Use of Simulink for Dynamic Air Quality Modeling in Interior Alaska

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Abstract: Interior Alaska has indoor air quality (IAQ) issues similar to those of other far northern communities associated with long cold winters and reduced ventilation rates. We also have some interesting issues associated with elevated radon in homes built in the hills around Fairbanks, as well as elevated particulate levels created by smoke from forest fires that occur in the vast uninhabited portions of the state. To better understand the influences of critical variables such as indoor source strengths and ventilation rates on IAQ associated with forest fires and radon, we have developed a Simulink-based mass conservation model. Using data gathered at two homes in Fairbanks during 2003, we have used this model to predict indoor radon and $PM_{2.5}$ levels (particles less than 2.5 μm in diameter). We find that we are able to predict both the rise of radon following the shutdown of a radon mitigation system and the variation of indoor $PM_{2.5}$ by using ventilation rates consistent with what we have measured, $PM_{2.5}$ source strengths associated with individuals in residences, and penetration and deposition rates compatible with what others have found. We have used situation-specific algorithms for subsurface radon source strength as well as particulate generation associated with cooking.

DOI: 10.1061/(ASCE)0887-381X(2005)19:1(3)

CE Database subject headings: Indoor air quality; Models; Ventilation; Radon; Alaska; Remedial action.

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Note. Discussion open until August 1, 2005. Separate discussions must be submitted for individual papers. To extend the closing date by one month, a written request must be filed with the ASCE Managing Editor. The manuscript for this paper was submitted for review and possible publication on December 2, 2003; approved on July 20, 2004. This paper is part of the ***Journal of Cold Regions Engineering***, Vol. 19, No. 1, March 1, 2005. ©ASCE, ISSN 0887-381X/2005/1-3–18/$25.00.

Introduction

Indoor air quality (IAQ) is influenced by many factors, including indoor sources of pollutants, outdoor air quality, ventilation rates, and building use patterns. There are no overall national standards for IAQ in homes, including those used for business activities. Allowable levels of critical constituents in the outdoor ambient air are quantified by the National Ambient Air Quality Standards (NAAQS). The six NAAQS constituents are sulfur dioxide (SO_2), ozone, nitrous oxides, carbon monoxide (CO), fine particulate matter (PM_{10}), and lead. Allowable levels are typically around a hundred parts per billion (ppb), except for CO at 9 parts per million (ppm) and lead at a few ppb (Johnson et al. 2002a). Major indoor pollutants include radon, formaldehyde, combustion products, biological contaminants, tobacco smoke, organic gases, lead, pesticides, and asbestos. Researchers established for the first time in the mid-1960s that air pollutants generated indoors might be responsible for adverse health effects originally attributed to outdoor air (Namiesnik et al. 1992).

Interest in specific indoor air contaminants has evolved over time, including an emphasis on carbon dioxide (CO_2) in the mid-1800s, SO_2 in the mid-1900s, and, in the last 30 years, half a dozen different contaminants or classes of contaminants. These include volatile organic compounds (VOCs), nitrous oxides (NOx), formaldehyde, radon, asbestos, particulate matter (PM), lead, and microbial matter (Namiesnik et al. 1992; Seifert 2002). Many studies have been conducted concerning the health effects of air pollution over the past 50 years. In the Harvard Six-Cities Study (Dockery et al. 1993), researchers found indoor PM levels were higher than outdoors in five of the six cities, with respiratory PM ($<3.5\ \mu m$) ranging from lows of 10 to 20 $\mu g \cdot m^{-3}$ indoors and outdoors to a high of 300 $\mu g \cdot m^{-3}$ indoors and 60 $\mu g \cdot m^{-3}$ outdoors. Work such as the U.S. Environmental Protection Agency (EPA's) Total Exposure Assessment Methodology (TEAM) and the Harvard Six-Cities Study considered the various components of personal exposure and found indoor sources to be significant.

Substantial uncertainties still exist, such as the chemistry of complex mixtures that occur indoors and the doses of microorganisms and their derivatives required to produce adverse health effects (Spengler et al. 2001). Fisk (2001) estimated that potential annual savings plus productivity gains from improved indoor environments exceed \$40 billion in the United States. This includes, for example, reduced symptoms of sick building syndrome and respiratory disease as well as direct improvement in worker performance unrelated to health.

Because of the expense of conditioning outside air during a large part of the year, there is an incentive to build "tight" homes in northern climates. In a prior study (Johnson et al. 2002a), we found the average air exchange rate for eight homes studied in detail was 0.24 air charges per hour (ACH). Koontz and Rector (1995) compiled data from various projects (2,971 measurements) across the United States where perfluorocarbon tracer techniques (PFT) were used. From this composite data, they documented 0.45 ACH as the 50th percentile. Pandian et al. (1993) reviewed data on 1,836 U.S. residences (locations not specified), where ACH values were as low as 0.1, with approximately half of the observations ranging from 0.35 to 2.35 and an arithmetic mean of 2.0. ACHs changed considerably with seasons; mean values for fall (0.4), winter (0.5), spring (1.9), and summer (5.4) were reported. Stricker Associates Inc. (1994) found ACH ranging from 0.1 to 0.38 with an average value of 0.23 for 30 representative houses in Quebec.

