15

Indoor Air Pollution
Sources, Levels, Chemistry, and Fates

At the beginning of this book, we presented some discussion of health-based air quality standards. In the final chapter, which follows this one, the scientific bases of control measures for various pollutants are discussed. In between, the complex chemistry that occurs in both polluted and remote atmospheres, and that converts the primary pollutants into a host of secondary species, has been detailed. To provide further perspective on airborne gases and particles and human exposure levels, we briefly treat indoor air pollution in this chapter. As we shall see, for many species it is simply a question of emissions leading to elevated levels indoors. However, there is some chemistry that occurs in indoor atmospheres as well, and it is of interest to compare this to that occurring outdoors.

From the point of view of health impacts, it is the combination (not necessarily linear) of concentrations and duration of exposure to a pollutant or combination of pollutants that is important. In this regard, it is noteworthy that most individuals spend the majority of their time indoors, even in relatively moderate climates. For example, in California, people spend 87% of their time indoors on average, 7% in enclosed transit systems, and only 6% outdoors (Jenkins *et al.*, 1992). Similarly, Quackenboss *et al.* (1986) report that only 15% of the average day in Portage, Wisconsin, was spent outdoors in summer and less than 5% in winter. As a result, elevated concentrations of air pollutants indoors can have a significant impact on human health and can lead to enhanced chemical sensitivities (Hileman, 1991) as well as other health impacts such as cancer. Establishing the contribution of indoor air pollution to carcinogenicity and assessing the relative risks (e.g., Tancrède *et al.*, 1987) are complex and difficult. An interesting approach suggested recently is pet epidemiology, discussed in detail by Bukowski and Wartenberg (1997).

The term "indoors" is used in the literature to refer to a variety of environments, including homes, workplaces, and buildings used as offices or for recreational activities. In addition, a number of studies have been carried out to measure various compounds inside vehicles during commutes. As we shall see, and consistent with expectations, levels measured indoors are characteristic both of the particular sources present and, to a significant extent, of the outdoor concentrations of the species. We shall not, in general, distinguish in this chapter between the various types of indoor environments but rather focus on the sources of various compounds and their indoor chemistry.

Table 15.1 summarizes the major species of concern for indoor air pollution and some of their sources (Su, 1996). We focus in this chapter primarily on those species common to indoor and outdoor air environments, including oxides of nitrogen, volatile organic compounds (VOC), CO, ozone, the OH radical, SO_2, and particles. In addition, a brief discussion of radon is included since this has been one of the major foci of concern in the past with respect to indoor air pollution.

A. RADON

Radon (^{222}Rn) is formed by the radioactive decay of uranium, ^{238}U (Fig. 15.1a). As a result, the highest concentrations tend to be associated with soils derived from rocks with a high uranium content (Nazaroff and Nero, 1988; Boyle, 1988; Nero, 1989; Mose and Mushrush, 1997). Because radon is a gas that diffuses out of the soil, it can enter homes through cracks in the foundation, around loose-fitting pipes and wall joints, and through floor drains (e.g., Nero, 1989). The concentrations found in a home depend on the type of soil (including the moisture content) on which it sits and the extent of Rn penetration into the house. They also depend on the house ventilation rate and the particular location in the house in which the measurement is

TABLE 15.1 Some Indoor Air Pollutants and Their Sources[a]

| Species	Sources
Radon (^{222}Rn)	Soil and some masonry building materials
Oxides of nitrogen	Combustion
Volatile organic compounds (including HCHO)	Building materials, carpets, solvents, paints, household and personal care products, air fresheners, pesticides, mothballs, humans, treated water
CO	Combustion
O_3	Outside air, photocopying machines, electrostatic air cleaners
SO_2	Combustion
Particles	Combustion
Asbestos	Building materials, handheld hair dryers
Microorganisms	Air conditioners, cool-mist humidifiers

[a] Adapted from Su (1996) and Nero (1988).
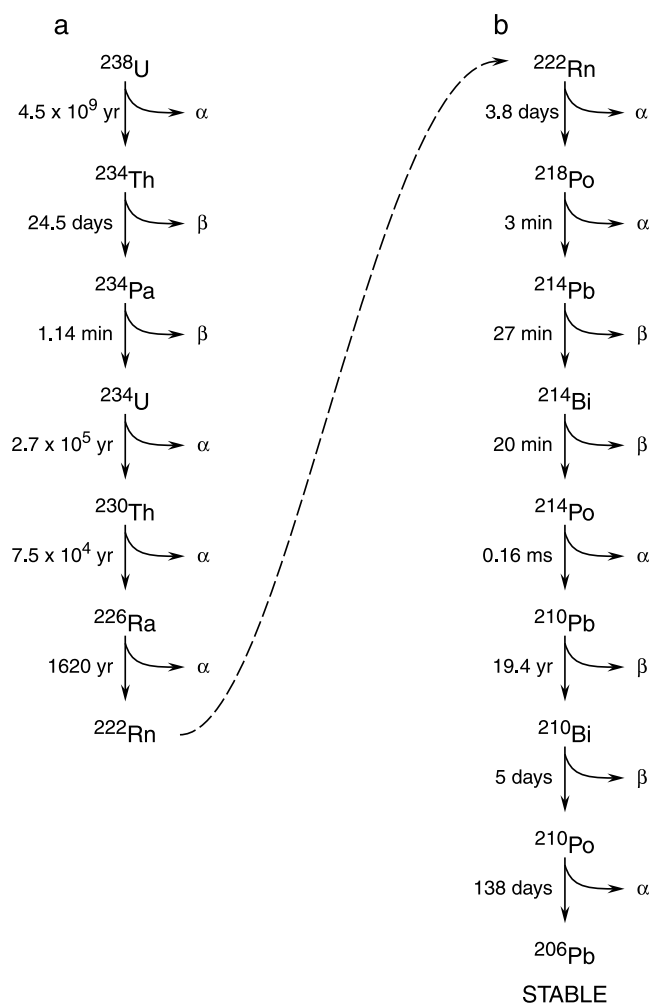
made, with the basement typically having the highest concentrations (e.g., Liu *et al.*, 1991). Interestingly, because homes are often warmer than their surroundings, a "chimney effect" occurs that draws gases, including radon, into the house from the surroundings (e.g., Osborne, 1987; Nero, 1989; Turk *et al.*, 1990; Hintenlang and Al-Ahmady, 1992).

Other sources include building materials such as concrete that are made from the earth's crustal materials and hence can contain significant amounts of uranium and radium (Nazaroff and Nero, 1988). Radon dissolves in water, and hence degassing from household water can also be a source. For example, Osborne (1987) reported that the radon concentration in a bathroom increased by more than two orders of magnitude during a 15-min period that a shower was running.

The health concerns associated with ^{222}Rn are primarily associated with its radon daughters. As a noble gas, radon is unreactive in air and is both readily inhaled and exhaled. However, a significant portion of its daughters are positively charged ions that are expected to attract water vapor and become hydrated; the formation of clusters with other ions is also likely (e.g., Castleman, 1991). Uptake on existing aerosol particles also occurs readily, and such particles can then be deposited in the respiratory tract, providing a source of radioactive emissions directly to the lung. Effects such as lung cancer may then ensue.

The health effects associated with radon, as well as sources and mitigation measures, are discussed in detail in several National Research Council reports (1988, 1991), in the book edited by Nazaroff and Nero (1988), and in the International Commission on Radiological Protection Report (1994). Initial risk assessments were based on data from underground miners who were exposed to relatively high levels of radon and its progeny. However, there has been considerable controversy over the extrapolation to lower levels in homes [e.g., see summaries by Nazaroff and Teichman (1990) and Peto and Darby (1994)].
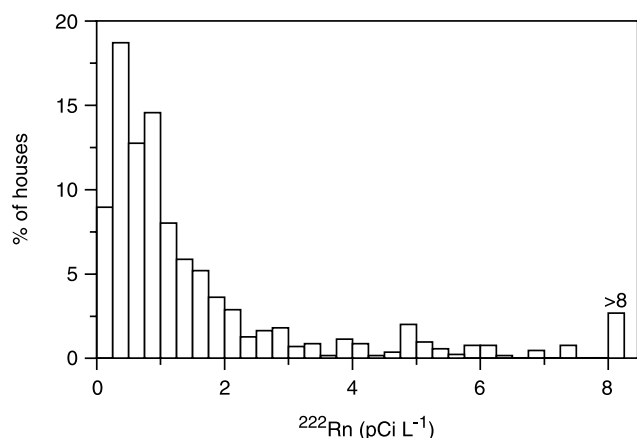

FIGURE 15.1 The major radioactive decay paths in (a) the uranium series leading to the formation of ^{222}Rn and (b) of ^{222}Rn to form ^{206}Pb. Half-lives are shown to the left of each process.

FIGURE 15.2   Distribution of ^{222}Rn measured in homes in the United States (adapted from Nero *et al.*, 1986).

A number of studies have been carried out to determine the concentrations of radon in homes (e.g., see Nero *et al.*, 1986; Alter and Oswald, 1987; Turk *et al.*, 1990; Liu *et al.*, 1991; and Mose and Mushrush, 1997). Units used to express the concentration of ^{222}Rn are picocuries per liter (pCi L^{-1}), with 1 pCi being the amount of substance that gives 2.2 radioactive decays per minute, or becquerels per cubic meter (Bq m^{-3}), where 1 pCi L^{-1} = 37 Bq m^{-3} (e.g., see Nazaroff and Teichman, 1990).

Figure 15.2 shows the distribution of levels measured in homes in the United States (Nero *et al.*, 1986). As might be expected given the variables that affect ^{222}Rn concentrations in homes, the concentrations vary widely, from <0.1 to >8 pCi L^{-1}. Indeed, Alter and Oswald (1987) report a few single measurements of up to 4000 pCi L^{-1}. For comparison, the U.S. EPA recommends levels below 4 pCi L^{-1} (150 Bq m^{-3}), which is exceeded by about 7% of U.S. homes (Nero *et al.*, 1986), and the International Commission on Radiological Protection (1993) recommends 5.4–16 pCi L^{-1} (200–600 Bq m^{-3}). Typical outdoor concentrations in continental areas are 0.1–0.4 pCi L^{-1} (Nazaroff and Nero, 1988).

B. OXIDES OF NITROGEN

1. Levels of NO$_x$

A large number of studies of NO and NO$_2$ have been carried out in many different indoor air environments. Because of air exchange, indoor levels are generally higher when outdoor levels increase (e.g., Hoek *et al.*, 1989; Rowe *et al.*, 1991; Hisham and Grosjean,

1991a; Spengler *et al.*, 1994; Weschler *et al.*, 1994; Baek *et al.*, 1997). However, enhanced indoor levels are found when combustion sources are present. These include gas stoves, kerosene heaters, water heaters, and cigarette smoke (e.g., Wade *et al.*, 1975; Marbury *et al.*, 1988; Ryan *et al.*, 1988; Petreas *et al.*, 1988; Hoek *et al.*, 1989; Pitts *et al.*, 1989; Spengler *et al.*, 1994; Levy *et al.*, 1998). While combustion generates primarily NO, the focus indoors has been on NO$_2$ because of its health impacts (see Chapter 2.).

For example, Spengler *et al.* (1994) report that personal exposures to NO$_2$ were 10 ppb higher in homes in the Los Angeles area having gas ranges with pilot lights compared to those having electric ranges and 5 ppb higher if the gas range did not have a pilot light. Levy *et al.* (1998) report NO$_2$ concentrations indoors and outdoors, as well as personal exposures, in 18 different cities in 15 countries in the Northern Hemisphere over a two-day period in February and March when the use of combustion was expected to be higher. The ratio of indoor to outdoor concentrations ranged from 0.3 ± 0.2 in Berlin, Germany, to 2.8 ± 2.8 in Tokushima, Japan. Average NO$_2$ concentrations over the two-day period indoors ranged from 5.5 ppb in Kuopio, Finland, to 63 ppb in Mexico City. Outdoors, the range of concentrations was from 12 ppb in Geneva, Switzerland, to 52 ppb in Seoul, Korea.

Again, the use of gas stoves was highly correlated with indoor NO$_2$, with an indoor/outdoor concentration ratio of 1.19 for homes with a gas range compared to 0.69 for those without a gas stove. The ratio was even higher for homes with a kerosene space heater, 2.3 compared to 0.85 without such a heater (Levy *et al.*, 1998). Both the indoor and outdoor concentrations of NO$_2$ were higher in cities where at least 75% of the homes had gas stoves; for example, the mean outdoor NO$_2$ concentration in such gas-intensive cities was 38 ± 20 ppb, compared to 14 ± 6 ppb in cities where fewer than 25% of the households had gas ranges.

High concentrations of NO$_2$ have also been measured in indoor skating rinks where the use of ice resurfacing machines powered by propane, gasoline, or diesel fuel results in significant emissions (e.g., Brauer and Spengler, 1994; Brauer *et al.*, 1997; Pennanen *et al.*, 1997). Mean concentrations of NO$_2$ of ~200 ppb have been reported, with some rinks having concentrations up to 3 ppm! The indoor-to-outdoor ratios of the arithmetic mean concentrations varied from about 1 to 41, with an overall mean of 20. Figure 15.3 shows the effect of the particular fuel used on the indoor NO$_2$ concentrations (Brauer *et al.*, 1997). Propane gave the highest values, followed by gasoline and diesel. The NO$_2$ concentrations when the resurfacer was electric were similar to those outdoors.
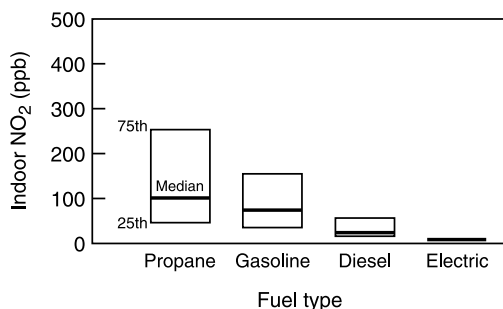
FIGURE 15.3 Indoor concentrations of NO$_2$ measured in ice skating rinks using different types of power for the ice resurfacers. The median and the 25th and 75th percentiles are shown in each case (adapted from Brauer *et al.*, 1997).

In the absence of such sources of NO$_x$, indoor and outdoor concentrations are quite similar (e.g., Weschler *et al.*, 1994), since removal of NO and NO$_2$ indoors, e.g., on surfaces, is relatively slow. However, as discussed shortly, although the surface reaction of NO$_2$ is relatively slow, it is still of interest since it generates nitrous acid (HONO). Different surfaces found inside homes have been found to have different removal rates for NO$_2$. Figure 15.4, for example, shows measured rates of removal of NO$_2$ by a number of common household materials (Spicer *et al.*, 1989). Large variations in removal rate (and hence the formation of products such as NO and HONO; see later) are evident, varying from negligible for plastic storm windows to quite large for wallboard.

In short, there is a variety of evidence that there are higher levels of NO$_2$ indoors when combustion sources are present and that the concentrations generated indoors can be quite substantial in some circumstances. One word of caution is in order, however, particularly with regard to earlier measurements of NO$_2$. As discussed in the following section, significant concentrations of HONO are generated both by a heterogeneous reaction of NO$_2$ on surfaces and by direct emissions from combustion sources. In some measurement methods used for NO$_2$, HONO is also detected and hence reported as NO$_2$. This is particularly true for the O$_3$ chemiluminescence method and for electrochemical sensors (e.g., Spicer *et al.*, 1994), so that NO$_2$ reported using these techniques should be regarded as upper limits to its concentrations. This problem can be circumvented through the use of denuders to remove HONO prior to sampling into the instrument.

2. HONO and HNO$_3$

As discussed in Chapter 7.C.1, HONO is formed by the reaction of NO$_2$ with water on surfaces. The reac-
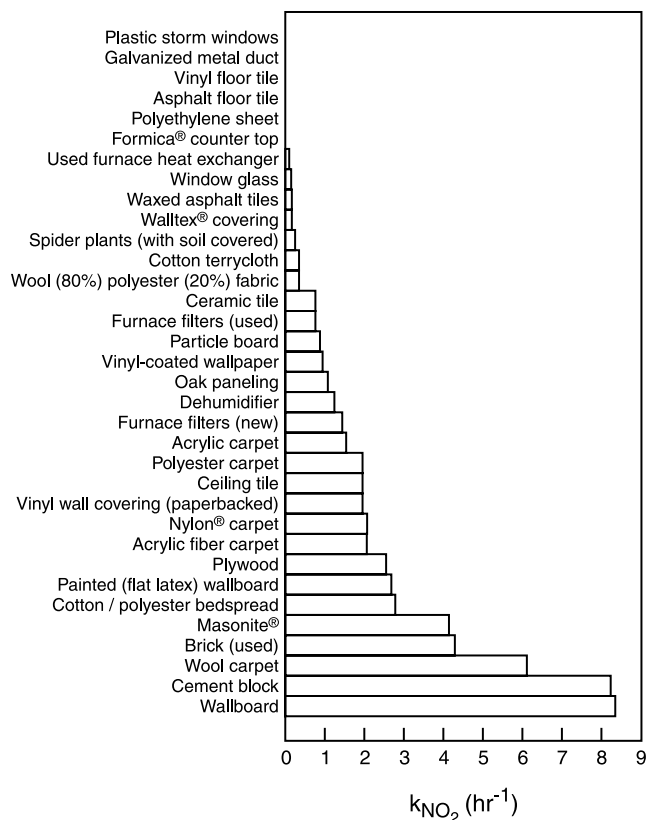

FIGURE 15.4 Rate constants for NO$_2$ removal on 3.3-m^2 samples of different materials commonly found in homes (adapted from Spicer *et al.*, 1989).

tion is usually represented as

$$2NO_2 + H_2O \xrightarrow{\text{surface}} HONO + HNO_3, \quad (1)$$

although the detailed mechanism is not known; gaseous HNO$_3$ is not generated in equivalent amounts, which has been attributed to its remaining adsorbed on the surface. This overall reaction occurs on a variety of surfaces in the laboratory and hence might be expected to also occur on surfaces in other environments, such as homes. This, indeed, is the case.

