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Chapter 3
Indoor air quality and health

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Abstract

During the last two decades there has been increasing concern within the scientific community over the effects of indoor air quality on health. Changes in building design devised to improve energy efficiency have meant that modern homes and offices are frequently more airtight than older structures. Furthermore, advances in construction technology have caused a much greater use of synthetic building materials. Whilst these improvements have led to more comfortable buildings with lower running costs, they also provide indoor environments in which contaminants are readily produced and may build up to much higher concentrations than are found outside. This article reviews our current understanding of the relationship between indoor air pollution and health. Indoor pollutants can emanate from a range of sources. The health impacts from indoor exposure to combustion products from heating, cooking, and the smoking of tobacco are examined. Also discussed are the symptoms associated with pollutants emitted from building materials. Of particular importance might be substances known as volatile organic compounds (VOCs), which arise from sources including paints, varnishes, solvents, and preservatives. Furthermore, if the structure of a building begins to deteriorate, exposure to asbestos may be an important risk factor for the chronic respiratory disease mesothelioma. The health effects of inhaled biological particles can be significant, as a large variety of biological materials are present in indoor environments. Their role in inducing illness through immune mechanisms, infectious processes, and direct toxicity is considered. Outdoor sources can be the main contributors to indoor concentrations of some contaminants. Of particular significance is Radon, the radioactive gas that arises from outside, yet only presents a serious health risk when found inside buildings. Radon and its decay products are now recognised as important indoor pollutants, and their effects are explored. This review also considers the phenomenon that has become known as Sick Building Syndrome (SBS), where the occupants of certain affected buildings repeatedly describe a complex range of vague and often subjective health complaints. These are often attributed to poor

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air quality. However, many cases of SBS provide a valuable insight into the problems faced by investigators attempting to establish causality. We know much less about the health risks from indoor air pollution than we do about those attributable to the contamination of outdoor air. This imbalance must be redressed by the provision of adequate funding, and the development of a strong commitment to action within both the public and private sectors. It is clear that meeting the challenges and resolving the uncertainties associated with air quality problems in the indoor environment will be a considerable undertaking.

1. Introduction

Until recently, the health effects of indoor air pollution have received relatively little attention from the scientific community. Prior to the 1970s, problems with indoor air quality in residences and the non-industrial workplace were occasionally investigated, but the level of interest was low (Stolwijk, 1992). Even today, the bulk of public concern continues to be directed at the health impacts of outdoor pollution. Numerous studies suggest that most members of the public perceive the risks from poor quality outdoor air as being substantially higher than those from indoor contamination (LHEA, 1997). These perceptions are formed despite the fact that, in developed societies, many people pass most of their time indoors. For example, a recent investigation of time budgets amongst US residents found that, on average, individuals spent 88% of their day inside buildings, and 7% in a vehicle. Only 5% of participants'time was actually spent outside (Robinson and Nelson, 1995). Whilst this does not per se mean that indoor exposures will produce more harmful health effects, the evidence is that indoor concentrations of many pollutants are often higher than those typically encountered outside.

A preoccupation with the relationship between outdoor air quality and health is understandable. Particularly in more economically developed and rapidly developing countries, recent and unprecedented changes in lifestyles and environmental quality have meant that an increasing number of people are exposed to the contaminants of urban air (Lipfert, 1997). There is certainly evidence that, especially amongst more vulnerable members of society, outdoor pollutants may pose a real health risk (Burr, 1995). Furthermore, the frequently invisible nature of indoor air quality problems must be contrasted with the often only too obvious photochemical and particulate substances that characterise outdoor contamination.

Historically, problems with indoor air were unquestionably much more apparent than they are today. Soot found on the ceilings of prehistoric caves provides ample evidence of the high levels of pollution that was associated with inadequate ventilation of open fires (Spengler and Sexton, 1983). Whilst chimneys first began to appear in European homes in the late twelfth century, most

large medieval houses still had a central hearth in the great hall, ventilated by a louvre in the roof. It was only during the sixteenth century that chimney stacks and fireplaces against walls came into general use (Brimblecombe, 1987; Burr, 1997). The blackened roof timbers in many buildings that predate these innovations bear testimony to the severe pollution problems that their inhabitants faced.

Today, concern over the health effects of poor quality indoor air is increasing. Despite the fact that the vast majority of buildings exhibit no immediately apparent problems, a wide spectrum of symptoms and illnesses are attributed to non-industrial indoor air pollution (Redlich et al., 1997). Indeed, problems associated with shortcomings in indoor air may be one of the most common environmental health issues most doctors now face (Seltzer, 1995).

Over recent decades, there have been many changes in the way buildings are constructed and operated. To some extent, modifications in building design have been driven by the need for increased energy efficiency, largely brought about by higher fuel costs since the 1970s oil crisis (Jones, 1998). Modern homes and offices are much better insulated than was previously the case. Houses with ventilation rates as low as 0.2 to 0.3 air exchanges h^{-1} are now widespread (Platts-Mills et al., 1996). In older properties, particularly those with open fireplaces, ventilation rates above 1 air exchange h^{-1} are more common. Improved insulation has been accompanied by numerous other modifications to the management of indoor environments, and advances in construction technology have led to a much greater use of synthetic building materials (D'Amato et al., 1994). All these changes have undoubtedly meant that buildings are more comfortable. However, they have also provided an environment in which airborne contaminants are readily produced and may build up to substantially higher concentrations than are typically encountered outside (Teichman, 1995).

Indoor air pollutants emanate from a range of sources (Table 1). They are emitted by the fabric of buildings, and may also be a by-product of the activities that are undertaken within them. Sources can be broadly classified as being associated with the activities of building occupants and other biological sources, the combustion of substances for heating or fuel, and emissions from building materials. For some contaminants, infiltration from outside, either through water, air, or soil, can also be a significant source.

The concentration of a pollutant indoors depends on the relationship between the volume of air contained in the indoor space, the rate of production or release of the pollutant, the rate of removal of the pollutant from the air via reaction or settling, the rate of air exchange with the outside atmosphere, and the outdoor pollutant concentration (Maroni et al., 1995). However, actual human exposures are often difficult to quantify. This is largely because the behaviour and activity patterns of individuals can strongly affect their levels of

Table 1. Major indoor pollutants and emission sources[a]

Pollutant	Major emission sources
Allergens	House dust, domestic animals, insects
Asbestos	Fire retardant materials, insulation
Carbon dioxide	Metabolic activity, combustion activities, motor vehicles in garages
Carbon monoxide	Fuel burning, boilers, stoves, gas or kerosene heaters, tobacco smoke
Formaldehyde	Particleboard, insulation, furnishings
Micro-organisms	People, animals, plants, air conditioning systems
Nitrogen dioxide	Outdoor air, fuel burning, motor vehicles in garages
Organic substances	Adhesives, solvents, building materials, volatilisation, combustion, paints, tobacco smoke
Ozone	Photochemical reactions
Particles	Re-suspension, tobacco smoke, combustion products
Polycyclic aromatic hydrocarbons	Fuel combustion, tobacco smoke
Pollens	Outdoor air, trees, grass, weeds, plants
Radon	Soil, building construction materials (concrete, stone)
Fungal spores	Soil, plants, foodstuffs, internal surfaces
Sulphur dioxide	Outdoor air, fuel combustion

[a]Taken from Spengler and Sexton (1983).

exposure (Harrison, 1997). Results from the TEAM (Total Exposure Assessment Methodology) studies undertaken by the US Environmental Protection Agency (EPA) during the 1980s consistently show that personal exposures to many pollutants can markedly exceed those anticipated from concentrations in ambient air (Wallace, 1991a). This phenomenon has become known as the 'personal cloud effect' (Furtaw et al., 1996).

Problems in quantifying exposure also arise because many of the advanced technologies that have been developed for measuring outdoor pollution are not suitable for indoor use. In part, this is due to considerations of cost, size, and the amount of air they displace (Maroni et al., 1995). Although miniaturised measuring devices have been developed for indoors, they often only record average pollutant concentrations over several hours or days. This can create uncertainties, especially for pollutants with health effects thought to be related to short-term extreme exposures. In the case of recording pollutants from sources such as micro-organisms an opposite problem can apply, whereby monitoring programmes must be operational for a very long time period to provide accurate estimates of exposure (Blomquist, 1994).

Because of the difficulties in assessing public exposure to indoor air pollution, coupled, perhaps, with the fact that the resurgence of scientific interest into the links between indoor air quality and health is still relatively recent, we know much less about the health risks from polluted indoor air than we do

about those associated with outdoor contamination. Indeed much of what has been learnt about the health impacts of indoor air comes from studies of the outdoor environment. Exposure to indoor toxicants can potentially lead to a variety of adverse health outcomes (Bascom et al., 1995). The likelihood that an individual will become ill from the presence of a contaminant depends upon factors such as the individual's sensitivity to that contaminant, the contaminant concentration, the current state of their psychological and physical health, and the duration and frequency of exposure (Seltzer, 1997). Indoor air pollutants have the potential to cause transient morbidity, disability, disease, and even death in extreme cases (Berglund et al., 1992). Recent research into these health outcomes has involved human, animal, and in vitro studies (Maroni et al., 1995).

Human studies entail the observation and measurement of symptoms and other effects in individuals exposed to pollutants, either routinely (as in epidemiological studies) or in experimental circumstances. The main advantage of studies of routine situations is that the conditions under which exposure takes place are realistic, although the power of these investigations is sometimes insufficient to signify the likely causality of associations. Experimental exposures have the advantage that exposure conditions and subject selection may be determined by the investigator. However, they are only suitable for studying slight, reversible, short-term effects in healthy or only moderately ill individuals.

Animal studies involve the assessment of the health effects of exposures in laboratory animals. Their main limitation is that it is necessary to extrapolate the findings to humans. Such extrapolations are fraught with difficulties, particularly as results are often based on much higher doses than would be commonly encountered in the indoor environment.

In in vitro research, the effects of pollutants on cell or organ cultures are examined. They have cost advantages over animal studies and are quick to undertake, but there are problems associated with the use of their findings to predict effects on whole organisms.

It is the aim of this review to draw together the results from these investigations and present a picture of our current understanding of the relationship between indoor air pollution and health. Most of the evidence presented will be taken from epidemiological studies. Animal and in vitro research has provided many important findings. However, the conditions of these studies may be rather removed from those that typically prevail when individuals are exposed to pollutants indoors. Major progress has recently been achieved in the field of developing guidelines for volatile organic compounds from bioassays (ECJRA, 1997), but their use for many substances is less well developed. Hence, this is not the place for a detailed consideration of their findings. Neither will research into the health impacts of occupational exposures to pollutants be discussed.

Such exposures are usually associated with high contaminant doses, and the health outcomes are often severe. Hence, the aetiology and mechanisms of industrial exposures are sufficiently different from those commonly encountered in the domestic or office environment to warrant consideration elsewhere.

In this account, pollutants are considered according to whether they arise from biological or non-biological sources. Of course, dividing them in this way provides a simplistic picture of true patterns of exposure; in reality individuals are often concurrently exposed to a wide variety of contaminants from a range of different sources. In some situations, they may react with each other, leading to patterns and levels of exposure that are rather different from the apparent sum of their parts. However, the use of a simplified division is necessarily adopted for the sake of clarity.

Also considered in this text is the phenomenon that has become known as 'sick building syndrome'(SBS), where the occupants of some buildings repeatedly describe a complex range of vague and often subjective health complaints (Horvath, 1997). Since the early 1970s, numerous outbreaks of SBS have been reported. These have usually been in offices, but sometimes schools, hospitals, homes for the elderly, or apartments have been involved. The symptoms of SBS are often relatively minor, non-specific, and common amongst the general population, but are more frequent amongst the occupants of 'sick' buildings (Lahtinen et al., 1998). Despite their minor nature, SBS symptoms may have a greater impact on public health and cost to the economy than some major diseases due to widespread absenteeism and lowered productivity amongst affected workers (Wallace, 1997). For example, it has been estimated that the annual 'cost' of headaches amongst the employees of the US Environmental Protection Agency may be as high as $2 million (Wallace, 1997).

Sick building syndrome is normally distinguished from the term 'building related illness' (BRI) (Ryan and Morrow, 1992). Whilst BRI has a known aetiology, many studies of SBS have failed to determine any specific factor that may explain the health complaints (Lahtinen et al., 1998). Outbreaks of SBS have been known to occur in older, naturally ventilated buildings. However they are more common in modern energy-efficient 'airtight' buildings, in particular those served by mechanical heating, ventilation, and air-conditioning (HVAC) systems (Redlich et al., 1997). This observation has led many to conclude that SBS must in some way be associated with indoor air quality problems.

2. Pollution from non-biological sources

It is important to note that outdoor sources may be the main contributors to indoor concentrations of a number of non-biological pollutants commonly found

in indoor air. This is especially the case for contamination in buildings situated in urban areas and close to industrial zones or streets with heavy traffic. The factors which determine the contribution of outdoor pollution to indoor air quality include the type of ventilation in use (natural or forced), the ventilation rate (air changes per hour), and the nature of the contaminants in question (Wanner, 1993). The major outdoor sources of important indoor air pollutants are given in Table 2. The USA EPA TEAM studies have shown that reactive gasses such as ozone tend to occur at lower concentrations indoors than outdoors because they react rapidly with indoor surfaces (Wallace, 1987). Non-reactive gases may accumulate indoors and exposures there may be greater than outside.

