

1. INTRODUCTION AND BACKGROUND

The Air Resources Board (ARB) prepared this report to the Legislature on indoor air quality in response to requirements of Assembly Bill 1173 (Keeley, 2002; Health and Safety Code Section 39930; see Appendix I). As required by the legislation, this report summarizes the best scientific information available on indoor air pollution, including:

- Common indoor pollutants and their sources.
- The potential health impacts of indoor pollutants and associated costs.
- Existing regulations and current industry practices.
- Options for mitigation in schools, non-industrial workplaces, homes, and other indoor locations.
- State and federal efforts related to control of biological and radiological substances.

In preparing the report, the ARB is consulting with the Department of Health Services (DHS), the Office of Environmental Health Hazard Assessment (OEHHA), the California Energy Commission (CEC), the Department of Industrial Relations (DIR, Cal/OSHA), relevant industries and other stakeholders, and interested members of the public. On April 4, 2003 a public workshop was conducted to inform stakeholders about the report and the anticipated schedule, and to solicit comments from them. On June 30, 2004, another workshop will be held to receive comments on the draft report. A website and an email list serve have been established to keep stakeholders informed of progress on the report preparation. The website is available at <http://www.arb.ca.gov/research/indoor/ab1173/ab1173.htm>, which also includes information on how to sign up for the email list serve. The draft report will be reviewed by a University of California scientific peer review panel and considered by ARB Board members before being forwarded to the Legislature.

This report reflects key points from the large body of knowledge that has been generated about indoor air quality since it became a concern in the 1970s. Researchers worldwide have made great progress in identifying indoor pollutants and understanding their relationship to human exposure and resultant health effects. Several international conferences are now held annually, and several journals are now available that are devoted exclusively to the field of indoor air quality. This report summarizes key findings of the most recent literature, with emphasis on the major trends identified by multiple investigators.

1.1 INDOOR POLLUTION POSES A SIGNIFICANT HEALTH RISK

State and federal comparative risk projects have repeatedly ranked indoor pollutants and sources in the high-risk categories of their analyses relative to other environmental health problems. In the 1994 *California Comparative Risk Project* (CCRP, 1994), the Human Health Committee ranked risks by two different methods: by sources and media, and by pollutant. In the sources and media ranking, the “residential and consumer product releases to air (indoor air)” category was ranked in the high risk group, along with some outdoor air pollution categories. In the pollutant rankings, ETS, radon, PM, and VOCs were all ranked in the high-risk category, and carbon monoxide and lead were ranked in the medium risk category. The results of the CCRP had a somewhat more substantial scientific basis than most other comparative risk projects, because it was based to a greater degree on actual measurement data, and used distributions of exposure and risk, rather than just population averages. Additionally, it was reviewed by an external scientific review committee.

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The U.S. EPA's 1987 national comparative risk project also ranked indoor air pollution high relative to other environmental areas: indoor pollution other than radon was ranked fourth among the top 13 national environmental problem areas assessed for cancer risk (U.S. EPA, 1987a). Radon was ranked first, but the risk from radon has been subsequently re-examined and reduced. A number of other states and regions have conducted comparative risk projects as well, with results similar to those of U.S. EPA and California.

The high ranking of indoor pollution relative to other environmental problems is not surprising, because there are numerous sources of pollutants indoors, indoor air concentrations of some pollutants are often high enough to pose a health risk, and people spend most of their time indoors. The total quantity of air pollutants emitted indoors is much less than that emitted by outdoor sources. However, once emitted, indoor pollutants are much less diluted than pollutants from outdoor sources. When this factor is combined with the fact that Californians, like others from industrialized nations, spend most of their time indoors, there is a much higher likelihood that people will be exposed to pollutants emitted indoors than those emitted outdoors.

One investigator has calculated that pollutants emitted indoors have a 1000-fold greater chance of being inhaled than do those emitted outdoors (Smith, 1988). Thus,

The Rule of 1000:

A typical pollutant release indoors is 1000 times as effective in causing human exposure as the same release to urban air.

KR Smith, in *Air Pollution: Assessing Total Exposure in the United States, Environment*, 30 (8): 10, 1988.

reducing indoor emissions by a given amount could have a proportionately greater impact on reducing exposure than would reducing outdoor emissions by that same amount. Regulation of outdoor sources such as motor vehicles and industrial plants has notably reduced many outdoor pollutant levels in California. Now, there are significant gains to be achieved in public health protection from reductions in indoor source emissions and other measures that might be taken to reduce indoor concentrations and exposures.

The health effects of indoor pollutants range from irritant effects to respiratory disease, cancer, and even sudden death. Indoor sources of pollutants are numerous, such as building materials, consumer products of all types, combustion appliances, and even some so-called "air fresheners". Common indoor activities such as cooking, cigarette smoking, burning candles, and vacuuming also generate pollutants. The health effects of indoor pollutants, and indoor pollutant sources and concentrations in California, are discussed in Chapter 2 of this report.

1.2 PEOPLE SPEND MOST OF THEIR TIME INDOORS

A key reason indoor pollution is so critical to health is that Californians, like others from industrialized nations, spend most of their time indoors—about 87%, on average. So, if pollutants are present indoors, there is a high likelihood that people will be exposed to them. As shown in Table 1.1, California adults spend an average of about 62% of their time in their home. Children spend even more time in their home: infants up to 2 years of age spend 85% of their time inside the home, on average. Thus, the home is a critical exposure microenvironment for all, and especially for children.

Table 1.1: Average Percent of Time Californians Spend in Major Locations^{1, 2}

AGE	AVERAGE PERCENT OF TIME			
	Inside the Home	Other Indoors	Outdoors	Inside a Vehicle
Children				
0 - 2	85	4	7	4
3 - 5	76	9	10	5
6 - 11	71	12	13	4
All Children (0 - 11)	76	10	10	4
Adults and Teens	62	25	6	7

¹From: *Study of Children's Activity Patterns* (Wiley et al., 1991a, ARB Contract no. A733-149; Phillips et al., 1991).

²From: *Activity Patterns of California Residents* (Wiley et al., 1991b, ARB Contract no. A6-177-33; Jenkins et al., 1992a).

1.3 CHILDREN'S HEALTH – VULNERABILITY IMPLIES INCREASED INDOOR AIR QUALITY IMPACT

In recent years there has been an increasing awareness that children may be more susceptible than adults to the harmful effects of air pollution. Additionally, children are more highly exposed to some indoor contaminants than are adults. Thus, children are likely at greater risk from indoor pollution than adults.

- **Children's physiology and developing lungs and bodies make them more susceptible to chemicals that affect development and lung function.** Children's immune systems are not fully developed and their growing organs and structures are more easily harmed. For example, lead is more easily deposited in the fast-growing bones of children. Pollutants that cause irritation or inflammation in the airways are more likely to obstruct a child's airways because they are narrower than airways of an adult. Results from the ARB-funded Children's Health Study indicate the lungs of children in high-pollution communities develop more slowly and move air less efficiently than lungs of children in low-pollution communities (Gauderman et al., 2000). When exposed to ETS, children are at greater risk than adults for developing lower respiratory tract infections, bronchitis, pneumonia, fluid in the middle ear, and asthma symptoms (NCI, 1999). Additionally, young children appear to be more susceptible to the effects of environmental tobacco smoke (ETS) than older children (IOM, 2000).
- **Infants and children inhale more air relative to their size** than do adults at a given level of activity (ARB/OEHHA 2000, Adams 1993). Additionally, children often breathe through their mouths, bypassing the filtering effect of the nose and allowing more pollutants to be inhaled. Thus, for the same amount of time spent in a given location/activity as an adult, a

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child will inhale a greater quantity and thus receive a greater dose of the chemicals in the air. Children also have a larger lung surface area per unit of body weight, relative to adults: thus, chemicals are absorbed more rapidly (Thurlbeck, 1988; Plopper and Thurlbeck, 1994).

- **Children's activities bring them into close proximity to indoor sources.** First, infants and young children spend more time indoors at home (see Table 1.1) than do adults. Additionally, younger children spend more time near indoor sources such as operating gas stoves, e.g., near the parent while cooking (Phillips *et al.*, 1991), leading to higher exposures to nitrogen dioxide and other cooking emissions.
- **Also, children spend more time on floors and more frequently put fingers and objects into their mouth** (Zartarian *et al.*, 1998; Zartarian and Leckie, 1998). This can lead to additional dermal and ingestion exposure to airborne TACs deposited and adsorbed onto floor dust, such as lead and other toxic metals, PAHs, and pesticides. Because the breathing zone for an infant or small child is several inches to a foot or so above the floor, while that for adults is several feet above the floor, particles stirred up by activity may become available to be inhaled by the child but are not elevated in the breathing zone of adults (Bearer 1995).
- **Formaldehyde levels are generally higher in California schools than in office buildings and homes (as shown in Figure 2.4).** Thus, children may receive greater exposures to formaldehyde due to exposures at school.

Concerns about children's health prompted the California Legislature to pass the Children's Environmental Health Protection Act (SB 25, Escutia) in 1999. This Act requires Cal/EPA to specifically consider children when setting Ambient Air Quality Standards (AAQS) and reviewing pollutants for identification or regulation as Toxic Air Contaminants (TACs). As required by this legislation, ARB has re-evaluated the AAQS for particulate matter and ozone, and the standard for nitrogen dioxide is currently under review. OEHHA has identified dioxin, lead, polycyclic organic matter, diesel exhaust particles, and acrolein as the top 5 priority TACs that may impact infants and children the most (OEHHA, 2001). The ARB is beginning to assess the adequacy of existing control measures for these compounds relative to the health of children.

1.4 ENVIRONMENTAL JUSTICE CONSIDERATIONS RELATIVE TO INDOOR AIR QUALITY

Senate Bill 115 (1999) directs the California Environmental Protection Agency to design an environmental justice mission statement for boards, departments, and offices within the agency. State law (California Government Code § 65040.12c) defines environmental justice as the fair treatment of people of all races, cultures, and incomes with respect to the development, adoption, implementation, and enforcement of environmental laws, regulations, and policies. In response to SB 115, the ARB approved *Policies and Actions for Environmental Justice* in December, 2001. This document establishes a framework for incorporating environmental justice into all ARB programs, policies, and regulations (ARB, 2001a).

The ARB has taken several steps to directly and indirectly address environmental justice concerns related to indoor air quality. Special air-monitoring studies have been conducted in classrooms, homes, and at schools in communities located near industrial sources of pollution and/or heavy vehicular traffic. Preliminary results indicate pollutant levels in the selected communities are similar to levels in other communities. To assure that information is available

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to all stakeholders, documents designed for public education, such as fact sheets and a school advisory, have been published in both English and Spanish.

Research is needed to determine the complex relationship between socioeconomic-status (SES), environmental factors, and health status, particularly as they relate to indoor air pollution. To date, only factors related to the prevalence of asthma have received substantial study. The prevalence of asthma appears to be more strongly correlated with lower socioeconomic status than with race and ethnicity (IOM, 1999). Yet, California data show that African Americans, American Indians, and Alaska Natives experience a higher prevalence of lifetime asthma than other groups in the population (Meng *et al.*, 2003). Auinger *et al.* (2003) report that the increase in asthma prevalence is greatest in children from lower income homes, probably due to an increase in prevalence of allergic conditions in their homes. Dust mites and cockroaches are important triggers for asthmatics that are more likely to be present in urban settings (IOM, 1999).

Some researchers have concluded that people of color and people with low income are more likely than others to live in substandard housing and experience disproportionate exposure to indoor air pollutants (e.g. Krieger and Higgins, 2002). Dampness, mold, dirty carpeting, and pest infestations are often components of substandard housing, each leading to associated health problems, especially allergy symptoms and exacerbation of asthma attacks in asthmatics. Homes with moisture and dampness contributing to mold growth are more likely to be older, poorly insulated buildings that lack central heating, and are often located in low-income communities (IOM, 2004). Those who live in substandard housing are exposed to more pests, as well as pesticides to get rid of them (Flynn *et al.*, 2000). Pesticides adsorb onto particles and accumulate in the carpet, where children and others can be exposed to them. In an effort to improve their environment, a higher percent of low-income individuals use room fresheners – products that may introduce additional toxic chemicals to the indoor environment (Wiley *et al.*, 1991b).

Children living in urban areas are disproportionately exposed to lead, primarily from lead-containing paint that has been used on older houses. Exposure to lead in both house dust and air takes a toll on children. Research indicates blood lead levels are higher for poor and minority children in central cities (IOM, 1999).

Exposure to formaldehyde is likely not equivalent across the California population. Formaldehyde levels historically have been highest in mobile homes, which are more often occupied by lower income families. Efforts are underway to limit formaldehyde emissions from composite wood products through an Air Toxics Control Measure (ATCM). This measure would reduce health risks associated with formaldehyde emissions from composite wood products in all types of construction.

2. HEALTH IMPACTS, SOURCES AND CONCENTRATIONS OF INDOOR AIR POLLUTANTS

Emissions from indoor sources contribute to exposure and risk in two ways. Some are emitted in substantial quantities over extended periods from large surfaces, such as formaldehyde from composite wood products made with urea-formaldehyde resin. These sources contribute to elevated indoor pollutant levels in many buildings where a large portion of the population spends their time. Large numbers of individuals can be affected due to the large quantity of indoor emissions. Other products, such as aerosol sprays or solvents, emit much smaller quantities of pollutants, and are used by a subset of the population. However, those who use such products use them in such a manner that the chemicals emitted are released near the user's breathing zone (area near the nose and mouth). A high concentration of the chemical is consequently inhaled during product use before the chemical has a chance to become more dilute in the air.

Hundreds of substances representing a range of chemical, physical, and biological species have been identified as indoor air pollutants. Indoor air pollutants include volatile organic compounds, inorganic gases, particulate matter, and complex mixtures such as environmental tobacco smoke (ETS). Indoor concentrations of many pollutants are often higher than outdoor concentrations due to the use of indoor pollutant sources in a confined space. Some pollutants such as formaldehyde, radon, asbestos, cigarette smoke, and mold have received a substantial amount of study. However, only a fraction of indoor pollutants have been identified and fully characterized.

Indoor air pollution can cause a variety of adverse impacts on human health, from irritant effects to respiratory disease, cancer, and death. The major health effects that can occur from exposure to common indoor pollutants are indicated in Table 2.1. These effects have a major impact on Californians' health each year, and most can be readily avoided. Asthma, cancer, irritancy, and sick building syndrome are discussed below. Later in this chapter the more common indoor pollutants, their sources, and indoor concentrations are discussed.

2.1 KEY HEALTH IMPACTS

2.1.1 Asthma

Asthma is a chronic inflammatory lung disease that results in partially reversible constriction of the airways. It is characterized by episodes of wheezing, shortness of breath, or coughing that may occur at any time. Asthma is a critical health issue because of its negative impact on the quality of life, increased morbidity and mortality, and substantial economic impact.

A number of indoor pollutants can cause or exacerbate asthma and chronic bronchitis. Indoor biological agents are clearly associated with these diseases; however, it is clear that biological agents alone cannot explain the tremendous increase in asthma over the last few decades. The recent rise in asthma incidence has been too rapid to be attributed to genetic factors and biological allergens alone: indoor and outdoor air pollution have been identified as potentially important contributors to the increase of asthma (Platts-Mills and Carter, 1997; Duhme *et al.*, 1998; Karol, 2002).

Table 2.1. Sources and Potential Health Effects of Major Indoor Air Pollutants

POLLUTANT	MAJOR INDOOR SOURCES	POTENTIAL HEALTH EFFECTS
Organic Chemicals (benzene, styrene, para-dichlorobenzene, chloroform, perchloroethylene, methylene chloride, phthalates, etc.)	Solvents; glues; cleaning agents; pesticides; building materials; paints; treated water; moth repellents; dry-cleaned clothing; air fresheners	Cancer; eye, nose, throat irritation; possible worsening of asthma; headaches; at high levels; loss of coordination; damage to liver, kidney and brain
Formaldehyde, other aldehydes	Composite wood products such as plywood and particleboard; furnishings; wallpaper; durable press fabrics; paints	Cancer; eye, nose, and throat irritation; headache; allergic reactions; worsening of asthma
Particulate Matter	Cigarettes, wood stoves, fireplaces, cooking, candles, aerosol sprays, house dust	Increased mortality and hospital admissions; lung cancer; eye, nose, throat irritation; increased susceptibility to sinus and respiratory infections; bronchitis; worsening of asthma
Environmental Tobacco Smoke (ETS)	Cigarettes, cigars, and pipes	Respiratory irritation, bronchitis and pneumonia in children; lung cancer; heart disease; worsening of asthma
Carbon Monoxide	Unvented or malfunctioning gas and propane appliances, wood stoves, fireplaces, tobacco smoke	Headache; nausea; angina; impaired vision and mental functioning; fatal at high concentrations
Nitrogen Dioxide	Unvented or malfunctioning gas appliances, other combustion appliances	Worsening of asthma; eye, nose, and throat irritation; increased respiratory disease in children
Radon	Soil under buildings, ground-water, construction materials	Lung cancer (especially in smokers)
Biological Agents (bacteria, fungi, house dust mites, animal dander; cockroaches)	House dust; pets; bedding; poorly maintained air-conditioners, humidifiers and dehumidifiers; wet or moist structures; furnishings	Allergic reactions; asthma; eye, nose, and throat irritation; humidifier fever, influenza, and other infectious diseases
Polycyclic Aromatic Hydrocarbons (PAH)	Cigarette smoke, cooking, burning wood	Cancer; gene mutation
Endocrine Disrupters (phthalates, DDT, chlordane, o-phenylphenol, heptachlor, PBDEs)	Plastics, computers, pesticides, flame retardant	Mimic or block natural effects of hormones (estrogen and others); developmental abnormalities

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Over the past three decades, asthma prevalence has been on the rise in industrial nations, and the mortality rate due to this disease has doubled (Karol, 2002). The Centers for Disease Control and Prevention (CDC) reported that in the year 2000, asthma was responsible for 4,487 deaths in the U.S., as well as approximately 465,000 hospitalizations, 1.8 million emergency department visits, and 10.4 million visits to physicians across all age groups (CDC, 2003). Currently, about 7.2% of adults in the U.S. have asthma (CDC, 2003), while 11.9% of Californians, or 3.9 million people, have asthma (CHIS, 2003). The reason for the higher prevalence rate in California is not known, but California also has a higher rate of mortality from asthma than the national rate. Children have been particularly hard-hit; from 1980 to 1994, there was a 160% increase in asthma in those up to 4 years of age in the U.S. (Mannino *et al.*, 1998). In California, asthma prevalence is highest among children 12-17 years of age. Asthma is the leading cause of school absenteeism in children, and results in missed workdays and lost productivity in adults. Asthma affects all races and ethnic groups, and both genders. However, in California prevalence is highest in American Indians, African Americans, Alaska natives, and Pacific Islanders. Although the causes for the observed increase in asthma prevalence are unknown, indoor air pollution has been identified as a contributing factor.

In a recent report by the National Academy Institute of Medicine, entitled *Clearing the Air: Asthma and Indoor Air Exposures* (IOM, 2000), the Committee on the Assessment of Asthma and Indoor Air examined the scientific literature relating indoor air pollutants and other factors to asthma. Their key findings are shown in Tables 2.2 and 2.3.

