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EFFECTS OF RESIDENTIAL WOOD COMBUSTION ON INDOOR AIR QUALITY: A CASE STUDY IN WATERBURY, VERMONT

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Abstract—An indoor/outdoor monitoring study was conducted during January–March 1982 in Waterbury, Vermont. Respirable particle measurements were made inside and outside 24 homes (all occupants were nonsmokers), 19 with wood-burning appliances and 5 without. Data were also obtained on seasonal air-exchange rate, heating fuel consumption, and relevant home characteristics. Findings indicate that indoor particle levels are consistently higher than outdoor values regardless of heating-fuel type. No statistical difference was observed between 24-h average respirable particle levels in wood- and nonwood-burning homes. A linear regression model, incorporating information on air-exchange rate, house volume, fuel use, and outdoor levels, accounted for about 20% of the variance in indoor particle concentrations.

Key word index: Indoor air quality, indoor air quality models, indoor aerosol, indoor/outdoor measurements, residential wood combustion, respirable particles, wood-stove emissions.

INTRODUCTION

Consumer concerns about rising prices and uncertain availability of petroleum and natural gas have led to increased use of alternative fuels for residential space heating. An estimated 2 million woodstoves were sold in 1982, a 10-fold increase since 1972. It is now estimated that approximately 10% of the total space-heating input for the United States is from firewood (Lipfert and Dungan, 1983). Because wood-burning stoves, furnaces, and fireplaces have been shown to emit significant quantities of particles, carbon monoxide, and polycyclic aromatic hydrocarbons (DeAngelis *et al.*, 1980a; DeAngelis *et al.*, 1980b; Hall and DeAngelis, 1980; Dasch, 1982; Butcher and Sorenson, 1979; Butcher and Ellenbecker, 1982; Sanborn and Blanchet, 1982; Barnett and Shea, 1982; Peters *et al.*, 1982; Cooke and Allen, 1982; Kowalczyk *et al.*, 1982; Cooper, 1980), this shift in space-heating fuels has raised questions about possible adverse consequences for public health.

Ambient air quality impacts due to emissions from residential wood combustion (RWC) have been documented by several investigators (Sexton *et al.*, 1984a; Custin and Murphy, 1978; Murphy and Buchan, 1982; Imhoff *et al.*, 1982; Carlson, 1982; Otis, 1977; Decesar and Cooper, 1982; Kowalczyk and Greene, 1982; Cooper *et al.*, 1981; Duncan *et al.*, 1979; Hornig *et al.*, 1982; Dalton *et al.*, 1977; Romero *et al.*, 1978). One issue which has received relatively little attention is the effect of RWC on indoor air quality. Woodsmoke can be emitted directly into the living space from improperly installed, maintained, or operated stoves and fireplaces. Alternatively, woodsmoke of outdoor origin (emitted from chimneys) can penetrate indoors as a consequence of natural or forced ventilation. Because urban residents typically spend 60–70% of their time at home (Szalai, 1972; Chapin, 1974), indoor residential environments might be the major route of exposures to emissions from wood combustion.

Although relatively few studies have examined air quality inside wood-burning residences, the evidence suggests that indoor concentrations of particles, organics, carbon monoxide, nitrogen oxides and sulfur dioxide can increase during operation of wood-burning appliances (Moschandreas *et al.*, 1980; Colome and Spengler, 1982; Colome *et al.*, 1981; Benton *et al.*, 1982; Neulicht and Core, 1982; Traynor *et al.*, 1982; McGill and Miller, 1982). However, the limited number of measurements (e.g. all published studies were conducted in 8 or fewer houses) seriously hinders attempts to evaluate associated health risks.

This paper discusses results of an intensive indoor monitoring program in Waterbury, Vermont, during the winter of 1981/1982. Respirable particles (RSP) were measured inside and outside 19 wood-burning (two with kerosene-fired heaters) and five nonwood-burning homes over 2-week periods. Data were also obtained on home characteristics, ventilation rates and energy consumption. Results are presented emphasizing the effects of residential wood combustion on indoor air quality.

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EXPERIMENTAL

The Harvard wood-burning study was conducted from 29 January to 11 March 1982, in Waterbury, Vermont. Waterbury is a rural community of about 2000 people located along the Winooski River in the northwestern section of the state. Approximately 50% of the private residences use wood fuel as either a primary or secondary heating source (Sanborn *et al.*, 1981; Sanborn *et al.*, 1982). Although there are no industrial emission sources in the vicinity, wintertime temperature inversions frequently promote overnight pollution buildup within the valley. Ambient air quality during the study period has been described in a previous paper (Sexton *et al.*, 1984a).

Indoor/outdoor measurements

Information from the Vermont Agency of Environmental Conservation wood-burning survey was used to develop a mailing list for Waterbury. Requests for volunteers to participate in the 1982 Harvard Wood-Burning Study were mailed to residents of the community during November–December 1981. Twenty-four homes (19 with wood-burning appliances) were selected from the pool of volunteers for inclusion in the indoor/outdoor monitoring program. No attempt was made to obtain a random sample. Instead, selection was based on (1) willingness of occupants to participate in the personal sampling portion of the study and (2) use of wood fuel as a primary or secondary heating source. Only homes with nonsmoking occupants were included in the study in order to minimize the confounding effects of tobacco smoke on indoor RSP concentrations.

Monitoring operations were divided into two sampling periods, with 12 homes studied from 29 January to 11 February and another 12 homes investigated from 28 February to 11 March. Twenty-four hour (i.e. 0800–0800) RSP samples were collected inside and outside each dwelling every other day for 2 weeks. An additional residence (a new 'energy efficient,' all-electric condominium) was included in the energy consumption and ventilation portion of the study, but no indoor pollution measurements were obtained.

Respirable particle concentrations were measured with Bendix cyclone preseparators attached to Harvard/EPRI portable sampling pumps (Turner *et al.*, 1979). Two RSP samplers were placed inside each residence and one was installed outside. The outdoor pump was encased in a heated box and connected to an external cyclone and filter. Flow checks were performed with a calibrated rotometer on all units before and after each run.