Pitts *et al.* (1985) first used differential optical absorption spectrometry (DOAS) to establish unequivocally that NO$_2$ injected into a mobile home forms HONO. Interestingly, the dependence of the rate of HONO generation on the NO$_2$ concentration was similar to that measured in laboratory systems, consistent with production in, or on, a thin film of water adsorbed on surfaces. A number of studies have confirmed that the behavior is similar to that in laboratory systems; i.e., the rate of production of HONO increases with NO$_2$ and with relative humidity. Indoor levels of HONO as high as 8 ppb as a 24-h average and 40 ppb as a 6-h

average have been reported in normal, in-use buildings and homes (Febo and Perrino, 1991; Spengler *et al.*, 1993; Weschler *et al.*, 1994).

The ratio of HONO to NO_2 indoors can be quite large, up to ~0.15 (e.g., Febo and Perrino, 1991; Brauer *et al.*, 1990, 1993; Spengler *et al.*, 1993). This can be compared to typical values of a few percent outdoors (see Chapter 11). High levels of HONO (up to ~30 ppb) have also been measurd in automobiles in use in polluted urban areas, and again, the ratio of HONO to NO_2 was quite large, ~0.4, compared to 0.02–0.03 measured outdoors in the same study (Febo and Perrino, 1995).

Figure 15.5, for example, shows one set of measurements of NO_2, HONO, and NO after injection of NO_2 into a home used for research purposes (Spicer *et al.*, 1993). As NO_2 decays, HONO is formed. Small amounts of NO are also generated, as has been observed in laboratory studies (e.g., Spicer *et al.*, 1989). Delayed release of HONO was observed, suggesting that HONO was adsorbed in part on the surfaces and was subsequently released to the gas phase. For example, when the house was purged with outside air, the HONO levels dropped; however, on closing up the house again, the levels increased immediately to a few ppb. Degassing of HONO from the interior surfaces of automobiles has also been reported (Febo and Perrino, 1995). Interestingly, in measurements made in a commercial office building, indoor HONO concentrations were observed not to be enhanced under conditions of high indoor O_3, suggesting that dissolved HONO in a surface film of water is readily oxidized to nitrate by O_3 (Weschler *et al.*, 1994).

The generation of NO in Fig. 15.5 was attributed by Spicer and co-workers to a reaction of gaseous NO_2 with adsorbed HONO:

$$NO_{2(g)} + HONO_{(ad)} \rightarrow H^+ + NO_3^- + NO_{(g)}. \quad (2)$$
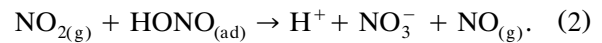
The same process was hypothesized to explain some time periods in a commercial office building when indoor NO actually exceeded outdoor NO (Weschler *et al.*, 1994).

As is the case in laboratory systems, equivalent amounts of HNO_3 are not observed as might be expected from the stoichiometry of reaction (1), likely due to HNO_3 remaining on the surface after formation and/or being taken up by surfaces. For example, Spicer *et al.* (1993) used atmospheric pressure ionization mass spectrometry (see Chapter 11.A.2), which could measure HNO_3 with a sensitivity of 50 ppt, during the NO_2 decay and HONO formation in the research home. HNO_3 was only observed, and in small concentrations (<1 ppb), when an unvented space heater was operating, and it decayed rapidly in the absence of sources. Similarly, Salmon *et al.* (1990) measured HNO_3 indoors and outdoors at five museums in the Los Angeles area and found that indoor concentrations were less than ~40% of those outdoors, and typically about 10%. The accumulation of nitrate on indoor surfaces in a commercial building has been reported by Weschler and Shields (1996a) and attributed to the formation and uptake of HNO_3 via reactions of NO_3 (see later) and/or oxidation of nitrite (i.e., adsorbed HONO) in an aqueous surface film.

Subsequently, it was shown that HONO is also directly emitted by gas stoves (Pitts *et al.*, 1989). For example, Fig. 15.6 shows the concentrations of NO_2 and HONO measured using DOAS when two top burners of a new, residential gas kitchen stove were turned on in a mobile home with both the central ventilation and air conditioning running. NO_2 from the gas stove emissions reached almost 300 ppb, and HONO about

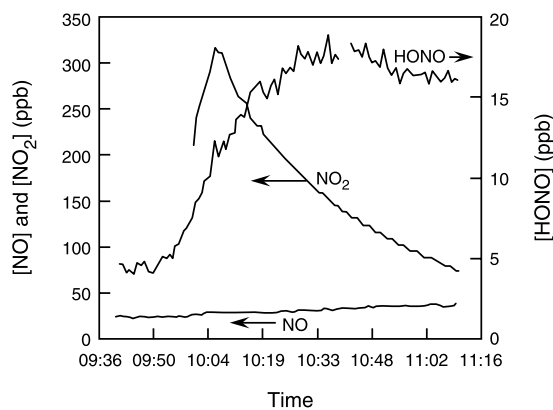**FIGURE 15.5** Concentrations of NO_2, HONO, and NO after injection of NO_2 into a research house (adapted from Spicer *et al.*, 1993).

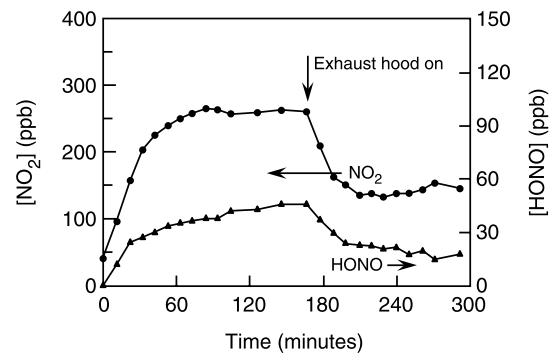**FIGURE 15.6** NO_2 and HONO measured in a mobile home with two burners of a kitchen stove on; the above-range exhaust was turned on at the time shown (adapted from Pitts *et al.*, 1989).
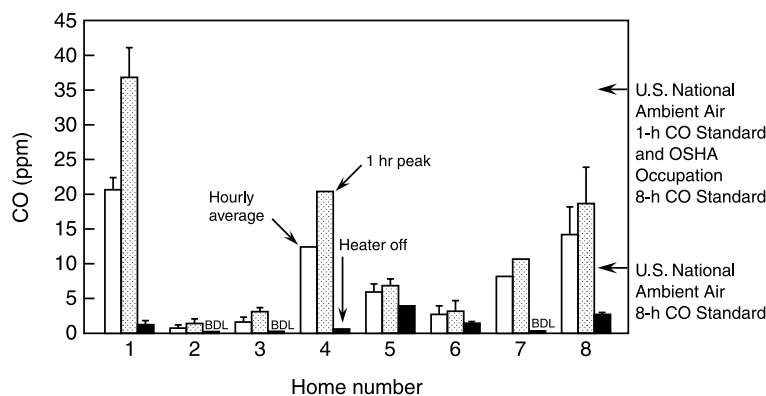
FIGURE 15.7 Measured concentrations of CO in eight homes using unvented kerosene heaters: dark shading, hourly average concentrations with heater off; no shading, hourly average concentrations with heater on; light shading, peak 1-h concentration (adapted from Mumford *et al.*, 1991).

45 ppb. When the exhaust hood (which was externally vented) above the range was turned on, the NO$_2$ and HONO decreased substantially. Similar production of HONO from kerosene and propane space heaters has been observed (e.g., see Pitts *et al.*, 1989; Brauer *et al.*, 1990; Febo and Perrino, 1991; and Vecera and Dasgupta, 1994). While the mechanism generating such substantial concentrations is not known, it may involve the recombination of OH with NO as the combustion gases cool.

In a house used for investigating indoor air pollution that had natural gas fueled applicances (a convective heater, a radiant heater, and a range with four burners), both the surface reaction of NO$_2$ and the direct combustion emissions contributed significantly to the measured indoor HONO. When an appliance was operational, the contribution of direct emissions was the more important source (Spicer *et al.*, 1993).

In short, the "dark reaction" of NO$_2$ with water on surfaces is ubiquitous and occurs not only in laboratory systems but also indoors. The combination of this heterogeneous reaction with combustion sources of HONO can produce significant concentrations of HONO indoors. As a result, there is a concern regarding the health impacts of nitrous acid, not only because it is an inhalable nitrite but also because it is likely the airborne acid present in the highest concentrations indoors.

C. CO AND SO$_2$

As for NO$_x$, combustion sources such as gas stoves and kerosene heaters can be significant sources of indoor CO. Figure 15.7, for example, shows measured

CO concentrations in eight mobile homes with unvented kerosene heaters either off or on (Mumford *et al.*, 1991). Both the hourly average and peak 1-h concentrations are shown for the situation with the heater on, whereas only the 1-h average for the heater off is shown. Also shown are the United States 1- and 8-h standards and the Occupational Safety and Health Adminstration (OSHA) 8-h standard. In three of the homes, the average CO concentrations exceeded the 8-h standard and in one home, the 1-h ambient air and OSHA standard was exceeded.

The ratio of indoor to outdoor concentrations of CO in homes using gas stoves has been measured to be 1.2–3.8 (Wade *et al.*, 1975), with the highest ratios found close to the source. Similarly, higher CO levels indoors compared to outdoors have been reported for restaurants in Korea, with those using charcoal burners as well as gas giving much higher concentrations (Baek *et al.*, 1997). Figure 15.8, for example, shows the mean indoor-to-outdoor concentrations of CO and some other air pollutants measured in restaurants in Korea using either gas only or a combination of gas and
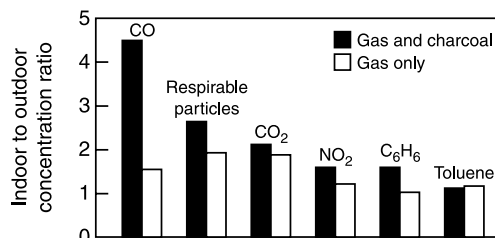

FIGURE 15.8 Median ratios of indoor-to-outdoor concentrations of CO and some other air pollutants measured in restaurants in Korea where either gas alone (□) or gas with charcoal (■) was used for cooking (adapted from Baek *et al.*, 1997).

charcoal. The use of charcoal increases this ratio to more than four; in one such restaurant, an indoor concentration of 90 ppm CO was measured.

In buildings where motor vehicle exhaust can be entrained from outdoors or attached parking garages, elevated indoor CO levels may also result (e.g., Hodgson *et al.*, 1991).

On the other hand, in homes and offices where there was no direct indoor source of CO, the indoor-to-outdoor ratio was about one, and sometimes less. For example, in Riyadh, Saudi Arabia, CO concentrations were measured indoors and outdoors; the indoor-to-outdoor ratio varied from 0 to 2, but was typically below one (Rowe *et al.*, 1989).

There have been a number of measurements of CO in the "indoor environment" of automobiles. Given that cars are major CO sources in urban areas, one might expect higher concentrations of CO during commutes and this is indeed the case. Typical CO concentrations of ~9–56 ppm have been measured inside automobiles during commutes in major urban areas (Flachsbart *et al.*, 1987; Koushki *et al.*, 1992; Ott *et al.*, 1994; Dor *et al.*, 1995; Fernandez-Bremauntz and Ashmore, 1995). This can be compared to peak outdoor levels of ~10 ppm in highly polluted urban areas (see Chapter 11.A.4c). Thus, a significant enhancement of CO inside automobiles during commutes is common. For example, Chan *et al.* (1991b) report a ratio of the in-vehicle CO concentration to that outdoors of ~4.5 in Raleigh, North Carolina.

As is the case for CO, SO_2 levels indoors and outdoors tend to be similar if there are no combustion sources indoors. For example, Hisham and Grosjean (1991b) report that the ratio of indoor-to-outdoor SO_2 concentrations averaged 0.89 for three museums in southern California, with a range from 0.36 to 1.92. On the other hand, quite high SO_2 concentrations can result when there are indoor combustion sources. For example, in China, where unvented stoves using coal are used extensively for cooking and heating, average SO_2 levels indoors are typically 250 $\mu g\ m^{-3}$ (96 ppb) during the summer and 750 $\mu g\ m^{-3}$ (287 ppb) during the winter, compared to average annual outdoor concentrations of 72–94 $\mu g\ m^{-3}$ (28–36 ppb) (Florig, 1997; Ando *et al.*, 1996).

D. VOLATILE ORGANIC COMPOUNDS

Volatile organic compounds (VOC) are ubiquitous components not only of ambient air but also of indoor air environments, including offices, commercial and retail buildings, and homes (Shah and Singh, 1988). There are three sources/categories for VOC: (1) entrainment of air from outside the building, (2) emissions from building materials, and (3) human activities inside buildings.

As might be expected given the nature of the sources, a very large variety of organic compounds have been identified and measured indoors (e.g., Brown *et al.*, 1994; Crump, 1995; Kostiainen, 1995). These number in the hundreds of different compounds, with the particular species and their concentrations depending on the particular sources present as well as the air exchange rates. Table 15.2 summarizes some of the types of organics that have been measured in indoor air and typical sources (Tichenor and Mason, 1988; Crump, 1995). Because of the VOC sources present indoors, the indoor-to-outdoor concentration ratios are quite large for many compounds.

For example, Table 15.3 shows some typical ratios of indoor-to-outdoor concentrations for specific compounds found in each of the classes shown in Table 15.2, which are frequently present indoors (Brown *et al.*, 1994). These data are based on a review of the literature and include data from a number of different countries. The ratio is for all but one compound substantially greater than one. Also shown in Table 15.3 is a typical range of concentrations expressed as the overall weighted average of the geometric mean, where the weighting was done using the number of available measurements. Some of the compounds associated with the three sources—entrainment from outdoors, emissions from building materials, and anthropogenic activities—are now briefly reviewed.

Entrainment of air from outdoor sources. Entrainment of outdoor air through ventilation systems brings with it the species found in ambient air, which have been discussed throughout this book. Some of them, such as HNO_3 and to a lesser extent O_3, can be removed on surfaces such as those in air conditioning systems, and hence the indoor concentrations tend to be lower than those outdoors. Others such as NO tend to have similar concentrations indoors and outdoors if there are no significant combustion sources indoors (e.g., Weschler *et al.*, 1994). In the case of hydrocarbons, the concentrations of compounds that do not have significant indoor sources tend to be about the same as the outdoor concentrations. For example, Lewis and Zweidinger (1992) measured VOC in 10 homes in winter and showed that the concentrations of ethene, benzene, 2-methylpentane, methylcyclopentane, 2,2,4-trimethylpentane, and 2,3-dimethylbutane indoors were within experimental error of those outdoors.

TABLE 15.2   Some VOCs Measured Indoors and Their Sources[a]

Class of Compounds	Species	Typical Sources
Hydrocarbons	Aliphatic hydrocarbons	Paints, adhesives, gasoline, combustion products, floor waxes
	Aromatic hydrocarbons (toluene, xylenes, ethyl benzene, trimethylbenzenes, styrene, benzene)	Insulation, textiles, disinfectants, plastics, paints, smoking
	Terpenes (limonene, α-pinene)	Scented deodorizers, polishes, fabrics, fabric softeners, cigarettes, food, beverages
	PAHs	Combustion products (smoking, wood burning, kerosene heaters)
Oxygenated organics	Acrylic acid esters, epichlorohydrin	Monomers may escape from polymers
	Alcohols	Aerosols, window cleaners, paints, paint thinning, cosmetics, adhesives
	Ketones	Lacquers, varnishes, polish removers, adhesives
	Ethers	Resins, paints, varnishes, lacquers, dyes, soaps, cosmetics
	Esters	Plastics, resins, plasticizers, lacquer solvents, flavors, perfumes
	Ethylene oxide	Sterilizers (hospitals)
Other organics	Toluene diisocyanate	Polyurethane foam aerosols
	Phthalic acid anhydride	Epoxy resins
	Sodium dodecyl sulfate	Carpet shampoo
Chlorinated organics	Benzyl chloride	Vinyl tiles
	Tetrachloroethylene	Dry-cleaned clothes
	Chloroform	Chlorinated water
	1,1,1-Trichloroethane	Dry-cleaned clothes, aerosol sprays, fabric protectors
	Carbon tetrachloride	Industrial-strength cleaners
	p-Dichlorobenzene	Moth crystals, room deodorants

[a] Adapted from Crump (1995) and Tichenor and Mason (1988).

There are, however, some specific outdoor sources that can lead to higher concentrations of certain VOCs indoors than in the general outdoor air environment. For example, gases generated in landfills or from petroleum contamination can migrate through the soil and groundwater to adjacent buildings and homes to give larger indoor concentrations, particularly in basements and crawl spaces, than otherwise expected (e.g., Moseley and Meyer, 1992; Hodgson et al., 1992; Fischer et al., 1996). In one such case, the total hydrocarbon concentration was measured to be 120 ppm in a crawl space beneath the floor of a school where petroleum contamination was present from adjacent sources, compared to <80 ppb outdoors (Moseley and Meyer, 1992). Although concentrations in various rooms were lower, they were still elevated compared to outdoors, ranging from 0.13 to 3.4 ppm.

