INDOOR AIR QUALITY CONTROL TECHNIQUES

Radon, Formaldehyde, Combustion Products

by

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1. Introduction

A. General Context of Indoor Air Quality Problems

The environmental concern for air pollution has been largely focussed on questions of outdoor air contamination. Recently, however, attention has begun to shift to concerns about the quality of air within buildings. In the United States today, people spend only 10-15% of their time outdoors; the rest is spent at home, at work, or traveling in between. Yet existing air quality regulations are based solely on outdoor conditions, specifically on large-scale, highly visible, outdoor air-pollution sources, such as industrial effluents and vehicle exhaust. Buildings were assumed to shelter occupants from outdoor pollutants, and little thought was given to pollutants generated or trapped indoors. Recent studies have shown that concentrations of certain pollutants inside houses exceed standards set for outdoor concentrations. Because health effects are often correlated with exposure over time, it is now clear that air quality indoors requires far more attention than it has yet received. As an emerging health problem, contamination of indoor air has been linked with a wide variety of building materials and consumer products, as well as strategies that reduce the amount of infiltrating air as a means of promoting energy conservation.

Two main factors govern the concentration of any given pollutant indoors: the source strength of the pollutant and its removal rate. Residential buildings have various sources of indoor air pollution. People (and their pets) generate carbon dioxide, moisture, odors, and microbes simply through normal biological processes. Other more important potential sources of indoor air pollution are combustion appliances (gas stoves, forced-air furnaces, unvented space heaters, wood stoves) building materials (used in construction, furnishings, and insulation) tobacco, and the soil under and around the structure.

Air flow into and out of a building is usually the principal means of removing these indoor contaminants. Reducing air flow rates, however, is one of the most cost-effective strategies for improving the energy efficiency of buildings. Although effective in conserving energy, lowering the air exchange rate in a structure generally slows the transport of indoor-generated air pollutants to the outside, trapping the pollutants inside the structure. Many indoor air quality problems surfaced when air exchange rates were drastically reduced (in the interests of energy efficiency) and the pollution effects became obvious to building inhabitants.

B. *Major Indoor Pollutants*

The pollutants of most concern in residential buildings are radon, formaldehyde, and certain combustion products (nitrogen dioxide, carbon monoxide, carbon dioxide, and respirable particles). Even though their health effects in many cases are only now being systematically documented, these pollutants have been singled out by the World Health Organization, National Academy of Sciences, and other research communities because they are suspected of being common in homes and because high indoor concentrations have been measured. The reader should not infer any priority to the order in which the pollutants are described here, for to attach priorities would require an understanding of relative health effects (and synergistic effects) that does not yet exist.

1. Radon and Radon Progeny

Radon and its immediate radioactive decay products are ubiquitous contaminants of indoor air. The two radon isotopes of interest here are ^{222}Rn and ^{220}Rn which have half lives of 3.8 days and 55 seconds, respectively. These isotopes arise as part of the ^{238}U and ^{232}Th decay series, respectively, which are naturally occurring elements found in the earth's crust. Decay products of ^{222}Rn are more important than those of ^{220}Rn since diffusion times from the soil limit the concentration of ^{220}Rn that can accumulate in most situations. The average dose from ^{220}Rn progeny has been estimated to be about 25 percent of that from ^{222}Rn progeny (UNSCEAR, 1982). In the remainder of this paper, discussions of radon and radon progeny will refer to ^{222}Rn and its decay products. The radon decay chain is illustrated in Fig. 1.1. The major health risks associated with radon, and discussed in the next section, result from the alpha decays of the two short-lived progeny, ^{218}Po and ^{214}Po. These progeny are physically and chemically active and can attach to airborne particles. Progeny that are attached or unattached to particulates may be inhaled and retained in the respiratory system.

Health Effects: Much of our knowledge about the human health effects of radon and its progeny is based on experience with underground uranium miners. These miners are generally exposed to radiation at concentrations much higher (100 times) than would ordinarily be experienced by occupants of residential buildings. The main health hazard from radon and its progeny is an increased risk of lung cancer because of the alpha radiation dose to the lung tissue. Because the progeny have short half-lives (30 minutes or less), their inhalation and deposition in the lungs can expose the surrounding tissue to alpha radiation before the body's lung-clearance mechanisms can remove them. Miners, exposed daily for many hours, do, in fact, develop lung cancer at a much higher rate than the general population. Individual health risk from radon progeny in residential buildings depends on the person's time-integrated exposure to radon progeny.

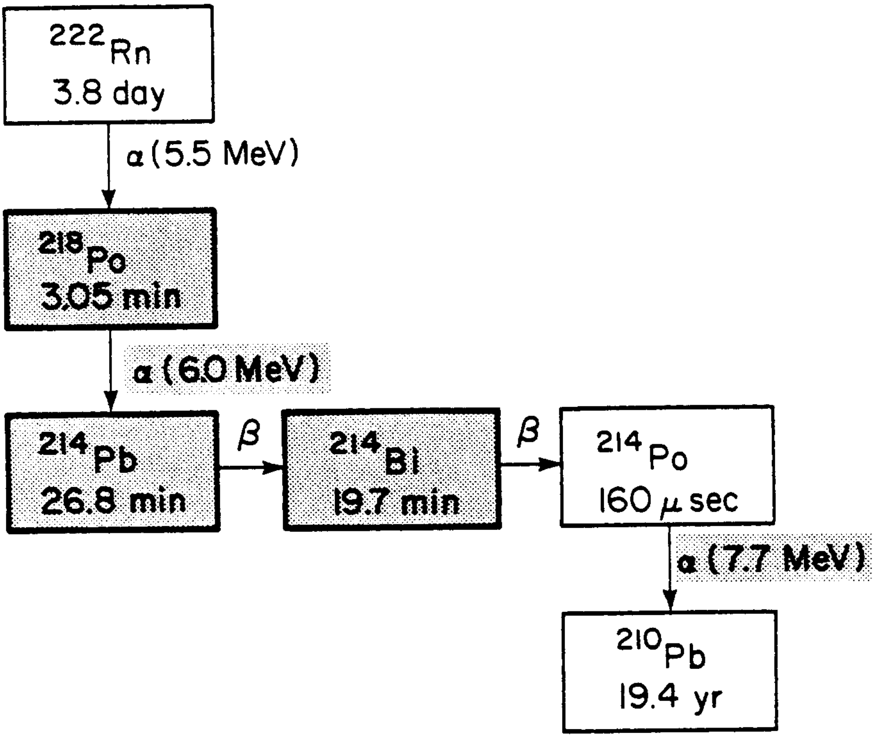

Figure 1.1 Radon decay chain.

Radon progeny concentrations are expressed in terms of total alpha energy emitted by the decay of the short-lived progeny. This potential alpha energy concentration (PAEC) is expressed in units of working levels (WL). (One working level has a value of 1.3 x 10^5 MeV/l, the potential alpha energy per unit volume that would be associated with air containing approximately 100 pCi/l of each of the short-lived progeny.) Radon progeny in U.S. homes concentrations vary from 0.001 to 0.1 WL or more.

Because the incidence of radiation-induced cancer is related to cumulative exposure, the risk factor must be evaluated in terms of duration of exposure. Again, based on the exposure of uranium miners, total exposure is expressed in terms of the working level month (WLM), where exposure to 1 WL for 173 hours (the number of hours in one work month) yields 1 WLM. The annual exposure associated with a constant radon concentration of 1 pCi/l is approximately 0.25 WLM. (This result is based on using an equilibrium factor of 0.5, defined to be the ratio of the actual PAEC to the PAEC if each progeny was to have the same activity concentration as that of radon. Ventilation and plate-out of progeny to walls and other surfaces reduce the equilibrium factor to a value less than one.)

The cumulative exposures (in WLM) at which human and animal cancers are observed are generally higher by an order of magnitude or more than those characteristic of the general indoor environment.

Thus, in order to predict the health effects of decreased indoor ventilation and a corresponding increase in exposure to radon progeny, it is usually necessary to extrapolate beyond the range of exposures for which effects are known. In a small number of cases, however, radon concentrations indoors have been observed to be so high that individual exposure is comparable to the maximum exposure allowed for a miner.

Sources: Radon can enter a building in the following ways (Fig. 1.2): by transport from soil around the foundation through cracks and openings in the structure, through emanation from earth-derived building materials such as concrete, and via spring or well water.

The presence and concentration of radon in a residence depends on several factors: the geographical location of the building, the materials used in construction, its foundation type, the presence of pathways for soil gas to enter the building, the water supply source, and the ventilation rate of the structure. In those cases where there was reason to be concerned about radon levels, studies indicate that the soil under a house is most frequently the principal contributor to indoor radon concentration (Nero and Nazaroff, 1983). Measurements taken in living spaces of homes throughout the United States show that concentrations of radon vary over nearly three orders of magnitude (0.1-50 pCi/l) with an average concentration of approximately 1 pCi/l. In cases where the building materials and/or soil surrounding the

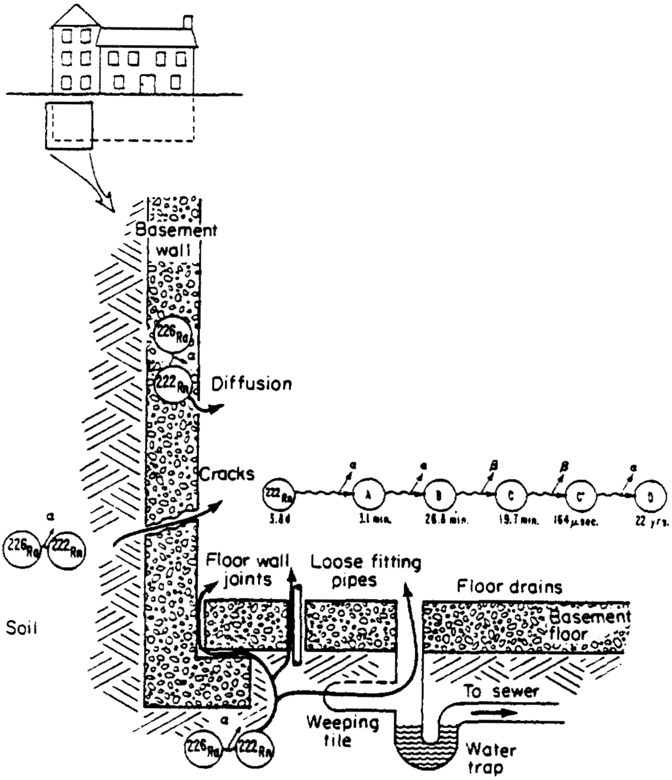

Figure 1.2 Pathways for radon entry in buildings.

building contain high-radium industrial or mining products (e.g., uranium mill tailings), even higher indoor concentrations of radon are occasionally found.

Other factors which can influence the concentration of radon and its progeny in residences are the type of basement or foundation and whether groundwater is used for the domestic water supply (in which case radon concentration in the water may be relatively high and radon in the water may be released indoors). Houses having vented crawl spaces may have lower concentrations than those with unvented crawl spaces, basements, or built as slab-on-grade. Substantial evidence suggests that the mechanisms driving air infiltration also drive radon entry. This coupling between air infiltration rate and radon entry rate will reduce the degree to which indoor radon concentrations vary with the air exchange rate.

2. Formaldehyde

Formaldehyde (HCHO), the simplest of the group of compounds classified as aldehydes, is a pungent, colorless gas the odor of which can be detected at concentrations as low as 0.05 ppm, although the odor threshold is usually around 1 ppm. Primarily an indoor-generated pollutant, its sources are building and insulating materials, furniture, and numerous consumer products. Formaldehyde is used in a wide variety of building materials, mainly in the urea, phenolic, melamine, and acetal resins present in insulation materials, particleboard, medium-density fiberboard, plywood, textiles, adhesives, and so forth. Although building materials such as particleboard can emit formaldehyde continuously for a period of years, its emission rate is known to decrease with time. The half-life of the formaldehyde emission from various materials is, however, not well known.

Health Effects: Relating specific health effects to specific concentrations of formaldehyde is difficult because people vary widely in their tolerances and subjective reactions. Furthermore, irritation and/or sensitization tolerance may develop after several hours of exposure and modify an individual's response. As with other toxins, the duration of the exposure is an important factor in the severity of any associated health effects. Health effects from acute formaldehyde exposure and long-term continuous exposure can be quite different, and knowledge of one cannot be used to predict the other.

Exposure to formaldehyde at concentrations of 0.05 - 0.5 ppm can cause burning of the eyes and irritation of the upper respiratory passages, depending on individual sensitivity and environmental conditions such as temperature and humidity. High concentrations (>few ppm) often produce coughing, constriction in the chest, and wheezing. Studies in rats and mice have shown that concentrations of formaldehyde of a few ppm for several months induce nasopharyngeal carcinoma. The most stringent guidelines or proposed standards for indoor formaldehyde concentrations are generally around 0.1 ppm.

Sources: Major sources of formaldehyde are urea-formaldehyde foam insulation (UFFI), particleboard, medium-density fiberboard, plywood and, to a lesser extent, combustion processes. Urea-formaldehyde (UF) foam insulation has been used extensively in the past, both here and in Europe, as thermal insulation in the walls of existing residential buildings. Installation involves mixing partially polymerized UF resin, a foaming agent (containing an acid catalyst), and air to create a foam that is injected into wall cavities. Some residual formaldehyde in the foam may enter the interior of the building and, under some circumstances, considerable amounts of formaldehyde can be released.

The superior bonding properties and low cost of formaldehyde resins have made them commercially attractive for use in such building materials as plywood and particleboard. In fact, urea-formaldehyde resin is the most common adhesive used in interior-grade plywood and particleboard. Interior-grade plywood, for example, is composed of several thin sheets of wood glued together with UF resin and the primary components of particleboard are small wood shavings glued together with UF resin. In buildings in which these wood products are used for subfloors, cabinets, partition walls, paneling, or furniture, formaldehyde can reach concentrations sufficient to cause irritation of the eyes and upper respiratory tract and possible chronic health effects.

Formaldehyde is also produced during simple combustion processes associated with gas cooking and heating and smoking. Studies conducted in special environmental chambers have indicated that a poorly adjusted combustion appliance can contribute significant quantities of formaldehyde to the indoor environment (Girman et al., 1982).

3. Combustion Products

A wide range of combustion products can be emitted by indoor combustion appliances and cigarette smoke: carbon monoxide (CO), carbon dioxide (CO_2), water vapor (H_2O), nitric oxide (NO), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and formaldehyde (HCHO), as well as respirable particles. Common appliances found in residential buildings are gas-fired stoves, indoor gas-fired water heaters, unvented gas-fired space heaters, gas-fired furnaces, and portable kerosene-fired space heaters. Wood-burning stoves and fireplaces and coal- or wood-burning furnaces can also contribute hydrocarbons and polycyclic organic matter (POM) in addition to the aforementioned pollutants. The combustion byproducts emitted by an automobile allowed to idle in an attached garage can gain entry to the living space of the house.

Health effects: Carbon monoxide enters the body through the respiratory system and reacts primarily with the hemoglobin of the blood. Because the affinity of hemoglobin for CO is more than 200 times that of oxygen, the absorption of CO reduces the oxygen-carrying capacity of blood. Experimental exposure of nonsmokers to 50 ppm for 90 minutes revealed that their discrimination of time intervals was impaired (NRC, 1977). This exposure is likely to produce a carboxyhemoglobin (COHb) level in the blood of about 2.5%, corresponding to that associated with continuous exposure to 10-15 ppm for eight or more hours. Current national ambient air quality standards for carbon monoxide are 9 ppm for an eight-hour average exposure and 35 ppm for a one-hour average exposure.

The present Occupational Safety and Health Administration limit for carbon dioxide exposure in the workplace is 5000 ppm, for a time-weighted eight-hour daily average. Much of the research on physiological effects of CO_2 exposure has been done to establish safe limits for submarine crews and astronauts. The National Aeronautics and Space Administration limit for a six-month exposure in spacecraft is 10,000 ppm. The Bioastronautics Data Book states that "for prolonged exposures of 40 days, concentrations of CO_2 in air less than 5000 ppm cause no known biochemical or other effect. Concentrations between 5,000 and 30,000 ppm cause adaptive biochemical changes which may be considered a mild physiological strain; and concentrations above 30,000 ppm cause pathological changes in basic physiological functions" (NASA, 1973).

Nitric oxide (NO) and nitrogen dioxide (NO_2) are both formed as products of combustion. Of the two, nitrogen dioxide is the more reactive, and poses the greater health hazard. Exposure to concentrations of 5 ppm can cause respiratory distress; concentrations of about 50 ppm can cause chronic lung disease; exposure greater than 150 ppm can cause death. In controlled laboratory studies, concentrations in the range of 0.10 to 0.50 ppm were associated with respiratory distress in individuals having chronic respiratory ailments. The EPA National Ambient Air Quality Standard (NAAQS) for outdoor exposure to NO_2 is 0.05 ppm.

Epidemiological studies in large groups of persons exposed to long-term, low-level concentrations of nitrogen dioxide in their homes are inconclusive. Two studies, both of primary school children in England, suggest a relationship between respiratory illness and the NO_2 exposure from gas-stove cooking (Florey et al., 1979; Melia et al., 1979). On the other hand, according to similar studies by Keller et al., (1979, 1979b) neither an increase in respiratory disease nor a decrease in pulmonary function was associated with the use of gas stoves. Another ongoing study seems to indicate that children raised in homes with gas stoves have a greater history of respiratory illness before age 2 than do children raised in homes with electric stoves (Speizer et al., 1980).

Individuals inhale a wide variety of particulate matter into their respiratory systems. Where particles are deposited and how long they are retained depends on their size, chemical composition, and density. Particles larger than 15 microns in diameter are filtered out by the nose and dislodged within several days. Smaller particles settle on the tissues of the upper respiratory tract, the smallest particles ($<$ 2.5 microns) penetrate to the alveoli, the small air sacs in the lungs. Particles deposited in the lungs can remain there for months, and can cause adverse health effects for long periods of time.

Particulate matter may exert a toxic effect because: (1) it is intrinsically toxic due to its chemical or physical characteristics (e.g., lead and asbestos) or (2) it acts as a carrier of an adsorbed toxic substance (e.g., radon progeny can be attached to particles). Carbon particles such as those emitted in combustion processes are efficient adsorbers of many organic and inorganic compounds and can carry toxic gases such as SO_2 into the lungs. This can have a potentiating effect on the human body; that is, particles that contain an attached toxic substance can increase a person's physiological response to that substance to a level above what it would be otherwise.

Sources: There have been many field studies monitoring the concentrations of various pollutants in residential buildings having gas stoves. In almost all cases, indoor levels of CO, NO_2, and NO have exceeded outdoor levels. In a number of cases, indoor levels comparable to existing outdoor air quality standards for NO_2 and CO have been reported.

Several studies have been performed to assess the level of air contamination resulting from operation of unvented gas-fired and kerosene-fired space heaters. Experiments on a portable, convective-type kerosene space heater operated for one hour in a 27-m^3 chamber showed that at a ventilation rate of 0.4 ach, CO and particle concentrations were low but CO_2 and NO_x concentrations were high (Traynor et al., 1983c). After 1 hour of operation, CO_2 concentrations reached 10,000 ppm, twice the eight-hour occupational standard established by the U.S. Occupational Safety and Health Administration (OSHA), and the NO_2 concentration was greater than 1,5 ppm over background, or six times the California peak one-hour outdoor standard.

Wood-burning stoves, wood-burning furnaces, and fireplaces represent a significant indoor source of CO, NO_x (i.e., NO_2 plus NO), hydrocarbons, and respirable particles, including the carcinogenic polycyclic organic matter (POM) compounds. Laboratory and field measurements show that pollutant emission rates from wood combustion can vary over a wide range. For example, CO emissions can vary from 4 to 400 grams per kilogram (g/kg) of wood burned; particulate emissions from 0.5 to 63.5 g/kg; total hydrocarbons from 0.2 to 48.5 g/kg; NO_x from 0.2 to 7.3 g/kg; and POM from 0.004 to 0.37 g/kg (Hubble et al., 1981).

One of every three persons between the ages of 17 and 64 smokes cigarettes regularly. Surveys in eight cities show that from 54 to 76% of homes have one or more smokers (Ferris et al., 1979; Lebowitz and Burrows, 1976). In addition, smokers and nonsmokers are often exposed to smoke at their workplaces and during social activities. Exposure to other peoples' tobacco smoke is now referred to as passive smoking. Closely connected to "active" and "passive" smoking are the terms "mainstream" and "sidestream" smoke. Mainstream smoke is the smoke drawn through the tobacco during inhalation, resulting in higher temperatures, more complete combustion, and a greater degree of absorption by the tobacco and, if present, the filter. Mainstream smoke accounts for approximately 4% of the total smoke produced. Sidestream, defined as smoke arising from the smoldering tobacco, accounts for the majority (96%) of gases and particles produced. The composition of mainstream smoke exhaled by a smoker varies considerably depending on smoking behavior, burning temperature, and type of filter. Because of the length of the burn and the burn temperature, sidestream smoke is a more important source of local air contamination for many substances such as CO, nicotine, ammonia, and aldehydes, than mainstream smoke. The passive smoker, of course, does not receive nearly as large a dose of smoke as does the smoker.

A number of indoor air pollutants arise from tobacco smoke; some of the more important ones are particles, CO, benzo-a-pyrene, nicotine, nitrosamines, acrolein, and other aldehydes. Spengler and coworkers measured levels of respirable suspended particles (RSP) in 80 homes, with and without smokers, across six cities (Spengler et al., 1981). RSP levels were essentially the same ($\sim$23 µg/m^3) indoors and outdoors in the homes without smokers. The mean RSP concentrations indoors for homes with one and two smokers were 43 to 75 µg/m^3, respectively; however, 24-hr average indoor RSP concentrations frequently exceeded 200 µg/m^3 in homes with cigarette smokers. These data, collected over a three-year period, illustrate the significant contribution of cigarette smoke to indoor particulate concentrations.

Vented combustion appliances can be a significant indoor source of combustion products if there is leakage of these products into the indoor air. The fact that woodstoves and fireplaces can be significant pollutant sources was mentioned above. Leakage can also occur from other appliances if the vent system performs improperly or if there is a crack in the heat exchanger of a furnace.

C. Role of Control Techniques in Solution of Indoor Air Quality Problems

Many techniques exist to control the concentrations of the pollutants introduced above and other indoor pollutants that are only now being recognized as significant. The purpose of this report is to provide a current review and evaluation of existing control techniques. However, this activity must be placed in the context of the wider research framework in which the study of indoor air quality is found.

Effective and efficient control of indoor pollutant concentrations depends upon an understanding of the chemical and physical properties of the sources of the individual pollutants being considered. Failure to understand these factors may lead to the use of a control technique that exacerbates rather than reduces an indoor air quality problem or that does not solve the problem in an efficient manner. Coupled with the need to improve our understanding of sources is the continuing need to understand the health effects of indoor pollutants. A control strategy that holds the concentration of a pollutant below 1 ppm may be adequate for one pollutant but totally inadequate for another. Thus the development and evaluation of indoor air quality control techniques must be performed in parallel with research on pollutant source characterization and health effects.

2. Ventilation

A. Background

Ventilation, the exchange of indoor air for outdoor air, is a widely used technique to help maintain acceptable indoor air quality. The connections between ventilation rate and human comfort and health were discussed in writing as early as the middle of the 18th century (Billington, 1982), and were probably recognized long before that time. The historical and present popularity of ventilation may be due to a number of factors. While the effectiveness of many IAQ control techniques is limited to one or a few pollutants, ventilation is effective in reducing the indoor concentrations of all pollutants, whenever the outdoor concentration is lower than the indoor concentration. In addition, ventilation is often the simplest or only technique that is readily available. A major drawback of ventilation, however, is the energy required to heat or cool the ventilation air and this factor is becoming increasingly important as energy prices rise.

In the past, residences have been ventilated primarily by infiltration, the uncontrolled leakage of air through cracks in the building envelope, and by the natural flow of air through open windows and doors. Mechanical (i.e., fan-forced) ventilation has often been used in larger buildings, sometimes in addition to natural ventilation. A recent trend is to use mechanical ventilation more frequently in residential buildings.

Before discussing various methods of ventilation in detail and reviewing available performance data, several mass balance models are presented and used to illustrate the relationships between ventilation rate and indoor pollutant concentrations.

B. Mass Balance Models

1. Background

It is generally not possible to estimate the impact of changes in ventilation rate on indoor concentrations without consideration of other factors that also affect the indoor concentration. One method of estimating the impact of ventilation is to use mathematical equations that are derived from mass balances for the pollutants in the indoor air. Each equation presented below is based on assumptions of a perfectly mixed indoor space and steady-state conditions, although more complex equations that describe indoor concentrations during transient periods are available in the literature. Although the equations are presented in the context of a discussion of ventilation, they are also useful for estimating the impact of changes in many other variables on indoor concentration.

2. Models

The most simple relationship between indoor pollutant concentration and an exchange rate is expressed by the equation

$$C_i = S/a \qquad (2.1)$$

where: C_i = indoor pollutant concentration

S = pollutant source strength per unit volume of indoor air (i.e., the rate at which pollutants are emitted or transported into the indoor air divided by the volume of indoor air), and

a = air exchange rate (i.e., flow rate of ventilation air divided by volume of indoor air).

This equation is valid only when the outdoor pollutant concentration is negligible, the pollutant source strength is unaffected by the ventilation rate, and there are no pollutant removal mechanisms other than ventilation.

In most cases, the assumptions made for derivation of Equation 2.1 are not valid. To account for the outdoor pollutant concentration and additional pollutant removal processes, we can use the following equation which is similar to that used earlier by Alzona et al.

(1979), Dockery and Spengler (1981), and Traynor et al. (1982, 1983)

$$C_i = (S + a\ PC_o) / (a + k + \lambda + R) \qquad (2.2)$$

where: P = fraction of outdoor pollutant that penetrates the building envelope or ventilation system,

C_o = concentration of pollutant in the outdoor air,

k = the net pollutant removal rate (per unit volume of indoor air) by plate-out on or reaction with indoor surfaces or by reaction within the indoor air expressed as a first order decay constant,

λ = the rate of pollutant removal by radioactive decay, and

R = the rate of pollutant removal by operation of an air cleaner.

The actual relationship between ventilation rate and indoor air concentration is often more complicated than indicated by Equation 2.2. For example, the pollutant source strength may be a function of ventilation rate, indoor pollutant concentration, indoor temperature and humidity, occupants' activities, time, and other factors. In addition, most indoor spaces are not perfectly mixed as assumed for derivation of the equation and imperfect mixing can greatly affect the relationship between ventilation rate and indoor concentration as discussed later.

To utilize Equation 2.2, values for the various input parameters are required and only limited data are available. Here we present only values for the reaction constant k and penetration factor P. For gases that are nearly non-reactive in indoor environments, the reaction constant and penetration factor are essentially zero and unity, respectively. Measured values of these constants for nitrogen dioxide, formaldehyde, and particulates are listed in Table 2.1. Pollutant removal by reaction and plate-out is poorly understood. The measured reaction constants (Table 2.1) have been used to account for pollutant removal that is largely unexplained. It is likely that these removal mechanisms are a function of the nature and quantity of indoor surfaces and the amount of indoor air movement. For the case of removal by chemical reactions, the removal rate may be a function of the indoor pollutant concentration. In addition, available data for the reaction constants are generally from transient experiments which do not permit the process of physical adsorption of pollutants followed by later desorption to be distinguished from permanent removal processes. Thus, the average and range of pollutant removal by reaction and plate-out is largely unknown. However, the reported removal rates significantly affect the relationship between ventilation rate and indoor pollutant concentration as shown later.

Table 2.1. Reported values for the reaction constant and penetration factor.

Pollutant	Reaction or Deposition Constant ± Standard Deviation (h^{-1})	Penetration Factor	Source	Comments
nitrogen dioxide	0.18, 0.29		Traynor et al., 1983b	
nitrogen dioxide	0.20 ± 0.13		Traynor et al., 1983	
nitrogen dioxide	1.29 ± 0.67		Traynor et al., 1982	
nitrogen dioxide	1.39		Moschandreas and Stark, 1978	
nitrogen dioxide	0.16 to 0.5		Ozkaynak et al., 1982	
nitrogen dioxide	0.83		Wade et al., 1975	
formaldehyde	0.4 ± 0.24	1.0	Traynor et al., 1982b	
particles	0.48 ± 0.21	0.4 ± 0.1	Traynor et al., 1982b	particle diameter < 0.5 μm
particles	0.03 to 0.35		Offermann et al., 1984	function of particle size

For several reasons, the impact of ventilation rate on radon progeny concentrations must be modeled by a more complex equation. The source of radon progeny is the radioactive decay of radon. Because a change in the ventilation rate will change the indoor radon concentration, it clearly has an effect on the source strength for radon progenies. (In some instances, the source strength for radon itself may be affected by the ventilation rate as discussed later, however, for the equation presented below, a radon source strength that is independent of ventilation rate has been assumed.) Another factor that must be accounted for is the removal of radon progeny by radioactive decay. In addition, most researchers use the potential alpha energy concentration (PAEC) to express the total concentration of radon's short-lived progeny in units of working levels (WL), thus, each decay product is not treated independently. Actually, the PAEC is not a direct measure of concentration but instead indicates the potential alpha energy emission from all radon progeny present in the air. An equation that relates PAEC to ventilation rate and other factors is

$$\mathrm{PAEC} = S\,\lambda_A / [(\lambda_R + a)(\lambda_A + a + k_A)] \times \tag{2.3}$$
$$[0.00103 + (\lambda_B/(\lambda_B + a + k_B))(0.00507 + 0.00373(\lambda_C/(\lambda_C + a + k_C)))]$$

where: S = radon source strength (pCi/-h),

λ_R = radioactive decay constant for radon = 0.00758 h^{-1},

λ_A, λ_B, λ_C = radioactive decay constants for progeny A, B, and C, i.e., respectively, $\lambda_A = 13.7\ h^{-1}$, $\lambda_B = 1.55\ h^{-1}$, $\lambda_C = 2.11\ h^{-1}$,

a = the air exchange rate (h^{-1}), and

k_A, k_B, k_C = the first order reaction constants (usually called plate-out or deposition constants) for progeny A, B, and C, respectively (h^{-1}).

This equation follows from work by several researchers (see, for example, Rudnick et al., 1983). A major difficulty in using the equation is to determine appropriate values for the plate-out constants. The rate of radon progeny removal by plate-out has been investigated or reported by a number of researchers (for example: Bruno, 1983; George et al., 1983; Knutson et al., 1983; Rudnick et al., 1983; Scott, 1983; Swedjemark, 1983; and Offermann et al., 1984). The rates of plate-out depend highly on the fraction of progeny that are attached to particulates which, in turn, is a function of the indoor particle concentration. Offermann et al., (1984) give values of $k_A = 7.8\ h^{-1}$, $k_B = 1.8\ h^{-1}$, and $k_C = 0.5\ h^{-1}$ for a low indoor particle concentration of 3000 particles/cc and values of $k_A = 1.4\ h^{-1}$, $k_B = 0.4\ h^{-1}$, and $k_C = 0$ for a high indoor particle concentration of 30,000 particles/cc. An important factor not accounted for in Equation 2.3 is that the indoor particle concentration and thus the plate-out constants for radon progeny will generally be a function of the ventilation rate; this is taken into account in a more complete model that is presented by Offermann et al. (1984).

The relationship between ventilation rate and indoor formaldehyde concentration can also be complex for the following reasons:

1. Multiple sources of formaldehyde may be present.
2. Some sources of formaldehyde may be separated from the indoor air by a permeation barrier (e.g., a carpet).
3. The emission rate of formaldehyde from many sources varies with indoor formaldehyde concentration and the relationship between emission rate and indoor concentration varies between sources.
4. The emission rate of formaldehyde depends on the temperature of the source and the humidity of the surrounding air.
5. It is possible formaldehyde is removed by reaction with indoor surfaces, although there is insufficient information to determine if reaction actually occurs.

The following equation from work by Matthews et al. (1983) accounts for multiple formaldehyde sources:

$$C_i = C_o + (1/aV)(\sum_i S_i A_i - \sum_j k_j A_j) \tag{2.4}$$

where: S_i = the source strength per unit area of source material i,

A_i = the area of source material i,

k_j = the reaction constant for formaldehyde with surface j, per unit area

V = the volume of indoor air, and

A_j = the area of surface j

The magnitude of many of the formaldehyde sources will depend on the surrounding formaldehyde concentration (i.e., the source strength will increase when the surrounding concentration is decreased). For sources that are not separated from the indoor air by a permeation barrier, the following linear equation is given (Matthews et al., 1983b):

$$S_i = - M_i C_i + b_i \tag{2.5}$$

where M_i and b_i are constants that are determined experimentally. The linear relationship between source strength and concentration in Equation 2.5 will generally be valid only for a restricted range of concentration. The equation for the source strength of a formaldehyde emitter that is covered by a permeation barrier is (Matthews et al., 1983b)

$$S_i = - M_i C^* + b_i = D/L\ (C^* - C_i) \tag{2.6}$$

where: C^* = the concentration of formaldehyde under the permeation barrier,

D = the diffusion coefficient for formaldehyde diffusion through the barrier, and

L = the thickness of the barrier.

If changes in the ventilation rate affect the temperature of the formaldehyde emitter or the indoor humidity, then modifications of the model to account for the impact of these parameters on formaldehyde source strength would be required. We have not included these modifications in this paper--further information is presented by Andersen et al., 1975; Long et al., 1979; Berge et al., 1980; and Matthews et al., 1983. The model presented here has not been extensively verified in field studies and will be very difficult to use when several sources of formaldehyde are present because of the large amount of input data required and because much of this data is not readily available. It is presented here because the validity of less complex models is questionable and also to illustrate the difficulties in determining the impact of ventilation or other control measures on formaldehyde concentration.

3. Discussion of Models

The mass balance models presented above have been used to demonstrate various potential relationships between ventilation rate and indoor pollutant concentration and the effects of various factors on these relationships. Figure 2.1 illustrates that the rate of pollutant removal by processes other than ventilation can have a substantial impact on the indoor concentration when air exchange rates are moderate or low; at high air exchange rates, however, these removal processes have a much smaller effect on indoor concentration.

The relationships between a change in ventilation rate and a change in indoor concentration are better illustrated in Figure 2.2. The sensitivity of indoor concentration to air exchange rate is reduced as the rate of pollutant removal by processes other than ventilation is increased (compare curves C, E, and F or curves A and B). The PAEC (curves A and B) is shown to be highly sensitive to the ventilation rate for the case shown of a constant radon source strength because a change in ventilation affects both the source strength and removal rate of radon progeny. The rate of progeny plate-out does not have a large impact on the relationship between a change in ventilation rate and a change in PAEC (compare curves A and B).