One-time surveys were made in each home to perform sulfer hexafluoride (SF_6) ventilation measurements and record important structural parameters (e.g. ceiling height, floor space). A fan pressurization test (i.e. blower door technique) provided additional data on air infiltration. More complete information about home characteristics was obtained by having occupants complete a questionnaire. Energy consumption for each residence was estimated from information about utility costs during the September 1981 to May 1982 heating season. A more detailed description of ventilation and energy-use measurements is available elsewhere (Turner *et al.*, 1983).

RESULTS

Ambient (outdoor) RSP measurements

Particle measurements at the three fixed-location monitoring sites have been discussed in an earlier paper (Sexton *et al.*, 1984a). Findings indicate that there was little spatial variation in particle concentrations between the two in-town stations (Eldredge and Pool). Mean RSP concentrations were 21 $\mu g\,m^{-3}$ at Eldredge and 19 $\mu g\,m^{-3}$ at the Pool and no statistical difference was recorded based on a paired t-test. The average RSP level at the more remote Radio site was 11 $\mu g\,m^{-3}$ and a statistical comparison with the valley floor stations revealed a highly significant ($P < 0.0001$) dissimilarity.

Respirable particle samples were also collected outside all homes in the indoor monitoring program. The location of each residence and the Eldredge, Pool, and Radio sites is shown in Fig. 1. Twenty-two of the 24 homes studied were less than 1 km from either the Eldredge or Pool monitoring stations. Of the two homes outside Waterbury, one (residence P) was approximately 1.6 km north in a cluster of several wood-burning homes, while the other (residence X) was an isolated structure in the midst of open fields about 1.6 km to the east of town.

Data for all outdoor RSP measurements are presented in Tables 1 and 2. Values in parentheses are predicted concentrations using the General Linear Models (GLM) Output option developed by Statistical Analysis Systems (SAS, 1982). This procedure computes expected values in each cell of the matrix using a least-squares regression technique, which takes account of both day and home effects. It is clear that predicted RSP concentrations are in excellent agreement with observed values.

As shown in Tables 1 and 2, there is close agreement among outdoor RSP measurements across both homes and days. The evidence suggests that ambient particle concentrations were relatively uniform within Waterbury during the 1982 Harvard Wood-Burning Study. The slightly elevated levels recorded at residences Q and S are probably due to singular micrometeorological conditions near the monitors. These data also indicate that outdoor respirable particle concentrations at the two homes outside the town (P and X) are not substantially different from levels in Waterbury. Because outdoor values were similar, the community average (mean ambient RSP level for all 12 residences on a particular day) was chosen as the appropriate measure of outdoor RSP at each home.

Indoor RSP measurements

Two in-home RSP samples were collected on each sampling day in each residence. One monitor was located in the same room with the wood-burning appliance (if applicable) and the second was placed in a separate room, but not the kitchen. A comparison of RSP concentrations in the two rooms, using a paired t-test, showed statistically significant differences ($P < 0.05$) in 5 out of the 24 homes. Both residences with kerosene heaters (P and S) had significantly elevated particle concentrations in the room containing the unvented heater. Three dwellings (i.e. C, G, W) had higher values in the room without a wood-burning stove. This could result from differences in monitor placement, effects of in-home mixing patterns, or unidentified emission sources.

Although significant room-to-room differences

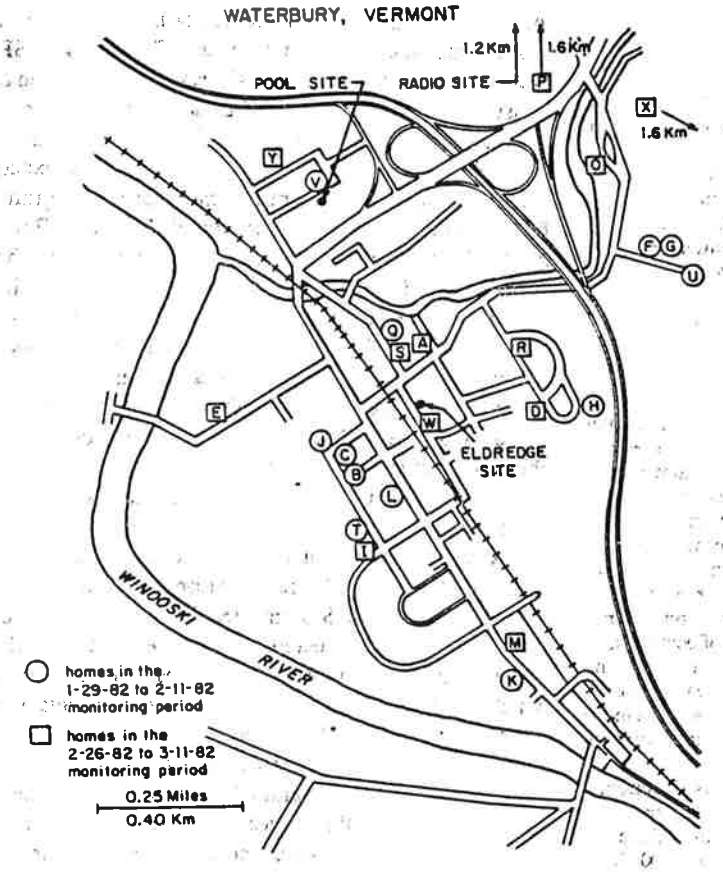

Fig. 1. Map showing location of fixed-site monitoring stations and the 24 homes in the study.