The use of pesticides *outside* buildings can also lead to enhanced concentrations of these compounds indoors. For example, Anderson and Hites (1988) measured the concentrations of chlorinated pesticides indoors and found elevated levels inside, e.g., a factor of 7 times higher for γ-chlordane compared to outdoor levels. One home that had the highest indoor concentrations had been treated with chlordane about a decade earlier, presumably by subsurface injection from which the pesticide migrated into the house through cracks in the basement walls. Enhanced levels of chlorpyrifos were observed indoors in homes where soil surrounding the home had been treated on a regular

TABLE 15.3   Ratio of Indoor-to-Outdoor Concentrations for Some VOCs and Typical Ranges of Concentrations[a]

Compound	Indoor / outdoor ratio (range of concentration (μg m^{-3}))	Compound	Indoor / outdoor ratio (range of concentration (μg m^{-3}))
n-Alkane		Alcohols	
n-Pentane	3	2-Propanol	>73
n-Hexane	9	n-Butanol	5 (<1)
n-Heptane	4 (1–5)		
n-Octane	7 (1–5)	Aldehydes	
n-Nonane	14 (1–5)	Acetaldehyde	5
n-Decane	19 (5–10)	Butanal	2 (1–5)
n-Undecane	20 (1–5)	Hexanal	>5
n-Dodecane	20 (1–5)	Nonanal	5 (5–10)
n-Tridecane	>6		
n-Tetradecane	16 (1–5)	Ketones	
n-Pentadecane	>5 (1–5)	Acetone	12 (20–50)
		Methyl ethyl ketone	4 (1–5)
Branched cycloalkanes			
2-Methylpentane	2	Esters	
2-Methylhexane	2	Ethyl acetate	15 (5–10)
3-Methylhexane	3		
Cyclohexane	4 (1–5)	Aromatic hydrocarbons	
		Styrene	10 (1–5)
Halogenated compounds		Benzene	3 (5–10)
Trichlorofluoromethane	10	Toluene	6 (20–50)
1,2-Dichloroethane	12 (<1)	Ethylbenzene	6 (5–10)
Dichloromethane	6 (10–20)	m- and p-xylene	6 (10–20)
Chloroform	5 (1–5)	o-Xylene	6 (5–10)
Carbon tetrachloride	2 (1–5)	n-Propylbenzene	4
1,1,1-Trichloroethane	9 (20–50)	1,3,5-Trimethylbenzene	4 (1–5)
1,1-Dichloroethene	13 (1–5)	1,2,4-Trimethylbenzene	15 (5–10)
Trichloroethylene	6 (1–5)	(1-Methylethenyl)benzene	5
Tetrachloroethylene	5 (5–10)	Naphthalene	4 (<1)
p-Dichlorobenzene	5 (5–10)		
m-Dichlorobenzene	0.4 (<1)	Terpenes	
		Camphene	20 (10–20)
		α-Pinene	23 (1–5)
		Limonene	80 (20–50)

Total VOC   7

[a] Adapted from Brown et al. (1994) and references therein.

basis. Similarly, aldrin and dieldrin (used as termiticides in the United States from the 1950s to the 1970s) were shown to be present indoors in a home in which the surrounding soil and/or concrete blocks used for the basement walls had been treated during construction; as might be expected given the source, the highest levels were in the basement (Wallace et al., 1996).

Another source of VOC is motor vehicle emissions, which can be drawn into buildings from outdoors or parking garages (e.g., Perry and Gee, 1994; Daisey et al., 1994). For example, motor vehicles were major sources (responsible for ≥75%) of 12 of 39 individual compounds measured in a dozen buildings by Daisey et al. (1994). Of the 12 compounds, 5 were alkanes and 7 were aromatics. Similarly, Baek et al. (1997) report that

vehicle emissions are important VOC sources indoors in Korea during the summer in homes and offices, as has been reported in the United States (e.g., Hodgson et al., 1991; Daisey et al., 1994).

Building materials. Emissions associated with building materials are major contributors to indoor levels of VOC. Table 15.4 summarizes some individual VOCs that have been associated with some building materials (Tichenor and Mason, 1988; Crump, 1995). The overall weighted average geometric mean concentrations of some individual organics measured indoors in buildings that were more than 3 months old are shown in Table 15.5 (Brown et al., 1994). New buildings often have higher concentrations of certain compounds compared to older buildings. For example, enhanced levels of

TABLE 15.4   Some VOCs from Building Materials[a]

Source	Compounds
Carpet adhesive	Toluene
Floor adhesive (water based)	Nonane, decane, undecane, dimethyloctane, 2-methylnonane, dimethylbenzene
Particleboard	Formaldehyde, acetone, hexanal, propanol, butanone, benzaldehyde, benzene
Moth crystals	p-Dichlorobenzene
Floor wax	Nonane, decane, undecane, dimethyloctane, trimethylcyclohexane, ethylmethylbenzene
Wood stain	Nonane, decane, undecane, methyloctane, dimethylnonane, trimethylbenzene
Latex paint	2-Propanol, butanone, ethylbenzene, propylbenzene, 1,1'-oxybis[butane], butylpropionate, toluene, formic and acetic acids
Water-based acrylic wall paint	1,2-Propanediol, isomers of 2,4,4-trimethyl-1,3-pentanediol monoisobutyrate
Furniture polish	Trimethylpentane, dimethylhexane, trimethylhexane, trimethylheptane, ethylbenzene, limonene
Polyurethane floor finish	Nonane, decane, undecane, butanone, ethylbenzene, dimethylbenzene
Room freshener	Nonane, decane, undecane, ethylheptane, limonene, substituted aromatics (fragrances)
Particleboard	Formaldehyde, acetone, hexanol
Vinyl flooring	Alkyl aromatics, dodecane, 2,2,4-trimethyl-1,3-pentanediol diisobutylate, 2-ethyl-1-hexanol, phenol, cresol, ethyl hexyl acetate, ammonia
Floor varnish	Butyl acetate, N-methylpyrrolidone
Laminated cork floor tile	Phenol
Carpets	4-Phenylcyclohexene, styrene, 4-ethenylcyclohexene, 2-ethyl-1-hexanol, nonanol, heptanol
Silicone caulk	Methyl ethyl ketone, butyl propionate, 2-butoxyethanol, butanol, benzene, toluene
Paint	Dibutyl phthalate
Acrylic sealant	Hexane, dimethyloctanols
Creosote-impregnated timber	Naphthalene, methylnaphthalenes

[a] Adapted from Tichenor and Mason (1988), Crump (1995), Reiss et al. (1995b), and Wolkoff (1998).

n-dodecane, n-decane, and n-undecane, the xylenes, and 2-propanol have been measured in new buildings, and the total VOC concentration is generally larger (by factors of 4–23) compared to established buildings (Brown et al., 1994).

Kostiainen (1995) identified more than 200 individual VOCs indoors in 26 houses. In addition, they compared the VOC concentrations in normal houses to those where complaints of odors or illness had been registered. A number of different VOCs were present at increased concentrations in the houses with complaints compared to the normal houses; these included a variety of aromatic hydrocarbons, methylcyclohexane, n-propylcyclohexane, terpenes, and chlorinated compounds such as 1,1,1-trichloroethane and tetrachlorethene.

Carpets are a major source of VOCs in homes. For example, Sollinger et al. (1993, 1994) have identified 99 different VOCs emitted from a group of 10 carpet samples, and Schaeffer et al. (1996) identifed more than 100 different VOCs emitted from the carpet cushion alone. Emissions come not only from the carpet fibers but also from the backing materials and the adhesives used to bind the carpet to the backing. As a result, the individual compounds emitted by carpets can vary substantially, depending on the carpet construction. For example, Table 15.6 shows some of the compounds emitted by three different types of carpets: (1) with a styrene–butadiene rubber (SBR) latex adhesive to bind the primary and secondary backings; (2) with a polyvinyl chloride (PVC) secondary backing and made in the form of tiles; and (3) with a polyurethane foam secondary backing. With the SBR adhesive, the major VOCs emitted are 4-phenylcyclohexene (responsible for the "new carpet" smell) and styrene. With the PVC backing, the major compounds are vinyl ac-

TABLE 15.5   Overall Weighted Average Geometric Mean Concentrations
of Some Individual VOCs Indoors[a]

Compound	Type of building	Number of measurements	Concentration ($\mu g\ m^{-3}$)
Hydrocarbons			
n-Hexane	Dwelling	656	5
	Office	26	12
n-Nonane	Dwelling	592	5
n-Decane	Dwelling	1085	5
Camphene	Mobile home	44	14
Limonene	Dwelling	584	21
Benzene	Dwelling	2171	8
Toluene	Dwelling	792	37
Ethylbenzene	Dwelling	1867	5
o-Xylene	Dwelling	1518	6
m- and *p*-xylene	Dwelling	1587	18
1,2,4-Trimethylbenzene	Dwelling	619	6
1,2,3-Trimethylbenzene	Office	152	9
m-Methylethylbenzene	Office	168	8
Oxygenated organics			
Acetone	Dwelling	86	32
Methanol	School	11	29
Ethanol	Dwelling	39	120
Acetic acid	School	5	12
Butyric acid	School	5	25
Methyl ethyl ketone	Dwelling	316	4–21
Diethyl ketone	School	12	6
Phenol	School	5	9
Nonanal	Dwelling	15	7
Ethyl acetate	Dwelling	302	8
	School	12	10
Chlorinated organics			
Dichloromethane	Dwelling	101	17
Chloroform	Office	20	10
1,2-Dichloroethene	Dwelling	35	11
1,1,1-Trichloroethane	Dwelling	1580	24
Tetrachloroethene	Dwelling	1919	7
p-Dichlorobenzene	Dwelling	1881	8

[a] Adapted from Brown *et al.* (1994).

etate (used as a copolymer in the production of PVC), 1,2-propanediol, and 2,2,4-trimethylpentane, whereas the polyurethane backing emitted primarily 2,6-di-*tert*-butyl-4-methylphenol (BHT), a compound often used as an antioxidant. Many of the compounds emitted are known to be used in the manufacturing processes (e.g., *ε*-caprolactam is used in Nylon-6 production) and/or are common solvents. For example, such species as 1,2-propanediol, 2,2,4-trimethylpentane, 1,1,1-trichloroethane, toluene, 1-butanol, and the dipropylene glycol methyl ethers may be used during the production of various carpet components.

Emissions of VOC from carpets tend to decrease with time and increase with temperature. For example,

Fig. 15.9 shows the concentrations in a test chamber of 4-phenylcyclohexene, styrene, and 4-ethenylcyclohexene, another compound associated with SBR adhesives (Weschler *et al.*, 1992b), from one carpet as a function of time (Hodgson *et al.*, 1993). While styrene and 4-ethenylcyclohexene decrease rapidly with time, the decay of 4-phenylcyclohexene is much slower. In a house used for field studies, the concentration of 4-phenylcyclohexene remained above 2 ppb almost two months after installation, well above the odor threshold of 0.5 ppb.

Figure 15.10 shows the temperature dependence of the concentrations of two VOCs emitted by carpets, styrene and benzothiazole, into a test chamber (Sol-

TABLE 15.6 Some Organics Emitted from Carpets[a]

Compound	SBR[b] latex backing adhesive	PVC[b] backing	Polyurethane backing
4-Phenylcyclohexene	D	−	−
Styrene	D	−	−
Vinyl acetate	−	D	−
1,2-Propanediol	−	D	−
2,2,-4-Trimethylpentane	−	D	−
Formaldehyde	−	+	−
2-Ethyl-1-hexanol	−	+	−
2,6-Di-tert-butyl-4-methylphenol (BHT)	−	−	D
1-Butanol	−	−	+
Hexamethylcyclotrisiloxane	−	−	+
Dipropylene glycol methyl ethers	−	−	+
2-Methyl-1-propene	+	−	+
Acetaldehyde	−	+	−
Acetone	+	−	+
1- and 2-propanol	−	−	+
Trimethylsilanol	−	−	+
1,1,1-Trichloroethane	−	−	+
Acetic acid	−	+	−
Alkane HCs	−	+	−
2,2,5-Trimethylhexane	−	−	+
4-Ethenylcyclohexane	+	−	−
ε-Caprolactam	−	−	+

[a] Adapted from Hodgson et al. (1993); D = dominant emission, + = observed, − = not observed.
[b] SBR = styrene−butadiene rubber latex adhesive; PVC = polyvinyl chloride.

linger et al., 1994). Both increase with temperature, but styrene at a much slower rate. Given the temperature control in many homes, the change in emissions over most temperature ranges encountered in dwellings may not be large. However, in homes without temperature control or in automobiles, for example, the temperature dependence could be significant.

Emission rates from other building materials such as flooring, paints, varnishes, and sealants also tend to increase, not surprisingly, with temperature (e.g.,
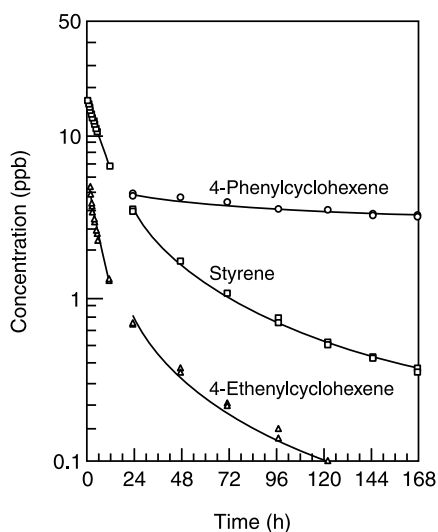

FIGURE 15.9 Concentrations of three VOCs emitted from a carpet sample made using SBR adhesives as a function of time in a test chamber (adapted from Hodgson et al., 1993).
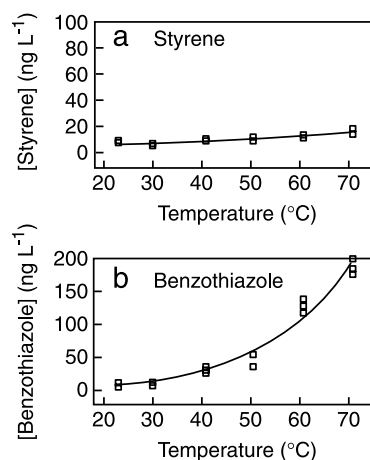

FIGURE 15.10 Temperature dependence of styrene and benzothiazole emitted from carpets into a test chamber (adapted from Sollinger et al., 1994).

TABLE 15.7   Effect of Temperature, Relative Humidity, and Replacement of Air by N_2 on VOC Emissions from Some Common Building Materials[a,b]

Building product	Temperature (°C)		RH (%)	Air replaced by N_2
	35	60		
Carpet				
2-Ethylhexanol	+	+	+	+
4-Phenylcyclohexene	+	+	NC	NC
PVC flooring				
2-Ethylhexanol	NC	+	NC	+
Phenol	NC	+	NC	+
Sealant				
Hexane	+	+	NC	NC
Dimethyloctanols	+	+	+	+
Varnish				
Butyl acetate	+	+	NC	NC
N-Methylpyrrolidone	+	+	NC	NC
Wall Paint				
1,2-Propanediol	NC	+	+	NC
Texanol[c]	NC	+	NC	NC

[a] Adapted from Wolkoff (1998).
[b] A "+" sign means an increase of 20% or more in concentration. NC means less than 20% change.
[c] Texanol = isomers of 2,4,4-trimethyl-1,3-pentanediol monoisobutyrate.

Wolkoff, 1998; Haghighat and de Bellis, 1998). Table 15.7, for example, shows the effects of temperature and relative humidity on the emissions of particular compounds associated with carpet, PVC flooring, sealants, varnish, and wall paint (Wolkoff, 1998). Interestingly, exposure of these samples to N_2 rather than air also increased the emissions in some cases. However, using increased temperatures to "bake-out" buildings and hence lower the concentrations of indoor VOCs does not appear to be particularly effective. For example, Bayer (1991) reports that the total VOC concentrations from particleboard are about the same after as before a 5-day bake-out at 88°C. Similarly, significant levels of HCHO have been observed in a mobile home even after 20 years of use in a hot ambient air environment (Pitts et al., 1989; see later).

The dependence of VOC emissions from building materials on relative humidity is more complex, with some emissions increasing with relative humidity, but others not. For example, Sollinger et al. (1994) report that the VOC emissions from carpets did not change with relative humidity over the range from 0 to 45% RH. On the other hand, the emissions of formic and acetic acids from latex paints have been reported to increase dramatically with relative humidity; for example, for one paint sample the emission rate for acetic acid almost tripled when the relative humidity was changed from 4–5% to 5–23% (Reiss et al., 1995b).

A number of different aldehydes have been measured indoors (e.g., see Crump and Gardiner, 1989; Lewis and Zweidinger, 1992; Zhang et al., 1994a; Daisey et al., 1994; and Reiss et al., 1995a), some of which are directly emitted and some of which are formed by chemical reactions indoors of VOCs such as styrene (see Section F). Of these, there is an enormous amount of evidence for direct emissions of HCHO from building materials. Interest in formaldehyde emissions and levels in homes and other buildings stems from its well-known health effects, which include possible human carcinogenicity and eye, skin, and respiratory tract irritation (Feinman, 1988). Formaldehyde is emitted from urea–formaldehyde foam insulation as well as from resins used in reconstituted wood products such as particleboard and plywood (Meyer and Reinhardt, 1986); urea–formaldehyde resins comprise about 6–8% of the weight of particleboard and 8–10% of medium-density fiberboard (Meyer and Hermanns, 1986). Other sources include permanent press fabrics (such as draperies and clothing), floor finishing materials, furniture, wallpaper, latex paint, varnishes, some cosmetics such as fingernail hardener and nail polish, and paper products (Kelly, 1996; Howard et al., 1998).