One possible relationship between air exchange rate and indoor formaldehyde concentration is also illustrated in Figure 2.2 (curve D). This curve is for a structure with two formaldehyde sources (particleboard and medium-density fiberboard) that are not separated from the indoor air by a permeation barrier. It is based on

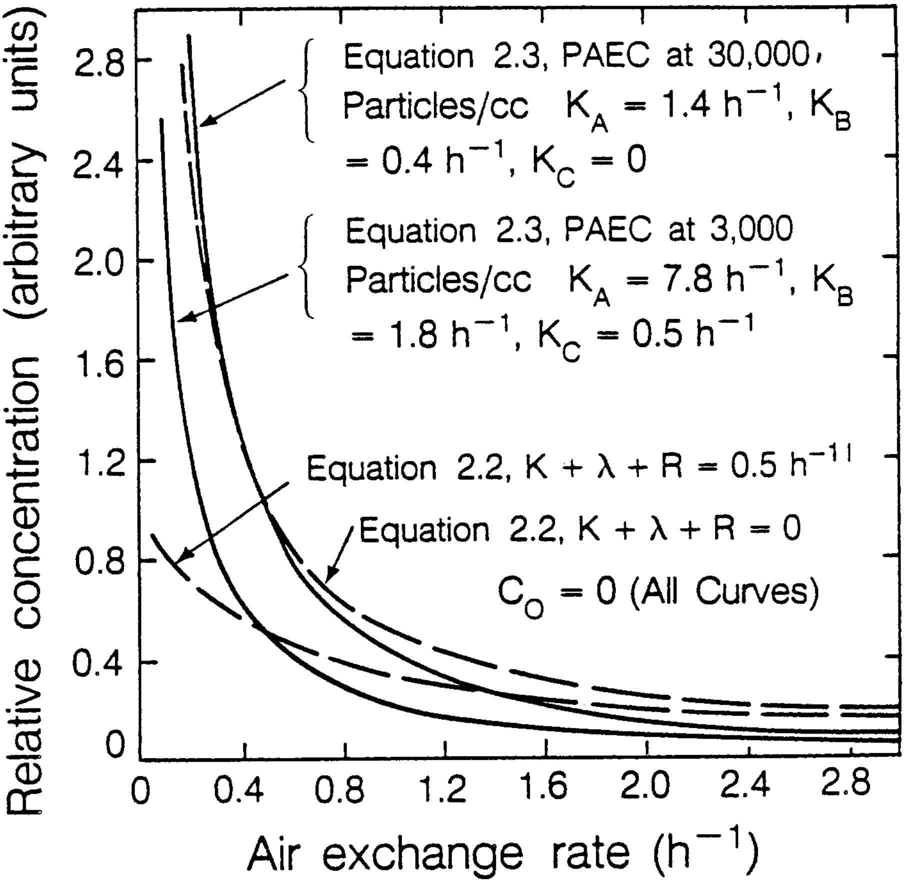

Figure 2.1 Relative concentration versus air exchange rate for variable pollutant removal processes other than ventilation.

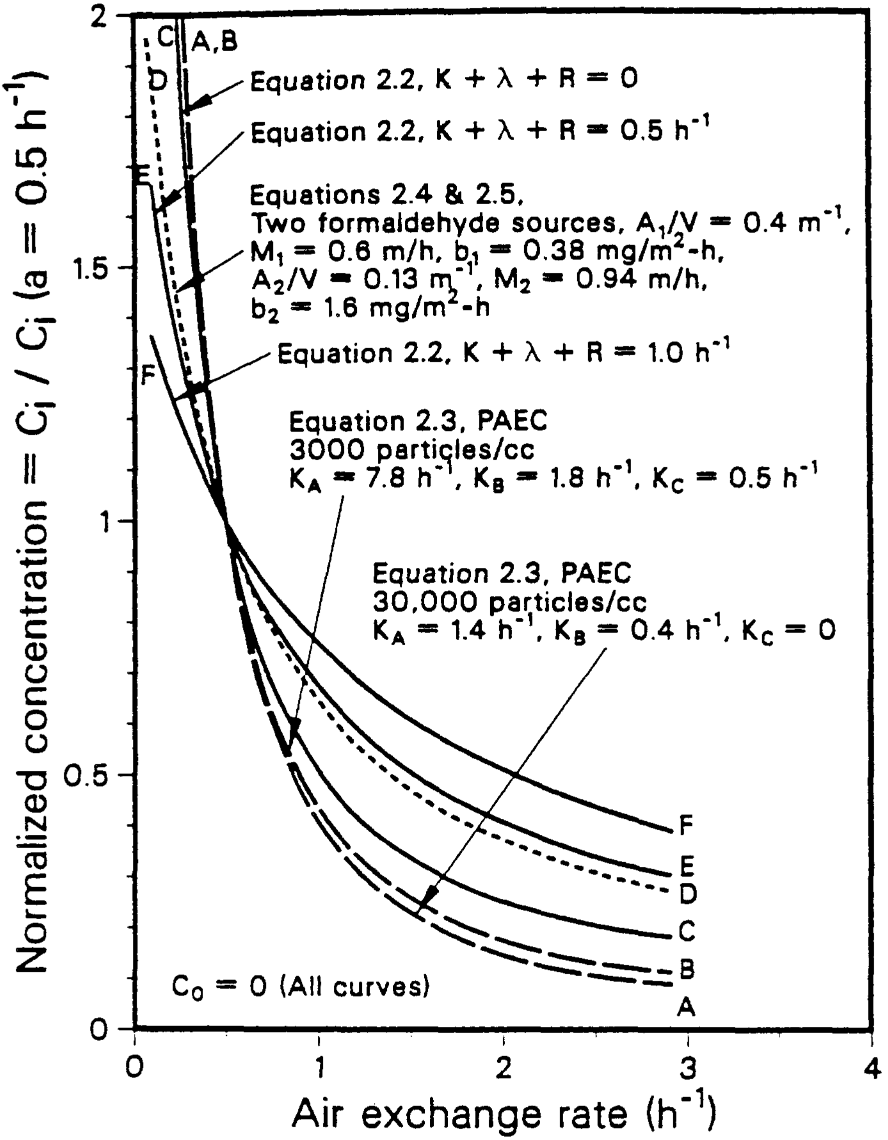

Figure 2.2 Relationship between a change in indoor concentration and a change in air exchange rate.

assumptions of no formaldehyde removal by reaction, no change in indoor humidity, and a linear increase in formaldehyde source strength with a decrease in indoor concentration. Particular sets of values of M_i, b_i, and A_i/V (see Equations 2.4 - 2.6) were chosen to generate the curve based on data from Matthews et al. (1983b). The important point to note is that the coupling between formaldehyde source strength and indoor concentration can substantially reduce the impact of ventilation rate on concentration.

While the theoretically derived relationships differ substantially, in general, the indoor concentration increases rapidly as air exchange rates become low. Avoiding unusually low air exchange rates is, therefore, an important component of strategies for maintaining acceptable indoor air quality. Because of the large range in pollutant source strengths, however, maintaining a typical ventilation rate does not ensure low indoor pollutant concentrations. In addition, the energy requirements associated with ventilation make it generally impractical to increase the ventilation rate by more than about one air change per hour; an increase of this magnitude will only have a large impact on indoor concentration when the initial air exchange rate is low. Thus, the strategy of avoiding unusually low ventilation rates should be combined with explicit efforts to minimize pollutant source strengths and, in some instances, utilization of pollutant removal processes other than ventilation. A detailed discussion of pollutant control measures other than ventilation is presented in later sections of this report.

C. Impact of Ventilation Rate on Pollutant Source Strengths

In the previous section, the relation between ventilation rate and both radon daughter and formaldehyde source strengths were mentioned. The source strength for radon may also depend on the ventilation rate or the method of ventilation. In most structures with elevated radon concentrations, the primary source of indoor radon is thought to be radon that emanates from soil and is transported into structures through cracks or other openings by pressure-driven flow. (Nazaroff and Nero, 1984). The rate of entry may increase if the interior of the structure, adjacent to these entry locations, is depressurized. A depressurization of this type may be caused by the stack effect which is one of the primary driving forces for infiltration. The stack effect results from the force of gravity and the difference between the density of outdoor air and the heated indoor air; it increases with the indoor-outdoor temperature difference. Thus, the rate of ventilation by infiltration and the rate of radon entry (i.e., the radon source strength) may, in some instance, both increase with an increase in the indoor-outdoor temperature difference.

Nazaroff et al. (1981b) present data from a number of residences that indicates a positive relation between radon source strength and ventilation rate. The data from one residence is presented in Figure

2.3. In a later report, Nazaroff et al. (1983) discussed results from an intensive study of a single residence. Based on the data from a five-month period, the authors developed a model for radon entry into the residence that included entry due to molecular diffusion of radon through the concrete floor and walls of the basement and also to pressure-driven flow of soil gas containing radon through various openings. In the model, the rate of radon entry due to flow of soil gas was proportional to the air exchange rate. Additional evidence that depressurization can increase the radon source strength was obtained in the same study when large increases in radon source strength coincided with increases in the ventilation rate and depressurization of the house interior caused by fireplace operation.

Balanced mechanical ventilation provided by a residential air-to-air heat exchanger (described later) apparently did not increase the radon source strength in studies by Nazaroff et al. (1981) and Offermann et al. (1982, 1982b). In these studies, the radon concentration followed the trend indicated by the simple Equation 2.1 which is based on an assumption of constant source strength. One would not expect a change in radon source strength when balanced mechanical ventilation systems (i.e., systems that supply and exhaust equal amounts of air) are operated because these systems will not depressurize the residence unless they are operated with an imbalance in flow rates. In contrast, mechanical exhaust ventilation, which is common in Europe, will depressurize the ventilated structure and, therefore, may increase the

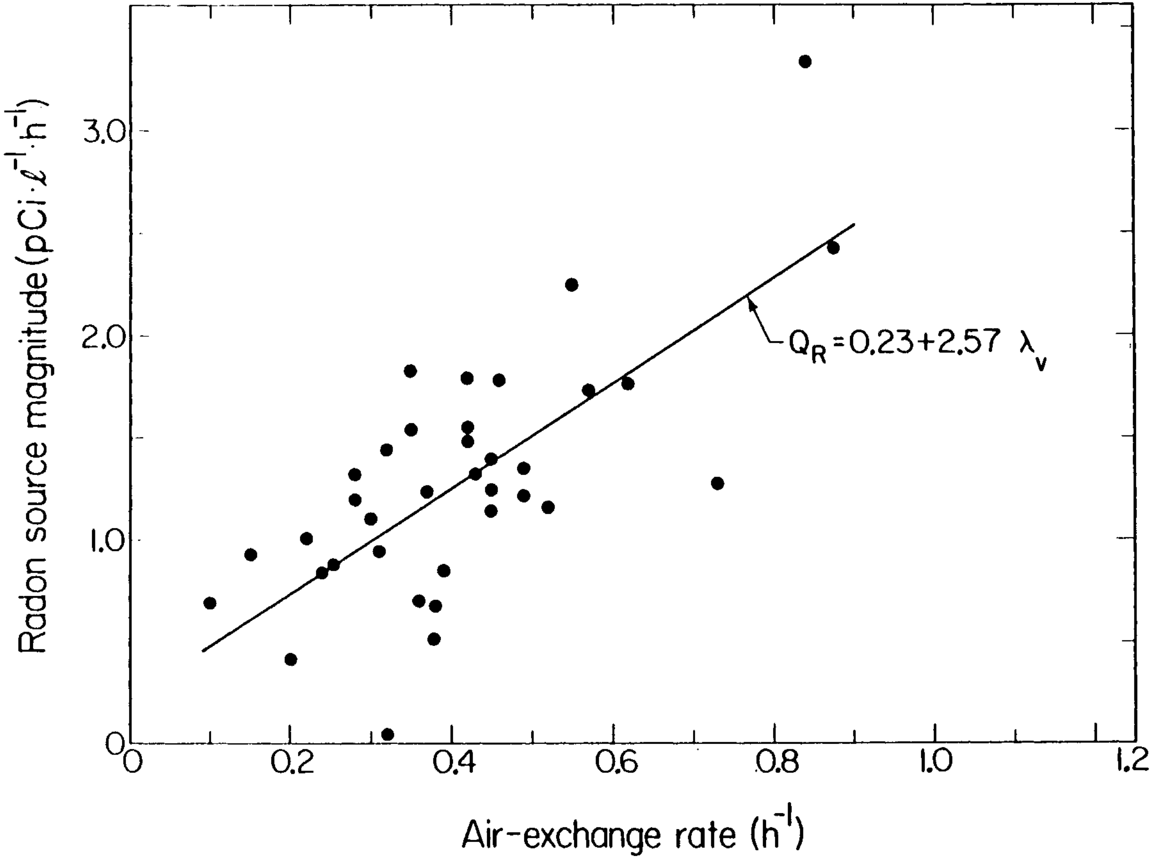

Figure 2.3 Example of coupling between radon source and infiltration rate (Source: Nazaroff et al., 1981b).

radon source strength, although no data are available to directly relate exhaust ventilation and radon source strength.

Based on the limited available information, it may be possible to prevent increases in radon entry with increased ventilation by taking measures to reduce the proportion of entering air or soil gas that has a high concentration of radon. Appropriate measures could include sealing paths for radon entry, e.g., cracks in the floor or slab, providing openings for incoming ventilation air, or ventilating a crawl space as discussed later.

It is clear from the literature that, in some instances, there is a positive relationship between ventilation rate and radon source strength. At this time, however, it is not known whether this relation holds in a small or large percentage of houses; in addition, there are insufficient data to quantify the relationship except perhaps in isolated structures.

An additional instance in which the ventilation rate can affect the pollutant source strength is noted. In a structure that is very air-tight, severe depressurization caused by fireplace operation, mechanical exhaust ventilation, or some other factor may reverse the flow of air through the vent stacks of combustion appliances. If such a reversal of flow occurs, exhaust products from the combustion appliances will enter the indoor air with potentially hazardous results. Installing combustion appliances in a room isolated from the occupied space and having its own supply of combustion air is one method of preventing this type of problem. Another option is to use a high-efficiency combustion appliance which employs a fan or a technique of pulsed combustion to force the exhaust products out the vent.

D. Methods of Ventilation

In the preceding discussion, simplified models to relate ventilation rate and pollutant concentration were presented and the impacts of ventilation on pollutant source strengths were discussed. In this section, several techniques of ventilation are described (with an emphasis on ventilation of residences) and literature on the performance of each technique is reviewed.

1. Natural Ventilation

The uncontrolled leakage of air through cracks and other openings in the building envelope (i.e., infiltration) and the flow of air through open windows, doors, and stacks constitute natural ventilation. Natural ventilation is driven by naturally occurring differences between the air pressure interior and exterior to these openings in the building envelope. The pressure differences are caused by wind and also by the stack effect (described earlier). An obvious method to reduce indoor pollutant concentrations is to increase the rate of natural ventilation. This can be accomplished by building the structure so that it is less air-tight (e.g., has more cracks), by opening windows or doors, or by using a ventilation stack.

The literature on natural infiltration is fairly extensive. A number of models are available for prediction of infiltration rate; a comparison of several of these models is provided by Liddament and Allen (1983). Infiltration rates of houses can vary from less than 0.1 to several air changes per hour (ach). An average air exchange rate for U.S. residences with no open windows or doors has been estimated to be in the range of 0.5 to 0.9 ach (Grimsrud et al., 1983). Major disadvantages of infiltration are that it is not controllable (i.e., it varies with weather conditions) and that recovery of energy from the exfiltrating air is not possible. The major advantage of infiltration is that no investment is required for purchase, maintenance, or operation of ventilation equipment.

In contrast to infiltration, the rate of ventilation by window opening can be controlled to some degree by the occupants. Window opening behavior increased with outdoor temperature and with the number of openable windows (Conan, 1982). It is not likely that window opening practices are suitable for maintaining acceptable indoor concentrations of all pollutants. Some pollutants can be sensed by the occupants (e.g., by smell) who may respond by opening windows, but other pollutants (e.g., radon) cannot be detected.

A method to increase the rate of natural ventilation is to utilize a ventilation stack which is essentially an open chimney that extends above the structure. Air is driven through the stack due to wind and stack effects. The amount of ventilation provided by the stack varies with weather conditions. Energy recovery from the stack air is usually not considered practical because there is generally an insufficient driving force to move air through a restrictive heat recovery system.

2. Balanced Mechanical Ventilation With Heat Recovery in an Air-To-Air Heat Exchanger

Background: A major disadvantage of using ventilation to control indoor pollutant concentrations is the energy required to heat or cool the ventilation air. To overcome this disadvantage, techniques of ventilation have been developed in which energy is recovered from the exhausted ventilation air. One such technique is to provide balanced mechanical ventilation and recover energy in an air-to-air heat exchanger. Using fans, roughly equal amounts of air are supplied to and exhausted from the structure. In an air-to-air heat exchanger, heat is transferred from the warmer to the cooler airstream, ideally without any mixing between the two streams. Due to the transfer of heat, the incoming air is preheated in the winter and precooled in the summer, thus reducing the load on the heating or air conditioning system. In some air-to-air heat exchangers, only heat is transferred between airstreams; these are often referred to as "sensible exchangers". In other exchangers, often called "enthalpy exchangers", both heat and moisture are transferred between airstreams.

Air-to-air heat exchangers have been used for some time in industry and commercial buildings and a large variety of designs are available. A description and discussion of available designs is provided by the American Society of Heating, Refrigerating, and Air Conditioning Engineers (ASHRAE) (1983). In recent years, the use of mechanical ventilation systems with air-to-air heat exchangers (MVHX systems) have become increasingly common in residences. Since the use of MVHX systems is perhaps the most commonly cited technique for controlling air quality in residences, a detailed discussion of their performance is presented below.

Description: Residential MVHX systems are described in various papers (see, for example, Fisk et al., 1980 or The ECE Group and The Ontario Research Foundation, 1982). A residential heat exchanger (Fig. 2.4) is a system that generally consists of a core, where heat is transferred between the two airstreams, a pair of fans, and a case that contains the core, fans, and fittings for attachment to ductwork. Some models also contain coarse air filters within the case.

Most available models of residential air-to-air heat exchangers are designed to be attached to a ductwork system. Air is withdrawn from various locations throughout the residence and supplied to other

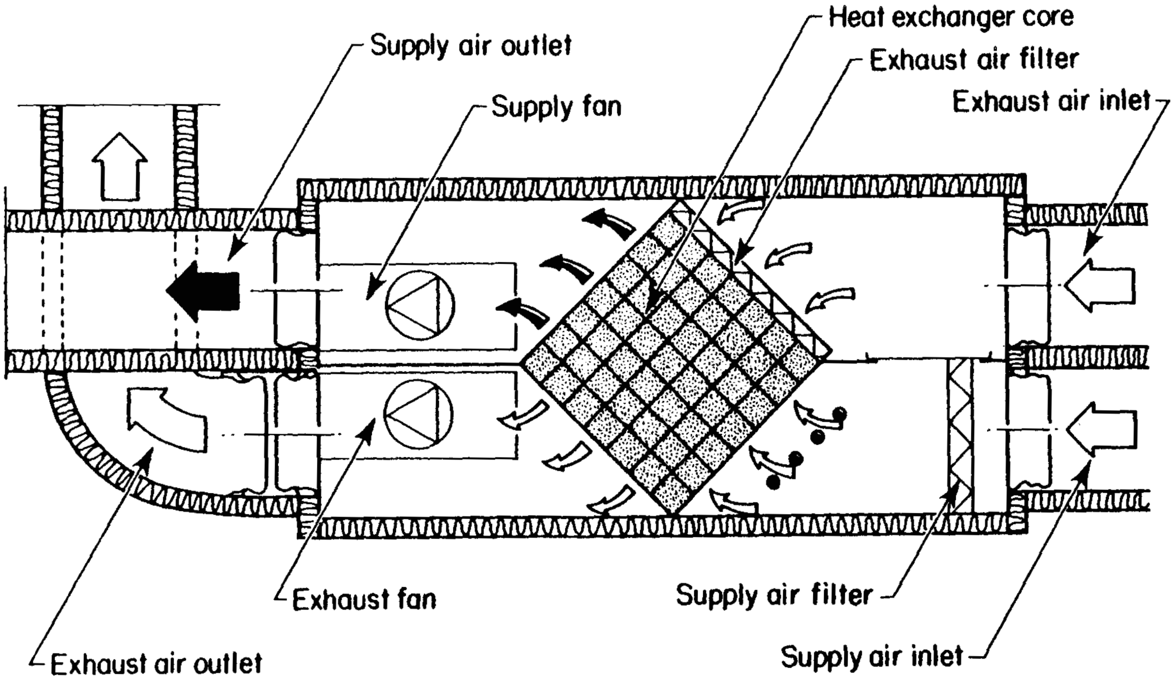

Figure 2.4 Schematic diagram of an air-to-air heat exchanger.

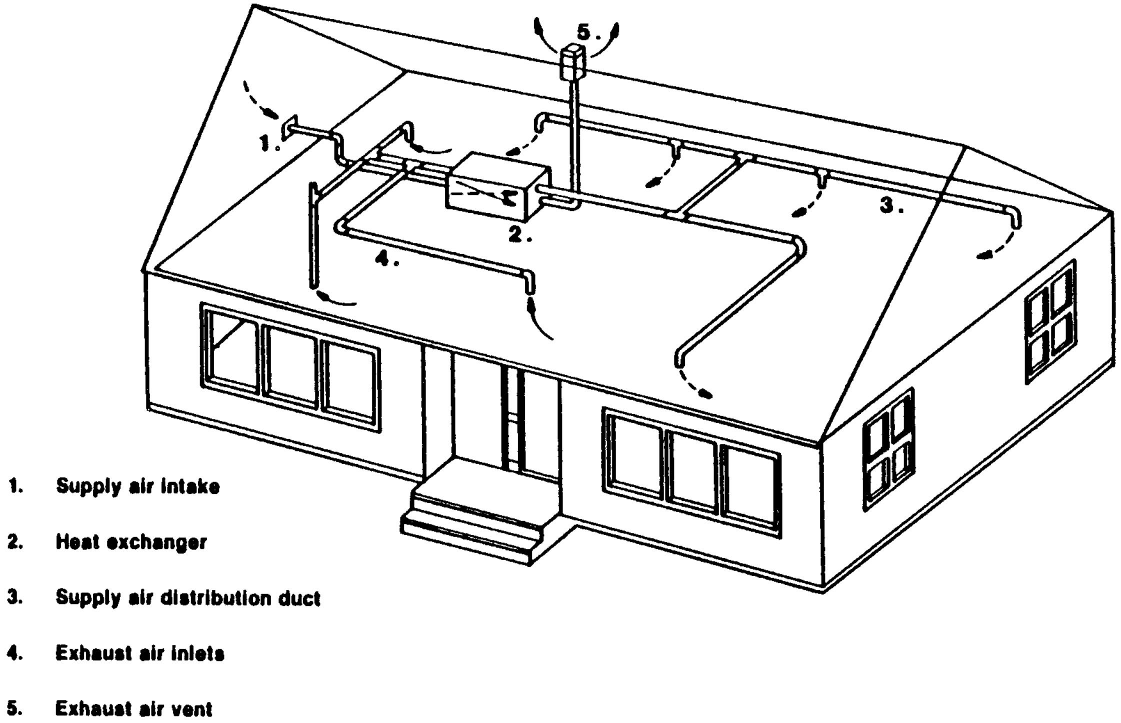

Figure 2.5 Fully ducted installation of an air-to-air heat exchanger.

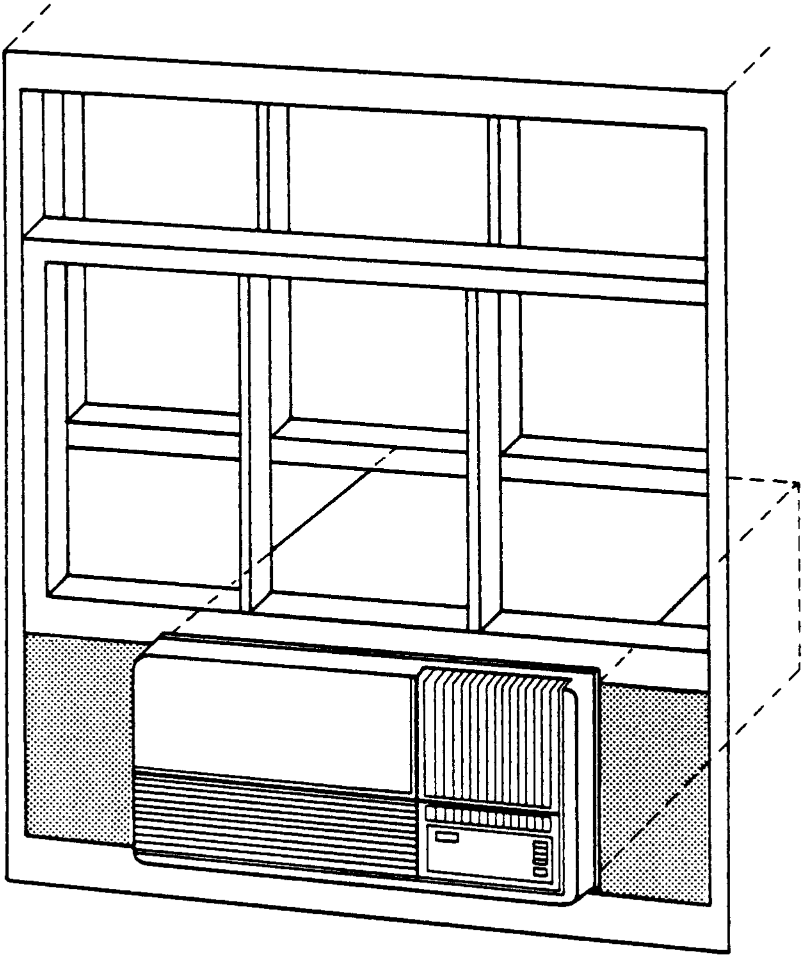

Figure 2.6 Window installation of an air-to-air heat exchanger.

locations via the duct system (Fig. 2.5). Hamlin and Besant (1982) make recommendations regarding the installation of such systems. Air-to-air heat exchangers that are installed through walls or windows (Fig. 2.6) and do not utilize a duct system are also available and command a large share of the U.S. market. The initial and installation costs of these unducted systems are generally lower than the costs for ducted systems.

Impact on Indoor Air Quality: Few investigations have been conducted to determine the impact of ventilation through MVHX systems on indoor air quality. Nazaroff et al. (1981) installed a ducted MVHX system in a Maryland home with high radon concentrations. The radon concentration in the home was monitored over a two-week period while varying the amount of ventilation. Radon concentration was inversely proportional to ventilation rate throughout most of the study, thus, the data followed the trend indicated by the simple Equation 2.1, presented above, which is based on the assumption of a constant pollutant source strength.

Offermann et al. (1982, 1982b) installed MVHX systems, including both ducted and unducted systems in nine occupied residences located in Rochester, New York. Air exchange rates and indoor pollutant concentrations were monitored in each residence for a ten-day period without operation of the MVHX systems followed by a ten-day period with operation of the systems. The average air exchange rate increased from 0.35 to 0.63 ach with operation of the MVHX systems. Average indoor radon concentrations decreased from 1.0 to 0.5 pCi/l which coincides fairly well with the average decrease predicted using Equation 2.1. The average measured reduction in formaldehyde concentration was 30% in houses with sensible heat exchangers which compares well with a predicted reduction of 27% if a constant formaldehyde source strength and a reaction constant (k) equal to 0.4 h^{-1} are assumed. (As mentioned earlier, however, a simple model of this type is not appropriate for formaldehyde.) On a house-by-house basis, however, the measured decreases in radon and formaldehyde concentrations often differed substantially from the predicted decreases.

In two of the houses monitored during this study in Rochester, enthalpy exchangers were installed, and a negligible reduction in formaldehyde concentration was observed. A possible explanation given was that formaldehyde may have been transferred between airstreams in this type of exchanger, however, the data was insufficient for a conclusion. A negligible reduction in formaldehyde concentration was also observed in one of the residences with a sensible exchanger. Data from this study, except that described below, are summarized in Figure 2.7.

Many of the houses in the Rochester study had insignificant indoor sources of combustion products. In the houses where the indoor concentrations of combustion products were significantly greater than

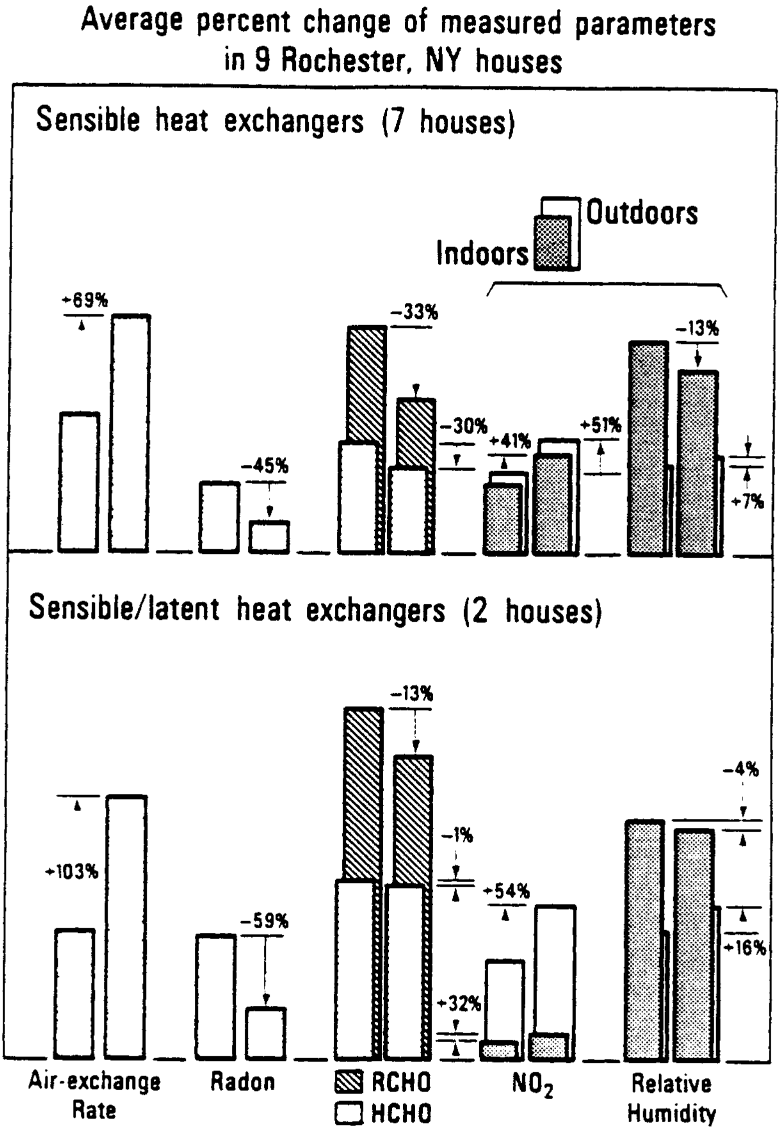

Figure 2.7 Changes in air exchange rate, indoor pollutant concentrations, and humidity following ventilation with sensible-type air-to-air heat exchangers in 7 houses and sensible-latent (i.e., enthalpy) -type air-to-air heat exchangers in 2 houses. The left bar of each pair is for the period without operation of the heat exchangers and the right bar is for the period with operation. (Source: Offermann, et al., 1982b).

outdoor concentrations, substantial reductions in both gaseous products and particulates were observed with operation of the MVHX systems.

Energy Performance: The capability of various residential air-to-air heat exchangers to transfer (recover) heat has been investigated in both laboratory and field studies. The most commonly quoted performance parameter is the temperature efficiency which is defined by the equation

$$\varepsilon = \frac{T_{c,o} - T_{c,i}}{T_{h,i} - T_{c,i}} \qquad (2.7)$$

where: T is an airstream temperature, and subscripts c, h, i, and o refer to the cold airstream, warm airstream, into the heat exchanger, and out of the heat exchanger, respectively. The temperature efficiency is a measure of the heat exchanger's ability to preheat or precool the incoming air and it is desirable for it to be as high as possible. Unfortunately, different investigators report values of temperature efficiency measured under different operating conditions so results are not always directly comparable.

In laboratory studies, Fisk et al. (1980, 1981, 1981b) have measured the temperature efficiency of several air-to-air heat exchangers and found it to range from 45 to 85%. Gezelius (1981) summarized results from laboratory measurements of the temperature efficiency of a large number of residential heat exchangers available in Sweden and indicated a range from 31 to 96%. Shoukri (1979) measured the temperature efficiency of a residential heat exchanger with a rotating heat-wheel core and found it to range from 80 to 90% under most operating conditions. Besant et al. (1977) determined in laboratory studies that the temperature efficiency of an easily constructed exchanger ranged from 50 to 100% over a wide range of air flow rates. Using a unique test method, Persily (1982) determined that the temperature efficiency of a wall-mounted residential heat exchanger was typically 72% at night and often higher during the day due to solar irradiation of the exchanger's case. Persily also concluded that, due to imperfect mixing of the indoor air and other factors, the actual heat recovery efficiency of the exchanger was only "55 to 60% of the theoretical maximum".

Data on the temperature efficiency of exchangers operating in actual residences are more limited. In the field study by Offermann et al. (1982, 1982b), the average temperature efficiency ranged from 49 to 89% with an average for all houses of 72%. To provide a number that better represented the impact of the exchangers on the heating load of the residences, a correction for flow imbalances was employed that brought the average temperature efficiency down to 65%. An additional correction for heat released by the fan motors further reduced the average value to 56%. The temperature efficiency of several different exchangers installed and operated under a variety of conditions was measured by Hamlin and Besant (1982) and ranged from 63 to 111% without any corrections. They indicated that some of the measured results were essentially meaningless due to leakage of air between airstreams and other factors.

The energy performance of residential MVHX systems depends highly on fan power requirements as well as temperature efficiency. In laboratory measurements by Fisk et al. (1980, 1981b) specific fan power ranged from 1.4 to 7.6 W/l/s of air flow; a variation of approximately a factor of five. Based on product literature, many manufacturers have recognized the importance of minimizing fan power and are designing or modifying their equipment accordingly.

The impact of MVHX system operation on the energy requirements of an actual residence depends on average temperature efficiency, air flow rates, fan power, and other factors. Using a computer model, Fisk and Turiel (1983) compared energy requirements in typical houses (with all ventilation provided by infiltration) to energy requirements in tightly constructed houses with ducted MVHX systems. The major assumptions and results are summarized in Table 2.2 and Table 2.3, respectively. The predicted reduction in ventilation heat load (i.e., the reduction in amount of energy required to heat ventilation air) ranged from 5.3 to 18 G J. (50 to 171 therms) and depended highly on the climate and the heat exchanger's performance. Predicted energy consumption by the heat exchanger's fan system ranged from 2.2 to 3.2

Table 2.2 Major assumptions for energy and economic analysis of mechanical ventilation with ducted air-to-air heat exchangers in new residences. (Source: Fisk and Turiel, 1983).