The above information indicates that Alaskan and Canadian buildings tend to be ”tighter“ than those in the continental United States. The tighter a building, the more readily indoor-generated pollutants can accumulate. But infiltration (uncontrolled flow of air through unintentional openings) varies not only from one building to another and with local climate, but also with time in a given building. The standard deviation for the latter can be half the mean. Although energy loads are proportional to the ventilation rate (due to forced ventilation plus infiltration), pollutant concentrations are not as indoor sources can vary markedly with time. It was found that summer infiltration was dominated by the rapidly varying wind speed and winter infiltration by the much more slowly varying indoor-outdoor temperature difference $\Delta T_{\text{in-out}}$ (Sherman and Wilson 1986).

Radon, arising from the decay of uranium, is a naturally occurring radioactive gas found in pores in the soil at concentrations sometimes exceeding 1,000 ppm (ambient air concentrations are typically <1 ppm). Rn-222 is the isotope of concern, and its progeny, polonium-218 and 214, are of concern because they emit alpha particles, which can initiate lung cancer. The recommended maximum annual average indoors is 4 $\text{pCi} \cdot \text{L}^{-1}$, with 7% of homes in the United States and 30% in the hills around Fairbanks estimated to exceed this (Johnson et al. 2002a).

Objectives

One objective was to assess the IAQ for a home in Goldstream Valley, 5 mi north of the University of Alaska (UAF) and away from the downtown area, and another home 2 mi southeast of the UAF and about 2 mi from downtown Fairbanks. The first building was chosen both because of the active cooperation of the owners and because it had a high occupancy density during weekdays (up to 13 people) since it serves as a place of business. Its “livable” volume of 630 m^3 (excluding crawl space, mechanical room, and storage), contains three occupied levels, each differing by about 1 m in elevation with a floor area of 211 m^2. These consist of a midlevel area near the entrance, a lower level with more offices and a bathroom, and a kitchen, bathroom, two offices, and a conference room upstairs. A propane-fired range in the kitchen is used occasionally. Under the kitchen, adjacent to the occupied lower level, is the mechanical room. In addition to the boiler for the hydronic heating system and domestic hot water, the water accumulator for the well water, and the heat recovery ventilator (HRV), the mechanical room has a large amount of frequently accessed storage, an area for air drying employee’s winter gear, and computer servers. Combustion air is supplied directly to the mechanical room from the outside.

The second home was chosen because we had already obtained data for it in 1999 and wanted to focus our efforts for this updated study on particulate matter, especially during periods when elevated smoke levels occur in interior Alaska due to forest fires. This home consists of a 134 m^2 main floor plus an 88 m^2

daylight basement with an attached garage. Heat is supplied by an oil-fired forced air furnace, with combustion air being the indoor air. The home also has an electric water heater and stove.

A second objective was to use the detailed measurements plus building occupancy patterns to help calibrate a Simulink-based IAQ model. This model, which previously had been used to model CO_2 as well as $PM_{2.5}$ in a two-room cabin in Fairbanks (Johnson et al. 2002b), assumes completely mixed flow (CMF) in a given zone similar to a nongraphical model developed by the EPA (Gao 2000). The particular applications in this paper are for radon and particulates. We collected data on other IAQ parameters to establish that these two homes had acceptable IAQ.

Because of the special environmental consciousness of the building inhabitants in Goldstream Valley (no smokers, motor vehicle idling minimized, minimal interior carpeting, environmentally friendly cleaning protocols followed), we wanted also to compare levels of IAQ parameters with those we had observed in other Fairbanks homes during a prior study (Johnson et al. 2002a), where none of the homes were occupied by smokers. We also wanted to observe and model the relationships between indoor and ambient air quality during periods of smoke caused by forest fires.

Methods

We deployed instruments to measure 14 different physical/chemical and 2 microbiological IAQ parameters over a 3-week period at House 1 during the winter of 2003. The former included CO, CO_2, relative humidity (RH), temperature (T), radon, $PM_{2.5}$, measured at intervals ranging from 5 min to 1 h, formaldehyde averaged over an 8 h period, benzene, toluene, and hexane averaged over a 7-day period, black (BC) or elemental (EC) carbon, and building walk-throughs for PM_1, total VOCs, and NO_2. Air samples, dust samples from carpets, and tape samples from surfaces were collected and sent to an outside laboratory for identification of total spores (viable and nonviable), pollen and mycelial fragments, or microscopic screen and fungi identification. These results (no apparent problems) will not be presented here. Indoor data on CO, CO_2, RH, and T were collected in the kitchen, lower-level office area, and crawl space.