Where meaningful, typical indoor–outdoor concentration ratios of pollutants are provided. However, to discuss the full impact of outdoor pollutants on indoor air quality would require a detailed consideration of the chemistry and processes that operate within the outdoor environment. This is not the place for such a discussion, although the health effects of one outdoor pollutant, radon, are considered. Although radon arises from outside, it merits consideration as it only presents a significant risk to health when found inside buildings.

Heating and cooking are often essential indoor activities but may produce smoke and gasses that present a problem of disposal. This difficulty is greatest in colder climates, where it is necessary to retain heat at the same time as removing combustion by-products (Burr, 1997).

A wide range of pollutants are associated with combustion. In the past, a high reliance on wood and coal for the production of heat meant that the indoor pollution profile was dominated by smoke from these sources (Wallace, 1996). In some countries, especially those of eastern Europe and China, the use of these fuels still predominates (Lan et al., 1993). In much of western

Table 2. Outdoor sources of major indoor air pollutants

Pollutant	Percentage of emissions associated with industry[a]	Percentage of emissions associated with transport[a]
Benzene	32	65
Carbon monoxide (CO)	3	90
Lead (Pb)	31	60
Oxides of nitrogen (NO_x)	38	49
Particulates (PM_{10})	56	25
Sulphur dioxide (SO_2)	90	2
Volatile organic compounds (VOCs)	52	34
Ozone (O_3)	Arises from atmospheric chemical reactions	

[a]Figures based on UK estimates (DOE, 1997).

Europe and the USA, coal and wood have largely been replaced by natural gas and electricity (Lambert, 1997), and approximately one-half of all American homes now use gas for cooking and heating (Hines et al., 1993). If properly vented to the outside, most appliances are typically not major contributors to indoor air quality problems. However, the use of unvented appliances or those that are malfunctioning or improperly installed can generate severe problems (Nagda et al., 1996).

Cigarette smoking is an important source of indoor combustion related pollution. Tobacco smoke is an aerosol containing several thousand substances that occur as particles, vapours, and gases. Exposure to cigarette smoke has been associated with a wide range of acute and chronic health impacts. Although the prevalence of smoking in western societies is much lower than it was in the 1950s and 1960s, tobacco smoke still contributes a significant proportion of the total indoor pollutant dose for many individuals.

Materials that comprise the fabric of buildings are another important source of non-biological pollutants. A large number of the chemical compounds which are found in indoor air originate from paints, varnishes, solvents, and wood preservatives used in buildings (ECJRC, 1997). Furthermore, deteriorating materials that comprise the fabric of a building can become friable and release contaminants into the air. Pollutants from these sources are often difficult to quantify because they are present in relatively low concentrations, and their sources are diffuse (Wanner, 1993). An additional complication arises where levels of contamination are mediated by climatic factors such as temperature and relative humidity. Hence, possible health risks of contaminants from building materials depend very much on the nature and concentration of the pollutants involved.

The following sections consider the typical concentrations and important health effects of the range of non-biological pollutants from the various sources that are most commonly encountered in the indoor environment.

2.1. Asbestos

Asbestos is a generic term that applies to a group of impure hydrated silicate minerals which occur in various forms, and are incombustible and separable into filaments (Maroni et al., 1995). Until recently, asbestos was widely used by the building industry for the production of many different products. It was particularly valued for its electrical and thermal insulating properties, and was heavily utilised in pipe and boiler insulation, cement-board, thermal tiles, paint, and wallpaper (Bigonon et al., 1989). Concern over the health effects of asbestos exposure has led to legislation being introduced to prohibit its use in many countries. Hence, most asbestos-related air quality problems are associated with the management of existing asbestos installations.

Accurate assessment of exposure to asbestos is particularly difficult because most airborne fibres are too small to be counted by optical microscopy, and hence scanning and transmission electron microscopy must be used to analyse samples (Hines et al., 1993). Because of this, relatively little information on typical indoor concentrations is available (Gaensler, 1992). In one of the few studies that have been undertaken, Lee et al. (1992) examined exposure to airborne asbestos in 315 public, commercial, residential, school, and university buildings in the USA. The average concentration of asbestos was 0.02 structures ml^{-1} of air, whilst the average concentration of fibres greater than 5 μm in length was 0.00013 fibres ml^{-1}. However, it is important to note that, in 48% of samples, no asbestos was detected. In general, asbestos-containing materials within buildings that are in good repair are unlikely to lead to exposure of occupants to concentrations higher than ambient levels. A recent literature review commissioned by the US EPA cost 4 million dollars and suggested that well-maintained asbestos in public buildings posed little risk to office workers (Hines et al., 1993).

Acute exposure to asbestos can cause skin irritation (Spengler and Sexton, 1983). However, the most serious health effects from asbestos are lung cancer, mesothelioma (cancer involving a proliferation of mesothelial cells) and asbestosis (a slowly developing but lethal fibrosis of the lung) (McDonald, 1991). Recent data suggest that the fibres must remain in the respiratory tract for approximately one year to produce effects, and those with a diameter of less than 1 μm and a length greater than 5–10 μm are believed to be particularly dangerous. There is generally a latent period of 20–50 yr between first exposure to asbestos, and the clinical manifestation of tumours (Doll and Peto, 1985).

Whilst the relationship between asbestos exposure and the incidence of mesothelioma and asbestosis is well documented, clarifying the relationship with lung cancer is complicated, and is still a controversial area (Huncharek, 1994). It is accepted that exposure to asbestos and tobacco smoke from active smoking has a marked synergistic effect. It is important to note that the majority of population risk from asbestos arises from industrial exposures. However, cases of disease associated with non-industrial exposures do occur. For example, Roggli and Longo (1991) highlight the instance of a teacher's aid with pleural mesothelioma who was for many years exposed to asbestos-containing acoustic plaster in a school building.

2.2. Carbon dioxide

Carbon dioxide (CO_2) is a colourless, odourless gas. Humans continuously exhale CO_2 formed in the body during metabolic processes and, where fuel is not being burnt, these emissions comprise the greatest contribution to indoor

concentrations (Wanner, 1993). Carbon dioxide is also the main combustion product from gas, kerosene, and wood or coal fuelled appliances and these can represent significant sources when in operation (Moriske et al., 1996). Typical indoor CO_2 concentrations range between 700 and 2000 ppm (approximately 3657 mg^{-3}) but can exceed 3000 ppm (5486 mg^{-3}) during the use of unvented appliances (Arashidani et al., 1996).

Carbon dioxide is a simple asphyxiant, and can also act as a respiratory irritant (Maroni et al., 1995). However, although indoor to outdoor ratios of the gas are typically in the range 1–3 for most environments, exposure to an extremely high CO_2 concentration (above 30,000 ppm or 54860 mg^{-3}) is required before significant health problems are likely (NASA, 1973). At moderate concentrations, CO_2 can cause feelings of stuffiness and discomfort, an effect noted by Pettenkofer as far back as 1858 (Pettenkofer, 1858). Respiration can be slightly affected at levels above 15,000 ppm (27430 mg^{-3}). Exposures above 30,000 ppm can lead to headaches, dizziness, and nausea (Schwarzberg, 1993). Yang et al. (1997) found that these concentrations also affect perception of motion. This may be because CO_2 has been shown to moderate the activity of cells within the visual cortex.

2.3. Carbon monoxide

Carbon monoxide (CO) is a toxic odourless gas produced by the incomplete combustion of fuel (IEH, 1996). Water heaters, gas or coal heaters, and gas stoves are all indoor sources of CO (Gold, 1992). High indoor levels of CO can also result from the entry of outdoor vehicle exhausts into the ventilation system of a building. This can be a particular risk in buildings that have delivery areas where vehicles are parked with their motors running. In addition, tobacco smoking is an important source of transitory indoor CO pollution (Sterling, 1991), and the burning of charcoal briquettes and the use of gasoline-powered electrical generators during disruptions to electrical services can present temporary problems (Houck and Hampson, 1997). Another source of CO is methylene chloride, a substance used as a paint stripper. It is metabolised within the body to form CO, and its use can lead to a significant dose in even well ventilated rooms (Gold, 1992).

In the absence of emission sources, concentrations of CO in indoor environments are generally lower than those outdoors. In a US Environmental Protection Agency (EPA) survey of the personal CO exposures of 1000 non-smoking adults, residential exposures were generally low, ranging from 2 to 4 ppm (approximately 2.3–4.7 μg m^{-3}) (Akland et al., 1985). Where gas stoves are in operation, hourly concentrations of CO are generally around 6 ppm (6.9 μg m^{-3}), and do not often exceed 12 ppm (13.8 μg m^{-3}) (Samet et al., 1987).

The toxic properties of CO are largely associated with its high affinity for oxygen-carrying proteins such as haemoglobin and myoglobin (Coultas and Lambet, 1991). Because its affinity for haemoglobin is approximately 200 times greater than that of oxygen, CO displaces oxygen, forming carboxyhaemoglobin (COHb), lowering the oxygen carrying capacity of the blood, and producing a left shirt in the oxyhaemoglobin dissociation curve (Roughton and Darling, 1994). Carbon monoxide can also interfere with oxygen diffusion into cellular mitochondria, and interfere with intracellular oxidation (Gold, 1992).

The health effects of exposure to CO are generally described relative to COHb levels (Madany, 1992). In non-smoking individuals unexposed to environmental CO, blood COHb levels are usually around 0.5% (Lambert, 1997). Various symptoms of neuropsychological impairment have been associated with acute low-level exposures. Amitai et al. (1998) found that subjects exposed to CO from residential stoves for up to 2.5 h showed declines in their learning and planning abilities, as well as a drop in their attention and concentration spans. Chronic exposure (at 10–30% COHb) often produces symptoms that are easily misdiagnosed or overlooked, such as headache, fatigue, dizziness, and nausea (Stewart et al., 1970). There is evidence from animal studies that some foetal damage may occur from maternal exposure to CO at these levels (Longo, 1977).

Carbon monoxide poisoning has its most toxic acute effects on the organs with high oxygen requirements; the heart and brain. Hence, individuals with ischaemic heart disease are at particularly high risk (USEPA, 1991). At moderate concentrations of CO, adverse cardiovascular effects may be observed amongst susceptible individuals (Dahms et al., 1993). Amongst 20 non-smoking men with ischaemic heart disease, Lambert (1994) found that the probability of occurrence of an episode of myocardial ischemia was 2.1 times higher at COHb levels of 2% relative to those below 1%.

Although non-reversible impairment is relatively rarely associated with CO exposure, the greatest danger of serious CO poisoning comes from faulty combustion appliances, or those with blocked or malfunctioning external vents (Howell et al., 1997). In such situations, COHb levels of between 50 and 60% can result in fainting and convulsions, whilst higher exposures can lead to coma and death. In England and Wales, there are on average around 60 deaths annually associated with accidental CO poisoning (Burr, 1997), and similar rates have been observed in the USA (Cobb and Etzel, 1991). Individuals who survive acute CO poisoning may still exhibit neurological and psychological symptoms many weeks or months after exposure, particularly if a period of unconsciousness has occurred (Choi, 1983).

2.4. Formaldehyde

Formaldehyde is the most widespread aldehyde found in the environment. Although it is a volatile compound, it is not detected by the gas chromatographic methods commonly applied to VOC analysis, and hence is often considered separately (Maroni et al., 1995).

At normal room temperatures, formaldehyde is a colourless gas with a pungent odour. The primary sources of formaldehyde are building materials such as particleboard, medium-density fibreboard, plywood, resins, adhesives, and carpeting (Hines et al., 1993). It is also used in the manufacture of urea formaldehyde foam insulation (UFFI) which is injected into wall cavities to supplement the insulation in existing buildings. However, because of health concerns, UFFI is little used nowadays.

As with all volatile organic compounds, the concentration of formaldehyde within a given indoor space will be very dependent upon the presence of important emission sources. The background concentration of formaldehyde in outdoor air is generally lower than 0.1 ppm (Maroni et al., 1995). Indoors, the rate of emission of formaldehyde varies according to conditions of temperature and humidity. Indoor formaldehyde concentrations usually exceed those observed outdoors. In an early study of 23 Danish homes by Anderson et al. (1975), the average formaldehyde concentration was 0.5 ppm (0.6 mg m^{-3}), with a range from 0.07 to 1.9 ppm (0.08–2.28 mg m^{-3}). Similar findings have subsequently been reported in Germany by Prescher and Jander (1987), in Finland by Niemala et al. (1985) and in the USA by Breysse (1984).

Adverse health effects from formaldehyde exposure may arise from inhalation, or direct contact. A range of acute health impacts have been attributed to the substance (Table 3). Exposure to concentrations of less than 1 ppm (1.2 mg m^{-3}) may result in sneezing, coughing, and minor eye irritation, although these symptoms often rapidly subside after the start of the exposure (Koeck et al., 1997). Numerous studies show that formaldehyde vapour is also an irritant of the skin (Eberlein-König et al., 1998) and the respiratory tract (Bardana and Montanaro, 1991).