Table 2.2. Indoor Exposures and Exacerbation of Asthma

<p>Sufficient Evidence of Causal Relationship</p> <ul style="list-style-type: none"> • Cat • Cockroach, House Dust Mite • ETS (preschoolers)
<p>Sufficient Evidence of an Association</p> <ul style="list-style-type: none"> • Dog • Fungi or molds, Rhinovirus • NO₂, NOX
<p>Limited or Suggestive Evidence of an Association</p> <ul style="list-style-type: none"> • Domestic birds • <i>Chlamydia pneumoniae</i>, <i>Mycoplasma pneumoniae</i>, Respiratory Syncytial Virus (RSV) • ETS (school-aged children and adults) • Formaldehyde, Fragrances
<p>Possible, but Insufficient Evidence</p> <ul style="list-style-type: none"> • Cow, Horse, Rodents • <i>Chlamydia trachomatis</i>, Endotoxins • Houseplants, Pollen • Pesticides, Plasticizers, VOCs • Insects other than cockroaches

Source: IOM, 2000

Table 2.3. Indoor Exposures and Development of Asthma

<p>Sufficient Evidence of Causal Relationship</p> <ul style="list-style-type: none"> • House dust mite
<p>Sufficient Evidence of an Association</p> <ul style="list-style-type: none"> • ETS (preschoolers)
<p>Limited or Suggestive Evidence of an Association</p> <ul style="list-style-type: none"> • Cockroach (preschoolers) • Respiratory Syncytial Virus (RSV)
<p>Possible, but Insufficient Evidence</p> <ul style="list-style-type: none"> • Cat, Cow, Horse, Dog, Domestic birds, Rodents • Cockroaches (except for preschoolers) • Fungi or mold, <i>Chlamydia pneumoniae</i>, <i>Chlamydia Trachomatis</i>, <i>Mycoplasma pneumoniae</i>, Endotoxins • Houseplants, Pollen • NO₂, NO_x • Pesticides, plasticizers, VOCs, Formaldehyde, Fragrances • ETS (school-age and older)

Source: IOM, 2000

The committee found that, in addition to the known biological asthma triggers such as mold, house dust mites, and animal dander, chemicals such as ETS, nitrogen dioxide, and formaldehyde can exacerbate asthma in sensitive individuals. Evidence for an association between formaldehyde exposure and wheezing is limited, often due to confounding factors such as exposure to multiple pollutants. Exposure to NO₂ and airway responses occur at high levels of NO₂ (400 to 700 ppb), levels that can occur in poorly ventilated kitchens with gas appliances in use. The scientists determined that there is sufficient evidence to conclude that ETS can exacerbate asthma in preschool-aged children (IOM, 2000), and identified an association between ETS exposure and the development of asthma as well (IOM, 2000). However, the scientists found only limited or suggestive evidence of ETS as an asthma trigger in older children and adults, and insufficient evidence to consider it a causal factor in these groups (IOM, 2000). The Committee noted that studies have linked outdoor PM with respiratory problems, and that outdoor fine particles, PM_{2.5}, readily enter the indoor environment. Studies that have addressed asthma specifically have found some evidence for asthma exacerbation due to outdoor PM exposure (IOM, 2000).

Although the Committee on the Assessment of Asthma and Indoor Air found inadequate evidence to link VOCs with exacerbation of asthma, more recent studies have linked indoor residential VOCs to asthma and its symptoms. Delfino (2002) published a review of the epidemiological evidence for links between air toxics and asthma. Delfino cites Swedish studies that showed that self-reported asthma prevalence in schoolchildren increased with increasing VOC levels, and asthmatic adult symptoms occurred in association with toluene, C₈-aromatics,

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terpenes, formaldehyde, and limonene. Adult asthma prevalence, wheeze, and blood eosinophil concentrations were higher in newly painted homes, consistent with higher VOC levels (particularly 2,2,4-trimethyl 1,3-pentanediol diisobutyrate and formaldehyde) measured in such homes. In another European study cited by Delfino, elevated levels of benzene and styrene were associated with respiratory infections in newborns at risk for atopic disease, and wheezing was related to house painting and carpet installation during the first year of life.

Delfino's (2002) review identified several links between asthma symptoms and indoor air pollutants, especially formaldehyde:

- A relationship exists between formaldehyde exposure and both occupational and non-occupational asthma.
- Children in homes with formaldehyde concentrations greater than 41 ppb are more often diagnosed with asthma and chronic bronchitis.
- Adults show more wheeze, chronic cough, and lower peak expiratory flow in homes with higher formaldehyde concentrations.
- Non-asthmatics in homes with formaldehyde levels of 50 ppb or higher have elevated levels of expired nitric oxide, a marker for lower airway inflammation.

Other reviews also support the association between VOCs and symptoms of asthma (Duhme *et al.*, 1998; Leikauf, 2002).

Studies conducted in the workplace also demonstrate an association between asthma symptoms and chemicals used indoors. Between 1993 and 1997, 12% of confirmed cases (236 of 1,915) of work-related asthma in California, Massachusetts, Michigan, and New Jersey were related to exposure to cleaning products. Chlorine bleach was identified as the cleaning agent associated with the greatest number of cases. Exposures were greatest in medical settings, schools, and hotels (all non-industrial workplaces) with janitors, cleaners and housekeepers experiencing the highest incidence of the disease (both new-onset and work-aggravated cases of asthma: 80% were new onset; Rosenman *et al.*, 2003).

2.1.2 Cancer

Many indoor air pollutants are known or suspected carcinogens. Formaldehyde, benzo(a)pyrene and other polycyclic aromatic hydrocarbons (PAHs), environmental tobacco smoke, benzene, chlorinated solvents such as tetrachloroethylene, and radon are a few of the identified carcinogens commonly found in indoor air, some at levels much higher than outdoor levels. Several technical documents provide summary data regarding the carcinogenic potential of these pollutants. Cancer unit risks and potency factors for 121 of the 201 carcinogenic substances for which emissions must be quantified in the California Air Toxics Hot Spots program are provided in the *Technical Support Document for Describing Available Cancer Potency Factors* (OEHHA, 2002; <http://www.oehha.ca.gov/air.html>). The U.S. EPA's Integrated Risk Information System (IRIS) provides a similar list of cancer potency values (<http://www.epa.gov/iris/>).

Risk assessments are conducted to estimate the increased risk of health problems in people who are exposed to different amounts of toxic substances. Risk is dependent on the amount of a pollutant people actually inhale, which depends on the air concentration of the pollutant in a given environment, the length of time a person is in that environment, and the person's breathing rate during that time. Since people spend the majority of their time indoors, moderate and high concentrations of indoor pollutants generally translate to elevated risk.

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Several field studies have measured indoor concentrations of carcinogenic chemicals in California (Wallace *et al.*, 1988; Wallace *et al.*, 1991a; Sheldon, *et al.*, 1992a; Avol *et al.*, 1996; and others). Results of these studies indicate that carcinogens are routinely found in most homes, often at levels higher than outdoor levels, due to the presence of indoor sources. Table 2.4 lists key pollutants from these California studies and indicates their cancer classification.

Table 2.4. Common Carcinogenic Indoor Air Pollutants

Compound	Cancer Status* U.S. EPA class
Acetaldehyde	California Prop 65 carcinogen
Acrolein	Group C, possible human carcinogen
Benzene	Group A, human carcinogen
Benzo(a)pyrene	Group B2, probable human carcinogen
Chloroform	Group B2, probable human carcinogen
<i>p</i> -Dichlorobenzene	California Prop 65 carcinogen
Di-2-ethylhexylphthalate	Group B2, probable human carcinogen; (IARC classification 2B, possible human carcinogen)
Formaldehyde	Group B1, probable human carcinogen
Methylene chloride	Groups B2, probable human carcinogen
Perchloroethylene	Group B2/C, probable human carcinogen
Styrene	Under consideration, possible human carcinogen
1,1,1-Trichloroethane	Group D, not classifiable
<i>o</i> -Xylene	Group D, not classifiable
<i>m,p</i> -Xylene	Group D, not classifiable

*Source: ARB, 1997a

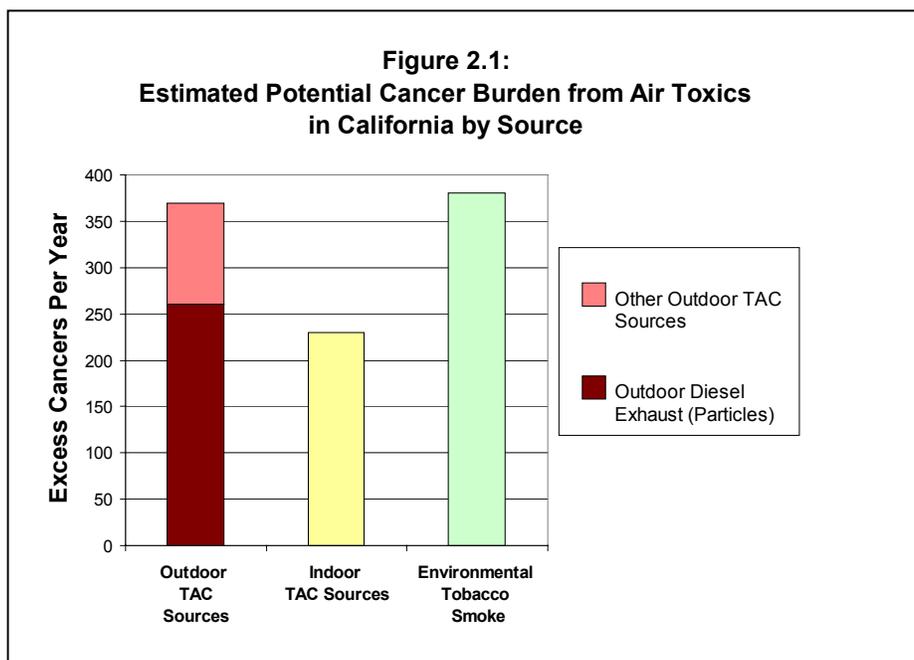
ARB staff estimate that about 230 excess cancer cases occur per year in California from indoor sources of toxic air contaminants, not including the excess cancer from exposure to radon, environmental tobacco smoke (ETS), and some other indoor carcinogens. This estimate of 230 cancer cases per year is based on risk estimates from the 1994 California Comparative Risk Project (CCRP, 1994), updated to reflect reduced exposure and risk from indoor formaldehyde. Those levels are estimated to have decreased by about one-half since the 1980s studies on which the 1994 estimate was based (see Appendices II and III). The chemicals with the highest estimated risk in the CCRP were formaldehyde (found in many building materials and consumer products) and para-dichlorobenzene (used in mothballs and air fresheners). Other chemicals included were perchloroethylene (used in dry-cleaning), chloroform (a by-product of water chlorination and use of chlorine in spas and washing machines), trichloroethylene, benzene, 1,3-butadiene, styrene, benzo(a)pyrene, and di-2-ethylhexylphthalate (a plasticizer). The latter

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organic chemicals are found in many different consumer products and building materials, and some also are produced by combustion processes (such as when cooking food or burning wood).

Since the time the studies used for the CCRP were conducted, some levels of indoor pollutants are estimated to have decreased while others likely have increased. However, there are insufficient new data available to refine the 1994 estimates for those chemicals. Because changes likely have occurred in both directions and are likely to be relatively small, and because the 1994 CCRP estimates did not include all known indoor carcinogenic pollutants (methylene chloride, other polycyclic aromatic hydrocarbons, and other phthalates were not included, for example), the 1994 estimates remain the best available estimates for the overall cancer risk posed by indoor chemical pollution in California, excluding that from radon gas, environmental tobacco smoke, and asbestos (discussed later in this document). The details of ARB's assessment using the CCRP results are provided in Appendix II.

This estimate of 230 excess cancer cases approaches the cancer burden from diesel exhaust particles, which is estimated to result in 260 excess cases per year in California, and exceeds the cancer risk for other outdoor pollutants, estimated at about 110 cancer cases per year (Figure 2.1). Indoor air cancer risk is also nearly two-thirds of the total cancer risk estimated for outdoor pollutants. It is estimated that cancer risk due to diesel exhaust particles will decrease 75% by 2010 (ARB, 2000a), leaving indoor air as a predominant source of air pollution cancer risk.



Exposure to environmental tobacco smoke (ETS) makes a significant contribution to the cancer burden from air pollution as well. Current exposure and risk estimates for ETS are not available; therefore estimates from the mid-1990s are shown here (OEHHA, 1997). Those ETS risk levels are similar to the total outdoor burden; however, because workplace exposure has decreased to nearly zero since the mid-1990s, and the prevalence of smoking has decreased substantially as

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well, the current cancer burden from ETS is expected to be substantially lower than shown in this graphic. Nonetheless, the contribution of ETS will remain significant, because some individuals including some children, are still exposed to substantial levels of ETS.

Other recently developed risk estimates demonstrate the carcinogenic risk posed by indoor pollutants present at average concentrations. In the absence of indoor standards, Hoddinott and Lee (2000) applied U.S. EPA Superfund risk assessment methodology to selected VOCs to determine if indoor concentrations produce significant risks. Indoor VOC concentrations from two studies completed in the 1980s (Wallace, 1987; Cohen *et al.*, 1989) were used to determine the level of risk associated with VOCs measured inside residences. U.S. EPA methodology indicates a “safe” level of risk is one increase in cancer incidence per 10,000 to 1,000,000 persons (U.S. EPA, 1989). Hoddinott and Lee (2000) calculated that the risk for adults and children, based on average levels found in homes, exceeded the acceptable risk level of one in a million for a number of pollutants. Those pollutants are found in dry-cleaned clothing, ETS, cleaning agents, glued carpet, gasoline, and degreasers. The authors concluded that “Chemical concentrations resulting from ‘off-gassing’ from normal household activities and materials can result in a health risk estimate that exceeds the benchmark used at hazardous waste sites”.

2.1.3 Irritant Effects

Many indoor pollutants cause eye, nose, and throat irritation (Wolkoff and Nielsen, 2001). Tables of irritant thresholds have been developed to provide guidance for non-irritating levels of pollutants at workplaces and in the home (Molhave, 1991; Devos *et al.*, 1990). Aldehydes, terpenes, and ozone are known irritants, as well as various other VOCs and oxidants. Formaldehyde vapors are highly irritating to the eye, nose, throat, and respiratory track. Acute exposure to these pollutants can cause headaches, difficulty breathing, and nausea. Chronic exposure is associated with respiratory symptoms and eye, nose, and throat irritation (U.S. EPA, 1994; ARB, 1992).

Terpenes are reactive chemicals (e.g., α -pinene, *d*-limonene, myrcene) that are frequently used in cleaning products for their favorable odor characteristics and solvent properties. Terpenes are associated with irritation, though it is believed that the irritant chemicals are a product of the reaction of terpenes with oxidants (Wilkins *et al.*, 2001; Weschler and Shields, 1997). Ozone, the primary ingredient in smog and a strong oxidant, also irritates the respiratory system, causing coughing, throat irritation, or a burning sensation in the airways. Ozone irritation can lead to a feeling of chest tightness, wheezing, and shortness of breath (ARB, 2000b).

Sick Building Syndrome (SBS) is a term used to describe a collection of irritant and neurological effects that occur while occupants are in a building, that generally disappear when affected people are out of the building. Specific causes of SBS have not yet been firmly identified. The most common symptoms include eye irritation, congested nose, headache, fatigue, difficulty concentrating, and dry skin (Tenbrinke *et al.*, 1998). SBS differs from building-related illness (BRI; see Biological Contaminants section) in which an identifiable factor causes a specific illness such as bacteria causing Legionnaires’ disease or humidifier fever.

Identifying a cause for SBS has been elusive. Mendell (1993) conducted a review of the epidemiological literature related to SBS. In reviewing 32 studies, he found consistent findings linking SBS symptoms with air-conditioning, carpets, more workers in a space, Video Display use, and ventilation rates at or below 10 liters/second/person. With specific causes unidentified,

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Mendell stressed the importance of using prudent design, operation, and maintenance practices to prevent sick building symptoms.

Tenbrinke *et al.* (1998) reported a new approach for using VOC exposure metrics as predictors of SBS. These authors were able to confirm a link between exposure to low level VOCs and SBS symptoms. Apte and Daisey (1999) used the methodology developed by Tenbrinke to identify an association between mucous membrane symptoms and photocopiers. Apte and Daisey also identified a relationship between sore throat symptoms and fresh paint.

Investigators are exploring potential reactions between indoor pollutants in an effort to more positively identify the cause of sick building health effects. Several reactions such as between ozone and unsaturated hydrocarbons, ozone and nitrogen oxides, and free radical reactions may lead to the formation of more irritating indoor compounds such as formaldehyde. These reactions can also lead to the production of submicron particulate matter, aldehydes and ketones with lower odor thresholds and greater irritancy than precursors, carboxylic acids such as formic acid and acetic acid, and free radicals (Weschler and Shields, 1997). Pollutants with reactive double bonds such as terpenes and alkenes react with ozone and nitrogen oxides to produce airway irritation similar to that of formaldehyde. Fan *et al.* (2003) confirmed the reaction of ozone with d-limonene and ozone with α -pinene to generate formaldehyde and ultrafine particles. It has been demonstrated that such mixtures can cause eye irritation, presumably due to the formation of new products through reactive chemistry (Wolkoff and Nielsen, 2001; Wallace, 2001).

2.2 Traditional (Criteria) Pollutants

Several pollutants for which outdoor air quality standards have been established occur at elevated levels indoors as well, and can pose a serious health risk. Ambient PM has been associated with premature mortality and serious respiratory and cardiovascular effects in numerous studies. CO can cause near-term death with high exposures of relatively short duration. All of these pollutants can impose serious, non-fatal health impacts: NO₂, from indoor combustion appliances, can harm the lungs and other mucous membranes and cause respiratory disease, and ozone can have similar effects at elevated levels. The health effects, indoor sources, and indoor air concentrations are discussed below.

2.2.1 Particulate Matter

Particulate matter (PM) is broadly defined as any non-gaseous material suspended in the air. PM can include solid material (i.e. dust), liquid material (i.e. a sprayed aerosol), or a combination of solid/liquid materials (i.e. a hydrated vehicle exhaust particle). PM is generally classified by its size. PM_{2.5} refers to all suspended matter having aerodynamic diameters less than 2.5 microns (μm : one μm = one millionth of a meter) and is commonly referred to as 'fine' PM. PM₁₀ refers to all suspended matter having aerodynamic diameters less than 10 μm . PM between 2.5 μm and 10 μm is commonly referred to as 'coarse' PM. Both federal and state ambient (outdoor) air quality standards incorporate these size distinctions. Recent studies suggest that PM_{2.5} mass may be the best indicator for predicting potential health effects resulting from ambient PM exposure (Williams *et al.*, 2000a,b,c; Schwartz and Neas, 2000), although some recent studies have specifically linked health impacts to the coarse fraction.