The detectors used infrared (IR) absorption to measure CO_2, electrochemical detection for CO, thin-film capacitive sensors for RH, thermistors for T, and absorption at 880 nm for BC. The three specific VOCs and formaldehyde samples were collected using passive badges and sent to an independent laboratory for desorption and analysis by gas chromatography/mass spectrometry (GC/MS) or spectrophotometry. Total VOCs were detected using a handheld instrument with a 0.4 L min^{-1} flow-through rate employing a photo ionization detector (PID). The aerosol monitor (TSI Dust Trak) comes with inertial separators for the front end that allow particles larger than 1, 2.5, or 10 μm to be excluded from entering the detection chamber. The monitor employs the scattering of laser light to infer aerosol mass concentration and is calibrated against a gravimetric reference test

Table 1. Baseline Air Quality Levels in Houses 1 and 2 (Excluding Multiday Periods When Number of Occupants=0)

House	CO_2 (ppm)	RH (%)	$PM_{2.5}$ ($\mu g \cdot m^{-3}$)	Average time (days)	When sampled
House 1					
Maximum	1,623	22	38	—	January–February 2003
Average	1,128	17	18	18	
Minimum	509	13	12	—	—
House 2					June 2003
Maximum	1,744	44	298	—	—
Average	845	37	12	4.5	—
Minimum	345	21	3	—	—

dust representative of a wide variety of ambient aerosols. The flow rate was 1.7 Lpm.

We used continuous radon monitors (CRMs) to measure radon levels in both the basement (mechanical room) and kitchen areas at 8 h intervals. We did this both during the 20 days the mitigation fan was operating and also for 9 days when neither it nor the heat recovery ventilator (HRV) were operating. The CRMs employed solid-state alpha particle detectors that recorded frequency of alpha particle impacts via a mechanism similar to one that uses the photoelectric effect such that an electric current is generated. Alpha track detectors deployed for 23 days in the kitchen (upper level) and 29 days in the basement (lower level) were colocated with the CRMs.

We deployed instruments to measure indoor and outdoor CO, CO_2, RH, T, and $PM_{2.5}$, plus outdoor soot and particle number density over several periods totaling 18 days at House 2 during the summer of 2003. We also collected outdoor and indoor samples on 0.8 μm polycarbonate filter paper to allow us both to infer particulate mass density gravimetrically and to examine particle shapes and size distributions via photomicrography.

Results

For House 1, we found the 5 min indoor CO_2 levels ranged from 509 to 1,623 ppm, with a mean of 1,128 ppm during the 18 days when the building was occupied. The corresponding RH and $PM_{2.5}$ data appear in Table 1. The average ambient $PM_{2.5}$ during this time of 15.8 $\mu g \cdot m^{-3}$ was 2.2 $\mu g \cdot m^{-3}$ lower than the indoor and statistically different than the indoor ($P = 1.49 \times 10^{-13}$). The average indoor 5 min CO was 0.5 ppm with a maximum 8-h average of 1.9 ppm.

With the radon mitigation system operating, the 8 h average radon levels in the house (using basement and kitchen data) ranged between 0.5 and 4.5 $pCi \cdot L^{-1}$ in January 2003. With the mitigation system disabled for 61.5 h starting on January 24, 2003, these levels rose to 42 and 18 $pCi \cdot L^{-1}$, respectively. The *r*-value between these two levels was 0.95 for all the data and 0.85 with the fan off. These values are consistent with those measured over a 3 to 4 week period using alpha trak detectors. Such detectors have been shown to produce data having a standard deviation of $\sim$0.14 $pCi \cdot L^{-1}$ when measuring levels of 4 $pCi \cdot L^{-1}$ over 30 days (E. Port, RSSI Inc., personal communication, March 24, 2004).

The ventilation rate inferred from CO_2 decay was 0.35 ACH with the HRV operating and averaged 0.22 with the HRV disabled. The range of values inferred from decay of CO_2 after the occupants left in the evening was 0.3 to 0.4 ACH. Blower door results indicated 3.4 ACH_{50}, leading to an estimated winter ventilation rate of 0.4 ACH.

The average ambient BC from January 28 at 9:40 a.m. until January 31 at 3:30 p.m. was 626 $ng \cdot m^{-3}$. If we look at the 10% highest and lowest values (out of 933 values), then the averages are 1,738 and 90, respectively, with all of the former occurring either near the noon hour or the end of a workday, while the latter all occurred from 8 p.m. Friday until 1:30 p.m. Saturday.