There is conclusive evidence that formaldehyde is an animal carcinogen (Morgan, 1997). In the 1980s, a number of occupational research projects were carried out to address the potential carcinogenicity of formaldehyde in humans (e.g. Wong, 1983). All subjects were workers exposed to high concentrations of formaldehyde, but none of the studies found strong evidence of a cancer risk. One of the few pieces of evidence of a risk to humans from typical indoor exposures comes from the work of Vaughan et al. (1986) who reported a significant correlation between formaldehyde exposure and nasopharyngeal cancer in mobile home residents.

Table 3. Acute health effects from formaldehyde exposure

Formaldehyde concentration (ppm)	Observed health effects
< 0.05	None reported
0.05–1.5	Neurophysiologic effects
0.05–1.0	Odour threshold limit
0.01–2.0	Irritation of eyes
0.10–25	Irritation of upper airway
5–30	Irritation of lower airway and pulmonary effects
50–100	Pulmonary edema, inflammation, pneumonia
> 100	Coma, death

Sources: Hines et al. (1993).

2.5. Nitrogen dioxide

Nitrogen dioxide (NO_2) is a water-soluble red to brown gas with a pungent acrid odour. It is formed from the combination of nitrogen and oxygen during combustion at high temperatures (Maroni et al., 1995). Hence, the production of NO_2 is particularly associated with the operation of gas appliances, kerosene heaters, and woodburning stoves, as well as the smoking of cigarettes. Additionally, outdoor air can act as an important source for indoor NO_2 pollution (Chan et al., 1990) although, in many areas of the USA and Europe, ambient outdoor NO_2 levels are relatively low.

There have been numerous studies of indoor concentrations of NO_2. In the absence of emission sources, levels generally correlate well with those observed outdoors (Monn et al., 1998). In homes with gas cooking stoves in Albuquerque, New Mexico, Lambert et al. (1993a) found that 2-week average NO_2 levels were 21 ppb (39 $\mu g\ m^{-3}$) in bedrooms and 34 ppb (63 $\mu g\ m^{-3}$) in kitchens. In comparison, bedroom concentrations in homes using electric stoves averaged just 7 ppb (13 $\mu g\ m^{-3}$). On average, normal use of a vented gas cooking range adds 25 ppb (47 $\mu g\ m^{-3}$) of NO_2 to the background concentration in a home (Samet et al., 1987). In homes with unvented kerosene space heaters, 1-week average concentrations exceeding 45 ppb (84 $\mu g\ m^{-3}$) have been observed (Leaderer et al., 1986). One-week average levels of greater than 50 ppb (94 $\mu g\ m^{-3}$) have been reported in homes with unvented gas space heaters (Ryan et al., 1989).

Transient peak concentrations of NO_2 during the operation of appliances can greatly exceed average measurements. Whilst cooking with a gas range, peak levels in the kitchen may be as high as 400–1000 ppb (752–1880 $\mu g\ m^{-3}$) (Spengler et al., 1981). As individuals operating gas ranges often stay close to the appliance, personal exposures may be even greater.

Nitrogen dioxide is an oxidising agent that can be very irritating to the mucous membranes of the lung (Spengler, 1993). It is highly soluble in water, and

a large proportion of inhaled NO_2 is removed in the respiratory tract (Lambert, 1997). It is thought to combine with water in the lungs to form nitric acid (HNO_3) and may react with lipids and proteins to form nitrite anions and hydrogen ions (Postlethwait and Bidani, 1990). Nitrogen dioxide within the airways is also converted into vapour phase nitrous acid (HONO) via heterogeneous reactions involving water vapour, invoking the formation of free oxygen radicals and lipo-peroxidation (Spicer et al., 1993). Whilst it has been suggested that oxidant injury is the principal mechanism by which NO_2 damages the lung, substantial uncertainty remains (Lambert, 1997).

Evidence from experimental research suggests that exposure to NO_2 may increase respiratory infections, and adversely affect lung function (Frampton et al., 1991). However, there is rather little evidence from epidemiological studies that exposure to NO_2 has deleterious health effects amongst the majority of the population. Two important projects have discovered possible associations. In Italy, Viegi et al. (1992) found the use of bottled gas for cooking was associated with increased reporting of cough and phlegm in males. In England, Jarvis et al. (1996) found that a general population sample of females who reported they used mainly gas for cooking were more likely to report respiratory symptoms in the 12 months prior to survey. These women were also found to have reduced lung function and increased airway obstruction. No effects were observed amongst males, suggesting women may be more susceptible to NO_2, or are exposed to higher concentrations.

There is evidence that the health effects of NO_2 may be greater amongst certain vulnerable population subgroups, such as children and asthmatics (Li et al., 1994). In the Harvard Six Cities Study, involving 10,106 children aged 6–10 yr, serious illness in infancy was more common amongst infants from homes with gas cooking (Ware et al., 1984). Recent meta-analyses suggest that the risk associated with NO_2 exposure is probably only significantly increased for children aged over 2 yr (Hasselblad et al., 1992). In Australia, Pilotto et al. (1997) monitored NO_2 exposures amongst 388 children aged between 6 and 11 yr. They found hourly peak levels of 80 ppb (150 $\mu g\ m^{-3}$) and above were associated with significant increases in the reporting of sore throats, colds, and absences from school, and they concluded that it was important to consider short-term peak exposures. However, in another study, Samet et al. (1993) found no association between personal exposures and symptoms amongst infants, and Brunekreff et al. (1990) failed to find an association between NO_2 and the pulmonary function of a sample of children.

Amongst asthmatic women and children, Goldstein et al. (1988) examined the relationship between mean 48 h NO_2 concentrations in the subjects' kitchens, and their spirometric lung function. They found that exposure to NO_2 levels ranging between 300 and 800 ppb (564 and 1504 $\mu g\ m^{-3}$) was associated with a reduction in lung capacity (measured by FEV) of the order of 10%.

Salome et al. (1996) also found that the experimental exposure of 600 ppb (1128 μg m^{-3}) of NO_2 over a period of one hour was associated with a slight increase in airway hyperresponsiveness amongst a sample of 20 asthmatics.

Exposure to NO_2 may act as a trigger for asthma in one of two ways (Jones, 1997). One possibility is that the pollutant causes a direct effect on the lungs by inflicting toxic damage. Another is that it may irritate and sensitise the lungs, making individuals more susceptible to allergic response upon contact with indoor allergens. Evidence for the sensitisation mechanism comes from a study by Tunnicliffe et al. (1994); in asthmatics previously exposed to 400 ppb (752 μg m^{-3}) NO_2 for a period of 1 h, lung function dropped by 19% after the inhalation of house dust mite allergen. This was compared to a reduction of just 14% in those exposed only to air.

2.6. Sulphur dioxide

Sulphur dioxide (SO_2) is produced by the oxidation of sulphur impurities during the burning of coal and other fuels that contain sulphur (Burr, 1997). It is a colourless gas with a strong pungent odour that can be detected at about 0.5 ppm (0.9 mg m^{-3}). It is readily soluble in water and can be oxidised within airborne water droplets (Maroni et al., 1995). Sulphur dioxide levels are generally lower indoors than outdoors, and indoor/outdoor concentration ratios between 0.1 and 0.6 are commonly observed in buildings without indoor sources. As a result of reductions in emissions, annual mean levels of ambient SO_2 in most major cities in Europe and the USA are below 20 ppb (52 μg m^{-3}). However indoor SO_2 concentrations can be higher for homes with kerosene heaters and poorly vented gas and coal appliances. Leaderer et al. (1993) recorded average concentrations of 30 ppb (78 μg m^{-3}) in a study of 33 homes with kerosene space heaters in Connecticut, USA. Mean values of 57 ppb (149 μg m^{-3}) have been measured in homes equipped with both kerosene heaters and gas stoves (Leaderer et al., 1984).

From a health effects viewpoint, two substances are important; the SO_2 itself, and the acid aerosols that may result from its oxidation in the atmosphere. Absorption of SO_2 in the mucous membranes of the nose and upper respiratory tract occurs as a result of its aqueous solubility. The deposition pattern of acid aerosols within the respiratory tract will be dependent upon the size distribution of the droplets and the level of humidity indoors (Maroni et al., 1995).

Whilst evidence from animal experiments and occupational exposures suggests that exposure to extreme concentrations of SO_2 and acid aerosols can precipitate an acute reduction in lung function (Islam and Ulmer, 1979), there are relatively few indications of short-term health impacts associated with typical indoor concentrations. However, long term exposures to indoor SO_2 may

be associated with elevated reporting of chronic respiratory complaints. In the UK, Burr et al. (1981) found that residents of South Wales who had open coal fires were more likely than others to suffer from breathlessness and wheezing. They speculated that a high death rate from respiratory conditions amongst miners wives may have been associated with exposure to SO_2 from the burning of concessionary coal, although particles may have also played a part. Recent studies in China, where the domestic burning of coal is still widespread, have also associated exposure to SO_2 with impaired lung function, and a range of other respiratory symptoms (Qin et al., 1993; Jin et al., 1993).

2.7. Radon

Radon is an inert radioactive gas that arises directly from the decay of radium-226 contained in various minerals (Lyman, 1997). It has a half-life of 3.82 d. As radon undergoes further radioactive decay, it produces a series of short-lived radioisotopes, known as radon daughters or progeny (Table 4). Radon itself is inert and causes little damage as most of it is exhaled in the breath. However, the progeny, Po-218 and Po-214 are electrically charged and can be inhaled either directly or through their attachment to airborne particles (Cohen, 1998). Once inhaled, they tend to remain in the lungs, where they may eventually cause cancer (Polpong and Bovornkitti, 1998). As activities such as the smoking of cigarettes can lead to considerably elevated levels of airborne particles, smokers are at particular risk from the inhalation of radon progeny (Hampson et al., 1998). Indeed, the US EPA has estimated that the cancer risk from radon for smokers is as much as 20 times the risk for individuals who have never smoked (US EPA, 1992).

In the past, contamination of air by radon and the subsequent exposure to radon daughters were believed to be a problem only for uranium and phosphate miners. However, it has recently been recognised that homes and buildings far away from uranium or phosphate mines can exhibit high concentrations of radon. As a consequence, radon and radon progeny are now recognised as important indoor pollutants (Létourneau, 1997).

Radon formed in rocks and soils is released into the surrounding air. Typical rates of radon release from soils throughout the world range from about 0.0002 to 0.07 bequerels (Bq) m^{-3} s^{-1}. Production rates from any soil are very dependent upon the geological characteristics of the soil itself and its underlying geological strata (Lévesque et al., 1997). Porous soils overlaying uranium rich alum shales, granites and pegmatites are a particularly high risk for radon, while gas-impermeable soils consisting of fine sand, silt, and moist clay present a low risk (IARC, 1988).

Outdoors, radon emanating from the ground is quickly dispersed, and concentrations never reach levels that may be a threat to health. However, inside

Table 4. The uranium-238 decay series

Nuclide	Half-life	Decay particle
U-238	4.5×10^9 yr	Alpha
Th-234	24 d	Beta
Pa-234	1.2 min	Beta
U-234	2.5×10^5 yr	Alpha
Th-230	8×10^4 yr	Alpha
Ra-226	1620 yr	Alpha
Radon-222	3.8 d	Alpha
Po-218	3 min	Alpha
Pb-214	27 min	Beta
Bi-214	20 min	Beta
Po-214	160 μ	Alpha
Pb-210	22 yr	Beta
Bi-210	5 d	Beta
Po-210	138 d	Alpha
Pb-206	Stable	-

Source: Lyman (1997).

confined areas, low rates of air exchange can result in a build-up of radon and its daughters to concentrations tens of thousands of times higher than those observed outside (Wanner, 1993). Radon concentrations within a building depend very much on both the concentration of radon in the soil surrounding the structure, and the presence of entry points that allow the gas to infiltrate from outside (Jedrychowski et al., 1995). Some of the common entry points of radon into buildings include foundation joints, cracks in floors and walls, drains and piping, electrical penetrations, and cellars with earth floors (Nielson et al., 1997).

In the USA, Nero et al. (1986) summarised the results of 19 studies of indoor radon concentrations, covering 552 single-family homes. They found the mean indoor concentration was 56 Bq m^{-3}. As part of the more recent US National Residential Radon Survey, Marcinowski et al. (1994) estimated an annual average radon concentration of 46.3 Bq m^{-3} in US homes. They also calculated that around 6% of homes had radon levels greater than the US EPA action level for mitigation of 148 Bq m^{-3}. Outside the USA, Albering et al. (1996) found a much higher average concentration of 116 Bq m^{-3} in 116 homes in the township of Visé in a radon prone area in Belgium. In Italy, Bochicchio et al. (1996) reported an average concentration of 75 Bq m^{-3} in a sample of 4866 dwellings, and observed concentrations exceeding 600 Bq m^{-3} in 0.2% of homes. Yun et al. (1998) recently undertook one of the relatively few studies of radon concentrations in the office environment. In 94 Hong Kong office buildings they recorded radon concentrations similar to those that have been observed in domestic situations, with a mean of 51 Bq m^{-3}.