Many measurements of HCHO have been made in indoor air environments. In conventional homes, average concentrations are typically about 10–50 ppb (e.g., Stock, 1987; Zhang et al., 1994a; Reiss et al., 1995a). Sexton et al. (1989) measured concentrations of HCHO in 470 mobile homes in California and found geometric mean concentrations of 60–90 ppb, although maximum values of over 300 ppb were recorded in some cases. In a similar study in Wisconsin, levels up to 2.8 ppm were measured (Hanrahan et al., 1985). Higher levels are typically found in mobile homes because of the reconstituted wood products (e.g., particleboard and plywood) used in their construction. Interestingly, HCHO does not appear to be a significant product of natural gas combustion, as levels in dwellings with and without gas stoves turned on are not significantly different (e.g., Pitts et al., 1989; Zhang et al., 1994a).

Temperature is again an important determinant of HCHO levels. Figure 15.11, for example, shows the concentrations of HCHO as well as of formic acid and methanol measured using FTIR in a research mobile home as function of time as the temperature increased. At 70°F, the average HCHO concentration was 27 ppb but increased to 105 ppb at 100°F (Pitts et al., 1989).
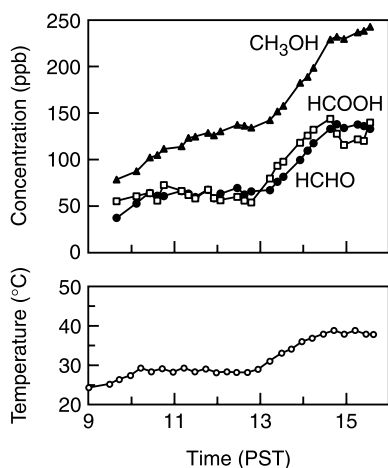
FIGURE 15.11   Concentrations of HCHO, HCOOH, and CH$_3$OH measured using long-path FTIR in a mobile home as a function of temperature. Although the gas stove burners were on, these were shown in separate experiments not to be the source of these organics (adapted from Pitts *et al.*, 1989.)

HCOOH has also been observed from outgassing in a mobile home trailer by chemical ionization mass spectrometry during measurements of HNO$_3$ (Huey *et al.*, 1998) and at lower concentrations (mean of 10 ppb) in conventional homes (Reiss *et al.*, 1995a).

Figure 15.12 summarizes the ratio of indoor-to-outdoor concentrations of HCHO and higher aldehydes as well as formic and acetic acids measured in some conventional homes. Concentrations of all of these compounds, except possibly propionaldehyde, are significantly higher indoors, suggesting that not only

HCHO but higher aldehydes and ketones as well as acids have significant indoor sources (Lewis and Zweidinger, 1992; Zhang *et al.*, 1994a, 1994b; Zhang and Lioy, 1994; Reiss *et al.*, 1995a). As discussed later, reactions of hydrocarbons with ozone indoors is a potential source, in addition to direct emissions.

It should be noted that while building materials are sources of a variety of VOCs, they can also adsorb organics as well (e.g., Van Loy *et al.*, 1997). As a result, building surfaces and contents may act as reservoirs of organics, slowly releasing compounds over a period of time.

Anthropogenic activities. There are many sources of VOCs associated with human activities in buildings. For example, mixtures of C$_{10}$ and C$_{11}$ isoparaffinic hydrocarbons, which are characteristic of liquid process copiers and plotters, have been identified in office buildings in which these instruments were in use (Hodgson *et al.*, 1991). Emissions of a number of hydrocarbons and aldehydes and ketones have been observed during operation of dry-process copiers; these include significant emissions of ethylbenzene, *o*-, *m*-, and *p*-xylenes, styrene, 2-ethyl-1-hexanol, acetone, *n*-nonanal, and benzaldehyde (Leovic *et al.*, 1996). Enhanced levels of acetaldehyde in an office building in Brazil were attributed to the oxidation of ethanol used as a cleaning agent (Brickus *et al.*, 1998), although levels outdoors were also enhanced due to the use of ethanol as a fuel (see Chapter 16.D.4). Pyrocatechol has been measured in an occupational environment where meteorological charts are mapped on paper impregnated with this compound (Ekinja *et al.*, 1995), and
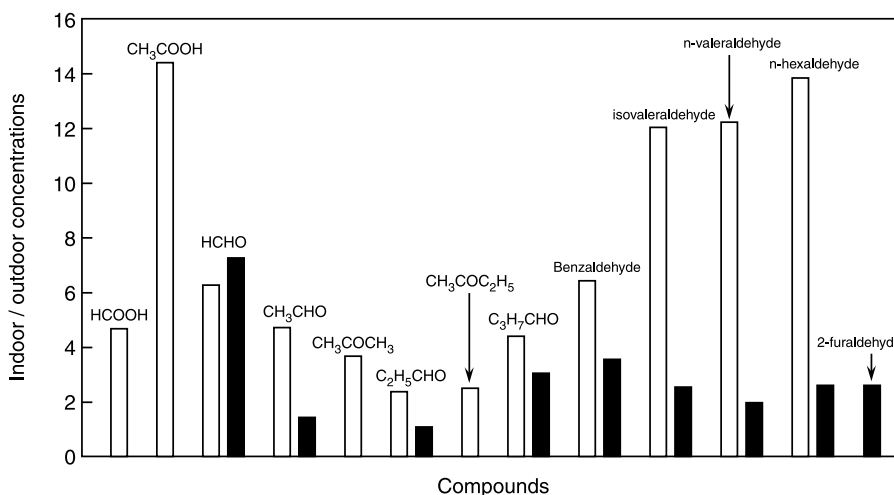

FIGURE 15.12   Ratio of indoor-to-outdoor concentrations of formic and acetic acids and some carbonyl compounds in some homes: unshaded, data from Reiss *et al.*, 1995a, for summer data; shaded, data from Zhang *et al.*, 1994a.

p-dichlorobenzene is observed when mothballs containing this compound are in use (e.g., Tichenor *et al.*, 1990; Chang and Krebs, 1992).

Elevated concentrations of the n-C_{13} to n-C_{18} alkanes and branched-chain and cyclic analogs were measured in a building having a history of air quality complaints; the source was found to be volatilization from hydraulic fluids used in the building elevators (Weschler *et al.*, 1990).

Enhanced levels of chlorinated compounds have been observed indoors due to human activity as well. For example, increased levels of perchloroethylene have been observed from unvented dry-cleaning units (e.g., Moschandreas and O'Dea, 1995) and volatilization of chlorinated organics such as chloroform from treated tap water can occur (e.g., McKone, 1987). Other sources include the use of household products. For example, chloroform emissions have been observed from washing machines when bleach containing hypochlorite was used (Shepherd *et al.*, 1996). It is interesting that emissions of organics associated with the use of washing machines are decreased when the machine is operated with clothes inside (Howard and Corsi, 1998).

Of course, activities such as smoking result in enhanced levels not only of nicotine (e.g., Thompson *et al.*, 1989) but also of a variety of other gases associated with cigarette smoke (e.g., California Environmental Protection Agency, 1997; Nelson *et al.*, 1998). For example, using 3-ethenylpyridine as a marker for cigarette smoke, Heavner *et al.* (1992) estimated that 0.2–39% of the benzene and 2–49% of the styrene measured in the homes of smokers were from cigarette smoke.

Humans emit a variety of VOCs such as pentane and isoprene (e.g., Gelmont *et al.*, 1981; Mendis *et al.*, 1994; Phillips *et al.*, 1994; Jones *et al.*, 1995; Foster *et al.*, 1996). In addition, emissions from personal care products have been observed. Decamethylcyclopentasiloxane (D5), a cyclic dimethylsiloxane with five Si–O units in the ring, and the smaller D4 analog, octamethylcyclotetrasiloxane, are used in such products as underarm deodorant and antiperspirants at concentrations up to 40–60% by weight (Shields and Weschler, 1992; Shields *et al.*, 1996). Increased concentrations of D5 have been measured in offices and are correlated to human activity, as expected if personal care products were the major source (Shields and Weschler, 1992). In some cases, increased concentrations attributable to emissions from silicone-based caulking materials were also observed (Shields *et al.*, 1996).

The use of pesticides indoors can lead to very large concentrations not only of the pesticide but of the additional VOCs used as a matrix for the pesticide, which represent most (>95%) of the mass of the material as purchased. For example, Bukowski and Meyer (1995) predict that VOC concentrations immediately after the application of a fogger could reach levels of more than 300 mg m^{-3}!

However, lower levels of pesticides themselves are common after use inside homes. For example, Lewis *et al.* (1988) reported the presence of 24 pesticides in homes, ranging in concentration from 0.002 to 15 μg m^{-3} (the latter for chlorpyrifos), while outdoor levels measured simultaneously were much lower, from <0.001 to 0.4 μg m^{-3}. The compounds present indoors at the highest concentrations were those used recently at the home. Similarly, Whitmore *et al.* (1994) reported 22 pesticides in homes in Jacksonville, Florida, and Springfield, Massachusetts, at indoor levels up to 0.5 μg m^{-3} indoors, but only 0.04 μg m^{-3} outdoors. The pesticides not only were present as gases but also adsorbed to dust particles in the home, particularly for the less volatile compounds. Indeed, higher concentrations of some pesticides have been found in dust than in air (e.g., Roinestad *et al.*, 1993).

There are a few data that suggest that pesticides can undergo reactions indoors. For example, Wallace *et al.* (1996) observed that the aldrin levels inside a home decreased with time, whereas those of dieldrin did not. Dieldrin had been applied with aldrin but is also an oxidation product of aldrin. One of the reasons for the lack of change in dieldrin may be that it was being formed as the aldrin decayed; however, this could not be differentiated from the effects of a lower vapor pressure of dieldrin, which could lead to lower overall removal rates. In the same study, pentachloroanisole was also measured inside the home and attributed to formation by degradation of pentachlorophenol, which is used as a wood preservative and termiticide.

Not surprisingly, some indoor organics are readily taken up on building surfaces, such as carpets and wallboard, and are subsequently released into the room. For example, Chang *et al.* (1998) showed that some of the alcohols found in indoor air environments are taken up by carpets and gypsum board and could be desorbed back into the gas phase later. However, revolatilization was observed to be slow, and Chang and co-workers estimated that it would take more than a year to remove the adsorbed organics. Similarly, Van Loy *et al.* (1998) showed that nicotine from environmental tobacco smoke can be readily adsorbed and then desorbed and that surfaces can hold significant amounts of nicotine. As a result of this reversible adsorption–desorption process, measurable levels of organics can be maintained indoors after the initial exposure by slow degassing from surfaces.

Not surprisingly, the concentrations of VOCs from automobile exhaust are higher in the "indoor environment" of automobiles during commutes. For example,

Duffy and Nelson (1997) report during commutes in Sydney, Australia, that the benzene concentrations inside vehicles were 10–25 times those in ambient air and that the concentrations of 1,3-butadiene were more than 55–115 times greater. The source appeared to be primarily from the exhausts of surrounding vehicles. Similar enhancements of benzene and other VOCs such as toluene, ethylbenzene, and the xylenes in automobiles and buses have been reported in many countries, including Korea (Jo and Choi, 1996), Taiwan (Chan *et al.*, 1993), and the United States (e.g., Chan *et al.*, 1991a,b; Lawryk and Weisel, 1996).

FIGURE 15.13 Measured concentrations of outdoor O$_3$ and indoor O$_3$ with and without a gas stove on: horizontal bars, outdoor O$_3$; solid squares, indoor O$_3$ (adapted from Zhang and Lioy, 1994).

E. OZONE

Because O$_3$ decomposes on surfaces, indoor levels are usually lower than those outdoors due to the decomposition that occurs as the air passes through air conditioning systems and impacts building surfaces (e.g., Reiss *et al.*, 1994). The measured ratio of indoor-to-outdoor concentrations of ozone vary from 0.1 to 1, but are typically around 0.3–0.5 (e.g., Druzik *et al.*, 1990; Hisham and Grosjean, 1991b; Liu *et al.*, 1993; Weschler *et al.*, 1989, 1994; Gold *et al.*, 1996; Jakobi and Fabian, 1997; Avol *et al.*, 1998; Drakou *et al.*, 1998; Romieu *et al.*, 1998). Buildings with low air exchange with outside air tend to have lower ratios, ~ 0.1–0.3 (Druzik *et al.*, 1990; Weschler *et al.*, 1994; Romieu *et al.*, 1998). For example, Gold *et al.* (1996) estimate that at outdoor ozone concentrations of 170 ppb in Mexico City, the indoor-to-outdoor ratio of O$_3$ at a school was 0.71 ± 0.03 with the windows and doors open, which maximized the exchange with outside air, 0.18 ± 0.02 with the windows and doors closed and the air cleaner off, and 0.15 ± 0.02 with the windows and doors closed and the air cleaner on.

There are some additional sources of O$_3$ indoors. These include dry-process photocopying machines, laser printers, and electrostatic precipitators (e.g., Leovic *et al.*, 1996; Wolkoff, 1999). Indeed, it is not unusual to detect O$_3$ by its odor during operation of some copy machines and laser printers.

In the "indoor environment" in cars, ozone levels tend to be significantly less than in the surrounding area. For example, Chan *et al.* (1991b) report that in-vehicle O$_3$ concentrations during commutes in Raleigh, North Carolina, were only about 20% of those measured in the local area at a fixed station. There are several contributing factors to these low concentrations. One is that NO concentrations are higher near roadways, so that O$_3$ is titrated to NO$_2$ by its rapid reaction with NO. A second is that O$_3$ can decompose on the surfaces of the automobile air conditioning system.
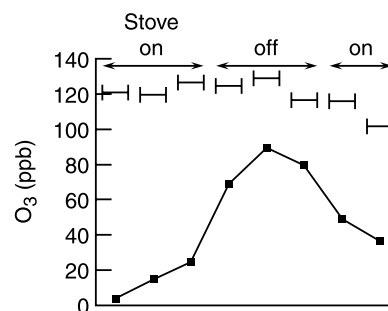
A similar titration effect has been observed inside homes where there are combustion sources of NO. Figure 15.13 shows measured outdoor and indoor levels of O$_3$ in one study with a gas stove turned on or off. Although the outdoor concentration remained relatively constant, the indoor levels were much lower when the gas stove was on compared to when it was off. Although NO$_2$ was not reported in this study, its levels presumably rose when the stove was on as the O$_3$ reacted with NO to form NO$_2$.

F. INDOOR VOC – NO$_X$ – O$_3$ CHEMISTRY

As discussed in detail throughout this book, there is rich and complex chemistry involving volatile organic compounds (VOCs), oxides of nitrogen, and ozone in ambient air. One might therefore anticipate similar chemistry in indoor air environments, and although there are far fewer studies, this does indeed appear to be the case. Weschler and Shields (1997b) and Wolkoff *et al.* (1997, 1999) review VOC–NO$_x$ chemistry that could potentially be important in indoor air enviroments and the implications for human exposures.

Although the chemistry occurring indoors is fundamentally the same as that occurring outdoors and discussed throughout this book, there are some important differences as well. For example, the time available for the chemistry to occur indoors is determined by the ventilation rate of the building. In addition, the light intensity and spectral distributions are quite different from those outdoors, decreasing the relative importance of photochemical reactions and increasing that of "dark reactions." For example, Pitts *et al.* (1985) followed the loss of methyl nitrite (CH$_3$ONO) in brightly lit rooms indoors in a mobile home used for the HONO

studies; no loss was observed over a 5-h period, whereas this compound photolyzes rapidly outdoors.
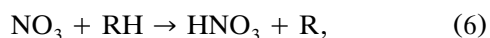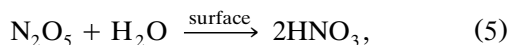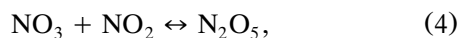
Finally, the concentrations of the reactants, both absolute and relative, are different from those in most outdoor environments. As we have already seen, the concentrations of VOCs tend to be larger due to the large number of sources indoors, whereas that of O_3 tends to be smaller due to its removal on surfaces, e.g., of air conditioning systems. In the absence of combustion sources, concentrations of oxides of nitrogen tend to be similar to those outdoors.

A number of models that incorporate chemical reactions have been developed and applied to indoor air environments. For example, Özkaynak et al. (1982) developed a model that incorporated simple NO_x chemistry. A more comprehensive model that included VOC–NO_x chemistry as well was developed by Nazaroff and Cass (1986) for indoor air environments (it included ventilation, emissions, and also removal by wall loss) and applied to museum environments. Measurements of NO, NO_2, and O_3 both indoors and outdoors were also made for testing and refining the model predictions. As expected based on the discussion of ozone in Section E, O_3 concentrations indoors, both measured and predicted, were lower than those outdoors due to removal at surfaces; however, removal by reaction with NO, particularly in the morning when NO concentrations were high, was also a significant contributor. The model tended to consistently underpredict NO indoors, which they attribute to its generation in the surface reaction of NO_2 discussed earlier, which was not included in the model chemistry. Although other species were not measured, the results of the model suggest that NO_3 and N_2O_5 levels indoors are greater than those outdoors and that under some circumstances, the same may be true for HNO_3 and H_2O_2.

Weschler and co-workers (Weschler et al., 1992a, 1994) suggested that the formation of NO_3 indoors,

$$O_3 + NO_2 \rightarrow NO_3 + O_2, \qquad (3)$$
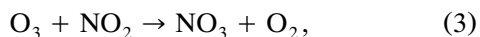
followed by its well-known secondary chemistry to form HNO_3,

$$NO_3 + NO_2 \leftrightarrow N_2O_5, \qquad (4)$$

$$N_2O_5 + H_2O \xrightarrow{\text{surface}} 2HNO_3, \qquad (5)$$

$$NO_3 + RH \rightarrow HNO_3 + R, \qquad (6)$$

may be an important indoor source of HNO_3. Indirect evidence for an indoor source is the measurement of the indoor-to-outdoor ratio (I/O) of HNO_3 and SO_2 in residences in several locations on the east coast of the United States during the summer months. The I/O ratio for HNO_3 was about the same as that for SO_2 at one location and larger at a second site. This is not expected, since HNO_3 is much more rapidly lost to surfaces than is SO_2. In a third location, the I/O ratio for HNO_3 was smaller than that of SO_2, but not by as much as would be expected from its rapid wall loss. Such data are indicative of an indoor source of HNO_3, and Weschler et al. (1992a) suggest reactions (3)–(6).