For House 2, we found indoor CO_2 levels varied from near ambient to about fivefold ambient, with an average value of about twice ambient (assumed 370 ppm) during 4.5 days in early June 2003. The RH varied between 21 and 44%, while the $PM_{2.5}$ averaged 12 $\mu g \cdot m^{-3}$. Details not shown on Table 1 reveal the indoor $PM_{2.5}$ to be uncorrelated ($r < -0.01$) with the ambient for this period, when the ambient air average $PM_{2.5} \sim 5$ $\mu g \cdot m^{-3}$. On the other hand, during 46 h of light smoke in late June and 19 h of heavy smoke in July, $r = 0.77$ and 0.78, respectively. In this July period, the average and maximum 5-min $PM_{2.5}$ outside = 99 and 199 $\mu g \cdot m^{-3}$ with 45 $\mu g \cdot m^{-3}$ as the average inside. For both the smoky and nonsmoky days, the mean $PM_{2.5}$ inside and outside were significantly different ($P < 10^{-8}$).

The soot levels, as indicated by EC in the ambient air at House 2, averaged 210 $ng \cdot m^{-3}$ with a maximum of 2,354 $ng \cdot m^{-3}$ (with a car starting near the sensor) for the June 6–11 period. During a 19-h period of heavy smoke beginning at 10:30 p.m. on July 23, the EC averaged 620 $ng \cdot m^{-3}$ with a maximum of 1,396 $ng \cdot m^{-3}$. Fig. 1 shows $PM_{2.5}$ and BC levels during the initial 19 h of data we collected during the heavy smoke period in July. Both are at elevated levels compared with a nonsmoke period such as the first week in June. The correlation coefficient between these two variables is $r = 0.63$. Fig. 2 reveals the particle number concentration is well correlated with the $PM_{2.5}$, with an r-value of 0.96 for the 11 h period shown.

Ventilation rates varied from 0.15 to 0.24 ACH with doors and windows closed as inferred from CO_2 decay following the departure of the occupants or from deliberate injection of CO_2.

A walkthrough on two different occasions showed elevated PM_1 levels associated with specific activities. For example, on June 8, the PM_1 number density in House 2 over a 10-min period without cooking activities varied between 5 and 10 K/cm^3. Later that day after cooking hamburgers, the PM_1 levels in the kitchen ranged from 10 to 60 K/cm^3 and stayed close to that level for 2 h. Two days later, after baking cookies, the PM_1 in the kitchen rose to 160 K/cm^3. It should be noted that no exhaust fan was operating during these periods. On the same day, the PM_1 rose to over 300 K/cm^3 in the garage immediately after a car drove in

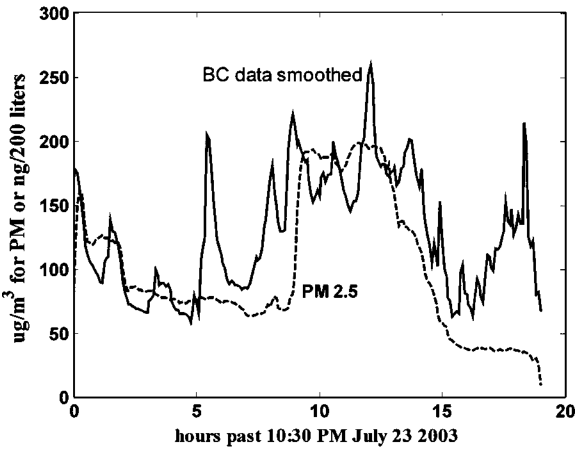

Fig. 1. Ambient BC and $PM_{2.5}$ at House 2 during forest fire event

and then decayed rapidly. This is in contrast to 9:40 a.m. on July 24 during the 19-h smoke period, when the PM_1 outside was $\sim 9\ K/cm^3$.

Discussion of Field Data

The fact that the maximum CO_2 levels for House 1 were close to those for House 2 (in spite of having more occupants from 8 a.m. till 5 p.m.) is due to the consistent ventilation provided by the HRV. The average level for House 1 while occupied would have put it in the upper third of the eight homes monitored year-round in 1999–2001 in Fairbanks, yet the maximum was less than half that found in bedrooms in the prior study (Johnson 2002a). The average CO_2 level for House 2 (measured in the living room) of 845 ppm was close to the summer average of 741 ppm measured in 1999 (Johnson et al. 2002a) when the year-

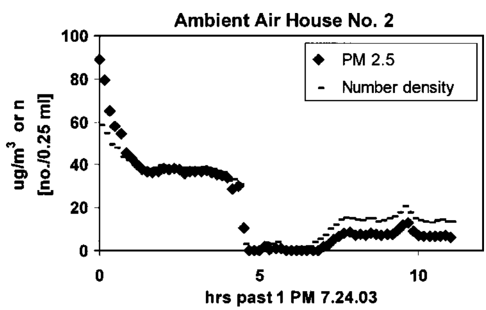

Fig. 2. $PM_{2.5}$ and particulate number density for House 2

round average was 890 ppm. The average summer RH of 37% for House 2 was lower than a 10-day average of 45% measured in the summer of 1999. The $PM_{2.5}$ levels in both houses were less than the proposed 24-h NAAQS of 65 $\mu g \cdot m^{-3}$.

