2), nitric oxide (NO), formaldehyde (HCHO), and respirable particles have all been observed. In FY 1983, the BVIAQ Program completed the data analysis of its controlled field studies of pollutant emissions from unvented gas-fired space heaters[1,2] and conducted similar studies of unvented kerosene-fired space heaters and wood-burning stoves.

In general, our research efforts are devoted to: (1) characterizing the emission rates and pollutant concentrations of the many types of combustion devices used in buildings, (2) assessing the health implications of exposures to combustion-related pollutants, and (3) exploring techniques to reduce pollutant concentrations while maintaining energy efficiency in buildings.

Accomplishments During FY 1983

Data from our controlled field study of indoor air pollution from unvented gas-fired space heaters (UVGSH) were analyzed in detail to project steady-state CO_2, CO, NO_2, NO, and O_2 indoor concentrations resulting from the use of UVGSHs and to determine the time-dependent relationship of the CO, NO_2, and NO_x (NO + NO_2) emissions from the UVGSHs. Eighteen field tests were conducted on four UVGSHs, coded as 12A, 16B, 30A, and 30C. [The numbers, multiplied by 1000, are the manufacturers' heat input rating in units of Btu/h (1.054 kJ/h), and the letters stand for the manufacturers.] The tests were conducted at a research house maintained by LBL at Walnut Creek, CA.

"Projected" Steady-State Pollutant Concentrations

Since steady-state concentrations were not always reached before the UVGSH was turned off, the final pollutant concentration is a percentage of the steady-state concentration. This percentage was theoretically determined using the measured air exchange rate of the house, the pollutant reactivity rate, and the combustion time of the appliance. A "projected" steady-state value was then determined by correcting the final pollutant concentration. All concentrations reported here are averages of the kitchen, living-room and bedroom values. Except for two short 2-hour tests of the 12A and 30C heaters, all pollutant concentrations reached 78% to 98% of their steady-state concentration.

Graphs depicting the projected steady-state pollutant concentrations for CO_2, CO, and NO_2 versus the air exchange rate of our research house are given in Figs. 1–3. The curved lines in Fig. 1 represent

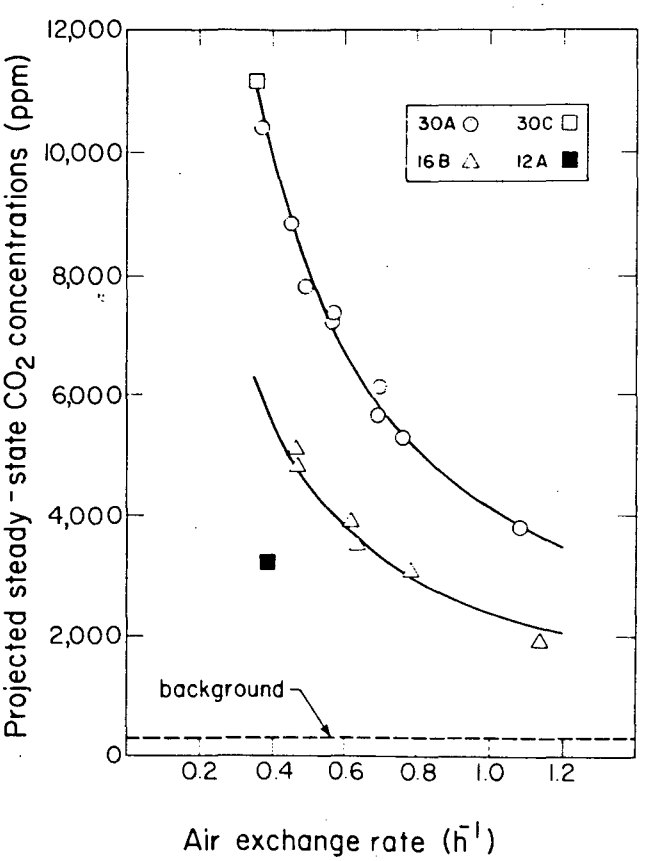

Figure 1. Projected steady-state CO_2 concentrations associated with the use of unvented gas-fired space heaters vs. the air exchange rate of the research house (a 240 m^3 unoccupied building). The solid lines are empirical fits of the data to the reciprocal of the air exchange rate for the 30A/30C and 16B heaters. (XBL 834-9040A)

empirical fits of the data to the reciprocal of the air exchange rate for the 30A/30C heaters and the 16B heater. These curves demonstrate that the steady-state concentrations of nonreactive gases whose source strength is constant follow the theoretical dependence on air exchange rate. Figure 1 shows that the CO_2 concentrations generated range from 2,000 ppm to 11,000 ppm; for comparison, the U.S. Occupational Safety and Health Administration (OSHA) 8-hour time-weighted workplace CO_2 standard is 5,000 ppm.[3] The steady-state CO_2 levels generated from the 30A/30C heaters exceeded the OSHA standard for all experiments where the air exchange rate was below 0.8 ach. The steady-state CO_2 levels generated by the 16B heater approached the OSHA standard at 0.46 ach.

Steady-state concentrations of oxygen were always above 19%. The lowest oxygen level measured was 19.5%; this occurred when a 31,600 kJ/h heater was operated for approximately 4 hours at an air exchange rate of 0.37 h^{-1}.

Our results show that the CO levels in a house with an UVGSH can exceed outdoor air quality standards (Fig. 2). Nine of the 18 tests resulted in projected steady-state CO levels exceeding the Environmental Protection Agency's 8-hour standard of 9 ppm.[4] Seven tests involved the 16B heater. The 30A heater was tested under an excess-air (maltuned) condition and exceeded the EPA's one-hour standard of 35 ppm.

Nitrogen dioxide concentrations were high relative to the only existing short-term standard for NO_2 that has been promulgated by a state or federal agency; this is the State of California standard of 0.25 ppm for a one-hour average exposure.[5] All projected NO_2 steady-state concentrations exceeded this one-hour standard (see Fig. 3).

Variable Emission Rates

An important observation was that the emission rates of CO, NO, NO_2, and N (of NO_x) varied throughout the operating period of the heater. A technique was developed to determine an emission rate every 24 minutes (the time to complete a set of bedroom, living room, kitchen, and outside measure-

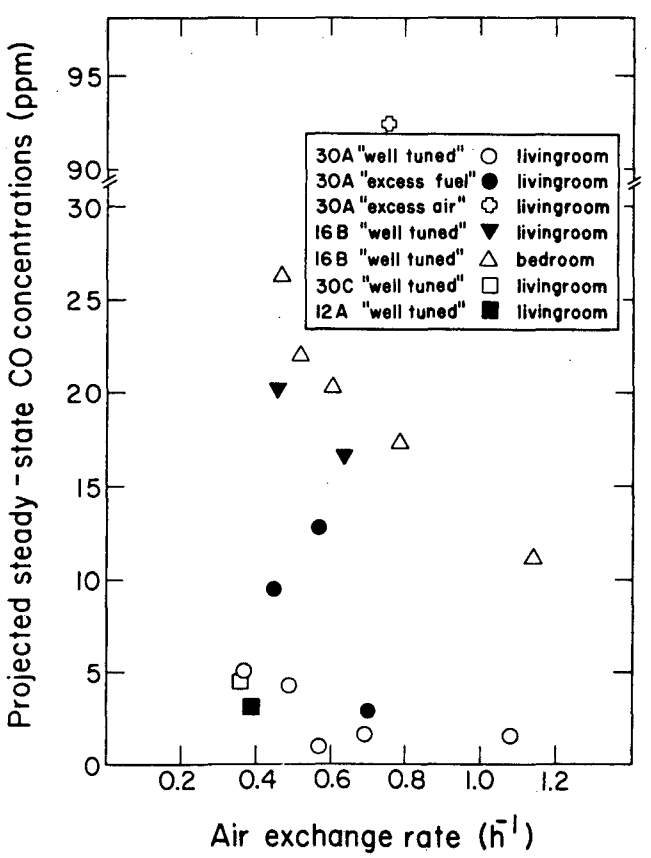

Figure 2. Projected steady-state CO concentrations associated with the use of unvented gas-fired space heaters vs. the air exchange rate of the research house.
(XBL 834-9042)

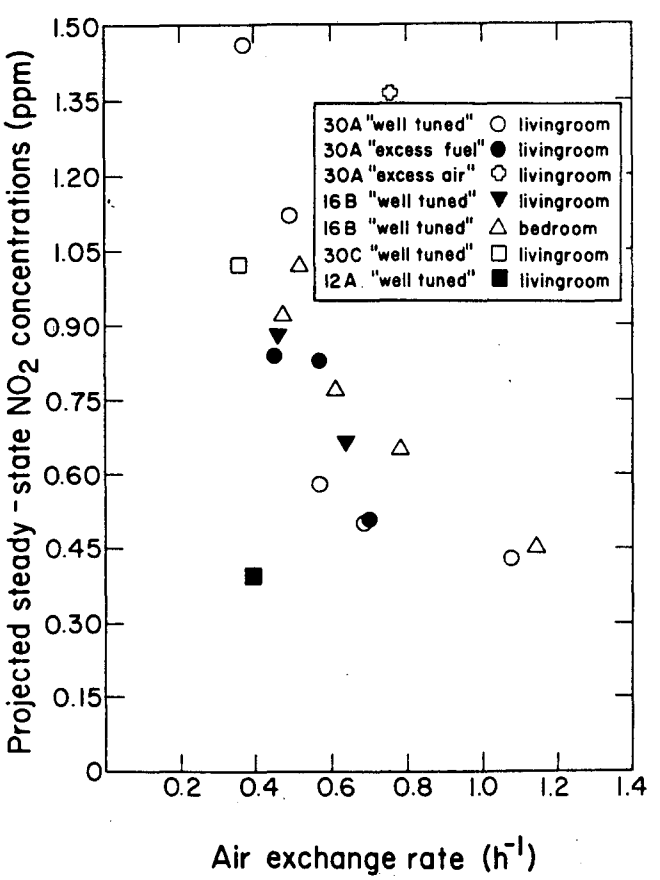

Figure 3. Projected steady-state NO_2 concentrations associated with the use of unvented gas-fired space heaters vs. the air exchange rate.
(XBL 834-9043)

ments).[1,2] The results of this "semi-continuous" emission rate analysis of CO, N (of NO_x), and NO_2 are shown on Figs. 4–6 for representative tests.

The results show that the CO emission rate can change drastically with time. The initial emission rate of CO, determined within 5 to 15 minutes of ignition, can be very different from the final steady-state emission rate. Figure 4 shows this great variability in the CO emission rate for representative tests. The causes of this variability over time are not well understood. In some tests, the emission rate drops during an initial warm-up period (see tests 30A-2, 30A-6, 30C-1, and 12A-1 in Fig. 4). After warm-up, changing O_2, CO_2, and H_2O concentrations may affect the CO emission rate profile by slightly changing the flame temperature or other flame characteristics. This may explain the rise in the CO emission rate for the 30A-2, 30A-6, and 30C-1 tests between approximately 20 and 90 minutes.