Table 1. Outdoor RSP concentrations ($\mu g\,m^{-3}$) during monitoring period A (29 January–11 February 1982)

Home	Outdoor RSP concentrations ($\mu g\,m^{-3}$) 24-h values							
	29 Jan.	31 Jan.	2 Feb.	4 Feb.	6 Feb.	8 Feb.	10 Feb.	Average
B	—(22)*	8(8)	13(17)	22(16)	15(18)	25(25)	24(24)	18(19)
C	13(20)	9(6)	17(14)	15(14)	15(16)	23(23)	21(22)	16(16)
F	—(20)	—(6)	19(14)	13(14)	18(16)	20(23)	18(22)	18(16)
G	17(20)	6(5)	17(14)	11(14)	15(15)	27(22)	18(22)	16(16)
H	—(17)	2(3)	13(12)	10(11)	15(13)	18(20)	18(19)	13(14)
J	20(20)	9(6)	16(15)	14(14)	11(16)	23(23)	—(22)	16(17)
K	24(20)	—(6)	—(15)	9(14)	18(16)	19(23)	25(22)	19(17)
L	—(17)	—(3)	—(12)	—(12)	14(13)	—(20)	18(19)	16(14)
Q	22(23)	—(9)	—(17)	—(17)	11(19)	26(26)	25(25)	24(19)
T	—(17)	—(3)	—(12)	10(11)	—(13)	20(20)	20(19)	17(14)
U	25(23)	5(8)	12(17)	18(17)	—(18)	26(25)	29(25)	19(19)
V	26(22)	4(8)	13(17)	17(16)	17(16)	—(18)	26(25)	19(19)
Community average	21(20)	6(6)	15(15)	14(14)	15(16)	23(23)	22(22)	17(17)
Range	13–26	2–9	12–19	9–22	14–18	18–27	18–29	13–24
Average of Eldredge and Pool sites for each day	26	9	17	20	19	34	33	23

* Predicted value using linear regression procedures.

Table 2. Outdoor RSP concentrations ($\mu g\,m^{-3}$) during monitoring period B (26 February–11 March 1982)

	Outdoor RSP concentrations ($\mu g\,m^{-3}$) 24-h values							
Home	26 Feb.	28 Feb.	2 Mar.	4 Mar.	6 Mar.	8 Mar.	10 Mar.	Average
A	17(17)*	19(20)	10(10)	12(12)	13(19)	19(17)	31(31)	18(18)
D	16(14)	14(17)	8 (7)	11(10)	20(16)	13(14)	24(28)	15(15)
E	12(14)	18(16)	— (7)	11(9)	16(16)	—(14)	—(28)	15(15)
I	11(15)	18(17)	6 (7)	11(10)	15(16)	16(14)	30(28)	15(15)
M	16(15)	18(17)	2 (7)	14(10)	13(16)	—(14)	30(28)	16(15)
O	—(17)	20(20)	12(10)	11(12)	15(19)	20(17)	31(31)	18(18)
P	—(16)	—(18)	— (9)	—(11)	17(17)	17(15)	28(29)	21(16)
R	10(18)	26(20)	10(10)	13(13)	20(19)	—(17)	32(31)	19(18)
S	36(26)	27(29)	25(19)	14(22)	25(28)	23(26)	—(40)	25(27)
W	16(19)	18(21)	13(12)	16(14)	25(20)	17(18)	31(32)	19(19)
X	—(15)	—(17)	— (8)	8(10)	24(17)	9(15)	—(28)	14(16)
Y	20(17)	17(19)	7(10)	13(12)	14(18)	19(16)	32(30)	17(17)
Community average	17(17)	20(19)	11(10)	13(12)	19(18)	17(16)	30(30)	18(17)
Range	11–36	17–27	2–25	8–16	13–25	9–23	24–32	14–25
Average of Eldredge and Pool sites for each day	15	18	11	13	20	23	28	18

* Predicted value using linear regression procedures.

were observed for homes with unvented kerosene heaters, respirable particles were relatively well mixed inside both wood- and nonwood-burning residences. For the purposes of this paper, indoor RSP concentrations are defined as the mean of both in-home measurements. A linear relationship between the two indoor monitors was used to predict missing values when appropriate. Indoor RSP concentrations for each home by sampling day are given in Tables 3 and 4.

Comparisons

Summary statistics for indoor and outdoor RSP measurements are presented in Table 5. Outdoor ambient concentrations were uniformly low (mean $= 17\,\mu g\,m^{-3}$) and exhibited slight variation (standard deviation $= 6\,\mu g\,m^{-3}$). Indoor values were significantly higher (mean $= 25\,\mu g\,m^{-3}$) and displayed a much broader distribution (standard deviation $= 13\,\mu g\,m^{-3}$). The difference is even more striking when one considers that outdoor RSP varied from 6 to $30\,\mu g\,m^{-3}$, while indoor levels ranged from 6 to $69\,\mu g\,m^{-3}$. The mean excess RSP indoors (indoor–outdoor) across all 24 homes was $8\,\mu g\,m^{-3}$ and the average indoor/outdoor ratio was 1.6.

These data clearly indicate that there are significant particle emission sources in homes where occupants are nonsmokers. Among the potentially important factors are discharges from combustion devices, such as fireplaces, woodstoves, and kerosene-fired heaters. To examine the impact of wood combustion on indoor

Table 3. Indoor RSP concentrations ($\mu g\,m^{-3}$) for each home by sampling day and average indoor, indoor–outdoor, and indoor/outdoor values (29 January–11 February 1982)

	RSP concentrations ($\mu g\,m^{-3}$) 24-h values							Averages		
Home	29 Jan.	31 Jan.	2 Feb.	4 Feb.	6 Feb.	8 Feb.	10 Feb.	Indoor	Indoor–outdoor	Indoor/outdoor
B*	21	28	16	24	17	27	27	23	6	1.7
C*	33	19	57	23	30	36	31	32	16	2.1
F*	9	44	33	10	19	15	21	21	5	1.9
G*	28	19	11	23	25	21	24	21	5	1.5
H*	35	32	10	17	19	18	16	21	4	1.7
J*	18	14	20	21	15	25	19	19	2	1.3
K	13	11	11	10	20	12	15	13	−4	0.9
L*	24	21	19	16	14	18	16	18	1	1.3
Q*	26	26	26	—	26	26	26	26	9	1.9
T*	7	41	12	42	39	51	19	30	14	2.4
U*	13	20	12	18	22	27	17	18	2	1.3
V*	20	19	16	17	18	35	20	20	4	1.4

* Wood-burning appliance.