	Case A typical house	Case A tight house	Case B typical house	Case B tight house
Infiltration rate (ach)	0.60	0.20	0.75	0.20
Ventilation through HX* (ach)	--	0.40	--	0.55
House volume (m^3 ($ft^3 \times 10^{-3}$))	339.6 (12)	339.6 (12)	339.6 (12)	339.6 (12)
Balance point (°C(°F))	12.8 (55)	12.8 (55)	12.8 (55)	12.8 (55)
Indoor temperature (°C (°F))	20 (68)	20 (68)	20 (68)	20 (68)
Apparent HX effectiveness	--	0.65	--	0.75
Total HX fan power (W)	--	200	--	140
Percent of total fan power that is delivered to residence in the form of heat (%)	--	50	--	75
Outdoor temperature at onset of freezing in HX (°C (°F))	--	-6.7 (20)	--	-6.7 (20)
Defrost time fraction (%)	--	20	--	20
Initial capital cost ($)	--	1300	--	1150
Yearly maintenance cost ($)	--	30	--	10

*HX = Heat Exchanger

Table 2.3 Results of energy analysis of mechanical ventilation with ducted air-to-air heat exchangers in new residences. (Source: Fisk and Turiel, 1983).

City	Case	[*]Vent. heat load in typical house GJ	Therms	[†]Vent. heat load in tight house with heat exchanger GJ	Therms	[‡]Vent. heat load reduction GJ	Therms	[§]Fan energy consumption GJ	Therms
Minneapolis	A	26.3	249	14.3	136	11.9	113	3.6	34
	B	32.8	311	14.8	140	18.0	171	2.5	24
Chicago	A	21.9	208	11.4	108	10.6	100	3.6	34
	B	27.5	261	11.5	109	16.0	152	2.5	24
Washington D.C.	A	15.2	144	7.5	71	7.8	74	3.1	30
	B	19.1	181	7.4	70	11.7	111	2.2	21
Atlanta	A	9.9	94	4.6	44	5.3	50	3.1	30
	B	12.3	117	4.5	43	7.8	74	2.2	21

[*] Heat load imposed on heating system during the specified heating season due to uncontrolled ventilation.

[†] Heat load imposed on heating system due to uncontrolled ventilation, heat exchanger inefficiency, and heat exchanger defrost cycles minus the amount of energy that is released by fan system and delivered to the residence.

[‡] Ventilation heat load in typical house minus that in tight house.

[§] Total electrical energy consumption by both fans.

G J. (21 to 34 therms) which is significant compared to the amount of heat recovery. Using essentially the same computer model, Turiel et al. (1983) estimated the energy savings when electrically heated houses in the Pacific Northwest are weatherized to reduce infiltration and unducted MVHX systems are installed and operated. The major assumptions and results are summarized in Table 2.4 and Table 2.5, respectively. Net electrical energy savings for the heating season of 1.7 to 2.7 G J. (16.5 to 25.9 therms) were predicted. The computer model was used again with data from the field study by Offermann et al. (1982b) as inputs (Table 2.6). The predicted reduction in ventilation heat load ranged from 8.2 to 16.0 G J. (78 to 152 therms) and predicted fan energy consumption ranged from 875 to 1283 kwh (2.99 to 4.38 therms).

A preferred method to determine energy savings when MVHX systems are used is to actually measure energy consumed for heating in houses with and without these systems. It is difficult to actually determine energy savings by this method, however, unless a large number of houses are monitored. Svensson (1982) measured energy consumption in 15 Swedish homes with MVHX systems and compared it to the measured consumption in 15 control houses that had mechanical exhaust ventilation without heat recovery. The energy savings was approximately 1000 kwh/yr (0.155 kwh/degree celsius day) in single-story houses and 1250 kwh/yr (0.17 kwh/degree celsius day) in larger and more air-tight two-story houses. These savings were less than might be expected because

Table 2.4 Major assumptions for energy and economic analysis of mechanical ventilation with unducted air-to-air heat exchangers in weatherized, electrically-heated residences. (Source: Turiel et al., 1983).

	untightened house	tightened house
1 Infiltration rate (ach)	0.65	0.50
2 Ventilation through HX[†] (ach)	---	0.15
3 House Volume (m^3 ($ft^3 \times 10^{-3}$))	339.6 (12)	339.6 (12)
4 Balance Point (°C (°F))	12.8 (55)	12.8 (55)
5 Indoor temperature (°C (°F))	20 (68)	20 (68)
6 Apparent HX effectiveness	---	0.60
7 Total HX fan power (w)	---	42
8 Percent of total fan power that is delivered to residence in the form of heat (%)	---	0.50
9 Outdoor temperature at onset of freezing in HX (°C (°F))	---	-9.4 (15)
10 Initial capital cost ($)	---	800[‡]
11 Yearly maintenance Cost ($)	---	10

[†] HX = heat exchanger

[‡] In Portland, the initial capital cost is $700.

Table 2.5 Results of energy analysis of mechanical ventilation with unducted air-to-air heat exchangers in weatherized, electrically-heated residences that are located in four Pacific Northwest cities. (Source: Turiel et al., 1983).

	[†]Ventilation heat load in untightened house		[‡]Ventilation heat load in tightened house with heat exchanger		[§]Ventilation heat load reduction		Fan energy consumption		Preheat System Energy consumption		[¶]Annual energy savings	
	GJ	Therms	GJ	therms	GJ	therms	GJ	therms	GJ	therms	GJ	therms
Portland	15.2	144.3	12.8	121.5	2.4	22.8	.66	6.3	na	na	1.7	16.5
Lewiston	21.1	200.5	17.7	168.3	3.4	32.2	.77	7.3	.2	1.6	2.5	23.3
Spokane	22.7	214.9	19.1	181.3	3.6	33.6	.77	7.3	.1	0.8	2.7	25.6
Helena	24.4	231.2	20.3	192.6	4.1	38.6	.86	8.2	.5	4.5	2.7	25.9

[†]The heat load imposed on the heating system during the specified heating season due to uncontrolled ventilation. [‡]The ventilation heat load in the tightened house with heat exchanger is equal to the heat load imposed on heating system due to uncontrolled ventilation, and heat exchanger inefficiency, minus the amount of energy that is released by fan system and delivered to the residence. [§]The ventilation heat load reduction is equal to the ventilation heat load in the tightened house minus that in untightened house. [¶]The annual energy savings equals the ventilation heat load reduction minus fan energy and preheat energy.

Table 2.6 Results of energy and economic analysis of increasing the ventilation with air-to-air heat exchangers based on data from a field study in Rochester, New York. (Source: Offermann, et al., 1982 b).

House ID #	Heating[a] System	Capital[b] Cost ($)	Heat[c] Exchanger Effectiveness (%)	Increase[d] in Ventilation (ft^3/min)	Ventilation[e] Heat Load Reduction (therms)	Fan Energy[f] Consumption (kwh)	Net Present[g] Benefit ($)	Benefit-[g] Cost Ratio	Discounted Payback Period (years)	Increase in[h] Ventilation Heat Load (therms)	Net Present[i] Value of Costs ($)
1	E	1441	45	69	78	875	-779	0.67	>30	65	3490
6	E	1087	67	66	116	1050	243	1.11	16	21	2365
10	E	1040	68	65	116	1050	286	1.14	15	19	2273
33[j]	HP	1753	57	117	152	875	-851	0.68	>30	91	3562
	G	1753	57	117	152	875	-82	0.97	21	91	4023
45	G	2074	44	92	104	1283	-1511	0.54	>30	87	4567
49	G	1170	67	55	101	1050	-494	0.78	>30	14	2245
52	E	1051	48	95	103	560	345	1.20	14	94	3461
60[j]	HP	1343	57	55	89	1050	-1313	0.45	>30	25	2487
	G	1343	57	55	89	1050	-860	0.64	>30	25	2615

[a] E = electric heat with efficiency of 100%, HP = heat pump with C.O.P. = 1.7, G = natural gas heat with furnace efficiency of 70%

[b] Cost for purchase and installation of heat exchanger plus $200 estimated cost for house tightening features

[c] From Table 13.

[d] Increase in air-exchange rate multiplied by house volume (see Table 7).

[e] House where increase in ventilation is supplied by heat exchanger is compared to house with increase in infiltration.

[f] Electrical energy consumption by heat exchanger's fan system.

[g] 20 year life for heat exchanger assumed.

[h] House with additional ventilation supplied by heat exchanger compared to house without additional ventilation.

[i] Total cost for purchase, installation, maintenance, and operation of heat exchanger, 20 year life assumed.

[j] Home heating system is electric heat pump with natural gas assist, results are given assuming heat pump supplies all heat and assuming all heat supplied by natural gas furnace.

the average air exchange rate was lower in the control houses than in the houses with MVHX systems. Many residents of the control houses had reduced the ventilation rate in their house by closing slots through which air entered the residence. The reported purpose of these actions was to reduce cold drafts. Svensson calculated that energy savings would have been approximately 2400 and 3050 kwh/yr for single- and two-story houses respectively, if both experimental and control houses had an air exchange rate of 0.5 ach. Based on measurements of air temperature, he reported an average temperature efficiency of 69%. After correcting for fluctuations in flow rate and the requirement to periodically defrost the exchanger, he arrived at an average heat recovery efficiency ("energy obtained/available energy") of 51%.

Economics: To evaluate the economics of MVHX system utilization, one must compare two different alternatives. It is difficult to quantify economically the benefits of reduced pollutant concentrations due to increases in ventilation. Therefore, economic evaluations of MVHX systems have been based on a comparison of costs when these systems are employed to costs for an alternate ventilation technique with the same average ventilation rate.

Economic calculations were performed by Fisk and Turiel (1983) using a computer model along with the previously described calculations of energy savings. Construction of low-infiltration residences and utilization of ducted MVHX systems was compared to the more typical practice of constructing less air-tight houses and relying on natural infiltration for ventilation. Depending on the climate, the cost of the heating fuel, and the MVHX system performance, the net present benefit for MVHX system utilization ranged from $ -1358 to $ +1963 and the discounted payback period ranged from 5 to over 30 years. The calculations, which were based on the point of view of the homeowner, indicated that MVHX systems were economical only in cold climates and with high performance MVHX systems. The systems were much more likely to be economical when the home was heated with an expensive heating fuel (e.g., electricity). The major assumptions and results are summarized in Table 2.2 and Table 2.7.

In the analysis by Turiel et al. (1983) of using unducted MVHX systems in weatherized, electrically heated houses located in the U.S. Pacific Northwest, the net present benefit ranged from $ -534 to $ -689 and the discounted payback period was always greater than 30 years (Table 2.4 and Table 2.8). Thus, the option of not reducing infiltration during weatherization appeared much more economical than reducing infiltration and using a MVHX system. The low cost of electricity in the cities considered (2.3 to 4.0 cents/kwh in June of 1982) was a major reason for the negative economic results. The cost of conserved energy was also calculated and it ranged from 7.1 to 9.7 cents/kwh. The authors concluded that in other regions of the U.S. with colder climates and higher electricity prices, the MVHX systems

Table 2.7 Results of economic analysis of mechanical ventilation with ducted air-to-air heat exchangers in new residences. (Source: Fisk and Turiel, 1983).

City (Heating °C Days*)	Heating fuel	Net Present† Benefit $		Benefit-Cost† Ratio		Discounted Payback Period (years)	
		Case A	Case B	Case A	Case B	Case A	Case B
Minneapolis (4657)	Gas	-844	+670	0.65	1.40	>30	12
	Oil	-105	+1790	0.96	2.07	22	7
	Electricity	+9	+1963	1.00	2.17	20	6
Chicago (3688)	Gas	-1358	+83	0.49	1.05	>30	19
	Oil	-520	+1348	0.80	1.73	>30	9
	Electricity	+194	+2425	1.07	2.31	17	5
Washington, D.C. (2347)	Gas	-1099	+246	0.54	1.15	>30	16
	Oil	-767	+743	0.68	1.45	>30	11
	Electricity	-620	+963	0.74	1.58	>30	9
Atlanta (1645)	Gas	-1352	-247	0.40	0.84	>30	29
	Electricity	-1284	-147	0.43	0.91	>30	25

*yearly total degree days, 18.3 °C base.

†based on a 20 year life for heat exchanger and zero salvage value.

Table 2.8 Results of economic analysis of mechanical ventilation with unducted air-to-air heat exchangers in weatherized, electrically-heated residences that are located in four Pacific Northwest cities. (Source Turiel, et al., 1983).

City (Heating °C Days)[†]	Net present benefit[‡]	Discounted benefit/cost ratio[‡]	Discounted payback period (years)	break-even[§] fuel price, ¢/kWh	Cost of conserved energy, ¢/kWh[¶]	Cost of Heat exchanger utilization ($)[‖]
Portland (2644)	-534	.45	>30	10.7	9.7	837
Lewiston (3018)	-689	.34	>30	8.8	7.9	893
Spokane (3779)	-676	.35	>30	7.9	7.1	884
Helena (4532)	-621	.45	>30	8.4	7.1	1035

[†] The yearly total degree-days are computed for an, 18.3°C base temperature. [‡] The benefit to cost ratio and the net present benefit are based on a 20-year life for the heat exchanger and zero salvage value. [§] The break-even fuel price is the electricity price which sets the NPB = 0 in 20 years.
[¶] The cost of conserved energy is the amortized capital cost divided by the net energy saved. [‖] The cost of heat exchanger utilization is the present value of heat exchanger purchase price, installation, maintenance and operation.

would have been economical.

When the computer model was again used to evaluate the economics of using MVHX systems with data (e.g., performance and costs) from the previously discussed Rochester, New York study (Offermann et al., 1982b) as inputs, the results were similar. The model predicted that increasing the ventilation rate with MVHX systems, in contrast to increasing ventilation by some other means without heat recovery, was marginally economical in three of four electrically heated houses and not economical in four other houses (Table 2.6). The estimated total cost for purchase, installation, and operation of the MVHX systems for a 20 year period ranged from $2200 to $3600.

Economic calculations were also performed by Svensson (1982) based on data from his field study that compared MVHX use in Swedish houses to mechanical exhaust ventilation without heat recovery. He concluded that with an energy cost of 0.20 Skr/kwh ($\sim$ $0.027/kwh) and an initial cost for the MVHX system of 8000 Skr ($\sim$ $1070), the annual savings amounted to only five to seven percent of the initial investment. Thus, the systems did not appear economical but he noted that they did improve the comfort of the occupants because there were fewer drafts in houses with MVHX systems compared to houses with exhaust ventilation.

<u>Ventilation</u> <u>Efficiency</u>: Other indicators of the performance of

MVHX systems include their ventilation efficiencies. This aspect of MVHX system performance will be discussed later in the general section on ventilation efficiency.

Performance Problems: Before concluding the discussion of MVHX systems, several performance problems must be noted. Perhaps the most important performance problem is the formation of ice or frost in the cores of the heat exchangers. The accumulation of ice or frost, which occurs in the passages for the air exiting the structure, can severely reduce performance. The American Society of Heating, Refrigerating, and Air Conditioning Engineers (1983) indicates that freezing is initiated in counterflow heat exchangers with balanced airstream flow rates and with the temperature and relative humidity of the exhaust airstream typical of indoor spaces (i.e., 21°C and 30 to 60%), when the outdoor temperature is in the range of -8 to -18°C. In an experimental study by Sauer et al. (1981) the onset of freezing in a counterflow exchanger operating with balanced flow rates and the same range of humidities was -11 to -22°C. In a study by Ruth et al. (1975), the onset of freezing of an enthalpy exchanger with a rotating wheel core was -12 to -25°C for the same range of inlet humidities. Freezing was observed with higher outdoor temperatures (approximately -8°C) than in the above studies, however, in the field study by Offermann et al. (1982 and 1982b). In laboratory studies of freezing in three different residential heat exchangers by Fisk et al. (1983b), the onset of freezing ranged from -3 to -11°C. The report also includes data on the impact of freezing on performance and on the use of a defrost freeze protection technique. Many manufacturers are now marketing residential heat exchangers with freeze protection systems of various designs. Even with these freeze protection systems, however, reductions in performance occur.

Other performance problems noted throughout the literature include imbalances in flow rate, leakage of air between airstreams, fouling of heat exchange surfaces by grease, dust, or lint, obstruction of air intake and exhaust grills by snow or leaves, and transfer of contaminants between airstreams (mentioned earlier). At this time the general reliability of MVHX systems and maintenance requirements are not well known.

3. Mechanical Exhaust Ventilation

Local Exhaust: The purpose of local exhaust ventilation is to minimize the transport of pollutants from their source to the occupied space. This process is accomplished by exhausting air from locations close to a concentrated pollutant source. If the system is effective, the concentration of pollutant in the exhausted air will be significantly greater than the average concentration in the occupied space. When pollutant sources are concentrated, local exhaust ventilation can maintain low indoor pollutant concentrations much more efficiently (i.e., with less exhausted air) than general dilution ventilation of the entire structure. In many instances, the pollutant is not emitted continuously and the exhaust ventilation system is operated only when needed.

Local exhaust ventilation is common in industry where exhaust hoods are often employed (see, for example, American Society of Heating, Refrigerating, and Air Conditioning Engineers, 1980). Restaurants often use exhaust hoods above cooking surfaces. Typical examples of local exhaust ventilation in residences are range hoods that exhaust air to the outdoors and bathroom fans. In commercial buildings, air from certain rooms (e.g., bathrooms, blueprint rooms) is often exhausted directly to the outside (i.e., it is not mixed with the general return air that may be partially recirculated).

The use of local exhaust ventilation is related to the concept of maximizing ventilation efficiency (discussed later). In addition, local exhaust ventilation can be considered a source control measure because it effectively reduces pollutant source strengths. Literature applicable to the use of local exhaust ventilation in residential and commercial buildings is discussed below.

In residential buildings, range hoods that exhaust air to the outside are one of the most common local exhaust ventilation systems. Several investigations of range hood performance have been completed. In a study by Macriss and Elkins (1977), the increase in the kitchen

NO_x concentration due to operation of a gas stove was reduced by 30% when a range hood with an air flow rate of 93 l/s was operated. However, in a later study by Traynor et al. (1982), operation of a range hood with air flow rates from 42 to 117 l/s caused 60 to 87% reductions in the amount of CO, CO_2 and NO_x that entered the occupied space of a research house. Range hood performance was also studied by Revzan (1984) who used a heated and unheated tracer gas to simulate emission of pollutants from a range. For the flow rates investigated (10 to 60 l/s), Revzan found a linear relationship between range hood efficiency and flow rate, with efficiency increasing from 15 to 77%. He defined efficiency as $(C_o - C_m)/C_o$ where: C_o equals the concentration of tracer that would have occurred in the test space if the indoor air had been perfectly mixed throughout the test, and C_m equals the measured average concentration of tracer in the test space after the test.

It is evident from these studies that range hoods can be highly effective in reducing the transport of pollutants from a gas range to the occupied space. Because these hoods only need to be operated when the range is used, they constitute an energy efficient pollution control technique. Range hoods are only effective, however, if they are used.

There appears to be few available data on the effectiveness of local exhaust ventilation from specific rooms in residences or commercial buildings. One study was performed by Revzan (1984) who injected tracer gas in the center of a room, exhausted air from the same room, and monitored tracer gas concentrations in the exhaust and at several points in both the room containing the tracer source and in an adjoining room. (The door between the rooms remained open.) With exhaust air flow rates up to 1.7 ach (based on the total volume of both rooms), the concentration of tracer gas in the exhaust and in the two rooms were roughly equal. With exhaust flow rates of 2.1 and 2.4 ach, the concentrations in the two rooms were still roughly equal but the concentrations in the exhaust duct were two to four times the average indoor concentration. The results provide evidence that local exhaust ventilation can reduce the transport of tracer gas to an adjoining room even when the door between the rooms remains open; however, significant air flow rates were required in these experiments. We expect that much lower air flow rates would be required if the door between the rooms remained closed. The practicality of exhausting air in residences from rooms containing actual contaminant sources has not been established.

Dilution Exhaust: The goal of dilution exhaust ventilation is to ventilate the entire structure or at least major portions of the structure. In Sweden and other Scandinavian countries, this method of ventilation is commonly used in tightly constructed residences. A fan exhausts air to the outdoors and outdoor air enters the residence through openings in the building envelope. Slots are often provided through the walls of each room and outdoor air enters the residence through these slots. The slots may be designed so that they can be manually closed (either partially or completely). With some slot systems, the size of the opening and, thus, the ventilation rate is automatically reduced in response to a reduction in outdoor temperature.

In comparison to ventilation by natural infiltration, there are several significant advantages to mechanical exhaust ventilation including the following:

1. The ventilation rate can be adjusted to equal the desired value and increased or decreased when desired.

2. The ventilation rate can be highly stable in an air-tight, exhaust-ventilated structure (i.e., without large slots). This stability is possible because the rate of air leakage through an opening (e.g., a crack) in the building envelope is proportional to the square root of the pressure difference across the opening. Mechanical exhaust ventilation will depressurize the interior of the house relative to outdoors and, if significant depressurization occurs, the impact of weather-induced pressures on ventilation rate will be small. Because the ventilation rate is more stable with exhaust ventilation, periods of time with very low ventila-

tion (and correspondingly much higher than average pollutant concentrations) are avoided. Large increases in ventilation rate during periods of cold weather (with associated high energy demands) are also avoided if the ventilation rate is constant. Another advantage results from the fact that indoor pollutant concentration is roughly proportional to the inverse of the air exchange rate. Because of this fact, a lower time-averaged pollutant concentration occurs when the ventilation rate is stable compared to the case when it is highly variable assuming that the mean ventilation rate is the same for both cases.

3. *When mechanical exhaust ventilation is employed, air can be* drawn from locations near pollutant sources which is a more efficient method of controlling pollutant concentration.

4. A final advantage of mechanical exhaust ventilation is that energy can be recovered from the exhausted airstream -- this topic will be discussed in more detail later in this section.

Along with the advantages, there are some important disadvantages to mechanical exhaust ventilation. A capital cost is associated with the mechanical system and with building a more air-tight residence. Electricity is required to operate the exhaust fan. Also, if large amounts of cold air is drawn into the residence at any single location, cold drafts may result as noted in the study by Svensson (1982). Another significant disadvantage is that the radon source strength may be increased by depressurization of the structure when exhaust ventilation is employed as discussed in a previous section.

When mechanical exhaust ventilation is utilized, energy can be recovered from the exhausted airstream. This energy recovery is generally accomplished by using a heat pump that transfers heat from the exhaust airstream to the domestic hot water supply. If additional energy is available, it can be transferred to water used in a hot water heating system or to the indoor air. Suitable heat pumps are available in Europe from a number of European manufacturers and it may be possible to utilize heat pump systems that are available in the U.S. for hot water heating. This technique of ventilation with heat recovery is being used in Sweden and other European countries but estimates of the number of systems installed are not available and almost no relevant literature is available that is published in English.

Heat recovery with a heat pump can be compared qualitatively to heat recovery with an air-to-air heat exchanger. The heat pump system will require more electrical energy than the heat exchanger. In a tightly-constructed structure, the heat pump can recover energy from virtually all the ventilation air since almost all of this air will exit the structure through the exhaust fan. In contrast, a significant amount of air will exfiltrate through the building envelope of a structure with an air-to-air heat exchanger (unless the structure is constructed very tightly) and the heat exchanger cannot recover energy from this exfiltrating air. The heat pump remains useable year round as a source of heat for the domestic hot water, although it would be thermodynamically more efficient to use the warmer outdoor air as the heat source during periods of air conditioning. The air-to-air heat exchanger can provide useful pre-cooling of ventilation air during the summer but the amount of energy saved is generally much less than in the winter when the difference between indoor and outdoor temperatures is greater. The heat pump system is more complex than a heat exchanger, thus, one would expect a higher initial cost for the heat pump although mass-produced heat pumps in this size range should not be prohibitively expensive. Finally, the amount and cost of ductwork required in a house with a heat pump should be significantly less than in a house with a ducted air-to-air heat exchanger.

The strategy of mechanical exhaust ventilation with and without heat recovery and other ventilation strategies were treated in a paper by Sherman and Grimsrud (1982). This paper provides theoretical predictions of ventilation rate and ventilation heat loads in structures with three different leakage areas (leakage area is a measure of building tightness) for a range of outdoor temperatures and windspeeds. In addition to mechanical exhaust ventilation, the authors provide predicted data for ventilation by natural infiltration, a ventilation stack, and for balanced mechanical ventilation with an air-to-air heat exchanger. Figure 2.8 provides a comparison of the ventilation rate and additional ventilation heat load as a function of weather for a fairly air-tight structure (200 cm^2 leakage area) ventilated by the various methods described above. (The additional ventilation heat load is defined as the total ventilation heat load minus the ventilation heat load that would occur in the structure with the smaller amount of ventilation that occurs only by natural infiltration.) The ventilation rate is least stable when a ventilation stack is employed and is highly stable when a mechanical exhaust system is used. Mechanical exhaust ventilation with heat recovery by a heat pump (exhaust air cooled by 20°C) causes the smallest additional ventilation heating load followed in order by: the air-to-air heat exchanger (70% temperature efficiency), mechanical exhaust ventilation with heat recovery in a heat exchanger (30% temperature efficiency) exhaust ventilation without heat recovery, and the ventilation stack. Figure 2.9 provides the same comparisons but for a less air-tight structure (500 cm^2 leakage area). In this case, the air-to-air heat exchanger becomes somewhat less attractive compared to the exhaust ventilation systems. If the authors had based their results on an integration of performance over the range of weather during a heating season, instead of comparing the systems under the same conditions of weather, the performance of the exhaust ventilation systems should be improved relative to the performance of the other systems.

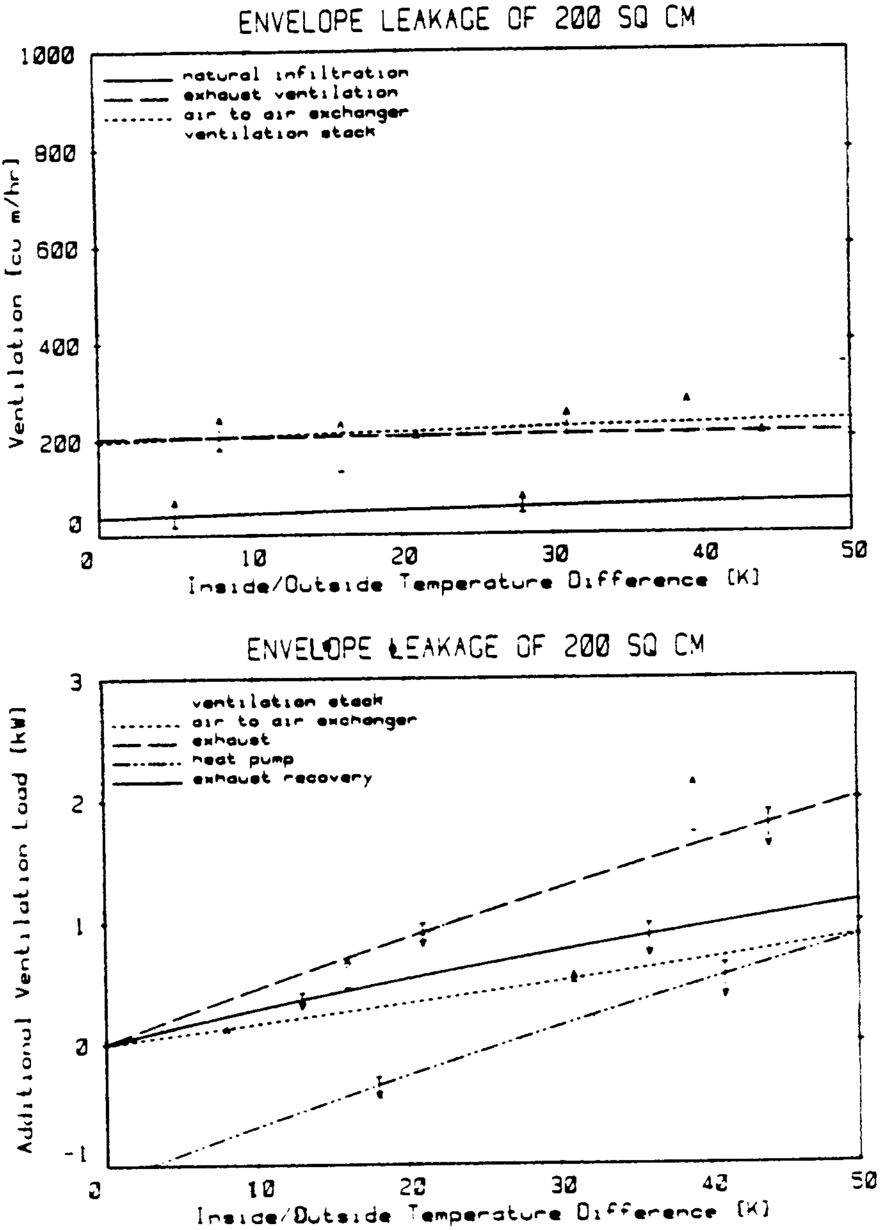

Figure 2.8 Comparison of theoretically predicated ventilation rate (top) and additional ventilation heat load (lower) for a house with 200 cm^2 leakage area and various ventilation systems. The lines are for an assumed wind speed of 4 m/s. The pair of arrows that cross each curve show the effect of wind speed; the base of the arrow was computed for zero wind speed and the tip for a wind speed of 8 m/s. (Source: Sherman and Grimsrud, 1982).

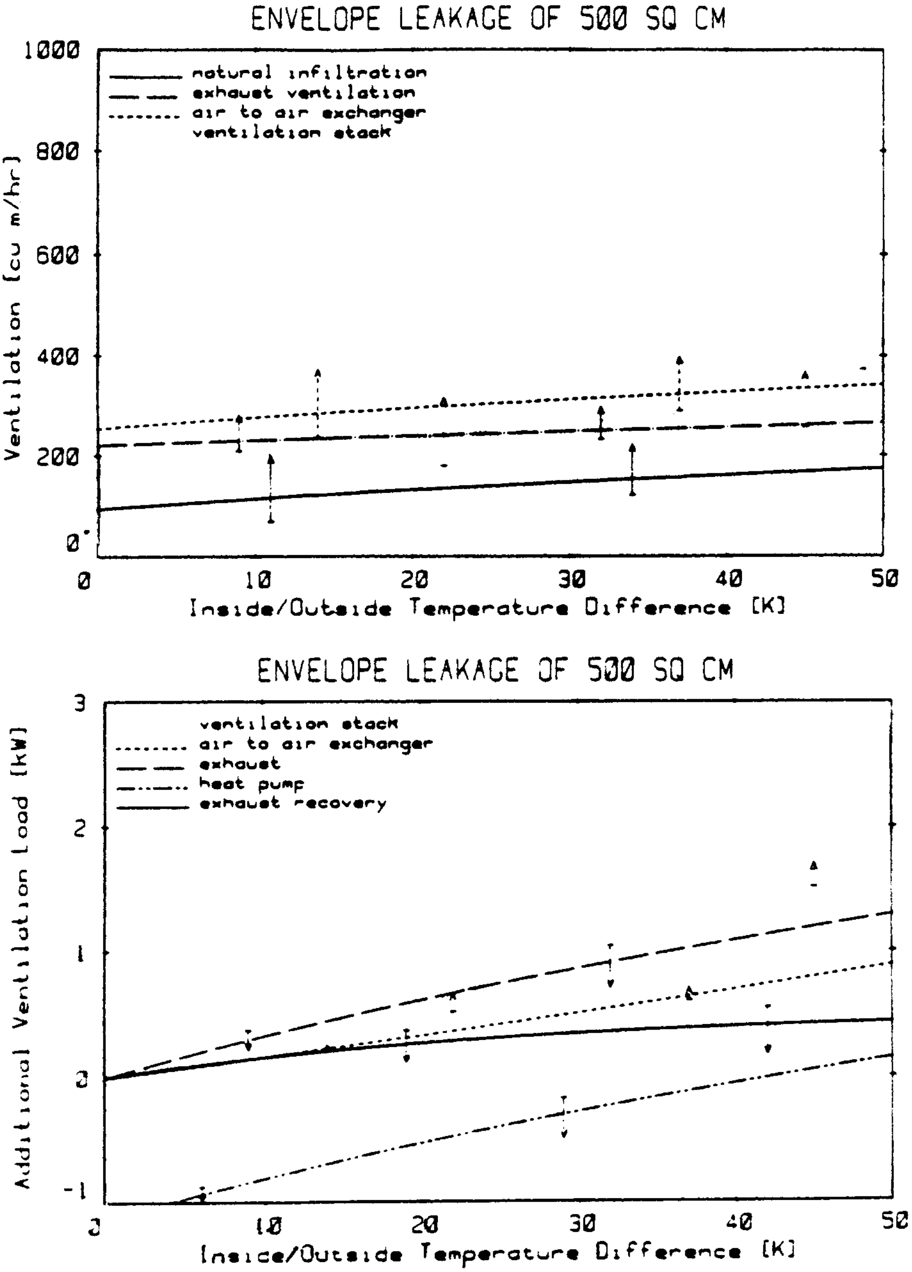

Figure 2.9 Comparison of theoretically predicted ventilation rate (top) and additional ventilation heat load (lower) for a house with 500 cm^2 leakage area and various ventilation systems. The lines are for an assumed wind speed of 4 m/s. The pair of arrows that cross each curve show the effect of wind speed; the base of the arrow was computed for zero wind speed and the tip for a wind speed of 8 m/s. (Source: Sherman and Grimsrud, 1982).

The paper does, however, illustrate some of the advantages of mechanical exhaust ventilation even without a seasonal integration of performance.

E. Concentration-Based Control of Mechanical Ventilation Systems

The ventilation systems of large buildings are often designed so that a variable volume of outdoor air is supplied to the indoor environment. Typically, the system is designed to supply a minimum amount of outdoor air during all periods of occupancy -- this minimum amount is often the amount of outdoor air needed to meet a ventilation standard, e.g., 2.5 l/s-person at design (i.e., maximum) occupancy. Control of the amount of additional outside air supplied is generally based on thermal loads and outdoor air temperature.

Ventilation standards for commercial and institutional buildings generally specify minimum outdoor air requirements on a per-person basis although other alternatives are sometimes permissible. The standards are based on the assumption that the primary sources of indoor-generated pollutants in these densely occupied buildings are CO_2 exhalation, odors generated from people, and tobacco smoking -- each a source that varies in proportion to occupancy.

If occupants are the most important sources of indoor pollutants, then, the minimum quantity of outdoor air supplied could be controlled in response to the actual number of occupants, or in response to the indoor concentration of O_2, CO_2, or tobacco smoke. Such a control strategy would be expected to save energy by eliminating or minimizing excessive ventilation especially during periods of low occupancy. The largest energy savings would occur in buildings with highly variable occupancy.