Evidence for such indoor chemistry has also been obtained by measuring NO, NO_y, and O_3 indoors and outdoors in an office building (Weschler et al., 1994). For example, as O_3 rises and NO decreases during the morning hours, the O_3 reaction with NO indoors leads to faster decay of NO than otherwise expected and a slower rate of increase of O_3. On the other hand, when NO is rising and O_3 falling in the early evening, this reaction speeds up the decay of O_3 and slows the increase in NO.

Weschler and Shields (1997b) suggest that with the higher concentrations of VOCs indoors, their reactions with O_3, NO_3, and OH may be important. There is some experimental evidence that this is indeed the case. For example, carpet exposed to O_3 in a chamber generated HCHO, benzaldehyde, benzoic acid, and acetophenone, all expected products from the reaction of O_3 with styrene (Zhang et al., 1994b) emitted from the latex adhesive used to bind the backing to the carpet; styrene decreased simultaneously (Weschler et al., 1992b). The formation of a series of C_5–C_{10} aldehydes was observed, which appeared to be from the reaction of O_3 with nonvolatile organics associated with the carpet fibers. Concentrations of HCHO increased by up to a factor of 3 and CH_3CHO by up to a factor of 20 in the presence of O_3. Interestingly, no additional effect was observed when NO_2 was also present, suggesting that the nitrate radical was not a significant contributor to the formation of these aldehydes and ketones.

Salthammer et al. (1999) examined emissions from commonly available coatings used on furniture and identified numerous oxidation products. These were observed without the addition of oxidants such as ozone, indicating that oxidation in air (perhaps including photodecomposition for some compounds) under typical conditions is sufficient to generate such products. For example, emissions of 2-ethylhexanol were identified from di-2-ethylhexyl phthalate, used as a plasticizer in many coatings.

Reiss et al. (1995b) exposed latex paint to O_3 and observed the production of HCHO as well as CH_3CHO and CH_3COCH_3 for some paint samples. They proposed that these were formed by the reactions of O_3 with some remaining double bonds that were not fully reacted during the process in which the $CH_2=CHR$

was polymerized to form the latex paint. Similarly, Chang and Guo (1998) and Fortmann *et al.* (1998) report emissions of hexanal during the drying of an alkyd paint; since hexanal was not a component of the paint itself, they proposed that it was formed by the oxidation of unsaturated fatty acid esters in the alkyd resin.

A correlation between indoor ozone and the concentrations of carbonyl compounds and organic acids in homes has been reported in several studies and attributed at least in part to indoor O_3 reactions (e.g., Reiss *et al.*, 1995a; Zhang and Lioy, 1994; Zhang *et al.*, 1994a–c). Not only HCHO and CH_3CHO but also larger aldehydes have been measured indoors, with indoor concentrations for all but possibly propionaldehyde being much larger than those outdoors (Fig. 15.12). The same is true for formic and acetic acids, which can be formed by ozone reactions from reaction of the Criegee biradicals with water vapor (see Chapter 6.E.2). Zhang *et al.* (1994c) report that indoor formic acid concentrations increased with the indoor concentration of O_3 and with relative humidity, as expected if the reaction of the HCHOO Criegee biradical with water vapor was a significant indoor source.

However, as discussed by Reiss *et al.* (1995a), separating the contribution of ozone reactions from other factors such as temperature and relative humidity, which also affect direct emissions, is difficult. For example, while the production rate of oxygenated organics is correlated with the ozone removal rate, the latter is also correlated with temperature. As a result, both reaction and increased direct emission rates due to higher temperatures may be contributing to these enhanced indoor levels.

Ozone can also react with components found in air ducts. For example, Morrison *et al.* (1998) reported that the sealant and neoprene gaskets used in the ducts emitted VOCs into the airstream, but at relatively low levels compared to the typical concentrations found indoors. However, reaction with O_3 led to increased emissions of aldehydes, particularly the C_5–C_{10} aldehydes.

The mechanism of the reaction of O_3 with alkenes was discussed in detail in Chapter 6.E.2. It was seen there that these reactions serve as an indoor source of OH through decomposition of the Criegee biradical. Weschler and Shields (1996b) proposed that the indoor reaction of O_3 with alkenes could serve as a source of OH and calculated that at 20 ppb O_3 and average indoor alkene concentrations, a steady-state OH concentration of about 2×10^5 molecules cm^{-3} might be expected.

Subsequently, they measured OH concentrations in an office building using the rate of decay of 1,3,5-tri-

methylbenzene, which reacts with OH but not O_3 (Weschler and Shields, 1997a). Although O_3 and *d*-limonene were injected during the experiment, their concentrations were chosen to be similar to that measured under normal operating conditions. An average OH concentration of 7×10^5 molecules cm^{-3} was obtained, which is about an order of magnitude smaller than daytime peak OH concentrations outdoors but also more than an order of magnitude larger than those outdoors at night (see Chapter 11.A.4i).

As discussed in Chapter 9.C.2, some of the larger alkenes such as terpenes form particles containing low-volatility organics on oxidation with ozone. Hence particle formation might be expected indoors in the presence of such compounds, and indeed this has been observed (Weschler and Shields, 1999).

In short, much of the chemistry that has been observed outdoors also occurs indoors. However, the relative importance of various reactions may be somewhat different due to the different absolute and relative concentrations of the reactants, the lower photolysis rates, the exchange of air with outdoor air, and the presence of relatively large surface areas, which can both remove various species and act as substrates for heterogeneous reactions.

G. PARTICLES

With the epidemiological studies suggesting increased mortality associated with particles (see Chapter 2), there has been increasing interest in indoor particle concentrations compared to outdoor levels. A number of studies have examined this over the years and are summarized in a review by Wallace (1996).

In general, if there are no indoor sources of particles, the levels indoors tend to reflect those outdoors. For example, application of a mass balance model to measurements of indoor and outdoor particle concentrations in Riverside, California, indicated that 75% of $PM_{2.5}$ and 65% of PM_{10} in a typical home were from outdoors (Wallace, 1996). Similar conclusions were reached by Koutrakis *et al.* (1991, 1992) for homes in two counties in New York. For example, they report that 60% of the mass of particles in homes is due to outdoor sources. However, the contribution to various individual elements in the particles varies from 22% for copper to 100% for cadmium.

There are some differences in indoor levels of particulate matter in areas with low outdoor compared to high outdoor levels. In the case of high outdoor levels, the indoor concentrations tend to be somewhat lower than those outdoors; for example, Colome *et al.* (1992)
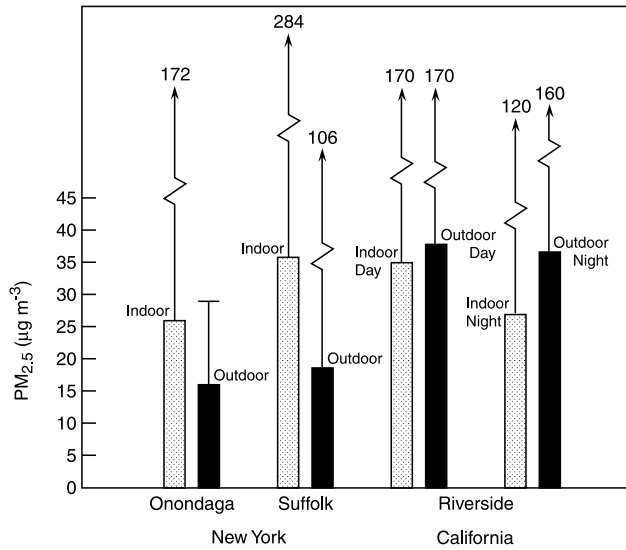
FIGURE 15.14 Geometric mean $PM_{2.5}$ concentrations indoors and outdoors in Onondaga and Suffolk Counties in New York state and in Riverside, California. The Riverside data are weighted means. The error bars represent the maximum values for the New York sites and the 98th percentile for the Riverside studies (data from Wallace, 1996; based on Sheldon *et al.*, 1989, and Pellizzari *et al.*, 1993).
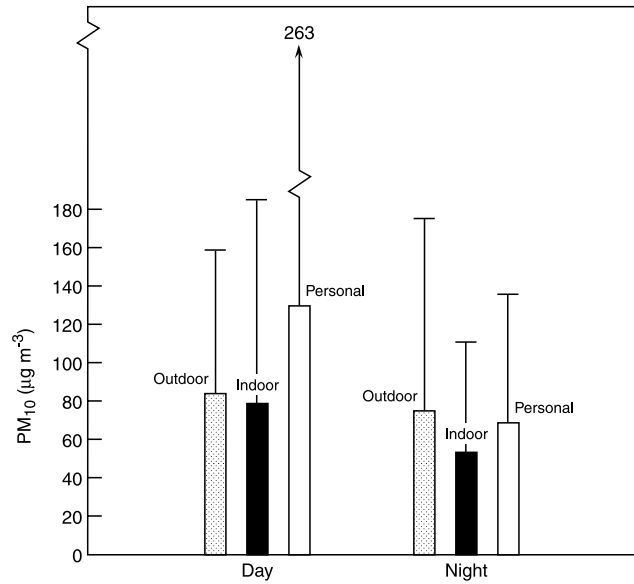

FIGURE 15.15 Geometric mean PM_{10} concentrations outdoors, indoors, and by personal exposure monitors during the day and at night in Riverside, California, weighted to provide estimates of concentrations for household-days or person-days. The error bars are 90th percentile (data from Clayton *et al.*, 1993).

report that the ratio of indoor-to-outdoor median concentrations of PM_{10} is 0.7 in residences in southern California. On the other hand, when outdoor levels are low, indoor levels tend to be higher. This is illustrated by the data in Fig. 15.14, which shows indoor and outdoor levels of $PM_{2.5}$ measured in two counties in New York state (Sheldon *et al.*, 1989) and in Riverside, California (Pellizzari *et al.*, 1993), as summarized by Wallace (1996).

Nighttime mass concentrations indoors tend to be smaller than those during the day, probably because of the decreased activity. Interestingly, when individuals wear personal exposure monitors to measure their actual exposure to particles, the measured mass concentrations tend to be higher than those measured with fixed monitors located indoors. Both of these are illustrated by the data in Fig. 15.15, which shows the geometric mean concentrations of PM_{10} in Riverside, California, measured outdoors, indoors, and with personal exposure monitors (Clayton *et al.*, 1993). Because of the study design, the measurements were weighted to provide estimates of the concentrations in terms of household-days or person-days, respectively. Indoor levels at night are about 50–70% of those during the day. However, the personal exposure concentrations are 165% of the measured indoor concentrations during the day and 128% at night. This has been dubbed the "body cloud" or "personal cloud" (Rodes, 1991). The reasons for the higher concentrations are not

clear. While skin flakes and clothing fibers contribute to this, their levels do not appear to be sufficiently high to explain the observed levels (Özkaynak *et al.*, 1996). In addition, the elemental composition of the personal exposure particles in some studies has been observed to be very similar to that of indoor aerosol samples, indicating that different sources are not likely to be responsible (Özkaynak *et al.*, 1996; Wallace, 1996). It may be that the activity of the individual is responsible, at least in part. For example, carpets have been shown to be a reservoir of dust particles that can be suspended when walking, vacuuming, etc. (Thatcher and Layton, 1995; Leese *et al.*, 1997). Such activity leads to resuspension of the larger particles preferentially. For example, cleaning activities increased the airborne concentrations of 10- to 25-μm particles by a factor of 30, of 5- to 10-μm particles by a factor of 11, and of 0.5- to 1-μm particles insignificantly (Thatcher and Layton, 1995). Enhanced levels of particulate sulfate and H^+ in personal exposure monitors compared to indoor values (ratios of 1.25 and 3.1, respectively) have been reported (Suh *et al.*, 1992), although Özkaynak *et al.* (1996) report that sulfur in particles in the "personal cloud" was the only element not enhanced compared to indoor levels.

A major source of increased particles indoors is cigarette smoking. (e.g., Spengler *et al.*, 1981; Quackenboss *et al.*, 1989; Neas *et al.*, 1994). Figure 15.16 shows
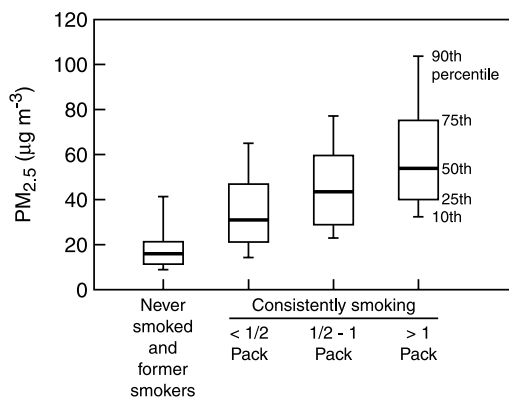
FIGURE 15.16 Annual average concentrations of $PM_{2.5}$ indoors as a function of smoking in the home. The data are shown in percentiles as marked. (Adapted from Neas *et al.*, 1994.)

the annual average concentrations of $PM_{2.5}$ measured indoors in homes with nonsmokers and for homes occupied by smokers as a function of the number of packs of cigarettes smoked per day (Neas *et al.*, 1994). There is a steady increase in the indoor levels of $PM_{2.5}$ as a function of the number of cigarettes smoked, with the increase in $PM_{2.5}$ estimated to be in the range of $25-45$ μg m^{-3} (Wallace, 1996).

In addition to the contribution to the *mass* concentrations of indoor particles, cigarette smoke is of concern because of the mutagens, carcinogens, and toxic air contaminants that are emitted (e.g., see Löfroth *et al.*, 1991; Chuang *et al.*, 1991; California Environmental Protection Agency, 1997; and Nelson *et al.*, 1998). Thus, a variety of both gaseous and particulate polycyclic aromatic hydrocarbons (PAH) and compounds (PAC) have been identified in buildings with cigarette smoke (e.g., Offermann *et al.*, 1991; Mitra and Ray, 1995). Indeed, in the homes of smokers, almost 90% of the total PAH was from tobacco smoke (Mitra and Ray, 1995). Higher levels of mutagenic particles have also been shown to be associated with indoor air containing cigarette smoke (e.g., Lewtas *et al.*, 1987; Löfroth *et al.*, 1988, 1991; Georghiou *et al.*, 1991).

Other significant sources identified in a number of studies are cooking, the use of kerosene heaters, wood burning, and humidifiers. For example, a study carried out under the auspices of the U.S. Environmental Protection Agency, the TEAM study (*T*otal *E*xposure *A*ssessment *M*ethodology), indicated that an increase in PM_{10} of $\sim 10-20$ μg m^{-3} could be attributed to cooking (Wallace, 1996). This source will obviously depend on the amount of cooking, the types of cooking, and the ventilation. For example, Löfroth *et al.* (1991) measured emissions of particles ranging from 0.07 to 3.5 mg per gram of food cooked, depending on the

particular food. Baek *et al.* (1997) measured indoor and outdoor concentrations of particles in homes, offices, and restaurants in Korea and report ratios of 1.3, 1.3, and 2.4, respectively. The higher value in restaurants, even those using only gas and not charcoal, suggests a significant contribution from cooking.

Kerosene heaters can be significant sources of particles under some circumstances. For example, kerosene heaters were reported to contribute to indoor $PM_{2.5}$ in homes in Suffolk County, New York, but not Onondaga County; wood stoves and fireplaces and gas stoves did not contribute in either case (Koutrakis *et al.*, 1992; Wallace, 1996). A similar conclusion was reached in a study of eight mobile homes in North Carolina (Mumford *et al.*, 1991).

However, it should be noted that even where kerosene heaters do not contribute significantly to particle *mass* concentrations, they may still be important in terms of health effects. This is because of the composition of the particles emitted, which include polycyclic aromatic compounds and other mutagenic species, as well as sulfate (Leaderer *et al.*, 1990; Traynor *et al.*, 1990). For example, Traynor *et al.* (1990) studied the emissions from unvented kerosene space heaters and identified a number of PAHs (naphthalene, phenanthrene, fluoranthene, anthracene, chrysene, and indeno[*c,d*]pyrene) and nitro-PAHs (1-nitronaphthalene, 9-nitroanthracene, 3-nitrofluoranthene, and 1-nitropyrene), in addition to a host of other gaseous species. Baek *et al.* (1997) also reported increased levels of a number of gases indoors in homes and offices in Korea due to the use of kerosene heaters.

In studies of indoor air in eight mobile homes, Mumford *et al.* (1991) identified the PAHs and nitro-PAHs measured in emissions from kerosene heaters by Traynor *et al.* (1990), as well as a number of compounds that may be animal carcinogens, such as cyclopenta[*c,d*]pyrene, benz[*a*]anthracene, benzofluoranthenes, benzo-[*a*]pyrene, and benzo[*ghi*]perylene. While the mass concentrations of PM_{10} did not increase with the kerosene heater on in six of the eight homes studied, the particles in five of the homes had increased mutagenicity using TA98 with or without S9 added (see Chapter 10.C.4).

In short, not only the mass emissions but also the nature of the compounds emitted must be taken into account in assessing the health effects of indoor particles.