A substantial portion of indoor particles originate outdoors from outdoor sources. However, there are numerous indoor sources of PM as well. These include combustion devices and

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activities such as stoves, fireplaces, cigarette smoking, cooking, and candle burning, all of which can produce PM with harmful components similar to those from outdoor air (Lofroth *et al.*, 1991). Indoor particles also include fibrous materials, pollen, mold spores and fragments, and tracked-in soil particles (Wallace, 1996a). These particles become trapped in carpets and have been shown to persist for a very long time, due to the difficulty of removing all particles by vacuuming, and may be re-suspended into the air. Some can trigger asthma attacks and allergy symptoms, as discussed above. Others can have a mix of toxic components such as PAHs, lead, and pesticides adsorbed onto them; these components may contribute to serious health effects such as cancer and developmental effects.

Major epidemiologic studies have shown a strong association between ambient (outdoor) PM concentrations and increased mortality and morbidity (e.g., Dockery *et al.*, 1993; Pope *et al.*, 1995) and an increase in the rate of death from cardiovascular and respiratory disease (Samet *et al.*, 2000). Because they are based on particle size and include a mix of particles from combustion sources, soil, and other sources, these effects from ambient PM and their magnitude are directly relevant to assessing the potential risk from indoor PM. These effects of ambient PM are summarized below.

2.2.1.1 Mortality

Both acute and chronic PM exposure have been associated with an increased risk of mortality, or premature death, primarily in older adults with preexisting heart and/or lung disease. Studies conducted in California, the U.S., and in diverse cities worldwide suggest that risk of mortality increases about 0.25 to 3.5 per cent with each 10 $\mu\text{g}/\text{m}^3$ increase in DAILY PM_{2.5} concentration (e.g., Burnett and Goldberg, 2003; Dominici *et al.*, 2003; Fairley, 2003; Goldberg and Burnett, 2003; Moolgavkar, 2003; Schwartz, 2003; Ponka *et al.*, 1998). Long-term cohort studies suggest that the increase in risk of mortality is about 4 per cent with each 10 $\mu\text{g}/\text{m}^3$ increase in ANNUAL mean PM_{2.5} concentration (Dockery *et al.*, 1993; Pope *et al.*, 1995). Meta-analyses of earlier mortality studies suggest that the effects on mortality are fairly consistent (Ostro, 1993; Dockery and Pope, 1994; Schwartz, 1994), regardless of where the study was performed.

2.2.1.2 Morbidity (Non-lethal Health Impacts)

Several hundred studies have been published examining the association between various measures of ambient PM and a variety of adverse health effects other than premature death. The health outcomes associated with PM include hospitalization and emergency room visits for respiratory or cardiovascular disease; respiratory symptoms, including asthma symptoms; restrictions in activity and school absenteeism; and reduced lung function and other effects in children. Although these effects are not necessarily as serious as death, they affect a greater proportion of the population, and thus have a major impact on public health. The key results from some of the morbidity studies include the following:

- **Hospitalization and serious respiratory disease**
 - ✓ Studies consistently report associations between both PM_{2.5} and PM₁₀ and hospital admissions for respiratory and cardiovascular illness (e.g., Atkinson *et al.*, 2003; Sheppard *et al.*, 1999; Sheppard, 2003; Ito, 2003; Zanobetti and Schwartz, 2003; Moolgavkar, 2003; Le Tertre *et al.*, 2003). These effects have been reported mainly for people over age 65 who already have some form of cardiopulmonary disease. Respiratory causes of admission include pneumonia, COPD and asthma, while cardiovascular causes have included general cardiovascular disease, congestive

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heart failure, stroke, and ischemic heart disease. Overall, PM₁₀ has been associated with an estimated increase in risk of hospitalization of 1.25% - 5% per 10 $\mu\text{g}/\text{m}^3$ increase in the daily mean PM₁₀ concentration for respiratory endpoints, and 0.3% to 2.6% for cardiovascular endpoints.

- ✓ Associations have also been reported between PM₁₀ and PM_{2.5} and emergency department visits, primarily for asthma exacerbation, which may or may not result in hospital admissions (e.g., Lipsett *et al.*, 1997; Delfino *et al.*, 1997).

• Respiratory symptoms

- ✓ Studies have associated ambient PM₁₀ and PM_{2.5} exposure with asthma and respiratory symptoms, for example cough, phlegm, chest pain, or wheeze (e.g., Delfino *et al.*, 2003; Mortimer *et al.*, 2002; Schwartz and Neas, 2000), asthma exacerbation (e.g., Whittemore and Korn, 1980), and use of asthma medications (e.g., Delfino *et al.*, 1996; Pope *et al.*, 1991).
- ✓ Cellular level effects have also been identified. Exposure to respiratory irritants can result in local airway inflammation, altered epithelial permeability, increased mucus secretion, and bronchoconstriction. Disease states such as asthma and chronic bronchitis can adversely affect particle clearance or removal (e.g., Foster, 1999). Also, the viability and functional integrity of cells in the lungs can be adversely affected by PM exposures (e.g., Soukup and Becker, 2001).

• Work loss, absenteeism, reduced productivity:

- ✓ Ostro (1987) and Ostro and Rothschild (1989) reported an association between PM_{2.5} and respiratory-related restrictions in activity that implied about a 10 to 15% change in reduced activity per 10 $\mu\text{g}/\text{m}^3$ of PM₁₀.
- ✓ Ransom and Pope (1992) reported about a 4% increase in absenteeism per 10 $\mu\text{g}/\text{m}^3$ of PM₁₀ increase at an elementary school in Utah.

• Effects on children

- ✓ Investigators with the ARB-sponsored Children's Health Study found that, among children with asthma, respiratory symptoms increased with increasing particle levels (McConnell *et al.*, 1999). Results also suggest that children who live in communities with high concentrations of PM may have decreased lung function growth compared to children living in communities with lower concentrations of PM (Gauderman *et al.*, 2000; Peters *et al.*, 1999a,b). However, both of these results were also true for NO₂ and acid vapor, and the independent effects of the different pollutants cannot be assessed because of high inter-pollutant correlations. Similar results have also been reported by Horak *et al.* (2002) in Austrian children.
- ✓ Several recent studies have suggested that the unborn may also be at risk of adverse effects from PM pollution, based on statistically significant relationships between outdoor PM concentration and low birth weight (Ritz *et al.*, 2000; Bobak, 2000), prematurity (Bobak, 2000), neonatal mortality (Penna and Duchade, 1991; Woodruff *et al.*, 1997; Bobak and Leon, 1998), and fetal growth retardation (Dejmek *et al.*, 1999). However, except for Ritz *et al.* (2000), these studies have been

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conducted outside the U.S., in areas with higher PM concentrations than those typically observed in the U.S.

Based on this literature, about 6,500 deaths and more than 17,000 serious, non-fatal illnesses (such as hospital admissions and non-fatal asthma attacks) due to outdoor particulate pollution have been estimated by ARB to occur each year in California (ARB/OEHHA, 2002). Indoor PM is comprised of varying proportions of PM of indoor and outdoor origin (discussed below). Because the additional PM burden from indoor sources is generally not well represented in epidemiology studies, indoor PM emissions may be significant contributors to the adverse impacts seen in the epidemiology studies, and may contribute to premature mortality, hospital admissions, chronic bronchitis, and other effects beyond the levels quantified in the epidemiology studies.

ARB recently convened a small panel of indoor air quality and PM experts to review and assess what is known regarding the impacts of indoor PM on health. They concluded that available studies document the known effects of ETS particles and infectious and allergenic indoor biological contaminants, but that few studies have been designed to specifically identify effects of other types of indoor-generated PM. Research is sorely needed in this area. The few scientific studies available provide only suggestive evidence of the health effects of combustion emissions from woodburning and gas stoves and other types of PM. Investigators in one study (Long *et al.*, 2001) of rat alveolar macrophages found that indoor-generated particles triggered greater production of tumor necrosis factor than did a comparable amount of outdoor PM, suggesting that indoor-generated PM may be more bioactive than ambient particles. This may be due to PM emissions from indoor combustion sources being relatively “fresh” (smaller in size and possibly more chemically reactive) when breathed by people indoors compared to the more “aged” PM from outdoor sources (by the time it reaches people), thus having potentially greater toxicity. Additionally, the multiple toxics and respiratory pollutants in the indoor PM mix may make synergistic and cumulative effects more likely, although those cannot yet be quantified. Alternatively, outdoor PM may be much more toxic, due to emissions from complex sources such as diesel trucks and industrial plants.

There are no federal or California standards for indoor PM levels. The National Ambient Air Quality Standards (NAAQS) for PM_{2.5} are 65 ug/m³ (24-hr average) and 15 ug/m³ (annual mean), and for PM₁₀ are 150 ug/m³ (24-hr) and 50 ug/m³ (annual mean). The California ambient air quality standard for PM_{2.5} is 12 ug/m³ (annual mean), and the PM₁₀ standards are 50 ug/m³ (24-hr) and 20 ug/m³ (annual mean). These levels are often exceeded in California's indoor environments, particularly when indoor sources are used or when particle- or aerosol-generating activities occur.

2.2.2 Indoor PM Sources and Emissions

Indoor PM concentrations are typically equal to or higher than concurrently measured outdoor levels (see next section), depending on the sources and activities that are present indoors. Outdoor air infiltration and indoor combustion sources such as smoking and cooking are typically the greatest sources of indoor PM (Wallace, 1996a; Ozkaynak *et al.*, 1996a,b; Brauer *et al.*, 2000; Abt *et al.*, 2000; Fortmann *et al.*, 2001). Prominent indoor sources include cigarettes, woodstoves, and candles; cooking and cleaning activities (Ozkaynak *et al.*, 1996a,b; Abt *et al.*, 2000, 2001; Long *et al.*, 2000); the presence and activities of occupants (Abt *et al.*, 2000, Rodes *et al.*, 2001); the use of personal care products (Conner *et al.*, 2001); and indoor chemical reactions (Weschler and Shields, 1997; Nazaroff and Weschler, 2004). Fibrous

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materials, pollen, mold spores and fragments, and tracked-in and blown-in soil particles also are components of indoor PM (Wallace, 1996a).

The contribution of outdoor PM to indoor PM concentrations can be substantial but highly variable. For residential buildings, the main entry routes of outdoor air are open windows and doors, cracks in the building shell, and mechanical ventilation systems such as swamp coolers and whole house fans. Investigators of a large, population-based study in California, the Particle Total Exposure Assessment Methodology Study (PTEAM Study) estimated that residential indoor PM₁₀, on average, is roughly comprised of about 66% outdoor PM₁₀; 75% for PM_{2.5} (Ozkaynak *et al.*, 1996a,b). In a study of four Boston homes with air exchange rates below 1.0 hr⁻¹, Abt *et al.* (2000) estimated that only 20-43 percent of indoor PM₂ to PM₁₀ were from outdoors, while 63-92 percent of indoor PM 0.02-0.3 μm were from the outdoors. Abt *et al.* (2000) and Long *et al.* (2000) also found that the relative contribution of outdoor PM to indoor levels varied by particle size, with outdoor air generally contributing a majority of the smaller particles measured indoors, while indoor sources contributed more to the coarse (2-10 micrometers) fraction. Because these studies examined primarily older individuals who are less active in their homes than younger families might be, and were conducted on the east coast, the results of these studies may not reflect typical California proportions; however, they show that reductions in outdoor PM levels could have a major effect on indoor concentrations.

Indoor combustion source emissions, such as those from smoking and cooking, are often intermittent and highly variable, but emissions can be very high, resulting in exposures that can have significant impacts on people's total daily exposure to PM (Long *et al.*, 2000). In the PTEAM Study homes with smokers, it was estimated that 30% of the PM_{2.5} mass and 24% of the PM₁₀ mass came from smoking. For homes in which cooking occurred during the monitoring period, 25% of the PM_{2.5} and PM₁₀ was estimated to come from the cooking activity (Ozkaynak, 1996b). These results are consistent with those of previous indoor studies that examined the impact of cigarette smoking on indoor PM levels, and they are consistent with subsequent studies of indoor cooking emissions that confirmed the high impact of cooking on indoor and personal PM levels (Abt *et al.*, 2000; Wallace, 2000b; Brauer, *et al.* 2000; Fortmann, *et al.* 2001).

For example, in an ARB-sponsored study, Fortmann *et al.* (2001) measured indoor and outdoor PM during 32 types of cooking activities with both gas and electric ovens and stovetops. Although concurrent outdoor levels reached only 20 μg/m³, indoor PM levels during and after cooking often exceeded 50 μg/m³, ARB's indoor air quality guideline level and ambient air quality standard for PM₁₀ for 24 hours. Kitchen PM₁₀ levels ranged to more than 1400 μg/m³ during frying, broiling, and baking activities with the gas stove (Fortmann *et al.* 2001). PM levels during cooking with the electric stove were generally lower; however, cooking with the electric stove produced much higher PM levels in two cases -- frying tortillas and stovetop stir-frying. The highest concentrations of PM were produced when using the self-cleaning cycle of the oven for several hours: PM₁₀ was over 3,600 μg/m³ (over 2,000 μg/m³ PM_{2.5}) for the gas stove, and nearly 400 μg/m³ PM₁₀ for the electric stove. Measurement of particle counts during cooking activities and oven cleaning indicated that particles were primarily less than 0.1 μm in size.

The burning of wood, candles, and incense can also be important combustion sources of residential indoor PM, especially in the 2.5 μm size range and below (Wasson *et al.*, 2002; Jetter *et al.*, 2002; Brauer *et al.*, 2000; Guo *et al.*, 2000; Lofroth *et al.*, 1991). Individual candles, for example, released 200-3600 μg/hr of PM₁₀, and 100-1700 μg/hr of lead from the lead wick (Wasson *et al.*, 2002). Guo *et al.* (2000) modeled indoor PM_{2.5} concentrations ranging from 4.3 – 1173 ug/m³ based on PM emission measurements from incense burning. These indoor

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combustion sources produce PM with potentially harmful components similar to those from some outdoor PM combustion sources (Lofroth *et al.*, 1991).

Physical generation or re-suspension of particles also can contribute to airborne indoor PM levels. Indoor surfaces such as carpets and draperies attract and re-emit particles (Thatcher and Layton, 1995, Kamens *et al.*, 1991). Particle concentrations from carpets can be high even in homes where good cleaning practices are used. The particles can become re-entrained in the indoor air when people walk or play on the carpeted surface (Wallace 2000a; Roberts and Dickey 1995; Abt *et al.*, 2000, Vette *et al.*, 2001).

Particles in carpet pose an additional risk to children. House dust particles include vapors, metals, and semi-volatile chemicals, such as pesticides and some PAHs, that have their own toxic properties (Rothenberg *et al.*, 1989; Roberts and Dickey 1995; Lewis *et al.*, 1999; U.S. EPA, 1999a). Particles of house dust with these contaminants can be re-emitted to the air and subsequently inhaled, and may be ingested by children through hand-to-mouth behavior or dermally absorbed when they spend time on the floor (Lewis *et al.*, 1994; Zartarian *et al.*, 1998; Zartarian and Leckie, 1998). For toxics such as lead, floor dust levels can be a major determinant of exposure.

Biological contaminants such as fungi, bacteria, house dust mites, and pollen also contribute to indoor particle concentrations, especially in buildings with moisture problems from flooding or roof leaks that have not been properly repaired. Bioallergens, such as pollen, in outdoor air can also penetrate indoor spaces. Re-entrained road dust may be a particularly important source of bioallergens in both indoor and outdoor air (Miguel *et al.*, 1998).

2.2.3 Indoor and Personal PM Concentrations

Indoor PM concentrations sometimes exceed outdoor air concentrations, due to the presence of indoor sources of PM, as discussed above. Additionally, people's personal exposures to PM often exceed both indoor and outdoor concentrations, primarily because people tend to spend time very near pollutant sources, such as when cooking or cleaning. This has been called the "proximity effect", and reflects the fact that pollutant levels are highest near the source than farther away where emissions have become diluted in the air (McBride *et al.*, 1994). A small portion of the elevated personal exposure levels seen across PM studies also is attributed to the existence of a "personal cloud" of PM surrounding a person, due to re-suspension from clothing, the use of personal care products (Conner *et al.*, 2001), and skin flakes. Williams *et al.* (2000b) estimated a mean personal cloud of 3.1 ug/m³, and Rodes *et al.* (2001) estimated personal clouds of 3 ug/m³ in two separate studies. However, Wallace (2000a) examined a breadth of studies and estimated an average personal PM₁₀ cloud of 30 ug/m³, ranging from 3-67 ug/m³. Personal PM_{2.5} clouds were estimated to range from 6-27 ug/m³.

Consequently, because proximity to PM-emitting sources and activities can significantly increase people's actual exposures, most studies of indoor residential PM concentrations have included, or even focused on, measurement of occupants' personal exposures to PM. These have been obtained using personal samplers worn by study participants for one or more days as they go about their daily routines. These measurements provide a more accurate measure of people's exposure to PM, because PM levels are often higher very near people than at the location of indoor air sampling equipment in a room. Table 2.5 summarizes the major indoor and personal exposure PM studies conducted in the U.S. in recent years.

Table 2.5. Summary of Recent Indoor PM Exposure Studies (means)

LOCATION	GROUP	YEAR	SEASON	PM2.5			REFERENCE
				PERSONAL ($\mu\text{g}/\text{m}^3$)	INDOOR ($\mu\text{g}/\text{m}^3$)	OUTDOOR ($\mu\text{g}/\text{m}^3$)	
7 U.S. Cities	Asthmatic Children	UNK	UNK		27.7	13.6	Wallace et al., 2003
Raleigh, NC	African Americans	2000-2001	multiple	23.0	19.3	19.2	Williams 2003
Seattle, WA	COPD Subjects	2000-2001	multiple	10.5	8.5	9.2	Liu 2003
	Healthy Subjects			9.3	7.4	9.0	
	Asthmatics			13.3	9.2	11.3	
	Coronary Heart Dis			10.8	9.5	12.6	
Boston, MA	Asthmatic Children	2000	multiple		12.3 ¹	9.0 ¹	Brugge 2003
					33.8 ²	13.1 ²	
Detroit, MI	Asthmatic Children	1999-2000	multiple		34.4 ± 21.7	15.6 ± 8.2	Keeler et al., 2002
Baltimore, MD	Elderly	1999	winter	19		5.6	Samat 2000
		1997	summer	27		25	
Boston, MA	Residential Homes	1996	multiple		13.9	11.7	Abt 2000
		1998	multiple		11.9	11.1	Long 2000
Boston, MA	Residential Homes (people with COPD)	1996-1997	winter	21.6	17.2	10.9	Rojas-Bracho 2000
			summer	21.5	17.7	16.4	
Los Angeles	Elderly w/ COPD	1996-1997	winter	24	25	25	Linn 1999
Baltimore, MD	Retirement Facility	1997	winter	34.4	17.4	17.0	Williams 2000a
		1998	summer	12.9	10.0	22.0	Williams 2000c
Fresno, CA	Retirement Facility	1999	winter	13.3	9.7	20.5	Evans 2000
			spring	11.1	8.0	10.1	
Birmingham, AL	Residential Homes ¹	1997-1998	summer	18.6 ± 6.4	16.1 ± 9.6	26.5 ± 9.5	Lachenmeyer and Hidy 2000
			winter	10.0 ± 3.3	11.2 ± 5.4	12.2 ± 5.1	

NOTES: 1) non-smoker homes, 2) smoker homes

LOCATION	GROUP	YEAR	SEASON	PM10			REFERENCE
				PERSONAL ($\mu\text{g}/\text{m}^3$)	INDOOR ($\mu\text{g}/\text{m}^3$)	OUTDOOR ($\mu\text{g}/\text{m}^3$)	
Seattle, WA	COPD Subjects	2000-2001	multiple		14.1	14.3	Liu 2003
	Healthy Subjects				12.6	14.5	
	Asthmatics				19.4	16.4	
	Coronary Heart Dis				16.2	18.0	
Detroit, MI	Asthmatic Children	1999-2000	multiple	68.4 ± 39.2	52.2 ± 30.6	25.8 ± 11.8	Keeler et al., 2002
Baltimore, MD	Elderly	1999	winter	28		7.5	Samat 2000
		1997	summer	34		34	
Boston, MA	Residential Homes	1996	multiple		19.6	17.1	Abt 2000
		1998	multiple		19.4	12.7	Long 2000
Boston, MA	Residential Homes (people with COPD)	1996-1997	winter	40.7	37.3	18.5	Rojas-Bracho 2000
			summer	34.7	28.3	24.8	
Los Angeles	Elderly w/ COPD	1996-1997	winter	35	33	40	Linn 1999
Baltimore, MD	Retirement Facility	1998	summer	11.0	13.5	30.0	Williams 2000c
Fresno, CA	Retirement Facility	1999	winter		15.1	28.2	Evans 2000
			spring	37.3	16.7	28.7	
Riverside, CA	Residential Homes	1996	fall	150	95	95	Ozkaynak 1996a

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The first major study to measure indoor, outdoor, and personal PM concentrations was the PTEAM Study. Investigators measured PM₁₀ and PM_{2.5} for 12-hour daytime and nighttime periods in 178 homes during the fall in Riverside. They found 12-hour daytime personal PM₁₀ concentrations to be about 50% higher than simultaneously measured daytime residential indoor or outdoor concentrations. Daytime personal concentrations averaged 150 µg/m³, while indoor and outdoor concentrations both averaged about 95 µg/m³ (Clayton *et al.*, 1993, Ozkaynak *et al.*, 1996a,b). Most importantly, 12-hour daytime personal PM₁₀ concentrations exceeded the California 24-hour ambient air quality standard level of 50 µg/m³ for about 90% of the monitoring days, and exceeded the federal PM₁₀ standard level of 150 µg/m³ for 25 percent of the monitoring days. During nighttime, personal PM₁₀ concentrations decreased and were similar to concurrent indoor and outdoor concentrations (roughly 80 µg/m³), reflecting the importance of the proximity of people to PM sources during normal daytime activities in determining personal exposure concentrations.