For House 1, we found the ambient soot levels reached a peak at times coinciding with motor vehicles leaving the parking lot at the end of the workday and were at a minimum on a weekend. This is consistent with soot (as measured by BC) being associated with the burning of carbonaceous fuels such as gasoline and diesel fuel, and the aethalometer being placed to sample just outside a garage that was about 20 m from where up to 15 motor vehicles were parked when people were at work. People working in a second building used some of these vehicles. The fact that the radon levels were much lower when the active subslab depressurization mitigation system was operating rather than disabled showed the effectiveness of this technology.

LaRosa et al. (2002) collected 10 months of data at a home in northern Virginia in 1999 and 2000 and found a mean EC of 680 $ng \cdot m^{-3}$, with 99% less than 2,929 for ambient air. The average indoor to outdoor ratio was 0.35, with the strongest indoor source being the burning of citronella candles. They reported median EC levels in ambient air in the United States were 700 $ng \cdot m^{-3}$ in 24 rural areas and 1,400 $ng \cdot m^{3}$ in 15 urban areas. During the period of heavy smoke in July, our maximum was 1,396 $ng \cdot m^{-3}$. Although considered inert, materials found to coat BC have been found to be mutagenic and carcinogenic.

During a 19-h period of heavy smoke in Fairbanks, the r-value between the indoor and outdoor $PM_{2.5}$ at House 2 was 0.78. Our average ambient $PM_{2.5}$ during this time was just above the 80–90 $\mu g \cdot m^{-3}$ measured at a number of sites in eastern New York on July 7, 2002, following a forest fire in Quebec (Rupprecht & Patashnick 2002). For the next 9 h, the r-value was −0.10, showing the dominance of indoor sources under many conditions. The ambient $PM_{2.5}$, PM_{10}, particulate number concentration, and BC all had positive correlations during these first 19 h. The r-value between the number of particles less than 2 μm in diameter and $PM_{2.5}$ was 0.97 for the hours from 1 p.m. till 5:20 p.m. on July 24, 2003. As seen in Fig. 2, even after the smoke had decayed appreciably (by midnight on July 24), the correlation is still substantial. Wetzel et al. (2003) found there were around 1,000 to 20,000 PPL between 0.3 and 2 μm at a site 20 mi north of Fairbanks during a 50-day period in spring 2001. Typically values were around 10,000 PPL. Our values in Fig. 2 are $>200k/L$ during high smoke periods but $<4k/L$ on normal days.

Ramachandran et al. (2003) found the average indoor $PM_{2.5}$ gravimetric concentration to be 13.9 $\mu g \cdot m^{-3}$ in a study of 30 residences in three communities in Minnesota. They also found the R^2 between 15-min indoor and outdoor $PM_{2.5}$ values was 0.49 in the spring and summer and 0.13 in the fall. Patterson and Eatough (2000) found the outdoor $PM_{2.5}$ to be correlated with outdoor soot, with $R^2=0.47$ for an elementary school. They also found $R^2=0.50$ for outdoor $PM_{2.5}$ versus total particulate number and found a penetration factor, p, of 0.3 could be used to explain the indoor $PM_{2.5}$ on a day (a school holiday) when there were presumably no indoor sources. Our ratio of indoor to outdoor $PM_{2.5}$ during a smoky period was 0.45.

The total volatile organic compounds (TVOCs) levels were normally

$<$300 ppb (as isobutylene) measured by the handheld photo ionization detector (PID). Baking bread elevated these levels to 600 ppb in the kitchen of House 2 (electric oven with no exhaust fan). Normal indoor air has a TVOC level of 100 to 400 ppb isobutylene units (RAE Systems 2002). Wallace (2001) found about half of 750 homes sampled in the United States had TVOC levels greater than 1,000 $\mu g \cdot m^{-3}$. Building materials and consumer products such as air fresheners are major VOC sources in nonsmoking homes. For both houses, benzene and *n*-hexane levels were less than 10 $\mu g \cdot m^{-3}$ ($\sim$3 ppb) and toluene less than 18 $\mu g \cdot m^{-3}$ ($\sim$5 ppb). In a study in Valdez, Alaska, involving 58 homes, benzene levels averaged 16 and 25 $\mu g \cdot m^{-3}$, respectively, in the summer and winter (Wallace 2001).

Even though we want to focus our efforts now on using Simulink to model some interesting dynamic effects, it is important to take note of these observations to verify that several air quality parameters—ranging from CO_2 to TVOCs and including particulates—were typically found to be in normal ranges for residences. Two notable exceptions discussed below are (1) elevated radon levels in House 1 when the radon mitigation system was disabled for 61 h and (2) elevated particulate levels in House 2 during a period of forest fire activity.