Humidifiers can be a significant source of airborne particles if tap water is used because as the water evaporates from the aerosol, the solids that were in water are left as particles. For example, Highsmith *et al.* (1988, 1992) showed that the airborne particle concentration increased linearly with the total dissolved

solids in the water used in the humidifier. When an ultrasonic humidifier was operated in a closed single room using tap water with total dissolved solids of 303 mg L^{-1}, concentrations of fine and coarse particles of 6307 and 771 $\mu g \ m^{-3}$, respectively, were generated. When distributed over the whole house, the corresponding levels were 593 and 65 $\mu g \ m^{-3}$, respectively. The whole-house values fell to 41 and 13 $\mu g \ m^{-3}$ when bottled water with total dissolved solids of 24 mg L^{-1} was used (Highsmith et al., 1992).

On the other hand, the use of evaporative ("swamp") coolers appears to decrease particulate matter indoors. For example, Quackenboss et al. (1989) report levels of $PM_{2.5}$ and PM_{10} in homes having such coolers (for both smokers and nonsmokers) that are about half that of homes without them.

Where indoor heating and cooking involves the use of coal or biomass, indoor particle concentrations can be extremely large. For example, Florig (1997) and Ando et al. (1996) report that in China typical indoor total suspended particle (TSP) concentrations can be in the range from 250 to 900 $\mu g \ m^{-3}$ in homes using coal and 950–3500 $\mu g \ m^{-3}$ in those using biomass fuels. These levels can be compared to annual average outdoor concentrations of 250–410 $\mu g \ m^{-3}$. The high concentrations associated with coal burning combined with the mutagenic nature of the emissions have been suggested to be responsible for enhanced lung cancer in China (Mumford et al., 1987). Similarly, Davidson et al. (1986) measured TSP concentrations of 2900–42,000 $\mu g \ m^{-3}$ in homes in Nepal that used biomass fuels, compared to outdoor levels of 280 $\mu g \ m^{-3}$. For particles with diameters less than 4 μm, the levels ranged from 870 to 14,000 $\mu g \ m^{-3}$.

Similar conclusions regarding the relative indoor and outdoor concentrations have been reached in studies of office and commercial buildings. For example, Ligocki et al. (1993) measured indoor and outdoor concentrations of particles and their components at five museums in southern California. The indoor-to-outdoor ratios of particle mass varied over a wide range, depending to a large extent on the ventilation and filtration systems in use. Ratios varied from 0.16 to 0.96 for particles with diameters less than 2.1 μm and from 0.06 to 0.3 for coarse particles with diameters greater than this.

The chemical composition of particles collected in the museums was also compared to that outdoors using a mass balance model. The results indicated that there were significant indoor sources of fine particle organics and that this source(s) was a significant fraction of the total indoor fine particle organic concentration (Ligocki et al., 1993). A similar conclusion was reached by Naik et al. (1991) in measurements made in a telephone switching office. The levels of the n-C_{27} through n-C_{33} alkanes in the fine particle fraction were all elevated, suggesting indoor sources such as waxes, polishes, and lubricants. Enhanced levels of dibutyl phthalate and di(2-ethylhexyl) phthalate were also observed in fine particles and attributed to plasticizers used in floor polishes and vinyl products.

Turk et al. (1989) measured particle concentrations in 38 buildings that had both smoking and nonsmoking areas. The average mass concentration of respirable particles in smoking areas was 70 $\mu g \ m^{-3}$ compared to 19 $\mu g \ m^{-3}$ in nonsmoking areas, whereas the outdoor concentrations were essentially the same, 19 $\mu g \ m^{-3}$.

In another study, Ott et al. (1996) measured particle concentrations inside and outside a tavern before and after a smoking ban was instated. Average respirable suspended particle mass concentrations were 57 $\mu g \ m^{-3}$ above the outdoor concentrations prior to the ban, compared to 6–13 $\mu g \ m^{-3}$ afterward. Cooking and resuspended dust also contributed to the indoor particle mass concentration, but at concentrations about 20–25% of that due to cigarette smoke.

In short, the indoor concentrations of particles depend on the outdoor levels, the ventilation system and exchange rates, and the presence of indoor sources such as cigarette smoke.

In many nonresidential buildings, deposition is of particular interest because of the potential for damage to materials in museums, offices, cultural objects, and industrial sites (e.g., Sinclair et al., 1988, 1990a,b; Nazaroff et al., 1990a; Salmon et al., 1994, 1995). Deposition onto both horizontal and vertical surfaces is of concern, but these can show different behaviors as a function of particle size. For example, deposition of particles to vertical and horizontal surfaces was measured inside five museums. Horizontal deposition velocities increased from $\sim 10^{-6}$ to 10^{-3} m s^{-1} as particle size increased from about 0.1 to 30 μm as expected for gravitational settling (see Chapter 9.A.3). However, the dependence on particle size of the uptake onto vertical surfaces, which is influenced by thermal and air flow fields (Nazaroff et al., 1990b), was variable, increasing with particle size in some cases, decreasing in others, and, in some, showing no dependence on particle size (Ligocki et al., 1990).

A number of models have been developed for particles indoors (e.g., Nazaroff and Cass, 1989a; Sinclair et al., 1990b; Nazaroff et al., 1990a; Weschler et al., 1996; Wallace et al., 1996, and references therein). This is a complex problem, given the number of potential sources, different deposition velocities for particles of different sizes (e.g., see Chapter 9.A.3 and Nazaroff and Cass (1989b)), the different particle compositions, and the effects of outdoor concentrations and ventila-

tion rates. However, reasonably good agreement has been obtained in many cases between modeled and observed concentrations.

H. PROBLEMS

1. As discussed in this chapter, a relatively new area in indoor air pollution is that of hydroxyl radical chemistry. However, the importance of indoor OH chemistry (as well as O_3 and NO_3 chemistry) is determined by the rates of the reactions compared to the rate of air exchange. An OH concentration of 7×10^5 cm^{-3} has been reported in an indoor air environment by Weschler and Shields (1997a). Assess the importance of the OH reaction for the removal of limonene indoors compared to its removal by reaction with O_3. The O_3–limonene rate constant is 2×10^{-16} and the OH–limonene rate constant is 1.7×10^{-10} cm^3 molecule^{-1} s^{-1}. Take the O_3 concentration to be 20 ppb. How do these compare with a typical air exchange rate of 0.75 h^{-1} used in these experiments?

References

Alter, H. W., and R. A. Oswald, "Nationwide Distribution of Indoor Radon Measurements: A Preliminary Data Base," *JAPCA, 37,* 227–231 (1987).

Anderson, D. J., and R. A. Hites, "Chlorinated Pesticides in Indoor Air," *Environ. Sci. Technol., 22,* 717–720 (1988).

Ando, M., K. Katagiri, K. Tamura, S. Yamamoto, M. Matsumoto, Y. F. Li, S. R. Cao, R. D. Ji, and C. K. Liang, "Indoor and Outdoor Air Pollution in Tokyo and Beijing Supercities," *Atmos. Environ., 30,* 695–702 (1996).

Avol, E. L., W. C. Navidi, and S. D. Colome, "Modeling Ozone Levels in and around Southern California Homes," *Environ. Sci. Technol., 32,* 463–468 (1998).

Baek, S.-O., Y.-S. Kim, and R. Perry, "Indoor Air Quality in Homes, Offices, and Restaurants in Korean Urban Areas—Indoor/Outdoor Relationships," *Atmos. Environ., 31,* 529–544 (1997).

Bayer, C. M., "The Effect of 'Building Bake-Out' Conditions on Volatile Organic Compound Emissions," in *Indoor Air Pollution—Radon, Bioaerosols, & VOC's* (J. G. Kay, G. E. Keller, and J. F. Miller, Eds.), Chap. 9, Lewis Publishers, Chelsea, MI, 1991.

Boyle, M., "Radon Testing of Soils," *Environ. Sci. Technol., 22,* 1397–1399 (1988).

Brauer, M., P. B. Ryan, H. H. Suh, P. Koutrakis, J. D. Spengler, N. P. Leslie, and I. H. Billick, "Measurements of Nitrous Acid inside Two Research Houses," *Environ. Sci. Technol., 24,* 1521–1527 (1990).

Brauer, M., T. R. Rasmussen, S. K. Kjaergaard, and J. D. Spengler, "Nitrous Acid Formation in an Experimental Exposure Chamber," *Indoor Air, 3,* 94–105 (1993).

Brauer, M., and J. D. Spengler, "Nitrogen Dioxide Exposures inside Ice Skating Rinks," *Am. J. Public Health, 84,* 429–433 (1994).

Brauer, M., K. Lee, J. D. Spengler, R. O. Salonen, A. Pennanen, O. A. Braathen, E. Mihalikova, P. Miskovic, A. Nozaki, T. Tsuzuki, S. Rui-Jin, Y. Xu, Z. Quing-Xiang, H. Drahonovska, and S. Kjaergaard, "Nitrogen Dioxide in Indoor Ice Skating Facilities:

An International Survey," *J. Air Waste Manage. Assoc., 47,* 1095–1102 (1997).

Brickus, L. S. R., J. N. Cardoso, and F. R. De Quino Neto, "Distributions of Indoor and Outdoor Air Pollutants in Rio de Janeiro, Brazil: Implications to Indoor Air Quality in Bayside Offices," *Environ. Sci. Technol., 32,* 3485–3490 (1998).

Brown, S. K., M. R. Sim, M. J. Abramson, and C. N. Gray, "Concentrations of Volatile Organic Compounds in Indoor Air—A Review," *Indoor Air, 4,* 123–134 (1994).

Bukowski, J. A., and L. W. Meyer, "Simulated Air Levels of Volatile Organic Compounds Following Different Methods of Indoor Insecticide Application," *Environ. Sci. Technol., 29,* 673–676 (1995).

Bukowski, J. A., and D. Wartenberg, "An Alternative Approach for Investigating the Carcinogenicity of Indoor Air Pollution: Pets as Sentinels of Environmental Cancer Risk," *Environ. Health Perspect., 105,* 1312–1319 (1997).

California Environmental Protection Agency, Air Resources Board, Research Division, "Determination of Formaldehyde and Toluene Diisocyanate Emissions from Indoor Residential Sources," Contract No. 93-315, Final Report, November 1996.

California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, "Health Effects of Exposure to Environmental Tobacco Smoke," Final Report, September 1997.

Castleman, A. W., Jr., "Consideration of the Chemistry of Radon Progeny," *Environ. Sci. Technol., 25,* 730–735 (1991).

Chan, C.-C., J. D. Spengler, H. Özkaynak, and M. Lefkopoulou, "Commuter Exposures to VOCs in Boston, Massachusetts," *J. Air Waste Manage. Assoc., 41,* 1594–1600 (1991a).

Chan, C.-C., H. Özkaynak, J. D. Spengler, and L. Sheldon, "Driver Exposure to Volatile Organic Compounds, CO, Ozone, and NO_2 under Different Driving Conditions," *Environ. Sci. Technol., 25,* 964–972 (1991b).

Chan, C.-C., S.-H. Lin, and G.-R. Her, "Student's Exposure to Volatile Organic Compounds While Commuting by Motorcycle and Bus in Taipei City," *J. Air Waste Manage. Assoc., 43,* 1231–1238 (1993).

Chang, J. C. S., and K. A. Krebs, "Evaluation of Para-Dichlorobenzene Emissions from Solid Moth Repellant as a Source of Indoor Air Pollution," *J. Air Waste Manage. Assoc., 42,* 1214–1217 (1992).

Chang, J. C. S., and Z. Guo, "Emissions of Odorous Aldehydes from Alkyd Paint," *Atmos. Environ., 32,* 3581–3586 (1998).

Chang, J. C. S., L. E. Sparks, and Z. Guo, "Evaluation of Sink Effects on VOCs from a Latex Paint," *J. Air Waste Manage. Assoc., 48,* 953–958 (1998).

Chuang, J. C., G. A. Mack, M. R. Kuhlman, and N. K. Wilson, "Polycyclic Aromatic Hydrocarbons and Their Derivatives in Indoor and Outdoor Air in an Eight-Home Study," *Atmos. Environ., 25B,* 369–380 (1991).

Clayton, C. A., R. I. Perritt, E. D. Pellizzari, K. W. Thomas, R. W. Whitmore, L. A. Wallace, H. Özkaynak, and J. D. Spengler, "Particle Total Exposure Assessment Methodology (PTEAM) Study: Distributions of Aerosol and Elemental Concentrations in Personal, Indoor, and Outdoor Air Samples in a Southern California Community," *J. Exposure Anal. Environ. Epidemiol., 3,* 227–250 (1993).

Colome, S. D., N. Y. Kado, P. Jaques, and M. Kleinman, "Indoor–Outdoor Air Pollution Relations: Particulate Matter Less Than 10 μm in Aerodynamic Diameter (PM 10) in Homes of Asthmatics," *Atmos. Environ., 26A,* 2173–2178 (1992).

Crump, D. R., and D. Gardiner, "Sources and Concentrations of Aldehydes and Ketones in Indoor Environments in the UK," *Environ. Int., 15,* 455–462 (1989).

Crump, D. R., "Volatile Organic Compounds in Indoor Air," in *Issues in Environmental Science and Technology* (R. E. Hester and R. M. Harrison, Eds.), Chap. 4, pp. 109–124, Royal Chem. Soc., Letchworth, UK, 1995.

Daisey, J. M., A. T. Hodgson, W. J. Fisk, M. J. Mendell, and J. Ten Brinke, "Volatile Organic Compounds in Twelve California Office Buildings: Classes, Concentrations, and Sources," *Atmos. Environ., 28,* 3557–3562 (1994).

Davidson, C. I., S.-F. Lin, J. F. Osborn, M. R. Pandey, R. A. Rasmussen, and M. A. K. Khalil, "Indoor and Outdoor Air Pollution in the Himalayas," *Environ. Sci. Technol., 20,* 561–567 (1986).

Dor, F., Y. Le Moullec, and B. Festy, "Exposure of City Residents to Carbon Monoxide and Monocyclic Aromatic Hydrocarbons during Commuting Trips in the Paris Metropolitan Area," *J. Air Waste Manage. Assoc., 45,* 103–110 (1995).

Drakou, G., C. Zerefos, I. Ziomas, and M. Voyatzaki, "Measurements and Numerical Simulations of Indoor O_3 and NO_x in Two Different Cases," *Atmos. Environ., 32,* 595–610 (1998).

Druzik, J. R., M. S. Adams, C. Tiller, and G. R. Cass, "The Measurement and Model Predictions of Indoor Ozone Concentrations in Museums," *Atmos. Environ., 24A,* 1813–1823 (1990).

Drye, E. E., H. Özkaynak, B. Burbank, I. H. Billick, J. D. Spengler, P. B. Ryan, P. E. Baker, and S. D. Colome, "Development of Models for Predicting the Distribution of Indoor Nitrogen Dioxide Concentrations," *J. Air Waste Manage. Assoc., 39,* 1169–1177 (1989).

Duffy, B. L., and P. F. Nelson, "Exposure to Emissions of 1,3-Butadiene and Benzene in the Cabins of Moving Motor Vehicles and Buses in Sydney, Australia," *Atmos. Environ., 31,* 3877–3885 (1997).

Ekinja, I., Z. Grabaric, and B. S. Grabaric, "Monitoring of Pyrocatechol Indoor Air Pollution," *Atmos. Environ., 29,* 1165–1170 (1995).

Febo, A., and C. Perrino, "Prediction and Experimental Evidence for High Air Concentration of Nitrous Acid in Indoor Environments," *Atmos. Environ., 25A,* 1055–1061 (1991).

Febo, A., and C. Perrino, "Measurement of High Concentration of Nitrous Acid inside Automobiles," *Atmos. Environ., 29,* 345–351 (1995).

Feinman, S. E., *Formaldehyde Sensitivity and Toxicity*, CRC Press, Boca Raton, FL, 1988.

Fernandez-Bremauntz, A. A., and M. R. Ashmore, "Exposure of Commuters to Carbon Monoxide in Mexico City—I. Measurement of In-Vehicle Concentrations," *Atmos. Environ., 29,* 525–532 (1995).

Fischer, M. L., A. J. Bentley, K. A. Dunkin, A. T. Hodgson, W. W. Nazaroff, R. G. Sextro, and J. M. Daisey, "Factors Affecting Indoor Air Concentrations of Volatile Organic Compounds at a Site of Subsurface Gasoline Contamination," *Environ. Sci. Technol., 30,* 2948–2957 (1996).

Flachsbart, P. G., G. A. Mack, J. E. Howes, and C. E. Rodes, "Carbon Monoxide Exposures of Washington Commuters," *JAPCA, 37,* 135–142 (1987).

Florig, H. K., "China's Air Pollution Risks," *Environ. Sci. Technol., 31,* 274A–279A (1997).

Fortmann, R., N. Roache, J. C. S. Chang, and Z. Guo, "Characterization of Emissions of Volatile Organic Compounds from Interior Alkyd Paint," *J. Air Waste Manage. Assoc., 48,* 931–940 (1998).

Foster, W. M., L. Jiang, P. T. Stetkiewicz, and T. H. Risby, "Breath Isoprene—Temporal Changes in Respiratory Output after Exposure to Ozone," *J. Appl. Physiol., 80,* 706–710 (1996).

Gelmont, D., R. A. Stein, and J. F. Mead, "Isoprene—The Main Hydrocarbon in Human Breath," *Biochem. Biophys. Res. Commun., 99,* 1456–1460 (1981).

Georghiou, P. E., P. Blagden, D. A. Snow, L. Winsor, and D. T. Williams, "Mutagenicity of Indoor Air Containing Environmental Tobacco Smoke: Evaluation of a Portable PM-10 Impactor Sampler," *Environ. Sci. Technol., 25,* 1496–1500 (1991).

Gold, D. R., G. Allen, A. Damokosh, P. Serrano, C. Hayes, and M. Castillejos, "Comparison of Outdoor and Classroom Ozone Exposures for School Children in Mexico City," *J. Air Waste Manage. Assoc., 46,* 335–342 (1996).