Control of the outdoor air supply based on indoor CO_2 concentrations has been the subject of several investigations. Turiel et al. (1979) discussed this control strategy and concluded that substantial energy savings could result from its implementation. For example, the yearly energy savings if this strategy was used in a typically sized school building in the Northwest or Northcentral regions of the U.S., was estimated to vary from 0.273 to 0.358 million kilowatt hours. The payback period was estimated to be three to four years based on 1978 prices.

Experiments were later performed in actual buildings by incorporating CO_2-based control systems for the outdoor air supply. Sodergren (1982) monitored a two-person office room and estimated 10 to 40% energy savings, depending on the CO_2 set point, compared to a constant supply of outdoor air. Janssen et al. (1982) conducted experiments in the music wing of a school building and determined that energy savings were about 20% compared to an improved, temperature-based method of controlling the existing ventilation system.

Another example of controlling mechanical ventilation systems based on an indoor pollutant concentration is the use of a humidistat to control the fan operation or fan speed of residential air-to-air heat exchangers. This practice is common in cold regions of Canada and the U.S. The basis for this control strategy is the assumption that ventilation sufficient for humidity control will also be sufficient to control concentrations of other pollutants. We are unaware of any estimates of the energy savings that result from water vapor based control.

The major drawback to using CO_2 or water vapor concentration as the basis for controlling the rate of outdoor air supply is that these strategies will not always ensure acceptable concentrations of other pollutants. It is known that the source strengths for indoor-generated pollutants can vary widely, thus, ventilation requirements may often be dictated by the concentration of some pollutant other than CO_2 or water vapor. However, in situations where it can be concluded that acceptable concentrations of other pollutants will be maintained, these techniques appear highly attractive. Control of ventilation systems based on the indoor concentrations of other pollutants (e.g., Rn, HCHO, CO, NO_2) may also become attractive in the future if inexpensive sensors for these pollutants become available.

F. Ventilation Efficiency

The objectives of ventilation are to remove or supply heat and moisture and to remove pollutants from the indoor air. The performance of a ventilation system depends on the patterns of air flow throughout the ventilated space and on the locations and emission characteristics of the pollutant sources. By utilizing ventilation air more effectively, indoor air quality can be improved and/or the energy required for ventilation can be reduced.

Several "ventilation efficiency parameters" have been defined that are useful for characterizing various aspects of the performance of ventilation systems. If the locations and emission characteristics of pollutant sources are not known, the relevant ventilation efficiencies are ones that describe the flow and distribution of air. Other ventilation efficiencies have been defined that describe the capability of a ventilation system in removing pollutants and these efficiencies will depend on characteristics of the pollutant source. Ventilation efficiency is often studied by using tracer gases to label the indoor air or the supplied ventilation air, or by using tracers to simulate the emission of pollutants within the ventilated space. A large amount of literature is available on the subject of ventilation efficiency. Only some of the major topics and results are reviewed here -- more detailed information is available from the original literature. (See, for example: Boman, 1982; Freeman et al., 1982;

Malmstrom and Ahlgren, 1982; Offermann et al., 1982c, 1983; Revzan, 1984; Sandberg, 1981, 1982; Sandberg et al. 1982b; Sandberg and Sjoberg, 1983; Skaret and Mathisen 1982, 1983; Skaret, 1983b.)

1. Short-Circuiting versus Displacement of Air

The pattern of air flow within a ventilated space can range from one extreme with a large amount of short-circuiting of air from supply to exhaust to the other extreme where the flow from supply to exhaust approaches a piston-type or displacement flow. Between these extremes is the case of perfect mixing of the indoor air. It is generally desirable to achieve a displacement-type flow to the degree possible since, with this type of flow pattern, the mean amount of time that air molecules remain in the structure is minimized and, thus, indoor pollutant concentrations will often be near their minimum.

A parameter that characterizes the overall air flow pattern in a structure is the nominal ventilation efficiency (also called the overall ventilation efficiency or simply the ventilation efficiency). If the indoor air is perfectly mixed, the nominal ventilation efficiency will equal unity. Short-circuiting reduces the nominal ventilation efficiency below unity. As the air flow pattern ranges between complete mixing and perfect displacement flow, the corresponding nominal ventilation efficiency will range between unity and a maximum value of two. For a hypothetical pollutant that is generated at a uniform rate throughout the ventilated space, the nominal ventilation efficiency would equal the concentration of pollutant in the exhaust airstream divided by the average indoor concentration. Another interpretation is provided by Sandberg and Sjoberg (1983) who have applied age distribution concepts to characterize ventilation and has demonstrated that the nominal ventilation efficiency equals the age of the indoor air if this air was perfectly mixed (i.e., structure volume/air flow rate) divided by the actual mean age of the indoor air. (The age of a parcel of air is defined as the amount of time that has elapsed since the air entered the structure.) One last interpretation is that this nominal ventilation efficiency is the effective ventilation rate divided by the nominal ventilation rate where the latter equals the flow rate of outdoor air into the structure divided by the volume of the ventilated space.

The largest set of data on the nominal ventilation efficiency of various ventilation systems in a single-chamber space is provided by Sandberg et al. (1982b). Data is presented from more than one hundred experiments with six different ventilation systems. The nominal ventilation efficiency varied widely -- from approximately 30 to 125%. The highest ventilation efficiencies occurred when the supply and exhaust registers were located at different heights (i.e., supply at ceiling and exhaust at floor level or supply at floor and exhaust at ceiling level) and when, simultaneously, the relationships between supply and exhaust temperature was such as to promote a displacement-type flow pattern. (The air temperatures had an impact on the flow pattern because of the variation in air density with temperature. A vertical displacement flow can be promoted by: 1) supplying air near the ceiling that is warmer than the exhausted air and exhausting air near the floor, or 2) supplying air near the floor that is cooler than the exhausted air and exhausting air near the ceiling.) Additional factors that also have an impact on nominal ventilation efficiency are the ventilation rate and the supply diffuser design; the experiments by Sandberg et al. (1982b) also include variations in these parameters.

Other data on nominal ventilation efficiency are presented in many of the papers cited above. The nominal ventilation efficiencies of residential MVHX systems (described earlier) were measured in studies by Offermann et al. (1982c, 1983). The efficiencies of seven ducted MVHX systems installed in residences ranged from 56 to 73% and averaged 62%. An unexplained low efficiency of 22% was measured in an eighth house. The nominal ventilation efficiencies of unducted MVHX systems measured in laboratory studies ranged from 49 to 66% which is similar to the efficiencies of the ducted systems. In these MVHX systems, considerable leakage of air occurred between the supply and exhaust airstreams and this leakage caused reductions in their ventilation efficiencies. Improvements in ventilation performance might be achieved easily by manufacturing and design changes aimed at reducing leakage between airstreams.

2. Multiple Zones and Air Distribution

In the previous discussion, the ventilated space was treated as one zone and the nominal or overall ventilation efficiency for the entire space was discussed. The nominal ventilation efficiency does not, however, provide information to characterize the ventilation at specific indoor locations. Actual structures may be physically divided into many different zones (e.g., rooms) and the effective rate of ventilation can vary greatly between zones. Even in a single room, the effective rate of ventilation can vary widely from point to point. Thus, modeling efforts and measurement techniques have been developed for evaluating the ventilation at specific locations.

Ventilation researchers sometimes consider a ventilated room to be composed of two distinct zones. The occupied zone consists of the region from floor level to above the breathing level and the remaining region is considered to be the unoccupied zone. (Alternately, the two zones may be different rooms.) It is generally preferable, from an air-quality viewpoint, for the ventilation air to flow from the occupied to the unoccupied zone as this flow pattern minimizes the mean age of air (and in many situations the average pollutant concentration) in the occupied zone. If each of the zones is assumed to be perfectly mixed and the rate of air recirculation between zones is specified, the ventilation efficiency in each zone can be predicted using a simple model. The ventilation efficiency in this case (which may be called the local ventilation efficiency or the steady-state relative ventilation efficiency) equals the age of the indoor air if this air were perfectly mixed divided by the actual mean age of air in the occupied zone although other interpretations can also be made. In Figure 2.10, this predicted local ventilation efficiency is plotted versus the rate of recirculation between zones for two different locations of air supply and exhaust. The advantage of arranging the supply and exhaust terminals to promote a displacement flow and of supplying ventilation air directly to the occupied zone are clearly illustrated. As the rate of air recirculation between zones increases, however, differences in performance of the various systems becomes less distinct. The two-zone model used to generate Figure 2.10 is obviously a gross simplification of actual air flow patterns but the general trends indicated by the model are valuable. Also, experiments conducted by several researchers (Malmstrom and Ahlgren, 1982; Sandberg, 1981; Skaret and Mathisen, 1982; and Skaret, 1983b) indicated that significant stratification can actually occur in the vertical direction, thus, a two-zone model may provide a reasonable approximation of actual performance in some situations.

In addition to modeling ventilation efficiency in different zones, the ventilation efficiency at specific locations can be measured. (In this paper we use the term "local ventilation efficiency"

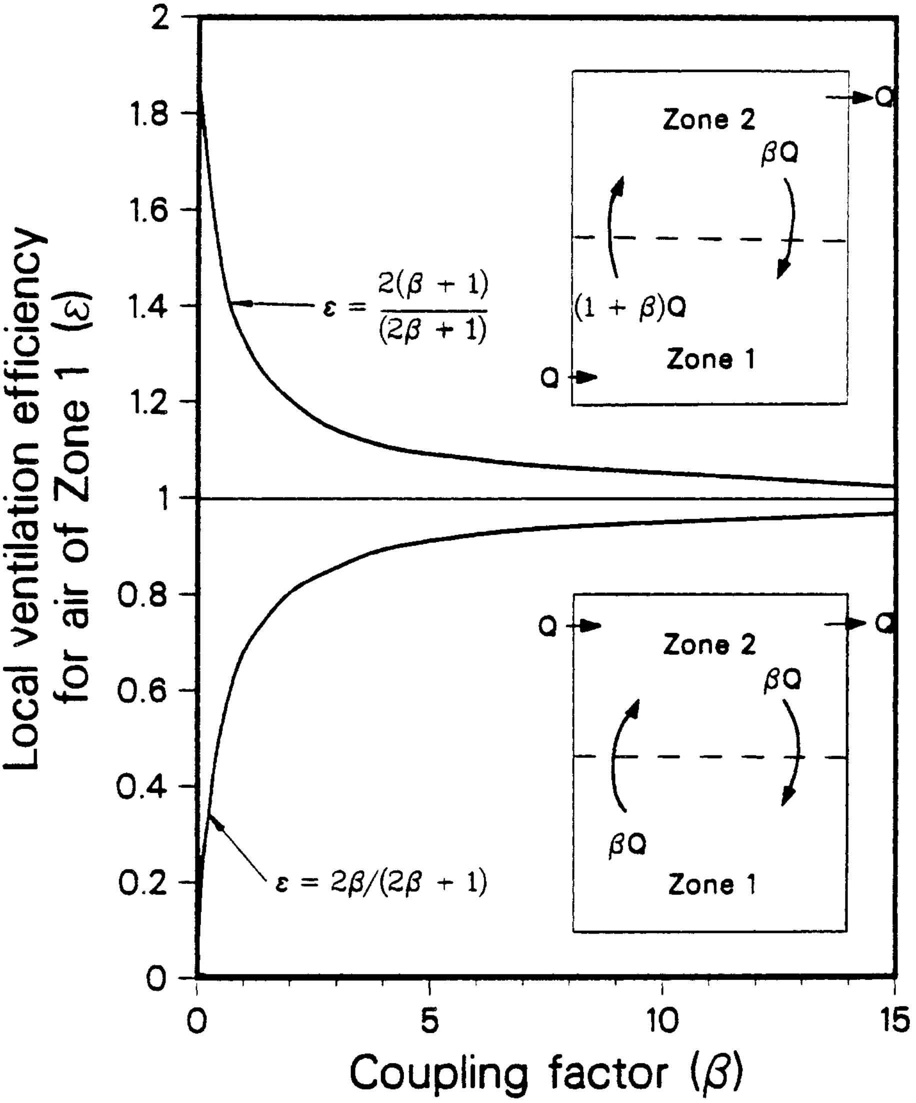

Figure 2.10 Local ventilation efficiency (for air) of zone 1 versus coupling factor and locations of air supply and exhaust, calculated from a two-zone model; perfect mixing was assumed within each zone.

to describe the ventilation efficiency at a specific location -- other investigators have sometimes used the term "relative ventilation efficiency".) The age-based interpretation of the local ventilation efficiency is the age of air with complete mixing divided by the actual local age of air. Some researchers have interpreted the local ventilation efficiency as a local ventilation rate divided by the nominal (i.e., with complete mixing) ventilation rate. Measured local ventilation efficiencies are strictly valid only at the measurement point although they are often considered to be representative of a region (e.g., room) surrounding the measurement point. The relationship between the local ventilation efficiencies and the nominal or overall ventilation efficiency is simple -- the nominal efficiency equals the average of the local ventilation efficiencies for the entire ventilated space.

Measured local ventilation efficiencies have been presented by a number of researchers. Sandberg et al. (1982b) catalog results from more than one hundred experiments in a single room demonstrating that the local ventilation efficiency can vary substantially even within a single room. The highest local ventilation efficiencies often occurred near the location of air supply and the degree of variation in efficiency depended on ventilation system characteristics. Additional data from Sandberg and Sjoberg (1983), and Boman (1982) leads to the same conclusions.

Fewer data are available for the local ventilation efficiencies in multi-room structures. Sandberg (1982) conducted experiments in a two-room structure with air entering primarily in one room and exiting the other. The local ventilation efficiency was always significantly higher in the room where ventilation air first entered. When the door between rooms was closed and air flowed through a slot under this door, the local ventilation efficiency was at least twice as high in the room where ventilation air entered. The average ventilation efficiency for the entire space was as high as 140% with a closed door between rooms. Freeman et al. (1982) measured the local ventilation rate (i.e., the inverse of the local age of air) in each room of a two-story residence under a variety of conditions. The mixing between rooms was generally very good (i.e., the local ventilation rate was nearly uniform). Offermann et al. (1982c, 1983) determined local ventilation efficiencies when unducted residential MVHX systems were operated in a three-room test structure and in a single-story research house. Variations up to ± 20% from the mean local ventilation efficiency occurred when internal doors remained open. Mixing between a room and the remainder of a house separated by a closed door was very poor unless a forced-air heating system was operating.

3. Ventilation Efficiency For Pollutants

The ventilation efficiencies discussed previously are valuable for characterizing overall air flow patterns and distribution of ventilation air. They are also good indicators of the efficiency of ventilation systems in removing pollutants that are generated uniformly throughout the ventilated space. In actual structures, however, pollutants are often generated or emitted nonuniformly (i.e., from points or surfaces). Optimally, the ventilation process should be designed for the most efficient removal of these pollutants. Experimental studies have, therefore, been undertaken to evaluate ventilation strategies when actual pollutant sources are simulated using tracer gases, and two-zone models have been developed for the case of nonuniform pollutant distribution. Ventilation efficiencies that indicate the efficiency of pollutant removal have also been defined.

Actual pollutant sources are often highly concentrated (e.g., point sources) and in such cases it is common to exhaust air from locations close to the pollutant source. The objective of this local exhaust ventilation process is to remove the pollutants to the degree possible before they can disperse throughout the structure. The performance of local exhaust systems is often assessed qualitatively by visual observations of smoke released near the point of local exhaust (e.g., the exhaust hood). A useful quantitative performance parameter is the steady-state relative ventilation efficiency for the pollutant which is defined by the equation

$$\eta_{ss} = (C_e - C_s) / \overline{C} - C_s) \tag{2.8}$$

where: C_e = the steady-state concentration in the local exhaust airstream,

C_s = the steady-state concentration in the general supply air, and

$\overline{C}$ = the average concentration in the indoor air or the occupied zone.

If the local exhaust system is effective, this efficiency will be significantly greater than unity. Another, but similar, ventilation efficiency for local exhaust systems is defined by Revzan (1984). Selected literature on local exhaust ventilation was reviewed previously in the section on mechanical exhaust ventilation.

The performance of general dilution ventilation systems in removing pollutants that are emitted nonuniformly has also been modeled and studied experimentally using tracer gases. The relative ventilation efficiency defined in the previous equation remains useful as a performance indicator. If the average pollutant concentration in the occupied zone is used in this equation, the result is the relative ventilation efficiency for pollutant removal in the occupied zone. This efficiency can be predicted using a two-zone model and typical predicted results are shown in Figure 2.11 for two supply and exhaust configurations and three locations for the pollutant source. The advantage of a displacement type ventilation are again evident.

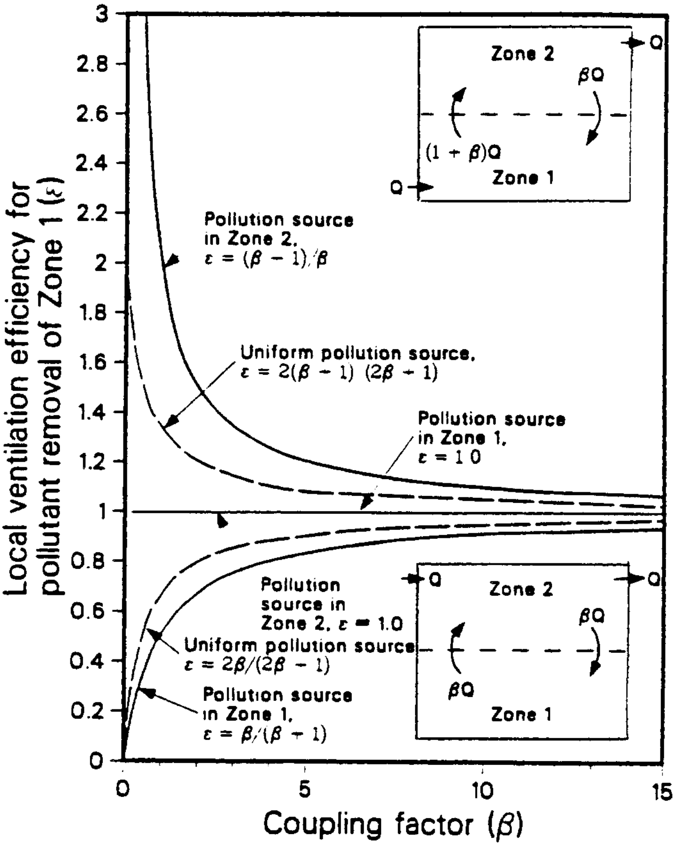

Figure 2.11 Local ventilation efficiency (for pollutant removal) of zone 1 versus coupling factor and locations of air supply and exhaust, calculated from a two-zone model; perfect mixing was assumed within each zone.

Measurements with simulated nonuniform pollutant sources and dilution ventilation systems have been conducted by Malmstrom and Ahlgren (1982), Skaret and Mathisen (1982, 1983), and Sandberg and Sjoberg (1983). At this time, the data appears too limited for general conclusions, but the results again suggest that ventilation system performance can be substantially improved compared to the performance of systems typically used today.

G. Ventilation of Large Buildings

While this chapter has focused primarily on ventilation of residential buildings, a brief description is provided below the techniques used to ventilate large (e.g., office) buildings and of potential problems with their ventilation that can adversely affect air quality.

1. Description

Large buildings almost always contain a mechanical ventilation system and some also have provisions for natural ventilation (e.g., openable windows). Uncontrolled infiltration (air leakage) can also be significant in these buildings. Several independent or semi-independent mechanical ventilation systems may be present in a single building. A number of ventilation system designs are in common use; only features that are common in many of these designs and that are relevant to this discussion are described here.

The typical mechanical ventilation system draws in outside air through an adjustable damper (outside-air damper) and this outside air is usually mixed with a larger amount of recirculated (return) air. The mixture of outside and return air (i.e., supply air) is directed through a supply-air duct system and eventually delivered through supply-air diffusers (usually located in the ceiling) to various regions within the building. At one or more points in the duct system, the air may pass through heating and/or cooling coils and other equipment. Control of the flow rate of the supply air may be accomplished by manual and/or automatic adjustment of dampers at various locations in the supply ductwork and near the supply diffusers. In some buildings, the flow rate may also be varied by controlling the speed of the supply-air fans. A common feature in relatively new buildings is a variable air volume (VAV) box, located upstream of a group of supply diffusers, which automatically regulates the flow of supply air out of the diffusers in response to thermal loads in the regions served by the diffusers.

In addition to the supply air system, return air is withdrawn from various locations in the building through exhaust registers that are also usually located in the ceiling. The return air is directed through ductwork, or through the plenums above a suspended ceiling, to the junction where, as mentioned earlier, a portion of this air is generally mixed with the incoming outside air. The remainder of the return airstream (i.e., the exhaust airstream) is exhausted to outside through an exhaust air damper. The flow rate of the exhaust airstream should approximately equal the flow rate at which outside air is brought into the building.

In the majority of large buildings, the rates at which outside air is brought into the building and indoor air is exhausted to outdoors are regulated automatically over time in response to heating or cooling loads and outside temperature. If outside air is at a temperature that makes it suitable for cooling, a large amount may be admitted to the building in a mode of operation called the "economizer cycle". When heating or cooling loads are high because of high or low outdoor temperatures, the flow rate of outside air may be adjusted to a minimum value in order to save energy. In some buildings, the minimum outside air is admitted through a separate set of dampers that are not controlled thermostatically in response to thermal loads; for example, these dampers may be subject only to manual adjustment.

2. Potential Problems

One potential problem with the ventilation of large buildings is that an insufficient amount of outside air may be delivered to the building as a whole during some or all periods of operation. This may be due to a number of factors including the following: 1) the control system that regulates the intake of outside air may be faulty or improperly adjusted (e.g., in some instances the outside-air damper may be fully closed due to adjustments intended to save energy and entry of outside air is due only to leakage through this damper or leakage through the building envelope); 2) the ventilation system or associated control system may be poorly designed, constructed, or maintained; 3) the building may have been enlarged or the occupancy increased dramatically without increasing the capacity of the ventilation system; 4) components of the ventilation system may be shut down (i.e., non-operating); and 5) the ventilation requirements may be higher than design values because of the presence of indoor pollutant sources that were not considered during the design process.

Another ventilation problem that can lead to poor air quality is that insufficient outside air may be delivered to certain regions within the building. A number of factors can lead to these instances of poor air distribution: 1) improper system design; 2) poor control or adjustment of flow rate through the various supply ducts; 3) insufficient delivery of air through specific supply-air diffusers; and 4) subdivision (after initial construction) of the building interior into compartments without provisions for the ventilation of each compartment. The rate of air supply to different regions is, in general, automatically controlled or manually adjusted based on the requirements for thermal comfort. Complaints regarding thermal comfort, drafts, or noise may motivate reductions in air supply to a specific region which in turn can lead to inadequate ventilation for the control of indoor pollutant concentrations.

Even when an adequate amount of outside air exits the supply-air diffusers, this air may short-circuit to the return registers and, thus, not be efficiently distributed to the breathing zone. The factors which affect the degree of short-circuiting of ventilation air and the efficiency at which pollutants are removed by ventilation are discussed in the previous section on ventilation efficiency.

3. Corrective Actions

A relatively small amount of published information is available on correcting the various problems noted above. Often, the indication of problems is a high incidence of complaints from building occupants and it is very difficult to determine if these complaints are caused by poor ventilation or other factors such as unusually high pollutant sources, thermal discomfort, or various psychological factors. In many buildings, the flow rate of outside air can be increased. A few problems with the ventilation system are easily detected by inspection (e.g., fully closed outside-air dampers) and can be easily corrected. In general, however, assessment of the performance of the ventilation system is difficult -- no convenient and inexpensive technique exists to determine how well specific regions within a large building are ventilated. The preferred long-term solution is to gain a better understanding of the performance of ventilation systems in large buildings and to modify designs, control strategies, construction practices, and maintenance practices to minimize the frequency of problems.

H. Summary

Ventilation is the most widely used and readily available technique for maintaining acceptable concentrations of indoor-generated air pollutants. The relationship between ventilation rate and indoor pollutant concentration is complex and, in general, best predicted using theoretical models. Large changes in indoor pollutant concentrations (i.e., factors of three or four) result from corresponding large changes in ventilation rate. It is clearly desirable to avoid extremely low ventilation rates because of the high pollutant concentrations that result. Indoor pollutant concentrations can still be unacceptably high in structures with normal ventilation rates, however, when pollutant source strengths are high. In such instances, it may be impractical to increase the ventilation rate sufficiently to achieve acceptable indoor air quality and other control measures should be considered, possibly in conjunction with increased ventilation. Only limited measurements have been performed to assess the changes in pollutant concentrations that result from changes in ventilation rate. There is a particularly strong need for data that relates the method and rate of ventilation to the source strengths for radon and the concentration of formaldehyde.

In the past, residential buildings have been ventilated primarily by infiltration during periods when the structure is heated or cooled. A major disadvantage of infiltration is that it is not easily controllable, i.e., it varies with weather conditions and building tightness. Construction of more air-tight residential buildings and incorporation of mechanical ventilation is, therefore, increasingly common in regions with cold climates. The use of mechanical ventilation has been a common practice in large buildings for many years.

One of the major disadvantages of large ventilation rates is the energy required to heat or cool the ventilation air, and this factor is becoming increasingly important as energy prices rise. To reduce energy requirements associated with ventilation, techniques have been developed to recover energy from the exhausted air. One such technique is to provide balanced mechanical ventilation and to transfer heat between the incoming and outgoing airstreams in an air-to-air heat exchanger. The technique has been employed for years in a limited number of commercial buildings and is now being used in residences. Available data on the energy performance of these systems is primarily from laboratory investigations. Using computer models and laboratory data, significant but variable energy savings are predicted when these systems are used in cold climates. Field studies to determine the energy savings that actually occur would be valuable but are difficult to perform. The economics of using air-to-air heat exchangers compared to reliance on infiltration is highly dependent on climate, ventilation rate, cost of heating fuel, heat exchanger performance, and capital costs.

An alternate technique to mechanically ventilate residences is to employ mechanical exhaust ventilation. This technique appears attractive from both an energy and air quality standpoint because of the controllable and stable ventilation rates that result and because air can be exhausted from specific locations. It is also possible to recover energy from the exhaust airstream using a heat pump and to transfer this energy to the domestic hot water supply or the indoor air. Measured data and theoretical predictions for the performance of these exhaust ventilation systems are very limited. A significant concern is that exhaust ventilation may, in some structures, increase the radon source strength. Further investigations of both the energy and environmental (i.e., air quality) performance of exhaust ventilation are required before conclusions can be made regarding its general applicability.

The technique of local exhaust ventilation has been shown to be highly effective in minimizing the transport of pollutants from a concentrated source to the occupied space. When pollutant sources are concentrated, such as the emission of combustion products from a gas range, local exhaust ventilation can maintain low indoor pollutant concentrations much more effectively (i.e., with less exhausted air) than general dilution ventilation of the entire structure. A useful systems performance indicator is the ratio of pollutant concentration in the local exhaust airstream to the average indoor concentration -- this ratio should be significantly greater than unity.

A method to reduce the energy requirements associated with ventilation is to improve the control systems that regulate the supply of outdoor air. In addition to improving the existing control systems that generally use thermal loads and outdoor temperatures as the basis for control, new control strategies that respond to indoor pollutant concentrations appear attractive in some situations.

Experimental investigations and theoretical models have shown that there is a wide range in the efficiencies of dilution ventilation processes. The overall air flow pattern through a structure can range from one extreme with a large amount of short-circuiting of air from supply to exhaust to the other extreme where the flow from supply to exhaust approaches a more desirable piston-type or displacement flow. The distribution of ventilation air throughout a single room or between rooms can be highly nonuniform. The effectiveness of ventilation systems in removing pollutants that are emitted nonuniformly is also highly variable. A number of "ventilation efficiencies" have been defined that are useful for characterizing ventilation system performance and that can be determined experimentally using tracer gases. Available data indicates that significant improvements in ventilation system performance can be obtained by supplying and exhausting air at different levels (i.e., floor and ceiling levels) and simultaneously controlling supply air temperature appropriately to promote a displacement flow pattern. There are also advantages to supplying the fresh ventilation air to the occupied zone. Most data on ventilation efficiency has been obtained in laboratory studies and further measurements are needed, particularly in large buildings. Available data suggests that significant reductions in energy consumption and improvements in air quality may be obtained by using more efficient ventilation systems.

3. Working Principles and Background Information on Pollutant-Specific Control Techniques

A. Overview

In this section we review the working principles for various techniques to control indoor pollutant concentrations (excluding the technique of ventilation) and comment on the energy and maintenance requirements, and costs of each technique. Control techniques that are not based on ventilation can be categorized in two basic groups: 1) source control measures that reduce the rate at which pollutants are emitted indoors or transported from their source to indoors, and 2) air cleaning, i.e., various methods for removing pollutants from the indoor air. The effectiveness of any specific source control or air cleaning technique, in contrast to that of ventilation, is generally limited to one or a few pollutants.

B. Source Control

Perhaps the most basic source control measure is to exclude the source of pollutants from indoors or minimize the quantities of source material. If transport of a pollutant from the source to indoors is possible (e.g., transport of radon from high-radium soil to indoors), then exclusion of the source from close proximity to the buildings is a potential control technique. Exclusion is generally more practical when it is accomplished during building construction, although removal or reduction of sources in existing buildings can also be a viable control measure. As an alternative to exclusion, pollutant sources can be modified to reduce the rate of pollutant emission; examples include basic modifications to unvented combustion processes or changing the composition of urea formaldehyde resins used in pressed wood products. Finally, various techniques can be employed to isolate the source from the indoor air; these techniques include the installation of physical barriers to inhibit transport of pollutants from the source (e.g., the soil) to indoors, and the use of local ventilation at the source to direct the pollutant to outdoors and, thus, prevent its entry into the indoor air.

The effectiveness of a source control measure is generally indicated by its impact on the pollutant source strength (i.e., rate of emission indoors or transport to indoors), or by its impact on the indoor pollutant concentration. In many cases it is feasible to reduce the indoor pollutant concentration by a large factor when employing source control. Obtaining equally large reductions in concentration by increasing the pollutant removal rate (i.e., ventilation or air cleaning rate) is generally practical only if the initial pollutant removal rate is unusually low.

In addition to the potentially large impact on indoor pollutant concentration, there are other important advantages to source control.

Many source control measures do not significantly increase building energy demands and require minimal or no maintenance. Source control may, however, involve the use of alternative and more expensive materials, appliances, construction practices, or manufacturing processes or may involve the expensive removal and replacement of materials. In cases where the initial costs for source control are reasonable, it is potentially the most cost effective type of control measure.

C. Removal of Particulate Phase Contaminants

A variety of different processes are available for removing particulate phase contaminants from ventilation air. These devices can be separated by working principles into two groups: mechanical filters and electrostatic filters. Mechanical filters remove particles from air as a result of the mechanical forces imposed on the particles by the airstream and filter media. Electrostatic filters on the other hand rely more on electrostatic forces to remove particles from the air.

1. Particulate Removal Mechanisms in Mechanical Filtration

The removal of particles from air by mechanical filtration is generally accomplished by passing the air through a fibrous media. There are five basic mechanisms by which particles can be deposited on the fibers in a filter: 1) inertial impaction, 2) interception, 3) diffusion, 4) electrostatic attraction, and 5) gravitational settling.

Inertial Impaction: Filtered air makes an abrupt change in direction as it passes around each fiber in the filter. Particles of sufficient size collide with the fiber because of their inertia. This is the predominant means of collection for larger particles (e.g. > 1.0 μm in diameter). Collection efficiency by inertial impaction increases with particle size and air velocity.

Interception: Interception occurs when a particle follows a gas streamline that is within one particle radius of the fiber. The particle makes contact with the fiber as it passes and is removed. Interception is the only mechanism that does not depend on air velocity and is an important removal mechanism in the particle size range of minimum removal efficiency. Collection by interception increases with increasing fiber density.

Diffusion: Brownian motion of small particles results from random collisions with the surrounding gas molecules. This motion increases the probability of a particle hitting a fiber while traveling past it on a nonintercepting streamline. Diffusion is the only deposition mechanism that increases with decreasing particle size and is the predominant collection mechanism of very small particles (e.g., less than 0.01 μm in diameter).

Electrostatic Attraction: As a charged particle passes close to an uncharged fiber it induces an equal and opposite charge on the surface of the fiber and the resulting electrostatic force attracts the particle toward the fiber. Similarly, when an uncharged particle approaches a charged fiber, the electrostatic image forces developed aid in the removal of the particle. The dielectric constant of the fiber material has an important effect on the development of image forces. Charged particles are also attracted to oppositely charged media by coulombic forces. These forces are much stronger than image forces and are the dominant collection force present in electrostatic filters.

Gravitational Settling: Gravitational settling of particles in fibrous filters is an insignificant removal mechanism for small particles (e.g., less than 1.0 µm in diameter).

The mechanisms of impaction, interception and diffusion predominate for different conditions of particle size and air velocity. The relationship between filter efficiency and particle size are shown in Figure 3.1 for a typical fibrous filter. For particles less than 0.01 µm in diameter, diffusion is the dominant removal mechanism while interception and inertial impaction dominate removal of particles greater then 1.0 µm in diameter. The overall filter efficiency calculated from equations for diffusion and impaction reveals that there is a minimum efficiency at an intermediate particle size where the particle is too large for diffusion to be effective and too small for interception to be effective. Because these two mechanisms dominate in different size ranges, for all filters there is a particle size for which the removal efficiency is a minimum. Depending on the fiber size, fiber density, and air flow rate, the particle diameter at which this minimum efficiency occurs can range from 0.05 to 0.5 μm (See Fig. 3.1).

2. Types of Mechanical Filters

Various types of fibrous media filters are commercially available for cleaning ventilation air. Three factors are important for characterizing the performance of these filters: fiber packing density, average filter diameter, and air velocity. Increasing the fiber density increases filter efficiency at the expense of an increase in air flow resistance. The air flow in fibrous filters is usually laminar and thus the resistance to airflow is directly proportional to the airstream velocity.