Studies conducted since the PTEAM study in other locations and seasons have measured average personal PM₁₀ concentrations from 11-68 µg/m³, and average personal PM_{2.5} concentrations ranging from 9-34 µg/m³. Average indoor concentrations of PM₁₀ have ranged from about 13-52 µg/m³, with indoor PM_{2.5} concentrations from 7-34 µg/m³. However, peak indoor levels have been high. For example, in a seven city study, Wallace *et al.* (2003) found that in all cities, at least 2% of all 1-hr measurements exceeded 1000 µg/m³. Long *et al.* (2000) measured indoor PM concentrations as high as 473 µg/m³ in study homes. These elevated levels reflect significant indoor source emissions.

Several studies have been conducted to examine the exposure of sensitive populations to PM. These have reported on PM₁₀ and PM_{2.5} exposures for subsets of individuals with chronic obstructive pulmonary disease (COPD), coronary heart disease, and asthma (Liu *et al.*, 2003; Williams *et al.*, 2000a,c; Rojas-Bracho *et al.*, 2000; Linn *et al.*, 1999). Liu *et al.* (2003) found that mean personal PM_{2.5} concentrations were higher than indoor and outdoor concentrations for each group in Seattle, and that PM₁₀ personal concentrations were higher than indoor and outdoor concentrations for asthmatics. Williams *et al.* (2000a,c) also found personal PM_{2.5} concentrations higher than indoor and outdoor concentrations in a Baltimore retirement facility in the winter of 1997. Rojas-Bracho *et al.* found that mean personal PM₁₀ and PM_{2.5} concentrations were above indoor and outdoor PM₁₀ and PM_{2.5} concentrations for 18 COPD patients in Boston.

Linn *et al.* (1999) monitored 15 COPD patients for PM_{2.5} exposures and 15 for PM₁₀ exposures in Los Angeles during the fall and winter. Unlike other studies, this study found personal concentrations similar to those indoors or outdoors. The authors suggest that the lack of increased personal PM in these subjects with severe COPD may be due to reduced personal activity, less time spent outside of the home, and other reasons. The pooled correlation of personal PM concentrations to ambient concentrations at a monitoring station were quite low.

A recently completed study by Suh (2003) has increased our understanding of Californians' exposures to PM in a sensitive subpopulation. The investigators examined the relationships among outdoor and indoor concentrations and personal exposures across different seasons. They examined daily PM exposures of a group of 15 individuals with COPD in Los Angeles over seven sequential days in the summer-fall and/or winter. Personal, indoor, and outdoor PM_{2.5}, NO₃⁻ and elemental carbon (EC) concentrations varied by season, with the exception of outdoor NO₃⁻. For summer-fall and winter PM levels, respectively, the personal PM_{2.5} means (19.6, 25.1 µg/m³) and maxima (63.5, 137.8 µg/m³) were higher than the indoor means (16.9, 18.1 µg/m³) and maxima (49.5, 94.8 µg/m³) and outdoor means (13.5, 19.3) and maxima (56.5, 53.5

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ug/m³). For NO₃⁻ and EC, higher outdoor (2.8-3.1 ug/m³), as compared to indoor (1.1-1.7 ug/m³) and personal (1.2-1.6 ug/m³) levels, were found in both seasons, reflecting the fact that motor vehicles are their major source and that loss of NO₃⁻ may occur indoors due to its high reactivity. NO₃⁻ and EC comprised a small fraction of personal, indoor, and outdoor PM_{2.5} (max. 28.5% and 17%, respectively). The indoor PM levels were low compared to those measured in studies of the general population, consistent with the subjects' limited personal activity and very little time spent near smoking, cooking, vehicles, or other major PM sources. Personal PM was highly correlated with indoor PM, and indoor and outdoor PM correlations were significant as well. Statistical modeling results indicated that significant predictors of higher personal PM were time near ETS, location near a major road, higher population density, cooking activity, and location in an inland area (vs. coastal).

A companion study to the above study by Suh and Koutrakis (2004) is nearing completion. The same investigators are using a similar design for 16 healthy persons in Los Angeles during the summer or winter, and it involves even more detailed characterizations of the PM levels and ventilation characteristics of the subject's homes. This study is focusing on determining the contribution of outdoor concentrations to personal exposures, and on diurnal variations of personal and indoor PM. The results from this study are expected to provide important new information on the relationships between outdoor and indoor concentrations as well as personal exposures for several components of PM, for healthy persons in southern California.

Far fewer studies have been conducted in public buildings than in residences. PM concentrations in public and commercial buildings appear to often be lower than ambient concentrations. Lower indoor PM concentrations in public and commercial buildings are due to the use of particle filters in mechanical ventilation systems, un-openable windows, and the lack of many indoor sources typically present in residences. However, as with residences, the presence of indoor sources in public and commercial buildings can produce indoor concentrations that exceed concurrent ambient concentrations, especially if smoking is allowed in the building. For example, in a study of 38 commercial buildings in the Pacific Northwest, Turk *et al.* (1987) found that buildings where smoking was permitted had average indoor PM levels 3.5 times higher than concurrent outdoor levels and indoor levels in buildings where smoking was prohibited. Sheldon *et al.* (1988) measured PM in six buildings in the eastern U.S., and found indoor PM concentrations generally lower than outdoors where there was no smoking, but much higher indoor concentrations where smoking was allowed. A more recent study, U.S. EPA's BASE study of office buildings across the U.S., should yield useful, more recent information on PM levels inside medium and large office buildings.

Lillquist *et al.* (1998) reported indoor and outdoor PM₁₀ measurements in three Utah hospitals over one winter season. Significant variability in indoor PM levels was found both among room types and among hospitals, and the relationship between indoor PM₁₀ levels and outdoor levels was highly variable. The ICUs had significantly lower PM₁₀ levels than other types of rooms, after adjusting for hospital differences. Thus, the most critically ill individuals may experience some protection in hospitals from ambient PM; however, in general, hospitals do not offer regular protection from ambient PM.

2.2.4 Carbon Monoxide

Carbon monoxide (CO) is a colorless, odorless, tasteless gas. It is a product of incomplete combustion, emitted from sources such as vehicles (in exhaust), gas and propane appliances, woodstoves, kerosene heaters, and cigarettes. CO can trigger acute health effects, even death, at very high levels, or flu-like symptoms and other effects at lower levels over periods of time.

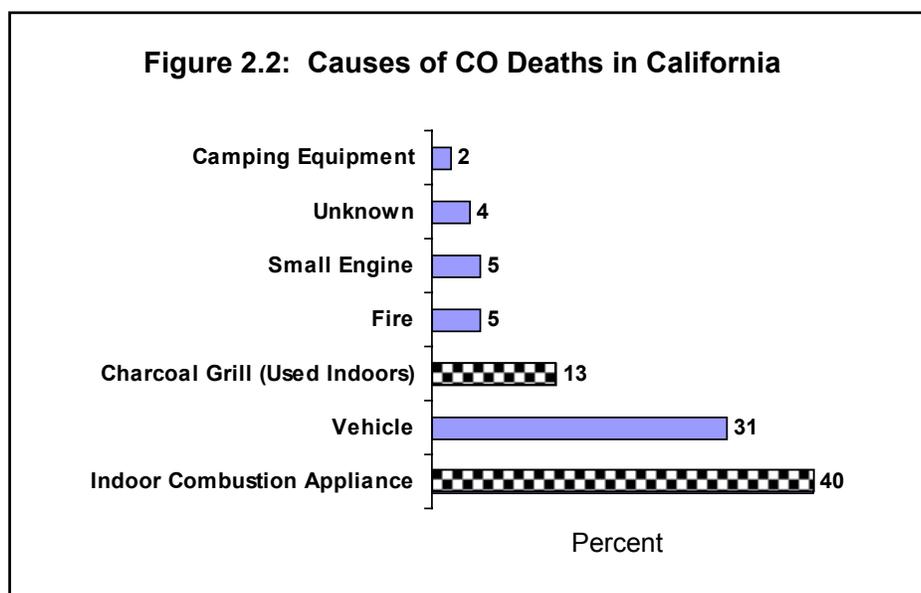
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CO is regulated in the outdoor, but not the indoor, environment. Transient elevated concentrations in outdoor places such as tunnels and parking garages also are not widely regulated. The state of California has an 8-hour average ambient air quality standard of 9.0 ppm (10 mg/m³) and a one-hour standard of 20 ppm (23 mg/m³). These are also ARB's recommended indoor guideline levels for CO. In one large California study, about 5% of homes were found to have CO levels that exceeded the State 8-hour ambient air quality standard for CO of 9 ppm (Wilson *et al.*, 1993).

2.2.4.1 Mortality

The acute health effects of CO exposure have been well documented in multiple studies since the 1970s. Acute CO poisoning results from a lack of oxygen in the bloodstream due to formation of a CO-hemoglobin complex (carboxyhemoglobin) that prevents oxygen from binding to hemoglobin. Symptoms of acute poisoning include headache, nausea, lethargy and inability to concentrate, unconsciousness, and death at very high concentrations. An estimated 600 deaths per year in the United States throughout the 1990s were attributable to unintentional CO poisoning (cited in Raub *et al.*, 2000).

A California study of ten years of death certificates showed that about 30 - 40 deaths occur in California each year, on average, due to unintentional carbon monoxide (CO) poisoning (Girman *et al.*, 1998; Liu *et al.*, 1993a, 2000). About two-thirds of those deaths were attributable to indoor sources. As shown in Figure 2.2, the indoor sources most implicated in the CO poisonings were combustion appliances (usually malfunctioning or poorly tuned) and charcoal grills and hibachis used indoors. Together these accounted for over 50% of the deaths. The types of cooking and heating appliances included in the indoor combustion appliance category were wall heaters (37%), free-standing heaters (19%), stoves (16%), water heaters (9%), furnaces (9%), and floor heaters (7%). Natural gas was associated with 75% of the deaths, propane with 24%, and kerosene with less than 1%. Motor vehicles, such as those left running in an attached garage, also took a substantial toll.



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2.2.4.2 Morbidity

In an examination of 1991-1994 California CO deaths and hospitalization discharge data, Waldman (1996) found that about 3 to 7 times as many hospitalizations for non-fatal CO poisoning occurred as did deaths from CO poisoning. This totals about 100-300 documented hospitalizations for an average year. However, uncertainties in the patient discharge database and the omission from the database of emergency room discharges that did not result in hospitalization led the investigator to conclude that the actual number of hospitalizations attributable to CO poisoning was at least several times higher. From the pertinent literature, ARB staff and Waldman estimate that five to twenty times (175-700) as many individuals as die from accidental CO poisoning are treated in emergency rooms or hospitalized each year due to serious, non-fatal CO poisonings, and that hundreds to thousands more suffer from undiagnosed heart problems, headache, flu-like symptoms, and other illnesses attributable to CO exposure (Cook *et al.*, 1995; Mah, 2000; Kirkpatrick, 1987; MMWR, 1982; Waldman, 1996). Figure 2.3 illustrates the likely magnitude of these non-lethal effects on the population.

Recent studies have further documented chronic health effects following acute exposures and those due to prolonged exposures to CO. These include lethargy, headaches, concentration

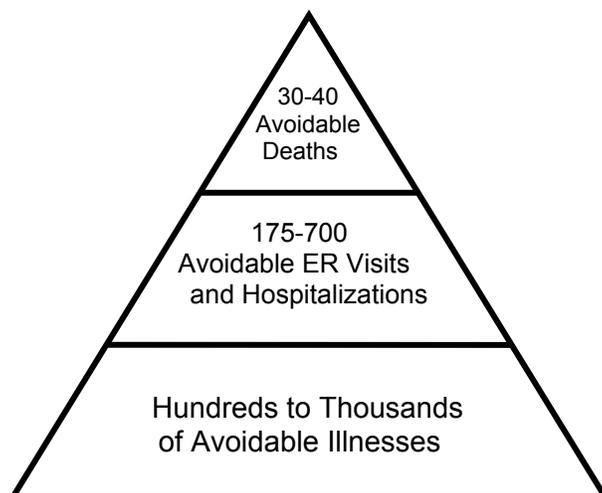


Figure 2.3: Annual California CO Cases

problems, amnesia, dementia, psychosis, Parkinsonism, memory impairment, personality alterations, signs of parietal dysfunction, and other minor symptoms (Townsend and Maynard, 2002; Mathieu-Nolf, 2002). In a study of cardiovascular hospital admission data in Los Angeles between 1992 and 1995, Linn *et al.*, (2000) found that CO showed the most consistently significant relationship among the pollutants considered, and concluded that a wintertime increase in CO of 1.1 to 2.2 ppm predicted an increase of 4% in cardiovascular admissions (~20 extra admissions per day). The elderly are especially susceptible to these effects of CO.

Children are especially susceptible to harm from CO. Because they inhale more air per unit of body weight than do adults at similar activity levels, children will inhale a higher dose proportionately than adults, and will experience symptoms more quickly than adults. The unborn appear to be susceptible to adverse effects after exposure to CO as well. In a study of the

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relationship between ambient air pollution and low birth weight in the northeastern U.S., Maisonet *et al.* (2001) observed increased odds of low birth weight for every 1-ppm increase in CO during the third trimester for the entire population, and a similar increased risk in the African-American population across all three trimesters. In an unrelated study, Ritz *et al.* (2000) estimated that the risk of pre-term birth increases by 12% per 3 ppm increase in CO averaged over six weeks before birth, and by 4% averaged over the first month of pregnancy. Ritz *et al.* (2000) found an increased risk of heart defects with increased ambient CO exposure during the second trimester of pregnancy. Because these effects are seen with small increases in CO, the presence of indoor sources of CO in homes with pregnant women is a serious concern.

2.2.4.3 Carbon Monoxide Emissions and Concentrations

Most homes have relatively low CO levels, except for short intermittent elevations during use of an indoor source. Indoor CO concentrations can increase rapidly when a highly emitting source is present in an enclosed environment. For example, Jetter *et al.* (2002) measured CO emission rates from incense burning from 159 - 531 mg/hr, resulting in an estimated peak CO concentration of 9.6 mg/m³ in a modeled typical room in a home. This is just at the California 8-hour standard level of 9 ppm (10 mg/m³); in combination with outdoor CO and any additional indoor sources, this home would have exceeded the standard. Pelham *et al.* (2002) reviewed CO levels in indoor ice arenas in a review article, stressing that a U.S. and global CO exposure problem remains in these locations (only three U.S. states regulate CO levels in ice arenas).

In a study of 277 Californian homes in 1992, Wilson *et al.* (1993) measured indoor and outdoor CO levels. Thirteen homes had indoor 8-hour CO concentrations above the California 8-hour outdoor standard and indoor air quality guideline of 9 ppm. Several homes had indoor CO 1-hr values greater than the state standard of 20 ppm; in one case this was attributed to the (dangerous) use of gas burners for residential heating. In a second California study (focused on PAHs) of 280 homes in northern California, only two homes exceeded California CO standard levels, one due to use of a fireplace and the other from gas heat (Sheldon *et al.*, 1993). However, in many homes, short-term excursions up to 42 ppm were observed, such as when a gas space heater was turned on.

CO levels in well-maintained indoor environments where appliances are operated properly tend to be relatively low. In surveys of 136 'non-problem' buildings in New York City between 1997 and 1999, Springston *et al.* (2002) measured indoor CO concentrations ranging from 0.2 to 10.3 ppm; only 0.04% of all readings exceeded 10 ppm.

2.2.5 Nitrogen Dioxide

Nitrogen dioxide (NO₂) is another colorless, tasteless gas emitted from combustion sources such as gas and propane appliances, wood burning, kerosene heaters, charcoal grills and motor vehicles. Adverse health effects attributable to NO₂ include exacerbation of asthma (especially in children), respiratory symptoms and infection, lung damage, and lung disease after long periods of exposure. NO₂ is regulated as a pollutant in outdoor air. California has a one-hour ambient standard for NO₂ of 0.25 ppm; this also serves as an indoor air quality guideline. Based on reports that 20-30% of the population use their gas stove (despite clear warnings against this) for space heating (Phillips *et al.*, 1990), indoor NO₂ is estimated to exceed the State ambient air quality standard at times in 10%-30% of California homes.

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2.2.5.1 Health Effects

As discussed earlier, the NAS Institute of Medicine's Committee on the Assessment of Asthma and Indoor Air determined that there is sufficient scientific evidence documenting that indoor NO₂ can exacerbate asthma (IOM 2000). A number of studies also have shown that nitrogen dioxide (NO₂) exposure can result in detrimental effects in the lung. The recent literature discusses the effects of NO₂ on airway antioxidant status, inflammatory cell and mediator responses, and lung function (Becker and Soukup, 1999; Barck *et al.*, 2002; Blomberg *et al.*, 1999; Delvin *et al.*, 1999). Data from Delvin *et al.* (1999), for example, demonstrate that 2 ppm NO₂ can induce a mild inflammatory response in the airways, and that NO₂ may cause a mild impairment of lung antibacterial capacity. This study suggests that possible increases in viral clinical symptoms associated with NO₂ may result from effects of the NO₂ on host defenses that normally prevent the spread of virus. Data from European studies indicates that NO₂ is a pro-inflammatory air pollutant under conditions of repeated exposure (Blomberg *et al.*, 1999) and suggests that ambient NO₂ can enhance allergic inflammatory reaction in the airways (Barck *et al.*, 2002).