Table 4. Indoor RSP concentrations ($\mu g\,m^{-3}$) for each home by sampling day and average indoor, indoor–outdoor and indoor/outdoor values (26 February–11 March 1982)

Home	RSP concentrations ($\mu g\,m^{-3}$) 24-h values							Averages		
	26 Feb.	28 Feb.	2 Mar.	4 Mar.	6 Mar.	8 Mar.	10 Mar.	Indoor	Indoor–outdoor	Indoor/outdoor
A*	19	20	13	14	19	19	27	18	1	1.1
D	12	13	10	11	12		23	13	−5	0.8
E*	6	16	11	8	20	23	25	16	−3	0.9
I	63‡	24	11	16	17	14	28	24	6	1.4
M	9	24	10	15	31	32	21	20	2	1.2
O*	19	22	30	22	29	52	28	29	11	1.8
P†	—	—	38	39	42	45	50	43	25	2.7
R	31	63	23	64	59	45	60	49	31	2.9
S†	24	27	24	25	45	33	25	29	11	1.7
W*	48	34	41	58	45	69	49	49	31	3.0
X*	6	13	23	38	26	28	38	24	7	1.5
Y*	27	—	—	30	19	29	27	26	7	1.5

* Wood-burning appliance.
† Both wood- and kerosene-fired appliances.
‡ Elevated value due to cooking fire.

Table 5. Summary statistics for all indoor and outdoor RSP measurements

Parameter	N	$\bar{x}$	S.E.	S.D.	Min.	Max.
Outdoor RSP concentration ($\mu g\,m^{-3}$)	14	17* (16)†	1.6 (1.1)	6 (1.5)	6	30
Indoor RSP concentration ($\mu g\,m^{-3}$)	163	25 (22)	1.0 (1.0)	13 (1.7)	6	69
Indoor–outdoor RSP concentration ($\mu g\,m^{-3}$)	163	8	1.1	14	−14	52
Indoor/outdoor RSP concentration	163	1.6	0.09	1.2	0.3	7.3

* Arithmetic.
† Geometric.
S.E., Standard error.
S.D., Standard deviation.

RSP, the sample population was divided into two categories: wood-burning and nonwood-burning residences.

Mean respirable particle concentrations for the 19 wood-burning and 5 nonwood-burning homes are compared graphically in Fig. 2. Mean values of indoor, indoor–outdoor and indoor/outdoor RSP concentrations for the 2 groups were not statistically different based on t-tests. While the sample size is relatively small, the data indicate that average indoor RSP levels were virtually identical in both sets of residences.

Although average values were alike, nonwood-burning homes exhibited substantially higher variation. The cause of this disparity was Home R (nonwood category), which had the highest mean indoor RSP concentration ($49.3\,\mu g\,m^{-3}$) in the entire sample population. Elevated particle concentrations are at least partially explicable in terms of the 'tightness' of Home R, since estimated seasonal air-exchange rate was 0.4 air changes h^{-1}. While this is one of the lowest ventilation rates among the 24 homes (range 0.3–1.7 air changes h^{-1}), other tight homes did not exhibit similarly elevated levels. Efforts to identify the source of these consistently high values were unsuccessful (e.g. occupants were nonsmokers, cooking stove was electric, oil was used as the space-heating fuel). While other potential indoor sources, such as cooking emissions, vacuuming discharges, and pet dander, could not be ruled out, no specific cause was identified.

Data given in Table 6 provide a comparison of homes by heating-fuel categories. Removing Home R from consideration reduces the mean indoor RSP concentration in the nonwood class from 24 to $18\,\mu g\,m^{-3}$ and the standard deviation from 17.6 to $11.0\,\mu g\,m^{-3}$. Mean values in dwellings with kerosene-fired appliances were $10\,\mu g\,m^{-3}$ higher than those in other wood-burning homes and $17\,\mu g\,m^{-3}$ higher than average values in nonwood residences.

The observed relationship among four categories of residences (i.e. nonwood, except Home R; all nonwood; wood without kerosene; and wood with ker-

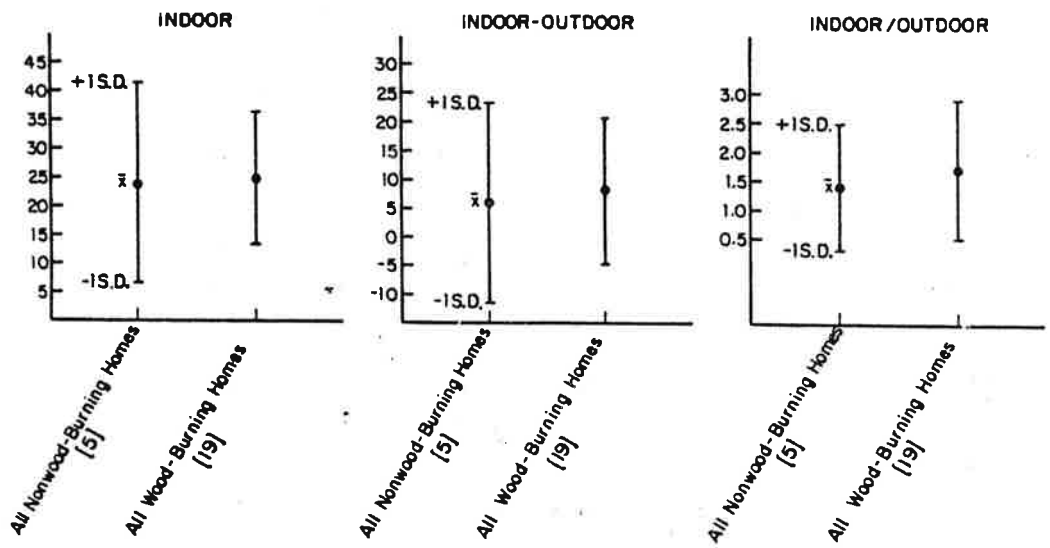

Fig. 2. Comparison of indoor, indoor–outdoor and indoor/outdoor RSP concentrations ($\mu g\,m^{-3}$) for wood-burning and nonwood-burning homes.

osene) is presented graphically in Fig. 3. Neither the wood-burning nor wood-burning plus kerosene houses were significantly different ($P < 0.05$) from the aggregated nonwood-burning homes (including Home R). However, when Home R is removed from consideration, both groups are statistically dissimilar from the 4 remaining nonwood dwellings. It is important to note that even if the wood-burning residence (Home W) with the highest average indoor concentration ($49.0\,\mu g\,m^{-3}$) is excluded, the four nonwood homes (Home R deleted) are still significantly different from the remaining 16 wood-burning homes.