Panel Filters: These filters have a low packing density of coarse glass, animal hair, vegetable, or synthetic fibers. The filter fibers are coated with a viscous substance, such as oil, which acts as an adhesive on impinging particles. The filters are characterized by low pressure drop, low cost, high removal efficiency for very large particles such as lint, but a negligible removal effi-

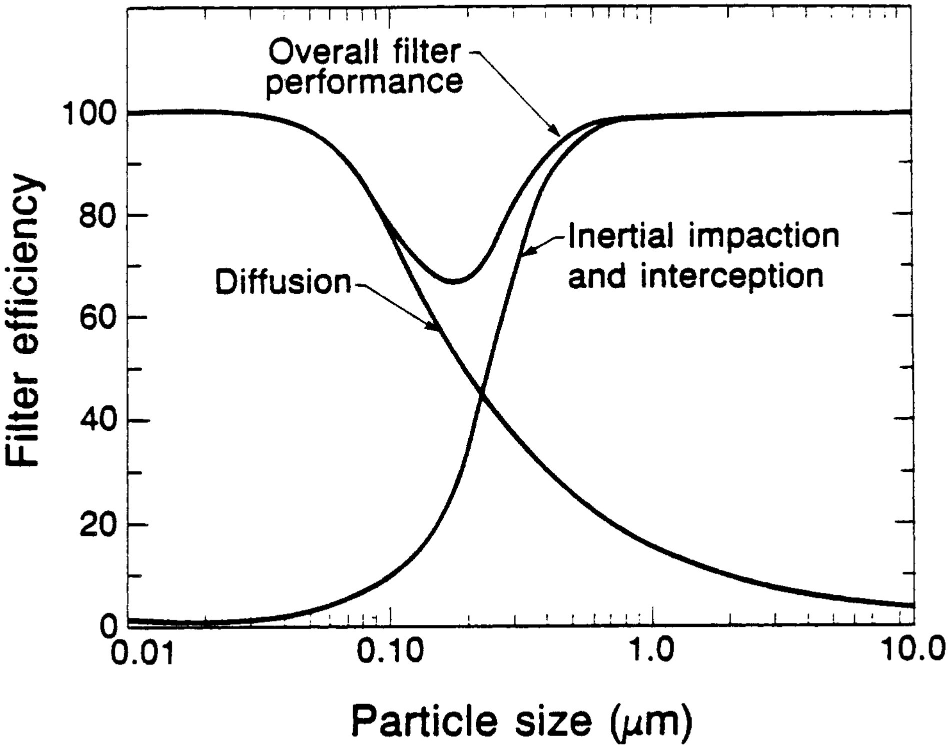

Figure 3.1 Particle removal efficiency as a function of particle size for a typical fibrous filter. (Adapted from Hinds, 1982).

ciency for particles smaller than 10 μm in diameter. The common residential furnace filter is an example of this type of filter.

Extended Surface Filters: Increased particle collection efficiency can be achieved by decreasing the fiber size and increasing the fiber packing density; however, these measures also increase air flow resistance. By extending the surface area of the filter media, the flow velocity through the media is reduced, which, in turn, reduces the pressure drop across the filter. One way of extending the media surface area is to deploy the media in a pleated or folded form. The much larger ratio of medium surface area to face area in these filters allows use of much denser and hence more efficient filter media while maintaining acceptable pressure drops. Extended surface filters also offer much higher dust holding capacities.

HEPA Filters: High Efficiency Particulate Air (HEPA) filters, are special types of extended surface filters characterized by their very high efficiency in removing submicron particles. Initially developed for use in nuclear material processing plants to control concentrations of fine airborne radioactive particles, a HEPA filter is defined as a disposable dry-type extended-surface filter having a minimum particle removal efficiency of no less than 99.97% for 0.3 μm particles and a maximum resistance, when clean, of 249 Pa (1.0 IWG) when operated at rated air flow rate (Institute of Environmental Sciences, 1968). The filter core generally consists of a continuous web of filter media folded back and forth over corrugated separators that add strength to the core and form the air passages between the pleats. HEPA filter media are composed of very fine submicron glass fibers in a matrix of larger diameter (1-4 μm) fibers. A number of grades of high efficiency fibrous filters are available with minimum efficiencies ranging from 95 to 99.99% for HEPA grade.

3. Working Principles of Electrostatic Filters

Various electrostatic filtration processes have been developed for removing particles from air. While removal as a result of mechanical effects such as diffusion and inertial impaction still occur to some degree, the major removal mechanism is electrostatic attraction. As a class these devices are characterized by a low pressure drop which is nearly independent of dust loading and by high removal efficiency for small particles. Three topics which are important for understanding the operation of electrostatic filters are: air ionization, particle charging, and particle migration velocity.

Air Ionization: A convenient method of creating a large source of ions for particle charging in electrostatic filters is to produce a corona discharge by supplying a high voltage to a thin wire or a sharply pointed electrode. The high voltage creates an electric field that is sufficiently strong near the surface of the wire to ionize gas molecules. In this region, called the corona, free electrons are accelerated sufficiently to strip electrons from surrounding gas molecules, creating positive gas ions and additional electrons. These additional electrons are, in turn, accelerated and cause further impact ionization. This chain-reaction process called a corona discharge, produces large quantities of electrons and positive ions. If the electrode is positive, the electrons will move rapidly to the electrode and the positive ions be repelled from the wire. If the electrode is negative, the positive ions will be attracted to the electrode and the electrons will be repelled. These free electrons then attach to electronegative gases such as oxygen and water vapor thereby producing negative ions. Because of the high energies in the corona region, it is possible for some ozone, an air contaminant, to be produced from oxygen. Most electrostatic precipitators designed for cleaning indoor air use a positive corona since it produces much less ozone than a negative corona.

Particle Charging: Two mechanisms by which aerosol particles can acquire charge are diffusion charging and field charging. In diffusion charging the particles pick up charges as a result of the random collisions between the ions and particles. Field charging results from collisions of particles with the rapidly moving ions in a strong electric field. The ions move along the electric field lines and strike particles which intersect those lines. The charge acquired is directly proportional to the particle diameter for diffusion charging and to the particle diameter squared for field charging. Field charging is usually the dominant charging mechanism for particles larger than 1.0 μm while diffusion charging usually dominates for particles smaller than 0.1 μm in diameter.

Particle Migration Velocity: The motion of a particle in an electric field is governed primarily by electrostatic and aerodynamic forces. The terminal velocity of a particle due to these forces is called the migration velocity and is analogous to the settling velocity of a particle falling in a gravitational field. Migration velocity is directly proportional to the charge on the particle and the strength of the electric field. Figure 3.2 shows a typical plot of theoretical migration velocity as a function of a particle diameter. As can be seen from this figure, for a given charging condition a minimum migration velocity occurs at an intermediate particle size. The increase in migration velocity as particle diameter decreases below 0.1 μm is largely a result of the decreased aerodynamic drag force (i.e., slip) experienced by these particles. The particle collection in electrostatic filters is determined primarily by the particle migration velocity, collection surface area, and air flow rate.

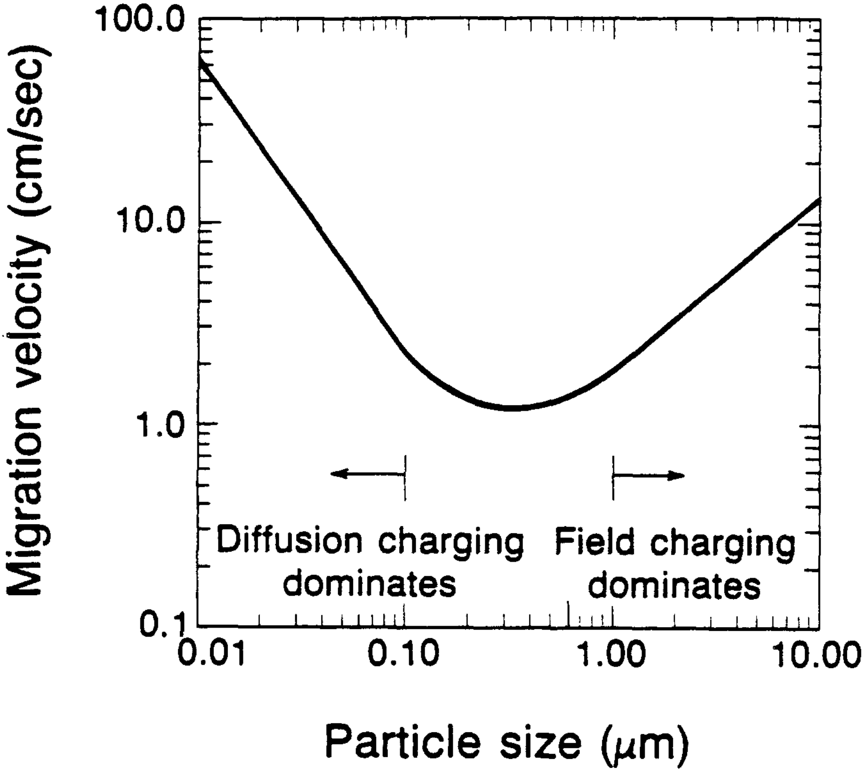

Figure 3.2 Particle migration velocity as a function of particle size for a typical set of charging and electric field conditions. (Adapted from Stern, 1977).

4. Types of Electrostatic Filters

Three types of electrostatic filters for removing airborne particles are described below.

Ionizing-Flat Plate Precipitators: This type of filter is often referred to as an electrostatic precipitator. Most residential electrostatic precipitators are two-stage precipitators, that is, they have a separate ionization stage preceding the precipitation state (see Fig. 3.3). Airborne particles are first charged by ions produced with an electric corona and then collected as they pass between a series of alternately charged and grounded collection plates.

The basic performance of electrostatic precipitators is described by the Deutsch-Anderson equation,

$$E = 1 - \exp\left[\frac{-V_m A}{Q}\right] \tag{3.1}$$

where: E = collection efficiency of particles with a migration velocity V_m,

A = area of the collection, and

Q = airstream flow rate.

This equation is based on assumptions of a uniform particle distribution throughout the cross section, fully charged particles entering

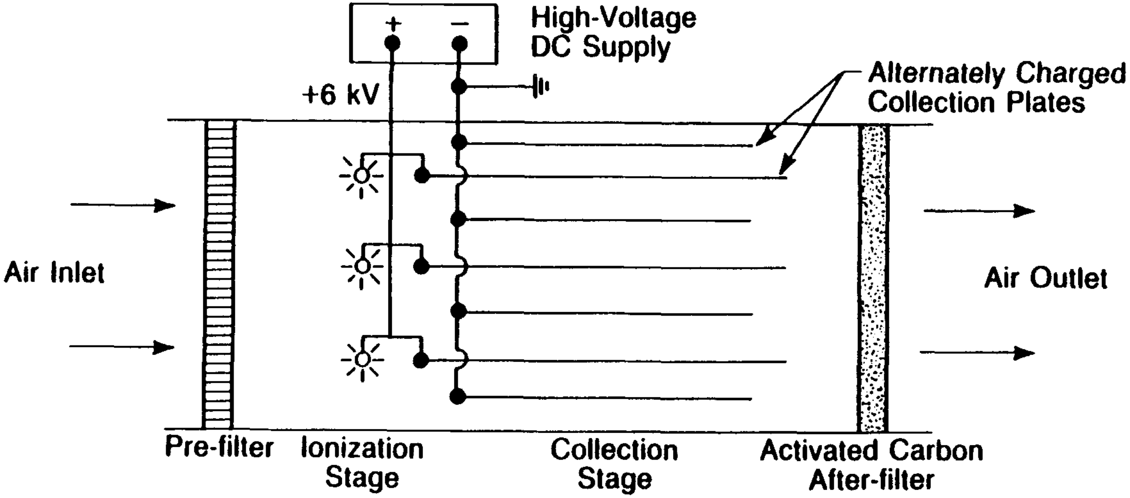

Figure 3.3 Schematic drawing of a two-stage flat plate electrostatic precipitator.

the collector, and no reentrainment of collected particles. From this equation, it is apparent that efficiency can be increased by increasing the collector area, A, or decreasing the airstream flow rate, Q. Similarly, increasing the particle migration velocity, V_m, by increasing the charge on the particle or the field strength will increase collection efficiency. As seen in Figure 3.2, there exists a particle size for a given charging condition that results in a minimum migration velocity. Precipitators designed to capture this size particle with 100% efficiency will be 100% efficient for all particle sizes.

Charged-Media Filters: The charged-media air cleaner combines certain characteristics of both mechanical and electrostatic filters. These devices augment the normal mechanical removal mechanisms attributed to fibrous filters by charging the fibers. Airborne particles passing close to the charged fibers are polarized and drawn to the fibers. Charged media filters which use a high voltage power supply normally use a filter medium constructed from a dielectric material such as glass or cellulose fibers. A gridwork of alternately grounded and charged members is in contact with the media, thus creating an intense and nonuniform electrostatic field.

A new filter media that is gaining popularity among manufacturers of air cleaning equipment employs a special fibrous material which is imbedded with permanent electrostatic charges called "electrets".

Similar to the externally charged media, electret media offers increased particulate removal efficiency as a result of the electrostatic forces imposed on the particulates, but no high voltage power supply is required for electret media. Tests performed with charged filter media have demonstrated high particulate removal efficiencies with relatively low pressure drops, however, there is some controversy regarding their performance after they have been loaded with particles.

Charged media filter devices may also incorporate an ionization stage so that particles are first charged in a corona-discharge ionizer, then collected on a charged media filter mat. This configuration provides higher efficiencies than would be possible if the charged media was used without a preceding ionization stage.

Ion-Generators: While not a filter in the same sense as other air cleaners, ionizers remove particles by charging them, after which they are attracted to surfaces at or near ground potential, such as walls, table tops, draperies, occupants, etc. In some cases, an oppositely charged collection surface is integrated as part of the device, which, in principle, reduces the problem of soiling of surfaces.

5. Performance Testing of Particulate Air Cleaners

Three operating characteristics which are important for the selection of particulate air cleaners are the following: 1) particulate collection efficiency 2) air flow resistance and 3) dust-holding capacity.

The American Society of Heating, Refrigeration and Air Conditioning Engineers (ASHRAE) has recommended several standard tests for evaluating filter performance (ASHRAE, 1976):

Weight Arrestance Test: A standard, relatively coarse, synthetic test dust is suspended in air and blown at a controlled rate through the test filter. The entering concentration is determined from measured air flow rate and the weight of the dust fed. The concentration of the dust in the air leaving the filter is determined by passing the entire airflow through a high efficiency after-filter and measuring the gain in filter weight. The arrestance, A, is determined as:

$$A = 100 \left[1 - (\text{weight gain of after-filter/weight of dust fed}) \right] \% \quad (3.2)$$

Because the dust used in this test is relatively coarse (mass median diameter $\sim$ 5-10 μm) this test primarily determines the ability of a filter to remove large dust particles (e.g., > 10 μm) and gives little information of the filter performance in removing the small respirable size particulates (e.g., < 1.0 μm) which constitute a significant mass fraction of indoor suspended particulates.

Dust Spot Efficiency Test: In this procedure, ambient air is blown at a controlled rate through the test filter. The naturally occurring particulate matter in the ambient air, atmospheric dust, serves as the test aerosol. Samples are drawn upstream and downstream of the filter being tested at equal flow rates through identical glass fiber sample filters. The downstream test air is sampled continuously while the upstream test air is sampled intermittently. The amount of upstream air sampled is adjusted so that there is equal discoloration of both sample filters as determined by an opacity meter. The dust-spot efficiency, E, is determined as:

$$E = 100 \left[1 - \frac{\text{volume of upstream air sampled}}{\text{volume of downstream air sampled}} \right] \% \qquad (3.3)$$

While this test gives more information than the weight arrestance test regarding the filter performance in removing small particles, neither of the current ASHRAE filter efficiency tests provide quantitative data regarding filter efficiency as a function of particle size.

DOP Penetration Test: This is a test developed by the United States Military (U.S., 1956) for high efficiency filters of the type used in clean room and nuclear applications. In this method, a smoke cloud of essentially uniform size droplets of DOP (di-octyl phthalate) is produced by condensation from DOP vapor. The particle diameter is controlled to be 0.3 µm and the cloud concentration is 80 mg/m^3. This smoke cloud is passed through the filter at a controlled flow rate and the concentration is measured upstream and downstream of the filter by a simple light scattering photometer. The filter penetration, P, is calculated as:

$$P = 100 \left[\frac{\text{Downstream concentration}}{\text{Upstream concentration}}\right] \% \qquad (3.4)$$

Fractional Efficiency Tests: No standardized tests have been established for determining the efficiency of air cleaners as a function of particle size. Some fractional efficiency evaluations have been conducted in laboratories using optical particle counters. Single-particle optical particle counters have recently found widespread use in clean room monitoring, pollution research, and laboratory aerosol studies because of their ability to rapidly measure both the concentration and size distributions of particles suspended in air. A beam of light inside the instrument is focussed onto a viewing volume through which airborne particles pass one at a time. The amount of light scattered from each particle is measured by a photosensitive detector and used to calculate the particle size. Fractional efficiency tests of filters are typically conducted by passing a test aerosol of known optical and physical characteristics through the filter and measuring the size distribution of particles upstream and downstream of the filter. Efficiency is calculated for each size range of particles as:

$$E_i = 100 \left[1 - \frac{\text{downstream concentration of size range } i}{\text{upstream concentration of size range } i} \right] \% \quad (3.5)$$

Laser-based optical particle counters can typically separate particles in the 0.10 to 10.0 µm size range into 20 or more size categories. By challenging a filter with a polydisperse aerosol, filter efficiency can be determined for a large number of particle sizes in one test. Recent developments in the design of optical particle counters has improved their reliability and availability so that this type of measurement may soon be adopted as a standard test protocol.

Dust Holding Capacity Test: A standard synthetic test dust is suspended in air and continuously fed to the filter. The test is normally terminated when the pressure drop across the filter exceeds the maximum value set by the manufacturer or when the particle removal efficiency drops below 85% of the initial efficiency. The dust holding capacity is calculated as the difference in the filter weight measured before and after the test.

Air Flow Resistance Test: The filter is installed in a standard 0.6 x 0.6 m (2 ft x 2 ft) test duct and measurements of the pressure drop across the filter are made at a number of air flow rates. From these data a curve of air flow rate versus resistance (i.e., pressure drop) can be generated.

6. Energy, Maintenance, and Cost Considerations

A significant advantage of removing particles from air by filtration rather than ventilation is that increased heating and air conditioning loads are avoided. Some additional fan energy is, however, required to overcome the resistance to air flow through the filter. The principle maintenance requirement is to periodically replace the filter media or clean the precipitator collection plates. The replacement/cleaning frequency is a function of the indoor particle concentration, air flow rate, and the filter dust holding capacity. An important consideration regarding tobacco smoke is the volatility of the removed particles. Tobacco smoke particles are liquid drops containing many odorous and volatile substances which continue to off-gas long after they have been collected. Thus, filters often require more frequent replacement or cleaning, as judged by odor emissions, when being used to control tobacco smoke particles. For this reason electrostatic precipitators are often recommended for tobacco smoke applications since they can be easily washed, in contrast to high efficiency fibrous filters which must be replaced.

D. Control of Gas Phase Contaminants

1. Adsorption

The principles of adsorption are reviewed in a number of texts; the material presented below is based primarily on the chapter on adsorption by Turk in the text by Stern (1977) and on the text by Treybal (1980). The processes of physical adsorption and chemisorption are described below followed by brief discussions of adsorbents, adsorbent impregnations, adsorbent performance data, and equipment used for adsorption.

Physical Adsorption: Gases and liquids will adhere on the surface of any solid but the amount of material that is retained is highly variable. The process of adherence is called adsorption, the solid is the adsorbent, and the material adsorbed on the solid is the adsorbate. Physical adsorption is due to forces of attraction between molecules of the adsorbed material and molecules of the adsorbent. The adsorbed molecules do not dissolve in the solid but instead remain on its surface. The adsorption process is exothermic; the amount of heat released with the adsorption of a gas is comparable to the heat of sublimation of the gas. Adsorbed gases may condense on the surface of the solid. The quantity of material that can be physically adsorbed is a function of several factors including: the surface area and polarity of the adsorbent, the volume and size of pores in the adsorbent, the molecular weight, size, shape, and polarity of the molecules in the gas or liquid that surrounds the adsorbent and the concentration of other molecules that may be competitively adsorbed. The adsorbent will eventually become saturated with adsorbate after which no net removal of adsorbate molecules occurs from the surrounding gas or liquid. Desorption of adsorbate can occur if the concentration of these molecules surrounding the adsorbent is decreased, the adsorbent temperature is increased, or some other material is introduced that is preferentially adsorbed.

Chemisorption: Chemisorption occurs when molecules become chemically bonded to the surface of the adsorbent. It is often an irreversible process. The amount of heat released during chemisorption is comparable to the heat of chemical reaction. The rate of chemisorption may increase with temperature, unlike that of physical adsorption. Chemisorption is also generally a more selective process than physical adsorption (e.g., only molecules with selected chemical properties are chemisorbed).

Adsorbents: Adsorbents that are used commercially have a very large surface area per unit mass due to the presence of microscopic pores -- for example, the surface area of a typical activated carbon may be 1000 m^2/gm. Pore diameters may be as small as a few angstroms. Adsorbents are generally used in a granular or powdered form.

The most commonly used adsorbents for air cleaning applications are activated carbons (also called activated charcoals) which are produced by carbonization of coconut shells, wood, coal, and a variety of other materials. Activation, which is a partial oxidation of the carbon, is accomplished by treatment with steam or hot air. Activated carbons are nonpolar and will preferentially adsorb nonpolar molecules, particularly those of high molecular weight. They remain effective in removing nonpolar pollutants from humid air because the polar water molecules attract each other, although the effectiveness of activated carbons may decrease somewhat if the relative humidity of the surrounding air is high (i.e., greater than 50%). Activated carbons are not highly selective, therefore, they have a variety of applications including removal of odors, a broad spectrum of organics, and solvents from air. They can be regenerated by heating to drive off the adsorbed material.

Other materials used for adsorption of gases include silica gels, activated alumina, and molecular sieves. These adsorbents are polar and, thus, preferentially adsorb polar molecules. Their usefulness for removing pollutants from moist air is restricted because of their affinity for polar water vapor molecules. (Adsorbents that are particularly useful for adsorbing water vapor are called desiccants.) A unique feature of molecular sieve adsorbents is that the size of the pores can be controlled. This feature may be advantageous -- for example, a molecular sieve with small pores may be chosen to exclude adsorption of large molecules. Molecular sieves are available with pores of various diameters between 3 and 10 angstroms.

Adsorbent Impregnations: Adsorbents can be impregnated with a variety of substances to improve their performance in removal of certain materials (e.g., pollutants). The impregnant may react chemically with the pollutant to produce a product that is less harmful or more readily adsorbed. Alternately, the impregnant may act as a catalyst for oxidation or decomposition processes that occur continuously during operation or periodically upon application of heat. One of the most common impregnants is potassium permanganate which oxidizes a variety of gases including formaldehyde and is often used to impregnate activated alumina. Other impregnants include bromine, phosphoric acid, sodium sulfite, sodium carbonate or bicarbonate, and a variety of metal oxides.

Adsorption Performance Data: Much of the available experimental data on the physical adsorption of gases are equilibrium data that indicate the capacity of an adsorbent for a particular gas. These data indicate, for a particular temperature, the concentration of adsorbate (i.e., mass adsorbate/mass adsorbent) that is in equilibrium with a specified partial pressure of the surrounding gaseous molecules of adsorbate. The equilibrium concentration of adsorbate is sometimes called the activity. In some cases, data are available on the simultaneous equilibrium between the adsorbent and two or more gases.

For air cleaning applications, the adsorbent must be replaced or regenerated before equilibrium is reached, because the efficiency at which the adsorbent removes adsorbate (e.g., pollutant) molecules from the airstream will decrease as equilibrium is approached. Also, the concentration of adsorbate molecules at the inlet of the adsorbent bed may vary over time and it is desirable to avoid desorption of the adsorbate during periods with a low inlet concentration. A more practical measure of the adsorbents capacity is, therefore, the "retentivity". There appears to be no uniformly accepted definition for retentivity, but it is an indicator of the maximum concentration of adsorbate that is strongly held by the adsorbent. For example, one test for retentivity is to saturate an adsorbent, then pass dry, adsorbate-free air through the adsorbent for six hours, and to determine the concentration of adsorbate retained after the six hours have elapsed (Strauss, 1975).

Still other measures of an adsorbent's capacity may be given. A common test is to pass an airstream with a specific temperature, humidity, flow rate, and concentration of adsorbate molecules through a bed of adsorbent granules and to determine the amount of adsorbate retained prior to breakthrough (i.e., detection of adsorbate molecules in the air downstream of the adsorbent). Alternately, the capacity is expressed by the amount of adsorbate retained before the removal efficiency of the bed has fallen below a prescribed value. These last two measures of capacity are dependent on characteristics of the airstream and the adsorbent bed.

In addition to the data on adsorbent capacity as noted above, information may be available on the pollutant removal efficiency and resistance to airflow for a particular bed of adsorbent; these data will also be dependent on airstream characteristics.

Performance data for chemisorbents are in many cases similar to those for physical adsorbents, although, unlike physical adsorbents, the capacity of a chemisorbent may be relatively independent of the surrounding concentration of the adsorbed gas. In some instances, however, only qualitative indicators of performance may be available (e.g., rapid, slow, or no reaction).

A major limitation of available data on adsorbent performance is its specificity. If a large and potentially unknown number of gases are available for adsorption, as is often the case when the application is the cleaning of indoor air, the available equilibrium and retentivity data may be inadequate for performance characterization.

Adsorption Equipment for Air Cleaning: A variety of techniques can be employed to promote contact between the adsorbent and an airstream containing pollutants. For a high pollutant removal efficiency, the residence time for pollutant molecules in the adsorbent bed must be sufficiently long to assure a high probability of contact with the adsorbent. (The airstream flows through the bed of granulated adsorbent and pollutant molecules are transported to the surface of the adsorbent by diffusion.) Other requirements of the contact system include the following: an acceptable resistance to airflow, a fairly uniform velocity distribution through the adsorbent bed, a prefilter which is sometimes advisable to remove particles, and a provision for regenerating or replacing the adsorbent. For air cleaning applications, a common technique is to pass the air through a stationary bed of granulated adsorbent and to periodically replace the bed with fresh or regenerated adsorbent. Beds may be up to a few centimeters thick (much thicker beds are sometimes used industrially) and are available in a variety of shapes (e.g., flat, pleated, etc.). Industrial adsorbent systems may contain moving beds of adsorbent, fluidized beds, and provisions for continuous regeneration.

Energy, Maintenance, and Cost Considerations: A significant advantage of adsorption air cleaning, when compared to ventilation, is that heating or air conditioning loads and the associated energy demands are not increased substantially when adsorption is employed. Some additional fan energy is, however, required to overcome the resistance to air flow through the adsorbent bed and any associated prefilter. The principle maintenance requirement is to periodically replace or regenerate the adsorbent. Without actual measurements of inlet and outlet pollutant concentrations, the determination of when to replace the adsorbent is difficult. The primary cost of adsorption air cleaning is the cost of the adsorbent and its periodic replacement or regeneration.

2. Absorption

Absorption is another air cleaning process that is used primarily for gaseous contaminants. In an adsorption process, also referred to as "gas scrubbing" or "air washing", an airstream contaminated with a gaseous pollutant is brought into contact with a washing liquid and the pollutant is dissolved into the solution. Typically, the contaminated air is passed through a liquid spray or over the wetted surfaces of a media. The pollutant is absorbed from the air only if it is soluble in the liquid and a driving force exists for transfer of the pollutant from the air to the liquid. To prevent saturation of the washing liquid with the pollutant and, thus, to maintain a driving force for pollutant transfer, the washing liquid must be periodically or continuously replaced or regenerated. In addition to gaseous pollutant removal, some air washers may be designed to remove particles by physical means.

The washing liquid to be employed for air washing depends on the gaseous pollutant to be removed. Water is commonly used, sometimes with additives, although other liquids may be more suitable for many pollutants. A primary criterion for choosing a washing liquid is the solubility of the gas in the liquid. The pollutant solubility should be high. Liquids which react with the pollutant, such that they effectively have very high capacities for absorbed pollutants, may be particularly useful washing liquids. Air washing is not likely to be effective for some indoor pollutants (e.g., radon) because of their insolubility in practical washing liquids.

Air washing is frequently employed for removing pollutants from industrial waste-gas streams and, consequently, equipment is commercially available. Available equipment includes smaller air washing devices designed for air flow rates appropriate for residential use. Equipment similar to air washing equipment is sometimes also used for evaporative air cooling or humidity control in commercial buildings. It is not known how suitable any of these commercially available devices would be for residential use; many of these devices are not designed to prevent humidification of the cleansed air.

The pollutant removal efficiency of an air washer is commonly defined as the fraction of the incoming pollutant that is removed from the contaminated airstream. This efficiency is determined by a number of factors including: the surface area for mass transfer, the flow rates and directions of air and washing liquid, the convective mass transfer coefficients (which are a function of geometry and flow rates), and the driving force (i.e., concentration difference) for transfer of the pollutant from the air to the liquid relative to the maximum potential driving force for this pollutant transfer. The driving force for mass transfer will depend on the concentration of the pollutant in the air and in the washing liquid and on the relationship between these concentrations at equilibrium. The maximum potential driving force for pollutant transfer occurs when there is no pollutant dissolved in the liquid. As was mentioned previously, replacement or regeneration of the washing liquid is necessary to maintain a driving force for pollutant removal. The total rate of pollutant removal from an indoor space will depend, as for all air cleaners, not only on the pollutant removal efficiency, but also on the air flow rate through the air cleaning device.

The power requirement of an air washer will be determined by the power requirements of the individual components. A fan is needed to force air through the air washer. Circulation of the washing liquid through the air also requires power, usually a liquid pump is employed. The washing liquid replacement or regeneration process will also contribute to the total power demand. Additionally, if water is employed as the washing liquid, a refrigeration system is needed to reduce or prevent evaporation of the water which would result in humidification of the indoor air. Because of the relative complexity of air washing equipment, the maintenance requirements for an air washer may be substantial compared to the requirements associated with many other indoor air quality control techniques.

The capital costs of air washing equipment as well as the maintenance costs will probably be significant because of the complexity of the required equipment. The costs for the energy consumed by air washers may also be considerable, however, the energy consumed can be rejected indoors during the winter to reduce the building's heating load. An air washer can also be configured so that it acts as an air conditioner during the summer. Thus, in cases where a large fraction of the energy consumed by the air washer provides usable heat and/or cooling, the net increase in energy consumption due to air washer operation can be small.

3. Air Circulation

A technique to reduce the indoor concentrations of some pollutants is to increase the rate of movement or circulation of indoor air. Increased air movement will facilitate the removal of some pollutants by reaction with or attachment to indoor surfaces because it increases the rate at which these pollutants are transported to the surfaces.

It is known that both particles and radon progeny will attach to surfaces. In addition, significant removal of nitrogen dioxide and, to a lesser extent, formaldehyde, has been attributed to reactions that occur with or on indoor surfaces. Little is known about the rates of pollutant removal by these processes or about details of the processes (i.e., the forces of attachment or the reactions that occur). It is possible that some gases physically adsorb to indoor surfaces at a significant rate but that these gases later desorb from the surfaces at a time when the indoor concentration is reduced.

In cases where air circulation is effective in reducing the indoor concentration, it could be a cost-effective measure because of the low costs involved. Also, if the removal processes were better understood, it might be possible to combine air circulation with the choice of specific surfaces or surface coatings. At the present time, however, large decreases in indoor pollutant concentrations due to air circulation have not been demonstrated. An exception is the substantial reduction in radon daughter concentrations with increased air circulation that can occur under some conditions, as described in the next section.

E. Removal of Radon Progeny by Air Cleaning

1. Introduction

The radon decay products, unlike their inert radon parent, are chemically active and can attach to surfaces, such as airborne particles, room surfaces, and lung tissue. Not all of the progeny are attached to surfaces, and based on dosimetric modeling (reviewed in James et al., 1981), the alpha dose to the lung tissue from progeny deposited in the lungs and not attached to aerosols is 9 to 35 times the calculated dose from deposited progeny attached to aerosols.

A commonly used method of parameterizing the concentration of radon progeny in terms of the health risks due to their alpha decays is the potential alpha energy concentration (PAEC), which is given by:

$$\text{PAEC} = k_1A_1 + k_2A_2 + k_3A_3 \tag{3.6}$$

where the subscripts 1, 2, and to 3 refer to ^{218}Po, ^{214}Pb, and ^{214}Bi respectively. The coefficients, k_i, are k_1=0.00105, k_2=0.00516, and k_3=0.00380 for progeny concentrations, A_i, measured in pCi/l. (The very short half-life of ^{214}Po, 160 x 10^{-6} sec, makes the number concentration of this isotope negligibly small, thus it makes no significant contribution to the PAEC.) The PAEC is then given in units of working level (WL), where 1 WL is defined as any combination of radon progeny in one liter of air such that the ultimate decay to ^{210}Pb will result in 1.3 x 10^5 MeV of alpha energy. Another useful term is the equilibrium factor, F, which is a measure of the degree of equilibrium established between radon and its decay products:

$$F = 100\ \text{PAEC}/A_0, \tag{3.7}$$

where A_0 is the concentration of radon in pCi/l. As we noted above, radon progeny can attach to room surfaces, which effectively reduces the progeny concentration in the air. Thus, F is seldom close to unity, and is usually in the range of 0.4 to 0.7 for most typical indoor situations.

2. Radon Progeny Behavior

In order to understand the effect of air cleaning on radon progeny concentrations, it is necessary to understand the interactions of radon progeny with particles and other surfaces. Radon progeny behavior is illustrated in Figure 3.4, which shows the various attachment and removal processes. The associated rates are shown in parentheses. Other than the initial radioactive decay of radon, subsequent radioactive decays of the progeny are not shown. Radon progeny are chemically active elements, and can undergo chemical or physical reactions. Airborne progeny not attached to particles are thought to form metal oxides with molecular sizes on the order of 5 nm in diameter (see, for example, Busigin et al., 1981 and Knutson et al., 1983). As might be expected, attachment of radon progeny to airborne particles depends upon the concentration of the particles and their size distribution. Based on experiments conducted by Porstendoerfer and Mercer (1978), the mean attachment rate coefficient is 4.3×10^{-3} hr^{-1} x $(particles/cm^3)^{-1}$ for particles typically found in indoor air. The attachment rate is the product of this coefficient and the particle concentration. As noted in Figure 3.4, the alpha decay of attached ^{218}Po can impart sufficient momentum to detach the recoil nuclide from the particle. The detachment probability, r, has

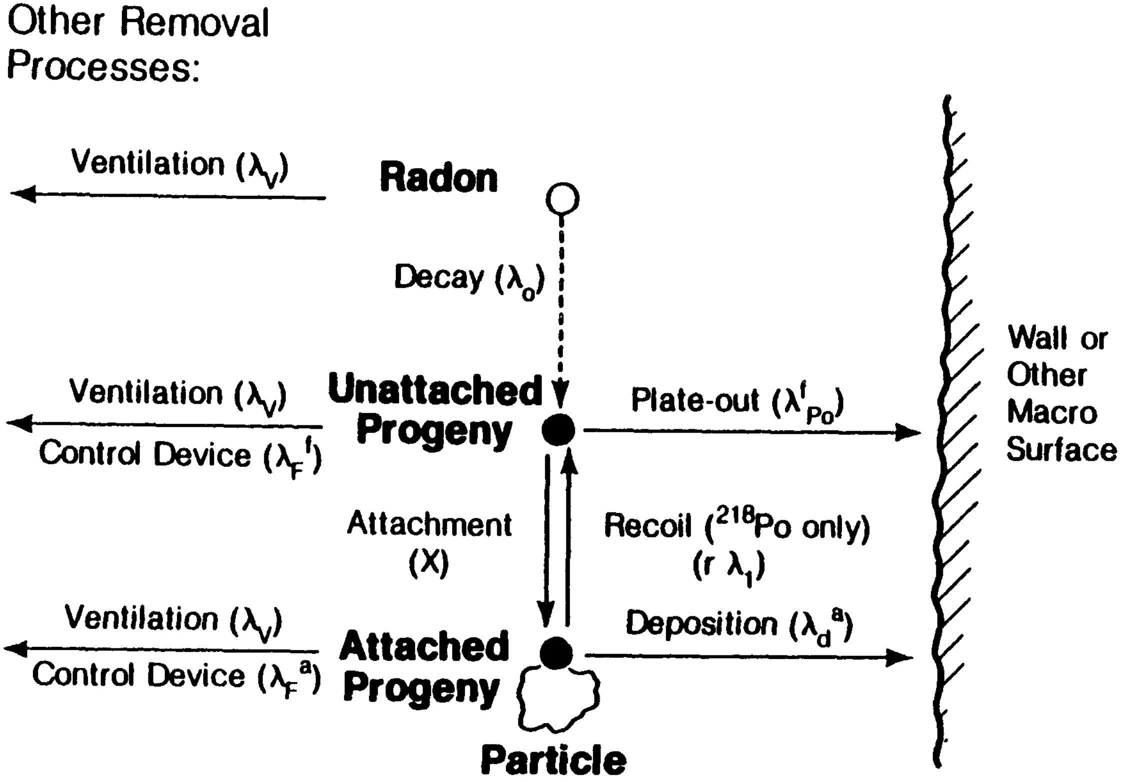

Figure 3.4 Schematic diagram of various decay and removal processes (and their associated rates) affecting concentrations of radon and radon progeny. The radioactive decay pathways for radon progeny are not explicitly noted in this diagram.

been estimated to be 0.83 (Mercer, 1976). The beta decay of other progeny produces less momentum and hence does not lead to detachment.