The health effects of NO₂ exposure on children have also been reported. Peters *et al.* (1999a, b) found NO₂ significantly associated with reduced lung function in female children within a cohort of 3,292 school children in twelve Southern California communities. Increases in bronchitis symptoms of children with pre-existing asthma also were associated with increases in ambient NO₂ levels (McConnell *et al.*, 2003). Within a cohort of 846 asthmatic children residing in 8 urban areas of the U.S., Mortimer *et al.* (2002) found a 48% increase in the likelihood of asthma symptoms associated with an increase in the 6-day average NO₂ levels. Within a panel of 138 children in central Los Angeles, Ostro *et al.* (2001) found increased odds for shortness of breath and wheezing associated with a 5-ppm increase in the 1-hr maximum NO₂ concentrations.

2.2.5.2 Nitrogen Dioxide Sources, Emissions, and Concentrations

Wall furnaces, floor furnaces, and unvented gas logs have been found to produce elevated indoor NO₂ levels (Wilson *et al.* 1986; Wilson *et al.*, 1993). In the California Residential Indoor Air Quality Study, indoor NO₂ levels were measured up to 177 ppb as 48-hour averages (Wilson *et al.*, 1993). At this level, it is very likely that the 250-ppb California one-hour standard was exceeded for at least some portion of that time.

Gas cooking stoves, particularly when not properly maintained, also have a high potential to lead to elevated indoor NO₂ exposures. Dennekamp *et al.* (2001) measured 5-minute peaks up to 1000 ppm NO₂ when cooking with a 4-burner gas stove (measured at face level in front of the cook). In an ARB-funded cooking study, Fortmann *et al.* (2001) measured indoor levels up to 20 times greater than outdoor levels over a one-hour cooking period, and indoor NO₂ levels exceeded the standard level during several frying, baking, and oven cleaning activities. Lee *et al.* (2002) measured indoor NO₂ and HONO (nitrous acid) levels within 119 residences in Southern California. The average indoor and outdoor concentrations of NO₂ were 28 and 20.1 ppb. The presence of a gas range was significantly associated with indoor NO₂ and HONO levels.

Indoor levels of NO₂ levels often exceed concurrent outdoor levels. In addition to the studies noted above, Zipprich *et al.* (2002) found that measurements in bedrooms and living rooms surpassed measurements taken outdoors (means 18, 19, and 15 ppb, respectively), and that statistical models that best predicted personal exposure included indoor measurements, but

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outdoor measurements contributed relatively little to personal exposure. Zipprich also found that adults had higher NO₂ exposures than children (means 16 vs. 13 ppb). In a Boston study, Brugge *et al.* (2003) found indoor NO₂ levels were either close to or exceeded the NAAQS level of 50 ppb.

2.2.6 Ozone

Ozone (O₃), the primary component of smog, is an invisible, yet highly corrosive, odorous, and chemically reactive gas. Ozone is chemically unstable, and so it breaks down or reacts with many surfaces, liquids, and chemicals. Ozone is typically higher outdoors than indoors. It becomes elevated indoors most commonly from the infiltration of outdoor ozone through doors, windows, and swamp coolers, and sometimes from direct emissions indoors by devices such as certain types of copy machines, laser printers, and “air purifiers”.

2.2.6.1 Health Impacts of Ozone

Ozone is an oxidizing pollutant and strong irritant that attacks the respiratory system, leading to the damage of lung tissue. Exposure to ozone damages the alveoli, the individual air sacs in the lung where the exchange of oxygen and carbon dioxide between the air and blood takes place (ARB/OEHHA, 2004). Increased respiratory symptoms, such as cough, pain on deep breath, and difficulty taking a deep breath are associated with exposure to ozone (Schelegle and Adams, 1986; McDonnell *et al.*, 1999; Kulle *et al.*, 1985; Folinsbee *et al.*, 1977; Seal *et al.*, 1993). Ozone also induces cellular and biochemical changes indicative of lung inflammation (Devlin *et al.*, 1991, 1996; Balmes *et al.*, 1996; Aris *et al.*, 1993). A major result from the Children’s Health Study indicates that children living in high ozone communities who actively participate in several sports, are more likely to develop asthma than children in these communities not participating in sports (Gauderman *et al.*, 2000).

Epidemiological studies have found statistically significant associations between outdoor O₃ concentrations and various adverse health impacts, including increased asthma symptoms (Wittemore and Korn, 1980; Thurston *et al.*, 1997; Delfino *et al.*, 1996; Mortimer *et al.*, 2002), increased respiratory symptoms (e.g., Brunekreef *et al.*, 1994), and reduced lung function (Brunekreef *et al.*, 1994; Brauer *et al.*, 1996). Some studies also report statistically significant associations between O₃ and hospital admissions or emergency room visits, primarily for asthma or other respiratory causes including COPD and bronchitis (Sheppard *et al.*, 1999; Schwartz, 1995; Delfino *et al.*, 1997, 1998; Burnett *et al.*, 1997; Anderson *et al.*, 1997). Ozone has also been linked to increased school absenteeism for respiratory illnesses (Gilliland *et al.*, 2001), and reduced lung function growth in children (Frischer *et al.*, 1999).

California has an ambient air quality standard for O₃ of 0.09 ppm for one hour, while the federal standard is 0.08 ppm averaged over eight hours. Both standards are currently under review.

2.2.6.2 Indoor Sources and Concentrations of Ozone

Outdoor air is the most common source of indoor ozone (Weschler *et al.*, 1989). Outdoor ozone (smog; formed by the photochemical reaction of volatile organic compounds and nitrogen oxides emitted primarily by motor vehicles and industries) enters homes through doors, windows, and numerous air leaks in buildings and their ventilation systems. Studies have shown that indoor ozone levels generally follow the diurnal and seasonal patterns of outdoor ozone, with higher levels in the daytime and summer months (Liu *et al.*, 1993b; Weschler *et al.*, 1994; Liu *et al.*, 1995; Avol *et al.*, 1998; Geyh *et al.*, 2000; Lee *et al.*, 2002). Like outdoor concentrations, indoor

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ozone levels can also remain elevated for long periods of time, eight hours or more, and display peak variations throughout the day (Weschler *et al.*, 1989). Indoor ozone levels typically range from 20 to 80 percent of outdoor ozone levels (Weschler *et al.*, 1989). Using a swamp cooler or whole-house fan on a high-ozone day can increase air exchange rates enough to produce indoor ozone levels very close to outdoor levels for hours at a time (Avol *et al.*, 1996).

The most common indoor sources of O₃ are poorly maintained laser printers and photocopiers, and O₃ generating-devices that are marketed as various types of room deodorizers and air cleaners. In particular, ozone generators that are marketed as “air purifiers” have been found to produce hazardous levels of indoor ozone. Several studies have shown that ozone at levels produced by ozone generators does not effectively control indoor air pollution, odors, or mold growth on surfaces (Boeniger, 1995; Kissel, 1993; Foorde *et al.*, 1997). Ozone generators can destroy microorganisms and gases, but only at concentrations unsafe for occupied spaces. In addition, ozone from ozone generators can react with indoor surfaces, such as latex paint and carpet, or airborne chemicals, including the fragrance compounds from commercial air fresheners, to produce toxic and irritating byproducts such as formaldehyde (Kleno *et al.*, 2001; Wainman *et al.*, 2000; Weschler, 2000; Weschler and Shields, 1999; Moriske *et al.*, 1998; Reiss *et al.*, 1995a,b; Weschler *et al.*, 1992). Most importantly, these devices can result in levels of ozone well above health-based standards and guideline levels. For example, a “personal air purifier” tested for its emissions when used according to manufacturer’s directions resulted in ozone levels in the users breathing zone that exceeded various standard levels, including the California ambient air quality standard (Phillips *et al.*, 1999).

Other sources of indoor ozone include other types of electronic air cleaners and office equipment that uses electrostatic processes. Negative ion generators and electrostatic precipitators (ESP) and dry-process copiers, laser printers, and fax machines can generate significant levels of indoor ozone as a by-product (U.S. EPA, 1995; Kissel, 1993; Selway *et al.*, 1980; Allen *et al.*, 1978).

2.3 TOXIC AIR CONTAMINANTS AND OTHER INDOOR AIR POLLUTANTS

There are a number of other important indoor pollutants that are somewhat unique in their exposure parameters or health effects. Toxic air contaminants (TACs) are air pollutants “ which may cause or contribute to an increase in mortality or an increase in serious illness, or which may pose a present or potential hazard to human health” (HSC Section 39655). Once a compound is identified as a toxic air contaminant, the ARB determines the need and appropriate degree of regulation for the compound. Regulations have been implemented to control the release of numerous TACs to outdoor air; however, regulations do not presently exist to control their release to indoor air. Foremost among these pollutants are formaldehyde, VOCs, environmental tobacco smoke, radon, and polycyclic aromatic hydrocarbons (PAHs). Most have been identified as toxic air contaminants based on carcinogenicity. Pesticides and endocrine disrupters are other pollutants that can be released indoors, or tracked indoors from outdoors. The health effects, indoor sources, and indoor air concentrations of these pollutants are discussed below.

2.3.1 Formaldehyde

Formaldehyde is a pungent smelling gas emitted from numerous indoor sources. These include many building materials (especially pressed wood products), most carpets, composite wood furnishings, consumer products, personal care products and cosmetics, permanent pressed

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clothing, and combustion sources. Formaldehyde is listed as a Toxic Air Contaminant and a Proposition 65 substance, based on its potential carcinogenicity. It is also an upper respiratory tract irritant that produces eye, nose, and throat irritation and exacerbates asthma. Indoor formaldehyde concentrations nearly always exceed outdoor levels, and are often many times higher, due to the many indoor sources. Indoor and urban ambient levels typically exceed the OEHHA Chronic Reference Exposure Level (REL) of 2.4 parts per billion (ppb), which is based on irritant effects on the mucous membranes of the upper airways and eyes, and levels sometimes exceed the 8-hour REL of 27 ppb designed to protect against the same effects. Nearly all indoor environments also exceed the one-in-a-million cancer risk level.

2.3.1.1 Health Effects of Formaldehyde

A number of adverse health effects in humans have been associated with formaldehyde exposure. Short-term effects include eye, nose, throat and skin irritation; nausea; headache; allergic sensitization; and exacerbation of asthma. People vary substantially in their sensitivity to formaldehyde. For most individuals these effects typically occur at exposure levels between 0.037 and 3 ppm (ARB, 1992; U.S. EPA, 1987b). Sensitive individuals may experience acute symptoms related to irritation at lower concentrations.

The ARB identified formaldehyde as a TAC in 1992, based on its carcinogenic potential. Both the International Agency for Research on Cancer (IARC) and the U.S. EPA have classified formaldehyde as a probable human carcinogen, based on sufficient evidence for carcinogenicity in animals and limited evidence in humans. Formaldehyde has been shown to cause a number of genotoxic effects in a variety of cell culture and in vitro assays, including DNA-protein crosslinks, sister chromatid exchanges, gene mutations, single strand breaks, and chromosomal aberrations (ARB, 1992).

2.3.1.2 Sources of Formaldehyde

Many materials and products emit formaldehyde. However, emissions studies have shown that building materials, particularly composite wood products, are likely the greatest contributors to formaldehyde in indoor air. Kelly *et al.* (1999) reported the highest emission rates for numerous composite wood products manufactured with a urea-formaldehyde resin. These products, commonly used in home construction, cabinetry, and furniture, displayed formaldehyde emission rates ranging from 8.6 to 1,580 $\mu\text{g}/\text{m}^2/\text{hr}$. A covering over the wood such as a paper laminate, melamine laminate, or vinyl coating substantially reduced the emission rates to levels at or below 55 $\mu\text{g}/\text{m}^2/\text{hr}$ for all products tested. Composite wood products designed for outdoor use are made with phenol-formaldehyde resin. When tested, these products emitted 4.1 to 9.2 $\mu\text{g}/\text{m}^2/\text{hr}$ formaldehyde, a substantial reduction from the urea-formaldehyde resin products. Coated products and phenol-formaldehyde resin products are preferred alternatives to the urea-resin products.

After numerous lawsuits in the 1970s and 1980s, the composite wood industry developed voluntary emission standards for medium density fiberboard (MDF) and particleboard. A comparison of emission rates from Pickrell *et al.* (1983) and Kelly (1999) shows that the emission rates from current composite wood products average 49% lower than the emissions in the early 1980's. Of the products Kelly tested, all of the bare MDF products and most of the particleboard samples were below the industry limits.

In a recent study funded by the California Integrated Waste Management Board (CIWMB) (Alevantis, 2003), investigators identified nine products in six categories that emitted

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formaldehyde at levels high enough to exceed the Section 01350 guideline level, as described in the Energy Efficiency and Sustainable Building Measures (State of California, 2002). When modeling was conducted on emissions from acoustical ceiling panels, a carpet, medium density fiberboard, gypsum board, resilient flooring (non-rubber based), and thermal insulation, room concentrations were estimated to exceed $16.5 \mu\text{g}/\text{m}^3$ (13.5 ppb), the upper bound allowed for formaldehyde contribution from a single product under Section 01350 guidelines. The nine products with elevated emissions account for 11% of the items tested in this study. Formaldehyde was detected in additional product categories at lower levels.

Recent studies in other states provide additional information on source contributions of indoor formaldehyde. In a manufactured house produced in Florida, Hodgson *et al.* (2002) determined that the greatest contributors to indoor formaldehyde levels were a particleboard cabinet case, and passage doors, each contributing about 33% of the total house formaldehyde concentration. The house, which was a furnished sales model, had an indoor formaldehyde concentration of 77 ppb. The combined emission rate for all wood products in the house was approximately 10 mg/hr. The other 33% of formaldehyde was contributed by cabinet stiles, countertops, subflooring, and other sources. The authors recommend several techniques for reducing indoor residential formaldehyde levels:

- Use vinyl coated or alternative passage doors,
- Use fully coated particleboard in cabinet cases,
- Use frameless cabinets, and
- Apply a laminate backing to the underside of particleboard countertops.

Composite wood products can release high levels of formaldehyde to the indoor environment for long periods of time—for months to years. Brown (1999), an investigator in Australia, measured formaldehyde emission rates from particleboard, medium density fiberboard, and office furniture over several months. Emission rates measured in the study declined from a value of 300 - 400 $\mu\text{g}/\text{m}^2/\text{hr}$ for relatively new product to 80-140 $\mu\text{g}/\text{m}^2/\text{hr}$ for product 5 to 10 months old.

Floor finishing materials such as carpet and vinyl flooring may also emit formaldehyde. In a study funded by the ARB, Hodgson (1999) measured the formaldehyde emissions of several flooring products. Results for formaldehyde emissions at 24 hours showed:

- Carpet emissions were generally below the limit of detection of $1 \mu\text{g}/\text{m}^3$.
- One carpet cushion had an emission rate of $8 \mu\text{g}/\text{m}^2/\text{hr}$.
- Seam tape applied to carpet had an emission rate of $5 \mu\text{g}/\text{m}^2/\text{hr}$.
- Emissions from five different sheet vinyl flooring samples were less than or equal to $4 \mu\text{g}/\text{m}^2/\text{hr}$.
- Adhesives applied to sheet flooring and cove base had emission rates ranging from 72 to $258 \mu\text{g}/\text{m}^2/\text{hr}$.
- When the vinyl flooring or coving was placed on top of the adhesive, the surface product served as a relatively effective barrier, causing a drop in emissions.

Paint is another building material known to emit formaldehyde. Hodgson (1999) measured formaldehyde emissions from 10 different paints widely used in California. As with most wet products, concentrations peaked initially, then declined over several hours. At 96 hours, five paints had emission rates greater than $10 \mu\text{g}/\text{m}^2/\text{hr}$. To address the desire for lower-emitting products, many paint manufacturers are formulating low-VOC paints. These paints have substantially lower total VOC emissions than traditional paint; however, they may emit formaldehyde. Two of the paints in the study were identified as “non-VOC”, however their formaldehyde emissions rates at 48 hours were 43 and $12 \mu\text{g}/\text{m}^2/\text{hr}$. Chang *et al.* (1999) also

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evaluated emissions from low-VOC paints. In small chamber tests, two of four low-VOC paints emitted formaldehyde with peak concentrations at 3.15 mg/m³ and 5.53 mg/m³.

2.3.1.3 Indoor Formaldehyde Concentrations

Despite changes in manufacturing processes over the last two decades, indoor levels of formaldehyde still reach very high levels in some indoor locations. Current estimated average and maximum indoor formaldehyde concentrations are shown in Figure 2.4 for different California environments. Figure 2.4 illustrates that concentrations in some homes and schools exceed OEHHA's interim 8-hour REL of 27 ppb, established to identify levels above which sensitive individuals could experience acute eye, nose, and lung irritation. The figure also shows the relative levels of formaldehyde in different environments, indicating that homes are of greatest concern.

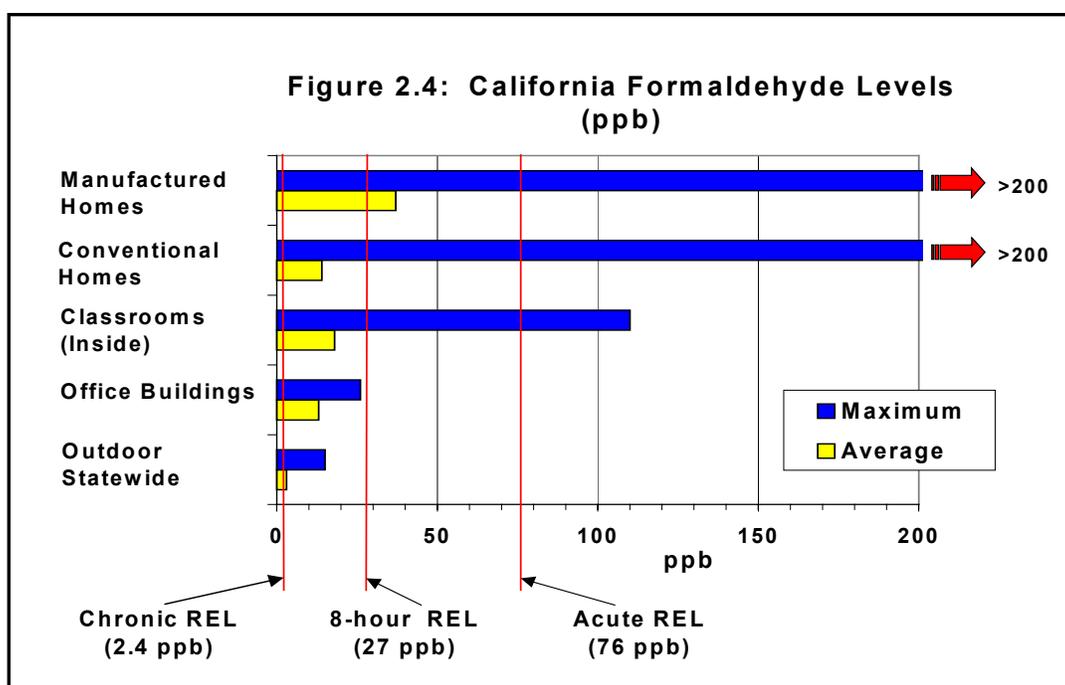


Figure 2.4 also illustrates that levels in virtually all indoor environments exceed OEHHA's chronic REL (2.4 ppb) for irritant effects and OEHHA's one-in-a-million excess lifetime cancer risk level (0.13 ppb) for formaldehyde. It is generally not feasible to achieve levels below these guideline levels because outdoor levels average about 3-5 ppb.

