

Indoor Air Quality of Acidic Air Pollutants at a Private House in Seoul During the Spring Months

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Abstract

Acidic air pollutants were collected to characterize indoor air quality at a private house in Seoul during the spring period. All indoor and outdoor samples were measured simultaneously using an annular denuder system. The data set was collected on twelve different days with a 24-hr sampling period in April and May 1997. The chemical species measured were HNO₃, HNO₂, SO₂ and NH₃ in the gas phase and PM_{2.5} (dp ≤ 2.5 μm), SO₄²⁻, NO₃⁻ and NH₄⁺ in the particulate phase. Indoor concentrations of HNO₂, NH₃, and PM_{2.5} were greater than outdoor levels. However, indoor concentrations of HNO₃, SO₂, NO₃⁻ and NH₄⁺ were less than those found from outdoors. In the case of SO₄²⁻, the indoor and outdoor concentrations were similar. Indoor concentrations of PM_{2.5}, SO₄²⁻ and NO₃⁻ were dependent upon the outdoor concentrations. A tracer-gas decay technique with sulfur hexafluoride (SF₆) as the tracer gas was used to estimate the air exchange rate of a private home in the spring. The average air exchange rate was computed to be 2.87 hr⁻¹.

Key words : Indoor and outdoor acidic air pollutants, Indoor/outdoor ratio, Air exchange rate, A private house, Seoul

1. INTRODUCTION

Seoul is well known for its high population density among the world's major cities. It is estimated that 25% (11 million) of South Korea's total population and 23% (2.24 million) of its total vehicles are placed in Seoul. However, its area is only 0.6% (605 km²) of South Korea. The decrease in outdoor air quality of this large city is easy to detect. Lee *et al.* (1999) have

shown that the levels of acidic air pollutants in Seoul are relatively high among other cities in the world. Deterioration of outdoor pollution levels in urban areas due to vehicular emissions, street dust, and industrial sources may have many impacts on the indoor air quality. However, many studies focused on outdoor air quality in South Korea. Most people spend 80~90% of their time indoors. Thus, the risks to health may be greater due to exposure to air pollution indoors than outdoors, especially in the case of acidic pollutants. Some individuals (e.g., the old, the very young and the infirm) who are most susceptible to the effects of pol-

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lutants may spend all their time indoors. However, there have been only a few studies addressing the issues of indoor air quality in South Korea (Baek *et al.*, 1997; Lee *et al.*, 1997). Indoor pollutants such as acidic gases and aerosols have not been studied in South Korea. The composition of the atmosphere (oxygen and nitrogen) is basically the same indoors and outdoors; however, pollutant concentrations will alter with time and space. The nature and quantities of indoor pollutants may differ considerably from those outdoors (Phillips *et al.*, 1993). Indoor pollution levels will be determined by a number of factors: (i) the air exchange rate between the indoor and outdoor environment; (ii) the outdoor air quality; (iii) any indoor sources of pollutants; and (iv) the removal of pollutants from the room by methods other than ventilation, e.g., adsorption of pollutants to surfaces (Phillips *et al.*, 1993; Wadden and Scheff, 1983).

Atmospheric releases of acidic pollutants can cause adverse health effects and they have the potential to cause other environmental damage (e.g., acid rain). Ambient fine particles (PM_{2.5}) and sulfate (SO₄²⁻) exposures are specifically responsible for the observed associations with daily mortality. This suggests that toxicological studies of mechanisms and control strategies to promote public health should focus on fine particles, which are produced by direct emissions and secondary reactions of combustion-related air pollutants (Schwartz *et al.*, 1996; Spengler *et al.*, 1990). PM_{2.5} showed a closer association with increased mortality than did PM₁₀ (Reichhardt, 1995).

The objectives of this study were: (i) to provide information on indoor air quality of acidic air pollutants in a residential area of Seoul; (ii) to determine quantitatively the relationship between indoor and outdoor air; and (iii) to estimate the air exchange rate between the indoor and outdoor environment. The chemical species measured were nitric acid (HNO₃), nitrous acid (HNO₂), sulfur dioxide (SO₂) and ammonia (NH₃) in the gas phase, and fine particles (PM_{2.5}), sulfate (SO₄²⁻), nitrate (NO₃⁻) and ammonium (NH₄⁺) in the particulate phase.