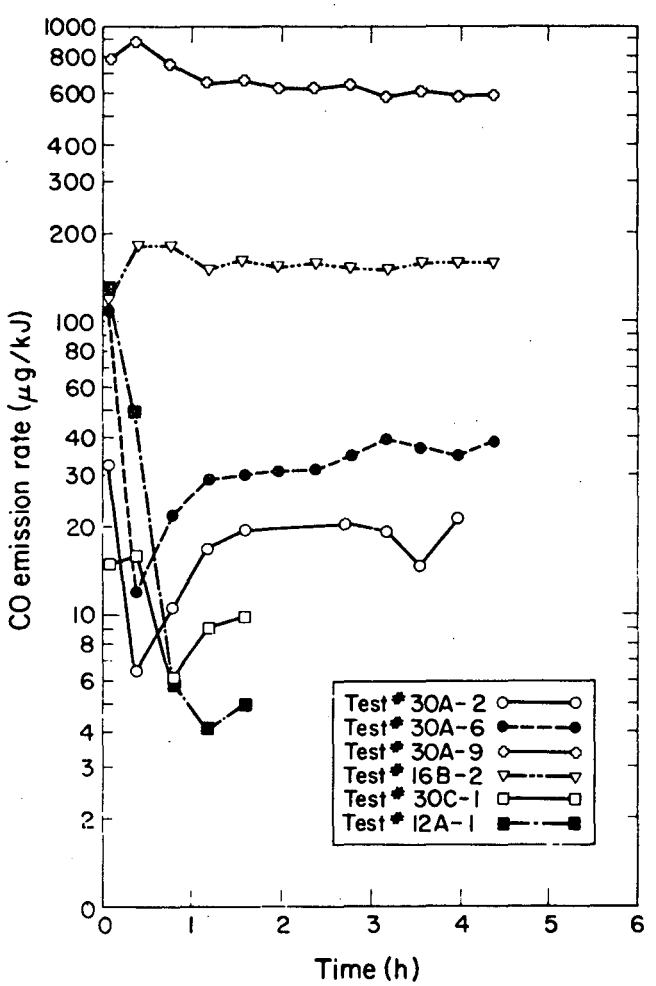

Figure 4. CO emission rates vs. time for representative controlled field tests. (XBL 835-9772)

Semi-continuous emission rate data for representative tests are shown graphically in Fig. 5 for N (of NO_x) and in Fig. 6 for NO_2. For N (of NO_x), there is a consistent trend in which the emission rate increases for an initial warm-up period, peaks between approximately 15 and 60 minutes of operation, then gradually decreases with time. Although not all factors that affect the temporal N (of NO_x) emission rate profile are known, it appears to be affected by the two phenomena that affect some of the CO emission rate profiles. First, the N (of NO_x) emission rate rapidly increases while the heater is warming up, when, presumably, the average and/or peak flame temperature is increasing as well. Second, the N (of NO_x) emission rate decreases as the O_2 level decreases and the CO_2, H_2O, and other combustion product concentrations increase. This change in supply air composition may cool the flame, thus accounting for the general decrease in the N (of NO_x) emission rate. Although consistent with some CO emission rate profiles, this explanation is speculative, and further research is needed to test it. The NO_2 profiles are more complicated than those of N (of NO_x), probably because the NO_2 emission

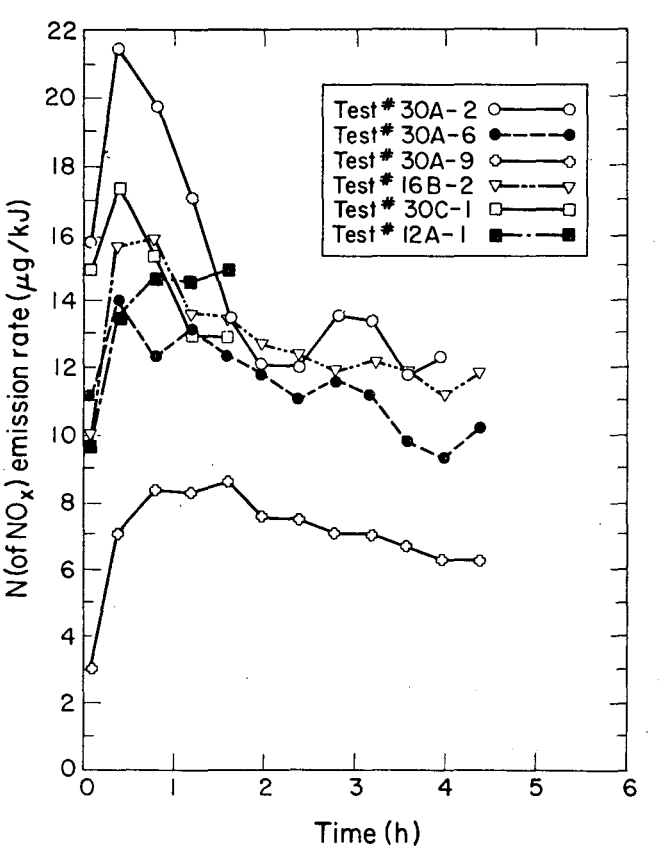

Figure 5. N (of NO_x) emission rates vs. time for representative controlled field tests. (XBL 835-9771)

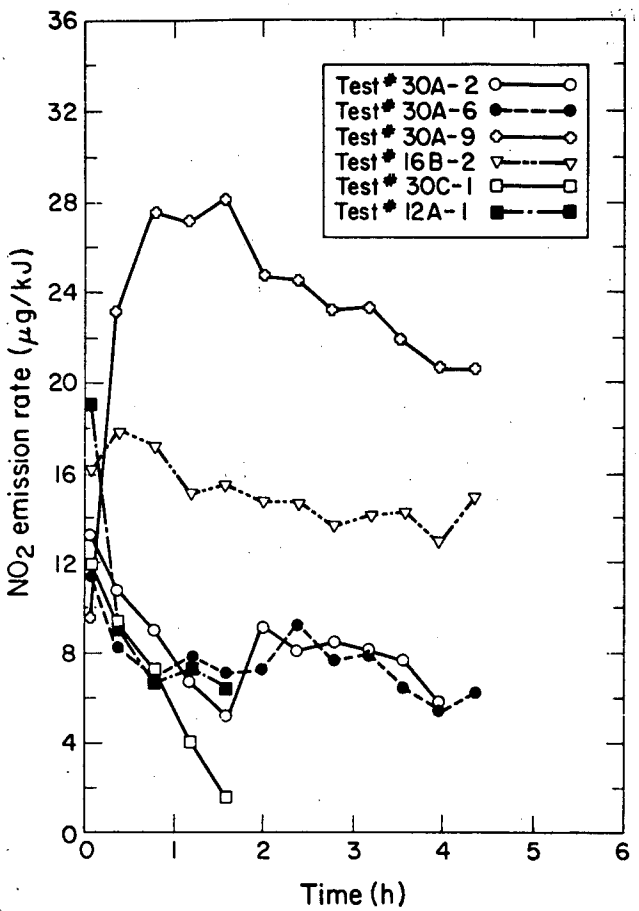

Figure 6. NO_2 emission rates vs. time for representative controlled field tests. (XBL 835-9770)

depends on factors other than flame temperature while the N (of NO_x) emission rate is driven primarily by the flame temperature.

Planned Activities for FY 1984

We plan to complete the data analysis of our field research on wood-burning stoves and portable kerosene-fired space heaters. For wood-burning stoves, data have been collected on CO, CO_2, NO_x, formaldehyde, and suspended-particle emissions. (Analysis for benzo (a) pyrene and other polycyclic organic compounds was conducted on the suspended particles.) Similar data have been collected on the field use of portable kerosene-fired space heaters, and analysis of these data will yield inter-room pollutant transport rates.

New work starting in FY 1984 includes laboratory tests on the new generation of unvented gas-fired space heaters that incorporate oxygen depletion sensors (ODS). In addition, protocols will be developed to study indoor air pollution from combustion appliances under actual field conditions. This research will complement our laboratory and field studies.

ORGANIC EMISSIONS FROM BUILDING MATERIALS

Because heating or cooling outdoor air as it enters an office building consumes a significant amount of energy, recent office construction practices have minimized the use of outdoor air for ventilation. These practices, along with the increased use of synthetic building materials and furnishings, have resulted in a large number of nonspecific health complaints by office workers that may be related to poor indoor air quality. The underlying causes of these complaints are unknown. Few studies of organic contaminants in indoor air in non-industrial buildings have been made; however, fragmentary evidence suggests that such contaminants can be major contributors to poor indoor air quality.[6,7] In the industrial indoor environment, elevated levels of organic contaminants have been shown to induce a variety of toxic, carcinogenic, and teratogenic effects. The possibility that chronic exposure to these same compounds may occur in non-industrial environments makes it imperative that these contaminants be identified and their levels and sources characterized.

Identification of contaminants will require studies of both organic emissions from building materials and organic concentrations in the air of office buildings. Only then can research begin into the health effects on occupants. These studies will also provide the range of organic concentrations that should be addressed and may also suggest which compounds are often associated with each other. The latter information will indicate whether synergistic effects should be examined. Finally, field studies of organic contaminants in office buildings under different conditions of ventilation, temperature, and humidity combined with similar studies of organic emissions from building materials will provide a data base for architects and builders so that materials with low potential for emissions can be used in new, energy-conserving buildings.

Accomplishments During FY 1983

Since the resumption of research in this project area, we have constructed and successfully used a system for the vacuum extraction of volatile organic compounds (VOC) from building materials. Depending on the material screened, there are two variations of the procedure. In the case of construc-

tion and interior finish materials, a sample of about 9 g is introduced into the vacuum system; in the case of adhesives, preweighed plates are coated with approximately 1 g of the material and dried for 9–14 days before introduction into the vacuum system. After cryogenic trapping, the VOC are taken up in solvent and injected into a gas chromatograph–mass spectrometer (GC-MS) for analysis. Compounds are identified by comparing unknown spectra with spectra contained in the EPA/NIH Mass Spectral Data Base.[8] This system allows quick screening of building material samples to determine which contain significant quantities of VOC of interest and thereby warrant further study. In addition, it has provided the basis for designing a more sophisticated system that includes temperature- and humidity-controlled flow of clean air and solid sorbent trapping.