The evidence summarized in Table 6 and Fig. 3 indicates that mean indoor RSP concentrations tend to be elevated in homes with wood-burning appliances, and even more so when unvented kerosene heaters are also used. Average values increased from $18\,\mu g\,m^{-3}$ in four nonwood dwellings (Home R deleted) to $24\,\mu g\,m^{-3}$ in 17 wood-burning homes (no kerosene) to $34\,\mu g\,m^{-3}$ in two residences using both wood and kerosene fuels. Because the mean increase in homes with woodstoves is small in absolute terms ($6\,\mu g\,m^{-3}$), it is difficult to detect given the small sample size (24 homes) and high variability of indoor RSP values. Indeed, if Home R is included in the calculations then no significant difference is apparent.

The problem is illustrated by the fact that the three dwellings with highest mean indoor concentrations

Table 6. Comparison of indoor and outdoor RSP concentrations by heating-fuel type

	RSP concentrations ($\mu g\,m^{-3}$)											
	Indoor				Indoor–Outdoor				Indoor/Outdoor			
	N	$\bar{x}$	S.D.	S.E.	N	$\bar{x}$	S.D.	S.E.	N	$\bar{x}$	S.D.	S.E.
(1) All non-wood burning homes (5)	35	24*(19)†	17.6(1.9)	3.0(1.1)	35	6	17.5	3.0	35	1.4	1.1	0.2
(2) Nonwood-burning homes except R (4)	28	18(16)	11.0(1.6)	2.1(1.1)	28	0	10.9	2.1	28	1.1	0.6	0.1
(3) All wood-burning homes (19)	128	25(23)	11.6(1.6)	1.0(1.0)	128	8	12.5	1.1	128	1.7	1.2	0.1
(4) Wood-burning homes without kerosene (17)	116	24(22)	11.4(1.6)	1.1(1.0)	116	7	12.4	1.2	116	1.7	1.2	0.1
(5) Wood-burning homes with kerosene (2)	12	34(33)	9.6(1.3)	2.8(1.1)	12	17	10.4	3.0	12	2.1	0.8	0.2

* Arithmetic
† Geometric
S.D., Standard deviation.
S.E., Standard error.

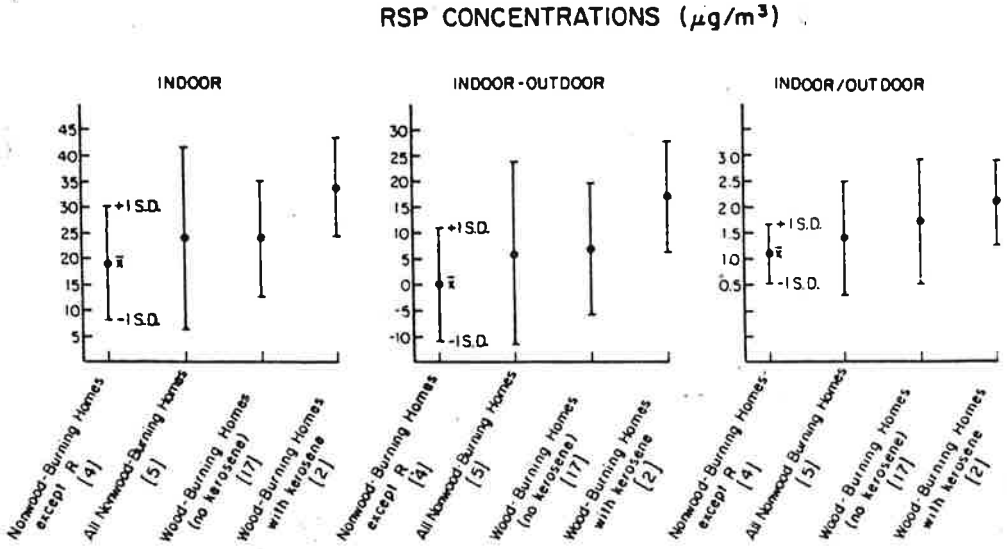

Fig. 3. Comparison of indoor, indoor–outdoor and indoor/outdoor RSP concentrations ($\mu g\,m^{-3}$) for four heating-fuel categories.

came from three different categories: Home R (nonwood) 49.3 $\mu g\,m^{-3}$, Home W (wood only) 49.0 $\mu g\,m^{-3}$, and Home P (wood + kerosene) 42.6 $\mu g\,m^{-3}$. Similarly, one of the three lowest 2-week averages (15.3 $\mu g\,m^{-3}$) was measured in the wood-burning residence (Home E) which had the highest reported wood consumption. Data on air-exchange rate for Home E (1 air change h^{-1}) suggest, that both the high wood consumption and low indoor RSP levels may be due to the 'leakiness' of the structure.

Evaluating potentially important determinants of indoor RSP

The preceding discussion highlighted the fact that within this set of 24 homes, information about type of space-heating fuel (e.g. wood, wood and kerosene, other) is not sufficient to characterize most of the variance associated with in-home RSP concentrations. Given that tobacco use, the suspected major indoor source of particles, had been largely eliminated by selecting only homes with nonsmokers, this raises questions about which other factors are critical for estimating indoor values. To investigate this issue further, a modeling approach was used to evaluate the predictive capability of potentially significant variables.

Intuitively, one would expect several parameters to be important determinants of indoor particle levels: air-exchange rate; outdoor concentration; indoor source strength; removal rate; and mixing volume (National Academy of Science, 1981). While it was not feasible to estimate removal rates, direct measurements were available for outdoor RSP concentrations, air-exchange rates (AER) and house volumes. Fuel use in BTU (e.g. wood, oil, kerosene, gas, electricity) served as a surrogate for source strength. A linear regression model was used to relate these explanatory variables to measured indoor RSP concentrations.