Radon exposure has been linked to lung carcinogenesis in both human and animal studies. It has also been associated with the development of acute myeloid and acute lymphoblastic leukaemia. However, the estimation of health risks from residential radon is extremely complex, and encompasses many uncertainties. Studies of smoking and non-smoking uranium miners indicate that radon is a substantial risk factor for lung cancer at high concentrations. Based on data on dose–response relationships amongst miners, it is estimated that between 5 and 15% of lung cancer deaths might be associated with exposure to residential radon (Steindorf et al., 1995). The relevance of data from mines to the lower-exposure home environment is often questioned (Lubin et al., 1997). Nevertheless, a meta-analysis of eight epidemiological studies undertaken by Lubin and Boice (1997) found that the dose–response curve associated with domestic radon exposures was remarkably similar to that observed amongst miners.

Ecological (geographical) study designs have been adopted by a number of recent epidemiological investigations into the health risks associated with non-industrial radon exposures. Lucie (1989) reported positive county level correlations between radon exposure and mortality from acute myeloid leukaemia in the UK, and Henshaw et al. (1990) found that mean radon levels in 15 counties were significantly associated with the incidence of childhood cancers and, specifically, all leukaemias. However, these reports have been met with considerable criticism because ecological designs can suffer from serious limitations (Wolff, 1991). In particular, the effects of migration are often difficult to account for, information on potential confounding variables can be unavailable, and estimates of exposure for populations of large areas may differ greatly from actual individual doses. Recent more refined ecological analyses, such as that undertaken by Etherington et al. (1996), have reported no association between indoor radon exposure and the occurrence of cancer.

An alternative to ecological analyses is the case-control study design, where radon exposures amongst individuals with cancer are compared to those of control subjects free from the disease. Most case-control studies have reported a small, but significant, association between radon exposure and lung cancer mortality (Pershagen et al., 1994). For example, in a recent examination of over 4000 individuals in Sweden, Lagarde et al. (1997) estimated that there is an excess relative risk of contracting lung cancer of between 0.15 and 0.20 per 100 Bq m^{-3} increase in radon exposure.

2.8. Respirable particles

Smoke from the burning of wood and fossil fuels produces an extremely complex mixture of pollutants, both in physical and chemical characteristics, and toxicological properties (Lambert, 1997). One of the primary constituents of

the smoke are respirable particles (Cooper, 1980). These are aerosols that are of a small enough diameter to enter and remain in the lung. Many are around 6–7 μm, or less in diameter (Martonen et al., 1992). The particulate mater comprises a mix of organic and inorganic substances including aromatic hydrocarbon compounds, trace metals, nitrates, and sulphates (Maroni et al., 1995). In developing countries, exposure to smoke is arguably the greatest indoor pollution problem, as in many buildings the burning of wood, charcoal, crop residues, or animal dung is often undertaken without adequate ventilation (Gold, 1992). For example, in India, Pandey et al. (1989) found that airborne particle concentrations, measured during cooking, were as high as 21,000 μg m^{-3}. Such figures are not typical of indoor exposures in the developed world.

In most developed societies, a movement away from the use of coal and wood to provide heat has meant that indoor particulate pollution associated with combustion is rather lower than in the past. However, the popularity of woodburning stoves has increased in both Europe and the USA in recent years (Samet, 1990). Whilst particulate pollution from modern airtight stoves is much lower than was previously common with open fires, problems can still occur during start-up, stoking, and reloading.

In the absence of significant indoor sources, indoor to outdoor ratios of respirable particles are generally slightly below unity (Wallace, 1996; Janssen et al., 1998). However, measurement techniques routinely used in reported studies are often not able to differentiate combustion related particles from those from other sources (Maroni et al., 1995). Given the heterogeneous nature of particulate material, this makes it impossible to formulate a direct relationship between exposure to specific compounds in the aerosol and health effects.

As part of the Harvard Six-City Study, Neas et al. (1994) recorded mean annual $PM_{2.5}$ (particles less than 2.5 μm in diameter) concentrations in the order of 17 μg m^{-3} in a sample of 470 non-smoking homes. The presence of a smoker can greatly add to indoor concentrations, and is discussed elsewhere in this review. In another sample of homes, Traynor et al. (1986) reported indoor concentrations of respirable particles slightly above background (up to 30 μg m^{-3}) during the use of airtight woodstoves and substantially higher with non-airtight stoves (200–1900 μg m^{-3}). When not in operation, homes with woodburning stoves have on average about 4 μg m^{-3} higher indoor particulate concentrations than homes without the appliances.

It is important to note that personal exposures to particles are often higher than ambient indoor concentrations. This is particularly the case during the daytime. For example, Clayton et al. (1993) observed that daytime mean personal PM_{10} (particles less than 10 μm in diameter) exposure in a sample of individuals was more than 50% higher than either indoor or outdoor levels. Similar findings have recently been reported by Janssen et al. (1998).

In general, the health impacts of outdoor exposures to respirable particles are rather better studied than those associated with indoor exposures (see Ostro and Chestnut, 1998). Hence, the most extensive data on the health effects from exposure to particles are derived from epidemiological studies of outdoor air. Given that total exposure to particles is greater indoors than outside by virtue of the time spent indoors, a good deal of the apparent effects of outdoor particles probably occurs due to exposure indoors. Whilst in vitro and animal experiments suggest that emissions from coal or woodburning stoves are mutagenic, epidemiological evidence from studies in the indoor environment is limited to a relatively small number of reports (Marbury, 1991).

Irritant effects from a range of inhaled particles may result in airway constriction. There is evidence that woodsmoke may be associated with respiratory illness, particularly amongst vulnerable groups such as children and those with pre-existing chronic respiratory disease. Honickey et al. (1985) examined respiratory symptoms amongst 62 children in Michigan, USA. They found 84% of children in homes with woodburning stoves recounted at least one severe symptom, compared to only 3% of children in homes without a stove. As part of the Six-City Study, Dockery et al. (1993) found an odds ratio of 1.32 (95% CI 0.99–1.76) for respiratory illness in households with woodburning stoves in comparison with those using other sources of heating. Koenig et al. (1993) reported that infants exposed to woodsmoke were more likely to recount asthma symptoms, and Abbey et al. (1998) observed a reduction in lung function in non-smokers exposed to high concentrations of indoor particles over a period of 20 yr.

Polycyclic aromatic hydrocarbons (PAHs) in woodsmoke are of particular concern because of their carcinogenic potential. These fat-soluble compounds are produced as a result of incomplete combustion, and include a large number of organic molecules which contain two or more benzene rings (Maroni et al., 1995). Once airborne, PAH compounds can be absorbed onto particles and inhaled into the lungs. It is thought that their carcinogenic properties arise as a result of their metabolism within the body (Sisovic et al., 1996). Evidence for this comes from studies such as that undertaken by Mumford et al. (1995). Amongst residents of Xuan Wei county, China, they found that urinary concentrations of PAH metabolites were highest in villages with correspondingly elevated lung cancer death rates.

2.9. Tobacco smoke

Although the health effects of smoking have been recognised for a number of decades, it is only relatively recently that concerns have been directed towards the inhalation of environmental tobacco smoke (ETS) by the non-smoking population. Tobacco smoke is an aerosol containing several thousand substances

that are distributed as particles, vapours, and gasses (Gold, 1992). Environmental tobacco smoke is possibly one of the most important indoor air pollutants in homes and offices (Wanner, 1993). This is because a substantial proportion of the population are exposed to high concentrations of ETS on a regular basis. In 1990, it was estimated that approximately 50 million US citizens (corresponding to 26% of the adult population) were smokers (Rando et al., 1997), and around 70% of all children may be living in homes with at least one parent who smokes (Weiss, 1986).

The constituents of ETS are commonly sub-divided into those associated with sidestream and mainstream smoke. Mainstream (MS) smoke is that which is directly exhaled from the smoker, whilst sidestream smoke (SS) is emitted from the smouldering tobacco between puffs (Maroni et al., 1995). As information has become known about mainstream and sidestream smoke generation and its chemical composition, many irritants and carcinogens have been identified (Rando et al., 1997). By inhaling ETS, non-smokers are exposed to most of the toxins inhaled by active smokers, as well as some additional substances. For example, *N*-nitrosodimethylamine, a potent animal carcinogen, is emitted in quantities 20–100 times higher in SS than in MS smoke (Guerin et al., 1992), and a 'passive' smoker at a distance of 50 cm from a cigarette may inhale more than 10 times the amount of carbonyl compounds actively taken up by the smoker (Schlitt and Knöppel, 1989).

Some of the more common compounds found in ETS are outlined in Table 5. However, the exact composition of the smoke is known to be very dependent upon the type of tobacco being consumed, its packing density, the composition of the wrapping paper, and the puffing rate of the smoker (Hines et al., 1993).

Because of the complex nature of ETS, measurements of respirable particles are often used as indicators of indoor concentrations. Reported cigarette smoke particle sizes have varied in the range from 0.1 to 1.5 μm (Chen et al., 1990). Spengler et al. (1981) observed respirable particulate concentrations in 80 homes over several weeks. They found that a smoker of one pack a day contributed around 20 μg m^{-3} to 24 h indoor concentrations. However, because cigarettes are not uniformly smoked throughout the day, the authors concluded that short-term particulate concentrations of 500–1000 μg m^{-3} were likely when cigarettes were actually ignited. Over a period of 3 months, Leaderer et al. (1990) investigated indoor aerosol mass concentrations in a sample of almost 400 houses. They found that homes with smokers exhibited mass concentrations that were approximately three times higher than those observed in non-smoking residences.

As well as being an important pollutant of the domestic environment, ETS can be a serious problem in the workplace. In a report on tobacco smoke discomfort published in the USA (Anonymous, 1992) 43.5% of non-smoking employees recounted at least some discomfort from ETS at their place of work.

Table 5. Emission factors for mainstream and sidestream smoke

Substance	Mainstream (μg per cigarette)	SS/MS ratio	Calculated sidestream (μg per cigarette)[a]
Carbon dioxide	10,000–80,000	8.1	81,000–640,000
Carbon monoxide	500–26,000	2.5	1200–65,000
Oxides of nitrogen	16–600	4.7–5.8	80–3500
Ammonia	10–130	44–73	400–9500
Hydrogen cyanide	280–550	0.17–0.37	48–203
Formaldehyde	20–90	51	1000–4600
Acrolein	10–140	12	100–1700
N-nitrosodimethylamine	0.004–0.18	10–830	0.04–149
Nicotine	60–2300	2.6–3.3	160–7600
Total particulate	100–40,000	1.3–1.9	130–76,000
Phenol	20–150	2.6	52–390
Catechol	40–280	0.7	28–196
Naphthalene	2.8	16	45
Benzo(a)pyrene	0.008–0.04	2.7–3.4	0.02–0.14
Aniline	0.10–1.20	30	3–36
2-Naphthylamine	0.004–0.027	39	0.02–1.1
4-Aminobiphenyl	0.002–0.005	31	0.06–0.16
N-nitrosonornicotine	0.2–3.7	1–5	0.02–18

Source: Rando et al. (1997).
[a]Calculated from ratio of SS to MS.

Occupational groups particularly at risk from ETS include those such as bartenders and waiters who work in a traditionally smoky environment. Lambert et al. (1993b) recorded a median respirable concentration of 53.2 $\mu g\ m^{-3}$ in a study of 7 restaurants, whilst a median concentration of 355 $\mu g\ m^{-3}$ has been noted in a billiard lounge (Rando et al., 1997).

The most common acute health effects associated with exposure to ETS are eye, nose, and throat irritation (Maroni et al., 1995). For some people, eye tearing can be so intense as to be incapacitating. Smoke also contains substances that can activate the immune system, and approximately half of allergy prone individuals react to various extracts of tobacco leaf or smoke in skin tests (Maroni et al., 1995).

Exposure to ETS has been particularly associated with the exacerbation of symptoms in asthmatics (Jones, 1998). In Italy, parental smoking and the self-reported prevalence of asthma symptoms was examined amongst 3239 children by Forastiére et al. (1992). They found that the prevalence of asthma symptoms was significantly elevated in children whose mother smoked. Similarly, Gortmaker et al. (1982) found the prevalence of severe asthma was 2.2% amongst a sample of children with a smoking parent, compared to only 1.1% amongst children whose parents did not smoke. There is some evidence that exposure to

ETS may be important in actually inducing the onset of asthma in very young children. On the Isle of Wight, UK, Arshad et al. (1992) reported an odds ratio for asthma at 12 months of 3.33 (95% CI 0.8–14.6) if one parent smoked, compared to 11 (95% CI 2.5–48.2) if both were smokers.

Strong evidence is available for establishing a link between acute childhood lower respiratory tract illnesses and exposure to ETS at home (Somerville et al., 1988). Furthermore, passive smoking may have a significant effect on the level and growth of lung function in children (NRC, 1986) and the onset of chronic obstructive pulmonary disease in later life. Smoke-exposed children also have increased rates of hospitalisation; as far back as 1974, Harlap and Davies observed elevated rates of hospital admission for bronchitis, pneumonia, and bronchiolitis amongst the infants of smokers (Harlap and Davies, 1974). An important current field of research is investigating whether the greatest effects of environmental tobacco smoke on the lung are established prebirth, or are associated with exposures during the first few years of life (Gold, 1992).