Organic compounds emitted by 15 building construction and interior finish materials and 15 adhesives with indoor applications were identified by vacuum extraction and GC-MS analysis. These included representative samples of structural and insulating materials; interior finish materials used for floor, wall, and ceiling treatments; and adhesives. A total of 68 major compounds with molecular weights ranging from 94 to 458 were identified in the vacuum extracts of construction and interior finish materials; the 13 most frequently occurring compounds are listed in Table 1. The lowest molecular weight compounds were phenol, alkyl benzenes, naphthalenes, and terpenes. The most frequently occurring compounds were plasticizers: dibutyl phthalate and 2,2,4-trimethyl-1,3-pentanediol diisobutyrate. The number of VOC varied greatly among the materials; for example, a carpet pad emitted only one compound, while a synthetic textile wall covering emitted more than 60.

Eight of the 15 adhesives studied continued to emit significant amounts of VOC after 1 week of drying. The major compounds identified in the vacuum extracts of the solvent-based adhesives are toluene (the most abundant), styrene, and a variety of cyclic, branched, and normal alkanes. A more complete listing of VOC emitted by adhesives is found in Table 2. Surprisingly, a large variety of alkanes were present in the extracts of three of the water-based adhesives; two of these, both from the same manufacturer, emitted an almost identical, complex mixture of normal and branched alkanes and cyclohexanes. Two other adhesives, S-6 and W-3, emitted a different mixture of higher-molecular-weight normal and branched alkanes. The similarity of the emissions from S-6 and W-3 is unexpected because one is a solvent-based asphalt adhesive and the other is a water-based natural rubber adhesive, the intended applications are very different, and they are supplied by different manufacturers.

Table 1. Most frequently occurring major compounds emitted by 15 representative building construction and interior finish materials.

Frequency (no. of samples)	Compound
10	Dibutyl phtalate
9	Di-isobutyrate (2,2,4-trimethyl-1,3-pentanediol)
7	n-Pentadecane
7	Diethyl phthalate
7	n-Heptadecane
7	n-Octadecane
7	Pristane
4	Dimethyl phthalate
4	n-Tetradecane
3	Benzophenone
3	n-Hexadecane
3	n-Nonadecane
3	Phytane

Adhesives have been suspected of causing widespread health complaints in office buildings[9]; for this reason, and because of the large variety and amount of VOC emitted by eight of the adhesives studied, we measured the VOC emission rates of these eight. A gas chromatograph equipped with a flame ionization detector was used with a flow-through exposure system to quantify VOC emissions. Humidified (40% relative humidity at 25°C), high-purity compressed air at room temperature and low pressure was passed over the adhesive samples, and the VOC emitted were trapped with a two-stage adsorption tube. After drying 9–14 days, only five of the adhesives had detectable solvent emission rates (Table 2). Three solvent-based adhesives had toluene emission rates between 0.6 and 62 mg g^{-1} h^{-1}. The two water-based adhesives, W-1 and W-2, emitted the same complex mixture of normal and branched alkanes and cyclohexanes in the emission rate tests as they had in the vacuum extraction screening tests. Because of the complexity of the mixture, emission rates were not determined for individual compounds. Instead, rates for total

Table 2. Volatile organic compounds emitted by adhesives with indoor applications. Emission rates are given for toluene or total alkanes for selected adhesives.

Adhesive chemical base	Application	Volatile organic compounds	Weight of adhesive (10^{-2}g cm^{-2})	Drying time (days)	Emission rate (μg g^{-1} h^{-1})
Solvent-based					*Toluene*
Synthetic rubber S-1	Rigid plastic foams for walls, ceilings	toluene; styrene	6.70	9	0.59
Unspecified polymer S-2	Styrene, polyurethane, cork, celotex, carpet	low-molecular-weight alcohol?; toluene			
Synthetic polymer S-3	Plasticized rubber, vinyl carpet	toluene	0.93 0.83	13 13	62 48
Asphalt S-6	Roof shingles, masonry cracks	*n*-decane, *n*-undecane, C_{10}–C_{11} branched alkanes(9+ compounds), C_{10} cyclohexanes (4 compounds)			
Unspecified polymer S-7	Subfloor bonding	methyl cyclopentane; cyclohexane; toluene	4.32 6.50	11 11	2.4 2.6
Water-based					*Total alkanes*
Synthetic rubber W-1	Foam-backed indoor and outdoor carpet	*n*-octane, *n*-nonane, C_8–C_9 branched alkanes (7+ compounds), methyl cyclohexane, C_8–C_9 cyclohexanes (10+ compounds)	2.54 2.85	14 14	740 760
Synthetic rubber W-2	Foam-backed indoor and outdoor carpet	same as compounds W-1	3.36 3.82	14 14	610 780
Natural rubber W-3	Foam-, sponge-backed indoor and outdoor carpet	toluene; *n*-nonane, *n*-decane, *n*-undecane, C_{10}–C_{11} branched alkanes (9+ compounds), C_{10} cyclohexane			

alkanes were calculated (Table 2). These emission rates ranged between 610 and 780 mg g^{-1} h^{-1}.

To assess the potential impacts of adhesives on indoor air quality, the emission rates determined in this study were used in a single-equation, mass-balance model that assumed well-mixed air. The model results demonstrated that the two water-based adhesives could produce significant concentrations of VOC in a hypothetical office space. However, these model results must be used with caution, since in an

office space the adhesives would be overlaid with carpet, and the carpet would undoubtedly moderate the VOC emissions. Thus the model results may represent an upper bound on VOC concentration. A more complete description of these studies can be found in a recent report.[10]

To assess occupant exposure to indoor pollutants in commercial and institutional buildings, we have developed an audit strategy for air quality and ventilation system performance.[11] As a first step, potential sources of pollutants (including radon, formaldehyde, carbon monoxide, carbon dioxide, suspended particles, and, perhaps most important, VOC) are recorded by the audit team on the basis of visual inspections and discussions with building managers. Then air sampling sites are selected to characterize the effect of different air handling systems, occupant activities, interior layouts, and proximity to potential pollutant sources. Air is sampled only during times of normal building occupancy. Mechanical ventilation systems are evaluated to determine the amount of outside air being delivered to the occupants.

Aspects of this audit strategy were tested in a building evaluation performed as a joint project with Public Works Canada. Various pollutants, including VOC collected on Tenax-GC, a solid sorbent, were sampled. After spiking with an internal standard, VOC samples were thermally desorbed into a GC-MS, where individual compounds were identified and quantified. Other pollutants were measured using standard sampling and analytical techniques. Ventilation system performance was evaluated using tracer gas, smoke sticks, detailed mapping, and thermography. Recommendations for actions required to improve air quality by both source removal and improved ventilation system performance are being developed.

Planned Activities for FY 1984

Studies of solid sorbent trapping materials exposed to an array of VOC and to thermal desorption will be conducted to optimize sampling conditions, not only when measuring emissions from building materials but also when measuring VOC concentrations in office buildings under field conditions. We plan to continue our studies of adhesives, with emphasis on determining emissions under a range of temperatures, humidities, and air flows while the adhesives are under simulated use, gluing appropriate materials such as carpets and tiles. We will also initiate a study determining the methylene chloride concentrations produced through the use of commercial paint strippers. Concentrations of major solvents other than methylene chloride will also be measured. The first phase of this study will be conducted in a large, recently constructed environmental chamber. In the second phase, we will attempt to scale down the test procedure to allow its use in small environmental chambers. Simple modeling will be an integral part of this study. Finally, concurrent with the activities outlined above, we will begin field sampling for VOC in a variety of commercial office buildings.

CHARACTERIZING THE SOURCES OF RADON IN BUILDINGS

Radon-222, a naturally occurring radioisotope and the heaviest noble gas, is produced by the decay of radium-226, a ubiquitous trace element in earth and earth-based materials. Concentrations of radon are often higher indoors than outside—commonly by an order of magnitude—because the house, acting like a leaky container, retards the transport of radon from radium-bearing materials to that part of the atmosphere that is higher than the breathing zone. Indoor radon is of concern because uranium miners developed a high incidence of lung cancer, an incidence associated with their inhalation of the radioactive progeny of radon. On the basis of studies of these miners, it has been suggested that exposure of the general population to the estimated mean indoor radon concentration of 1 picocurie per liter (pCi L^{-1}) (37 Bq m^{-3}) could account for 1000 to 20,000 lung cancers per year in the United States.[1] Because indoor radon concentrations vary by three orders of magnitude or more, a small fraction of the public is exposed to very high levels of radon progeny indoors: some exposures may even exceed the limit currently permitted for underground miners. For some individuals, the lifetime risk of lung cancer from indoor exposure to radon progeny may thus be as high as 10%.

Although ventilation rates affect indoor radon concentrations, the dominant cause of the wide range of concentrations is variability in the rate of entry from the sources, primarily the soil and rock under and near buildings.[6–13] Building materials, unless they contain unusually high concentrations of radium, contribute much less than 1 pCi L^{-1} to the indoor concentration. In some cases, domestic water from wells, especially those drilled in granitic areas, can account for high transport rates of radon from soil and rock into the house. Usually, however, radon from soil enters via pressure-driven flow through penetrations in the building substructure.

Our research efforts are directed toward providing a basis for (1) systematic identification of geo-

graphic areas where high indoor radon levels are endemic and (2) the identification or development of cost-effective construction techniques and control measures for preventing or reducing excessive entry rates. During the past year, our work has focused on three projects, two on the entry of radon into residences, and one on the use of aerial radiometric data for determining the geographic distribution of radon source potential. We have also continued some work on measurement of indoor radon concentrations and ventilation rates in the existing housing stock.[14] Studies of the behavior of radon decay products are described in a following section, "Particles and Radon Progeny."

Accomplishments During FY 1983

Radon Transport into a Detached One-Story House with a Basement[15]

During a 5-month period from late winter through early summer 1982, radon concentration and ventilation rate, as well as meteorological and radon-source parameters, were monitored continuously in a house near Chicago (which was known to have a higher-than-average indoor radon concentration), using an automated instrumentation system based on one previously described.[16] The mean radon concentration and mean ventilation rate were 2.6 pCi L^{-1} (96 Bq m^{-3}) and 0.22 h^{-1}, respectively; however, they both varied over a wide range. Radon activity measured at the cover of the basement sump varied bimodally (perhaps due to variations in the water level in the peripheral drain-tile system), and corresponding changes in indoor concentration were observed: in the higher mode, which persisted for 22% of the time, the sump activity ranged between 300 and 700 pCi L^{-1} (11,000–26,000 Bq m^{-3}), and the mean indoor radon concentration was 6.5 pCi L^{-1} (240 Bq m^{-3}); in the lower mode, the sump activity was less than 10 pCi L^{-1} (370 Bq m^{-3}), and the mean indoor radon concentration was 1.5 pCi L^{-1} (56 Bq m^{-3}).

As shown in Fig. 7, the indoor radon concentration in either mode shows little dependence on ventilation rate. The points were obtained as follows: Three-hour averages corresponding to periods of high and low sump activity were sorted according to ventilation rate. The data were then grouped into bins of 19–21 measurements. For each bin, the geometric means (GM) of the radon concentration and of the air-exchange rate were plotted; for some bins, error bars indicate the geometric standard error of the radon concentration. Two models of radon entry, as represented by the curves in Fig. 7, were then fit to the data.