Data on home characteristics and in-home fuel use are summarized in Table 7. Air-exchange rates are average values for the entire heating season calculated from blower door measurements (Turner *et al.*, 1983). Seasonal energy consumption figures were developed from utility bills and homeowner responses to a questionnaire. Measured indoor and outdoor RSP concentrations were presented earlier in Tables 1–4.

Measures of central tendency and variability for each of the parameters in a general multivariate linear regression model are given in Table 8. Results show that there was considerable variation for AER, house volume, and fuel use among homes. Outdoor RSP concentrations were low throughout the monitoring period, with values ranging between 6 and 30 $\mu g\,m^{-3}$. Average indoor RSP concentrations were consistently higher and varied from 13–49 $\mu g\,m^{-3}$. Mean indoor particle values (2-week averages) exceeded corresponding outdoor levels in 21 of the 24 homes studied.

The correlation matrix for model variables is displayed in Table 9. Results of pairwise correlations indicate that covariance among independent variables (i.e. wood, oil, kerosene, gas, electricity, house volume, AER, outdoor concentration) is not a problem. Only 2 sets of comparisons, oil use vs wood use ($r = -0.63$) and outdoor concentrations vs kerosene use ($r = 0.62$), had correlation coefficients (absolute values) greater than 0.50. The negative relationship between wood and oil use is reasonable, since 55% of the homes in Waterbury were heated primarily with oil and only 28% were heated entirely with wood (Sanborn *et al.*, 1981; Sanborn *et al.*, 1982). Therefore, if homeowners did not use wood, they were likely to rely on oil for space-heating fuel. The apparent positive correlation

Table 7. Summary of home characteristics and in-home fuel use by residence

Home i.d.	Home characteristics: House vol. (m^3)	Air exchange† rate (ACH)	1° heating* source	2° heating* fuel	Cooking* fuel	In-home fuel use†: Wood (cords)	Wood (10^6 BTU)	Oil (gal)	Oil (10^6 BTU)	Kerosene (gal)	Kerosene (10^6 BTU)	Propane (CCF)	Propane (10^6 BTU)	Elec. (kWh)	Elec. (10^6 BTU)	Total (10^6 BTU)	Total BTU $dd^{-1} m^{-2}$
A	326	0.8	wood stove	oil	elec.	4.2	90	102	14	0	0	0	0	10440	36	140	147
B	309	1.0	wood stove	oil	elec.	3.5	75	361	50	0	0	0	0	13210	45	170	180
C	548	0.6	wood stove	elec.	elec.	4.7	101	0	0	0	0	0	0	27016	92	193	177
D	467	0.7	oil furnace	oil	elec.	0.0	0	1243	170	0	0	0	0	3810	13	183	180
E	507	1.0	wood stove	oil	elec.	8.0	172	250	34	0	0	0	0	12360	42	242	152
F	187	0.5	wood stove	oil	elec.	3.5	75	50	7	0	0	0	0	7080	24	106	158
G	187	1.0	wood stove	oil	elec.	3.5	75	270	37	0	0	0	0	7780	27	139	224
H	359	0.4	wood house	oil	elec.	7.5	161	330	45	0	0	0	0	5470	19	225	189
I	692	0.3	oil furnace	oil	elec.	0.0	0	764	105	0	0	0	0	7960	27	132	92
J	404	0.9	wood furnace	wood	elec.	6.4	138	0	0	0	0	228	21	6640	27	181	144
K	397	1.7	oil furnace	oil	elec.	0.0	0	1609	220	0	0	0	0	15320	52	273	213
L	244	1.2	wood stove	gas	elec.	3.5	75	0	0	0	0	200	18	2680	9	103	123
M	227	0.7	gas furnace	gas	gas	0.0	0	0	0	0	0	1178	108	3240	11	119	155
N	290	—	elec. heaters	—		0.0	0	0	0	0	0	0	0	10329	35	35	53
O	402	0.6	wood stove	oil	elec.	6.9	148	525	72	0	0	0	0	6860	23	244	195
P	303	0.9	wood stove	kero.	elec.	5.8	125	0	0	65	9	0	0	6440	22	156	209
Q	323	1.1	wood stove	oil	gas	5.0	108	155	21	0	0	30	3	8030	27	159	142
R	424	0.4	oil furnace	oil	elec.	0.0	0	861	118	0	0	0	0	2750	9	127	93
S	411	1.1	wood stove	oil	elec.	6.5	140	0	0	100	14	30	3	10250	35	191	164
T	504	0.6	wood stove	oil	elec.	2.5	54	793	109	0	0	0	0	6870	24	186	143
U	331	0.9	oil furnace	wood	elec.	3.5	75	731	100	0	0	0	0	9630	33	208	207
V	303	0.5	wood stove	oil	gas	4.5	97	108	14	0	0	147	14	2520	9	134	128
W	320	0.9	wood stove	—	elec.	6.0	129	0	0	0	0	30	3	5240	18	150	177
X	229	0.6	wood stove	—	gas	4.0	86	0	0	0	0	224	21	5500	19	125	207
Y	340	0.9	oil furnace	oil	wood	1.0	2	843	116	0	0	309	28	1940	7	172	157

* Information obtained by field personnel during in-home survey.
† Predicted seasonal AER from blower door measurements (ACH = air change h^{-1}).
‡Estimated seasonal fuel use from utility bills.
BTU $dd^{-1} m^{-2}$ = BTU degree day^{-1} $meter^{-2}$.