2. METHODS

A private house, located in Seoul, the capital of South Korea, was used to collect acidic air pollutants in a residential area. The data set was collected on twelve different days with a 24-hr sampling period (from 8:00 a.m. to 8:00 a.m. the next day) using an annular denuder system (ADS) in the spring (3 April to 16 May 1997). A detailed characterization of the selected house is included in Table 1. Indoor samples were collected on the first floor (volume = 69.2 m³) in the living room. The room for our investigation had dimensions of 9.18 m × 3.35 m × 2.25 m (L × W × H). The ADS, so as not to bother the residents, was placed about 1 m from a window with the inlet roughly 1.5 m above floor level. There were five family members, three of whom were light cigarette smokers. Except for the wife, the others would go to the workplace in the morning and return to the house in the evening. The cooking and heating fuels were liquefied natural gas (LNG), which is known as clean fuel. The weather at sampling was generally mild and sunny during the daytime, and consequently the windows were often open. At night, the windows were closed due to the chilly. Outdoor samples were simultaneously collected using the ADS on the roof of the Konkuk University building (15 m above ground level).

Table 1. Sampling site descriptions.

Building description	three stories
Sampling room size (m ³)	69.2
Sampling floor size (m ²)	30.8
Floor covering	hard vinyl
Wall covering	wallpaper
Ceiling covering	wood
Number of windows	2
Number of doors	4
Ventilation	natural
Number of occupants	5
Number of smokers	3
Smoking rates per person	3 to 5 cigarettes per day
Plants	yes
Cooking fuel	LNG ^a
Heating type (fuel)	Central (LNG ^a)

^a Liquefied natural gas.

near the indoor sampling site. A detailed description of the outdoor sampling site was reported elsewhere (Lee *et al.*, 1999).

The ADS (University Research Glassware) consists of: (1) an inlet with an impactor or cyclone preseparator designed to remove all particles with a d_{50} of 2.5 μm or greater, (2) annular denuders to quantify acidic and basic gases, and (3) a filter pack for particles (U.S.EPA, 1997). Flow rates used at indoor and outdoor sampling were 4 L/min through an impactor and 10 L/min through a cyclone to remove coarse particles. The first denuder was coated with sodium chloride (NaCl) to collect HNO_3 and SO_2 . The second denuder was coated with sodium carbonate (Na_2CO_3) to collect HNO_2 and SO_2 . The third denuder was coated with citric acid to collect NH_3 . The particulates and their components ($\text{PM}_{2.5}$, SO_4^{2-} , NO_3^- and NH_4^+) were collected on the Teflon filter (1 μm pore size; Gelman Science). The Nylasorb filter (1 μm pore size; Gelman Science) was used to collect any particulate NO_3^- that evaporated from the Teflon filter. Since the Teflon filter is hydrophobic, 100 μl of ethanol was added to wet the filter before extraction. All filters were extracted with 10 mL of double-distilled water (DDW; Barnstead). Ion chromatography (Dionex DX-100) was used for the analysis of anion and cation concentrations from the filter extracts. The Dionex system was equipped with an analytical column preceded by a guard column. The anion and cation columns used were an IonPac AS9-SC (4 mm) and an IonPac CS12 (4 mm), respectively. Samples were injected via 25 μl loop into an eluent. The eluent used for anion analysis was a 2.0 mM carbonate (Na_2CO_3)/0.75 mM bicarbonate (NaHCO_3) solution. The eluent for cation analysis was a 20 mM methanesulfonic acid solution. Pump flow rates for the analysis of anion and cation were regulated to 2 and 1 mL/min, respectively. Limits of detection for SO_4^{2-} , NO_3^- and NH_4^+ , based on the sensitivity of ion chromatography and a 24-hr sample collected at a flow rate of 10 L/min, were 0.04, 0.02 and 0.03 $\mu\text{g}/\text{m}^3$, respectively. To determine $\text{PM}_{2.5}$ mass by gravimetric analysis using a microbalance (Cahn C-35), which is sensitive to changes as small as