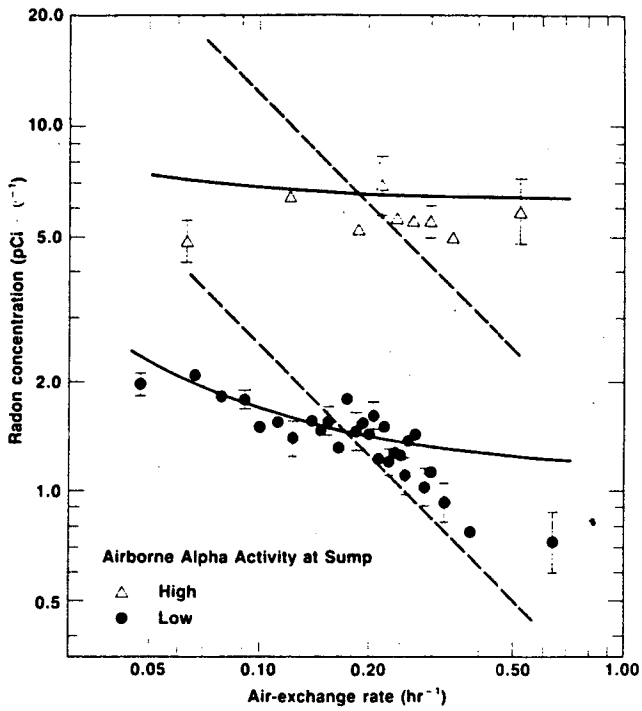

Figure 7. Scatter plot of radon concentration (in pCi/L) vs. air-exchange (ventilation) rate averaged over 3-hour periods. The triangles and circles represent periods of high and low airborne alpha activity at the sump, respectively. Each point reflects the geometric mean of 19–21 measurements, sorted according to ventilation rate. Error bars represent one geometric standard deviation of the mean; for clarity, only some are given. Dashed and solid lines reflect two models of radon entry, as discussed in the text. (XBL 836-2680)

In the first model (dashed lines), the radon entry rate is assumed to be independent of the ventilation rate so that the radon concentration and ventilation rate vary in inverse proportion. (The outdoor radon concentration, being much smaller, is ignored.) This model applies when radon entry is primarily via molecular diffusion or tap water. In the second model (solid lines), the entry rate has two components: a diffusive component that is independent of ventilation and a flow component that is proportional to the ventilation rate. The flow component models the entry of air from the soil; such air bears a high concentration of radon and is driven by the same mechanisms—temperature differences and wind—that cause infiltration. With the exception of a few points having high ventilation rates for periods of low sump activity, the second model fits the data far better than does the first.

For low sump activity, a least-squares fit to the 3-hour data determined that the flow and diffusive components were 1.15 pCi L^{-1} (43 Bq m^{-3}) and

0.06 pCi L^{-1} h^{-1} (2 Bq m^{-3} h^{-1}). For the high sump activity data, the diffusive component was assumed to be the same and the flow component was determined by a fit to be 6.3 pCi L^{-1} (230 Bq m^{-3}). On average, for low sump activity periods, the diffusive component accounts for only 20% of the radon entry rate, while for high sump activity periods, it accounts for only 3% of the total entry rate.

The diffusive component agrees well with the expected contributions from concrete and soil. To account for the flow component, a pressure difference of about 3.5 Pa must be able to drive a flow of 0.5 L/s through the soil and into the house. This appears possible if the soil permeability is relatively high. Alternatively, a comparable flow of air along the outside of the basement wall may sweep radon that has diffused through the soil into the house.

Radon Entry in Houses Having a Crawl Space[17]

Many single-family houses in the United States, particularly in areas with temperate climates, have a wooden floor over a vented crawl space. Radon, entering the crawl space from the soil, will pass either through the vents or through penetrations in the floor; in the latter case, it enters the living space. The fraction transmitted into the house depends not only on the size and location of vents and floor penetrations, but also on meteorological conditions: a large indoor-outdoor temperature difference will increase the convective flow of air into the living space through the floor, while high wind speeds will increase the cross-ventilation of the crawl space through the vents.

As part of a study investigating the transport of radon from soil into crawl spaces and its subsequent transmission into living spaces, detailed monitoring in two crawl-space houses was carried out in a manner similar to that in the basement house, except that ventilation rates were determined from an infiltration model rather than from tracer-gas decay measurements. The results from one house, located in Portland, Oregon, are displayed in Fig. 8. During the first 2 weeks of monitoring, the crawl-space vents were sealed with polyethylene sheets and tape. The average indoor radon concentration during this period was 2.2 pCi L^{-1} (81 Bq m^{-3}). This was significantly higher than the 1.2 pCi L^{-1} (44 Bq m^{-3}) average during the following 3 weeks when the vents were open, indicating that cross-ventilation of the crawl space is reduced when the vents are sealed.

Considerable evidence suggests that, for this house, radon transport from the soil into the crawl space does not result entirely from molecular diffusion, and that the presence of the house influences the spatial distribution of radon flux from the soil beneath it and nearby. Consider, for example, the results of March 29 and 30 (Fig. 8). Here, coincident with a day of heavy rainfall and a modest drop in barometric pressure, the radon concentrations in the living space and in the crawl space rise to the highest levels of the entire monitoring period; the radon entry rate is several times greater than can be accounted for by the diffusion of radon from the soil under the house. Apparently, the heavy rainfall reduces the permeability of the soil surrounding the house, so that the flow of air carrying a high concentration of radon out of the soil in response to the drop in barometric pressure or to the gravity-drawn flow of rainwater through the soil is focused into the crawl space.

In another house, a tracer gas was used to study air-flow rates from the crawl space into the living

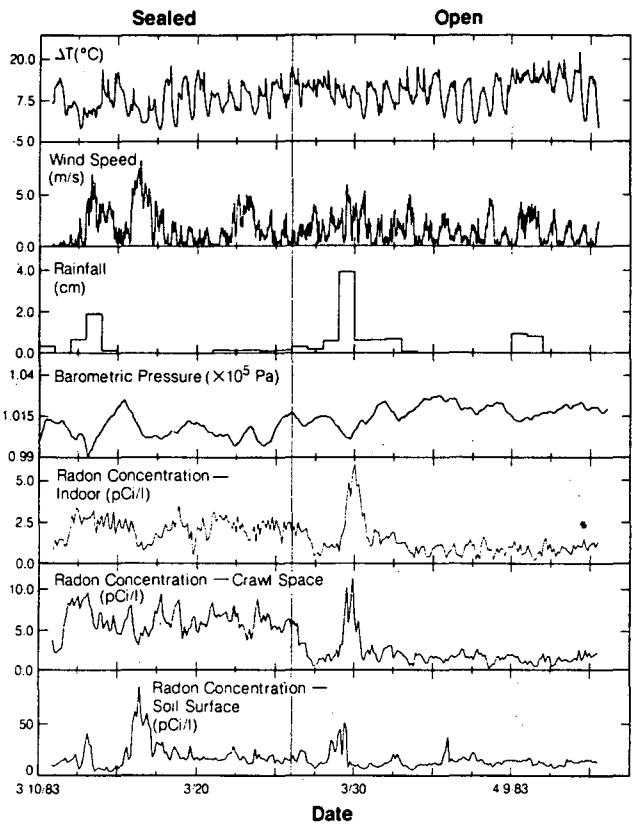

Figure 8. Measurements made over a 5-week period at a house in Portland, Oregon. Crawl-space vents were sealed during the first 2 weeks and open during the final 3. A positive temperature difference indicates the temperature is higher indoors than outside. Radon concentration at the soil surface was measured by sampling air from an accumulator that covered 0.95 m^2 of soil at a rate of 15 cm^3 s^{-1}. (XBL 837-457)

space. Over a range of vent and floor-leakage areas, 30% to 65% of the tracer gas released in the crawl space appeared in the living space. (The mean wind speed and indoor-outdoor temperature difference were 2.3 m s^{-1} and 9.1°C, respectively.) A two-person-day "house-doctoring" effort against floor leakage led to a 15% reduction in the leakage area of the entire house, but no increase in the indoor concentration of the tracer gas was subsequently observed. Thus, contrary to the common idea that lower ventilation rates lead to higher indoor pollutant concentrations, weatherization measures directed at tightening the floor may reduce infiltration in houses with crawl spaces without increasing indoor radon concentrations.

Radon Source Assessment Based on Aerial Radiometric Data[18]

From 1974 to 1981, the U.S. Department of Energy funded the National Uranium Resource Evaluation Program to improve estimates of potential uranium resources and uranium reserves. A major part of the program was the aerial measurement of gamma radiation from the earth's surface. One of the three radioisotopes measured directly in this manner was bismuth-214, a decay product of radium-226, the progenitor of radon. By itself, the radium content of the soil is not sufficient to determine radon entry rates, but it can, in principle, be used to locate some areas with high potential for elevated radon entry. In addition, by combining data on radium content with information relating to transport properties of air in soil, it may be possible to characterize, on a geographic basis, the potential entry rate of radon from the soil into buildings.

During the past year, the aerial data for the Pacific Northwest (Washington, Oregon, Idaho, and parts of Montana, Nevada, and Utah) and for California have been studied. Equivalent uranium (or radium) concentrations in soil have been characterized by calculating the mean and standard deviation for each of 92 quadrangles. Each quadrangle is delineated by one degree of latitude (approximately 72 miles north to south) and two degrees of longitude (120–140 miles east to west). Figure 9 shows the spatial distribution of these mean equivalent uranium concentrations. Generally, for the Pacific Northwest, radium concentrations in near-surface soil increase from west to east. In California, the San Francisco Bay Area and regions to the south appear to have higher radium concentrations than regions to the north of the Bay Area.

In two areas near Spokane, Washington, for which the aerial data exhibited great spatial variability, field measurements were also performed. Along one flight-line segment of 1.5 km, measurements at 20 sites were made of bismuth-214 gamma activity, radon flux from the soil surface, and emanating and total radium-226 content in the soil. Good agreement was seen between the aerial and ground-based determinations of bismuth, with somewhat greater spatial variation observed on the ground because measurements there integrate over a much smaller area of soil. The other parameters varied in rough correspondence to the ground-based bismuth measurements.

Planned Activities for FY 1984

Our experimental investigations of radon entry into residences will be expanded with a study of a house having a high indoor radon concentration. In this case, the objectives are to account for the source materials and transport processes, and to test the effectiveness of inexpensive control measures. Examination of the aerial radiometric data will continue; the local variability, which affects the extent to which the data should be aggregated to identify geographic areas with high source potential, will be investigated in more detail. In addition, strategies for using the data set covering the entire United States will be examined.