Table 8. Measures of central tendency and variability for model parameters

Variable	N	$\bar{x}$	S.E.	Min.	Max.
Air-exchange rate (ACH)	24	0.8	0.06	0.3	1.7
Home volume (m^3)	24	368	24.6	187	692
Wood use (10^6 BTU)	19	101	9.5	2	172
Oil use (10^6 BTU)	16	77	15.2	7	220
Kerosene use (10^6 BTU)	2	11	2.4	9	14
Gas use (10^6 BTU)	9	24	10.9	3	108
Electricity use (10^6 BTU)	24	27	3.7	7	92
Outdoor RSP conc. ($\mu g\,m^{-3}$)	24	17	0.6	6	30
Indoor RSP conc. ($\mu g\,m^{-3}$)	24	25	2.0	13	49

S. E., Standard error.
ACH = air change h^{-1}.

Table 9. Correlation matrix for model parameters

	Oil	Kerosene	Gas	Electricity	House volume	AER	Outdoor Conc.	Indoor Conc.
Wood	−0.63* (0.001)†	0.29 (0.17)	−0.31 (0.14)	0.20 (0.34)	−0.04 (0.67)	0.30 (0.87)	0.30 (0.15)	0.09 (0.68)
Oil		−0.25 (0.24)	−0.24 (0.26)	−0.02 (0.93)	0.41 (0.05)	0.15 (0.48)	−0.28 (0.19)	−0.19 (0.37)
Kerosene			−0.10 (0.65)	0.05 (0.83)	0.01 (0.96)	0.21 (0.33)	0.62 (0.001)	0.25 (0.19)
Gas				−0.32 (0.13)	−0.33 (0.12)	−0.04 (0.87)	−0.14 (0.51)	−0.15 (0.49)
Electricity					0.38 (0.07)	0.24 (0.26)	0.05 (0.83)	−0.09 (0.69)
House volume						−0.24 (0.27)	−0.08 (0.72)	0.07 (0.74)
AER							0.12 (0.56)	−0.27 (0.20)
Outdoor Conc.								0.27 (0.20)

* Pearson correlation coefficient.
† Probability.
AER = air-exchange rate.

between outdoor RSP and kerosene use is an artifact of the relatively high ambient levels measured outside Home S, one of the 2 residences using kerosene heaters.

The last column in Table 9 shows the correlation of indoor RSP with each of the explanatory variables. It is obvious that no single parameter exhibits a strong relationship with measured indoor values. Highest correlations were observed for outdoor RSP concentrations ($r = 0.27$), air-exchange rate ($r = -0.27$) and kerosene use ($r = 0.25$). Indoor particle levels were found to be unrelated to amount of wood burned and house volume. As expected, homes using primarily oil, gas, or electricity for space-heating had negative correlation coefficients, suggesting that they tended to have lower in-home RSP values.

Least-squares regression models relating indoor RSP to home characteristics, fuel use, and outdoor RSP are listed in Table 10. Best 1–4 variable models were selected using a stepwise regression procedure (SAS, 1982). The adjusted r^2 (corrected coefficient of determination) values provide a means of comparing regression equations which vary with respect to the number of explanatory variables (Kmenta, 1971). The lower adjusted r^2 observed for the 3 and 4 variable models suggests that the loss of additional degrees of freedom is not worth the marginal gain in unadjusted r^2. Thus air-exchange rate and kerosene use are the best predictors of indoor RSP concentrations included in the model. The most striking feature of these data, however, is the fact that it was possible to account for only about 20% of the variation in indoor concentrations.

Although the predictive capability of the model is low, computed regression coefficients provide some clues concerning the influence of underlying physical processes. For instance, these data indicate that an increase in the air exchange rate is associated with reduced indoor RSP concentrations, while the use of kerosene as a heating fuel is associated with increased indoor RSP. Contrary to expectations, outdoor particle values were not found to be a strong determinant of indoor concentrations. This is probably due to the fact that ambient RSP levels (24-h averages) exhibited relatively little variation during the study. The con-

Table 10. Best-fitting least-squares regression models for indoor RSP concentrations ($\mu g\,m^{-3}$)

Model	Terms	Coefficients	Std. error	F	$P > F$	r^2	Adjusted r^2
Best 1 variable model	Intercept	24	—	—	—	0.08	0.04
	Kerosene use	0.8	0.6	1.80	0.2		
Best 2 variable model*	Intercept	33	—	—	—		
	Air-exchange rate	−11	6.4	2.93	0.10	0.19	0.11
	Kerosene use	1	0.6	2.97	0.10		
Best 3 variable model	Intercept	33	—	—	—		
	Air-exchange rate	−11	6.5	2.88	0.11		
	Kerosene use	1	0.6	2.67	0.12	0.21	0.09
	Gas use	−0.06	0.1	0.41	0.53		
Best 4 variable model	Intercept	25	—	—	—		
	Air-exchange rate	−11	6.6	2.77	0.11		
	Kerosene use	0.8	0.8	0.89	0.36		
	Outdoor RSP conc.	0.5	0.9	0.30	0.59	0.22	0.06
	Gas use	−0.05	0.1	0.32	0.58		

* Best model based on adjusted r^2 value.

sistently large intercept term (25–35 $\mu g\,m^{-3}$) suggests that one or more important indoor emissions sources are not accounted for in the model, though some portion of this amount is certainly due to measurement error.

DISCUSSION

Previous studies

Information concerning effects of residential wood combustion on indoor air quality is limited due to a lack of measurements. Studies conducted to date have been designed to provide evidence about general air pollution levels in a few wood-burning homes, selected mainly for their proximity and accessibility. Because no systematic examination of this issue has been attempted, available evidence is fragmented, uneven in quality, and often contradictory. Findings from published studies are summarized in Table 11.

These data suggest that indoor concentrations of particles, benzo(a)pyrene (BAP), hydrocarbons, carbon monoxide (CO), nitric oxide (NO), nitrogen dioxide (NO_2), and sulfur dioxide (SO_2) can increase during operation of wood-burning appliances. However, elevated indoor levels of air pollution were not a consistent finding across all wood-burning homes. This is not surprising given the small sample sizes ($N \leq 8$) and the expected home-to-home variation in wood use, operating and maintenance procedures, air-exchange rates and mixing volumes.

Harvard 1982 indoor/outdoor monitoring study

Indoor RSP concentrations were relatively well mixed within homes, although higher values were observed in rooms with unvented kerosene heaters. Indoor concentrations were greater than corresponding outdoor levels for 65% of 163 paired indoor–outdoor samples. Twenty-three of the 24 houses in the study had at least one 24-h period when indoor particle levels exceeded those outdoors. These data highlight the ubiquitous nature of indoor RSP sources, even in homes where occupants are nonsmokers.