Haghighat, F., and L. de Bellis, "Material Emission Rates: Literature Review, and the Impact of Indoor Air Temperature and Relative Humidity," *Bldg. Environ., 33,* 261–277 (1998).

Hanrahan, L. P., H. A. Anderson, K. A. Daily, A. D. Eckmann, and M. S. Kanarek, "Formaldehyde Concentrations in Wisconsin Mobile Homes," *JAPCA, 35,* 1164–1167 (1985).

Heavner, D. L., M. W. Ogden, and P. R. Nelson, "Multisorbent Thermal Desorption/Gas Chromatography/Mass Selective Detection Method for the Determination of Target Volatile Organic Compounds in Indoor Air," *Environ. Sci. Technol., 26,* 1737–1746 (1992).

Highsmith, V. R., C. E. Rodes, and R. J. Hardy, "Indoor Particle Concentrations Associated with Use of Tap Water in Portable Humidifiers," *Environ. Sci. Technol., 22,* 1109–1112 (1988).

Highsmith, V. R., R. J. Hardy, D. L. Costa, and M. S. Germani, "Physical and Chemical Characterization of Indoor Aerosols Resulting from the Use of Tap Water in Portable Home Humidifiers," *Environ. Sci. Technol., 26,* 673–680 (1992).

Hileman, B., "Multiple Chemical Sensitivity," *Chem. Eng. News,* 26–42, July 22 (1991).

Hintenlang, D. E., and K. K. Al-Ahmady, "Pressure Differentials for Radon Entry Coupled to Periodic Atmospheric Pressure Variations," *Indoor Air, 2,* 208–215 (1992).

Hisham, M. W. M., and D. Grosjean, "Air Pollution in Southern California Museums: Indoor and Outdoor Levels of Nitrogen Dioxide, Peroxyacetyl Nitrate, Nitric Acid, and Chlorinated Hydrocarbons," *Environ. Sci. Technol., 25,* 857–862 (1991a).

Hisham, M. W. M., and D. Grosjean, "Sulfur Dioxide, Hydrogen Sulfide, Total Reduced Sulfur, Chlorinated Hydrocarbons, and Photochemical Oxidants in Southern California Museums," *Atmos. Environ., 25A,* 1497–1505 (1991b).

Hodgson, A. T., J. M. Daisey, and R. A. Grot, "Sources and Source Strengths of Volatile Organic Compounds in a New Office Building," *J. Air Waste Manage. Assoc., 41,* 1461–1468 (1991).

Hodgson, A. T., K. Garbesi, R. G. Sextro, and J. M. Daisey, "Soil-Gas Contamination and Entry of Volatile Organic Compounds into a House near a Landfill," *J. Air Waste Manage. Assoc., 42,* 277–283 (1992).

Hodgson, A. T., J. D. Wooley, and J. M. Daisey, "Emissions of Volatile Organic Compounds from New Carpets Measured in a Large-Scale Environmental Chamber," *J. Air Waste Manage. Assoc., 43,* 316–324 (1993).

Hoek, G., B. Brunekreef, and P. Hofschreuder, "Indoor Exposure to Airborne Particles and Nitrogen Dioxide during an Air Pollution Episode," *JAPCA, 39,* 1348–1349 (1989).

Howard, C., and R. L. Corsi, "Volatilization of Chemicals from Drinking Water to Indoor Air: The Role of Residential Washing Machines," *J. Air Waste Manage. Assoc., 48,* 907–914 (1998).

Howard, E. M., R. C. McCrillis, K. A. Krebs, R. Fortman, H. C. Lao, and Z. Guo, "Indoor Emissions from Conversion Varnishes," *J. Air Waste Manage. Assoc., 48,* 924–930 (1998).

Huey, L. G., E. J. Dunlea, E. R. Lovejoy, D. R. Hanson, R. B. Norton, F. C. Fehsenfeld, and C. J. Howard, "Fast Time Response Measurements of HNO_3 in Air with a Chemical Ionization Mass Spectrometer," *J. Geophys. Res., 103,* 3355–3360 (1998).

International Commission on Radiological Protection, *Ann. ICRP, 23,* No. 2 (1993).

International Commission on Radiological Protection, "Protection against Radon-222 at Home and at Work," *Ann. ICRP*, No. 65 (1994).

Jakobi, G., and P. Fabian, "Indoor/Outdoor Concentrations of Ozone and Peroxyacetyl Nitrate (PAN)," *Int. J. Biometeorol., 40,* 162–165 (1997).

Jenkins, P. L., T. J. Phillips, E. J. Mulberg, and S. P. Hui, "Activity Patterns of Californians: Use of and Proximity to Indoor Pollutant Sources," *Atmos. Environ., 26A,* 2141–2148 (1992).

Jo, W.-K., and S.-J. Choi, "Vehicle Occupants' Exposure to Aromatic Volatile Organic Compounds While Commuting on an Urban–Suburban Route in Korea," *J. Air Waste Manage. Assoc., 46,* 749–754 (1996).

Jones, A. W., V. Lagesson, and C. Tagesson, "Origins of Breath Isoprene," *J. Clin. Pathol., 48,* 979–980 (1995).

Kay, J. G., G. E. Keller, and J. F. Miller, *Indoor Air Pollution—Radon, Bioaerosols, & VOC's*, Lewis Publishers, Chelsea, MI, 1991.

Kelly, T. J., "Determination of Formaldehyde and Toluene Diisocyanate Emissions from Indoor Residential Sources," California Air Resources Board, Research Division, 2020 L Street, Sacramento, CA 95814, Final Report, Contract Number 93-315, November, 1996.

Kostiainen, R., "Volatile Organic Compounds in the Indoor Air of Normal and Sick Houses," *Atmos. Environ., 29,* 693–702 (1995).

Koushki, P. A., K. H. Al-Dhowalia, and S. A. Niaizi, "Vehicle Occupant Exposure to Carbon Monoxide," *J. Air Waste Manage. Assoc., 42,* 1603–1608 (1992).

Koutrakis, P., M. Brauer, S. L. K. Briggs, and B. P. Leaderer, "Indoor Exposures to Fine Aerosols and Acid Gases," *Environ. Health Perspect., 95,* 23–28 (1991).

Koutrakis, P., S. L. K. Briggs, and B. P. Leaderer, "Source Apportionment of Indoor Aerosols in Suffolk and Onondaga Counties, New York," *Environ. Sci. Technol., 26,* 521–527 (1992).

Lawryk, N. J., and C. P. Weisel, "Concentrations of Volatile Organic Compounds in the Passenger Compartments of Automobiles," *Environ. Sci. Technol., 30,* 810–816 (1996).

Leaderer, B. P., P. M. Boone, and S. K. Hammond, "Total Particle, Sulfate, and Acidic Aerosol Emissions from Kerosene Space Heaters," *Environ. Sci. Technol., 24,* 908–912 (1990).

Leese, K., "Indoor Air Measurements," *Environ. Sci. Technol. / News, 31,* 493A (1997).

Leese, K. E., E. C. Cole, R. M. Hall, and M. A. Berry, "Measurement of Airborne and Floor Dusts in a Nonproblem Building," *AIHA J., 58,* 432–438 (1997).

Leovic, K. W., L. S. Sheldon, D. A. Whitaker, R. G. Hetes, J. A. Calcagni, and J. N. Baskir, "Measurement of Indoor Air Emissions from Dry-Process Photocopy Machines," *J. Air Waste Manage. Assoc., 46,* 821–829 (1996).

Levy, J. I., K. Lee, J. D. Spengler, and Y. Yanagisawa, "Impact of Residential Nitrogen Dioxide Exposure on Personal Exposure: An International Study," *J. Air Waste Manage. Assoc., 48,* 553–560 (1998).

Lewis, C. W., and R. B. Zweidinger, "Apportionment of Residential Indoor Aerosol VOC and Aldehyde Species to Indoor and Outdoor Sources, and Their Source Strengths," *Atmos. Environ., 26A,* 2179–2184 (1992).

Lewis, R. G., A. E. Bond, D. E. Johnson, and J. P. Hsu, "Measurement of Atmospheric Concentrations of Common Household Pesticides: A Pilot Study," *Environ. Monitor. Assess., 10,* 59–73 (1988).

Lewtas, J., S. Goto, K. Williams, J. C. Chuang, B. A. Petersen, and N. K. Wilson, "The Mutagenicity of Indoor Air Particles in a Residential Pilot Field Study: Application and Evaluation of New Methodologies," *Atmos. Environ., 21,* 443–449 (1987).

Ligocki, M. P., H. I. H. Liu, G. R. Cass, and W. John, "Measurements of Particle Deposition Rates inside Southern California Museums," *Aerosol Sci. Technol., 13,* 85–101 (1990).

Ligocki, M. P., L. G. Salmon, T. Fall, M. C. Jones, W. W. Nazaroff, and G. R. Cass, "Characteristics of Airborne Particles inside Southern California Museums," *Atmos. Environ., 27A,* 697–711 (1993).

Liu, K.-S., S. B. Hayward, J. R. Girman, B. A. Moed, and F.-Y. Huang, "Annual Average Radon Concentrations in California Residences," *J. Air Waste Manage. Assoc., 41,* 1207–1212 (1991).

Liu, L.-J. S., P. Koutrakis, H. H. Suh, J. D. Mulik, and R. M. Burton, "Use of Personal Measurements for Ozone Exposure Assessment: A Pilot Study," *Environ. Health Perspect., 101,* 318–324 (1993).

Löfroth, G., P. I. Ling, and E. Agurell, "Public Exposure to Environmental Tobacco Smoke," *Mutat. Res., 202,* 103–110 (1988).

Löfroth, G., C. Stensman, and M. Brandhorst-Satzkorn, "Indoor Sources of Mutagenic Aerosol Particulate Matter: Smoking, Cooking, and Incense Burning," *Mutat. Res., 261,* 21–28 (1991).

Marbury, M. C., D. P. Harlos, J. M. Samet, and J. D. Spengler, "Indoor Residential NO_2 Concentrations in Albuquerque, New Mexico," *JAPCA, 38,* 392–398 (1988).

McKone, T. E., "Human Exposure to Volatile Organic Compounds in Household Tap Water: The Indoor Inhalation Pathway," *Environ. Sci. Technol., 21,* 1194–1201 (1987).

Mendis, S., P. A. Sobotka, and D. E. Evler, "Pentane and Isoprene in Expired Air from Humans—Gas Chromatographic Analysis of a Single Breath," *Clin. Chem., 40,* 1485–1488 (1994).

Meyer, B., and K. Hermanns, "Formaldehyde Release from Wood Products. An Overview," in *Formaldehyde Release from Wood Products,* ACS Symposium Series 316, Chapter 1, pp. 1–16, Am. Chem. Soc., Washington, D.C., 1986.

Meyer, B., B. A. Kottes Andrews, and R. M. Reinhardt, Eds., "Formaldehyde Release from Wood Products," in *ACS Symposium Series 316,* Am. Chem. Soc., Washington, DC, 1986.

Mitra, S., and B. Ray, "Patterns and Sources of Polycyclic Aromatic Hydrocarbons and Their Derivatives in Indoor Air," *Atmos. Environ., 29,* 3345–3356 (1995).

Morrison, G. C., W. W. Nazaroff, J. Alejandro Cano-Ruiz, A. T. Hodgson, and M. P. Modera, "Indoor Air Quality Impacts of Ventilation Ducts: Ozone Removal and Emissions of Volatile Organic Compounds," *J. Air Waste Manage. Assoc., 48,* 941–952 (1998).

Moschandreas, D. J., and D. S. O'Dea, "Measurement of Perchloroethylene Indoor Air Levels Caused by Fugitive Emissions from Unvented Dry-to-Dry Dry Cleaning Units," *J. Air Waste Manage. Assoc., 45,* 111–115 (1995).

Mose, D. G., and G. W. Mushrush, "Variable Spacial and Seasonal Hazards of Airborne Radon," *Atmos. Environ., 31,* 3523–3530 (1997).

Moseley, C. L., and M. R. Meyer, "Petroleum Contamination of an Elementary School: A Case History Involving Air, Soil-Gas, and Groundwater Monitoring," *Environ. Sci. Technol., 26,* 185–192 (1992).

Mumford, J. L., X. Z. He, R. S. Chapman, S. R. Cao, D. B. Harris, X. M. Li, Y. L. Xian, W. Z. Jiang, C. W. Xu, J. C. Chuang, W. E. Wilson, and M. Cooke, "Lung Cancer and Indoor Air Pollution in Xuan Wei, China," *Science, 235,* 217–220 (1987).

Mumford, J. L., R. W. Williams, D. B. Walsh, R. M. Burton, D. J. Svendsgaard, J. C. Chuang, V. S. Houk, and J. Lewtas, "Indoor Air Pollutants from Unvented Kerosene Heater Emissions in

Mobile Homes: Studies on Particles, Semivolatile Organics, Carbon Monoxide, and Mutagenicity," *Environ. Sci. Technol., 25,* 1732–1738 (1991).

Naik, D. V., C. J. Weschler, and H. C. Shields, "Indoor and Outdoor Concentrations of Organic Compounds Associated with Airborne Particles: Results Using a Novel Solvent System," in *Indoor Air Pollution—Radon, Bioaerosols, & VOC's* (J. G. Kay, G. E. Keller, and J. F. Miller, Eds.), Chap. 6, pp. 59–70, Lewis Publishers, Chelsea, MI, 1991.

National Research Council, "Health Risks of Radon and Other Internally Deposited Alpha-Emitters," Committee on the Biological Effects of Ionizing Radiations, Commission on Life Sciences, National Academy Press, Washington, D.C., 1988.

National Research Council, *Comparative Dosimetry of Radon in Mines and Homes*, Panel on Dosimetric Assumption Affecting the Applications of Radon Risk Estimates, Board on Radiation Effects Research, Commission on Life Sciences, National Academy Press, Washington, D.C., 1991.

Nazaroff, W. W., and G. R. Cass, "Mathematical Modeling of Chemically Reactive Pollutants in Indoor Air," *Environ. Sci. Technol., 20,* 924–934 (1986).

Nazaroff, W. W., and A. V. Nero, Jr., *Radon and Its Decay Products in Indoor Air*, Wiley-Interscience, New York, 1988.

Nazaroff, W. W., and G. R. Cass, "Mathematical Modeling of Indoor Aerosol Dynamics," *Environ. Sci. Technol., 23,* 157–166 (1989a).

Nazaroff, W. W., and G. R. Cass, "Mass-Transport Aspects of Pollutant Removal at Indoor Surfaces," *Environ. Int., 15,* 567–584 (1989b).

Nazaroff, W. W., L. G. Salmon, and G. R. Cass, "Concentration and Fate of Airborne Particles in Museums," *Environ. Sci. Technol., 24,* 66–77 (1990a).

Nazaroff, W. W., M. P. Ligocki, T. Ma, and G. R. Cass, "Particle Deposition in Museums: Comparison of Modeling and Measurement Results," *Aerosol Sci. Technol., 13,* 332–348 (1990b).

Nazaroff, W. W., and K. Teichman, "Indoor Radon," *Environ. Sci. Technol., 24,* 774–782 (1990).

Neas, L. M., D. W. Dockery, J. H. Ware, J. D. Spengler, B. G. Ferris, Jr., and F. E. Speizer, "Concentration of Indoor Particulate Matter as a Determinant of Respiratory Health in Children," *Am. J. Epidemiol., 139,* 1088–1099 (1994).

Nelson, P. R., S. P. Kelly, and F. W. Conrad, "Studies of Environmental Tobacco Smoke Generated by Different Cigarettes," *J. Air Waste Manage. Assoc., 47,* 336–344 (1998).

Nero, A. V., M. B. Schwehr, W. W. Nazaroff, and K. L. Revzan, "Distribution of Airborne Radon-222 Concentrations in U.S. Homes," *Science, 234,* 992–997 (1986).

Nero, A., "Earth, Air, Radon, and Home," *Phys. Today, 32–39,* April (1989).

Nero, A. V., Jr., "Controlling Indoor Air Pollution," *Sci. Am., 258,* 42–48 (1988).

Offermann, F. J., S. A. Loiselle, A. T. Hodgson, L. A. Gundel, and J. M. Daisey, "A Pilot Study to Measure Indoor Concentrations and Emission Rates of Polycyclic Aromatic Hydrocarbons," *Indoor Air, 4,* 497–512 (1991).

Osborne, M. C., "Four Common Diagnostic Problems That Inhibit Radon Mitigation," *JAPCA, 37,* 604–606 (1987).

Ott, W., P. Switzer, and N. Willits, "Carbon Monoxide Exposures inside an Automobile Traveling on an Urban Arterial Highway," *J. Air Waste Manage. Assoc., 44,* 1010–1018 (1994).

Ott, W., P. Switzer, and J. Robinson, "Particle Concentrations inside a Tavern before and after Prohibition of Smoking: Evaluating the Performance of an Indoor Air Quality Model," *J. Air Waste Manage. Assoc., 46,* 1120–1134 (1996).

Özkaynak, H., P. B. Ryan, G. A. Allen, and W. A. Turner, "Indoor Air Quality Modeling: Compartmental Approach with Reactive Chemistry," *Environ. Int., 8,* 461–471 (1982).

Özkaynak, H., J. Xue, J. D. Spengler, L. A. Wallace, E. D. Pellizzari, and P. Jenkins, "Personal Exposure to Airborne Particles and Metals: Results from the Particle TEAM Study in Riverside, CA," *J. Exposure Anal. Environ. Epidemiol., 6,* 57–78 (1996).

Pellizzari, E. D., K. W. Thomas, C. A. Clayton, R. W. Whitmore, R. C. Shores, H. S. Zelon, and R. L. Perritt, "Particle Total Exposure Assessment Methodolgy (PTEAM): Riverside, California Pilot Study," Vol. 1, NTIS #PB 93-166957/AS, National Technical Information Service, Springfield, VA, 1993.