A third research direction that supports these two major efforts is a study of soil characteristics: some effort will be devoted to developing and applying techniques for measuring radon production and transport properties in soil.

PARTICLES AND RADON PROGENY

Sources of particles in the indoor environment include combustion processes such as tobacco smoke, unvented gas ranges or heaters, kerosene heaters, and wood stoves and fireplaces. The health effects of inhaling such particles depend on the physical and chemical characteristics of the particles and on the regions of the lung where deposition occurs. Particles less than 3 microns in diameter are likely to be deposited in the lower lung—the tracheobronchial and pulmonary regions.[19] Inhalation of fine particles may also have long-term health effects because these particles transport adsorbed toxic materials such as benzo(a)pyrene or radon progeny.

Alpha decay of radon gas produces radioactive isotopes of polonium, bismuth, and lead. These decay products (progeny) can be deposited in the lung, where subsequent alpha decays can cause lung cancer. The decay products may be deposited as *unattached* atoms or small molecules, or they may be

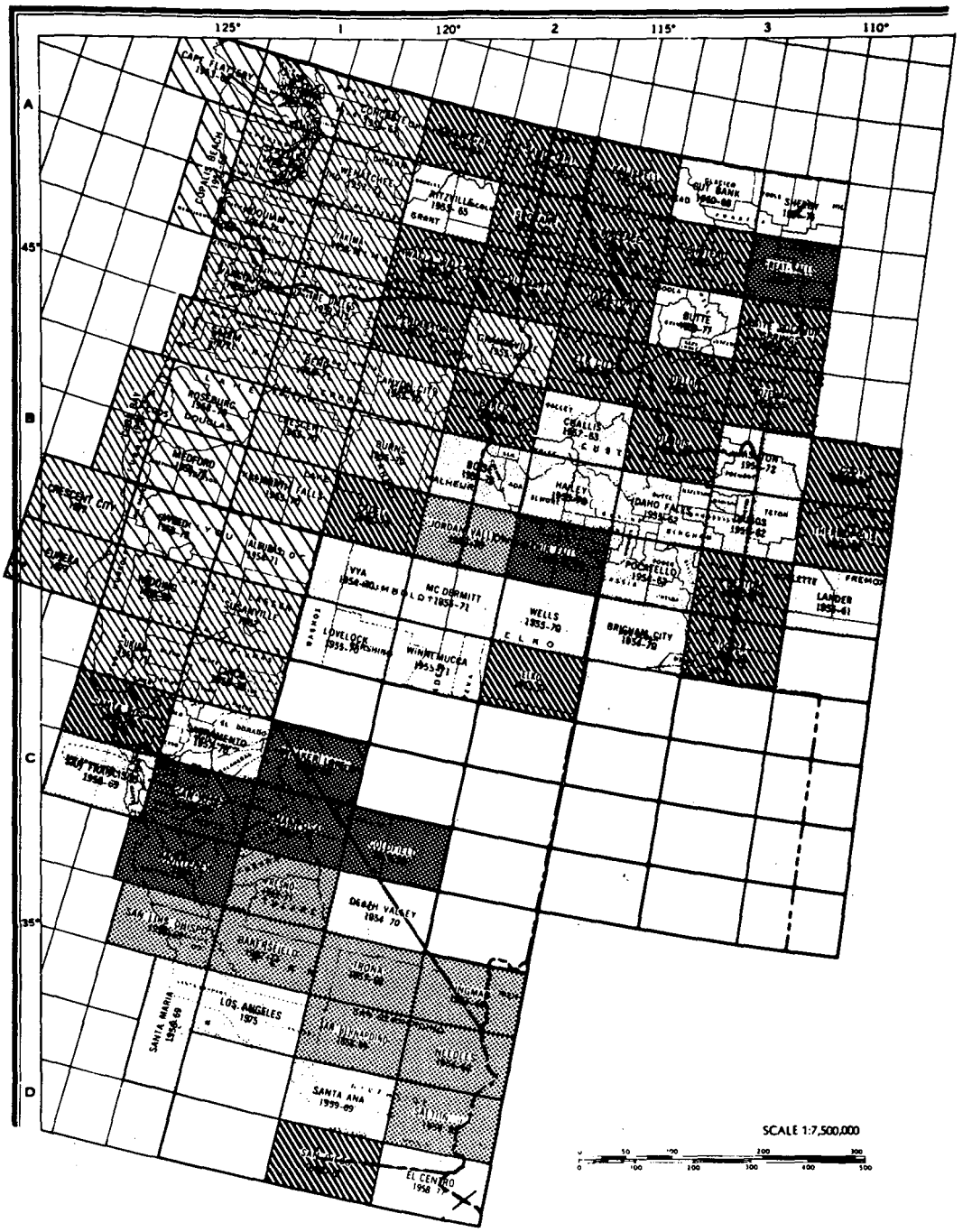

Figure 9. Mean near-surface radium-226 concentrations by quadrangle for the Pacific Northwest and California. The average for the entire region is 0.73 pCi g^{-1} (27 Bq kg^{-1}).

(XBL 841-9469C)

deposited *attached* to airborne particles. Estimates of the health effects of the decay products indicate that the alpha decay of unattached progeny produces a larger lung dose than does the decay of attached progeny.[20]

Our present research efforts have several objectives:

(1) to characterize the particle size and concentration profiles of aerosols from typical indoor sources;

(2) to study the interactions among particles and other chemical species, in this case, radon decay progeny;

(3) to investigate particle removal processes and the concomitant effects on radon progeny concentrations; and

(4) to develop an empirically based model for the behavior of radon progeny.

Accomplishments During FY 1983

Control of Particles and Radon Progeny[21,22]

There is growing interest in air cleaning for removal of airborne particles. Air cleaning devices, such as filters, electrostatic precipitators, or negative-ion generators, are available as stand-alone units or as part of a forced-air (ducted) furnace system. Experiments using both ducted and unducted control devices were performed at the Indoor Air Quality Research House to evaluate the effectiveness of different types and models of air cleaners and to investigate the effect of particle removal on the behavior of radon progeny. Results using the unducted devices are reported here.

The instrumentation employed in these experiments comprises a highly automated data acquisition, monitoring, and control system installed in the Research House. This system provides real-time data on particle sizes and concentrations and on radon and radon progeny concentrations, and allows programmable control over operation of the experiment. A more detailed description is provided below in the "Instrumentation" section.

Particles were generated by a cigarette-smoking machine. A typical 6-minute smoking cycle burned ~600 mg of tobacco and produced a peak concentration of up to 2×10^5 particles/cc. Radon was injected by passing air through a volume containing 200 μCi of ^{226}Ra stearate, resulting in initial radon concentrations of 150 to 400 pCi/liter. After the initial injection of pollutants, a 4-hour mixing and decay period established radioactive equilibrium among the radon and radon progeny and a constant decay rate for the particles. The control device was then turned on, usually for 3 to 5 hours.

Control device performance was measured by the difference in the rate of decay of particle concentrations (both total and by size fraction), with and without the control device operating. This net decay rate, multiplied by the volume of the test space, yielded an Effective Cleaning Rate (ECR). Dividing the ECR by the measured volume flow rate through the air cleaner provides a measure of overall system efficiency in removing particles. The results of these experiments are summarized in Fig. 10. As can be seen, the most effective devices were the extended surface filters and the electrostatic precipitators. Least effective were the panel filters and one of the negative-ion generators. A second negative-ion generator that operated at a higher voltage was moderately effective. In addition, two oscillating fans were operated as a test of the effect of rapid air circulation; no measurable reduction in particle concentrations was observed.

The effects of particle removal on radon progeny concentrations were determined by measurements of the concentrations before and at the end of device operation, when radioactive equilibrium was established. The particle concentrations and equilibrium factor, F, are tabulated in Table 3 for each pair of before-and-after measurements. F is the ratio of the potential alpha energy concentration (multiplied by 100) to the radon concentration. The table also lists the total removal rate Λ, measured for the first progeny, ^{218}Po. This rate consists of all removal

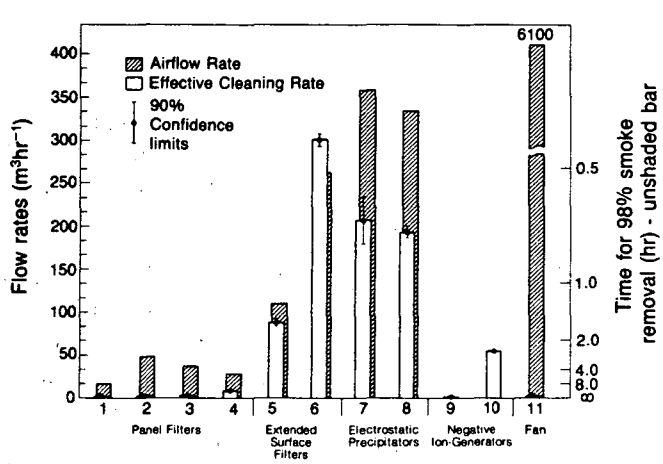

Figure 10. Performance measurements of unducted particle control devices. Measured air-flow rates through the device are shown by shaded bars, while unshaded bars give the effective air cleaning rate in cubic feet per minute (cfm). Superposition of the bars gives an indication of overall device efficiency. (XBL 8310-3343A)

Table 3. Effect of air cleaner operation on radon progeny equilibrium factors.

Air cleaner	Before operation		After operation		
	Particle concentration[a]	F[b]	Particle concentration[a]	F[b]	Λ_1[c]
Panel filters					
1	32,000	0.83	18,000	0.83	2.1
2	31,000	0.92	8,100	0.77	3.1
3	38,000	0.87	7,800	0.65	3.8
4	31,000	0.88	13,000	0.76	3.1
Extended surface filters					
5	28,000	0.81	120	0.068	25.
6	60,000	0.92	17	0.066	23.
Electrostatic precipitators					
7	30,000	0.86	<500	0.07	23.
8	24,000	0.82	230	0.076	20.
Negative-ion generator					
10	31,000	0.88	60	0.103	14.4
Fan					
11	33,000	0.81	15,000	0.58	6.1

[a]Concentration in particles/cm^3

[b]Ratio of potential alpha energy concentration ($\times$100) to radon concentration.

[c]Total removal rate for ^{218}Po, the first progeny.

mechanisms, including deposition of radon progeny on walls or other room surfaces. Based on these total removal rates, the deposition rates for unattached and attached ^{218}Po can be estimated as a function of particle concentration. Similar estimates for the succeeding radon progeny, ^{214}Pb and ^{214}Bi, can also be made, and the results are illustrated in Fig. 11. At typical indoor particle concentrations (5000 to 10,000 particles/cm^3) progeny deposition rates range from 1 to 6 hr^{-1}; as particle concentrations decrease, the deposition rates increase rapidly, approaching 15 hr^{-1}. Thus, while operation of an air cleaner can remove radon progeny by direct filtration, a more substantial effect is produced by reducing the particle concentration so that deposition of progeny on surfaces is the predominant removal mechanism.