Substantial variability in indoor RSP levels was observed among homes within each of the heating-fuel categories (i.e. nonwood, wood, wood and kerosene). Concentrations of respirable particles tended to be higher in wood-burning, as opposed to nonwood-burning residences. However, observed differences were not statistically significant unless 1 nonwood home, with elevated indoor concentrations of unknown origin, was deleted from the analysis. Indoor values were significantly higher in rooms with kerosene heaters.

Several investigators have shown that tobacco smoke is a major contributor to indoor particle concentrations (Spengler *et al.*, 1980; Dockery and Spengler, 1981; Spengler *et al.*, 1981; Spengler and Tosteson, 1981; Kinney *et al.*, 1982; Spengler *et al.*, 1982). To eliminate this source, only homes with nonsmoking occupants were included in the study. Measured RSP concentrations inside Waterbury residences were comparable to indoor levels reported for other nonsmoking homes in Topeka, KA (Spengler *et al.*, 1980; Spengler and Tosteson, 1981), Kingston-Harriman, TN (Spengler *et al.*, 1982), Boston, MA (Ju and Spengler, 1981), and six cities in the Harvard Air Quality/Lung Health Study (Dockery and Spengler, 1981; Spengler *et al.*, 1981; Kinney *et al.*, 1982).

The fact that indoor RSP concentrations in nonsmoking homes consistently exceed outdoor levels suggests the presence of other indoor emission sources. Data presented here indicate that wood- and kerosene-fired appliances can contribute to elevated indoor values, but are not the only source of RSP inside the 24 residences in this study. Because particles can be generated by combustion, condensation, abrasion, and resuspension, there are many potentially significant indoor emission sources. Among the more obvious are

Table 11. Description of previous studies which examine effects of residential wood combustion on indoor air quality

Reference(s)	Location	Number of homes	Type of wood-burning device(s)	Measurements	Findings
Moschandreas *et al.* (1980)	Boston, MA	3	Woodstove and fireplaces	NO, NO_2, CO_2, CO, O_3, SO_2, CH_4, THC, SO_4, NO_3, BaP, TSP, RSP, selected elements, air infiltration rates	Average indoor TSP values during wood-burning periods were 3 times nonwood-burning periods. 24-h NAAQS for TSP was exceeded once indoors during fireplace use and the secondary standard was exceeded by RSP alone. Indoor BaP concentrations during woodstove use averaged 5 times those measured during nonwood-burning periods.
Colome and Spengler (1982) Colome *et al.* (1981)	Steuvenville, OH and Portage, WI	10 (two with woodstoves)	Woodstoves	SO_2, NO_2, SO_4, RSP, selected elements	Outdoor measurements were consistent with proximity to outdoor sources. Indoor levels of SO_2, SO_4, Mn and V were lower than those measured outdoors. NO_2 concentrations were elevated in some gas-cooking homes, RSP levels were high in homes with smokers. Elevated indoor levels of Al, Br, Cl and Na did not follow any discernible pattern. One home with a woodstove and a gas-cooking stove had the highest NO_2 concentration recorded in Portage.
Benton *et al.* (1981)	Flint Hills, KS	5	Woodstoves, fireplace, wood-cooking stove, and a wood furnace	Hydrocarbons, RSP	Wood-burning devices caused indoor RSP levels to be higher than in nonwood-burning homes. Chromatograms for each home had unique features that were consistent from week to week. Most hydrocarbon peaks were unidentified; however, the presence of benzene, toluene, *p*-xylene, cumene isomers and cymene isomers was recorded.
Neulicht and Core (1982)	Portland, OR	5	Woodstoves	PNAs, particulate mass, Pb	No significant increase in the concentration of particulate matter and PNAs in 4 of 5 homes evaluated during wood-burning activities. A significant increase in both particulate matter and PNAs was recorded during wood-burning in 1 of 5 homes monitored.
Traynor *et al.* (1982)	Berkeley, CA	3	Woodstoves and wood furnaces	CO, NO, NO_2, SO_2, air-exchange rates	Levels of CO, NO, NO_2 and SO_2 increased during wood-burning periods, although concentrations were generally below occupational and outdoor air quality standards.
McGill and Miller (1982)	Topeka, KS	8	Woodstoves and fireplaces	Volatile and semi-volatile organics	Thermograms were reasonably characteristic of individual wood-burning appliances.

cooking, vacuum cleaning, sweeping, dusting, aerosol spray products, pets, and other activities within a house.

One potential source of error which deserves further consideration is the uncertainty associated with AER determination. Seasonal air-exchange rates were developed from one-time blower door measurements. Because ventilation rates are influenced by diurnal, seasonal, and regional factors as well as local meteorological conditions, there is obviously a wide margin for error. In addition, continuous measurements of infiltration rates have shown that opening doors, particularly for extended periods, can cause short-term AER to increase from less than 1 ACH to more than 3 ACH (Lamb *et al.*, 1982). Homes in Waterbury are expected to be relatively 'tight' during winter months, but occupant activities might have considerable effect on short-term AER. This in turn could have a substantial impact on measured indoor RSP concentrations.

CONCLUSIONS

Measurements inside and outside 24 homes in Waterbury, Vermont, indicate that RSP concentrations are frequently elevated indoors, even in the absence of tobacco smoke. No statistical difference was observed between indoor particle values in homes that did not burn wood and those that did. These findings suggest that airtight woodstoves can be installed, operated, and maintained in such a way so that direct release of particles to the indoor environment is negligible. The complexity of modeling indoor RSP concentrations is underscored by the fact that even when data on outdoor particle values, heating-fuel use, home characteristics, and air-exchange rates are included in the model, only about 20% of the variance in indoor RSP levels can be explained. The evidence suggests that particle concentrations in residential indoor environments (nonsmoking occupants) vary widely from home to home. Future studies should investigate the sources of these inter-home differences and determine to what degree indoor RSP concentrations are building- and occupant-specific.

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