0.1 μg , all Teflon filters were pre- and post-weighed. The net weights were corrected with three control filters not exposed to sampling. All Teflon filters were conditioned before and after sampling in a clean chamber (Nikko auto dry desiccator) with temperature $< 35^\circ\text{C}$ and humidity $< 50\%$ for at least 24-hr. The collection efficiencies of HNO_3 , HNO_2 , SO_2 and NH_3 were greater than 98%. The relative standard errors of SO_4^{2-} , NO_3^- and NH_4^+ analyzed using ion chromatography were within 5%.

3. RESULTS AND DISCUSSION

The indoor and outdoor samples were simultaneously collected in the spring of 1997 at a private house. Table 2 shows the summary of the results of this study. Outdoor concentrations in Table 2 were from elsewhere (Lee *et al.*, 1999). The mean concentrations of indoor pollutants were: 1.02 $\mu\text{g}/\text{m}^3$ for HNO_3 , 7.07 $\mu\text{g}/\text{m}^3$ for HNO_2 , 4.02 $\mu\text{g}/\text{m}^3$ for SO_2 , 11.0 $\mu\text{g}/\text{m}^3$ for NH_3 , 64.3 $\mu\text{g}/\text{m}^3$ for $\text{PM}_{2.5}$, 9.78 $\mu\text{g}/\text{m}^3$ for SO_4^{2-} , 3.79 $\mu\text{g}/\text{m}^3$ for NO_3^- and 2.93 $\mu\text{g}/\text{m}^3$ for NH_4^+ . Table 3 represents the mean indoor/outdoor concentration ratios (I/O ratio). The species with an I/O ratio greater than 1.00 were: HNO_2 , NH_3 and $\text{PM}_{2.5}$. The species with an I/O ratio less than 1.00 were: HNO_3 , SO_2 , NO_3^- and NH_4^+ . The indoor and outdoor SO_4^{2-} concentrations were similar. In the case of the Brauer *et al.* (1991) study, the private house sampled in the Boston area during the summer, these ratios were: 0.60 for HNO_3 , 2.66 for HNO_2 , 0.39 for SO_2 , 3.48 for NH_3 , 0.96 for SO_4^{2-} , 0.81 for NO_3^- and 1.90 for NH_4^+ . Table 4 shows the linear regression equations of the indoor concentrations (dependent variable) compared with the corresponding outdoor levels. The species that had non-significant regression coefficients ($p > 0.01$) among the species measured were not included in Table 4.

3.1 Indoor and outdoor concentrations

Indoor air quality will respond to changes in the outdoor air quality; but the rate of response and final indoor

Table 2. Summary statistics for indoor and outdoor concentrations.

Species	Location	Concentration ($\mu\text{g}/\text{m}^3$)			N ^b
		Mean	SD ^a	Range	
HNO ₃	Indoor	1.02	0.54	0.40~2.09	12
	Outdoor	1.17	0.69	0.39~2.32	12
HNO ₂	Indoor	7.07	3.07	2.31~13.3	12
	Outdoor	6.22	2.39	2.94~11.7	12
SO ₂	Indoor	4.02	2.31	0.57~7.83	12
	Outdoor	22.7	10.4	3.21~34.8	12
NH ₃	Indoor	11.0	2.26	6.45~14.2	12
	Outdoor	4.57	1.79	2.23~8.78	12
PM _{2.5}	Indoor	64.3	24.6	19.1~106.5	12
	Outdoor	55.9	24.3	10.1~85.5	12
SO ₄ ²⁻	Indoor	9.78	9.34	2.16~37.1	12
	Outdoor	9.75	7.94	0.29~31.1	12
NO ₃ ⁻	Indoor	3.79	2.42	0.94~9.05	12
	Outdoor	7.23	4.71	0.84~16.2	12
NH ₄ ⁺	Indoor	2.93	1.70	0.82~5.94	12
	Outdoor	4.37	2.81	1.30~9.82	12