The strongest evidence for the carcinogenic effects of ETS comes from research into cases of lung cancer. Based on the results from a case-control study of 191 individuals in the USA, Janerich et al. (1990) concluded that approximately 17% of lung cancers amongst non-smokers can be attributed to exposure to ETS during early life. The risks associated with exposure appear to be particularly high amongst women; a multi-centre national study of lung cancer in lifetime non-smokers found an increased cancer risk of approximately 30% in women whose husbands smoked (Rando et al., 1997). Although it is difficult to determine causality, there is evidence that cancer in sites other than the lung may be associated with passive smoking. Fukuda and Shibata (1990) discovered an association between ETS exposure and cancer of the sinus, and a relationship with the incidence of brain tumours has also been reported (Ryan et al., 1992).

2.10. Volatile organic compounds

Any chemical compound that contains at least one carbon and a hydrogen atom in its molecular structure is referred to as an organic compound. Organic compounds are further classified into various categories which include volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs) and non-volatile organic compounds (NVOCs). Volatile organic compounds are defined as having a boiling point that ranges between 50°C and 260°C (Maroni et al., 1995). Their low boiling point means that they will readily off-gas vapours into indoor air. However, the fact that materials containing them exhibit the desirable characteristics of good insulation properties, economy, fire-resistance,

Table 6. Sources of common volatile organic compounds in indoor air

Sources	Examples of typical contaminants
Consumer and commercial products	Aliphatic hydrocarbons (*n*-decane, branched alkanes), aromatic hydrocarbons (toluene, xylenes), halogenated hydrocarbons (methylene chloride), alcohols, ketones (acetone, methyl ethyl ketone), aldehydes (formaldehyde), esters (alkyl ethoxylate), ethers (glycol ethers), terpenes (limonene, alpha-pinene).
Paints and associated supplies	Aliphatic hydrocarbons (*n*-hexane, *n*-heptane), aromatic hydrocarbons (toluene), halogenated hydrocarbons (methylene chloride, propylene dichloride), alcohols, ketones (methyl ethyl ketone), esters (ethyl acetate), ethers (methyl ether, ethyle ether, butyl ether).
Adhesives	Aliphatic hydrocarbons (hexane, heptane), aromatic hydrocarbons, halogenated hydrocarbons, alcohols, amines, ketones (acetone, methyl ethyl ketone), esters (vinyl acetate), ethers.
Furnishings and clothing	Aromatic hydrocarbons (styrene, brominated aromatics), halogenated hydrocarbons (vinyl chloride), aldehydes (formaldehyde), ethers, esters.
Building materials	Aliphatic hydrocarbons (*n*-decane, *n*-dodecane), aromatic hydrocarbons (toluene, styrene, ethylbenzene), halogenated hydrocarbons (vinyl-chloride), aldehydes (formaldehyde), ketones (acetone, butanone), ethers, esters (urethane, ethylacetate).
Combustion appliances	Aliphatic hydrocarbons (propane, butane, isobutane), aldehydes (acetaldehyde, acrolein).
Potable water	Halogenated hydrocarbons (1,1,1-trichloroethane, chloroform, trichloroethane).

Source: Maroni et al., 1995.

and ease of installation, means that their use in construction projects is widespread (Burton, 1997).

Table 6 lists sources of common VOCs found indoors. Out of a total of more than 900 chemical and biological substances that have been identified in indoor air, more than 350 VOCs have been recorded at concentrations exceeding 1 ppb (Brooks et al., 1991). The typical concentrations of some of the most frequently encountered compounds in the home are summarised by Table 7.

Indoor concentrations of VOCs have been well quantified; they are mostly considerably below the odour threshold but often exceed outdoor levels by up to 5 times (Wallace, 1991a). One of the earliest studies was undertaken by Mølhave (1979). He targeted 29 chemicals in 14 office buildings in Denmark. Most VOCs identified were alkylbenzenes and had concentrations in the range of 0.03–2.8 $\mu g\ m^{-3}$. Shah and Singh (1988) found that levels of the majority

Table 7. Concentration (μg m^{-3}) and distribution of selected VOCs found in indoor air

Pollutant	Concentration				
	Percentile				Mean
	10th	50th	90th	98th	
Benzene	2	10	20	30	10
Toluene	30	65	150	250	80
n-Decane	3	10	50	90	20
Limonene	2	15	70		30
o-Xylene	3	5	10		10
1,1,1-Trichloroethane	2	5	20		10
p-Dichlorobenzene	1	5	20		
1,2,4-Trimethylbenzene		5	20		10
m- and *p*-Xylene	10	20	40		20
Undecane	3	5	25		10
1,3,5-Trimethylbenzene		2	5		5
Dichloroethane		< 10	< 10	600	
Trichloroethane	1	5	20	30	

Source: IEH, 1996.

of a sample of 66 indoor VOCs ranged from 0.4 to 4 μg m^{-3} and Brown et al. (1994) reported that concentrations of most VOCs they studied were below 5 μg m^{-3}.

In any given environment, the concentration of individual VOCs will be very variable and depend upon the presence or absence of an extremely wide range of potential emission sources. Hence, many publications report levels of 'Total Volatile Organic Compounds' (TVOCs) rather than individual values, although this of little help in determining the toxicological properties of substances (ECJRC, 1997). A large study of TVOC concentrations in a sample of 179 UK homes found that the mean concentration of all readings in the rooms measured was 200–500 μg m^{-3} (Brown and Crump, 1996). The maximum recorded concentration was 11,401 μg m^{-3} in a living room. Mean concentrations were highest in main bedrooms, and lowest in second bedrooms. Similar findings have been reported by research in Denmark (Wolkoff et al., 1991), Germany, (Adlkofer et al., 1993) Sweden (Norbäck et al., 1993), and the USA (Hartwell, 1987).

It is important to note that average VOC concentrations may give an inaccurate indication of personal exposures (Rodes, 1991). This is because individuals are often situated close to emission sources, and may also be subjected to pollutants emitted from substances on their person. As an illustration of this, Wallace (1991a) reported the results of measurements of personal exposure to 25 VOCs among 51 Los Angeles Residents. Whilst ambient indoor maxima

ranged from 10 to 100 μg m^{-3}, personal maxima were generally between 100 to 1000 μg m^{-3}.

VOC concentrations may be much higher than typical ambient levels in newly constructed buildings, or those in which building work or decoration has recently taken place. This is because many VOCs will off-gas a significant proportion of their volume in a relatively short time, and hence their concentrations will decline rapidly and exponentially. Because of this, builders and decorators may receive particularly high doses (Wieslander et al., 1997). For example, Wallace et al. (1991), found that breath concentrations of decane in a subject being studied increased by a factor of 100 (from 2.9–290 μg m^{-3}) after they were engaged in painting and the use of solvents.

Exposure to VOCs can result in both acute and chronic health effects. It is possible that asthmatics and other individuals with prior respiratory complaints may be particularly susceptible to low-dose VOC exposures. In a recent study, Norbäck et al. (1995) reported a positive association between levels of VOCs and the prevalence of nocturnal breathlessness amongst 88 Swedish asthmatics aged between 20–45 yr. However, most information on VOC toxicity has been established from animal and experimental studies at high concentrations, as levels in the majority of indoor environments are well below those required to demonstrate measurable health impacts.

At high concentrations, many VOCs are potent narcotics, and can depress the central nervous system (Maroni et al., 1995). Exposures can also lead to irritation of the eyes and respiratory tract, and cause sensitisation reactions involving the eyes, skin, and lungs. Mølhave (1991) reported complaints of unpleasant mucous membrane irritation amongst individuals exposed to a mix of 22 VOCs at a concentration of 8 μg m^{-3}. Because of the similarity of these symptoms, exposure to VOCs has frequently been attributed as a cause of sick building syndrome, discussed later in this text. To back this up, a number of studies have reported a strong association between mucous membrane irritation, central nervous system symptoms, and total exposure to VOCs amongst office workers (Hodgson et al., 1991).

At extreme concentrations, some VOCs may result in impaired neurobehavioral function (Burton, 1997). In an experimental study, Otto et al. (1992) noted that subjects exposed to a concentration of 22 VOCs at 25 μg m^{-3}, reported symptoms of headache, drowsiness, fatigue, and confusion. At concentrations as high as 188 μg m^{-3}, VOCs such as toluene may cause symptoms of lethargy, dizziness, and confusion. These may progress to coma, convulsions, and possibly death at levels in excess of 35,000 μg m^{-3} (Sandmeyer, 1982). However, these concentrations have never been recorded in the non-industrial environment.

Exposure to high concentrations of several VOCs commonly found in indoor air have been associated with cancers in laboratory animals. Based on

laboratory evidence, Wallace (1991b) estimated that typical concentrations of seven VOCs exceeded the 1×10^{-6} risk of cancer by at least a factor of 10. These include benzene, vinylidine chloride, *p*-dichlorobenzene, chloroform, ethylene dibromide, methylene chloride and carbon tetrachloride. However, it should be noted that many risk calculations involve an assumption of a linear relationship of the dose–response curve when extrapolating from high to low exposures, and hence they are subject to much uncertainty (Stolwijk, 1991).

One recent theory proposes that the products of chemical reactions involving VOCs may be more important than direct exposure to the VOCs themselves (Wolkoff et al., 1997). This proposition is based on the fact that the majority of sick building syndrome studies have recorded concentrations of VOCs at considerably lower levels than those required to induce symptoms. Whilst there is an assumption here that sick building syndrome must be associated with exposure to VOCs, there is increasing evidence that many chemical reactions take place on surfaces and in air in the indoor environment (Reiss et al., 1995). Results from epidemiological research suggests that reactions between indoor ozone and VOCs may produce irritant substances that could cause SBS symptoms (Groes et al., 1996). Reactions involving nitrogen dioxide (Grosjean et al., 1992), and particles (Schneider et al., 1994) may also be important.

3. Pollution from biological sources

Whilst discussions of indoor air pollution frequently concentrate on chemical pollutants, the health effects of inhaled biological particles should not be overlooked, as a large variety of biological material is present in indoor environments (Montanaro, 1997). Biological agents can cause disease through atopic mechanisms, infectious processes, or direct toxicity. House dust in carpets, on sofas, and in air ducts is a major source of a range biological allergens (Lewis et al., 1994). The growth of moulds is not only aesthetically unpleasant, but can pose serious health problems, and some indoor environments provide ideal conditions for the maintenance of populations of harmful viruses and bacteria. The health effects of these agents are discussed below.

3.1. Indoor biological allergens

Table 8 summarises the main biological allergens found in indoor air. Probably the best studied source of indoor allergens is the house dust mite. In temperate climates, ten species of mites can be found from the genera *Dermatophagoides, Euroglyphus, Malayoglyphus, Hirstia,* and *Sturnophagoides* (D'Amato et al., 1994). D. *pteronyssinus* and *E. Maynei* mites appear to comprise around 95%

Table 8. Common indoor allergic agents

Source	Genus	Species	Allergen
Dust-mite	*Dermatophagoides*	*pteronyssinus*	*Der p*
	Dermatophagoides	*farinae*	*Der f*
	Euroglyphus	*maynei*	*Eur m*
	Hirstia	*domicola*	*Hir d*
	Lepidoglyphus	*destructor*	*Lep d*
	Malayoglyphus	*intermedius*	*Mal I*
	Malayoglyphus	*carmelitus*	*Mal C*
	Sturnophagoides	*brasiliensis*	*Stu b*
Cat	*Felis*	*domesticus*	*Fel d*
Dog	*Canis*	*familiaris*	*Can f*
Rodent	*Mus*	*musculus*	*Mus m*
	Rattus	*norvegicus*	*Rat n*
Cockroach	*Blattela*	*germanica*	*Bla g*
	Periplanetta	*americana*	*Per a*
Fungi	*Alternaria*	*alternato*	*Alt a*
	Aspergillus	*fumigatus*	*Asp f*
	Cladosporium	*herbarium*	*Cla h*

of all mites found in a typical house (Hart and Whitehead, 1990). In topical climates the mite *Blomia tropicalis* is also commonly found. Undoubtedly the most researched mite is the common house dust mite *D. pteronyssinus*. They measure 250–350 μm in size, and grow from egg to adult in around 25 d (D'Amato et al., 1994). Most mite populations prefer constant temperatures of around 25°C and a relative humidity of between 70 and 80% (Arlain et al., 1990; Salerno et al., 1992). They do not survive well in cool, dry conditions. Mites inhabit a range of soft furnishings, including sofas, fabrics, carpets, sheets, duvets, pillows, and mattresses.

The droppings from the mite are a primary source of indoor antigens (Kaliner and White, 1994). Mite faeces are encased in a coating of intestinal enzymes including a protein which is a strong allergen. Although over 10 different antigens have been characterised for *D. pteronyssinus* so far, the first to be identified, *Der p* 1, is the most commonly reported.