Instrumentation

Instrumentation for the measurement of particle concentrations and size distributions has been installed at the Indoor Air Quality Research House. Total aerosol concentration is measured with a condensation nucleus counter (CNC). The CNC also samples the output concentration of an electrostatic classifier (EC), which provides particle size and concentration data for particles between 0.01 and 0.3 μm in diameter. Information on larger particles is provided by an optical particle counter (OPC), which has been adapted to measure the size and concentration of particles between 0.1 and 3 μm diameter.

Instrumentation control and data logging are done by a microcomputer system designed and assembled at LBL. This computer also controls the

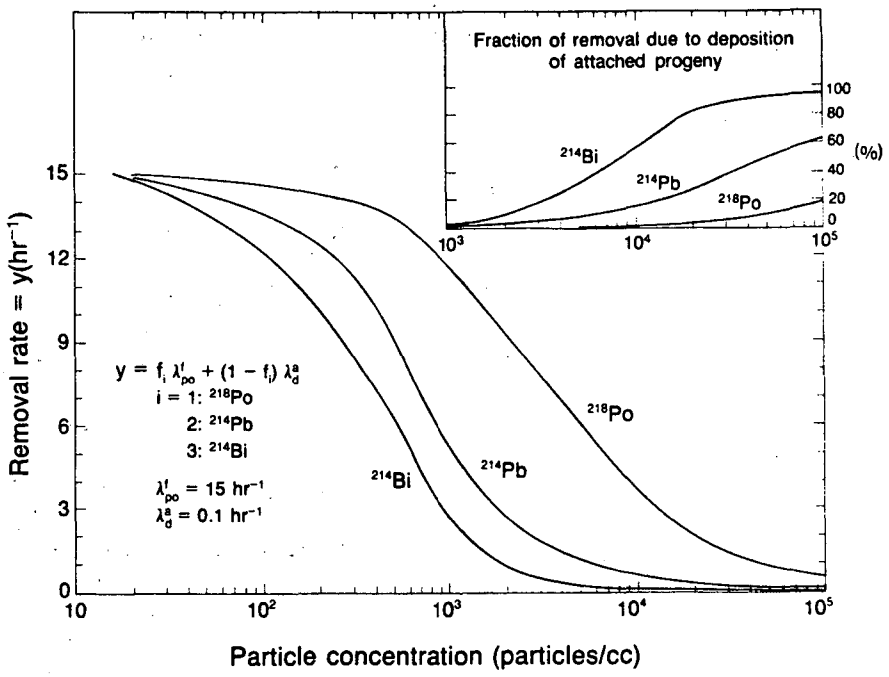

Figure 11. Removal of radon decay products due to plateout of unattached progeny and deposition of progeny attached to the environmental aerosol, as a function of particle concentration. Inset shows the fraction of removal rate due to deposition only. At particle concentrations below 1000 particles/cc, plateout of unattached progeny is dominant.

(XBL 8312-6708A)

sampling protocol and the timing for each sampling point. Data from the CNC and EC are stored in the computer, then written on a cartridge tape at the end of the sampling sequence. Data from an internal buffer in the OPC are written directly to tape after each sampling sequence.

An automated instrument for measuring indoor concentrations of radon progeny has been designed and built at LBL.[23] This instrument—the radon daughter carousel (RDC)—is microprocessor-controlled, using a filter sampler and a solid-state alpha particle detector to collect radon progeny and to subsequently analyze their alpha decays. Alpha spectroscopic data are stored internally, then transmitted at the end of each measurement sequence to the research house computer for storage on magnetic tape. Following data transmission, progeny collection on a new filter surface begins (total sampling time is typically 5 minutes). Total measurement times can be selected to optimize the operation of the device to the required precision. Concentrations as low as 0.5 pCi/L can be measured for individual progeny with a measurement time of 45 minutes.

A device to measure concentrations of unattached progeny has been constructed, using a 100- to 250-mesh wire screen backed by a filter sampler. Alpha decays from progeny collected on the screen and the backup filter are analyzed using a scintillator and photomultiplier package fabricated at LBL. This sampler is currently being checked out and will be used in future experiments on radon progeny behavior.

Modeling[24]

In order to examine the effects of air motion on the behavior of radon decay products in the indoor environment, a numerical model has been developed to simulate the generation, convective and diffusive transport, and removal mechanisms of radon and radon progeny. A number of basic flow configurations have been investigated, including flow over a flat plate, stagnation flow, and free convective flow on a vertical surface. The model is currently being used to examine the mechanisms by which radon progeny, both attached and unattached, are deposited on surfaces.

Planned Activities for FY 1984

Further evaluation of particle control and the resulting effect on radon progeny concentrations will

be performed. Of particular interest are measurements of unattached progeny as a function of particle concentration. Also of interest is the direct measurement of radon progeny plateout. Detailed characterization of particle concentrations and size distributions from a variety of combustion sources are also planned, with the possibility of using these results for indoor source apportionment.

Additional development of a model for progeny behavior will be undertaken, to include turbulent flow conditions. Since the model provides a theoretical framework for considering the general behavior and interactions of indoor pollutants, its extension to other indoor pollutants will be initiated.

PASSIVE SAMPLERS

Field studies of indoor air quality and the effects of reduced ventilation have been limited by the high cost of employing continuous monitors. Passive samplers capable of measuring daily or weekly integrated contaminant concentrations on site, without external power, accessories such as pumps and chart recorders, or the attendance of trained technicians offer a promising solution. Large-scale audits could be conducted with these samplers to insure that energy-conservation programs are implemented without an adverse effect upon occupant health and safety. In addition, passive samplers would make possible large-scale studies of the indoor air quality of existing buildings and facilitate epidemiological studies, which have typically had to rely upon exposures derived from distant, fixed-site monitors. With lightweight, inexpensive, and unobtrusive passive samplers, personal monitoring would be possible and a major uncertainty in epidemiological studies would be removed. The development and validation of passive samplers to meet these needs is the focus of this project.

Accomplishments During FY 1983

Nitrogen Dioxide Passive Sampler

The nitrogen dioxide (NO_2) passive sampler,[25] which was developed for industrial hygiene applications, is the most widely used sampler in studies of indoor air quality. Yet even this sampler has been used under conditions for which it has been inadequately tested. Because diffusion theory predicts only a 1.7% change in the sampling rate with a 10°C change in temperature at 21°C, temperature corrections have generally been ignored. However, the NO_2 absorbent employed in the sampler, triethanolamine, has a liquid-solid phase transition at 21°C.[26] Because this phase change occurs at a typical ambient temperature, the effect of temperature upon the performance of the sampler was investigated.[27]

In this study, the NO_2 concentration, exposure time, and face velocity were held constant while the temperature of the sampled air was varied from 7°C to 38°C. The collection efficiency of the sampler was found to decrease by 15% with a temperature decrease from 27°C to 15°C.

During this study, it became evident that face velocity effects produced by the low flow conditions in our exposure system could also be significant. Therefore, we conducted a preliminary study on the effect of low face velocity on the NO_2 passive sampler. The results supported a previously published theoretical treatment of sampler starvation.[28] After a correction for sampler starvation caused by low face velocity was applied, we obtained the temperature dependence for the NO_2 passive sampler illustrated in Fig. 12.

Carbon Monoxide Passive Sampler

We have continued the laboratory development of a carbon monoxide (CO) passive sampler based on a diffusion tube and a class of compounds sensitive to CO. These compounds, all containing palladium, are reduced upon exposure to CO, producing metallic palladium. The analytical technique employed is based upon detecting the amount of palladium present after exposure. Because the palladium compounds require water vapor as a reactant or catalyst, a major difficulty to overcome was the stabilization of the samplers' response to CO atmospheres over a wide range of humidities. Figure 13

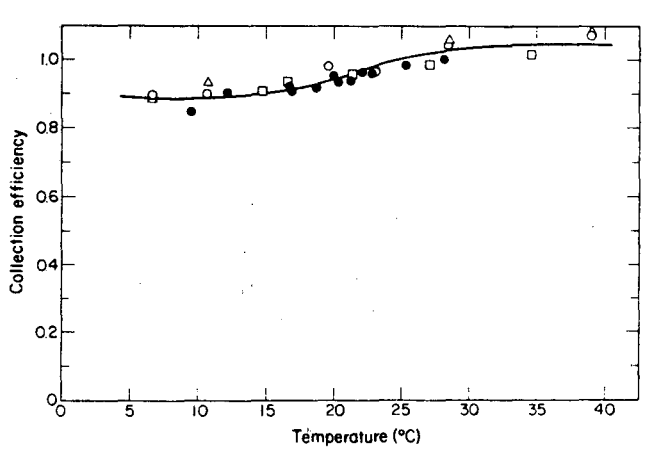

Figure 12. Collection efficiencies of the Palmes NO_2 passive sampler vs. temperature after correction for face velocity effects. Efficiencies are also corrected for the theoretical temperature dependence (see text). Symbols correspond to different sampler lots. (XBL 834-1570)

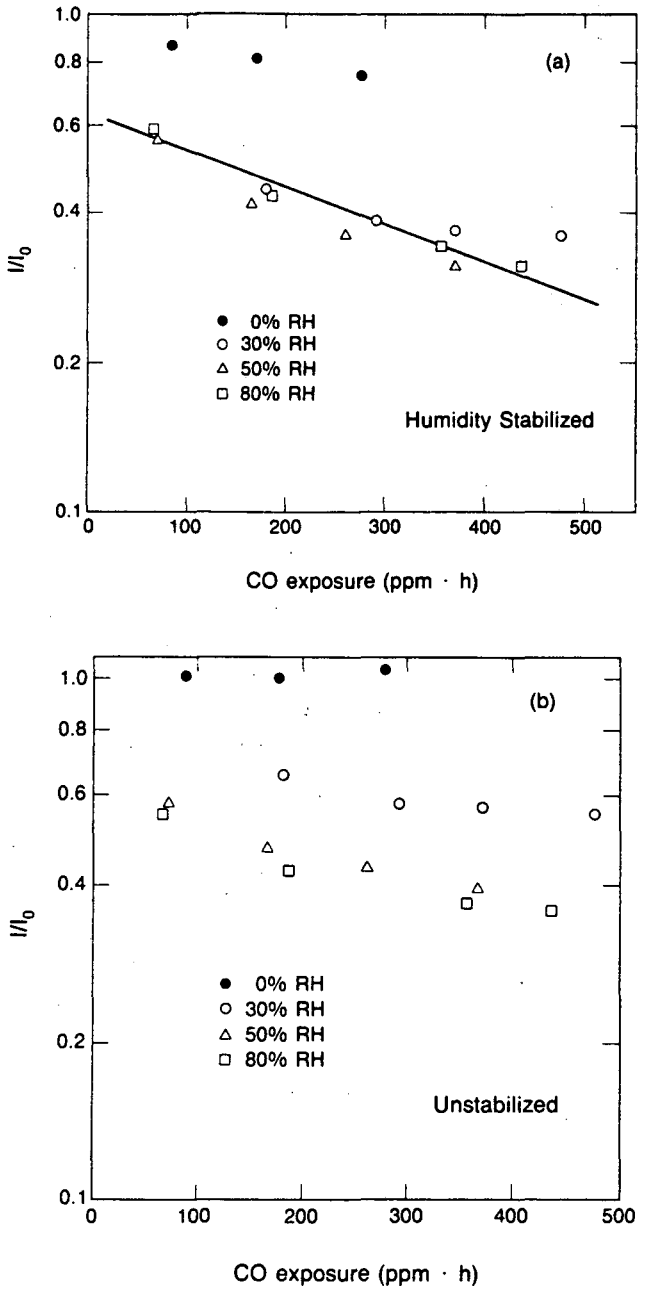

Figure 13. Comparison of the responses of carbon monoxide passive samplers (a) with and (b) without humidity stabilization; test atmospheres were at indicated relative humidities (RH) and near 5 ppm carbon monoxide.
[(a) XBL 8312-7402A; (b) XBL 8312-7401A]