The other factors which affect the airborne concentrations of radon progeny are the rate of deposition of the attached and unattached species on walls or other surfaces. The number-weighted particle deposition rate (and therefore the deposition rate for attached progeny) is 0.16 hr^{-1}, based on recent observations in a full-size room (Offermann et al., 1984). In a similar set of experiments, the deposition rate for unattached progeny is estimated to be 15 hr^{-1}, almost a factor of one hundred greater than that for attached progeny (Offermann et al., 1984).

These various rates can be compared; at a particle concentration of 10,000 particles/cm^3, for example, the attachment rate to particles is 43 hr^{-1}, compared with a rate for deposition on surfaces for unattached progeny of 15 hr^{-1}. At lower particle concentrations, the attachment rate is correspondingly lower, and deposition of unattached progeny becomes a more important process. The deposition rates for unattached and attached progeny as a function of particle concentration are illustrated in Figure 3.5 for each radon decay product of interest.

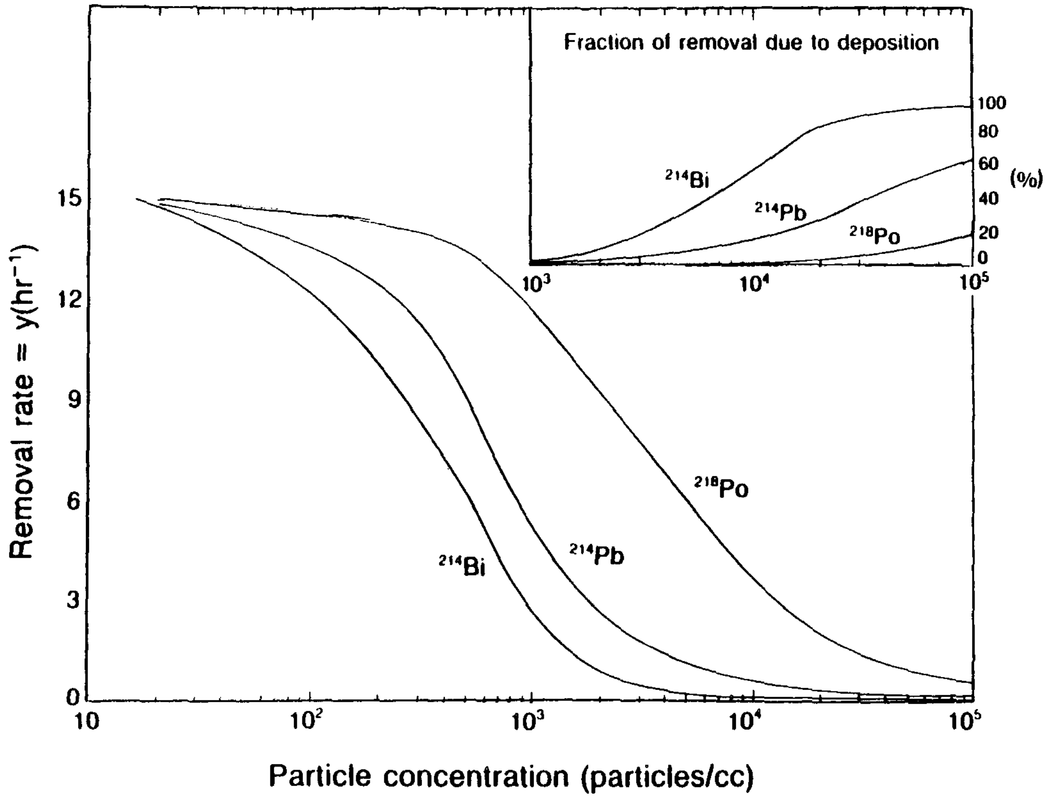

Figure 3.5 Removal rate of airborne radon decay products due to plate-out of unattached progeny and deposition of progeny attached to environmental aerosols. (Source: Offermann et al., 1984).

3. Removal by Air Cleaning

Air cleaning has two effects on airborne radon progeny. Air cleaning can directly remove attached progeny by removal of particulate matter. In addition, electrostatic precipitators and extended surface filters (such as HEPA-type filters) remove unattached progeny as well, although the performance of these devices in the particle size range below 10 nm is not well characterized (Offermann et al., 1984). Air cleaning also contributes to radon progeny removal by reducing the ambient particle concentration so that deposition of unattached progeny on indoor surfaces becomes a predominant removal mechanism.

The equilibrium factor, defined in Equation 3.7, is shown as a function of particle concentration in Figure 3.6. As can be seen, for a constant radon concentration, the total PAEC decreases as particle concentration is reduced. At the same time, the PAEC due to unattached progeny, indicated by the dashed line in the figure, increases as particles are removed (Sextro et al., 1984). Below ~500 particles/cm^3, the PAEC is almost entirely associated with unattached progeny. Since the lung dose associated with the alpha decay of unattached progeny is substantially greater than the dose expected from attached progeny, air cleaning may not significantly reduce the overall health risks of airborne radon progeny concentrations.

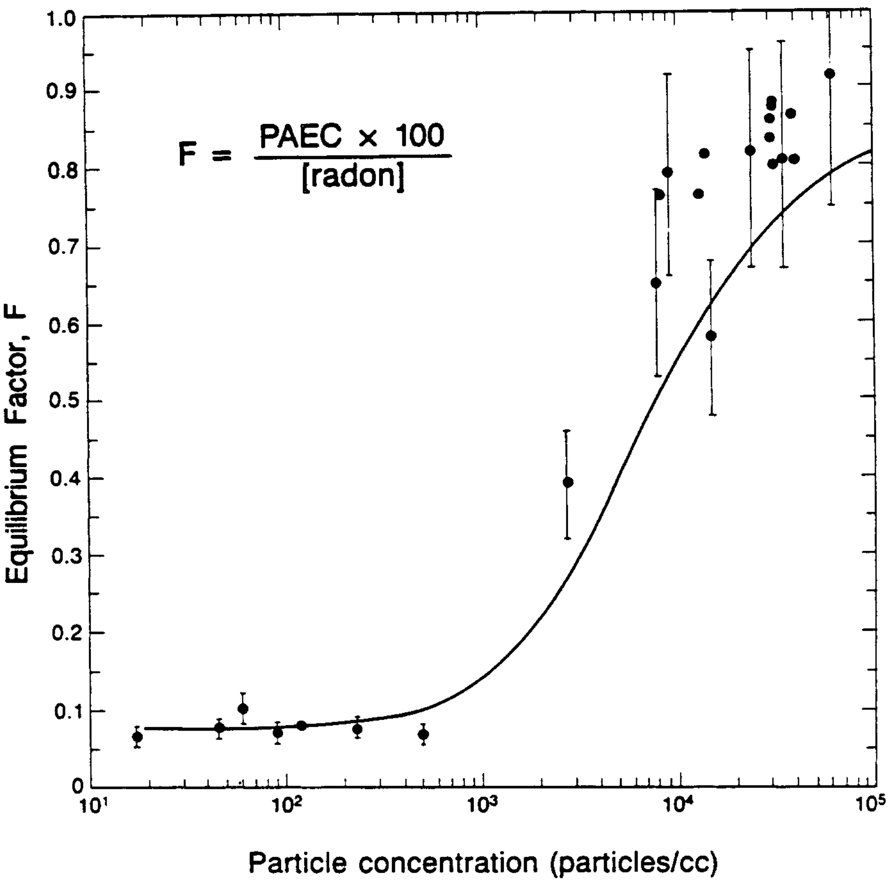

Figure 3.6 Equilibrium factor, F, versus particle concentration. Measured data and representative uncertainties are indicated by the solid circles and error bars. The solid line is based on calculated values for total radon progeny. (Source: Sextro et al., 1984).

4. Control of Radon and Radon Progeny: Literature Review

A. Background

1. Radon from Soil

The primary source of radon in the majority of U.S. homes with elevated radon concentrations is thought to be the radioactive decay of radium in soil located nearby to these homes (Nero and Nazaroff, 1983). This soil will, in general, be naturally occurring (i.e., indigenous), however, in some instances, this soil may consist of (or contain) waste products of mining or industrial processes (e.g., uranium mill tailings or refuse from phosphate mining) that may have an unusually high radium concentration.

An understanding of the locations and mechanisms of radon entry into buildings from the soil is critical for the development of many control strategies. Radon that originates in the soil and enters the indoor space must be transported through the soil to the interface between the soil and the building and also across or through this interface to indoors. One mechanism of radon transport is molecular diffusion, although diffusion is not thought to be sufficient to create highly elevated indoor radon concentrations (Nero and Nazaroff, 1983). Radon in soil gas or crawlspace air can also be transported by pressure-driven flow. This flow may be driven by changes in barometric pressure, wind-induced pressure, and the stack effect (Nazaroff and Nero, 1984). The stack effect, which is a primary driving force for infiltration, arises from the force of gravity and the difference between the density of the outdoor air and the heated indoor air. It causes depressurization at lower regions within the structure adjacent to locations where radon is likely to enter and, thus, provides a driving force for radon entry.

Radon transport and entry rates may be a function of several environmental parameters including indoor-outdoor temperature difference, rate of change in barometric pressure, and wind speed. The resistance to transport through the soil (which is a function of the soil permeability) and the resistance to transport through the interface between the soil and building may also strongly affect the radon entry rate. The permeability of the soil is a function of soil type and moisture content: soil saturated with water is less permeable; however, the soil beneath buildings may in some cases remain dry while the surrounding soil is saturated.

Locations where radon enters due to the pressure-driven flow are thought to include: cracks and joints in the foundation or in floors located above a crawlspace, penetrations for water and gas service and for drainage to sewage systems, and floor drains and sumps that are connected to subfloor drain tile systems (especially drains that are not occluded by a water trap).

2. Radon from Building Materials

Radium is present in trace amounts in earth-based building materials (e.g., concrete); thus, these materials can be a source of indoor radon. In most U.S. buildings, the building materials are a substantially smaller source of radon than the soil (Nero and Nazaroff, 1983). Building materials can be an important or dominant source of radon, however, if they are composed of materials with an unusually high radium concentration and if a significant portion of the radon produced within these materials enters the indoor space. In Sweden, alum shale (a naturally occurring material with a high radium content) was used as a component of concrete building materials in 350,000 to 700,000 dwellings between 1930 and 1975 (Swedjemark, 1978 1980). In the U.S., phosphate slag, which is high in radium was used as a component of concrete in an estimated 74,000 homes (Kahn et al., 1979).

3. Radon from Water

Substantial radon can be dissolved in ground water that is in contact with soil gas that has a high radon concentration. When this water is exposed to the indoor air, which has a much lower concentration of radon than the soil gas, radon can be released indoors. Water is usually not the predominant source of indoor radon, but it can be an important source in some buildings if the water has a highly elevated radon concentration. Radon concentrations in water vary by at least four orders of magnitude (Nero and Nazaroff, 1983).

4. Radon from Natural Gas

Because natural gas originates from below ground level it can contain significant amounts of radon and this radon can be released indoors during unvented combustion. The contribution to indoor radon from natural gas is considered to be minor (Johnson et al., 1973; Barton et al., 1973), thus, related control measures are not discussed in this report.

5. Overview of Control Techniques

A variety of techniques are potentially suitable for controlling indoor radon and/or radon progeny concentrations. The use of ventilation was discussed in a previous section. The radon source can be excluded or removed from close proximity to inhabited buildings. The transport of radon to indoors can be inhibited by sealing various openings or surfaces, or by subfloor ventilation. Buildings can be sited, designed, and/or constructed appropriately to reduce the potential for high indoor radon levels. Finally, radon daughters can be removed (i.e., cleansed) from the indoor air. Each of these techniques is discussed in following sections.

6. Sources of Data on Radon Control

A few sources of data on radon control techniques deserve specific mention. Data from a number of remedial action programs, designed to reduce radon exposures in particular communities or regions where indoor radon concentrations were frequently elevated, are the largest sources of information. These communities include: Grand Junction, Colorado where a large number of homes were built on uranium mill tailings (DOE, 1979); Port Hope, Ontario, where contamination of soil had occurred due to local radium processing (Case, 1980; Eaton, 1980); and Elliot Lake, Ontario, Bancroft, Ontario, and Uranium City, Saskatchewan which are located on porous sands in close proximity to radium-bearing bedrock (Eaton, 1980). There are also some reports of contamination from mining waste products in these last three cities. In addition, remedial techniques have been studied in a region of central Florida with natural phosphate deposits and waste products from phosphate mining (Scott and Findlay, 1983). These phosphate deposits are associated with elevated soil radium and indoor radon concentrations.

B. Source Removal, Exclusion, or Burial.

One strategy for controlling indoor radon concentrations is to prevent strong sources of radon from being in close proximity to the building. Several related control measures are discussed below.

1. Soil, Tailings, and Fill

As mentioned above, nonindigenous high-radium soils, mill tailings, and fill can be important sources of radon. An obvious approach is to monitor and plan the disposal of high-radium waste products so as to avoid siting of new structures on such materials. Even when buildings are not sited directly on disposal sites, the waste products (e.g., mill tailings) can be dispersed due to erosion and are, therefore, a potential source of contamination in nearby areas. Due to the enactment of a Uranium Mill Tailings Radiation Control Act in 1978, certain standards now must be met in regard to the disposal and treatment of mill tailings (Voorhees et al., 1983).

High radium soils or tailings can also be covered to reduce the radon emanation. Schiager (1974) calculated that radon emanation from mill tailings could be reduced by a factor of e (i.e., 2.72) with a 1.5 m cover of moist packed earth or a 5 cm cover of concrete. The possible impact of cracks developing in the concrete was not discussed.

For existing structures, an option is to remove nearby high-radium soils and to replace them with soils of lower radium content. This technique has been employed successfully in Port Hope, Ontario, (Case, 1980), and in Grand Junction, Colorado (DOE, 1979) where the primary sources of radon are nonindigenous materials. DOE (1979) gave an average cost per residence for fill removal of $3220. Fill removal has also been used in cases of contaminated fill in Bancroft, Ontario with costs estimated to be $2500 per residence for removal of material beneath the basement floor slab (James MacLaren, 1979). Keith Consulting (1979) reported that removal of waste material was effective in reducing radiation levels in three Uranium City structures but that costs were high -- $65 to $160 per cubic meter ($50 to $120 per cubic yard) for interior (i.e., beneath the floor) excavation and $33 per cubic meter ($25 per cubic yard) for backfill. Thus, it appears that fill removal is an effective control measure in cases of local contamination (in contrast to naturally occurring sources of radon), however, the costs for fill removal are very high. Advantages of this measure are its permanence and the lack of long-term energy or maintenance requirements.

2. Building Materials

An obvious technique to reduce future indoor radon exposures arising from high-radium building materials is to prohibit the use of such materials. According to Kahn et al. (1979), the states of Idaho and Montana have effectively prohibited future use of phosphate slag as a component of building materials and the E.P.A. is proposing similar prohibitions. Aerated concrete containing alum shale has not been produced in Sweden since 1975 (Swedjemark, 1980).

In existing buildings, the materials that are a significant source of radon could be removed, however, this option has received little attention perhaps because of the high costs involved. In addition, some of the measures described below are potentially suitable control options for buildings with high-radium building materials.

3. Water

When exhalation of radon from potable water is an important source of indoor radon, the simplest solution is to switch to a water supply with less dissolved radon; this option, however, will not always be available. Another potential solution is to aerate the water so that the radon is released to the outdoor air (James MacLaren, 1979); no data on this technique were found. A third possible solution is to store the water. The half life for radon is 3.8 days; thus, the radon concentration in the water could be reduced by approximately 85% by storage for 10 days. With a typical daily water consumption per person of 20 to 80 gallons (Partridge et al., 1979), significant storage capacity would be required. One final measure that is potentially helpful is to employ local exhaust ventilation near water vapor and radon sources (e.g., showers).

C. Site Selection

By not siting buildings in regions with known unusually high radon sources, such as areas with uranium mill tailings, other waste products noted earlier, or soil with a naturally high radium context, one can reduce excessive radon exposure. The uranium mining sites are often in the southwestern United States and Canada but smaller operations can be found elsewhere. Another important region where higher-than-average radon sources are present is an area of phosphate deposits and waste products from phosphate mining and processing in central Florida. The permeability of the soil may be an equally important consideration: soil gas is more readily transported to the building's substructure in highly permeable soil.

Data from airborne radiometric surveys of the entire United States and substantial regions of Canada taken for an assessment of uranium resources are a source of information on radon emanation rates from soil (Nero and Nazaroff, 1983; DSMA Atcon, 1983). Alternately, the radon release rate can be determined from local soil samples or by local gamma ray spectroscopy (DSMA Atcon, 1983). Combining these data on radon emanation rates with estimates of the local soil permeability (based on agricultural soil classification maps, soil grain size analysis, or in-situ permeability measurements) could indicate the potential for high radon concentrations at a building site or within a larger region. DSMA Atcon (1983) suggests development of a

"Radon Index Number" based on these data plus information on housing characteristics and climate. Such an index could provide a basis for site selection, and also indicate regions where specific attention could be given toward minimizing radon entry by appropriate building designs and construction practices.

D. Inhibiting Radon Entry

Various techniques can be used to inhibit the entry of radon into the indoor air; these techniques include: construction of a particular type of building substructure, application of sealants, and subfloor ventilation.

1. Building Substructures

The potential for high indoor radon concentrations may depend significantly on the design of the building substructure. Common substructural designs in the United States are the following: basement, concrete slab-on-grade, and wooden floor over a crawlspace, or some combination of these three designs. The potential for radon entry in houses with each substructure is discussed below.

Nazaroff and Nero (1984) suggested that structures with basements may be most susceptible to radon entry for two reasons: first, a large area of contact exists between the building and the soil. Second, they postulated that radon is drawn into the house from a larger volume of soil in houses with basements. It was suggested that the long flow-path between locations where air enters the soil and where soil gas may enter through the basement floor (and, consequently, the large resistance to flow per unit area of soil perpendicular to the flow), causes radon to be drawn from soil located at a greater horizontal distance from the outer walls in houses with basements compared to houses with other types of substructures.

Basement characteristics that may increase the susceptibility of radon entry are the presence of exposed soil or rock, unsealed penetrations for piping and other services, and the use of concrete block walls. (Concrete block walls often have gaps in the mortar joints, and soil gas can flow readily through the plenums; also, the block itself is composed of a fairly porous concrete.)

Substantial radon concentrations can also occur in housing with slab-on-grade foundations. Scott and Findlay (1983), in their investigation of radon control in houses location on Florida phosphate lands, describe various features typical of slab-on-grade foundations that allow radon entry; these features include the presence of numerous penetrations through the slab for plumbing and sanitary services, the use of concrete blocks below grade for walls, and the presence of a joint between the floor and foundation walls. A "monolithic slab" is described that contains no joints or connections to walls beneath grade level and that appeared significantly more resistant to radon entry based on a comparison to other houses sited on similar land. The combination of using a monolithic slab and sealing all penetrations through the slab (techniques of sealing are described later) appeared to be a highly effective radon control measure based on the data from a single residence. Scott and Findlay (1983) found, however, that training of the site foreman was not sufficient to ensure that radon entry paths would be sealed. They suggested that training of the construction crew and implementing an inspection program might be required to make construction of radon resistant slab-on-grade housing a routine practice.

The third substructure design is the crawl space which may be vented or unvented. Transport of radon from the soil into unvented crawl spaces can be enhanced by the stack effect and wind pressures (Nazaroff and Doyle, 1983). A significant fraction of the radon that enters both vented and unvented crawl spaces can leak into the living space (Nazaroff and Doyle, 1983), however, houses with vented crawl spaces are less susceptible to high radon concentrations. Insufficient data are available to determine the frequency at which elevated radon concentrations occur in houses with ventilated crawl spaces.

For buildings with all substructure designs, the integrity of the building substructure can have a large impact on indoor radon concentrations. As will become apparent in the next section, construction practices that minimize pathways for leakage through slab floors, basement walls, and floors located above crawlspaces are likely to be effective and efficient radon control measures.

2. Sealing Radon Entry Pathways

One of the most common radon control strategies employed in the various remedial action programs noted earlier is sealing the various paths through which radon enters the indoor air. There are two basic techniques: 1) selective sealing of cracks, floor joints, floor drains, penetrations for plumbing, and other openings through which radon enters by pressure-driven flow of soil gas or crawl-space air, and 2) sealing or coating the interior or exterior of entire surfaces (e.g., basement floor and wall). Because radon entry by pressure driven flow is thought to be the most common mechanism of radon entry in buildings with elevated concentrations, the first technique has been used more extensively. Coating of entire surfaces may be necessary only when radon enters at a substantial rate by transport (i.e., primarily diffusion) directly through building materials; one common example is the case of radon emanation from high-radium building materials. The following discussion of sealing techniques is based on a review of a large number of references (Chakravatti, 1979; Culot et al., 1973; DSMA ACRES, 1978, 1979, 1979b, 1979c, 1980; DSMA Atcon ,1983; Eichholz et al., 1980; Findlay, 1978, 1979; Findlay and Boychuck, 1980; Franklin et al., 1975; Hammon et al., 1975, 1977;

James Maclaren, 1979; Keith Consulting, 1979, 1979b, 1980; Kukacka and Isler, 1973; and Manley, 1979).

Sealing of Drains and Sumps: Perhaps the most commonly noted radon entry locations are floor drains and sumps that are connected to drainage- or weeping-tile systems in the soil beneath or surrounding the house. Radon can readily enter the building if there is not a functional water trap to isolate the tile systems from indoors. An often effective measure is, therefore, to rebuild the system so that it includes a water trap. In many instances water is directed into the trap at a slow rate to ensure that the trap remains full of water. James MacLaren (1979) estimated that it cost $500 to add a water trap to a floor drain and $1500 to add a water trap to and modify a sump.

Crack and Joint Sealing: Sealing of cracks, floor-to-wall joints, and other openings in basement walls or floors has often been performed in remedial action programs. For the sealant to adhere to the concrete, it is often necessary to cut or rout a notch into the concrete at the location of the crack to expose fresh surfaces. The crack is generally sealed with an epoxy, a non-shrink grout, or a flexible caulking that is protected by a cementitious material. Kukacka and Isler (1973) gave a cost of $2.17 per linear meter ($0.66/linear foot) of seal cracks. James MacLaren (1979) estimated that sealing of cracks and joints costs $2000 per house and Keith Consulting (1979) gave a cost of $800 to $1500 for houses with unfinished basements.

Sealing Exposed Rock or Soil: In some buildings there is a region of exposed rock or soil in the basement. To prevent radon entry, these exposed regions have been covered generally with a concrete patch that is sealed to the surrounding concrete.

Sealing of Plumbing and Other Service Penetrations: A variety of materials (e.g., grouts, epoxies, and asphalt roofing cement) have been used to seal penetrations in concrete through which various service conduits are placed. Some investigators have noted the importance of utilizing a slightly flexible sealant to allow for movement of the conduit relative to the concrete that is caused by thermal expansion and contraction.

Sealing Hollow Concrete Block Walls: Investigators have noted the difficulty in blocking radon entry through hollow concrete block walls, perhaps due to the fact that radon can enter through several paths, i.e., through cracks in the blocks or mortar joints, at the top of the wall through the plenums, and directly through the web of the block. The only technique frequently reported as successful in stopping the radon entry is to entirely fill the plenums with grout to above grade level, an expensive and labor-intensive procedure. Water from the grout sometimes leaks through cracks into the building interior and damages finished surfaces.

Sealing of Surfaces: Another method to inhibit radon entry is to seal or provide a barrier over entire surfaces. The barrier can be applied either to the interior or exterior of surfaces such as the faces of walls and the floor of a basement. Materials used for the barrier can be in the form of sheets or rolls; alternatively, a liquid that dries to form a solid can be painted or sprayed on surfaces. For the barrier to be effective, it must be sufficiently impermeable to radon diffusion and also be resistant to chemical or mechanical breakdown during use. Other considerations include the cost of the material, the cost for installation or application, and toxicity during application or in the case of accidental combustion.

A wide variety of materials have been considered for use as radon barriers. Many of the investigations have not been directed toward application in buildings, but, instead, toward reducing radon concentrations in mines or toward controlling the radon release from piles of uranium-mill tailings. In a recent review, Ketchum (1983) divided these materials into four general categories; 1) plastic polymers, 2) elastomeric polymers, 3) laminates and coated materials, and 4) coatings. Each category can be further subdivided; for example, coatings can be epoxies, urethanes, asphalts, etc. Based on the review by Ketchum (1983) and data from others, there is a large range in the performance of materials in preventing radon diffusion even within each category. A variety of materials are sufficiently impermeable to be useful. Depending on the barrier material and thickness, the estimated or laboratory-measured effectiveness of the barriers in preventing radon diffusion ranges from zero to 100%. (Data on the effectiveness of barriers in actual structures are discussed later.)

Kukacka and Isler (1973) listed costs of $1 to $2 per square foot for sealing of unfinished and finished basement floors, respectively. Guimond et al., (1979) estimated the cost of applying a polymeric sealant to be $600 to $1950 per structure. Application of an exterior sealant to basement walls (including excavation and refill of the soil) was estimated to cost $5500 by James MacLaren (1979) who also estimated costs of $6500 and $9500 for sealing the interior of basement floors and walls in unfinished and finished basements, respectively.

Effectiveness of Sealing Techniques: In many instances, the effectiveness of various techniques to inhibit radon entry by application of sealants is reported only in qualitative terms. Quantitative results are provided by DSMA ACRES (1979) and DSMA ACTON (1980) and reproduced here in Tables 4.1 - 4.6. The work described in these tables was performed in two stages. Stage I involved a visual inspection and closing of obvious radon entry routes along with closing of other visible but less obvious entry routes that were indicated to be important in tests (the tests were not described). If the estimated annual average PAEC for the house was still greater than 0.02 WL after

Table 4.1 Key to remedial actions performed at Elliot Lake, Ontario by DSMA Acton Ltd. and Acres Consulting Services Limited. This key applies for Tables 4.2 and 4.3.

Fix Number	Description	Number of times performed
1	Replace floor drain	60
1.1	Replace drained collection pit	9
2	Replace sump	12
2.1	Replace soaking pit	4
3	Close wall-floor joint	27
4	Close cracks and openings through poured concrete surfaces	31
5	Seal exterior walls	0
6	Cover exposed earth in crawl spaces	3
7	Cover exposed rock in basement	6
8	Coat masonry walls (interior)	1
9	Fill concrete block walls	1

Source: DSMA ACRES, 1979.

Table 4.2 Remedial actions performed and estimated annual average radon daughter concentrations in Elliot Lake, Ontario houses that complied with criterion (i.e., concentration less than 20 m WL) on December 31, 1978 after completion of Stage I remedial work. The key to the remedial actions is given in Table 4.1.

House Number	Contract Number	Fix Number											Estimated Annual Average (mWL)	
		1	1.1	2	2.1	3	4	5	6	7	8	9	before	after
1	0	x				x	x						29	7
7	0	x				x	x						29	11
10	2					x	x						61	11
12	0	x				x	x						62	7
17	2	x											24	7
18	0	x											78	10
19	0	x				x							54	2
20	2	x											31	7
22	0	x				x	x						25	13
23	1	x											50	8
25	1			x									33	12
27	1		x				x			x			44	8
31	0	x											D	7
32	0	x				x							35	7
34	0	x											D	7
39	0	x				x							28	7
40	1	x											D	3
44	1			x									31	9
45	3	x											24	13
52	2	x	x			x	x						39	12
53	0			x									D	3
55	1	x											21	6
57	1	x											33	6
65	1					x	x						38	14
66	2	x											21	9
70	0			x									32	5
72	1	x											37	4
73	3	x					x						21	15
80	1			x									29	8
83	0	x											D	4
87	2	x											D	9
89	2	x				x	x						24	8
91	0	x											D	4
92	2	x											34	9
94	0	x											26	5

Source:DSMA ACRES, 1979.

Table 4.2 (Continued)

House Number	Contract Number	Fix Number											Estimated Annual Average (mWL)	
		1	1.1	2	2.1	3	4	5	6	7	8	9	before	after
96	0	x			x	x							22	16
97	0	x					x						D	6
99	3	x					x						25	8
104	0			x									36	8
109	1	x											21	7
110	1	x											D	4
115	0	x											35	13
123	3	x											28	9
128	0				x								23	9
129	1	x											47	9
136	2	x											44	13
137	2	x					x						32	6
138	2	x											28	4
207	3	x											26	9
218	2	x											30	11
226	3	x											40	14
266	3	x			x		x						22	14
384	3	x					x						43	7
390	3	x											27	15
426	3			x			x						32	9
586	3	x											30	9
597	3						x						31	11

Total number of houses 57

Note: D indicates that the house was fixed as part of the remedial demonstration program. The annual average was believed to be greater than 20 mWL before the remedial work was carried out, but measurements were made over too short a period to properly estimate the annual average.

Source: DSMA ACRES, 1979 .

Table 4.3 Remedial actions performed and estimated annual average radon progeny concentrations in Elliot Lake, Ontario houses that complied with criterion (i.e., concentration less than 20 m WL) on December 31, 1978 after completion of Stage II remedial work. The key to the remedial actions is given in Table 4.1

House Number	Contract Numbers	1	1.1	2	2.1	3	4	5	6	7	8	9	Estimated Annual Average (mWL)	
													before	after
4	1/0					x	x			x	x		112	14
11	1/0	x	x			x			x	x			30	18
30	2/0	x				x	x						60	17
121	1/0	x					x						21	2
415	3/0	x				x							27	18
429	3/0			x		x	x			x			86	13
596	2/3/0	x				x	x						32	3

Total of houses 7

Source: DSMA ACRES, 1979 .

Table 4.4 Key to remedial actions performed at Elliot Lake, Ontario by DSMA Acton Ltd. and Acres Consulting Services Limited. This key applies for Tables 4.5 and 4.6.

Fix Number	Description
1	Water-trap weeping tile connected to floor drain
2	Water-trap weeping tile connected to sump
3	Close wall-floor joint
4	Close cracks and openings through poured concrete surfaces
5	Seal exterior surface of basement walls
6	Cover exposed earth in basements
7	Cover exposed rock in basements
8	Seal interior surface of basement walls
9	Fill concrete block walls with cement grout
10	Remove radioactive concrete or fill
11	Place shielding over active concrete
12	Install fan for improved ventilation

Source: DSMA Acton, 1980.

Table 4.5 Remedial actions performed and estimated annual average radon progeny concentrations for houses that complied with criterion (i.e., concentration less than 20 m WL) after Stage 1 remedial work during 1979. The key to remedial actions is given in Table 4.4.

House Number	Contract Number	Remedial Work (Fix number)	Estimated Annual Average (mWL)	
			Before	After
13	4	1,3	21	5
28	4	1	26	7
54	4	1,4	54	15
60	4	1,4	20	4
114	0	1	23	5
116	0	1	28	7
122	3	1,3,4	32	17
139	5	1,2,3,4,10	44	4
206	4	1	21	5
222	0	1	20	10
268	0	1,3,4	29	12
420	3	3,4	31	15
436	5	5	56	7
437	5	5	48	9
488	6	1,3,4,10	49	7
580	4	3,4,12	38	5
600	4	2,4	27	10
830	4	1	43	5
833	4	1	26	6
860	4	1	36	5
878	5	1	35	18
885	0	5	23	2

Total of Houses: 22

Source: DSMA Acton, 1980.

Table 4.6 Remedial actions performed and estimated annual average radon progeny concentrations in Elliot Lake, Ontario houses that complied with criterion (i.e., concentration less than 20 m WL) after Stage II remedial work during 1979. The key to the remedial actions is given in Table 4.4.

House Number	Contract Number	Remedial Work (fix number)	Estimated Annual Average (mWL) Before	After
14	1/4/5	1,2,3,4,7	53	13
29	0/5	3	94	5
35	2/0	3,10	24	13
38	0/0	2,4,9	41	7
43	4/4	2,4,9	21	13
50	0/1	1,3,4,7	43	15
64	1/2/3	1,3,7	32	4
67	0/3	1,2	23	11
81	0/0	1,3,4	35	11
88	1/2/3	1,3,4	36	3
120	0/2	2,4,9	41	8
413	0/0	2,4,9	45	8
427	3/0	3,4	29	16

Total of houses -	13
Total of houses complying in 1979 -	35
Total of houses complying to December 31 1979 -	98

Source: DSMA Acton, 1980.

completion of Stage I measures, further work was undertaken in a Stage II procedure to identify and seal additional radon entry pathways. Note that the tabulated data are only from houses where the measures were successful (i.e., the estimated annual average PAEC was reduced to below 0.02 WL).

An examination of the data in Tables 4.1 - 4.6 indicates that installation of water traps between drain tile systems and the sump or floor drain and repair or replacement of the sump can dramatically reduce the PAEC in many houses.

In most houses for which data is tabulated, crack and joint sealing was performed in conjunction with other measures; in the few houses where only cracks and joints were sealed, substantial reductions in PAEC were noted.