There may be up to 100,000 mite faecal particles, ranging between 10 and 40 μm in diameter, in a gram of house dust (Platts-Mills et al., 1991). The size of the particles, similar to that of pollen grains, means they do not remain airborne for long, although concentrations in disturbed rooms can be more than 1000 times higher than those observed in undisturbed environments (Kalra et al., 1990). A concentration of mite allergen above 2 μg *Der p* 1 g^{-1} (equivalent to 100 mites) of dust appears to represent a significant factor for mite sensitisation (WHO, 1995), and 10 μg *Der p* 1 g^{-1} increases the risk of triggering an acute or severe asthma attack in mite allergic individuals (IEH, 1996).

Typical *Der p* 1 concentrations are below 5 μg g^{-1} of dust (Verhoeff, 1994). However, the actual concentration of mite antigen in a home will strongly depend upon a range of factors, and is particularly influenced by climatic variables. This is illustrated by the work of Friedman et al. (1992). They reported mean concentrations of *Der p* 1 and *Der f* 1 as being 3.98 and 8.17 μg g^{-1} of dust in a sample of 15 New England homes during the month of June. In September the corresponding mean concentrations had risen to 21.43 and 14.53 μg g^{-1}.

Given that inhalation is the main route of exposure for the majority of allergens in indoor air, it is unsurprising that most research has concentrated on the effects on atopic (allergic) respiratory diseases, and in particular asthma. Over 70 yr ago, Ancona (1923) made the first documented observation that mites may be a cause of respiratory symptoms. Today, the evidence suggests that exposure to airborne mite allergen will exacerbate symptoms in up to 85% of asthmatics (Platts-Mills and Carter, 1997). Studies of the effects of dust mite allergen exposure are numerous. Recently, Björnsson et al. (1995) reported that Swedish subjects inhabiting homes with large mite populations were more likely to report nocturnal breathlessness and other asthma-related symptoms. Peat et al. (1994) also found an association between exposure to dust mite *Der p* 1 and wheezing in the previous year amongst Australian children.

As well as being a major trigger of attacks of asthma, there is some evidence that, particularly amongst infants, exposure to mite allergen may induce the onset of the condition in previously healthy individuals (Tariq et al., 1998). In an important investigation, Arshad et al. (1992) studied the role of early exposure to dust-mite allergens in the development of allergic disorders amongst infants on the Isle of Wight. One hundred and twenty infants were randomly allocated to prophylactic and control groups. In the prophylactic group, the infants' bedrooms and living rooms were treated to control mite populations. In the control group, there were no interventions. After 12 months, the odds ratio for asthma was 4.13 (95% CI 1.1–15.5) amongst the control group. After 24 months, the excess number of cases amongst the control group had fallen, but still remained (Hide et al., 1994).

Like dust mites, cockroaches have been associated with manifestations of symptoms in individuals with allergic asthma, and up to 60% of asthmatics test positive to cockroach allergen in the USA (Kuster, 1996). Although cockroaches are generally indigenous to warm tropical climates, some species are able to thrive elsewhere due to the presence of central heating. The most commonly found cockroach in homes in Europe is the German Cockroach, *Blattella Germanica*, from which at least 5 antigens have been isolated (*Bla g* 1–5). In the USA, the American Cockroach, *Periplaneta americana* is ubiquitous. Sources of cockroach allergens have been identified in body parts, as well as faecal extracts (Musmand et al., 1995).

Van Wijnen et al. (1997) studied concentrations of *Bla g* 1 allergen in 46 homes in Amsterdam. It was detected in over 44% of homes, with mean concentrations ranging between 1.3 and 11 ng g^{-1} of dust. The highest level recorded in a single sample was 3899 ng g^{-1}. In general, concentrations were elevated in rooms with textile floor coverings. Similar findings were made in the USA by Rosenstreich et al. (1997). They measured concentrations of the allergen in dust taken from 476 homes situated in various inner-city locations. Interestingly, they found that asthmatic children allergic to cockroaches were three times more likely to be hospitalised for their asthma if they lived in a home with a large cockroach population. The authors concluded that the problems of cockroach sensitisation might be particularly severe amongst the residents of poor quality inner-city housing, as these homes provide an ideal environment for cockroach populations.

Domestic cats and dogs are a further important source of allergens in indoor air. This is particularly so in colder climates, which are less favourable to dust mites. Cat allergen (*Fel d*) is found in saliva, skin and dander. Particles of the primary cat allergen, *Fel d* 1, are often small, and hence remain airborne for many hours (Luczynska et al., 1990). *Fel d* 1 concentrations in domestic dust can exceed 10 μg g^{-1} and airborne levels can vary between 2 and 20 ng m^{-3} (Luczynska, 1994). As with cat allergens, dog allergens are mostly found in saliva and dander. The major dog allergen has been termed *Can f* 1 (Montanaro, 1997). Concentrations of *Can f* 1 in dust are often similar to *Fel d* 1, although *Fel d* 1 can remain airborne longer due to its smaller particle size (Puerta et al., 1997).

Because *Fel d* and *Can f* are readily transported by becoming attached to clothing, exposure to these allergens can even be a problem in public places such as schools, trains, and hospitals, where pets are not allowed. Custovic et al. (1998) recently examined concentrations of mite, cockroach, dog, and cat allergens in a sample of 14 British hospitals. Although levels of mite and cockroach allergen were low, high concentrations of *Fel d* 1 (Mean 22.9 μg g^{-1}, range 4.5–58) and *Can f* 1 (Mean 21.6 μg g^{-1}, range 4–63) were found in upholstered chairs. Out of a total of 10 sampling days, airborne *Can f* 1 was detected on every occasion (range 0.12–0.56 ng m^{-3}), whilst airborne *Fel d* 1 was recorded on 7 d (range 0.09–0.22 ng m^{-3}).

Recently, it has been proposed that concentrations of dog and cat allergens over 8 μg g^{-1} in house dust represent a threshold for allergic sensitisation (D'Amato et al., 1994). Because of the generally small particle size, dog and cat allergen readily enters the lung and can produce a rapid and severe asthmatic response. For example, Pollart et al. (1991) found 30 out of 188 asthmatic patients admitted to a hospital emergency department were allergic to cats, compared to only 1 out of 202 control asthmatics who did not require emergency treatment.

3.2. Fungi, bacteria and viruses

Microorganisms are an important form of biological pollution in the indoor environment. A large number of species of fungi and bacteria are found indoors, where they are associated with the presence of organic matter (e.g. wall coatings, wood, foodstuffs) (IEH, 1996). The outdoor air is one of the major sources of fungi and bacteria in indoor environments, particularly during the summer and autumn (Wanner et al., 1993). It is also well documented that high levels of humidity favour fungal growth (Sterling and Lewis, 1998). They are frequently found in homes that contain damp conditions, especially those with structural faults, or basements and underfloor crawl spaces (Montanaro, 1997).

A major difficulty to the sampling of viable airborne fungi is the large variability with time, even over short periods (IEH, 1996). A particular problem arises as temporal variations in airborne concentrations are often much greater in magnitude than variations between homes (Flannigan and Miller, 1994). Despite these difficulties, indoor concentrations of bacteria and fungi have been relatively well studied.

In the UK, Hunter et al. (1996) used air filtration to monitor 163 homes for the presence of fungi and bacteria over the period November 1990 to December 1992. The geometric mean count was 234 colony forming units CFU m^{-3} air for fungi, and 365.6 CFU m^{-3} air for bacteria. In a more intensive study of 35 of the houses, mean counts were 912 and 818 CFU m^{-3} air for fungi in living rooms and bedrooms respectively, compared to 917 and 933 CFU m^{-3} air for bacteria. *Penicillium* was the most frequently isolated fungus, found in 53% of samples. It was followed by *Cladosporium* and *Aspergillus*. The dominant bacteria were *Bacillus*, followed by *Staphylococcus* and *Micrococcus*. In Norway, Dotterud et al. (1995) found that *Penicillium* was the most common microfungus in homes and schools, followed by *Aspergillus*, *Cladosporium*, and *Mucor*. In a study in Japan, Takahashi (1997) reported that concentrations of fungal CFU ranged from between 13 to 3750 CFU m^{-3} in samples of air from indoor environments. They also found that concentrations were strongly correlated with indoor temperature and relative humidity, as well as outdoor climatic factors. Macher et al. (1991) measured rather less extreme spore concentrations, of the order of 198 CFU m^{-3}, in an apartment in the USA.

Exposure to airborne bacteria and fungi is associated with a number of well-defined diseases, as well as various less well-defined symptoms (Peat et al., 1998). Table 9 summarises some of the main reported health impacts. Although many studies use indoor dampness as a surrogate measure of microorganism concentrations, some direct measurements have been attempted. In the UK, Platt et al. (1989) examined the relationship between fungal spore counts and self-reported symptoms amongst the occupants of almost 600 homes. They found higher numbers of CFU m^{-3} were associated with an elevated preva-

Table 9. Diseases and disease syndromes associated with exposure to bacteria and fungi

Disease/Syndrome	Examples of causal organisms cited
Rhinitis (and other upper respiratory symptoms)	*Alternaria, Cladosporium, Epicoccum*
Asthma	Various aspergilli and penicillia, *Alternaria, Cladosporium, Mucor, Stachybotrys, Serpula* (dry rot)
Humidifier fever	Gram-negative bacteria and their lipopolysaccharide endotoxins, Actinomycetes and fungi
Extrinsic allergic alveolitis	*Cladosporium, Sporobolomyces, Aureobasidium, Acremonium, Rhodotorula, Trichosporon, Serpula, Penicillium, Bacillus*
Atopic dermatitis	*Alternaria, Aspergillus, Cladosporium*

Source: IEH, 1996.

lence of reported wheeze and fever in children, and high blood pressure, and breathlessness in adults. Also in the UK, Potter et al. (1991) detected allergic responses to nine different fungal allergens in a survey of 2000 patients with allergic respiratory disease. In Sweden, Wickman et al. (1992) found that *Penicillium*, *Alternaria*, and *Cladosporium* moulds were more common in homes of children with allergies, and Neas et al. (1996) reported that morning lung function was inversely associated with *Epicoccum* and *Cladosporium* spore concentrations amongst a panel of 108 children. Contrary to many findings, Verhoff et al. (1994) were unable to find any relationship between microbial contamination and the directly measured pulmonary function of a sample of children living in 60 homes in the Netherlands. Although levels of CFU were higher in dust from mattresses in damp homes, they were not associated with objective measures of lung function amongst the infants.

Interestingly, the most recent research on moulds and fungi suggests that allergic reactions may not be the most important factor in the development of respiratory symptoms associated with exposure to spores (Howden-Chapman et al., 1996). Mycotoxins are toxic compounds produced naturally by many fungi (Hendry and Cole, 1993). They induce a wide range of acute and chronic systemic effects that cannot be attributed to fungal growth within the host. Samson et al. (1994) provide evidence that mycotoxins emitted by fungi are readily absorbed through the membranes of the respiratory tract, and it could be that their presence in the lung affects the immune system, precipitating the onset of symptoms.

Most bacterial and viral infections that spread within buildings are transmitted from human to human by airborne droplets (Ayars, 1997). More often than not, the building is an 'innocent bystander' and plays no role in harbouring populations of infectious agents, other than by providing a living environment for infected individuals. However, there are examples of serious bacterial and

viral infections where the building itself can act as a potential reservoir (Burrel, 1991).

Probably the best-known infection associated with the indoor environment is Legionnaire's disease. It is caused by the bacteria *Legionella pneumophila*. It was first recognised at a legionnaire's convention in Philadelphia in 1976. One hundred and eighty four legionnaires attending the symposium developed symptoms of the disease, and 29 died (Fraser et al., 1977). The disease has an incubation period of 2–12 d, and an attack rate of 1–7%. Typical symptoms include malaise and headache, followed by dry cough, chest pain, diarrhoea, and altered mental status (Ayars, 1997). Since that epidemic, numerous other outbreaks and a host of other *Legionella* species have been identified (Hanrahan et al., 1987).

Legionella bacteria thrive in a warm and humid environment, and a common denominator in most outbreaks is a source of water in or around the building. Typical reservoirs of the bacteria include air humidifiers, air conditioning cooling towers, warm water supplies, shower heads and plumbing systems (Hines et al., 1993). The mechanisms by which *Legionella* reach the lungs were, until recently, thought to be via the inhalation of aerosols. However, recent evidence suggests that many cases may be associated with the aspiration of potable water (Yu, 1993).

An influenza illness known as Pontiac fever has also been identified, and a number of outbreaks have been documented. It is caused by *Legionella* organisms, but is more benign than Legionnaire's disease (Glick et al., 1978). Q fever is another bacterial infection that can spread in buildings independent of humans (Ayars, 1997), and the bacterium that causes tuberculosis can also be transmitted via air-management systems in closed environments. One of the most studied examples occurred in 1965 aboard a US naval vessel when 7 crew members developed tuberculosis that was spread via the ship's ventilation system (Houk et al., 1968).

Although outbreaks of bacterial infections are more commonly associated with indoor environments, there are also a number of serious viral diseases that can be harboured within buildings. Included are the often deadly but rare viruses that produce haemorrhagic fevers such as the Marburg virus, Ebola virus, and Lassa fever (Ayars, 1997). Many of these agents appear to be spread by aerosolised animal products, particularly urine, and hence may pose particular problems in buildings with rodent infestations.