^a Standard deviation.^b Number of samples.

pollutant concentrations will depend on factors such as the permeability of the structure, the stability of the pollutant and indoor activities (Phillips *et al.*, 1993). Indoor air quality in a naturally ventilated house is linked closely with that outdoors. The indoor/outdoor ratio of a pollutant reflects not only the penetrating ability of the pollutant but also its decay in the indoor environment. The penetration ability is a function of a number of parameters such as the nature of the pollutant and mechanisms of the air exchange. The indoor decay rate of pollutants originating from the outdoor sources depends on each pollutant's physio-chemical properties and the characteristics, such as the house surface and the air exchange (Koutrakis *et al.*, 1992).

The indoor/outdoor ratio of HNO₃ was 0.87 (indoor and outdoor concentrations are 1.02 and 1.17 $\mu\text{g}/\text{m}^3$, respectively) (Tables 2 and 3). The low indoor concentration of HNO₃ may be due to the reaction with the high NH₃ and high HNO₃ reactivity. Nitric acid has been reported to have high solubility (Seinfeld, 1986) and was scavenged by interior surfaces at a relatively

Table 3. Indoor/outdoor concentration ratios.

Species	Indoor/outdoor ratios
HNO ₃	0.87
HNO ₂	1.14
SO ₂	0.18
NH ₃	2.41
PM _{2.5}	1.15
SO ₄ ²⁻	1.00
NO ₃ ⁻	0.52
NH ₄ ⁺	0.67

fast rate (Weschler *et al.*, 1992). Brauer *et al.* (1991) reported that the indoor/outdoor ratio of HNO₃ was 0.60, which is less than that of this study. This may be due to the different air exchange rates and interior materials at private houses between the Seoul and Boston areas.

Indoor HNO₂ production results from the reaction of NO_x which has penetrated into the indoor air from outdoors. The chemical mechanism of this process, it is suggested, involves a heterogeneous reaction between NO₂ and H₂O (Weschler and Shields, 1997; Brauer *et al.*, 1990; Jenkin *et al.*, 1988). Brauer *et al.* (1991) observed that indoor HNO₂ concentrations were greater than those outdoors even when indoor NO₂ concentrations do not substantially exceed those outdoor levels. Spengler *et al.* (1993) who monitored at houses in New Mexico, also reported that indoor concentrations of HNO₂ exceeded outdoors. Nitrous acid in the atmosphere is rapidly photolyzed during the day, resulting in the generation of OH radicals (Kitto and Harrison, 1992). As might be expected, the indoor HNO₂ concentration (7.07 $\mu\text{g}/\text{m}^3$) was greater than the outdoor value (6.22 $\mu\text{g}/\text{m}^3$) (Table 2).

The I/O ratio of SO₂ was the lowest (0.18) compared with the other species (Table 3). This result shows that the effective SO₂ penetration indoors is quite low, suggesting deposition of the gas on surfaces. Another possible explanation for the decreased indoor SO₂ levels is that the higher indoor NH₃ concentrations, which increase with the SO₂ oxidation rate, would serve to reduce indoor concentrations of SO₂ relative to outdoors (Genfa *et al.*, 1989). The indoor and outdoor concentrations of SO₂ were relatively correlated (p-value of

probability of no correlation = 0.06, $R^2 = 0.56$). If there were no indoor sources, one would expect a significantly smaller I/O ratio for HNO_3 than for SO_2 , since HNO_3 is scavenged by indoor surfaces several times faster than SO_2 (Weschler and Shields, 1997; Weschler *et al.*, 1992; Salmon *et al.*, 1990). This reflects the fact that HNO_3 is far more soluble in water than SO_2 (Finlayson and Pitts, 1986). From this study, the I/O ratio of HNO_3 (0.87) was much higher than that of SO_2 (0.18), suggesting that HNO_3 may be formed indoors. Weschler *et al.* (1992) reported that ozone may react with nitrogen dioxide with the indoor air infiltrated from outdoors to generate nitric acid. During the sampling periods, the mean outdoor concentrations of ozone and nitrogen dioxide were 28.2 and 48.5 ppb, respectively. These values may be enough for HNO_3 production indoors.