Data for Figure 2.4 have been compiled from several sources. Manufactured home levels are from a large, older study of manufactured homes (Sexton *et al.*, 1985), with levels adjusted downward based on the average reduction in emissions from manufactured wood products since 1983 (Kelly *et al.*, 1999). The conventional home data were obtained by combining results from a 1996 study of Southern Californian homes (Avol *et al.*, 1996) and the National Human Exposure Assessment Survey (NHEXAS) exposure study in Arizona (Gordon *et al.*, 1999). The maximum values estimated for current California manufactured and conventional homes are similar, between 220-240 ppb, although measured values have been obtained well above these

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levels in some studies. The classroom data are taken from the California Portable Classrooms Study (Whitmore *et al.*, 2003), using both Phase I and Phase II data. Concentrations in office buildings are from the U.S. BASE study (U.S. EPA, 2003a) of about 100 medium and large office buildings throughout the U.S., including buildings from California. The sources and derivations of the averages and maxima shown in Figure 2.4 are discussed in Appendix III.

New homes built with standard construction products are likely to have unhealthy levels of formaldehyde. Hodgson *et al.* (2000) measured formaldehyde concentrations inside homes two months old, then continued measurements for a seven and half-month period. The geometric mean concentration of formaldehyde in four manufactured homes was 34 ppb, and in seven site-built homes was 36 ppb. Both were substantially greater than the outdoor formaldehyde level of 6 ppb, and above health-based guidelines. The formaldehyde emission rates were fairly consistent over the seven to eight month sampling time, indicating that the off gassing of formaldehyde emissions from building materials in new homes extends for a long period of time. The author commented that these levels are approximately 50 percent lower than formaldehyde concentrations measured during the 1980's due to use of less plywood paneling in manufactured homes and reduced emission rates from composite wood products relative to 20 years ago. Emission rates were calculated for these homes and found to be 45 $\mu\text{g}/\text{m}^2/\text{hr}$ for the manufactured homes and 31 $\mu\text{g}/\text{m}^2/\text{hr}$ for the site-built homes.

In the most recent large-scale study (NHEXAS) of formaldehyde levels in homes, conducted in Arizona, twenty five percent of the homes sampled had formaldehyde levels above the 8-hour REL of 27 ppb. This is of concern because people are typically in their homes for longer than 8 hours and the study was designed to be representative of the population. The formaldehyde limit of detection was fairly high at 10 ppb, so that formaldehyde was detectable in only 69% of the indoor air samples (131/189) with a median concentration of 21 $\mu\text{g}/\text{m}^3$ (17 ppb). The 75th and 90th percentiles in residences were 34 $\mu\text{g}/\text{m}^3$ (28 ppb) and 46 $\mu\text{g}/\text{m}^3$ (37 ppb), respectively, with a maximum value at 408 $\mu\text{g}/\text{m}^3$ (332 ppb). These investigators also commented that mean values are lower than those reported in the 1980's (Gordon *et al.*, 1999).

Although not evident in Figure 2.4, the results from Phase II of the California Portable Classroom Study (PCS) indicated that formaldehyde levels in at least 4% of California classrooms exceed OEHHA's interim 8-hour REL of 27 ppb (Whitmore, *et al.*, 2003), the level at which an 8-hour exposure might result in irritant effects and initiation of an immune system response in a sensitive individual. This is equivalent to about 10,720 classrooms, or at least 214,400 children (assuming 20 children per classroom...usually there are many more) exposed to formaldehyde levels that could potentially result in irritant effects. Average and peak levels in the classrooms generally exceeded those measured in the U.S. EPA BASE study of public and commercial buildings, indicating that, during the day, children are likely experiencing greater exposures than are most adults. Data analyses from the PCS indicate that several factors were associated with indoor formaldehyde levels in classrooms including the presence of plywood and particleboard; vinyl tackboard; and bookcases and cabinets made of pressed wood; increased temperature and humidity; and classroom age (higher levels in newer classrooms) (Whitmore *et al.*, 2003).

Overall, as shown in Figure 2.4, indoor environments typically have formaldehyde levels above the recommended guideline levels for long-term exposures, and sometimes above levels protective against acute exposures as well. Homes in particular can have elevated levels.

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2.3.2 Volatile Organic Chemicals

Volatile organic chemicals (VOCs) are a large group of pollutants that include many of the toxic gases widely found in indoor environments. VOCs include a variety of chemical properties and functional groups: alkanes, alcohols, esters, ethers, and aromatic compounds. Indoor sources of VOCs include building and furnishing materials such as carpet, paint, and vinyl flooring; consumer products such as air fresheners, adhesives, and cleaning agents; water treated with chlorine; dry-cleaned clothing; environmental tobacco smoke; plastic products, computers, and others.

Some VOCs have been identified as California Toxic Air Contaminants (TACs) or federal Hazardous Air Pollutants (HAPs) due to their cancer-causing potential. Some VOCs also cause eye, nose, and throat irritation, and neurological effects such as headache. Indoor and personal exposure levels of these VOCs are typically higher than concurrent outdoor levels, and often exceed acceptable cancer risk levels and other health-based guideline levels.

Recent research has examined some reactive VOCs with double bonds such as styrene, pinene, limonene, and other terpenes. These can produce secondary pollutants such as ultrafine particles and compounds that are more irritating than the precursors (Weschler and Shields, 1997; Molhave, 2003). Research is also being directed to another group of pollutants that disrupt endocrine systems. In a recent study, Rudel *et al.* (2003) measured some of these endocrine disrupting compounds in residential air samples as well as residential dust samples.

2.3.2.1 Health Effects of VOCs

Many of the VOCs found in indoor air in California are carcinogenic. Accordingly, there is no level of exposure to these chemicals that is known to be absolutely safe. Some of the more common carcinogenic VOCs were listed earlier in Table 2.4. The cancer risk posed by some of the most prevalent VOCs is discussed above in Section 2.1.2, and a detailed discussion of those risks is provided in Appendix II.

Many indoor VOCs also can irritate the eyes, nose, and throat. These effects can be experienced at concentrations sometimes found in buildings, depending on the sensitivity of the individual. At higher concentrations (usually not encountered in homes or offices, but sometimes found in occupational settings), some VOCs, most notably aromatic hydrocarbons (such as benzene), chlorinated chemicals (such as perchloroethylene), and pesticides can impact the nervous system, causing acute effects such as nausea, tremors, drowsiness, dizziness, and headache (ARB, 1997a).

While neurotoxic effects are not normally encountered in homes and offices, special circumstances can lead to elevated exposures in some specialized non-industrial workplaces. The nature of activities and products used in medical offices, hospitals, beauty shops, high production copy shops, and other non-industrial workplaces can lead to unusually high concentrations of some pollutants. Business parks that merge industrial and non-industrial businesses may also create unique situations leading to exposure to elevated levels of pollutants.

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2.3.2.2 Sources And Emissions of VOCs

Studies have shown that some indoor sources emit substantial amounts of VOCs, and that groups of related VOCs often have common sources. Exposure to VOCs is influenced by people's activities and their proximity to sources of pollutants. Exposure studies indicate the greatest exposures generally result from close contact with specific sources.

Benzene, styrene, toluene, and xylene are aromatic hydrocarbons containing a hexagonal ring structure. Sources of these chemicals include environmental tobacco smoke, motor vehicle exhaust, evaporative emissions from vehicles in attached garages, and a variety of consumer products such as solvents, adhesives, glues, and paint (Wallace *et al.*, 1988; Akland and Whitaker, 2000; Guo *et al.*, 1999). They are also present in emissions from office copy machines and printers (Lee *et al.*, 2001). The overwhelming source of benzene exposure for smokers is mainstream cigarette smoke (Wallace, 1996b); nonsmokers with smokers in their households may experience a substantial benzene exposure due to ETS as well (Wallace, 2001). For nonsmokers with infrequent exposure to ETS, most benzene exposure comes from auto exhaust, especially while traveling on busy roadways (Rodes *et al.*, 1998) or gasoline vapor emissions during fueling at gas stations, in houses with an attached garage, and while traveling in a vehicle (Wallace, 1996b).

Toluene is emitted from a number of consumer products in California. In a study of over 100 products sold in California stores, Akland and Whitaker (2000) measured toluene most frequently in auto care products, glues, and personal care products, such as in fingernail polishes. Estimated maximum concentrations and daily doses were high, up to 4000 µg/day. In the Arizona NHEXAS study, toluene measurements demonstrated the infiltration of pollutants from attached garages. Investigators measured toluene levels at non-smokers' homes and found a level of 24 µg/m³ in homes with an attached garage (n=40), but only 5 µg/m³ in homes without an attached garage (N=9) (Gordon *et al.*, 1999).

Chlorinated hydrocarbons, another large group of VOCs, contain one or more chlorine atoms. Chemicals in this group have diverse sources. Para-dichlorobenzene (p-DCB), commonly used as an "air freshener," is on the California Proposition 65 list based on its carcinogenic potential. Data from the U.S. EPA's Total Exposure Assessment Methodology (TEAM) studies indicate that about one-third of homes use products containing p-DCB (Wallace, 1991). In one study, the indoor air concentration of p-DCB increased from 1 µg/m³ to more than 500 µg/m³ the day after a toilet bowl cleaner was introduced into a home (Wallace, 2001). In another study, the maximum p-DCB measured in Arizona homes was 3949 and 4400 µg/m³, presumably from the introduction of household consumer products (Gordon *et al.*, 1999). These examples illustrate the extremely high levels that can occur in residences.

Levels of perchloroethylene, another chlorinated hydrocarbon, increase astronomically when dry-cleaned clothes are brought into a house. Levels in a home containing recently dry-cleaned clothes can be 100 to 150 times greater than outdoor levels of perchloroethylene (Wallace 2001). Levels of another chlorinated hydrocarbon, methylene chloride, have been greatly reduced in consumer products; however, it is still common in paint strippers. Short-term exposures can be significant for individuals who use paint strippers (Wallace 1991).

Chlorine is intentionally added to domestic water for public health purposes. However, chlorine and organic matter in the water react to produce chloroform. Exposure to chloroform and other trihalomethanes occurs from drinking water, taking showers and baths, and operating washing

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machines (Wallace 2001). In one study, investigators measured chloroform concentrations while individuals actually took showers in residential shower stalls. Average concentrations during and immediately after the shower ranged from 67 $\mu\text{g}/\text{m}^3$ to 265 $\mu\text{g}/\text{m}^3$. Air concentrations of trihalomethanes were about three times higher on average for a typical shower compared to a bath. Higher concentrations were observed with hotter water (Kerger *et al.*, 2000). Chloroform can also be produced during the use of bleach and other cleaning agents that contain chlorine. Investigators predict that 5.3 mg to 9.8 mg of chloroform can be released to indoor air during a ten-minute wash cycle when a laundry bleach containing sodium hypochlorite is used in a residential washing machine (Shepherd *et al.*, 1996).

The pollutants discussed above are associated with one or a few predominant sources. But, many sources, such as building materials, are known to emit numerous VOCs. A number of studies have been conducted to examine VOC emissions from a variety of building materials. Most recently, emission testing funded by CIWMB indicates that building materials potentially can emit enough VOCs to produce potentially harmful levels (Alevantis, 2003). Several products exceeded the Section 01350 guideline levels as follows:

- Carpet exceeded the emission rate limits for naphthalene and acetaldehyde.
- Fiberboard, particleboard, and resilient flooring exceeded the limits for acetaldehyde.
- Resilient flooring products exceeded the limit for naphthalene and phenol.

Products exceeding guideline levels for formaldehyde are discussed in the formaldehyde section.

In another study, Hodgson (1999) identified 17 toxic air contaminants in the emissions from carpet, vinyl flooring, and latex paint. These toxic compounds are routinely emitted to the indoor environment, particularly in new or recently remodeled homes and offices. Table 2.6 contains a list of TACs identified by Hodgson in building material emissions. In addition to TACs, all of the bonded urethane carpet cushions emitted butylated hydroxytoluene (BHT), an irritant, and all carpet samples emitted 4-phenylcyclohexene (4-PC), the compound largely responsible for new carpet odor.

Table 2.6. Toxic Air Contaminants in Building and Finishing Materials

Carpet and Cushion	Paint	Vinyl flooring
Toluene	m,p-Xylene	Toluene
m,p-Xylene	Ethylene glycol	m,p-Xylene
o-Xylene	2-(2-Butoxyethoxy)ethanol	o-Xylene
Styrene	Formaldehyde	Styrene
Ethylene glycol	Acetaldehyde	1,2,4-Trimethyl benzene
Formaldehyde		Naphthalene
Acetaldehyde		Acetophenone
Acetophenone		Phenol
2-(2-Butoxy)ethanol		Formaldehyde
Ethylbenzene		Acetaldehyde
Tetrachloroethane		
Naphthalene		
Phenol		

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Emissions from polymeric building materials can be attributed to several factors (Yu and Crump, 1998):

- solvent residues from the manufacturing process,
- unreacted monomers trapped in the product structure,
- secondary products from reactions of monomers, and
- plasticizers used in production.

Measurement of VOCs in new homes is another method of determining the variety of compounds emitted from building materials. Hodgson *et al.* (2000) identified α -pinene, formaldehyde, hexanal, and acetic acid as the predominant compounds measured in 11 new homes. For manufactured houses and site-built houses, the geometric mean total volatile organic chemicals (TVOC) emission rate was 1.7 mg/m²-h, and 2.1 mg/m²-h, respectively. Individual compounds with the highest emission rates, greater than 50 μ g/m²-h include α -pinene, ethylene glycol, hexanal, acetic acid, β -pinene, and 2,2,4-trimethyl-1,3-pentanediol monoisobutyrate (TMPD-MIB).

Paint not only emits numerous TACs, but emissions can continue for extended periods of time. Sparks *et al.* (1999) estimated that less than 50% of the VOCs in latex paint are emitted in a year. Compounds studied include ethylene glycol, propylene glycol, texanol, and butoxyethoxyethanol. Alkyd paints contain substantially more VOCs than latex paints due to the use of mineral spirits as the solvent. Compounds emitted from alkyd paints include alkanes such as decane, nonane, octane, undecane, and aromatic compounds such as xylenes, toluene, and ethylbenzene (Guo *et al.*, 1999).

Consumer products such as cleaning products, personal care products, art supplies, and hobby supplies can release pollutants to the indoor environment and cause high personal exposures to pollutants. In fact, Akland and Whitaker (2000) found six products that they estimated would exceed the Proposition 65 no-significant-risk level for formaldehyde, including a nail finish, make-up, floor-cleaning spray, formica laminate, foaming tire care, and sheet flooring adhesive. Two products exceeded the no-significant-risk level for benzene, and one product each exceeded the levels for acetaldehyde and tetrachloroethylene.

Cleaning products have been studied to assess their impact on personal exposure in indoor environments. Zhu *et al.* (2001) tested 13 cleaning products to assess the potential for human exposure to three glycol ethers, 2-methoxyethanol (ME), 2-ethoxyethanol (EE), and 2-butoxyethanol (BE). Investigators calculated average daily inhalation exposure levels for an individual cleaning with these products at home. Calculations were based on product use scenarios developed by U.S. EPA and a "standard room". For two all-purpose spray cleaners the average daily inhalation exposure for 2-butoxyethanol was 0.075 and 0.186 mg/kg body weight/day, and for two spray glass cleaners it was 0.004 and 0.006 mg/kg body weight/day. Investigators identified numerous other VOCs emitted from a variety of consumer products. For example:

- Finger nail polish remover contained toluene, BE, and ethyl acetate.
- Permanent hair color contained BE, alcohols, and propylene glycol.

Other investigators also have studied emissions from consumer products and personal care products. Cooper *et al.* (1995) identified ethanol, α -pinene, camphene, β -pinene, diethylene glycol monoethyl ether, benzaldehyde and others as components of fragrance in two colognes, a perfume, a soap, and an air freshener. Toxicological data indicate the compounds may be

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irritants, mutagenic, teratogenic, or carcinogenic; however, these health effects generally occur at much greater levels than would be expected from the use of the products.

The impact of human occupancy and activities on VOC concentrations was documented in a study conducted by Shields *et al.* (1996). Investigators measured VOC concentrations indoors and outdoors at 70 offices with a range of occupant density. The results showed that VOC concentrations were associated with the density of occupancy and ventilation rate. The authors were able to identify six compounds associated with occupant density: limonene used in cleaning products and air fresheners; tetrachloroethylene from dry-cleaned clothes; n-dodecane through n-hexadecane, probably from cosmetics, hand lotions, and shaving creams; and octa- and deca-methylcyclpentasiloxanes, which are associated with underarm deodorants and antiperspirants.

Girman *et al.* (1999) identified VOCs in public buildings, but did not associate the presence of VOCs with a particular source. Girman analyzed data collected at 56 office buildings (most with no known indoor air quality problems) across the U.S. Of the 48 VOCs identified in indoor air, eight were found in all of the samples, another 26 were found in 81-99% of the samples. Indoor concentrations of 27 VOCs were at least twice the corresponding outdoor concentration. The 12 VOCs with the highest median indoor concentrations were acetone, toluene, d-limonene, m- and p-xylenes, 2-butoxyethanol, n-undecane; benzene, 1,1,1-trichloroethane, n-dodecane, hexanal, nonanal, and n-hexane. Four of the most abundant 12 VOCs are oxygenated, which may indicate a greater potential to cause irritant effects.

2.3.2.3 Indoor Concentrations of VOCs

Several major studies of concentrations of selected toxic pollutants in California residences have been conducted under the sponsorship of the U.S. EPA and/or ARB. Studies of VOCs were conducted over different seasons in three different years (1984, 1987, and 1990) in a total of nearly 500 California homes (Wallace 1991; Wallace *et al.*, 1988; Sheldon *et al.*, 1992a). Analyses of the VOC results across these and other studies indicate that indoor concentrations are typically two to five times higher, and sometimes many times higher, than outdoor air (Wallace 1991; Wallace *et al.*, 1988; Sheldon *et al.*, 1992a; Ott and Roberts 1998). Results also indicate that personal (actual) exposure levels are generally higher than either indoor or outdoor residential levels. These results were consistent across different seasons and different geographic locations.

To illustrate, the relative average personal, indoor, and outdoor concentrations of several VOCs measured in a northern California study are presented in Table 2.7 (Sheldon *et al.*, 1992a). Personal levels are usually higher than indoor and outdoor levels due to people's frequent use of, or proximity to, sources such as consumer products as they go about their daily activities.