4. Sick building syndrome

Over the past two decades, numerous field studies on indoor air quality and the sick building syndrome (SBS) have been conducted, mostly in office environ-

Table 10. Common symptoms of sick building syndrome

- Headache and nausea
- Nasal congestion (runny/stuffy nose, sinus congestion, sneezing)
- Chest congestion (wheezing, shortness of breath, chest tightness)
- Eye problems (dry, itching, tearing, or sore eyes, blurry vision, burning eyes, problems with contact lenses)
- Throat problems (sore throat, hoarseness, dry throat)
- Fatigue (unusual tiredness, sleepiness, or drowsiness)
- Chills and fever
- Muscle pain (aching muscles or joints, pain or stiffness in upper back, pain or stiffness in lower back, pain or numbness in shoulder/neck, pain or numbness in hands or wrists
- Neurological symptoms (difficulty remembering or concentrating, feeling depressed, tension, or nervousness)
- Dizziness
- Dry skin

Source: Wallace (1997).

ments. The symptoms of SBS are usually non-specific, and are often somewhat particular to the building being occupied by the workers (Table 10). Amongst employees, symptoms often worsen during working hours, and lessen or disappear after leaving the building. It may be that SBS does not warrant separate attention from the rest of the discussion of indoor air pollutants, as it is unlikely that its mechanisms are much different from those of some well defined pollution problems. However, the study of cases of SBS provides a valuable insight into the problems faced by investigators attempting to find evidence of causal relationships between any indoor air pollution and health.

For a time, exposure to VOCs was thought to be the major cause of SBS. This view was based on the results of experimental chamber studies in which persons exposed to mixtures of VOCs exhibited symptoms commonly associated with SBS (Mølhave et al., 1986). Some epidemiological studies have also supported the hypothesis that VOCs may be important. In a longitudinal study of a 'sick' library building, Berglund et al. (1990) found an association between the reporting of SBS symptoms and temporal variations in VOC concentrations. In a cross-sectional study of SBS symptoms amongst 147 office workers, Hodgson et al. (1991) found that VOC concentrations in the breathing zone of the building occupants were good predictors of mucous membrane irritation, and central nervous system complaints. However, despite these findings, a large number of studies have been unable to find any association between VOC exposures and SBS outbreaks. Indeed, one recent large-scale project actually reported a negative relationship (Sundell et al., 1993). Therefore it appears that, in many cases, VOCs may not be primarily to blame.

Some attention has recently been focused on the role that building ventilation systems may play in SBS (Bourbeau et al., 1997). There is much debate as to whether ventilation may actually be a cure or cause of SBS. Certainly, hot stuffy air is a common complaint in SBS studies, suggesting that efficient ventilation is desirable (Wallace, 1997). However, the exact role of building ventilation in this phenomenon is not well understood, and the results from studies that have been undertaken are often conflicting (Mendell, 1994). Although many previous reports have found reduced symptom prevalence to be associated with increased outside air ventilation (e.g. Nagda et al., 1991; Sundell et al., 1993) a more recent double-blind experiment by Jaakkola et al. (1994) found no significant difference in symptom prevalence amongst the occupants of buildings with ventilation rates between 13 and 42 cfm p^{-1} of outside air.

In an important meta-analysis of six previous studies, Mendell and Smith (1990) examined the relationship between symptom reporting and ventilation provision amongst office workers. Compared with those in naturally ventilated buildings, workers in air-conditioned offices consistently reported an increased prevalence of work related headache (OR = 1.3–3.1), lethargy (OR = 1.4–5.1), and upper respiratory/mucus membrane symptoms (OR = 1.3–4.8). Interestingly, the provision of mechanical ventilation without air conditioning was not associated with higher symptom prevalence.

The fact that some ventilation systems may themselves contribute to the concentrations of indoor pollutants is a possible explanation for the finding of Mendell and Smith (1990). Sources might include bacterial contamination, fungal mycotoxins, antiseptic agents and pesticides, or VOCs from air filtration systems or maintenance ducts. Harrison et al. (1987) found that, compared with the inhabitants of 8 naturally ventilated buildings, the occupants of 19 buildings with mechanical ventilation reported significantly higher frequencies of eye and nose irritation, headaches, attacks of lethargy, and dry skin. However, a number of more recent studies (e.g., Sundell, 1994) have failed to confirm these findings, and whilst recent research by Vincent et al. (1997) did find some evidence of an association between the mode of building ventilation and SBS symptoms amongst Parisian office workers, the authors concluded that ventilation only explained a relatively small proportion of non-specific symptoms.

Despite the voluminous research undertaken over the last two decades to investigate the reasons for SBS outbreaks, it has been estimated that no specific cause has been identified in over 75% of cases (Rothman and Weintraub, 1995). This may, in part, be due to methodological considerations. One problem is that currently available pollutant sampling techniques are costly and suffer from some technical limitations. This has often meant that only relatively limited data on the actual concentrations of pollutants that may be important

in SBS have been available (Bardana, 1997). This is particularly the case for studies of VOCs, where many investigations have measured concentrations of only a defined subset of a limited number of compounds. Potentially important VOCs such as aldehydes and carboxylic acids of low molecular weight are rarely analysed quantitatively (Wolkoff et al., 1997).

Another methodological limitation that has hampered many studies is the lack of a standardised system for the reporting and diagnosis of symptoms. Numerous different questionnaires have been used to elicit information on SBS symptoms, meaning that measures of symptom prevalence are often not comparable between studies. Furthermore, the lack of standards for objective findings or diagnostic criteria, coupled with the non-specific nature of many symptoms, may lead to the potential introduction of bias by respondents who already have strong beliefs concerning the origin of their illness (Burton, 1997).

Whilst perceived sensory irritation is often a primary determinant in the assessment of the quality of indoor air, there is an increasing recognition that any stimulus–response may be greatly influenced by the complex environment surrounding the exposure, which can include the social context or the perceiver's mental state (Dalton, 1996). To illustrate this, Dalton et al. (1997) exposed three groups of individuals to 800 ppm of acetone for a 20 min period. Before the exposure, the groups were given different information about the consequences of long-term exposure to the substance. The study found that subjects given a positive characterisation of the consequences of exposure perceived significantly less odour and irritation during exposure than did subjects given a negative or neutral characterisation ($R^2 = 0.72$, $p < 0.001$).

Whilst a reliance on the self-reporting of symptoms for the assessment of indoor air quality might be problematic, there are uncertainties in formulating more objective measures. For example, Hempel-Jorgensen et al. (1998) examined the relationship between self-reported symptoms, cytological changes, and conjunctival hyperaemia (redness) in the eyes of subjects exposed to mixtures of VOCs. Whilst perceived irritation and hyperaemia did increase with exposure, cytological changes in samples of eye fluid were unrelated to both the level of exposure and the reporting of perceived irritation.

Methodological issues aside, it may be that the likely causal mechanisms that underlie most outbreaks of SBS have not yet been correctly identified. The theory discussed earlier that reactions involving VOCs and substances such as ozone may be more important than direct exposure to the VOCs themselves could be significant (Wolkoff et al., 1997). If reactive mechanisms do prove to be one cause SBS, the prudent siting of electrostatic office equipment (such as photocopiers, and laser printers) that produce these substances may be particularly significant in preventing outbreaks (Wolkoff et al., 1997).

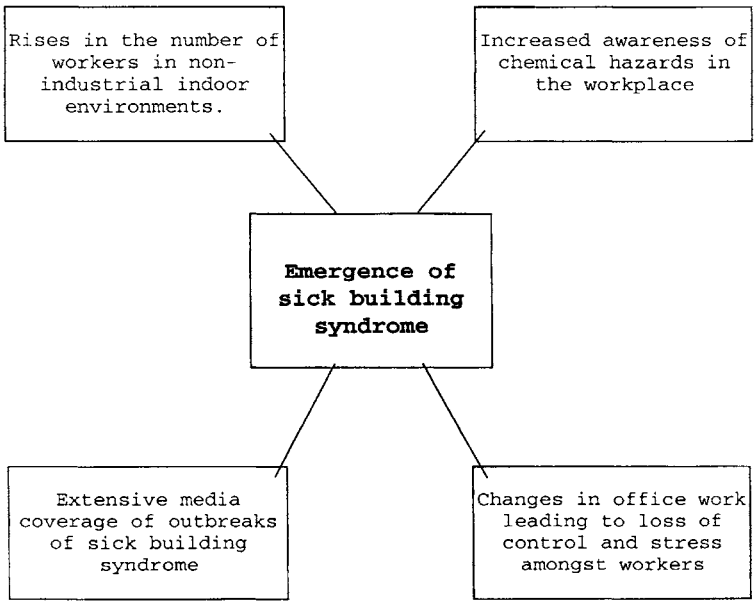
Another possibility that has been suggested is that the symptoms of SBS may, in some part, be associated with the syndrome of multiple chemical sen-

sitivity (MCS). Proponents of MCS believe that symptoms may be produced by exposure to many chemically distinct substances at very low doses (Nethercott, 1996). The concept of MCS has largely evolved out of concern over food allergies during the 1960s (Shorter, 1997), and recent research by Meggs et al. (1996) suggests that around one-third of US citizens consider themselves to be suffering from MCS. The existence of MCS could explain why outbreaks of SBS have been recorded in buildings with extremely low concentrations of individual pollutants. However, MCS is not generally recognised by traditional medicine and a number of recent studies have failed to identify any clinical mechanism that might underlie it (Wolf, 1996). For example, Simon et al. (1993) were unable to detect any immunological differences between 34 control subjects and a sample of 41 patients reporting MCS. Interestingly, the authors did, however, find that MCS patients were more likely to show symptoms of anxiety and depression. Hence it has been argued that MCS may have a much stronger psychological than clinical basis (Shorter, 1997).

The debate over MCS highlights the importance that psychological factors may play in SBS. The symptoms commonly associated with SBS also characterise the diagnostic criteria for generalised anxiety disorder and panic disorder. Furthermore, individual characteristics and non-toxicological job related factors often appear to be important correlates of SBS (Burton, 1997).

Research on occupational stress has repeatedly shown that factors such as work overload, role ambiguity, and low status can be important in determining the health and well-being of employees (Kivimäki, 1996). In their study of the significance of psychosocial factors on the prevalence of SBS symptoms amongst 450 Swedish office workers, Erikssonm and Höög (1993) found that the combination of a high workload, a perception of low control of the personal environment and workload, and lack of support from supervisors generated an increased risk of symptom reporting. In an extensive study of 4900 office employees in the USA, Wallace et al. (1993) also found that a heavy workload and conflicting demands in work were associated with headaches, eye irritations, dizziness, and nasal and chest discomfort. Amongst women, the researchers found that low educational status was also significant. Letz (1990) has produced a theoretical framework that explains how psychological mechanisms may explain the observed increases in the number of outbreaks of SBS in the last two decades. It is outlined in Fig. 1.

It seems, then, that SBS probably has a multi-factorial aetiology, in which chemical, physical, biological, and psychosocial factors all interact to produce symptoms and discomfort (Baker, 1989). Psychosocial factors may act as modifiers of the responses of individuals to chemical, physical, and biological challenges in the indoor environment (Lahtinen et al., 1998). It seems probable that work stress functions as a mediating factor around environmental symptoms, possibly by altering the body's sensitivity to perceived physical demands and



Source: Adapted from Letz (1990)

Figure 1. Possible socio-psychological factors responsible for the recent emergence of sick building syndrome.

toxic exposures (Hedge et al., 1992). However, despite the multitude of theories that have been proposed, the exact processes of SBS are still unknown, and many uncertainties remain.

5. Discussion

Differences in factors such as climatic conditions, lifestyles, and construction habits mean that indoor environments vary from region to region, and country to country. However, despite this variation, many industrialised societies are increasingly facing a growing number of very similar indoor air quality problems. Although considerable advances have been made in our knowledge and understanding of the sources of indoor air pollution, progress on the definition of the effects on human health and welfare has been on a much smaller scale (Stolwijk, 1992).

There is undoubtedly a need for more high-quality research to be undertaken to investigate further the links between indoor air quality and health. At present, projects concerning indoor air receive only around one tenth of

the funding allocated to those studying the health effects of outdoor pollution (Seltzer, 1995). Given the evidence for the health impacts of indoor pollutants, this imbalance clearly needs to be addressed. Future research must fully acknowledge the complex nature of the indoor environment. Indoor air quality is the result of an intricate series of interactions involving many indoor and outdoor ventilation, microbiological, toxicological, and physical systems. Their interdependence may lead to the instigation of a multitude of synergistic processes. Hence the current paradigm, of categorising air quality problems according to single independent fields of expertise, will probably not provide solutions to most of unanswered questions.

Protocols must be developed and validated for analytical techniques that can consider, not just individual classes of pollutants, but a whole range of combinations and mixtures. It must be recognised that research limited to knowledge of a single class of pollutant may produce interesting results in a laboratory situation, but is unlikely to explain accurately the human health problems that will result from several interacting contaminants (Seltzer, 1995). All of the required advances will demand more refined methods for measuring and characterising pollutant emissions and concentrations. At present, techniques for measuring the complex combinations of pollutants that exist in many indoor environments are not well developed. In the future, it is important that this information is available in a useable form that will facilitate study, and aid the development of pollution mitigation strategies (Farland, 1991).