Modeling

We employed both steady-state and transient mass conservation models to help quantify the relationships between critical variables for our studies involving radon, CO_2, VOCs, and particulates. The conservation of mass relation for a given zone may be expressed as

$$V dc/dt = Q(pc_a - c) - (k + k_d)Vc + \dot{S} \tag{1}$$

where V=zone volume; c=concentration; $\dot{S}$=source strength; Q=convective flow through the zone; p=penetration factor; c_a=concentration of air entering the zone; k_d=deposition rate coefficient; and k=decay rate coefficient. We utilized integration blocks within Simulink to integrate Eq. (1) and march downstream with time for the entire interior occupied volume, assuming a completely mixed flow (CMF) within. The air exchange rate, ambient concentration, penetration rate, and source strength can all be input as functions of time, while the decay and deposition rates are assumed constant. Baseline ventilation rates were estimated using CO_2 decay patterns when the number of building occupants was zero. With no sources, deposition, or decay, the solution of Eq. (1) is

$$\ln\left[\frac{c_o - c_a}{c - c_a}\right] = \mathrm{ACH} \cdot t \tag{2}$$

where c_o=concentration at t=0, and ACH=Q/V. Performing this calculation for data collected on 10 different evenings resulted in ACH varying from 0.2 to 0.4 for House 1. This rate, of course, includes the effect of the HRV, which was rated at 0.35 ACH for the “high” setting while it was operating. For House 2, the air exchange rate during the summertime sampling period was dependent on the

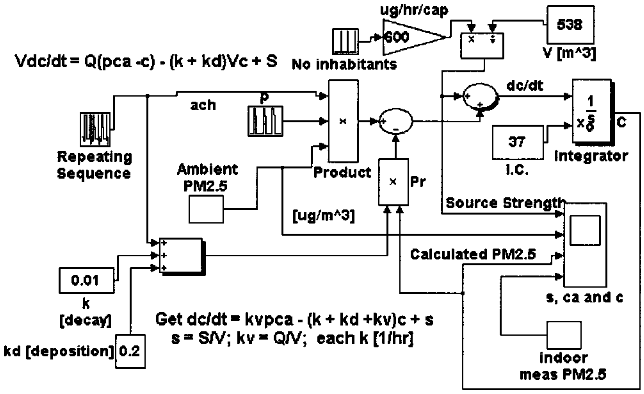

Fig. 3. Simulink-based IAQ model

numbers of open doors and windows and was estimated to range between 0.16 and 1.3 ACH.

Although we will focus here on the application of the model below to $PM_{2.5}$ and radon, we have also used it to model CO_2 variations (Johnson et al. 2000a).

In Fig. 3, this graphically based model solves Eq. (1) using a variable time step ordinary differential equation solver, depicted by the integration block on the righthand side near the top. For its application to $PM_{2.5}$, shown in Fig. 3, required inputs (each as functions of time) include ACH, the measured indoor and ambient $PM_{2.5}$, house volume, and number of occupants. We employed a variable time step and initially ran the simulation for 28 h, which coincided with a period of elevated particulate levels in the ambient air associated with a forest fire about 150 km to the southwest of Fairbanks. The ventilation rate was 0.2 to 0.45 ACH, except for 1.2 ACH during the first 45 min when the front door and five windows were open. We assumed the particle penetration to be 0.6, the $PM_{2.5}$ generation rate per capita to be 600 $\mu g \cdot h^{-1}$, and decay and deposition coefficients of 0.01 and 0.2 h^{-1}, respectively.

Allen et al. (2002) found $p = 0.53 \pm 0.25$ for 85 residences in Seattle for PM_1, with large differences by building types and by season. Long et al. (2001) found summer hourly p values >0.7 for $PM_{2.5}$ and for 73% of the winter <0.7. During a smoky period in Fairbanks in the summer of 2002, Reynolds (2002) found $p = 0.71$ for an older and 0.50 for a newer home, respectively. Freijer and Bloemen (2000) found an indoor/outdoor ratio of 0.33 for ozone via a modeling exercise for a 250 m^2 home. These researchers looked at the potential of mass balance models to predict the effects of various ventilation and source scenarios on IAQ. They commented on the data from three very large studies (over 1,000 homes total), showing profound differences between ventilation rates in homes, with 10 to 90% values ranging from 0.14 to 3.52 ACH. They also showed results illustrating the profound effect of opening doors and windows on air exchange rate (mean values from 0.45 to 2.62 ACH for an office room). Liao et al. (2003) found a particle deposition rate for PM_{10} to be ~ 0.7 h^{-1}. Using PTEAM data, Wilson et