In three houses listed in Tables 4.1 - 4.6, the only measure performed was to apply an exterior surface sealant to the basement walls and the estimated average PAEC dropped by a factor of five or more. Additional data on the use of surface sealants in actual structures are provided by Eichholz et al. (1980) who applied several materials (i.e., a plaster compound, a cement/fiberglass mixture, several paints, and polyethylene sheeting) over sections of a high-radium concrete block wall. Measurements indicated a reduction in radon emanation rate from the coated wall sections that ranged from 18 to 97%.

The DSMA data (Tables 4.1 - 4.6) also show that a combination of sealing techniques is often effective for radon control. The various sealing techniques (in combination or separately) sometimes reduced the PAEC by greater than a factor of ten and frequently by more than a factor of three; similarly large reductions would generally be difficult to achieve by ventilation or air cleaning. As noted above, however, the tabulated data does not include instances where sealing techniques failed. Numerous failures are noted in the literature and generally attributed to radon entry through alternative (often unidentified) locations, failure of sealants to adhere to surfaces, or future cracking of surfaces. In several references, the need for quality workmanship when implementing the various sealing measure was emphasized. The long term effectiveness of the sealing-based measures has not been documented. Also, the available data are primarily from houses with basements, thus, the effectiveness of these sealing measures in houses with slab-on-grade foundations or crawl spaces is not extensively documented.

3. Subfloor Ventilation

Another technique to inhibit radon entry is to ventilate the subfloor volume (i.e., the region beneath the slab-floor or crawl space) by passive or active means. The objective of these measures is to transport radon that originates in the soil to outdoors and, thus, prevent its entry into the building.

Crawl-Space Ventilation: Crawl spaces are relatively easy to ventilate. Passive ventilation is accomplished by placing a number of openings in the crawl-space walls between the crawl space and outdoors; ventilation of the crawl space then depends on natural forces (i.e., wind and stack effects). A fan can also be installed to mechanically ventilate crawl spaces. The fan consumes electricity but eliminates the reliance on natural forces to provide the ventilation. Crawl-space ventilation is sometimes coupled with application of a sealant to the underside of the floor to inhibit radon entry into the house. In cold climates it may be necessary to insulate the floor and the heating system's ductwork if it is located in the crawl space (to reduce heat losses) and also the water and drainage piping (to prevent freezing) since increasing the rate of ventilation of the crawl space can reduce its temperature. A spray-on closed-cell polyurethane foam may have applications as both a sealant and an insulation.

As for other control measures, the estimated costs for crawl-space ventilation vary widely. Guimond et al., (1979) estimated that it cost $550 for crawl-space ventilation in planned structures. James MacLaren (1979) estimated a cost of $4500 per residence to increase the natural ventilation of existing crawl spaces and spray a closed-cell urethane foam insulation on the underside of the floor.

Quantitative data on the effectiveness of crawl-space ventilation are limited. Chakravatti (1979) presents data that indicated low radon daughter concentrations in newly built Elliot Lake apartment buildings with mechanically ventilated crawl spaces (ventilated at one air change per hour) but does not show data for the same buildings without ventilated crawl spaces. Nazaroff and Doyle (1983) measured substantially higher (i.e., 67% and 83% higher) indoor radon concentrations in two houses during a week with the crawl-space vents sealed compared to a week with the vents open. Several other references note that crawl-space ventilation is effective but present no data. It is logical that crawl-space ventilation would be an effective measure if the crawl space is ventilated at a substantial rate and clearly this can be accomplished using mechanical ventilation.

Sub-Slab Ventilation: Ventilation of the region beneath a slab floor is more difficult than ventilating a crawl space. This technique, though has been studied in some detail.

Two sub-slab ventilation system designs are described by Vivyurka (1979) and variations of these designs are described in other references. The "weeping tile system" was used in existing houses that had weeping (i.e., drainage) tile placed in the ground parallel to both the inner and outer edges of the basement wall footings with the tile systems draining into an open sump located in the basement. A fan with rated capacity of 24 l/s (50 ft^3/min) was installed in a box that was placed over the sump (but not sealed to the floor) and the exhaust from the fan was directed outdoors using a flexible duct. This system was generally very effective in reducing radon daughter concentrations as indicated by the data in Table 4.7 (Fan System No. 1). Only a few problems were noted with this system over a 28-month period in 63 homes (e.g., five fans had to be replaced and four fans were unplugged).

In the second design described by Vivyurka (1979) (which was installed primarily during house construction), an extensive tile system was placed beneath the basement floor and connected to a vertical ventilation stack that extended above the roof of the house. Indoor radon concentrations were maintained at low levels during the winter months, presumably because of the good draw through the exhaust stack during cold weather, but the system did not perform well during warm weather. The design was modified to incorporate a fan rated at 38 to 71 l/s at 75 Pascals (80 to 150 ft^3/min at 0.3 inches water gauge) that drew air from the tile system and exhausted it to outdoors. The fan was installed in a variety of locations (e.g., in the stack, in a sealed off section between the floor joists with an exhaust through the basement wall, and with the fan installed on an outside wall). This system was also generally effective as shown by the data in Table 4.7 (Fan Systems No. 2, 3, and 5). The primary problems noted with this system after 18 months of operation in over 160 houses were noise, and leakage into or out of the plenum between

Table 4.7 A summary of radon progeny concentrations with and without fan operation in Elliot Lake, Ontario houses with various subfloor ventilation systems and various fan locations. (Source: Vivyurka, 1979)

Fan System[a]	Without Fan				With Fan Operating				
	Number of Units Where Readings Taken	Highest Value Obtained to Date (WL)	Lowest Value Obtained to Date (WL)	Average of all Units (WL)	Number of Units Where Readings Taken	Highest Value Obtained to Date (WL)	Lowest Value Obtained to Date (WL)	Average of all Units (WL)	# of Units Below 0.02 (WL)
1	63	0.890	0.000	0.075	63	0.227	0.000	0.009	60
2	55	0.233	0.000	0.031	163	0.327	0.000	0.006	162
3	18	0.138	0.001	0.025	18	0.070	0.000	0.009	18
4	1	0.154	0.000	0.067	1	0.113	0.000	0.018	1
5	5	0.089	0.012	0.036	12	0.154	0.000	0.006	1
6	0	----	-----	----	3	0.021	0.001	0.004	3

[a] Key:

1. Fan installed in box located above sump. Sump is connected to a weeping tile system placed parallel in inner and outer edges of basement wall footing.

2. Fan draws air from an extensive tile system placed beneath basement floor. Fan is installed in plenum between floor joists.

3. Fan draws air from an extensive tile system placed beneath basement floor. Fan is installed in stack.

4. Fan installed in crawlspace.

5. Fan draws air from an extensive tile system placed beneath basement floor. Fan is installed outside of house.

6. Fan system - other (not specified).

the floor joists containing the fan; each of these problems motivated movement of the fan to outside the building. A few fans also had to be replaced.

A third method of sub-slab ventilation has been described briefly by OSMA ACTON (1980) and Findlay and Boychuck (1980) for houses with no underfloor tile system. This method consists of sealing a plastic pipe into a hole placed in the center of the basement floor and connecting it to a vertical stack with or without a fan. This system was described as effective in the one house for which it had been tried.

Few estimates have been given on the cost of a sub-slab ventilation system. James MacLaren (1979) estimated a cost of $8000 to install a system with a natural ventilation stack; this includes the cost to remove and reconstruct the floor slab. Obviously, costs would be much lower if sub-slab ventilation systems were installed during initial construction, in fact, installation of a sub-slab tiling system with access for connection to an exhaust stack or fan might be a worthwhile precautionary measure.

Effectiveness of Subfloor Ventilation: Data on the effectiveness of each technique of subfloor ventilation for controlling indoor radon and indoor daughter concentrations were described above; the data on crawl space ventilation are fairly limited. As a class, these techniques appear sufficiently effective so that reductions in concentrations by a large factor can be achieved. Because these techniques are not always reported as effective, measurements are generally required to determine performance in each instance.

E. Radon Progeny Control Through Air Circulation and Particle Control

The interactions among radon progeny, aerosols, and macro surfaces have been discussed in Section E of Chapter III. These principles provide the basis for understanding this discussion on control of radon progeny by air circulation and particle control. Some of the studies of radon progeny control have been done in conjunction with efforts to limit the exposures of uranium miners (and other underground hard-rock miners) to radon decay products (see, for example, Lawter et al., 1981). The results of these investigations have only limited residential applications.

1. Air Circulation

Air circulation in residences can be accomplished by either ducted or unducted fans. The effects of air circulation on radon progeny concentrations have been investigated in mines (Wrenn et al., 1969), in small experimental radon chambers (Holub et al., 1979; Bigu, 1983) and in room-size chambers or houses (Windham et al., 1978; Jonassen, 1980b; Rudnick et al., 1983; Offermann et al., 1984).

Overall, the results of these studies are consistent; the effects of air circulation on radon progeny concentrations, as measured by changes in the potential alpha energy concentration (PAEC), depend critically upon the particle concentrations. Most of these air circulation studies were conducted at particle concentrations below 10,000 particles/cm^3. Reductions in PAEC between a factor of 2 and 20 have been reported. The most substantial decrease in PAEC was observed by Wrenn et al. (1969), where the particle concentration in the mine was ∿200 particles/cm^3. The investigations by Offermann et al., (1984) were conducted at high particle concentrations and the slight decrease in radon progeny concentrations was within the limits of experimental uncertainty.

In some of these studies of air circulation, reductions in particle concentrations were reported which would increase the effectiveness of air circulation in reducing the PAEC. However, in some cases, this reduction in particle concentration appears to have been caused by an increase in ventilation rather than a substantial increase in particle deposition. This effect has been noted by Windham et al. (1978) and discussed by Scott and Findlay (1983). In studies of air circulation with a central fan and air conditioner, both the radon and particle concentrations dropped, indicating that leakage in the duct system drew in additional ventilation air. Wrenn et al. (1969) reported decreased particle concentrations as a result of air circulation in mine shafts; however, it is not clear from their results whether the circulation enhanced the deposition of particles on the walls, or whether the additional air movement induced additional ventilation of the mine shaft, which would decrease the particle concentration. Offermann et al. (1984) explicitly investigated the effect of air circulation on particle concentration by using oscillating fans positioned to direct the flows onto the walls of a room. No detectable decrease in particle concentration was observed; these tests were conducted at particle concentrations greater than 40,000 particles/cm^3.

Based on these studies, air circulation appears to significantly reduce the radon progeny concentrations when the particle concentration is low enough so that a significant fraction of the progeny are not attached to airborne particles. In Figure 3.5, presented in the previous discussion of interactions between radon progeny and particles, the deposition rate for radon progeny can be seen to increase substantially as particle concentrations decrease below a few thousand particles/cm^3. The air circulation enhances the deposition of unattached progeny by circulating the progeny close to the walls where diffusion to the wall surfaces occurs, thus increasing the probability of particle deposition. Thus, in order for air circulation to have a significant effect on progeny concentrations, the particle concentration should be below ~10,000 particles/cm^3, which is not often the case in typical residences (especially in homes with smokers) without enhanced particle removal.

2. Air Cleaning

The various types of air cleaners and their effects on particle concentrations have been discussed in Chapter III. A number of devices have been evaluated for their effect on radon progeny concentrations. These include electrostatic precipitators, mechanical filters, such as extended surface filters (HEPA-type), and negative ion generators.

Although air cleaner device performance as a function of particle size has been measured (Offermann et al., 1984), no direct measurement of the removal of unattached progeny by air cleaners has been reported. However, the collection efficiency for these small molecules, estimated to be ∿ 10 nm or less in diameter (Busigin et al., 1981) has been inferred to be the same as for attached progeny (i.e., particles) for both extended surface filters and electrostatic precipitators (Offermann et al., 1984).

The effect of electrostatic air cleaning on PAEC was investigated in several studies (Lloyd and Mercer 1981; Miles et al., 1980; Rudnick et al., 1983; Scott and Findlay, 1983; Offermann et al., 1984). The reduction in PAEC ranged from a factor of 2 to 20, usually accompanied by a similar decrease in particle concentration (although particle concentration was not reported in all studies). In one study, operation of electrostatic precipitators in a house produced a reduction in particle concentration of less than a factor of two (Carr and Tappan, 1976); it appears that the air cleaner did not have a large enough effective flow rate to substantially reduce the particle concentration in the house. Tappan (1979) also investigated the use of electrostatic air cleaners in a large commercial building. While similar results were obtained for the reduction in PAEC (60 to 80%), some mechanical and operational difficulties were encountered. Offermann et al. (1984) evaluated two stand-alone (unducted) electrostatic precipitators, both of which reduced the PAEC by a factor of 13. These experiments were done with initial particle concentrations greater than 40,000 particles/cm^3; operation of the air cleaning devices in a tight test chamber reduced these concentrations to less than 200 to 500 particles/cm^3.

Fan-filter devices have been used for radon progeny control experiments (Carr and Tappan, 1976; Lloydand Mercerm, 1881; Rudnick et al., 1983; Jonassen, 1980b; Offermann et al., 1984). It should be noted, however, that the terminology used to describe mechanical filters in various reports is not always precise or consistent. For high efficiency mechanical filters, such as a HEPA or other extended surface filter, the results are similar to those reported for electrostatic filters, *namely that particle concentrations are significantly* reduced, and radon progeny concentrations drop correspondingly. In addition to high efficiency filters, Offermann et al. (1984) evaluated inexpensive, table-top-size fan-filter units. These devices had no measurable effect on particle concentration, and, as expected, produced no observable reduction in radon progeny concentrations.

Negative ion generators, some of which had positively charged collection surfaces as part of the air cleaning system, were also examined (Lloyd and Mercer, 1981; Bigu, 1983; Offermann et al., 1984). All three studies report only modest reductions in radon progeny concentration. The largest reported decrease in PAEC, a factor of 4 to 7 (Bigu, 1983), is confounded somewhat since the experiment was done in a small, plexiglass chamber whose surface electrical properties don't appear to duplicate those found in residential buildings. As noted by Offermann et al. (1984), the use of negative ion generators for either particle or radon progeny control depends upon the flux of negative ions and the location of collection surfaces (which in many cases are the room walls themselves). One ionizer evaluated as part of this study had an integral, positively-charged collector; no reduction in either particle or radon progeny concentration was observed. An ionizer with a higher operating voltage and no collection surface (i.e., collection was done by the walls of the room) did reduce the particle concentration and the PAEC. The experiments reported by Lloyd and Mercer (1981) showed no effect in a room with only one ionizer. Reductions in particle and radon progeny concentrations were seen only with two or more ionizers, used along with positively-charged collection surfaces.

As with air circulation, the results of air cleaning as a method of radon progeny control are dependent upon the particle concentrations. Offermann et al., (1984) concluded that, while the air cleaner removes unattached and attached progeny at the same rate as particles, the dominant factor in the reduction in radon progeny concentrations at low particle concentrations is the deposition rate of unattached progeny on room surfaces. Thus, operation of an air cleaner has a two-fold effect; it removes both particles and radon progeny directly, and at the resulting low particle concentrations, the fraction of progeny not attached to particles increases significantly. These unattached progeny have a high deposition rate on room surfaces.

3. Implications of Air Circulation and Air Cleaning

Effective air cleaning devices are not inexpensive; prices start at a few hundred dollars for the device itself (either standalone, or designed for duct installation) plus the additional expense of installation of a ducted device. Devices costing less than $100-150 typically were not effective in removing particles (Offermann et al., 1984). Questions of device reliability and proper operation led Tappan (1979) to conclude that air cleaning as a reliable, fool-proof method of reducing radon progeny concentrations was less desirable than radon source reduction strategies. Air circulation by means of a ceiling or box fan would be less expensive (Rudnick et al., 1983), however, as has been emphasized here, air circulation will reduce radon progeny concentrations only at relatively low particle concentrations (but will not, by itself, substantially reduce particle concentrations).

As discussed briefly in a previous section, unattached radon progeny are predicted to have a greater alpha dose to lung tissue than progeny attached to particles. While air cleaning can reduce the PAEC, perhaps by as much as an order of magnitude, the overall health risks may not decline correspondingly due to the resulting high unattached fractions for radon progeny (Sextro et al., 1984; Jonassen and McLaughlin, 1982).

F. Summary of Radon/Radon Daughter Specific Control Measures

In summary, a variety of pollutant-specific measures can be employed to help control indoor concentrations of radon and/or radon daughters.

In the future, it is clearly advisable to avoid the use of high-radium substances as components of building construction materials. Similarly, buildings should not be sited on waste materials that are high in radium. For existing structures, removal of these high radium materials can be an effective control measure, but is generally very expensive.

Attention to the design of the building substructure is also relevant for radon control. Houses with basements may be more susceptible to radon entry, although available data are too limited for definitive conclusions. Since high-permeability pathways between the soil or crawl space and indoors are generally thought to be the primary routes for radon entry in houses with high radon concentrations, it would appear logical to design and construct new structures so as to minimize these pathways within practical constraints.

Techniques to control radon entry by sealing can be divided into two basic categories -- sealing of entire surfaces and selective sealing of cracks, floor drains, and other openings. If the predominant source of radon is the soil, and radon enters due to pressure-driven flow of soil gas or crawl space air, then the sealing of entire surfaces is generally not necessary. A possible exception is that of highly porous concrete blocks through which significant flow of high-radium soil gas may possibly occur; insufficient data exist at this time to assess this possibility. In situations where the primary mechanism of radon entry is diffusion, such as the case of high-radium building materials, sealing of entire surfaces would appear to be a better strategy. In remedial action programs, sealing techniques have met with mixed success. Although they have often been successful, problems have been noted, including the presence of alternative routes of radon entry and poor adherence between the seal and the surfaces to be sealed.

A number of ventilation-based techniques can be used to inhibit radon entry. These techniques include crawl-space ventilation and exhausting air to outside that is drawn from beneath slabs (e.g., beneath basement floors), or from sumps or drain tile systems. These techniques have received limited investigation but have often been effective especially when a fan is used to provide the ventilation. The need for a fan is less clear when considering crawl-space ventilation. As a better understanding is developed of the mechanisms and locations of radon entry, more effective techniques of sealing and subfloor ventilation may be developed and construction practices may also be modified to inhibit radon entry.

In regard to removal of radon progeny from the indoor air, the technique that appears most promising is the use of particle control devices to remove progeny and enhance deposition of progeny on surfaces. This technique can substantially reduce airborne progeny concentrations but substantial air-cleaning rates are required. It is difficult to achieve very large decreases in daughter concentrations (e.g., an order of magnitude) by this technique. In addition, this technique may increase the airborne concentration of radon progeny that are not attached to particles. According to some lung dosimetry models, a substantially higher dose results from unattached daughters compared to attached progeny, thus air cleaning may not yield a significant decrease in health risks from radon progeny.

The potential for high indoor radon concentrations may depend substantially on geographic or geologic features. An improved understanding of these dependencies could be valuable for the future design of radon control measures and could also indicate areas where specific attention should be given to radon control.

Because of the complexity of the radon problem, it is apparent that a combination of the strategies noted above (and possibly new strategies) will be required to control radon or radon progeny concentrations in the most efficient manner.

5. Control of Formaldehyde Concentrations: Literature Review

A. Formaldehyde Release

Meyer et al., (1979b) describe three primary sources for formaldehyde release from particleboard: 1) free formaldehyde that is attached to the wood or dissolved in the form of methylene glycol; 2) unreacted low molecular weight polymethylene and polymethylol compounds; and 3) hydrolysis of the urea-formaldehyde resins. Others generally provide a less detailed breakdown: release of free (i.e., weakly bound) formaldehyde and hydrolysis of the urea formaldehyde and (UF) resin. The same processes are responsible for formaldehyde release from urea-formaldehyde foam insulation (UFFI). The rate of formaldehyde release has been shown to increase with temperature, wood moisture content, humidity of the surrounding air, and with decreased formaldehyde concentration in the surrounding air (Berge et al., 1980; Andersen et al., 1975; Jewell, 1980; Long et al., 1979; Meyer et al., 1979b; and Matthews et al., 1983). In general, the formaldehyde emission rate decreases rapidly just after manufacturing and then more slowly over a period of months or years. Presumably, the initial release of formaldehyde is due in large part to the release of free formaldehyde (Myers, 1983).

B. Source Exclusion and Removal

An obvious technique to control indoor formaldehyde concentrations is to exclude or minimize the quantities (often called the loading) of formaldehyde-emitting material. It is clearly more practical to implement this strategy during construction than to remove the materials after construction. Even when formaldehyde-emitting materials are excluded during construction, they may be introducted later as components of furniture and other furnishings.

Matthews et al. (1983b) point out that removal of selected formaldehyde-emitting materials may have a limited impact on the indoor formaldehyde concentration if another significant formaldehyde source is present. The rate of formaldehyde emission from the remaining source can increase after the removal because of the inverse relationship between emission rate and indoor formaldehyde-concentration. Removal of UFFI from houses has been a common practice in Canada despite the high costs involved; however, data indicating the impact of this measure on indoor formaldehyde concentrations were not found. Instances of removal of pressed wood products have not been documented.

An example of the exclusion of formaldehyde sources is the ban on UFFI for use in homes and schools by the Consumer Product Safety Commission in August of 1982. This ban was overturned, however, by court actions in April of 1983. As a result of this ban, problems with formaldehyde release from UFFI, publicity, and lawsuits, the UFFI industry is essentially inactive at this time.

Formaldehyde-emitting materials are often desirable products because of their low cost and high performance (e.g., particleboard is highly resistant to warpage and cracking); thus, as an alternative to exclusion, techniques to reduce the emission of rates of formaldehyde from these materials are being developed as discussed below.

C. Source Modifications

One of the most promising techniques to control indoor formaldehyde concentrations is to modify the source materials to reduce the rates of formaldehyde emission. A number of such modifications are described below. Because of the variety of techniques used to assess formaldehyde emission rates, the measures reported by different investigators are difficult to compare and the impact of the measures on indoor formaldehyde concentrations can not be readily assessed from the data provided. Much of the data on the effectiveness of source modifications relate more directly to the initial release of free formaldehyde than to the longer-term release due to hydrolysis.

1. Alternative Resins

A variety of resins can be used in place of the traditional urea-formaldehyde (UF) resins. In general, the cost of alternative resins is higher then that of UF resins and in some cases the mechanical performance is inferior.

Roffael (1978) showed that phenol-formaldehyde-bonded particleboard and disocyanate-bonded board emitted approximately one-tenth as much formaldehyde as UF board. Sundin (1978) and Matthews et al. (1983b) indicated that plywood with phenol-formaldehyde resins emitted minimal formaldehyde.

2. Resin Modifications

Various materials can be added to UF resins to reduce formaldehyde emission. In many cases these additives (called scavengers) react with free formaldehyde and, thus, prevent its release.

Partial substitution of lignosulfonates, especially ammonium lignosulfonates for UF resin in the outer layers of the particleboard reduced formaldehyde emission substantially (Roffael, 1978). Hanetho (1978) showed that partial use of phenol-resorcinol resins could reduce formaldehyde release but indicated that they were too expensive for practical use. Brunner (1980) described the addition of isocyanate to the core layer of the particleboard and urea to the outer layers to reduce emission. Meyer (1979) cites literature on a large number of such additives in his text on UF resins. Meyer et al. (1979b) discuss the addition of sulfur, lignosulfonate, and dithionite to reduce formaldehyde emission from particleboard. Myers and Nagaoka (1981) observed a factor of ten reduction in formaldehyde emission after adding 10 percent urea and 5 percent melamine to a particleboard resin; in a later paper Myers (1982) shows that the susceptability of UF resins to hydrolysis is reduced by neutralizing acidic conditions (acids catalyze the curing of UF resins).

Additives can also be used to reduce the formaldehyde emission from UF foam (e.g., UFFI). Meyer (1979) states that UF foams can be modified with the same reagents used for particleboard as long as these reagents do not interfere with the foaming and cites literature on various additives.

It is known that UFFI sometimes degrades physically, i.e., becomes friable especially after exposures to high humidity and temperature (Allan et al., 1980; Meyer, 1979; Long et al., 1979). When the foam breaks down, formaldehyde emission increases. The reported cause of the degradation is often improper quality control (discussed later), however, Allan et al. (1980) show that increased acidity in the foam, which results from curing of the catalyst, reduces the foam's stability.

3. Molar Ratio

A number of studies have indicated that reducing the molar ratio of formaldehyde to urea can substantially reduce formaldehyde emission (Mikhaylova et al., 1974; Roffael, 1978; and Brunner, 1980). According to Meyer et al. (1979b), however, while low molar ratios reduce the amount of free formaldehyde and, thus, the initial rate of formaldehyde release, resins of low molar ratio are more vulnerable to formaldehyde release by hydrolysis.

The molar ratio also affects the physical properties of particleboards. Numerous studies have shown a drop in bond strength and greater swelling (when wetted) of particleboards with low molar ratios of formaldehyde to urea (see, for example, Brunner, 1980).

4. Processing Changes

UF resins, are generally cured by subjection to high temperature and pressure. Myers (1982), in a study of the hydrolyic stability of UF resins, showed that formaldehyde release from two different resins due to hydrolysis can be decreased by at least a factor of two by increasing curing time from 5 to 15 minutes. Others have suggested benefits from increasing the temperature or pressure during curing. Increasing curing time or temperature would, however, also increase the cost of production.

5. Coatings and Barriers

Another strategy to reduce formaldehyde emission from pressed wood products is to coat their surfaces. The coatings may contain scavengers that react with formaldehyde and/or the coatings may provide a barrier to impede the transport of moisture into the board (which hydrolizes the resin) or the transport of formaldehyde out of the board. Since the formaldehyde release from the edges of boards can be significant, coating of the edges may also be desired.

The use of coatings or barriers is discussed in several papers (Roffael, 1978; Mikhaylova et al., 1974; Sundin, 1978, 1982; Imura and Minemura, 1977; Meyer, 1979; and Matthews et al., 1983, 1983b).

Polyurethane, lacquer, and unspecified paints are reported to be moderately effective in reducing emission; performance can be improved with the application of several coats. Standard latex paints are indicated to be ineffective (Matthews et al., 1983b). Impregnation of the particleboard surface with various scavangers prior to the painting can substantially increase effectiveness; Imura and Minemura (1977) chose urea as the best impregnant considering cost and effectiveness in experiments with ten scavengers. Other coatings or barriers that are indicated to be effective to varying degrees include: melamine-impregnated paper, acid curing lacquers, decorative laminate, veneer, polyacrylamide, vinyl wall paper, vinyl carpet, vinyl floor tiling, and Falima F (a paint with formaldehyde-reactive chemicals (see Roffael, 1978)).

6. Aging

The rate of formaldehyde release from materials containing UF resins generally decreases rapidly after production and then more slowly after a period of years. Storage and aging of the product is, therefore, a technique to reduce the formaldehyde emission into buildings. Matthews et al. (1983b) show data from a study of 40 homes that indicates substantially lower formaldehyde concentrations in homes older than 5 years compared to concentrations in newer homes. However, it is possible that aging may not be as beneficial for products containing scavengers, since these scavengers may be consumed over time.

The emission rate from UFFI will also generally decrease over time. As mentioned above, however, some foams will deteriorate physically after time particularly when exposed to high temperature and humidity and this deterioration will be accompanied by an increase in formaldehyde emission. Concerns have also been expressed, particularly by those involved with the UFFI problem in Canada, that upon aging, particulates originating from the UFFI can enter the indoor air and act as irritants.

7. Dehumidification

Dehumidification is a potentially useful technique for reducing formaldehyde emissions from building materials. Andersen et al.'s (1975) test results of particleboard in a closed chamber at 22°C and a 0.5 ach ventilation rate showed that a decrease in relative humidity from 70 to 30% will drop formaldehyde levels by 50%; they derived a linear correlation between formaldehyde concentration and absolute humidity (i.e., not relative humidity). Berge et al. (1980) found a linear relationship between the formaldehyde concentration in a closed system (no ventilation) and relative humidity which indicates a 17% increase in formaldehyde concentration with a 10% increase in relative humidity. Formaldehyde release from UFFI is also a function of humidity (Long et al., 1979). No studies have been performed in actual homes to determine the effect of dehumidification on indoor formaldehyde concentrations. Also, it is possible that dehumidification may slow the rate of formaldehyde emission from materials but not decrease the total amount of formaldehyde emitted over time.

Dehumidification can be accomplished by employing residential dehumidifiers, by local ventilation near humidity sources (e.g., use of bathroom fans that exhaust to outside), or by providing ventilation with outdoor air when the outdoor air is less humid than indoor air. While dehumidification will not solve severe formaldehyde problems, it may be combined with other measures and, by itself, might be able to counteract small increases in indoor formaldehyde concentrations caused by infiltration reduction from house tightening.

8. Ammonia Fumigation

Formaldehyde-emitting materials can be exposed to an atmosphere containing ammonia to reduce emission rates. According to Walker (1975), formaldehyde and ammonia react to yield hexamethylenetetramine.

An industrial process for treating pressed wood products with an ammonia atmosphere was described by Simon (1980). His data indicate dramatic decreases in free formaldehyde content due to the fumigation process.

In-situ ammonia treatment has also been investigated and appears to be one of the more promising in-situ techniques. In a procedure described by Jewell (1980, 1981), ammonium hydroxide is placed in shallow pans in every major room of a residence, and the house is sealed for at least twelve hours while fans circulate the indoor air and the indoor temperature is maintained at 27°C or higher. Jewell used this technique in twelve mobile homes and reported 45 to 90 percent initial reductions in formaldehyde concentrations and 39 percent to 81 percent reductions in four homes monitored after a 40-to-60 week period. Trained technicians may be required for in-situ ammonia fumigation because the process can be hazardous if performed improperly or without proper safety equipment.

Gesser (1983) determined in laboratory experiments that ammonia fumigation can also reduce the formaldehyde emission rate from UFFI to one tenth or less of the initial value. Repeated increases in temperature or humidity reduced the effect of the ammonia treatment, but repeated treatment with ammonia may be a viable control technique.

9. Quality Control

Improving the quality control during the manufacturing of pressed wood products or the installation of UFFI is an important component of strategies to reduce formaldehyde emission rates.

Particularly with UFFI, poor quality control has often been cited as the cause of large formaldehyde emission rates. UFFI is produced by mixing at the installation site, a UF resin containing plasticizers, a surfactant (foaming agent) containing catalysts or hardeners, and air using a special foaming gun and injecting the mixture into wall cavities. Because there were thousands of installers, quality control was difficult to maintain. The following quality control procedures have been suggested to reduce formaldehyde problems associated with UFFI (Rossiter et al., 1977; Committee on Indoor Pollutants, 1981): installation only by trained personnel; no foam in attics or ceilings; no foams directly exposed to indoor or outdoor environment; proper maintenance of foaming equipment; resins or foaming agents older than the manufacturers' recommended shelf life should not be used; resins and foaming agents should be stored at proper temperatures; temperature of resin and foaming agent during mixing and temperature of wall cavaties should be within a specified range; no foaming at high humidities; no excessive formaldehyde in resin or excessive catalyst in foaming agent; proper ratio of resin to foaming agent; and appearance and setting time of foam should be checked immediately before application.

Production of pressed wood products is an even more complex process than production of foam and obviously, poor quality control during manufacturing could lead to problems. Matthews et al. (1983) described both improved monitoring technology for quality control and statistical considerations for quality control. Myers (1983) concluded that two standard techniques to determine formaldehyde emission rates could not be used as the basis for reliable product standards unless a large factor of safety was incorporated.

10. Source Modification Summary

It is clear that substantial reductions in formaldehyde emission rates can result from the various source modifications noted above. Two obvious questions are: 1) will source modification and quality control procedures be sufficient to solve the formaldehyde problem, and 2) will the appropriate source modifications be incorporated into manufacturing processes. With available data, conclusive answers to these questions cannot be determined; an important factor is the indoor concentration above which a problem is considered to exist. Some information relevant to these questions is provided by Hawthorne and Gammage (1982) and Matthews et al. (1983b).

Hawthorne and Gammage determined the formaldehyde emission rates from nine UFFI test panels fabricated in 1980 and aged for over one year including one panel that incorporated phenol in the resin and one with a formaldehyde scavenger. Substantial variations in emission rate were measured; the emission rates of the two panels noted above were approximately 54% and 76%, respectively, of the average emission rate and roughly at the median of the emission rates for the nine panels tested. Using a simple theoretical model and the average emission rate (measured under simulated indoor conditions), they estimated indoor concentrations of 0.05 to 0.25 ppm due to the presence of UFFI in houses with typical air exchange rates. Substantially higher or lower concentrations could result in actual use if the humidity or temperature differed from their test conditions (~25°C and 50% RH).

Matthews et al. (1983b) predicted the indoor formaldehyde concentration in a hypothetical room with various combinations of formaldehyde sources and with air exchange rates of 0.25 to 5.0 h^{-1}. The emission rate data used for the predictions was fairly recent (i.e., taken between 1981 and 1983). The predicted indoor concentrations ranged from 0.07 to 0.28 ppm with an air exchange rate of 0.5 h^{-1} and were as high as 0.38 ppm at a ventilation rate of 0.25 h^{-1}.

Thus, with substantial loadings of the types of UF-based building materials produced during the past few years, available data suggests that indoor formaldehyde concentrations could frequently and significantly exceed the most stringent guidelines for formaldehyde (i.e., 0.1 ppm). Because of reductions in formaldehyde emission rates, however, the indoor formaldehyde concentrations resulting from the use of many recently or presently available formaldehyde-emitting building materials should be lower than the concentrations observed in the past in buildings with similar loadings and ventilation rates. Further improvements in the formaldehyde-emission characteristics of these UF-based materials may continue in the future.

D. Sealing Wall Cavities

For the case of UFFI, which is installed in wall cavities, a potential control technique is to apply sealants to inhibit the transport of formaldehyde from the UFFI to indoors. Sealing of cracks, electrical outlet boxes, and other penetrations through walls has been one of the most commonly employed techniques by the Canadian UFFI Center to reduce formaldehyde concentrations in Canadian homes with

UFFI (Bowen et al., 1981). The effectiveness of this technique will depend on how successfully the sealing process can reduce formaldehyde entry and on the presence of alternative formaldehyde sources. Because sealing will reduce infiltration rates, it can increase the indoor concentration of pollutants emitted from sources other than UFFI.

E. Removal of Formaldehyde from Indoor Air

Formaldehyde removal from the indoor air can be either an active or passive process. Active air-circulation devices can process large volumes of air but the fans used for the air movement cause noise, require maintenance, consume electricity, and may be costly. Passive techniques (e.g., coating of walls with formaldehyde scavengers) rely on the natural movement of indoor air and, therefore, may require little maintenance; however, the effectiveness of passive techniques has received little study. Removal of formaldehyde by adsorption and absorption are discussed below.

1. Adsorption

The technique of air cleaning by adsorption was described in a previous section. Most studies of adsorption-based formaldehyde control have focussed on the use of chemical adsorbents. Recent studies of the use of adsorbents for formaldehyde control are reviewed below.