There is also a need for research to refine our understanding and measurement of health impacts. A lack of knowledge has led to particular difficulties in identifying many of the causes of sick building syndrome. Studies of sick buildings also highlight the need to consider mechanisms that lie outside the traditional toxicological dose–response relationships. Among the factors that should be encompassed in the development of future methodologies are the impact of environmental variables such as temperature and lighting, as well as psychological indicators that include job dissatisfaction, controllability, and stress.

As the health effects of indoor air pollution become better understood, the legal and regulatory aspects of the issue will require careful consideration. Nowadays, indoor air quality and pollution are becoming concerns not just of scientists, but also of the legal community.

The legal community's focus is on efforts to control the quality of indoor air through the passage of legislation and the development of regulations, as well as to impose liability upon those who are allegedly responsible for the indoor environment (Tokarz and Potterfield, 1994). It is important to remember that we live in an increasingly litigious society, and the number of claims from workers who feel their health has been damaged by the contamination of their surroundings is increasing.

With the introduction of the US Clean Air Act in 1970, the US EPA was faced with the task of setting national ambient air quality standards for a set of common outdoor air pollutants. No consideration of the health risks of indoor contamination was included. The Act was substantially revised in 1990, but still does not provide the EPA with clear authority to address indoor pollution. Complicating the public-policy issue is the shortage of solid data. Within the United Kingdom, the National Environmental Protection Plan, published in 1996, identifies indoor air quality as a key area for action (DOE, 1996), but provides no mandatory regulations. Indeed, outside the Scandinavian counties, Holland, and Germany, policy on indoor air quality issues is generally poorly developed in Europe as a whole (Harrison, 1997).

The introduction of new legislation may go some way towards reducing levels of contamination. However, it is important to remember that simple interventions by home-owners and building operators can also have major impacts. Measures such as the avoidance of smoking indoors, the deployment of dehumidifiers in damp environments, and the use of natural ventilation are important. Here, education will be critical. Good education programmes are the key. The US Master Home Environmentalist Program (MHEP) is an example of a public education programme designed to reduce exposures amongst homeowners by providing personal contact with experts in the home (Roberts and Dickey, 1995). Volunteers operating the scheme provide free home assessments and follow-ups to families wishing to reduce their exposure to indoor air pollutants, house dust, and chemicals. Programmes such as this may be one of the most efficient ways to teach control methods to motivated individuals.

It might be necessary to re-evaluate the way in which buildings are designed and constructed in the future. Whilst problems with existing homes and offices are often difficult to overcome, architects and builders may need to be educated so as to create less contaminated structures. Simple measures, such as the selection of building materials and indoor fittings on the principle that the emission of pollutants such as VOCs and formaldehyde should be as low as reasonably achievable may prove beneficial. More radical ideas, such as a change of attitude with respect to the energy efficiency of buildings may also be required. Increased fuel costs could be offset against potential reductions in the considerable financial burden from the treatment of both acute and chronic indoor pollution related diseases. In these areas, a policy shift from a reactive to a proactive mode with respect to building regulations may be required. However, improved data on the links between building design and health will have to precede the development and enforcement of any new construction legislation.

Meeting the challenges that are posed by indoor air pollution will take some time. A strong commitment will be required within both the public and private sectors, and the provision of adequate funding will be essential. In the

meantime, we must manage our indoor environments using the best scientific advice available. Our accumulating knowledge of the sources, exposures, and health impacts of indoor pollutants will need to be put to good use to ensure that future indoor environments provide the healthiest possible conditions.

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Appendix

Since the completion of the Millennium Review, there have been a number of important references published in the areas of respiratory health (and in particular the effects of passive smoking on infants), and radon and cancer risk.

1. Respiratory health

A series of ten articles reviewing the health effects of passive smoking, largely written by Cook and Strachan, but with contributions from Anderson, Carey, and Coultas, have been published in the journal *Thorax*. All except one have focussed on the effects of parental smoking on children's health. The findings of these papers have been based on the results of a systematic review and quantitative meta-analysis of the existing literature.

In the first article of the series, Strachan and Cook (1997) examined the effects of parental smoking on lower respiratory tract illness in infancy and early childhood. When using a broad definition of symptoms that included wheeze, bronchitis, bronchoilitis or pneumonia, they found an odds ratio of 1.57 (95% CI 1.42–1.74) for the presence of symptoms in infants aged under two years

if either parent smoked. If the mother smoked, the odds ratio was raised to 1.72 (95% 1.55–1.91). The elevated maternal influence may be explained by the children spending more time with their mothers, but could also suggest that prenatal exposure to the products of smoking might compromise lung development and hence place infants at increased risk of contracting illnesses.

Cook and Strachan (1997) assessed the relationship between parental smoking and the prevalence of asthma and respiratory symptoms in school-aged children. They found an odds-ratio of 1.21 (95% CI 1.10–1.34) for cases where either parent smoked. This increased to 1.50 (95% CI 1.29–1.73) where both parents were smokers, illustrating the probable importance of environmental exposures. For either parent smoking, elevated symptom prevalence was also detected for the prevalence of wheeze (odds ratio = 1.24, 95% CI 1.17–1.31), cough (odds ratio = 1.40, 95% CI 1.27–1.53), phlegm production (odds ratio = 1.35, 95% CI 1.13–1.62), and breathlessness (odds ratio = 1.31, 95% CI 1.08–1.59).

Interestingly, some recent studies that have investigated the relationship between current parental smoking and the prevalence of respiratory symptoms in large samples of infants, have reported conflicting results. Research by Hu et al. (1997) in Chicago found that wheezing was inversely associated with current maternal smoking, whilst Forsberg et al. (1997) detected an inverse association between current smoking in the home and the prevalence of asthma attacks and dry cough. It may be that these negative associations are due to the avoidance of smoking amongst the parents of symptomatic children (Cook and Strachan, 1999). Further research is undoubtedly required in this area.

The link between common lower respiratory tract illnesses in infancy and asthma in later childhood is controversial. The *Thorax* review series presented evidence to suggest that parental smoking was more influential as a cause of early "wheezy bronchitis" than of later onset asthma. However, recent Norwegian research (Siersted et al., 1998) suggests that teenagers with asthmatic symptoms are less likely to receive a diagnosis of asthma if their parents smoke. This raises the possibility that the association between Environmental Tobacco Smoke (ETS) and asthma may have been underestimated by studies relying of physician diagnosed symptoms.

In contrast to much of the previous literature, Strachan and Cook (1998a) did not find evidence of a positive association between allergic sensitisation and parental smoking, either before or after birth. This may be because many studies included asthma and wheezing as measures of sensitisation, and these might be related to exposure to ETS by mechanisms other than allergy. In their review of the relationship between exposure to ETS and the natural history and severity of asthma and wheezing, Strachan and Cook (1998b), also found inconsistent results amongst both case-control and longitudinal studies. Early prognosis appeared to be worse if parents smoked, whereas the persistence

of symptoms into the teens and twenties was less common in the children of smokers. However, a meta-analysis of the relationship between bronchial reactivity, as assessed by challenge test and exposure to (largely maternal) ETS was more consistent, suggesting a small but statistically significant increase in bronchial hyper-responsiveness amongst the children of smoking mothers (Odds ratio = 1.29, 95% CI 1.10–1.50) (Cook and Strachan, 1998).

Cook et al. (1998) examined the relationship between parental smoking and infant's spirometric index scores. They concluded that maternal smoking was generally associated with small but statistically significant deficits in spirometric indices (for example a 0.9% (95% CI −1.2–0.7) reduction in forced expiratory volume in one second). The authors concluded that this was almost certainly a causal relationship, and that much of the effect might be due to maternal smoking during pregnancy which appears to have a particularly large influence on neonatal lung mechanics.

Anderson and Cook (1997) reviewed the evidence for a relationship between sudden infant death syndrome (SIDS) and passive smoking, and reported an odds ratio of 2.13 (95% CI 1.86–2.43) for maternal smoking. An effect of this magnitude is unlikely to be due to residual confounding. Based on the limited available evidence where mothers claimed to be non-smokers during pregnancy, it seems that postnatal exposure may play the most important role.

Strachan and Cook (1998c) reviewed studies of the relationship between parental smoking, and infant middle ear disease and adenotonsillectomy. Where either parent smoked, they reported odds ratios of 1.48 (95% CI 1.08–2.04) for recurrent otitis media, 1.38 (95% CI 1.23–1.55) for middle ear effusion, and 1.21 (95% CI 0.95–1.53) for referral for glue ear. All these results suggest that exposure to ETS may be a significant risk for the development of middle ear problems.

In summarising the findings from their meta-analyses, Cook and Strachan (1999) compared their results with those of a review undertaken by the Californian Environmental Protection Agency (CEPA) (Dunn and Zeise, 1997). Many of the conclusions were remarkably consistent, although the CEPA interpreted the inconsistent data on allergic sensitisation, as providing evidence that exposure to ETS may be a risk. The CEPA study did evaluate the effects of ETS on cystic fibrosis, a condition not covered in the *Thorax* articles. It reported that hospital admissions for cystic fibrosis exacerbations were significantly related to parental smoking in 3 of the 4 studies reviewed, and that ETS exposure also appeared to be linked to other measures of disease severity.

In the only article in the *Thorax* series that is not concerned with infants, Coultas (1998) examined the relationship between passive smoking and the risk of adult asthma and chronic obstructive pulmonary disease (COPD). Although the available literature is limited by difficulties in measuring a dose-response relationship, the survey found evidence that adults regularly exposed

to ETS at home or in the workplace appear to have a 40–60% increased risk of asthma compared to non-exposed individuals. A weak relationship was also detected between passive smoking and the occurrence of COPD, although the literature contains much variability in the adopted definitions of COPD.

In an important article published in *Allergy*, Garrett et al. (1999) examined the relationship between indoor exposure to formaldehyde and the prevalence of sensitisation to common aero-allergens amongst a sample of 148 Australian infants aged between 7–14 years. Formaldehyde measurements were taken with passive samplers on four occasions between March 1994 and February 1995. A respiratory questionnaire was completed, and skin prick tests were performed.

The median indoor formaldehyde level was 15.8 $\mu g\ m^{-3}$. Marginally higher mean formaldehyde levels were recorded in the bedrooms of atopic compared to non-atopic children (19 $\mu g\ m^{-3}$ versus 16.4 $\mu g\ m^{-3}$, $p = 0.06$). The age-sex adjusted odds ratio for atopy with an increase in bedroom formaldehyde levels of 10 $\mu g\ m^{-3}$ was 1.40 (95% CI 0.99–2.04), and this remained after controlling for passive smoking, the presence of pets, and concentrations of other indoor air pollutants. There were significant differences in the average number of positive skin prick tests between formaldehyde exposure levels (1.2 positive tests for $< 20\ \mu g\ m^{-3}$ versus 4 positive tests for $> 50\ \mu g\ m^{-3}$, $p = 0.004$). The average relative allergen wheal size for the largest allergen wheal was also associated with formaldehyde concentrations (0.4 allergen wheal ratio for $< 20\ \mu g\ m^{-3}$ versus 1.2 allergen wheal ratio for $> 50\ \mu g\ m^{-3}$). The findings of Garrett et al. (1999) suggest that low-level exposure to formaldehyde may increase the risk of allergic sensitisation to common aeroallergens in children. The mechanism could be associated with formaldehyde-induced changes to the upper respiratory tract, possibly via increased permeability of the respiratory epithelial layer or suppression of mucosal immune defences.

2. Radon and cancer risk

There has been an interesting debate in the literature concerning the relationship between population-level residential radon exposures and the prevalence of lung cancer. The debate largely arises from the publication of a report by Cohen (1997) of a strong negative relationship between residential radon measurements and US county rates of male and female lung cancers. This direction of relationship is contrary to what might be expected from studies on miners.

Goldsmith (1999) examined Cohen's findings, and claimed a number of flaws in the original study design. In particular, they argued that Cohen's results are confounded by population density (residential radon levels are higher in suburban areas leading to a negative association with population density,

whilst population density is positively associated with county-level lung cancer prevalence).

Using a more sophisticated approach to that of Cohen, Bogen (1998) fitted a cytodynamic two-stage (CD2) cancer model to age specific lung cancer mortality data and estimated radon-exposure in white females aged 40+ years who resided in 2821 US counties between 1950–1954. The results of this model suggest that residential radon exposure may have a non-linear U-shaped relation to lung cancer mortality risk, and hence linear extrapolations could be inaccurate. The presence of a U-shaped relationship has also been detected in field measurements by Tóth et al. (1998), in their study of residential radon exposure and lung cancer risk amongst women in Mátraderecske, Hungary. They found the lowest cancer risk was actually amongst women living in medium-high radon concentrations of between 110 and 165 Bq m^{-3}. Bogen (1998) postulates that the mechanism of such a relationship may be associated with the influence of radiation exposure on the growth kinetics of pre-malignant foci. In particular the competition between extinction (via alpha-induced cell killing) within pre-existing pre-malignant foci, and the induction of new pre-malignant foci might be important.

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