Ammonia emission indoors may be attributed to home factors such as: the building materials, cleaning fluids, cigarette smoke (especially sidestream) and human exhaled breath (Brauer *et al.*, 1991; Wadden and Scheff, 1983). Wadden and Scheff (1983) also reported that ammonia emissions are 32.2 mg/day per person. NH_3 concentrations were higher indoors ($11.0 \mu\text{g}/\text{m}^3$) than outdoors ($4.57 \mu\text{g}/\text{m}^3$) (Table 2). The I/O ratio of NH_3 was the highest (2.41) among the species measured (Table 3). This may be due to house factors. Brauer *et al.* (1991) also found higher indoor NH_3 levels (the I/O ratio of NH_3 was 3.48).

Indoor $\text{PM}_{2.5}$ concentrations ($64.3 \mu\text{g}/\text{m}^3$) were greater than outdoors ($55.9 \mu\text{g}/\text{m}^3$) (Table 2), which had an I/O ratio of 1.15 (Table 3). As in the case of NH_3 , the $\text{PM}_{2.5}$ mass may have been emitted from human activities, such as cigarette smoke and fuel usage in the indoor environment in addition to the infiltration of outdoor air. There were five family members; three among them are cigarette smokers. The indoor and outdoor concentrations of $\text{PM}_{2.5}$ were well correlated (p-value of probability of no correlation < 0.01, $R^2 = 0.84$) (Table 4). Sheldon *et al.* (1989) studied at 433 homes during the winter in two New York State counties, Onondaga and Suffolk, reported that mean indoor $\text{PM}_{2.5}$ concen-

trations were 36.7 and $46.7 \mu\text{g}/\text{m}^3$, respectively, and mean outdoor $\text{PM}_{2.5}$ concentrations were 16.8 and $21.8 \mu\text{g}/\text{m}^3$, respectively.

The indoor/outdoor ratio of SO_4^{2-} was 1.00 (Table 3). The Brauer *et al.* study (1991) found a similar value (I/O ratio of 0.96). The high indoor/outdoor SO_4^{2-} ratio is likely to result from a combination of penetration and indoor production of SO_4^{2-} . Some SO_4^{2-} may be emitted in combustion itself from oxidation of mercaptan tracer compounds in natural gas (Dockery and Spengler, 1981). Indoor SO_4^{2-} concentration is significantly associated with the outdoor concentration (p-value of probability of no correlation < 0.01, $R^2 = 0.98$) (Table 4).

The indoor/outdoor NO_3^- ratio (I/O ratio of 0.52) was much less than unity (Table 3). However, there was a high correlation between indoor and outdoor concentrations of NO_3^- (p-value of probability of no correlation < 0.01, $R^2 = 0.88$) (Table 4). As in the case of NO_3^- , the indoor concentration of NH_4^+ ($2.93 \mu\text{g}/\text{m}^3$) was much less than that outdoor ($4.37 \mu\text{g}/\text{m}^3$) (Table 2). Lower indoor NO_3^- and NH_4^+ values may be explained by a shift of the equilibrium from particulate phase ammonium nitrate (NH_4NO_3) to ammonia and nitric acid in warmer indoor environments (Lee *et al.*, 1999; Seinfeld, 1986; Stelson and Seinfeld, 1982).

3. 2 Estimation of air exchange rate

One of the factors affecting indoor air quality is outdoor air pollutants, which can be infiltrated from cracks and opened windows. The air exchange rate between the indoor and outdoor environment, therefore, was estimated by a tracer decay technique using the decay of sulfur hexafluoride (SF_6) concentration. SF_6 is an unreactive substance: not attacked by water, acids or bases at room temperature. Because the air exchange rate should be different between the daytime, in which the windows are open, and the nighttime, in which the windows are closed, SF_6 was released in the day and at night at the indoor sampling site. Samples were collected using vacutainer tubes with a hyperdermic needle at about five minute intervals. SF_6 samples were analyzed by gas chromatography (Hewlett Packard, Mo-

Table 4. Linear regression equations of indoor concentrations on the corresponding outdoor levels.