depicts representative results for samplers with and without this stabilization. The samplers in both cases were exposed to controlled atmospheres near 5 ppm of CO for different lengths of time. Four lots of samplers were exposed to atmospheres differing only in relative humidity. The response of the stabilized samplers is uniform for all atmospheres with relative humidities of 30% or greater. While linearity within a lot of stabilized samplers is good, variation with respect to lot is evident and must be improved.

Water Vapor Passive Sampler

We have improved the design and performance and nearly completed the field validation of a water vapor passive sampler first described by Palmes.[25] This sampler uses molecular sieve contained in the closed end of a diffusion tube to trap water vapor. After exposure, the weight gain of the sampler and the duration of sampling can be related to the time-weighted average humidity of the sampled atmosphere. The sampler is shown schematically in Fig. 14. Improvements include the addition of cloth backing to prevent leakage of molecular sieve during handling and sampling and the substitution of aluminum diffusion tubes for the acrylic tubes initially used. (Acrylic tubes adsorbed water nonreproducibly, so that stable blanks and adequate shelf life weren't obtained.) As illustrated in Fig. 15, which compares the results of 1- to 9-day exposures to relative humidities near 50% at room temperature with the theoretically expected results, linearity and accuracy are excellent. Average precision is better than 3%. Tests have demonstrated that sensitivity is adequate for sampling times as low as 1 day, yet capacity is still sufficient for over 7 days of sampling near 100% RH.

Planned Activities for FY 1984

The field validation of the water vapor passive sampler will be completed in the early part of the year, as will tests of pre- and post-exposure storage

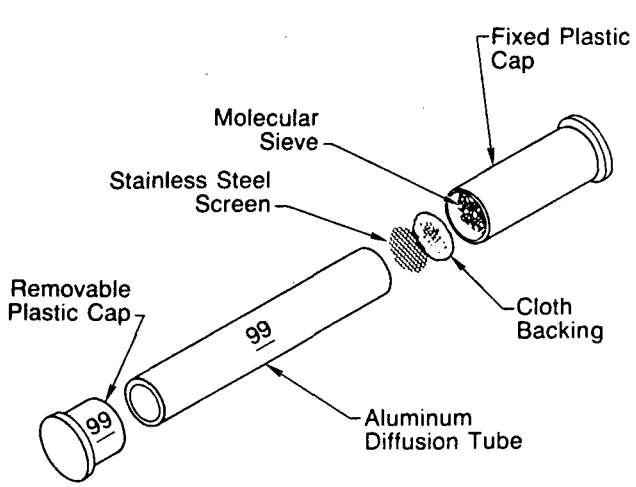

Figure 14. Exploded schematic of the water vapor passive sampler. Note that cap and tube are numbered to maintain a matched set. (XBL 8312-7408)

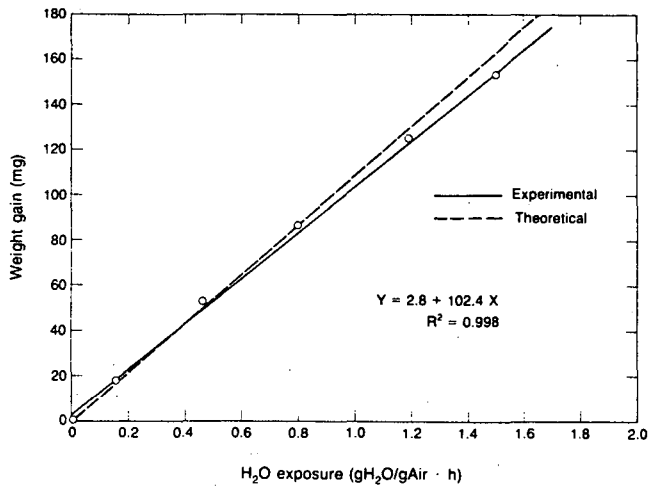

Figure 15. Comparison of experimental and theoretical weight gain of water vapor passive sampler after 1, 3, 5, 7, and 9 days of exposure to an atmosphere near 50% relative humidity. (XBL 8312-7400)

stability. We will begin the field validation of the carbon monoxide passive sampler, comparing its performance with that of a continuous analyzer in a variety of residential environments. The remaining laboratory tests of this sampler will also be completed with the goal of improving precision, sensitivity, and range.

A major uncertainty in the use of passive samplers is the air speed at the sampler face in various environments. While the air speed at which starvation occurs for a given sampler can be calculated from theory, no instrument at present can measure the low air speeds typical of non-industrial settings. To address this uncertainty, we will begin the development of a low-air-speed sampler to determine appropriate conditions for the deployment of passive samplers. Air speeds as determined by such a sampler would also be useful in validating models of pollutant transport.

INDOOR AIR QUALITY CONTROL TECHNIQUES

Control of the indoor concentrations of indoor-generated air pollutants can be accomplished by exclusion or suppression of pollutant sources, by air cleaning, and by ventilation. Residential mechanical ventilation with heat recovery in air-to-air heat exchangers has been the focus of a major effort by the Control Techniques Project. In earlier work, the energy performance, impact on indoor air quality, ventilation efficiency, and economics of this technique were investigated in laboratory and field studies and with computer models. During FY 1983, an experimental study of the freezing problem in residential air-to-air heat exchangers was completed. In addition, a method of removing formaldehyde from indoor air by absorption in water was tested, and the use of tracer gases to evaluate the performance of ventilation systems in removing pollutants was investigated.

Accomplishments During FY 1983

Freezing in Air-to-Air Heat Exchangers

An increasingly common and energy-efficient technique of ventilating residences is to mechanically supply and exhaust approximately equal amounts of air to and from the residence and to transfer heat between the incoming and outgoing airstreams in an air-to-air heat exchanger. Because of the transfer of heat, the incoming air is preheated or precooled before entering the residence. Thus, the energy loads imposed on the heating or cooling systems are less than for ventilation without heat recovery. Previous studies have shown that this technique of mechanical ventilation is most economically attractive in cold climates. However, when it is sufficiently cold outdoors, moisture in the exhaust airstream can freeze within the core of the heat exchanger, and the accumulated ice or frost can substantially reduce the flow rate of the exhaust airstream and the amount of heat recovery. A laboratory investigation[29] was therefore conducted with the following objectives: (1) to determine the environmental conditions (indoor and outdoor) that lead to freezing; (2) to determine the impact of freezing on heat exchanger performance; and (3) to evaluate a freeze protection strategy based on periodic defrosts.

Figure 16 shows some of the results of these experiments. The inlet temperature of the cold airstream, at which freezing was initiated in both a cross-flow and a counterflow sensible heat exchanger, is plotted versus the inlet relative humidity of the warm airstream. (A sensible heat exchanger transfers only heat between airstreams, while an enthalpy exchanger transfers both heat and moisture.) In our tests of the sensible heat exchangers and an enthalpy cross-flow exchanger (data for the latter are not shown), the temperature at the onset of freezing ranged from −3° to −12°C. The results for the counterflow exchanger are in fair agreement with our predictions based on a simple theoretical model and with observations of freezing in earlier field studies. Experimental studies with a sensible counterflow heat exchanger at the University of Missouri-Rolla[30]

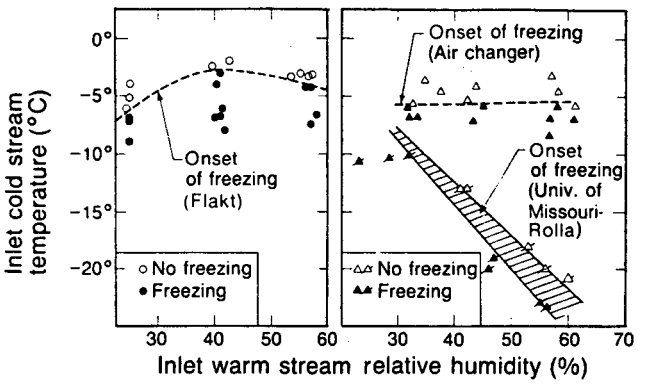

Figure 16. Results of tests to determine the onset of freezing for the cross-flow (Flakt) and counterflow (air changer) heat exchangers and previous results from tests of a counterflow heat exchanger at the University of Missouri at Rolla. (XBL 839-3294)

(Fig. 16) and data tabulated by the American Society of Heating, Refrigerating, and Air Conditioning Engineers,[31] however, indicate that freezing is initiated at significantly lower temperatures. We suspect that the major reasons for the discrepancies are differences between the heat exchangers used in the experiments and the different criteria used for the onset of freezing. For future studies, we have proposed new criteria for the onset of freezing that are based on a decrease in heat recovery performance.

The experimental studies of freezing included an evaluation of heat exchanger performance when periods with freezing were periodically interrupted by a defrost. The defrosts were accomplished by stopping the flow of cold air through the exchanger long enough to melt the accumulated ice or frost. The tests were conducted with two sensible exchangers (one cross-flow, the other counterflow), a variety of airstream humidities and temperatures, and a variable length of time between defrosts. Two parameters calculated from the data are based on the temperature efficiency E_T of the exchangers; this is a measure of the heat recovery performance and is defined by the equation:

$$E_T = (T_{s,o} - T_{s,i})/(T_{e,i} - T_{s,i}) \qquad (1)$$

Where T is an airstream temperature; subscripts s and e refer to the supply and exhaust airstreams, while subscripts i and o refer to flow into and out of the heat exchanger. In tests of a cross-flow exchanger, freezing caused the temperature efficiency to decrease at a rate from 1.5 to 13.2 percentage points per hour (Fig. 17). More rapid decreases in efficiency were associated with lower cold-airstream temperatures, higher warm-airstream humidities, and shorter periods of freezing. Temperature efficiency decreased at a slower rate (0.6 to 2.0%/h) in tests of a counterflow exchanger. It appeared that the performance of the counterflow exchanger decreased at a slower rate because the core contained larger air flow passages that are not easily obstructed by ice and because much of the water that condensed in the core drained to its warm end, where it did not freeze.