Several of these studies have focussed on the use of porous alumina impregnated with potassium permanganate available commercially under the tradename "Purafil".

Jewell (1980) built an air circulation/cleaning device containing 36 kg of Purafil that reduced formaldehyde levels in a mobile home from approximately 0.5 to 0.1 ppm for 130 days. The air circulation rate through the device was 730 l/s or 13.6 house volumes per hour. The cost for the Purafil was $112 (or $3/kg at 1980 prices) and the estimated total cost for the system was $442. The capacity (i.e., lifetime) of the Purafil under these conditions was not determined.

Eriksson et al. (1980) studied the removal of formaldehyde from air (with typical indoor formaldehyde concentrations) by Purafil and other adsorbents (activated carbon, activated aluminum oxide, treated calcium carbonate, activated carbon impregnated with urea and ammonium sulphate, activated aluminum oxide impregnated with urea and ammonium sulphate, and a ceramic material impregnated with potassium permanganate). They concluded that only Purafil and the ceramic material impregnated with potassium permanganate were promising for indoor formaldehyde control. The capacity of the Purafil was given as 4 g formaldehyde per kg Purafil based on field data and better than 16 g formaldehyde per kg based on laboratory data; this discrepancy indicated that gases other than formaldehyde may have consumed the Purafil during field use. In field tests both an active air cleaning device containing Purafil and a passive device, consisting of Purafil in a filter cloth suspended in the room air, substantially reduced formaldehyde concentrations (i.e., 17 to 72% reductions).

The Formaldehyde Institute sponsored an evaluation of adsorbents for formaldehyde by Arthur D. Little, Inc., (1981). In laboratory screening tests using higher inlet formaldehyde concentrations than typically found indoors, three adsorbents: Purafil, Calgon ASC charcoal (impregnated with copper, chromium, and silver oxides), and Barnebey-Cheney CI (proprietary modified charcoal) had a greater capacity for formaldehyde (-25 g formaldehyde per kg adsorbent at 50% formaldehyde removal efficiency) than the other adsorbents screened: Hopcalite (a copper oxide on support material); molecular sieve (3A); MSA KI charcoal (potassium iodide-impregnated); Amersorb XE-347 (an active carbonized polymer); and alumina (impregnated with urea, ammonium chloride, or chromia). Following the laboratory studies, a formaldehyde control device (estimated initial cost of $1000) with Purafil and/or Calgon ASC charcoal was fabricated and tested in a mobile home (Arthur D. Little, 1981b). Tests were conducted using each adsorbent independently and with charcoal upstream of the Purafil (- 8.6 kg of Purafil and 6.8 kg of charcoal at a total cost of approximately $100 or $6.5/kg). Approximately 270 l/s (575 ft^3/min) of air was circulated through the adsorbents using a 1500 watt (two-horsepower) blower and directed into the mobile home's furnace duct system. Initial formaldehyde concentrations of 1.3 to 1.8 ppm (with interior temperatures of 80 to 90°F) dropped to 0.2 to 0.3 ppm with initial operation of the control device, but after one to five days of operation, indoor formaldehyde concentrations returned to original levels. It is difficult to fully assess the performance of the control device because inlet and outlet formaldehyde concentrations were not generally measured and because the temperature in the mobile home, which greatly affects the formaldehyde emission rate, varied substantially. The formaldehyde removal efficiency of the control devices, based on a whole-house mass balance and a constant formaldehyde source strength dropped rapidly, i.e., from 44% to 5% over a 90-hour period. It was concluded that contaminants other than formaldehyde had reduced the Purafil's capacity for formaldehyde. Because of the high initial cost, the estimated cost of $100/month for adsorbent, and the high fan power requirement, the study was terminated.

One additional survey of adsorbents for formaldehyde control was conducted by Eian (1983). A granular carbon (Witco 965 12 x 20 mesh) was impregnated with various materials (30 total) by immersion in an impregnating bath and then dried. In addition, three commercially available adsorbents (Barnebey Cheney CI, Norton Formaldehyde Sorbent, and Purafil) were evaluated. Sorbent beds (61 cm^2 face area and 3.2 cm in depth) were challenged with an airstream (64 l/min, 27 to 30°C, 50% RH) containing 100 ppm formaldehyde. The "service life" of the adsorbent beds, defined as the time elapsed before 1 ppm formaldehyde was detected in the outlet air, ranged from 0 to 260 minutes. Significant service lifes (i.e., 49 minutes or greater) were measured for the following impregnates or materials: water-soluble non-volatile secondary amino alcohols; amino acid impregnants deposited as acid neutralized salts; amino sulfonic acids; methane sulfonamide; succinimide; aromatic ammonium salts; Norton Formaldehyde Sorbent; and Purafil. Data on the mass of adsorbents were not provided, thus, their capacity for formaldehyde cannot be determined precisely. By assuming an adsorbent bulk density of 0.75 g/cm^3, rough estimates of capacity at 99% removal efficiency can be made; these calculations yield a capacity of 2.6 g/kg for Purafil and 13 g/kg for the adsorbent with the greatest service life (activated charcoal impregnated with sulfamic acid sodium salt).

This data indicates that available chemical adsorbents can efficiently remove formaldehyde from air but they have a limited capacity for formaldehyde. Capacities of 0 to 25 grams formaldehyde per kilogram adsorbent are indicated for adsorbent beds in laboratory studies. Some field studies have indicated large reductions in capacity compared to laboratory results and this reduction has been attributed to loading of the adsorbents by gases other than formaldehyde. The amount of formaldehyde removal necessary per year to have a significant impact on the indoor concentration is generally at least a few hundred grams. For example, one air change per hour of ventilation in a 340-m^3 house with an indoor concentration of 0.1 ppm removes approximately 360 g/year of formaldehyde; removal rates of 370 to 8300 g/year are indicated in the Arthur D. Little (198a) report as necessary in mobile homes with highly elevated (1-3 ppm) formaldehyde concentrations. For an adsorbent with a field capacity for formaldehyde of 5 g/kg, a cost of $6/kg, and a required yearly removal rate of 400 g, the total cost of adsorbent per year would be $600. In many cases greater quantities of adsorbent would be required and additional costs incurred for the equipment and operation of a fan. Unless the data reviewed here is greatly in error (for example the Purafil product literature indicates a "field corrected" capacity of 150 g/kg), it is clear that adsorbents with a significantly greater capacity of formaldehyde or a lower cost are required for adsorption to be a practical formaldehyde control technique.

2. Absorption

Absorption air cleaning, also known as gas scrubbing or air washing, is commonly employed to remove contaminants from industrial waste-gas streams. In an air washing process the pollutant is removed from the air by dissolution in a liquid. The literature includes several investigations of air washing for the removal of formaldehyde from concentrated waste-gas streams; however, since these studies were concerned with industrial applications, it is not known if the processes studied would be suitable for indoor use.

One study of air washing for indoor formaldehyde control has been completed and a second investigation is in progress. The completed study, by Pedersen and Fisk (1984) included laboratory tests of two full-scale air washers, derivation of a mathematical model of an air washer, and a comparison of air washing and ventilation as indoor formaldehyde control techniques. The experimental air washers employed water as the washing solvent; a portion of the washing liquid was continuously replaced with fresh water. A refrigeration system was used to prevent evaporation of the water and resultant humidification of the indoor air. The air washers were tested with constant inlet formaldehyde concentrations between 0.07 - 0.4 ppm. Air flow rates through the devices were 100 - 160 l/s. The formaldehyde removal efficiencies (defined as the fraction of inlet formaldehyde that was removed from the airstream) of the two air washers were 0.30 - 0.63. The water requirements were 0.5 - 8.0 l/h and the power requirement for an air washer with a 140 l/s air flow rate was estimated to be 1500 - 1800 W (including the refrigeration cycle).

The study also found that the energy required to operate an air washer in a residence could be less than the energy required to remove an equivalent amount of formaldehyde by ventilation (even when the ventilation system incorporated heat recovery). This energy advantage results because air washer operation does not increase the heating load of a building by increasing the air exchange rate; in fact, air washer operation reduces the load on the heating system because the power consumed by the air washer is rejected to the indoor space. The energy advantage of air washing over ventilation is limited, however, to residences located in cold climates which use expensive forms of energy for heating.

An additional component of the study by Pedersen and Fisk (1984) was the measurement of formaldehyde removal by air dehumidifiers which condense water vapor on the cold evaporator coil of a refrigeration system. As with an air washer, a condensate-coated coil provides air-to-water contact and, thus, formaldehyde may be dissolved into the water. Two dehumidifiers were tested: one had a simple coil consisting of a coiled tube; the second had a coil that was finned to provide additional surface area. The tests showed that the finned coil had a moderate formaldehyde removal rate when the air supplied had a high humidity. However, when the humidity was low, such as is often the case in a building located in a cold climate, very little condensate was present so little formaldehyde was removed from the air. The unfinned coil with a small surface area had little formaldehyde removal capability, even when the supply air humidity was high. The authors suggested that an air conditioner operating where the air humidity was high would remove formaldehyde at a moderate rate since condensate would be present on the cold coil of the air conditioner.

Since it is known that indoor formaldehyde source strengths will increase as the indoor concentration is reduced, the authors concluded that effective air washers will often require even higher air flow rates and formaldehyde removal efficiencies than the experimental units tested. Testing of an air washer with a higher air flow rate is the goal of a study underway by the Saskatchewan Research Council in Canada. Water will also be employed as the washing liquid in this air washer.

F. Summary of Formaldehyde-Specific Control Measures

A variety of measures can be employed to help maintain acceptable indoor formaldehyde concentrations. It is clearly desirable to minimize the rate of formaldehyde emission into the indoor air. One effective technique is to exclude or minimize the quantities of formaldehyde-emitting materials in the building envelope and furnishings. Because these materials are often desirable, however, techniques to reduce formaldehyde emission rates have been developed. The use of alternative resins, additives to the traditional UF resins, changing the ratio of formaldehyde to urea in the resins and changes in the curing process can greatly reduce formaldehyde emission rates. Coating or impregnating the surfaces of particleboards with materials that scavenge free formaldehyde or that impede transport of formaldehyde and moisture through the board's surface is also an effective measure. The emission rates from pressed-wood products can also be reduced by aging and fumigation with ammonia prior to installation. Attention to quality control during manufacturing of pressed wood products and particularly during the installation of UFFI appears to be a crucial component of efforts to reduce formaldehyde emission rates. It is difficult to assess the impact of these source control measures on indoor formaldehyde concentrations. Available emission rate data suggest that with substantial loadings of the types of UF-based materials available in 1981-1983, indoor formaldehyde concentrations could frequently and significantly exceed the most stringent guidelines (i.e., 0.1 ppm). Manufacturers have reduced formaldehyde emissions rates, however, and further reductions may occur in the future.

In existing houses with elevated formaldehyde concentrations, fewer source-control options are available. Coating of surfaces may be effective based on available laboratory data but the preferred coatings which scavenge free formaldehyde are not readily available to homeowners. Dehumidification may cause moderate reductions in indoor concentration. Although the usefulness of ammonia fumigation for reducing the emission rate of formaldehyde from installed UFFI has not been established, in-situ ammonia fumigation deserves special attention based on studies of its effectiveness in mobile homes and because of its low cost.

Another control strategy is to remove the formaldehyde from the indoor air. In laboratory studies, it has been shown that numerous chemical adsorbents can remove formaldehyde from air. Only a few of these adsorbents have been evaluated in actual field studies. The data reviewed indicates that available adsorbents have limited capacities for formaldehyde and the capacities appear to be reduced in the field perhaps because the adsorbents are loaded by materials other than formaldehyde. The frequent replacement of adsorbents that is indicated as necessary from the available data would be very expensive; however, other adsorbents not yet developed or fully evaluated may prove to be more practical in the future.

Formaldehyde can also be effectively removed from air by air washing with water (i.e., absorption of formaldehyde into water). Because water is inexpensive, it can readily be disposed of and replaced with fresh water - a great advantage compared to adsorption. One of the drawbacks of current air washing strategies is that a refrigeration system is required to prevent humidification of the indoor air and this system consumes a significant amount of electrical energy. In addition, air washing equipment is likely to be fairly expensive and this technique has not yet been evaluated in the field. Because of the coupling between formaldehyde emission rate and indoor concentration, large amounts of air will generally have to be cleansed of formaldehyde (when either adsorption or absorption techniques are employed) to have a large impact on the indoor concentration.

As with the indoor radon problem, it is apparent that a combination of measures are required for efficient control of indoor formaldehyde concentrations.

6. Control of Gaseous Combustion Products: Literature Review

A wide range of combustion products can be emitted by indoor combustion appliances and tobacco smoking; carbon monoxide, carbon dioxide, nitric oxide, nitrogen dioxide, sulfur dioxide, water vapor, formaldehyde, and respirable particles. This chapter will concentrate on control of nitrogen dioxide, carbon monoxide, carbon dioxide, and sulfur dioxide. The other pollutants are either treated elsewhere in the document or are not considered to be major problems at this time.

Combustion pollutants are generated by clearly defined sources. This is an obvious advantage when attempting to control the concentrations of these pollutants since source removal can always be considered as an unambiguous first step. The discussion presented below assumes that this possibility has been considered and, for an appropriate reason, has been rejected.

A. Nitrogen Dioxide

Nitrogen dioxide, NO_2, is produced from nitrogen and oxygen present in the air in any process that exceeds a temperature of approximately 1800 K (Mitchell et al., 1975). These temperatures are seen in the flames of gas and kerosene combustion appliances; they are not seen in appliances using electrical resistance elements for heating. Source replacement, e.g., substituting electrical for gas or kerosene heaters, is therefore an appropriate technique for reducing NO_2 levels indoors.

Modifications to gas appliances are currently being studied to reduce average NO_2 concentrations indoors. The open pilot of a gas appliance has a small combustion rate. However, since it burns continuously and emits pollutants at a higher rate than the primary burners in an appliance (Macriss and Elkins, 1977), it makes a significant contribution to the total average pollutant concentration in a space. Pollutant concentrations will be reduced if the standing pilots are replaced by other types of ignition devices.

A second major area of investigation is burner modification. Mitchell et al. (1975) explored the use of wire meshes in the flame region to conduct heat away from the hottest portions of the flame and reduce the production of NO_2. This procedure must be done carefully, for if too much heat is extracted from the flame region, the carbon monoxide burnout reactions are quenched. (Carbon monoxide is primarily a product of incomplete combustion and its emission rate will increase if the flame is cooled too much.) Indeed, one mesh configuration that was tried reduced NO_2 emissions 70 percent but increased CO emissions 7500 percent. The authors were able to demonstrate a double cylinder configuration for the mesh that cooled the flame in stages, reducing the NO_x emissions by approximately 62% while allowing the CO burnout to proceed further to completion; data on the reduction in NO_2

emissions were not given. DeWerth and Sterbik (1983) conducted experiments to evaluate various configurations of steel-ring inserts in a range-top burner and demonstrated a 26% reduction in NO_2 emission without a dramatic increase in CO emission. Their experiments illustrated that "judicious" placement of the rings was necessary. The report by DeWerth and Sterbik (1983) also includes a review of other burner modifications; however, the use of ring-inserts appeared to be the most promising. This basic strategy of burner inserts has been adopted recently in a modification introduced by the gas industry to meet a new California Standard for controlling NO_2 emissions from furnaces and water heaters (Gas Research Institute, 1983).

In the past, the ovens within gas ranges were vented to the outside. While it is unlikely that ventilation ducts will once again be connected to modern ranges, the potential for improved utilization of the local ventilation provided by range hoods is an important strategy for reducing indoor NO_2 concentrations. Several research efforts (Macriss and Elkins, 1977; Traynor et al. 1982; and Revzan, 1984) have quantitatively demonstrated the substantial improvements in air quality when a range hood is used. The performance of range hoods was discussed in a previous section on local exhaust ventilation.

The final control strategy for NO_2 that we shall discuss is the deliberate use of its reactivity to control its concentration within a space. As several researchers (Wade et al., 1975; Moschandreas and

Stark, 1978; Yamanaka et al., 1979; and Traynor et al., 1982) have noted, nitrogen dioxide concentrations decay in an indoor space by processes other than just ventilation. The reactions that occur are not completely understood. The rates of NO_2 removal by reaction compare to rates of removal by ventilation in residences; the tabulated reaction constants for NO_2 in Table 2.1 indicate removal rates from 0.16 to greater than 1.29 h^{-1}. Understanding the mechanisms of this removal process and relating it to materials typically found within buildings is an area of current interest.

B. Carbon Monoxide

Another combustion pollutant with potentially serious health effects is carbon monoxide. Many of the comments made above for nitrogen dioxide apply equally well to this pollutant.

Since indoor CO concentrations may be the result of indoor combustion processes, source exclusion or the replacement of combustion sources with their appropriate electric counterpart is a first control step. An important CO source that is not a major NO_2 source is tobacco smoking. To emphasize the importance of smoking as a CO source, Jaeger (1981) points out the numerical similarity between the concentration of CO contained in automobile exhaust and that contained in mainstream cigarette smoke. Exclusion of this source of CO could result from providing no-smoking zones in buildings or from complete elimination of smoking in the building.

If source elimination is not possible, modification of the combustion processes to reduce CO generation can be considered. At this point, however, one encounters difficulty because of the complexity of the combustion process. Many strategies that reduce the emission of CO (i.e., techniques to increase the temperature of the flame) will tend to increase the NO_2 emission rate. Therefore, modifications must be evaluated carefully to determine their effect on emissions of all pollutants and, in particular, CO and NO_2.

Carbon monoxide is also a major exhaust pollutant from internal combustion engines. Therefore, it is essential that garages and loading docks be isolated from occupied areas of a building. A particular concern arises when a building is constructed above a parking garage. Ventilation systems and air leakage paths must be monitored carefully to assure that CO and other pollutants from the vehicles are not transported into the occupied space.

Johnson et al. (1975) studied excessive CO levels in enclosed ice arenas and discovered that the dominant source was the exhaust from periodically used ice-resurfacing machines. They reported success in controlling CO levels in these buildings by monitoring concentrations regularly, increasing ventilation rates, and installing catalytic emission control devices on the resurfacing machines.

Studies of Koepsel (1976) and Leaderer et al. (1981) have shown that CO is not removed from indoor air by processes other than ventilation. Thus, the major control strategies for a non-reactive pollutant such as carbon monoxide are source removal, ventilation, and source modification.

C. Carbon Dioxide

Carbon dioxide found within buildings has three primary sources: the CO_2 background concentration present in the outside air, the CO_2 from respiration processes of the building occupants, and the CO_2 from combustion processes within the space. Carbon dioxide is a primary combustion product. Measurements of the CO_2 emission rates from unvented combustion heaters give values that are commonly within several percent of the theoretical emission rates based upon fuel consumption. These results are independent of the tuning of the device. Figure 6.1 illustrates the stability of the CO_2 emission process in a plot of the steady-state concentration in a test house versus ventilation rate (Traynor et al., 1983). The solid line illustrates the inverse relationship expected between steady-state concentrations and ventilation rates if the emission rate of the source is constant and if ventilation is the only pollutant removal process.

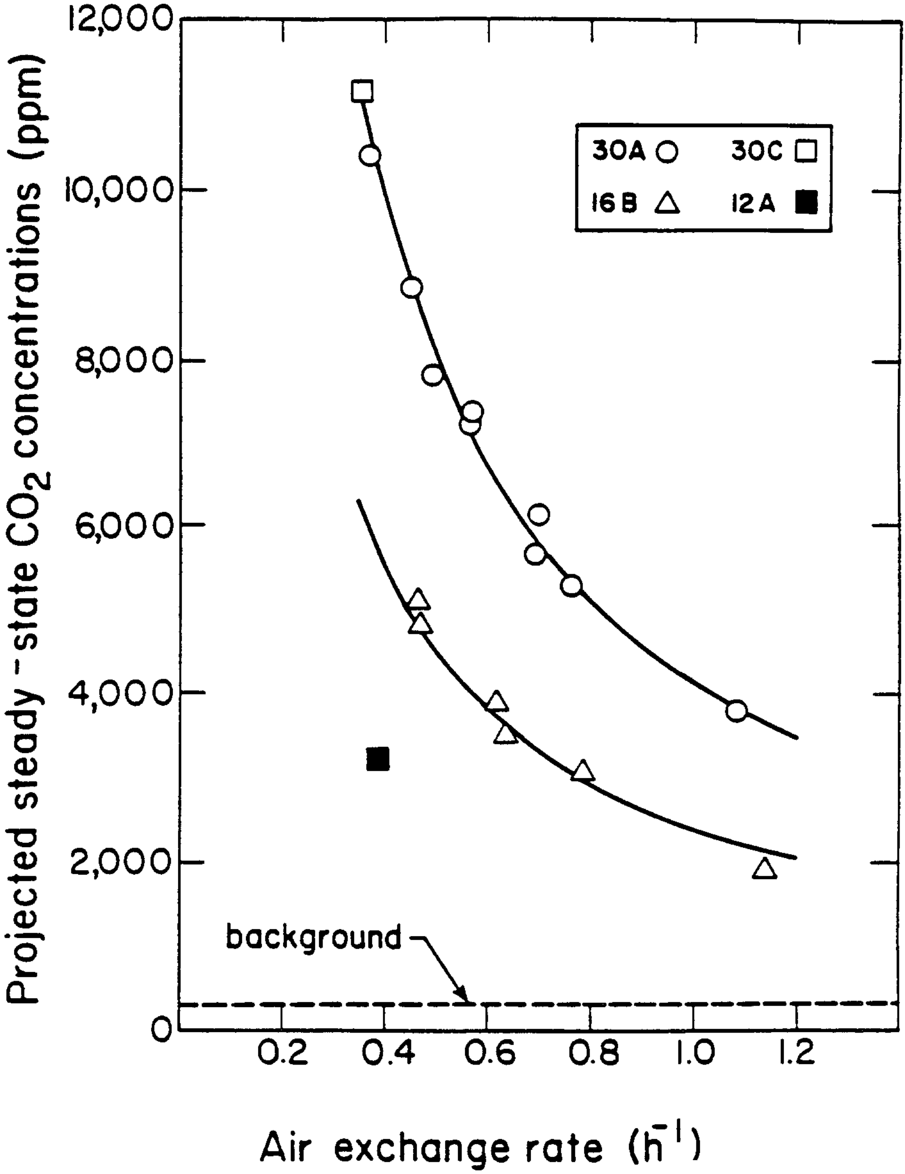

Figure 6.1 Projected steady-state CO_2 concentrations associated with the use of unvented gas-fired space heaters versus the air exchange rate of the house. Tests were conducted in a 240-m^3 unoccupied research house. The solid lines are empirical fits of the data to the reciprocal of the air exchange rate for the 30A/30C and 16B heaters. (Source: Traynor, et al, 1983).

This figure is also important for understanding the possibilities for controlling excess concentrations of CO_2 from unvented combustion appliances. As discussed above, source exclusion or removal is the first possibility to consider. If such a step is not possible, one must move to source modification. However, source modification takes on a different meaning when dealing with CO_2. The only possible modification to the building-combustion-device system that would reduce CO_2 emissions is a modification that would increase the thermal efficiency of the building shell and therefore reduce the amount of fuel required to produce the same thermal comfort within the space. This, to be sure, applies to all the combustion-generated pollutants we are considering -- however, it is the only modification that will affect CO_2 production from unvented combustion in the space.

Since CO_2 is non-reactive, control of excess CO_2 concentrations in the air is best done with ventilation. When this is not possible (e.g., in a submarine) air scrubbers using monoethanolamine have been employed (Carhart and Thompson, 1975).

D. Sulfur Dioxide

Sulfur dioxide is produced by combustion processes using fuels with a high sulfur content. The wide-spread adoption of unvented kerosene space heaters, the increasing use of wood burning stoves, and the high sulfur content of some of the fuels used in these appliances means that sulfur dioxide can be a significant indoor as well as outdoor pollutant (Leaderer, 1981; Traynor et al., 1982c).

Control of sulfur dioxide concentrations indoors entails either source elimination, a topic that has been discussed above, or fuel substitution. Often the sulfur content of a fuel is not known. Consumers can, if they are careful, find fuel sources to burn that are low in sulfur and thereby reduce sulfur dioxide concentrations indoors.

Sulfur dioxide is a widely studied outdoor pollutant. It was recognized early that indoor concentrations of SO_2 are substantially lower than outdoor levels. Several authors (Spedding and Rowlands, 1970; Spedding, 1970, 1972; Spedding et al., 1971; and Walsh et al., 1977) have studied the sorption of SO_2 by typical indoor surfaces. Deposition velocities and saturation effects have been measured for wallpapers, paints, and carpeting materials. Increasing this natural removal of SO_2, perhaps by air circulation or inclusion of specific surface materials in the indoor space, may also be helpful for controlling indoor SO_2 concentrations, however, this type of technique is not likely to be as effective or practical as source elimination or fuel substitution.

E. Reducing Leakage of Combustion Products to Indoors

Vented combustion appliances can be a significant indoor source of combustion products if there is leakage of these products into the indoor air. An obvious control measure under these circumstances is to reduce or eliminate the leakage. This may involve replacement of a cracked heat exchanger in a furnace or various improvements to a vent system. Substitution of air-tight wood stoves for models that are not air tight, or use of non-air-tight stoves that do not leak combustion products into the indoor air are other possible measures. Reducing the episodic leakage of pollutants into the indoor air during warm up, loading of fuel, or stoking of wood stoves or fireplaces may be more difficult; techniques that increase the draft during these operations may be helpful.

F. Summary

Techniques to control the indoor concentrations of combustion products fall into the following categories: 1) source removal or exclusion, 2) source modification, 3) ventilation, and 4) enhancing the removal of indoor pollutants by sorption on surfaces. Of these techniques, source removal or exclusion is the simplest and most effective, but the initial and operating costs and performance of the substitute products must be considered. Substantial success has been reported in modifying burner designs to reduce nitrogen dioxide emission rates. It appears difficult to reduce nitrogen dioxide emissions dramatically without increasing carbon monoxide emissions, but in some instances the reduction in nitrogen dioxide emissions may be more important. Ventilation is an effective method of reducing indoor concentrations of combustion products. Local exhaust ventilation at the pollutant source deserves particular attention because it is effective and energy efficient. Also, if the source is periodic, the local ventilation system need only be operated during times of pollutant emission. The last potential control measure described here, increasing the sorption of combustion products on indoor surfaces offers some potential for the control of nitrogen dioxide and sulfur dioxide concentrations; however, the effectiveness and practicality of this strategy has not been demonstrated.

7. Control of Respirable Particles: Literature Review

A. Source Exclusion or Reduction

A major indoor source of respirable particulate matter is tobacco smoke. Other indoor sources include unvented combustion appliances such as gas stoves, ovens, and space-heaters, as well as wood stoves, fireplaces, pets, cooking and cleaning activities, and general indoor traffic. As with any contaminant, exclusion of the indoor source is an effective and often economical means of reducing indoor concentrations.

1. Tobacco Smoke

Tobacco smoking is a major source of indoor respirable particles. The sidestream emissions of tobacco smoke are estimated to be as much as 30 to 40 mg per cigarette (Committee on Indoor Pollutants, 1981). The most effective strategy for reducing exposure from tobacco smoke contaminants is to exclude smoking from the indoor environment. When a complete ban on smoking is not acceptable, segregation of smokers and non-smokers may be effective in reducing non-smoker exposure. The effectiveness of this strategy varies depending on, among other things, the degree of mixing between the segregated zones and the configuration of the ventilation system. Measurements conducted by Repace and Lowrey (1982) indicate significant concentration differences between smoking and non-smoking sections. The indoor concentrations of respirable particulate matter were observed to be 43 and 60 percent lower in the non-smoking sections of two cafeterias and 50 percent lower in the non-smoking section of a sandwich shop.

2. Unvented Gas-Fired Appliances

The emission of respirable particulate matter from natural gas-fired appliances as reported by Girman et al. (1982, 1983) are 4.6 mg/hr per burner for a gas stove, 0.1 mg/hr for a gas oven, and 0.2 to 3.2 mg/hr for gas-fired unvented space heaters. Thus, the particulate emissions from unvented gas-fired appliances are small on a mass basis compared to cigarettes but may still be significant on a number basis (e.g., for radon progeny considerations). Electric appliances can be used as particulate-free alternatives to unvented gas-fired appliances.

3. Other Sources of Particles

Particle emissions from sources, such as wood stoves and fireplaces, vary widely, depending upon their design and operation. Reduction of particle emissions from these sources may be accomplished by structural changes, such as a wood stove insert in a fireplace or by replacing fireplace screens with a tempered glass door, or by operational changes. Some of these methods are discussed in the previous section on gaseous contaminant removal.

The necessity for source control of particle emissions associated with occupant activities, such as cooking or cleaning, will depend upon the source strength. Local ventilation, such as use of a range hood ducted to the outside, may often provide the simplest control method.

B. Removal of Respirable Particles from Indoor Air

The working principles of particulate air cleaning were described in a previous section. As discussed, particles in the size range of 0.1 to 0.5 μm are the most difficult to remove because they are too small for interception or impaction to be effective removal processes and too big for diffusion to be effective. Particles in this size range have a high probability of penetrating to the pulmonary compartment of the lung where adverse health effects are most likely to occur and also constitute a significant mass-fraction of indoor respirable particle matter.

Offermann et al. (1984) examined the performance of ten portable air cleaners. The experiments were conducted in a room-sized chamber (36 m^3) using tobacco smoke as the test aerosol. The two parameters used to characterize the performance of the air cleaners in this study are the effective cleaning rate (ECR) and the system efficiency. The

ECR is calculated as the difference in particle decay rates observed with and without operation of the air cleaner multiplied by the test chamber volume. Thus the ECR is the effective flow rate of particle-free air that would produce the observed reduction in particle concentration. It provides a measure of how fast the device cleans the air. The system efficiency is calculated as the ECR divided by the air flow rate through the device and thus provides information on how efficient a particular device is in removing a certain size particle. The small panel filters tested had low effective cleaning rates, between 0 and 3.3 l/s (0 - 7 cfm). The low particle removal rates of these devices are attributed to the low air flow rates and the large fraction of the air that bypasses the filters. The particle removal efficiencies ranged from 0 to 40 percent. The most efficient devices tested were the two extended-surface filters, these devices were 86 and 100 percent efficient for removal of cigarette smoke particles and had ECRs of 27 and 85 l/s (57 and 180 cfm), respectively. The two electrostatic precipitators tested were about 58 percent efficient and had ECR's of ~ 57 l/s (~120 cfm). In addition, the effects of forced-air circulation were investigated, and no measurable reduction in particulate concentration was observed.

McNall (1975) examined the performance of an electrostatic precipitator installed into the forced-air system of a 425-m^3 residence. In one test, where smoking was at the rate of 35 cigarettes per hour, the observed steady-state concentration with the precipitator running was more than a factor of three below the predicted steady-state concentration without the precipitator energized. This corresponds to a particulate removal efficiency of approximately 40 percent based on the reported flow rate through the precipitator of 350 l/s (740 cfm). In a similar manner, a particulate removal efficiency of 60 percent may be inferred from an experiment conducted at a smoking rate of 12 cigarettes per hour.

Silverman and Dennis (1956) conducted extensive tests with a portable electrostatic precipitator. Removal efficiencies for tobacco smoke ranged from 85 to 90 percent as determined by weight. In tests conducted with ragweed pollen, Ambrosia trifida, which has a count-median diameter of 20 μm, the removal efficiency was observed to be greater than 99.9 percent as determined by count. A removal efficiency of 98 percent as determined by count was observed for tests conducted with a bacteria, M. (Micrococcus) Candidus, which has a count-median diameter of 0.5-0.7 μm. The effect of humidity on collection efficiency was also examined. In the range of 40 to 60 percent relative humidity, no appreciable change in collection efficiency was detected. However, particle removal efficiency was observed to decrease from 99 to 90 percent as determined by repeated dust-spot efficiency tests made over a two-week period. Ozone concentrations were observed to range from 3-20 ppb in the outlet airstream of the precipitator which is at the threshold of odor perception but well below the National Ambient Air Quality Standard (NAAQS) of 120 ppb for outdoor air.

In a series of tests conducted by Decker et al. (1963), the efficiency of various air cleaning devices was examined for removal of biological particles (e.g., bacteria 1 - 5 μm diameter). Collection efficiencies ranged from 10% for coarse-fibered panel filters to 60 - 90% for electrostatic precipitators to 90 - 100% for high efficiency extended-surface (HEPA-type) filters. In a similar series of experiments conducted by Harstad et al. (1967), HEPA-type filters were shown to be nearly 100% effective at removing the submicron virus, T1 bacteriophage, which has a mass-median diameter of 0.17 μm.

The effect of negative ion-generators on indoor concentrations of respirable particulate matter has been the focus of a number of studies. In tests conducted by Offermann et al. (1984), a residential negative ion-generator with an integral charged collection surface failed to remove tobacco smoke from a 36-m^3 chamber unless circulating fans were turned on and even then the removal rate was very modest, with an ECR of 5 l/s (10 cfm). In tests with a higher-voltage commercial negative ion-generator an ECR of 14 l/s (30 cfm) was observed. Repace et al. (1983), have reported tests of negative ion-generators conducted with cigarette smoke in unventilated rooms. The effective cleaning rates as calculated from these data range from 25 to 38 l/s (54 to 80 cfm). In a test of a commercial multi-unit ion-generation system conducted in a 333-m^3 unventilated church hall, an effective cleaning rate of 147 l/s (311 cfm) was observed. Thus it appears that modest particulate removal rates may be achieved by operating negative ion-generators.

Another factor which deserves consideration is the effect of particle charging on deposition in the human respiratory system. In experiments conducted by Melandri et al. (1983), total respiratory deposition has been observed to increase linearly with an increase in the number of charges per particle. Further studies are needed to determine what effect ionizers have on the dose of respirable particles to humans.

C. Summary

In addition to ventilation, which was discussed in a previous section, two basic strategies were identified for the control of respirable particles. The first strategy to consider is source exclusion or reduction because of its effectiveness and the lack of associated long term energy or maintenance costs.

The second basic strategy is removal of the respirable particles from the indoor air. There is a wide variation in the abilities of various types of air cleaning devices to remove respirable particulate matter. Residential furnace filters and panel filters such as those used in some portable air cleaners are not effective in removing respirable-size particles. Particles in this size range are especially difficult to remove because of inherent aerodynamic considerations. Electrostatic precipitators have been demonstrated to be 50 to 90% efficient in removing a wide variety of respirable size particles, while extended-surface area filters may remove from 70 to 99.99% of these particles. As with any contaminant removal system, the location of the device inlet with respect to the contaminant source is an important factor that can significantly affect the reduction in indoor particle concentrations. We also note that with regards to tobacco smoke, removal of the particulate phase does not eliminate the odor and irritant effects of tobacco smoke, many of which result from exposure to the gas-phase contaminants produced.

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