Species	Slope \pm SE ^a	Intercept \pm SE	R ² ^b	Probability of no correlation
PM _{2.5}	0.85 \pm 0.17	16.9 \pm 10.5	0.84	p < 0.01
SO ₄ ²⁻	1.15 \pm 0.08	-1.42 \pm 0.99	0.98	p < 0.01
NO ₃ ⁻	0.45 \pm 0.08	0.52 \pm 0.66	0.88	p < 0.01

^a Standard error.^b Coefficient of determination.**Table 5. Air exchange rates (hr⁻¹)^a compared with other studies.**

Study	Season	Sampling site	Air exchange rate
Salmon <i>et al.</i> (1990)	Spring	Private home	3.6
Brauer <i>et al.</i> (1991)	Summer	Private homes	3.0
	Winter	Private homes	1.6
This study	Spring	Private home	
		Day	4.29
		Night	1.44

^a Cases where none of the sample sites used air conditioners during the sampling period.

del 5890 Series II) using an electron capture detector (ECD).

The air exchange rate, which is the decay rate of SF₆, was derived from the mass balance model as follows (Wadden and Scheff, 1983):

$$\frac{kq}{V} = -\ln\left(\frac{C_f}{C_i}\right)(t^{-1}) \quad (1)$$

where kq/V is the air exchange rate (hr⁻¹); k is the mixing factor; q in volume/time is the actual ventilation rate; V is the indoor volume; C_i and C_f are the initial and final concentrations of SF₆; and t is the time interval between the initial and final measurements.

From equation (1), the daytime air exchange rate calculated, when the windows were open, was 4.29 hr⁻¹. The nighttime air exchange rate calculated, when the windows were closed, was 1.44 hr⁻¹. The mean value was 2.87 hr⁻¹. Outdoor air quality may be a major significant influence on indoor air quality during the day, because the daytime indoor/outdoor air exchange rate was much higher than that at night. Brauer *et al.* (1991)

found that the summertime and wintertime air exchange rates sampled at private homes were 3.0 and 1.6 hr⁻¹, respectively. Salmon *et al.* (1990) also reported that in a private house, which was converted for use as a museum later, the air exchange rate was 3.6 hr⁻¹ in the spring (Table 5).

4. SUMMARY

The purpose of this study was to provide quantitative information on the indoor air quality of acidic air pollutants in a residential area of Seoul. Indoor pollutant concentrations depend on factors such as the air exchange rate between the indoor and outdoor environment, the pollutant's physio-chemical properties and indoor activities. Among the chemical species measured, indoor levels of HNO₂, NH₃ and PM_{2.5} exceeded their outdoor concentrations. The indoor/outdoor ratio of SO₄²⁻ was similar. Indoor concentrations of PM_{2.5}, SO₄²⁻ and NO₃⁻ were significantly related to the corresponding outdoor levels. The air exchange rates of the daytime and nighttime were calculated as 4.29 and 1.44 hr⁻¹, respectively.

REFERENCES

- Baek, S.-O., Y.-S. Kim, and R. Perry (1997) Indoor air quality in homes, offices and restaurants in Korean urban areas—indoor/outdoor relationships. *Atmospheric Environment*, 31(4), 529–544.
- Brauer, M., F. Koutrakis, G.J. Keeler, and J.D. Spengler (1991) Indoor and outdoor concentrations of inorganic acidic aerosols and gases. *Journal of Air Waste Management Association*, 41, 171–181.
- Brauer, M., B. Ryan, H.H. Suh, P. Koutrakis, and J.D. Spengler (1990) Measurements of nitrous acid inside two research houses. *Environmental Science and Technology*, 24, 1521–1527.
- Dockery, D.W. and J.D. Spengler (1981) Indoor–outdoor relationships of respirable sulfates and particles. *Atmospheric Environment*, 15, 335.
- Finlayson, B.J. and J.N. Pitts (1986) *Atmospheric Chemistry*.