Another example of California VOC indoor and outdoor concentrations is presented in Figure 2.5. Figure 2.5 summarizes the median levels (levels at which half of the homes are higher and half lower) of indoor and outdoor concentrations of selected VOCs using combined data from one of the California VOC TEAM studies (February and July, 1987 in Los Angeles; Pellizzari *et al.*, 1989) and a study conducted in Woodland, California that was similar to the TEAM studies (Sheldon *et al.*, 1992a). Figure 2.5 reflects the presence of indoor sources for all chemicals shown except for carbon tetrachloride, for which there are no longer indoor sources due to

**Table 2.7. Personal, Indoor, and Outdoor VOC Concentrations
From a Northern California Study
($\mu\text{g}/\text{m}^3$; means)**

Compound	Personal Concentration	Indoor Concentration	Outdoor Concentration
<i>para</i> -dichlorobenzene	21	18	0.30
Benzene	5	4.7	1.2
Perchloroethylene	1.6	1.1	0.53
Trichloroethylene	2.3	0.68	NQ
Styrene	2.4	2.8	0.24
1,1,1-Trichloroethane	22	6.5	1.5
<i>m,p</i> -xylene	9.3	6.3	1.8

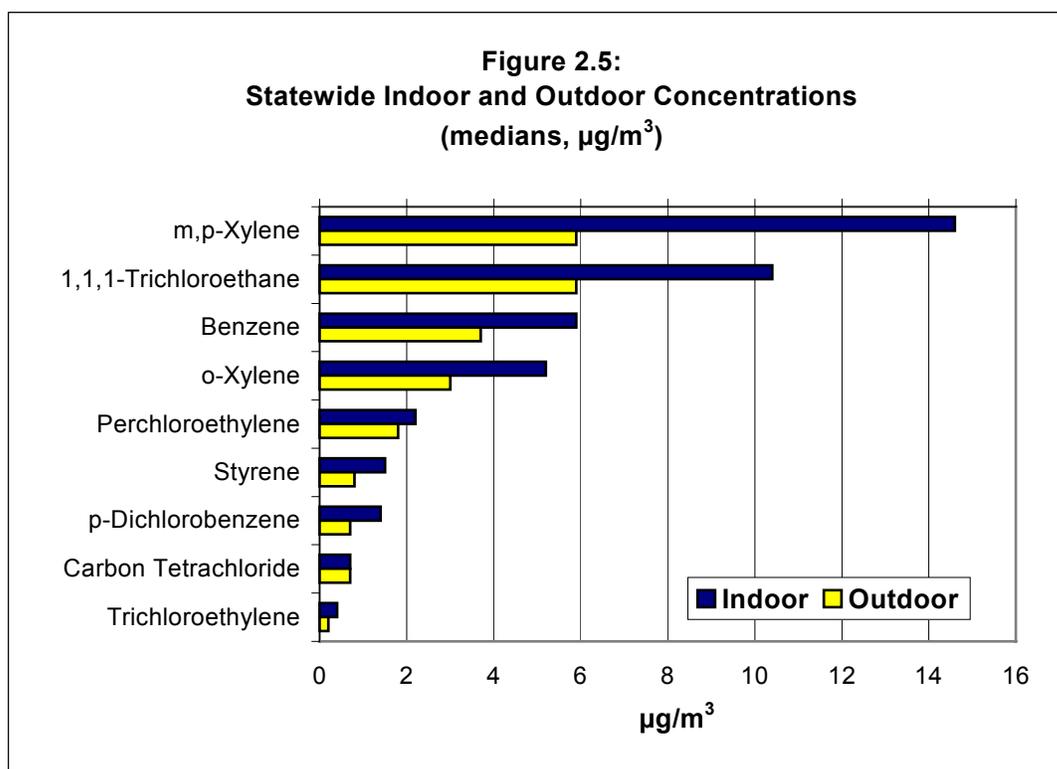
NQ = not quantifiable: below the method quantifiable limit.
From Sheldon *et al.*, 1992a.

product restrictions imposed at the national level (CPSC, 1987b). Benzene concentrations measured in Texas homes during 1993 (Mukerjee *et al.*, 1997a,b) were comparable to levels measured in the California studies.

Some spatial and temporal trends in VOC levels (not illustrated in the figure) also have been noted: indoor concentrations in southern California appear to be somewhat higher than indoor levels in northern California, and indoor levels in winter tend to be higher than indoor levels in summer (Jenkins *et al.*, 1992b).

Levels of some VOCs in California homes, such as benzene, have most likely decreased since these studies were conducted, due to changes in some building materials and consumer products, and the reduced smoking rates in homes. However, more recent studies in other states have continued to show higher levels indoors than outdoors, and levels above health risk guidelines.

For example, results from the recent Arizona National Human Exposure Assessment Survey (NHEXAS) indicate that indoor levels of pollutants continue to be greater than outdoor levels, and some are comparable to levels measured in the initial TEAM studies (Gordon *et al.*, 1999). In another recent study conducted in Minneapolis/St. Paul, investigators confirmed the continued risk due to elevated indoor concentrations of VOCs (Sexton *et al.*, 2004). Pollutant concentrations were compared to risk thresholds established in Minnesota. Median indoor concentrations for benzene and chloroform exceeded the level for a 70-year lifetime cancer risk of 1 in 100,000. The 90th percentile concentrations exceeded the cancer risk level for benzene, chloroform, and *p*-dichlorobenzene. The identified pattern of personal VOC concentrations exceeding indoor VOC concentrations, which in turn exceed outdoor VOC concentrations, also was reaffirmed by this study. This relationship held for 13 of the 15 VOCs measured by Sexton *et al.* in Minneapolis/St. Paul over three seasons in 1999.



Some additional trends found in indoor air VOC studies were identified by Brown *et al.* (1994) in their review of studies conducted between 1978 and 1990. The review encompasses measurements from residences, offices, schools and hospitals. Significant findings include:

- For all compounds, indoor concentrations were greater than outdoor concentrations by a factor of two to 73, indicating indoor sources were present. Mean VOC and TVOC concentrations were generally greater in established residences than established public buildings.
- New buildings had considerably higher TVOC concentrations than established buildings. For example, the weighted-average geometric mean TVOC concentration from 33 new residences was $4,500 \mu\text{g}/\text{m}^3$, whereas it was $1,130 \mu\text{g}/\text{m}^3$ from 1,081 established buildings.
- VOCs specific to new buildings included 2-ethoxyethylacetate, n-butanol, α -pinene, undecane, dodecane, tridecane, tetradecane and 2-propanol.

2.3.3 Environmental Tobacco Smoke

Environmental tobacco smoke (ETS) is the complex mixture of thousands of gaseous and particulate compounds emitted from the burning end of cigarettes and cigars, and in smokers' exhaled breath. ETS is also called "second-hand smoke", "passive smoke", and "involuntary smoking". Many of the chemicals in ETS have been identified as toxic air contaminants (ARB/OEHHA, 2003). Cigarettes are the most common source of ETS, but cigars and pipes are also sources.

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Passage of a statewide smoke-free workplace law in California in 1995 (AB 13, Labor Code 6404.5) eliminated smoking in nearly all California indoor workplaces—including restaurants, bars and gaming clubs. In 1999, 93.4% of California's indoor workers reported working in a smoke-free environment, compared to only 45% in 1990 (Gilpin *et al.*, 2001). Additionally, the percentage of Californians with children under the age of 18 who do not allow smoking in the household has increased substantially. In 1994, 63% of Californians with children did not allow smoking in the house; by 2001, 78% did not allow it (Gilpin *et al.*, 2001). Because of such laws and trends, California has lower ETS exposures than most other states.

2.3.3.1 Health Effects of ETS

Exposure to ETS has been linked with a variety of adverse health effects, including heart disease, asthma episodes, other respiratory illness, and lung cancer (OEHHA, 1997). Primary tobacco smoking is an established human carcinogen (IARC, 2002; U.S. DHHS, 1989). ETS has been identified as a cause of lung cancer in nonsmokers (U.S. DHHS, 1986; U.S. EPA, 1992; IARC, 2002). More recently, studies are confirming that ETS exposure also has a causal association with breast cancer in humans (ARB/OEHHA, 2003).

ETS has a number of serious impacts on children's health including exacerbation of asthma, increased respiratory tract infections, increased middle ear infections, low birth weight, sudden infant death syndrome (SIDS), and developmental impacts (ARB/OEHHA, 2003). In children, exposure can exacerbate asthma, and increases the risk of lower respiratory tract illness, and acute and chronic middle ear infection. Children under 18 months of age have greater tendency to suffer from bronchitis or pneumonia when parents smoke. A small reduction in birth weight has been repeatedly observed with mothers who are active smokers. Studies have shown fairly consistently that maternal smoking during pregnancy is adversely associated with measures of cognition and behavior in children (NCI, 1999). There is epidemiological evidence of a causal relationship between maternal smoking in general and risk of SIDS.

ETS affects many tissues and organs of the body. Eye and nasal irritation are the most commonly reported symptoms among adult nonsmokers exposed to ETS (ARB/OEHHA, 2003). There is compelling evidence that ETS is a risk factor for induction of new cases of asthma (in children and adults) and increases the severity of disease among children and adults with established asthma (IOM, 2000). ETS also is causally associated with coronary heart disease in smokers and nonsmokers exposed to ETS (NCI, 1999). Research suggests that chronic ETS exposure may increase the risk of stroke by about 82% (Bonita *et al.*, 1999).

2.3.3.2 ETS Sources

The variability in emissions between brands of cigarettes is relatively low, though emissions from cigars and cigarettes vary in magnitude. Daisey *et al.* (1998) conducted a chamber study testing six of the most popular commercial brands in California and one reference cigarette for emissions of 21 different air toxics and other airborne compounds, including volatile organic compounds (VOCs), nicotine, aldehydes, and airborne particulate matter (estimated to be PM_{2.5}). Diluted sidestream smoke (produced by a smoking machine that smoked three cigarettes sequentially) was used to approximate ETS aging in a room-sized chamber, and a mass-balance model was used to generate estimates of indoor concentrations. Among the VOCs, acetaldehyde and formaldehyde displayed the highest emission factors (average emission factors 3,340 ng/mg tobacco and 2,040 ng/mg tobacco, respectively), and PM showed an emission factor 12,400 ng/mg. The nicotine emission factor was 1,410 ng/mg tobacco.

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These results suggest that ETS has a substantial influence on indoor concentrations of these compounds.

Chuang *et al.* (1999) investigated children's exposure to polycyclic aromatic hydrocarbons (PAH) in low-income rural and inner-city areas in North Carolina. Indoor/outdoor air samplers and real-time PAH monitors were used to obtain measurements. The researchers determined that potentially carcinogenic PAH concentrations were significantly higher in smokers' homes than in non-smoking homes (geometric mean: 6.14 ng/m³ vs. 1.38 ng/m³, respectively). Additionally, the authors discovered that children in both rural and inner-city homes received higher potential doses of PAHs than adults, in part due to their lower body weights.

2.3.3.3 ETS Concentrations

Restrictions on smoking in California from the late 1980s to mid-1990s in workplaces and in public locations such as restaurants, bars, and gaming clubs have led to a reduction in smoking in indoor environments in California, with commensurate reductions in indoor concentrations of ETS and non-smokers' exposure levels. A number of studies published since 1996 have shown that ETS constituents are present at lower concentrations following smoking bans than they were prior to the bans (Ott *et al.*, 1996; Hammond, 1999; Switzer *et al.*, 2001; Repace, 2003), and that levels can be considerably higher in smoking versus comparable nonsmoking areas (Glasgow *et al.*, 1998; Hammond, 1999; Graves *et al.*, 2000).

Despite California's workplace smoking ban, high indoor ETS concentrations still can be found in smokers' homes and in private vehicles (Park *et al.*, 1998; Offermann *et al.* 2002; Ott *et al.*, 2003). Children's exposure to ETS is greatly impacted in these two environments when in the presence of a smoking parent or other adult. Children spend up to 85 percent of their time in the home (Wiley *et al.*, 1991a). Thus, the potential for exposure to ETS can be extremely high when smoking occurs in a child's home. As of 2001, 22% of smoking parents still allowed smoking inside the home (Gilpin *et al.*, 2001). Likewise, exposure in vehicles can be quite high due to the presence of a strong source in a relatively small volume of air. Recent residential respirable particulate matter (RSP, PM_{3.5}) measurement is limited to a single study (Ott *et al.*, 2003). A level of 300 µg/m³ was measured in the bedroom where one cigarette was smoked; 5,500 µg/m³ was the maximum bedroom level when 3 cigarettes were smoked. RSP levels ranging from 92 µg/m³ (with ventilation) to 1,195 µg/m³ (without ventilation) have been measured inside a minivan (Offermann *et al.*, 2002).

Three comprehensive review documents summarize nicotine and RSP concentrations measured in smoking environments prior to 1996 (U.S. EPA, 1992; Guerin *et al.*, 1992; NCI, 1999). Comparison of mean nicotine concentrations from these earlier reviews with data published after 1995 reveals that the means have decreased in workplaces and restaurants, but not in homes. In studies conducted before 1996, mean nicotine concentrations in offices and restaurants ranged from about 1 to 36 µg/m³. In a more recent review, Hammond (1999) reported means of 2 to 8 µg/m³ for these locations. Also, according to the Hammond review, nicotine levels were two- to six- times lower in indoor workplaces with smoking bans than in offices that allowed smoking (less than 1 µg/m³ vs. 2-8 µg/m³, respectively). It appears that nationally, as smoking has become a less accepted social behavior, individuals are not smoking in indoor public locations as much as they did a few years earlier (in California, smoking is prohibited in indoor workplaces).

However, the recent body of data indicates that those who choose to smoke in their home have remained consistent in their smoking patterns over the years. Current estimates of mean

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residential nicotine concentrations in homes with smokers are 0.5, 3.0, and 6.0 $\mu\text{g}/\text{m}^3$ for low, medium, and high concentrations, respectively, based primarily on measurements taken by Glasgow *et al.* (1998). The majority of California homes never or rarely contain smokers and would typically have a nicotine level lower than 0.5 $\mu\text{g}/\text{m}^3$.

Levels of RSP are generally comparable in both older and newer studies, but slightly lower in the newer studies (relative to 1996). Studies highlighted in the NCI review (1999) reported RSP concentrations consistent with other reviews. All measured levels tend to range from about 100 to 400 $\mu\text{g}/\text{m}^3$ in offices and restaurants that allow smoking. Switzer *et al.* (2001) measured RSP levels at a church bingo site of 87 to 348 $\mu\text{g}/\text{m}^3$ above outdoor levels, and at less than 15 $\mu\text{g}/\text{m}^3$ when smoking was banned. Similarly, PM3.5 concentrations at a sports tavern in California were 56.8 $\mu\text{g}/\text{m}^3$ with smoking, and 5.9 – 12.9 $\mu\text{g}/\text{m}^3$ with smoking banned (Ott *et al.*, 1996). Similar to nicotine levels, these recent RSP data from smoking locations are somewhat lower than the pre-1996 data. RSP levels are negligible (<15 $\mu\text{g}/\text{m}^3$) in indoor locations where smoking is prohibited. Table 2.8 summarizes current estimated indoor levels of nicotine and RSP for California. A more detailed discussion of indoor ETS concentrations is provided in the Draft *Technical Support Document – Part A, Proposed Identification of Environmental Tobacco Smoke as a Toxic Air Contaminant* (ARB/OEHHA, 2003).

Table 2.8. Estimates of Current California Indoor Concentrations of Nicotine and RSP

Environment	Nicotine Concentration $\mu\text{g}/\text{m}^3$	RSP Concentration $\mu\text{g}/\text{m}^3$
Homes with smokers present	0.5 – low ¹ 3.0 – medium 6.0 – high	300-5,500 ³
Offices/public buildings With smoking Smoking prohibited	2 – 8 ² <1	56.8 – 348 ⁴ <15
Vehicles With ventilation Without ventilation	NA NA	~100 ⁵ ~1,200

1 Glasgow *et al.*, 1998

2 Hammond, 1999

3 Ott *et al.*, 2003

4 Ott *et al.*, 1996 and Switzer *et al.*, 2001

5 Offermann *et al.*, 2002

2.3.4 Biological Contaminants

Biological contaminants are substances of plant, animal, or microbial origin. They are naturally abundant in the outdoor and/or indoor environments, but are considered contaminants when found in undesired locations or at elevated concentrations. Exposure to biological contaminants may cause a variety of health effects, including infection, irritation, allergy, and toxic responses.

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2.3.4.1 Health Effects of Biological Contaminants

The health effects of biological contaminants can be grouped into three major categories:

- Communicable disease transmission: Many infectious diseases are transmitted from person to person in indoor air. For example, inhalation of viruses is associated with influenza, measles, and chicken pox (while colds more are often due to direct contact). Inhalation of bacteria is associated with tuberculosis and Legionnaire's disease. Epidemiological studies have often found significant decreases in the prevalence of respiratory illness or surrogates for respiratory illness (sick leave, total absence from school) to be associated with higher building ventilation rates, reduced office sharing, and less crowding (Fisk, 2000; Myatt *et al.*, 2002; Shendell *et al.*, 2004). These and other studies indicate that a significant portion of common respiratory illness may be transmitted indoors by airborne particles; however, the proportion of total disease transmission that occurs through this route remains uncertain.
- Hypersensitivity reactions: Many biological agents can provoke an immunological response, which most frequently takes the form of allergic reactions to the agent. Allergic rhinitis (hay fever) and allergic asthma are examples of hypersensitive responses to biological contaminants. Common symptoms and signs are watery eyes, runny nose, sneezing, nasal congestion, itching, coughing, wheezing, difficulty breathing, headache, and fatigue. Fungal spores, microbial byproducts, house dust mites, cat allergens, and pollen are frequently associated with allergic and asthmatic responses.
- Toxic responses: Inhalation of endotoxin (a component of the outer membrane of Gram-negative bacteria) and mycotoxins (fungal toxins produced as secondary metabolites) may cause toxic reactions, such as inflammation, vomiting, diarrhea, and immunosuppression. This health impact is the least studied and understood.

People are a major source of infectious agents, but the environment is the primary source of bacteria, fungi (mold or mildew), animal dander, house dust mites, cockroaches, and plant pollen. Outdoor air that enters buildings carries bacteria, fungi, and pollen, but indoor concentrations typically are lower than outdoors. However, excessive moisture in buildings may support microbial growth on building materials and furnishings and result in elevated exposure to biological contaminants in residential, educational, and commercial buildings. Legionnaire's disease, for example, may be transmitted when excess moisture is collected in air conditioning drip pans or other moist locations in buildings, providing a suitable microenvironment for growth of bacteria and fungi. Such cases have been termed Building Related Illness (BRI), because a specific disease results from a specific organism associated with the building. The usual causes of BRI include viruses, bacteria, and fungi. Occupants complain of symptoms such as cough, chest tightness, fever, chills, and aches, depending on the specific organism involved. Symptoms may require prolonged recovery times after leaving the building. BRI is less common than SBS (Menzies *et al.*, 1994; Hedge, 1995; Hodgson, 1995; Menzies and Bourbeau, 1997).