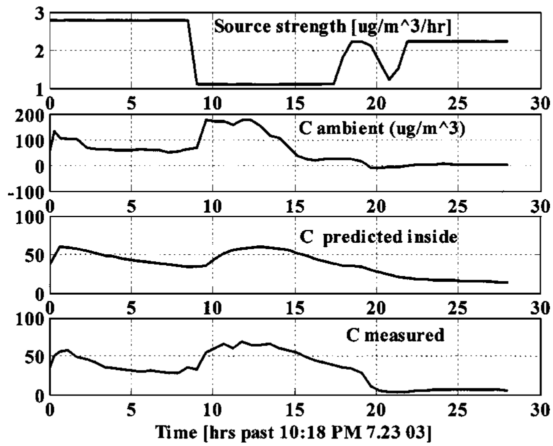

Fig. 4. Model predictions for $PM_{2.5}$ during period of heavy smoke

al. (2000) found deposition rate coefficients for $PM_{2.5}$ to be 0.27 h^{-1} during the day and 0.39 h^{-1} at night. They also found $p=1$ during the day at 1.14 ACH and 0.89 at night at 0.98 ACH.

Since the $PM_{2.5}$ real-time data were obtained using a TSI Dust Trak, based on the scattering of light, we used a correction factor (CF) of 0.37 to account for the differences between the actual distribution of particle sizes occurring during the period of interest and the distribution (Arizona test dust A1) used at the factory in calibrating the instruments. This CF is in the range found by others (Ramachandran et al. 2003) and represents an average of what we found using gravimetric data from two filter papers on which particulates were collected during the same time period the Dust Traks were operating. In 1999, with ambient $PM_{2.5}$ $=2.8\ \mu g \cdot m^{-3}$, we found the CF=0.75. As shown in Fig. 4, the inside $PM_{2.5}$ closely tracks the ambient, showing the dominance of outside sources during this time.

We also used this model to predict indoor $PM_{2.5}$ concentration during a period of some—but much less intense—smoke. Moreover, during this 52-h period in June 2003, from 1 to 7 windows were open so that not only was the ventilation rate increased but the particle penetration rate (p) also increased. As shown in Fig. 5, the indoor $PM_{2.5}$ approximately tracks the ambient but is also influenced by events inside, such as frying shrimp on hour 46. This caused the indoor $PM_{2.5}$ to rise dramatically while the ambient stayed about the same. To model this effect, we assumed the indoor source strength rose dramatically during the cooking process (to hundreds of milligrams per hour).

For the model's application to radon, we picked a 60-h period following the shutdown of a radon mitigation system on January 1, 2003, and used the model to predict the in-house radon concentration for House 1. The results presented in Fig. 6 show the model is able to approximate a rise in indoor radon as the subslab source strength increases, with the ventilation rate determined by tracer decay and

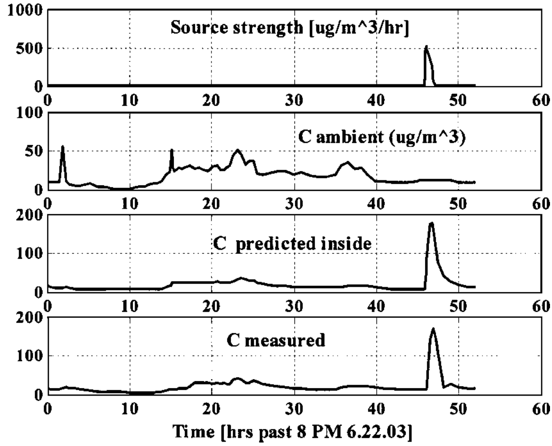

Fig. 5. Model predictions for $PM_{2.5}$ during period of light smoke

differential pressure ($\Delta P_{\text{in-out}}$) data. We have let the radon source strength be related to the submembrane radon concentration, following Scott (1992). For a given soil/membrane permeability and a given pressure difference between the soil below the membrane and dwelling, the source strength will vary linearly with the radon concentration in the soil pores. Other data we have indicate only small variations in pressure difference after the mitigation fan is disabled.