The "defrost time fraction"—the amount of time required to defrost these heat exchangers divided by the total operating time—ranged from 0.06 to 0.26 and did not differ greatly between the cross-flow and counterflow exchangers when corresponding tests were compared. This ratio was smaller after long periods of freezing, which indicates that the defrosting periods were used more efficiently after significant ice or frost had accumulated. The optimal amount of time between defrosts appeared to be in the range of 1 to 6 hours for the cross-flow exchanger and 6 to 13 hours for the counterflow exchanger. When the frequency of defrost was in the optimal range, the average temperature efficiency was approximately 62% for the cross-flow exchanger and 75% for the counterflow exchanger. It was estimated that the freezing and periodic defrosts reduced the time-averaged temperature efficiency of the cross-flow and counterflow exchangers by 15% and 10%, respectively, under nearly optimal operating conditions. Larger reductions in time-averaged performance are likely with currently available freeze protection systems. Heat exchanger designs that minimize the effects of freezing and improved controls for freeze-protection systems were recommended.

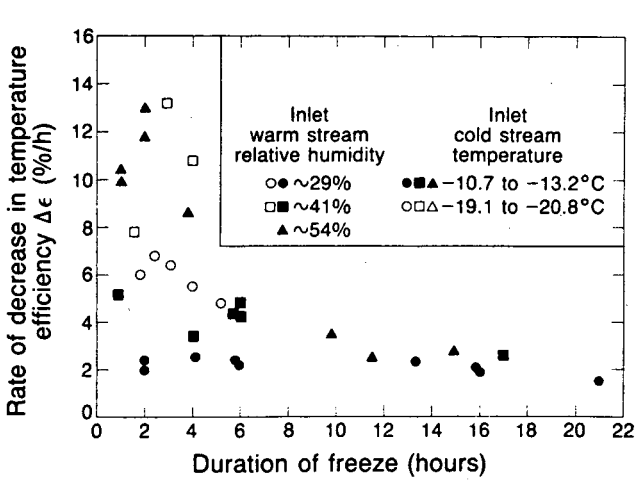

Figure 17. Rate of decrease in temperature efficiency of cross-flow (Flakt) heat exchanger versus duration of the freeze. (XBL 839-3288)

Air Washing for Formaldehyde Control

A potential control technique for indoor formaldehyde that had not been evaluated previously is removal by absorption in water; this is also referred to as air washing. Formaldehyde reacts rapidly with liquid water to form methylene glycol and polyoxymethylene glycol polymers; its effective solubility in water is therefore high. Air washing is accomplished by forcing air through an air-water contact system. To prevent saturation of the washing water with formaldehyde, a small amount of the water is continuously replaced with fresh water. Humidification of the air can be prevented by using a refrigeration system to cool the air or the water prior to the air-water contact. In some situations, the air washer can simultaneously remove formaldehyde and provide dehumidification. Energy consumed by the refrigeration system can be delivered to indoors, thus reducing the heating load, or can be rejected to outdoors, so that the air washer acts as an air conditioner.

Theoretical calculations indicated the feasibility of air washing for indoor formaldehyde control. Two air washers were therefore fabricated and tested in the laboratory under controlled conditions.[32] A semi-theoretical model of air washer performance was developed and used to correct experimental data from tests that were not conducted under steady-state conditions (i.e., when there was a net storage of formaldehyde within the water in the air washer). The model was also used to illustrate the relationship between various operational parameters and air washer performance.

Tests were first conducted with wetted porous mats as the air-water contact system. The mats were maintained wet by rotation with their lower ends suspended in the sump of the air washer. The measured formaldehyde removal efficiency (Table 4, tests 1–4 through 1–9) ranged from 38% to 49%. Small corrections to the measured efficiency were required to account for the storage of formaldehyde in the air washer. Another series of tests was conducted with a nonporous mass transfer medium used as the air-water contact system. This medium provided considerably more surface area for air-water contact. Water was continuously pumped from the sump to a distribution system above the medium. The measured formaldehyde removal efficiency of this system (Table 4, tests 2–1 through 2–7) ranged from 63% to 77%. For some tests, however, the corrected efficiency was substantially lower than the measured efficiency.

An overall measure of the formaldehyde removal performance of these air washers is the product of the actual air flow rate and the corrected efficiency (last column of Table 4). An air washer with an efficiency/flow-rate product equal to the largest value reported in Table 4 (74 L/s) would have approximately the same impact on indoor formaldehyde concentration as one air change per hour of ventilation (assuming an average house volume). Thus, the air washers removed formaldehyde at a significant rate, maintained their effectiveness with low inlet formaldehyde concentrations, and consumed a relatively small amount of water.

The air washer's refrigeration system, however, will require a significant amount of power. The requirement for a residential-sized unit with a 140 L/s flow rate was estimated to be 1500 to 1800 W. Also, the initial cost of the air washer would probably be fairly high, even if it is mass-produced. If the air washing process can be implemented without substantial increases in energy consumption, it may be an attractive formaldehyde control technique. Such situations may occur if the air washing process is integrated into an air conditioning system (such as that of a commercial building) or if most of the energy consumed provides usable heat.

Ventilation Efficiency Studies

A primary objective of ventilation is the control of indoor air pollutants. The performance of a ventilation system will depend on the patterns of air flow throughout the structure and on the location and emission characteristics (e.g., velocity and temperature) of pollutant sources. Air flow patterns will be a function of the locations of air supply and exhaust, the temperature and velocity of the supply airstream, and the various factors that affect indoor air movement.

Ventilation performance is often studied by using tracer gases. Either the air within the structure is labeled by injecting a tracer gas and mixing the air to obtain a uniform concentration or the incoming (supply) air is labeled. Measurements of tracer gas concentration versus time in the exhaust airstream and within the structure provide information on the distribution of the ventilation air and on the overall nature of air flow from supply to exhaust (i.e., the degree to which the flow pattern resembles short-circuiting of air or a displacement-type flow of air).

Recently, age distribution concepts have been applied in ventilation system studies. The age of a parcel of air is defined as the amount of time elapsed since the air entered the structure.[33] The average age of air within the ventilated space and the local age of air at various indoor locations can be calculated from the measured tracer gas data and compared to the ages that would result if the indoor air were per-

Table 4. Results from tests of air washers no. 1 and 2.

Test No.	Air flow rate (L/s)	Inlet HCHO conc. (ng/L)	Water repl. rate (L/h)	Water circ. rate (L/m)	HCHO removal efficiency		Product of efficiency × flow (L/s)
					Measured	Corrected	
1-4	119	116	7.6	-	0.42	0.41	49
1-5	119	106	4.5	-	0.40	0.40	48
1-6	118	218	7.4	-	0.42	0.42	50
1-7	119	222	5.7	-	0.49	0.47	56
1-8	120	250	4.1	-	0.38	0.36	43
1-9	157	360	4.1	-	0.38	0.36	56
2-1	117	269	2.3	57	0.77	0.63	74
2-2	117	252	2.3	23	0.68	0.56	66
2-3	160	85	1.7	38	0.65	0.46	74
2-4	116	80	1.4	38	0.70	0.51	59
2-5	116	161	2.3	38	0.72	0.60	70
2-6	116	102	0.7	38	0.74	0.35	41
2-7	116	136	0.5	38	0.63	0.30	35

fectly mixed. The resulting ventilation efficiencies (i.e., age with perfect mixing divided by measured age) are indicators of ventilation system performance. However, these ventilation efficiencies only provide information on the flow pattern of the ventilation air; they do not characterize the performance of the ventilation system in removing pollutants emitted from a nonuniform source.

Tracer gases can also be used to simulate indoor pollutant sources. The measured tracer concentrations can then be used to calculate the mean and local ages of the tracer within the ventilated space as well as the corresponding ventilation efficiencies. Only limited data are available from studies with concentrated pollutant (or tracer gas) sources. In addition, the usefulness and practicality of this method of characterizing ventilation system performance have not been determined. Two experimental studies were therefore conducted with tracer gases released in a manner to simulate the emission of pollutants.

The first study was conducted in a single chamber. The neutral-density tracer gases simulated pollutant emissions from either a single point or the surface of a wall. Three different mechanical ventilation systems were employed, and factors that have an impact on indoor air movement (e.g., operation of heating equipment within the chamber) were also varied between experiments. A total of 26 experiments were conducted. The data were analyzed to determine mean and local tracer ages, plus the corresponding ventilation efficiencies. This technique of data analysis did not yield satisfactory information on ventilation system performance, however, because of unstable patterns in tracer-gas transport. Therefore, other techniques of analyzing

the data are being considered.

In the second study,[34] the effectiveness of a range hood and a window fan in removing pollutants emitted from a point source was considered. Experiments were conducted in a two-room test space. The range-hood experiments were carried out with both a heated and unheated tracer gas introduced at the location of the burner. The air flow rate through the range hood (a commercially available model) was varied from 37 to 216 m^3/hr. With a heated tracer gas, the ventilation efficiency of the hood increased linearly with flow rate from 16% to 77%. (The ventilation efficiency was defined as $(C_o - C_m)/C_o$, where C_o is the concentration that would result in the test space if the indoor air had been perfectly mixed and C_m is the measured average tracer concentration in the test space.) With an unheated tracer gas, the ventilation efficiency depended highly on environmental conditions.

Window fan experiments were conducted with the fan's location fixed and the source of tracer gas in each of the two rooms. When the source and fan were in different rooms, the measured data agreed fairly well with predictions from a two-zone model in which mixing factors ranged from unity to 17.5. When the source and fan were in the same room, however, agreement with the model was poor. A measure of ventilation efficiency was the ratio of the steady-state concentration in the exhaust duct to that in the room with the fan. This ratio reached approximately four, with a flow rate of 163 m^3/hr.

Planned Activities for FY 1984

An experimental study will be conducted to determine the rates of formaldehyde transfer between airstreams of enthalpy-type air-to-air heat exchangers. These enthalpy exchangers are frequently employed in U.S. residences when ventilation with heat recovery is desired. A modest effort will also be devoted to identifying an air washing process for formaldehyde that does not require a refrigeration system. A significant effort will be devoted toward the development and validation of a practical tracer-gas-based system for studying the performance of ventilation systems in large buildings. Finally, work will begin on a computer model of residential exhaust ventilation with heat recovery.

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