- John Wiley & Sons, New York, NY.
- Genfa, Z., K. Dasgupta, and S. Dong (1989) Measurement of atmospheric ammonia. *Environmental Science and Technology*, 23, 1467–1474.
- Jenkin, M.E., R.A. Cox, and D.J. Williams (1988) Laboratory studies of the kinetics of formation of nitrous acid from the thermal reaction of nitrogen dioxide and water vapor. *Atmospheric Environment*, 22, 487–498.
- Kitto, A.N. and R.M. Harrison (1992) Nitrous and nitric acid measurements at sites in South-east England. *Atmospheric Environment*, 26A, 235–241.
- Koutrakis, P., S.L. Briggs, and B.P. Leaderer (1992) Source apportionment of indoor aerosols in Suffolk and Onondaga counties, New York. *Environmental Science and Technology*, 26(3), 521–527.
- Lee, H.S., C.-M. Kang, B.-W. Kang, and H.-K. Kim (1999) Seasonal variations of acidic air pollutants in Seoul, South Korea. *Atmospheric Environment*, 33(19), 3143–3152.
- Lee, H.S., B.-W. Kang, J.-P. Cheong, and S.-K. Lee (1997) Relationships between indoor and outdoor air quality during the summer season in Korea. *Atmospheric Environment*, 31(11), 1689–1693.
- Phillips, J.L., R. Field, M. Goldstone, G.L. Reynolds, J.N. Lester, and R. Perry (1993) Relationships between indoor and outdoor air quality in four naturally ventilated offices in the United Kingdom. *Atmospheric Environment*, 27A, 1743–1753.
- Reichhardt, T. (1995) Weighing the health risks of airborne particulates. *Environmental Science and Technology*, 29(8), 360A–364A.
- Salmon, L.G., W.W. Nazaroff, M.P. Ligocki, M.C. Jones, and G.R. Cass (1990) Nitric acid concentrations in Southern California museums. *Environmental Science and Technology*, 24, 1004–1013.
- Schwartz, J., D.W. Dockery, and L.M. Neas (1996) Is daily mortality associated specifically with fine particles?. *Journal of Air Waste Management Association*, 46, 927–939.
- Seinfeld, J.H. (1986) *Atmospheric Chemistry and Physics of Air Pollution*. John Wiley & Sons, New York, NY.
- Sheldon, L.S., T.D. Hartwell, B.G. Cox, J.E. Sickles II, E.D. Pellizzari, M.L. Smith, R.L. Perritt, and S.M. Jones (1989) An investigation of infiltration and indoor air quality. Final Report. NY State ERDA Contract No. 736-CON-BCS-85. New York State Energy Research and Development Authority, Albany, NY.
- Spengler, J.D., M. Brauer, J.M. Samet, and W.E. Lambert (1993) Nitrous acid in Albuquerque, New Mexico, Homes. *Environmental Science and Technology*, 27, 841–845.
- Spengler, J.D., M. Brauer, and P. Koutrakis (1990) Acid air and health. *Environmental Science and Technology*, 24, 946–956.
- Stelson, A.W. and J.H. Seinfeld (1982) Relative humidity and temperature dependence of the ammonium nitrate dissociation constant. *Atmospheric Environment*, 16, 983–992.
- U.S.EPA (1997) Determination of reactive acidic and basic gases and strong acidity of fine-particles (< 2.5 μm). U.S. Environmental Protection Agency Compendium Method IO-4.2.
- Wadden, R.A. and P.A. Scheff (1983) *Indoor Air Pollution*. Wiley Interscience, New York, NY.
- Weschler, C.J. and H.C. Shields (1997) Potential reactions among indoor pollutants. *Atmospheric Environment*, 31(21), 3487–3495.
- Weschler, C.J., M. Brauer, and P. Koutrakis (1992) Indoor ozone and nitrogen dioxide: A potential pathway to the generation of nitrate radicals, dinitrogen pentoxide, and nitric acid indoors. *Environmental Science and Technology*, 26(1), 179–184.