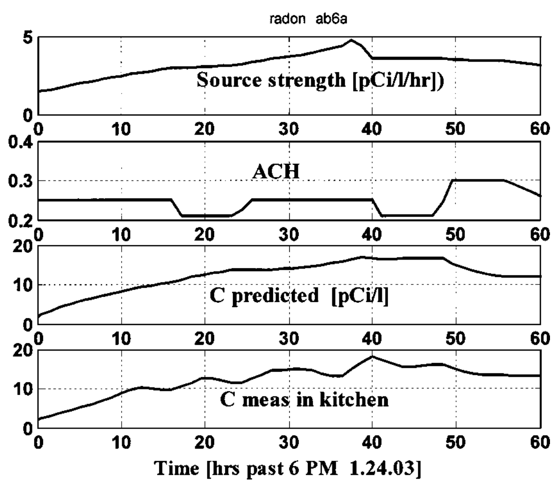

Fig. 6. Model results for radon in House 1

Discussion of Model Results

The results appearing in Fig. 4 indicate a good agreement between the model and data over the 28-h period shown. The indoor PM is dominated by the outdoor during the first 19 h (period of heavy smoke) with a correlation coefficient of 0.78. During the last 9 h, during typical ambient conditions, $r=-0.10$. The source strength varied because the number of occupants changed (2 to 3 people left the house 9 h into the simulation). The fact that the agreement is reasonable indicates our assumed ACH and indoor source strengths are consistent with the measurements. Our model treated the house as one zone and hence assumed the particles to be well mixed throughout the volume. This is consistent with the small particles having very low settling velocities plus the CO_2 levels being similar throughout the house in the absence of significant point sources (people).

Stricker Associates (1994) found the respirable suspended particles (RSP) (here as $PM_{2.5}$) average source strength was $3.21\ mg \cdot h^{-1}$ for 30 homes in Quebec, including those with smokers. The average $PM_{2.5}$ levels in nonsmoking homes was 23.3, compared with $52.9\ \mu g \cdot m^{-3}$ in smoking homes, with an average ACH of 0.22 for all homes. The data were obtained for homes in Quebec between December 1993 and March 1994. For the 13 homes with <10 cigarettes per week smoked and <30 h per week use of wood in a stove or fireplace, the average RSP source strength was $1.99\ mg \cdot h^{-1}$ and a specific source strength sdot $=680\ \mu g \cdot cap^{-1}\ h^{-1}$. Our model gives very similar results whether sdot $=600$ (as in Fig. 4) or $300\ \mu g \cdot cap^{-1}\ h^{-1}$, because infiltration-generated $PM_{2.5}$ dominates during this smoky period.

For the 52-h period in June, more windows and doors were open than during the heavy smoke period in July. As shown in Fig. 5, the indoor $PM_{2.5}$ was not dominated by the outdoor during a period of frying shrimp on hour 46, but during the preceding 46-h period, the correlation between the indoor and outdoor $PM_{2.5}$ was 0.77. During this time, the average indoor and outdoor $PM_{2.5}$ were 17.2 and $18.0\ \mu g \cdot m^{-3}$, respectively. Wallace (1996) found cooking to be the second most important source of indoor PM, next to environmental tobacco smoke (ETS).

The radon model used the same basic mass conservation model given by Eq. (1) with a decay rate $k=0.0075\ h^{-1}$, $kd=0$, $p=1$, the ambient radon concentration $=0.4$ pCi/L, and an underground source. Assuming one knows the ventilation rate, the critical independent variable is clearly the radon source strength. We assumed the ventilation rate is a function of $\Delta P_{\text{in-out}}$. The radon source strength appearing in Fig. 6 is consistent with the Scott (1992) model for soils with permeabilities between 10^{-10} and $10^{-9}\ m^2$ (sand/gravel mixtures) and submembrane data obtained in March 2003.

One could next use these models to perform sensitivity calculations for such purposes as designing mitigation systems for radon or subsurface VOCs, or for operating HRVs during periods of poor-quality ambient air. For example, one could reduce mechanical ventilation and/or increase filtration when ambient $PM_{2.5}$ levels rise above a certain threshold. High-efficiency air filters have been shown to remove 95% of $PM_{0.3}$ (Fisk 2001). For future work, use of perfluorocarbon tracer gas, for example, could help us better quantify variations in ventilation rates, and additional data on subslab differential pressure plus radon concentration would help provide better estimates for the radon source strength.

Conclusions

1. Particulate levels found in these homes in Fairbanks are consistent with those found in other northern residences.
2. Fine particulates generated by forest fires can travel hundreds of kilometers and dominate indoor sources that may normally be more significant than outdoor sources.
3. A Simulink-based model offers a graphically based means of predicting time-varying concentration of particulates and other IAQ parameters in indoor air.
4. During the summertime there is a strong correlation between indoor and outdoor $PM_{2.5}$ in Fairbanks on smoky days. Such is not the case during other periods, when indoor sources are more significant.
5. In a house used as an office during the day, with up to 15 motor vehicles parked nearby, the indoor air quality remained in a good range during 3 weeks in the winter with respect to the parameters we evaluated.
6. When homes are built in regions prone to indoor air quality problems with respect to radon, active soil depressurization (ASD) mitigation systems can perform well. Further, the performance of such systems can be quantified by a dynamic mass conservation model, providing sufficient information on the radon source strength and ventilation rates is available.
7. This model can be used to provide input to design and operation decisions for Alaskan and other homes.

Acknowledgments

The writers thank the U.S. Environmental Protection Agency and the Alaska Department of Health and Social Services for funding this study.

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