Pennanen, A. S., R. O. Salonen, S. Alm, M. J. Jantunen, and P. Pasanen, "Characterization of Air Quality Problems in Five Finnish Indoor Ice Arenas," *J. Air Waste Manage. Assoc., 47,* 1079–1086 (1997).

Perry, R., and I. L. Gee, "Vehicle Emissions and Effects on Air Quality: Indoors and Outdoors," *Indoor Environ., 3,* 224–236 (1994).

Peto, J., and S. Darby, "Radon Risk Reassessed," *Nature, 368,* 97–98 (1994).

Petreas, M., K.-S. Liu, B.-H. Chang, S. B. Hayward, and K. Sexton, "A Survey of Nitrogen Dioxide Levels Measured inside Mobile Homes," *JAPCA, 38,* 647–651 (1988).

Phillips, M., J. Greenberg, and J. Awad, "Metabolic and Environmental Origins of Volatile Organic Compounds in Breath," *J. Clin. Pathol., 47,* 1052–1053 (1994).

Pitts, J. N., Jr., T. J. Wallington, H. W. Biermann, and A. M. Winer, "Identification and Measurement of Nitrous Acid in an Indoor Environment," *Atmos. Environ., 19,* 763–767 (1985).

Pitts, J. N., Jr., H. W. Biermann, E. C. Tuazon, M. Green, W. D. Long, and A. M. Winer, "Time-Resolved Identification and Measurement of Indoor Air Pollutants by Spectroscopic Techniques: Gaseous Nitrous Acid, Methanol, Formaldehyde, and Formic Acid," *JAPCA, 39,* 1344–1347 (1989).

Quackenboss, J. J., J. D. Spengler, M. S. Kanarek, R. Letz, and C. P. Duffy, "Personal Exposure to Nitrogen Dioxide: Relationship to Indoor/Outdoor Air Quality and Activity Patterns," *Environ. Sci. Technol., 20,* 775–783 (1986).

Quackenboss, J. J., M. D. Lebowitz, and C. D. Crutchfield, "Indoor–Outdoor Relationships for Particulate Matter: Exposure Classifications and Health Effects," *Environ. Int., 15,* 353–360 (1989).

Reiss, R., P. B. Ryan, and P. Koutrakis, "Modeling Ozone Deposition onto Indoor Residential Surfaces," *Environ. Sci. Technol., 28,* 504–513 (1994).

Reiss, R., P. B. Ryan, S. J. Tibbetts, and P. Koutrakis, "Measurement of Organic Acids, Aldehydes, and Ketones in Residential Environments and Their Relation to Ozone," *J. Air Waste Manage. Assoc., 45,* 811–822 (1995a).

Reiss, R., P. B. Ryan, P. Koutrakis, and S. J. Tibbets, "Ozone Reactive Chemistry on Interior Latex Paint," *Environ. Sci. Technol., 29,* 1906–1912 (1995b).

Rodes, C. E., R. M. Kamens, and R. W. Wiener, "The Significance and Characteristics of the Personal Activity Cloud on Exposure Assessment Measurements for Indoor Contaminants," *Indoor Air, 2,* 123–145 (1991).

Roinestad, K. S., J. B. Louis, and J. D. Rosen, "Determination of Pesticides in Indoor Air and Dust," *J. AOAC Int., 76,* 1121–1126 (1993).

Romieu, I., M. C. Lugo, S. Colome, A. M. Garcia, M. H. Avila, A. Geyh, S. R. Velasco, and E. P. Rendon, "Evaluation of Indoor Ozone Concentration and Predictors of Indoor–Outdoor Ratio in Mexico City," *J. Air Waste Manage. Assoc., 48,* 327–335 (1998).

Rowe, D. R., K. H. Al-Dhowalia, and M. E. Mansour, "Indoor–Outdoor Carbon Monoxide Concentrations at Four Sites in Riyadh, Saudi Arabia," *J. Air Waste Manage. Assoc., 39,* 1100–1102 (1989).

Rowe, D. R., K. H. Al-Dhowalia, and M. E. Mansour, "Indoor–Outdoor Nitric Oxide and Nitrogen Dioxide Concentrations at Three Sites in Riyadh, Saudi Arabia," *J. Air Waste Manage. Assoc., 41,* 973–976 (1991).

Ryan, P. B., M. L. Soczek, J. D. Spengler, and I. H. Billick, "The Boston Residential NO$_2$ Characterization Study: I. Preliminary Evaluation of the Survey Methodology," *JAPCA, 38,* 22–27 (1988).

Salmon, L. G., W. W. Nazaroff, M. P. Ligocki, M. C. Jones, and G. R. Cass, "Nitric Acid Concentrations in Southern California Museums," *Environ. Sci. Technol., 24,* 1004–1012 (1990).

Salmon, L. G., C. S. Christoforou, and G. R. Cass, "Airborne Pollutants in the Buddhist Cave Temples at the Yungang Grottoes, China," *Environ. Sci. Technol., 28,* 805–811 (1994).

Salmon, L. G., C. S. Christoforou, T. J. Gerk, G. R. Cass, G. S. Casuccio, G. A. Cooke, M. Leger, and I. Olmez, "Source Contributions to Airborne Particle Deposition at the Yungang Grottoes, China," *Sci. Total Environ., 167,* 33–47 (1995).

Salthammer, T., A. Schwarz, and F. Fuhrmann, "Emission of Reactive Compounds and Secondary Products from Wood-Based Furniture Coating," *Atmos. Environ., 33,* 75–84 (1999).

Schaeffer, V. H., B. Bhooshan, S.-B. Chen, J. S. Sonenthal, and A. T. Hodgson, "Characterization of Volatile Organic Chemical Emissions from Carpet Cushions," *J. Air Waste Manage. Assoc., 46,* 813–820 (1996).

Sexton, K., M. X. Petreas, and K.-S. Liu, "Formaldehyde Exposures inside Mobile Homes," *Environ. Sci. Technol., 23,* 985–988 (1989).

Shah, J. J., and H. B. Singh, "Distribution of Volatile Organic Chemicals in Outdoor and Indoor Air," *Environ. Sci. Technol., 22,* 1381–1388 (1988).

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, "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 (1989).

Shepherd, J. L., R. L. Corsi, and J. Kemp, "Chloroform in Indoor Air and Wastewater: The Role of Residential Washing Machines," *J. Air Waste Manage. Assoc., 46,* 631–642 (1996).

Shields, H. C., and C. J. Weschler, "Volatile Organic Compounds Measured at a Telephone Switching Center from 5/30/85–12/6/88: A Detailed Case Study," *J. Air Waste Manage. Assoc., 42,* 792–804 (1992).

Shields, H. C., D. M. Fleischer, and C. J. Weschler, "Comparisons among VOCs Measured in Three Types of U.S. Commercial Buildings with Different Occupant Densities," *Indoor Air, 6,* 2–17 (1996).

Sinclair, J. D., L. A. Psota-Kelty, and C. J. Weschler, "Indoor/Outdoor Ratios and Indoor Surface Accumulations of Ionic Substances at Newark, New Jersey," *Atmos. Environ., 22,* 461–469 (1988).

Sinclair, J. D., L. A. Psota-Kelty, C. J. Weschler, and H. C. Shields, "Deposition of Airborne Sulfate, Nitrate, and Chloride Salts as It Relates to Corrosion of Electronics," *J. Electrochem. Soc., 137,* 1200–1206 (1990a).

Sinclair, J. D., L. A. Psota-Kelty, C. J. Weschler, and H. C. Shields, "Measurement and Modeling of Airborne Concentrations and Indoor Surface Accumulation Rates of Ionic Substances at Neenah, Wisconsin," *Atmos. Environ., 24A,* 627–638 (1990b).

Sollinger, S., K. Levsen, and G. Wünsch, "Indoor Air Pollution by Organic Emissions from Textile Floor Coverings. Climate Chamber Studies under Dynamic Conditions," *Atmos. Environ., 27B,* 183–192 (1993).

Sollinger, S., K. Levsen, and G. Wünsch, "Indoor Pollution by Organic Emissions from Textile Floor Coverings: Climate Test Chamber Studies under Static Conditions," *Atmos. Environ., 28,* 2369–2378 (1994).

Spengler, J. D., D. W. Dockery, W. A. Turner, J. M. Wolfson, and B. G. Ferris, Jr., "Long-Term Measurements of Respirable Sulfates and Particles inside and outside Homes," *Atmos. Environ., 15,* 23–30 (1981).

Spengler, J. D., M. Brauer, J. M. Samet, and W. E. Lambert, "Nitrous Acid in Albuquerque, New Mexico, Homes," *Environ. Sci. Technol., 27,* 841–845 (1993).

Spengler, J., M. Schwab, P. B. Ryan, S. Colome, A. L. Wilson, I. Billick, and E. Becker, "Personal Exposure to Nitrogen Dioxide in the Los Angeles Basin," *J. Air Waste Manage. Assoc., 44,* 39–47 (1994).

Spicer, C. W., R. W. Coutant, G. F. Ward, D. W. Joseph, A. J. Gaynor, and I. H. Billick, "Rates and Mechanisms of NO$_2$ Removal from Indoor Air Residential Materials," *Environ. Int., 15,* 643–654 (1989).

Spicer, C. W., D. V. Kenny, G. F. Ward, and I. H. Billick, "Transformations, Lifetimes, and Sources of NO$_2$, HONO, and HNO$_3$ in Indoor Environments," *J. Air Waste Manage. Assoc., 43,* 1479–1485 (1993).

Spicer, C. W., D. V. Kenny, G. F. Ward, I. H. Billick, and N. P. Leslie, "Evaluation of NO$_2$ Measurement Methods for Indoor Air Quality Applications," *J. Air Waste Manage. Assoc., 44,* 163–168 (1994).

Stock, T. H., "Formaldehyde Concentrations inside Conventional Housing," *JAPCA, 37,* 913–918 (1987).

Su, W.-H., "Indoor Air Pollution," *Resour., Conserv. Recycl., 16,* 77–91 (1996).

Suh, H. H., J. D. Spengler, and P. Koutrakis, "Personal Exposures to Acid Aerosols and Ammonia," *Environ. Sci. Technol., 26,* 2507–2517 (1992).

Tancrède, M., R. Wilson, L. Zeise, and E. A. C. Crouch, "The Carcinogenic Risk of Some Organic Vapors Indoors: A Theoretical Survey," *Atmos. Environ., 21,* 2187–2205 (1987).

Thatcher, T. L., and D. W. Layton, "Deposition, Resuspension, and Penetration of Particles within a Residence," *Atmos. Environ., 29,* 1487–1497 (1995).

Thompson, C. V., R. A. Jenkins, and C. E. Higgins, "A Thermal Desorption Method for the Determination of Nicotine in Indoor Environments," *Environ. Sci. Technol., 23,* 429–435 (1989).

Tichenor, B. A., and M. A. Mason, "Organic Emissions from Consumer Products and Building Materials to the Indoor Environment," *JAPCA, 38,* 264–268 (1988).

Tichenor, B. A., L. A. Sparks, J. B. White, and M. D. Jackson, "Evaluating Sources of Indoor Air Pollution," *J. Air Waste Manage. Assoc., 40,* 487–492 (1990).

Traynor, G. W., M. G. Apte, H. A. Sokil, J. C. Chuang, W. G. Tucker, and J. L. Mumford, "Selected Organic Pollutant Emissions from Unvented Kerosene Space Heaters," *Environ. Sci. Technol., 24,* 1265–1270 (1990).

Turk, B. H., D. T. Grimsrud, J. T. Brown, K. L. Geisling-Sobotka, J. Harrison, and R. J. Prill, "Commercial Building Ventilation Rates and Particle Concentrations," *ASHRAE Trans., 95,* 422–433 (1989).

Turk, B. H., R. J. Prill, D. T. Grimsrud, B. A. Moed, and R. G. Sextro, "Characterizing the Occurrence, Sources, and Variability of Radon in Pacific Northwest Homes," *J. Air Waste Manage. Assoc., 40,* 498–506 (1990).

Van Loy, M. D., V. C. Lee, L. A. Gundel, J. M. Daisey, R. G. Sextro, and W. W. Nazaroff, "Dynamic Behavior of Semivolatile Organic Compounds in Indoor Air. 1. Nicotine in a Stainless Steel Chamber," *Environ. Sci. Technol., 31,* 2554–2561 (1997).

Van Loy, M. D., W. W. Nazaroff, and J. M. Daisey, "Nicotine as a Marker for Environmental Tobacco Smoke: Implications of Sorption on Indoor Surface Materials," *J. Air Waste Manage. Assoc., 48,* 959–968 (1998).

Vecera, Z., and P. K. Dasgupta, "Indoor Nitrous Acid Levels. Production of Nitrous Acid from Open-Flame Sources," *Int. J. Environ. Anal. Chem., 56,* 311–316 (1994).

Wade, W. A., III, W. A. Cote, and J. E. Yocom, "A Study of Indoor Air Quality," *JAPCA, 25,* 933–939 (1975).

Wallace, J. C., L. P. Brzuzy, S. L. Simonich, S. M. Visscher, and R. A. Hites, "Case Study of Organochlorine Pesticides in the Indoor Air of a Home," *Environ. Sci. Technol., 30,* 2715–2718 (1996).

Wallace, L., "Indoor Particles: A Review," *J. Air Waste Manage. Assoc., 46,* 98–126 (1996).

Weschler, C. J., H. C. Shields, and D. V. Naik, "Indoor Ozone Exposures," *JAPCA, 39,* 1562–1568 (1989).

Weschler, C. J., H. C. Shields, and D. Rainer, "Concentrations of Volatile Organic Compounds at a Building with Health and Comfort Complaints," *Am. Ind. Hyg. Assoc. J., 51,* 261–268 (1990).

Weschler, C. J., M. Brauer, and P. Koutrakis, "Indoor Ozone and Nitrogen Dioxide: A Potential Pathway to the Generation of Nitrate Radicals, Dinitrogen Pentaoxide, and Nitric Acid Indoors," *Environ. Sci. Technol., 26,* 179–184 (1992a).

Weschler, C. J., A. T. Hodgson, and J. D. Wooley, "Indoor Chemistry: Ozone, Volatile Organic Compounds, and Carpets," *Environ. Sci. Technol., 26,* 2371–2377 (1992b).

Weschler, C. J., H. C. Shields, and D. V. Naik, "Indoor Chemistry Involving O_3, NO, and NO_2 as Evidence by 14 Months of Measurements at a Site in Southern California," *Environ. Sci. Technol., 28,* 2120–2132 (1994).

Weschler, C. J., and H. C. Shields, "Chemical Transformations of Indoor Air Pollutants," in *Indoor Air '96* (S. Yoshizawa, K. Kimura, K. Ikeda, S. Tanabe, and T. Iwata, Eds.), Organizing Committee of the 7th International Conference of Indoor Air Quality and Climate, Vol. 1, pp. 919–924, Nagoya, Japan, 1996a.

Weschler, C. J., and H. C. Shields, "Production of the Hydroxyl Radical in Indoor Air," *Environ. Sci. Technol., 30,* 3250–3258 (1996b).

Weschler, C. J., H. C. Shields, and B. M. Shah, "Understanding and Reducing the Indoor Concentration of Submicron Particles at a Commercial Building in Southern California," *J. Air Waste Manage. Assoc., 46,* 291–299 (1996).

Weschler, C. J., and H. C. Shields, "Measurements of the Hydroxyl Radical in a Manipulated but Realistic Indoor Environment," *Environ. Sci. Technol., 31,* 3719–3722 (1997a).

Weschler, C. J., and H. C. Shields, "Potential Reactions among Indoor Pollutants," *Atmos. Environ., 31,* 3487–3495 (1997b).

Weschler, C. J., and H. C. Shields, "Indoor Ozone/Terpene Reactions as a Source of Indoor Particles," *Atmos. Environ., 33,* 2307–2318 (1999).

Whitmore, R. W., F. W. Immerman, D. E. Camann, A. E. Bond, R. G. Lewis, and J. L. Schaum, "Non-Occupational Exposures to Pesticides for Residents of Two U.S. Cities," *Arch. Environ. Contam. Toxicol., 26,* 47–59 (1994).

Wolkoff, P., P. A. Clausen, B. Jensen, G. D. Nielsen, and C. I. Wilkins, "Are We Measuring the Relevant Indoor Pollutants?" *Indoor Air, 7,* 92–106 (1997).

Wolkoff, P., "Impact of Air Velocity, Temperature, Humidity, and Air on Long-Term VOC Emissions from Building Products," *Atmos. Environ., 32,* 2659–2668 (1998).

Wolkoff, P., "Photocopiers and Indoor Air Pollution," *Atmos. Environ., 33,* 2129–2130 (1999).

Zhang, J., and P. J. Lioy, "Ozone in Residential Air: Concentrations, I/O Ratios, Indoor Chemistry, and Exposures," *Indoor Air, 4,* 95–105 (1994).

Zhang, J., Q. He, and P. J. Lioy, "Characteristics of Aldehydes: Concentrations, Sources, and Exposures for Indoor and Outdoor Residential Microenvironments," *Environ. Sci. Technol., 28,* 146–152 (1994a).

Zhang, J., W. E. Wilson, and P. J. Lioy, "Indoor Air Chemistry: Formation of Organic Acids and Aldehydes," *Environ. Sci. Technol., 28,* 1975–1982 (1994b).

Zhang, J., W. E. Wilson, and P. J. Lioy, "Sources of Organic Acids in Indoor Air: A Field Study," *J. Exposure Anal. Environ. Epidemiol., 4,* 25–47 (1994c).