House dust mites, cockroaches, and animal dander are all known allergens for sensitive individuals. House dust mites are microscopic organisms that live on human skin cells and other organic material. Consequently they are found in carpets, bedding, and sofa cushions. They are more prevalent in older homes and in more humid locations such as near the coast. Cockroaches and animal dander can be eliminated from the indoor environment; however, residual bits of their body parts and animal dander brought into the home on clothing may be sufficient to trigger allergic or asthmatic responses in some individuals.

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2.3.4.2 *Indoor Mold*

Molds are simple, microscopic organisms, present virtually everywhere, indoors and outdoors. Molds, along with mushrooms and yeasts, are fungi, which play a critical role in nature by breaking down dead plant and animal matter and recycling nutrients in the environment. For molds to grow and reproduce, they need only a food source – organic material, such as leaves, wood, paper, or animal products such as leather – and moisture. Because molds grow by digesting organic material, they gradually destroy whatever they grow on. Mold growth on surfaces can often be seen in the form of discoloration and wooly or cottony mycelia, frequently green, gray, brown, or black but also white and other colors. Molds release countless tiny, lightweight spores, which travel easily through the air.

In recent years, media attention on indoor mold has surged, and this has led to rising concern about mold-related health effects. Everyone is exposed to some mold on a daily basis, typically without evident harm. It is common to find mold spores in the air inside homes, and most of the airborne spores found indoors come from outdoor sources. Mold spores primarily cause health problems when they are present in large numbers and people inhale an unusually high number of them. This occurs primarily when there is active mold growth within a home, office, or school where people live or work. People can also be exposed to mold by touching contaminated materials and by eating contaminated foods.

The most serious health effects of indoor mold chiefly include inflammation of the mucous membranes or respiratory tract and infection, however, allergic symptoms are the most common following mold exposure. Typical symptoms that mold-exposed persons report (alone or in combination) include: respiratory problems, such as wheezing, difficulty breathing, and shortness of breath; nasal and sinus congestion; eye irritation (burning, watery, or reddened eyes); dry, hacking cough; nose or throat irritation; and skin rashes or irritation. The Institute of Medicine reviewed the literature and found evidence that mold exacerbates asthma, but that there was insufficient evidence that mold exposure leads to the development of asthma. Allegations that indoor mold causes other health effects (e.g., memory loss or chronic fatigue) have not been substantiated.

2.3.5 Pesticides

Pesticides can be naturally occurring or synthetic chemicals designed to control and kill insects, weeds, and diseases in the home and surrounding landscaping. Ninety percent of American homes use pesticides (Gurunathan *et al.*, 1998). Schools commonly apply pesticides in or around the classroom (Addiss *et al.*, 1999; Kaplan *et al.*, 1998; Volberg *et al.*, 1993). Pesticides can be tracked in on clothing or drift in from outdoors, later becoming resuspended in air and accumulating in dust.

2.3.5.1 *Health Effects of Pesticides*

Two classes of widely used insecticides in the U.S. are the organophosphates and pyrethroids, both are neurotoxins. Neurological signs resulting from acute toxicity may include nausea, headaches, dizziness, and general weakness. Pesticides are often measurable in house dust and carpet dust; levels of contamination are discussed below. The effects of pesticides on children are a particular concern because their behavior can lead to greater exposure than to an adult. Children spend time on the floor where they contact dust that may contain pesticides. The hand-to-mouth behavior of young children may lead to ingesting pesticides. Furthermore,

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research has shown that low-level chronic exposure to organophosphates can adversely affect children's developing nervous systems (Eskenazi *et al.* 1999). There is currently insufficient evidence to determine if pesticides cause or exacerbate asthma (IOM, 2000). As with organophosphates, chronic exposure to pyrethroids has been linked to hormonal disruption, childhood cancer, and neurological damage (Landrigan *et al.*, 1999).

2.3.5.2 Sources of Pesticides

Chlorpyrifos and diazinon, two organophosphate insecticides, were the most widely used pesticide ingredients in common household ant and roach killers and lawn-care sprays (U.S. EPA, 2001; U.S. EPA, 2000d). The U.S. EPA banned the indoor use of chlorpyrifos and diazinon for non-agricultural settings in December 2000 and March 2001, respectively (U.S. EPA 2001; U.S. EPA, 2000d). Since the ban on indoor use of chlorpyrifos, a class of insecticides called pyrethroids has been used as substitutes for chlorpyrifos and other organophosphate pesticides. Permethrin, the most common pyrethroid, acts on a broad spectrum of insects, and is less persistent than chlorpyrifos in dust and soils.

2.3.5.3 Pesticide Concentrations

To date, only one published study has examined indoor pesticide concentrations in California homes. In September 1992, Bradman *et al.* (1997) measured pesticide levels in house dust samples from 5 farmworker homes and 5 non-farmworker homes in an agricultural area south of Fresno. This study was conducted before the ban on indoor use of chlorpyrifos and diazinon. However, it cannot be determined if the indoor levels were from infiltration of outdoor air and soil track-in, or from the use of indoor pesticide. Of the ten pesticides detected in house dust, only chlorpyrifos and diazinon levels frequently exceeded 1 µg/g in the homes. In general, farmworker homes had higher levels of chlorpyrifos and diazinon than non-farmworker homes. The highest detectable level of pesticide in non-farmworker homes was 2.5 µg/g of diazinon, and the highest in farmworker homes was 169 µg/g of diazinon.

During 2001-2002 (after the ban on indoor use of chlorpyrifos and diazinon), ARB and DHS funded a comprehensive statewide study of the environmental health conditions in portable (relocatable) classrooms. As part of the study, investigators summarized floor-dust pesticide concentration data for 20 different pesticides. Chlorpyrifos, cis- and trans-permethrin, ortho-phenylphenol, piperonyl butoxide (PBO), and esfenvalerate were detected in over 80% of the classrooms suggesting indoor contamination from outdoor sources. Excluding ortho-phenylphenol, these pesticides measured 95th percentiles above 1 µg/g. Portable classrooms did not differ significantly from traditional classrooms in their mean pesticide concentrations (Whitmore *et al.*, 2003).

The pesticides detected in California and their concentrations are similar to those found in other studies in other areas of the U.S. A summary of indoor and personal air concentrations and house dust levels is presented in Table 2.9. The following paragraphs provide a brief discussion of some of the table content.

The Non-Occupational Pesticide Exposure Study (NOPES) examined inhalation exposures for 32 different pesticides in 208 residences in Jacksonville, Florida, and 101 residences in Springfield/Chicopee, Massachusetts, over three different seasons from 1986 to 1988

Table 2.9. Comparison of Indoor-air, Personal-air, and House-dust Levels in U.S.

Location	Season	Ranges and median and mean values					
		Chlorpyrifos			Diazinon		
		Range	Median	Mean	Range	Median	Mean
Indoor air		Concentration (ng/m³)					
Minneapolis, MN ^a	All	NL ^b	1.742	NL	NL	0.29	NL
NHEXAS Arizona ^c	All	<3.2-3280	8	NL	<2.1-20,500	4.6	NL
Brownsville, TX ^d	Spring	2.5-115	7.6	NL	1.6-60	1.4	NL
	Summer	5.7-67	24	NL	2.5-78	3.5	NL
Jacksonville, FL ^e	Summer	<2.5-2170	182	366.6	15-13,700	73	420.7
	Spring	NL	NL	205.4	NL	NL	109.2
	Winter	<2.5-1043	69	120.3	30-1080	21	85.7
Springfield, MA ^e	Spring	<4.5-252	<4.5	9.8	<22-1810	<22	48.4
	Winter	<3.5-291	<3.5	5.1	<28-28	<28	2.5
Raleigh, NC ^f	Fall	NL	NL	0.08	NA ^g	NA	NA
New Jersey area ^h	Fall	151.2-154.2	NL	NL	5.7	NL	NL
Cape Cod, MA ⁱ	All	<1-92	<1	NL	<1-550	<1	NL
Personal air		Concentration (ng/m³)					
Minneapolis, MN	All	NL	1.577	NL	NL	0.275	NL
NHEXAS Arizona	All	<19-175	NL	NL	<15	NL	NL
Jacksonville, FL	Summer	NL	NL	280.4	NL	NL	321.6
	Spring	NL	NL	182.8	NL	NL	112.7
	Winter	NL	NL	118.2	NL	NL	89
Springfield, MA	Spring	NL	NL	7.5	NL	NL	10.1
	Winter	NL	NL	5.9	NL	NL	1.4
House dust		Concentration (µg/g)					
Fresno, CA ⁱ	Fall ^j	0.2-33	NL	NL	0.7-169	NL	NL
	Fall ^k	<1	NL	NL	0.2-2.5	NL	NL
NHEXAS Arizona	All	<0.004-119	0.16	NL	<0.020-66	0.13	NL
Brownsville, TX	Spring	0.1-1.7	0.3	NL	0.1-1.8	0.06	NL
	Summer	0.2-1.7	0.56	NL	0.1-0.8	0.07	NL
Wenatchee, WA ^l	Spring ^d	<0.02-3.6	0.27	0.43	NA	NA	NA
	Spring ^k	<0.02-0.5	0.05	0.17	NA	NA	NA
Jacksonville, FL	Winter	NL	4.7	5.8	NL	0.4	1.7
Raleigh, NC	Fall	NA	NA	1.6	NA	NA	NA
New Jersey area ^h	Fall	0.53-15	NL	NL	0.08-0.74	NL	NL
PCS ^m	All	NL	0.308	0.607	NL	0.035	0.358
Cape Cod, MA ⁱ	All	<0.2-228	<0.2	NL	<0.2-51.0	<0.2	NL

Source: Modified from Gordon *et al.*, 1999.

^aClayton *et al.*, 2003.

^bNL, not listed.

^cGordon *et al.*, 1999.

^dMukerjee *et al.*, 1997b.

^eWhitmore *et al.*, 1994.

^fLewis *et al.*, 1994.

^gNA, not applicable.

^hRoinestad *et al.*, 1993.

ⁱRudel *et al.*, 2003.

^jBradman *et al.*, 1997.

^kSamples from agricultural family homes.

^lSamples from non-agricultural family homes.

^mSimcox *et al.*, 1995.

ⁿWhitmore *et al.*, 2003.

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(Whitmore *et al.*, 1994). Seven pesticides—chlordane, chlorpyrifos, diazinon, dichlorvos, heptachlor, ortho-phenylphenol, and propoxur—had the highest mean concentrations across seasons in Jacksonville for both indoor and personal air. Mean concentrations of all detectable pesticides in carpet dust ranged from 0.01 µg/g to 15.4 µg/g. The mean concentrations for chlorpyrifos and diazinon in carpet dust were 5.8 µg/g and 1.7 µg/g, respectively.

During the 1992 spray season, Simcox *et al.* (1995) vacuum-sampled house dust for four organophosphorous (OP) insecticides, including chlorpyrifos, in 59 homes in Wenatchee, Washington (48 agricultural, 11 non-farming families). All four compounds were detected in 62% of households. Pesticide levels were significantly lower in non-farming homes than in agricultural homes (Table 2.9). OP levels ranged from non-detectable to 17 µg/g in agricultural homes and non-detectable to 0.82 µg/g in non-farming homes. Two-thirds of the agricultural homes contained pesticide levels above 1 µg/g for at least one OP compound.

During May to September 1997, the Minnesota Children's Pesticide Exposure Study (MNCPEs) characterized indoor and personal air levels for children 3 to 12 years old in 102 homes in Minneapolis/St. Paul, Minnesota (Clayton *et al.*, 2003). Although the median values for indoor air were slightly higher than personal air (Table 2.9), there was a highly significant correlation between personal air and indoor air samples for the organophosphates chlorpyrifos (0.81), malathion (0.51), and diazinon (0.62). Air levels for chlorpyrifos and diazinon were measurable for >90% and >65% of the participants, respectively. Permethrin was also detectable in air with a median concentration of <0.2 ng/m³.

In the National Human Exposure Assessment Survey (NHEXAS) performed in Arizona, Gordon *et al.* (1999) measured residential levels of chlorpyrifos and diazinon. Chlorpyrifos was found in dust more often than diazinon (88% vs. 53%), and had nearly the same occurrence in indoor air as diazinon (65% vs. 63%). There was a small association between chlorpyrifos in dust and air (Pearson = 0.096, Spearman = 0.773). Although the NHEXAS results are generally comparable to those from other studies, Gordon *et al.* (1999) found a higher maximum level for pesticides in residential samples than previously reported.

In a recent study, Rudel *et al.* (2003) analyzed 39 air and 38 dust samples (collected by vacuum cleaner) for pesticides in 120 homes in Cape Cod, Massachusetts. Of the 23 pesticides detected in air and 27 detected in dust, chlorpyrifos was one of the most abundant with 90th percentile concentrations of 12 ng/m³ in air and 1.87 µg/g in dust. Although not as pervasive in dust, diazinon had one of the highest 90th percentile concentrations in air (9.0 ng/m³). Cis- and trans-permethrin and PBO were also detected at relatively high levels in dust (7.04 µg/g, 16.5 µg/g, and 15.1 µg/g, respectively).

2.3.6 Lead

Lead is a toxic metal that has been used in paints, gasoline, pipes, ceramic glazes, and caulk. Lead paint chips and soil contaminated with lead poses a major hazard to children when they ingest them. Over the long term, exposure to lead can lead to brain damage, decreased growth, hyperactivity, impaired hearing, and reproductive effects.

2.3.6.1 Health Effects of Lead

The body mistakes lead for calcium and concentrates it in the bones, where it can leach into the blood. Both pre-natal and post-natal exposure, and exposure during childhood results in toxicity to the nervous system (Thacker *et al.*, 1992; Needleman *et al.*, 1979). Chronic exposure to lead

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can lead to decreased learning ability due to brain damage. Acute health effects, seldom seen, include seizures, paralysis, anemia, abdominal pain, constipation, vomiting, and decreased appetite.

Lead not only concentrates in bones, it also accumulates in the liver and kidneys. Lead has a half-life of 25 to 40 days in the blood and organs, and a half-life of 25 years in the bones. During pregnancy, lead mobilized from the bones and released into the bloodstream can pass through the umbilical cord and adversely affect the developing fetus. The U.S. Centers for Disease Control and Prevention established a lead safety threshold of 10 µg/dL in 1991. Significant decreases in IQ, behavioral problems, and learning impairment have been documented at blood-lead levels above this threshold. Recent research showing health effects at levels below 10 µg/dL now supports the growing consensus that there is no safe level for lead exposure.

2.3.6.2 Sources of Lead

In the U.S., lead was eliminated from paints and gasoline in the late 1970s. Today, major sources of lead include old paint in homes built before 1978, lead pipes placed before the 1930s, and soil by busy roads. Activities that disturb lead-based paint, such as remodeling or paint removal, can release large amounts of lead-bearing particles into the air. Often overlooked sources of lead include home remedies, cosmetics, hobby materials, and candle wicks (ATSDR, 1999a). Foreign-made, vinyl mini-blinds may also contain lead (CPSC, 1996). Under prolonged exposure to sunlight, the vinyl deteriorates and produces a lead-laden dust.

Much of the lead present in indoor air appears to result from the infiltration of lead particles in outdoor air. Tracked-in and infiltrated lead dust can accumulate in carpets that can serve as a reservoir for lead-laden dust (U.S. EPA, 1997a). Research has shown that lead-dust loadings and concentrations per unit mass of dust are correlated with blood-lead levels, but no causal effect can be inferred from this association. Children's mouthing behaviors and activities that put them in direct contact with lead-contaminated surfaces increase their probability for exposure to lead by ingestion.

2.3.6.3 Lead Concentrations

In a 1997 report, the U.S. EPA summarized findings from a literature review on the amount of lead in dust within carpets, furniture, and forced air ducts. A major drawback across studies was the inconsistency in the methods used to collect and analyze lead. In general, the average loadings are lower for furniture than for floors, window components, walls, and air ducts. Furniture, including upholstery and window treatments, has an averaging loading of less than 100 µg/ft². The average loadings for forced air ducts exceed 100 µg/ft², and can be as high as 1,000,000 µg/ft².

Dust lead levels at schools are an important factor when determining children's total exposure to lead. In a recent comprehensive study of the environmental health conditions in California portable (relocatable) classrooms, ARB and DHS summarized floor-dust metal concentration data for 18 different metals (Whitmore *et al.*, 2003). All the elements, except copper, had higher dust loading in portable classrooms, though the differences were not statistically significant. As stated in the report: "Since the portable classrooms are generally newer, the lower concentration of lead may reflect the number of years' accumulation of the particles in the classroom." Table 2.10 reports the median and 95th percentile concentrations for lead in the floor dust of all classrooms sampled.

Table 2.10. California Portable Classrooms Study Results for Lead in Floor Dust

Element	Room Type	Concentrations (µg/g)		Loadings (ng/cm ²)	
		Median	95 th Percentile	Median	95 th Percentile
Lead	All	61.6	189.5	6.6	58.4
	Portable	57.5	151.6	5.8	57.9
	Traditional	66.8	200.6	7.1	57.5

Between 1994 and 1998, DHS surveyed the lead content in paint and soil for a random sample of 200 California public elementary schools and child care facilities (DHS, 1998). Results were used to make predictions about the lead content in all of California’s 5,000 schools. The study found: “As in California’s housing, lead-containing paint is estimated to be in most California public elementary schools and child care facilities.” DHS estimated that one-third of schools contain deteriorating lead paint, and nearly 6% have soil that is contaminated with lead exceeding U.S. EPA Lead in Soil Guidelines. The highest levels of lead in soil were found close to school buildings and near buildings built prior to 1940.

2.3.7 Mercury

There are three types of mercury: elemental (or metallic), inorganic (or mercury salts), and organic. Inorganic mercury is intentionally added to latex paint, medicine, and disinfectants to impart antibacterial properties. Metallic mercury can be found in thermometers, electric switches, fluorescent lights, thermostats, and other products. The primary source of organic mercury, specifically methylmercury, is fish and other types of seafood.

2.3.7.1 Health Effects of Mercury

Organic and elemental mercury can cause permanent damage to the brain and central nervous system. Other health effects from exposure to mercury include tremors, changes in personality, loss of sensation and muscle coordination, vision and hearing impairments, and deficits in cognitive function. Elemental mercury (Hg⁰) is most toxic when inhaled. It can cause respiratory tract irritation, severe stomach discomfort, skin rashes, and elevations in blood pressure and heart rate. All three types have been linked with kidney damage. The risk from long-term, low-level exposure to inorganic mercury is not well understood. Carpi and Chen (2001) estimated that 10% of U.S. homes may have indoor mercury levels that exceed the U.S. EPA Reference Concentration of 0.3 µg/m³.

2.3.7.2 Sources of Mercury

Indoor air is the second most common route of exposure to mercury in the general U.S. population (fish consumption is first) (NJ Mercury Task Force, 2002; Carpi and Chen, 2001; WHO, 1990). Mercury is an effective fungicide and bactericide, so it has been used as an intentional additive in many household products, including latex paints (interior use banned in 1990), contact lens solutions, and nasal sprays (ATSDR, 1999b). Mercury may also be found as an accidental contaminant in detergents and cleansers due to its extensive use in the chlor-alkali industry (U.S. EPA, 1999b). Elemental mercury’s unique physical and chemical properties are the basis for its use in many household items. It expands uniformly with increases in temperature, does not cling to glass, and is liquid and can vaporize at room temperature. Mercury workers are exposed to greater levels of elemental mercury than the general

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population. Workers' families are also at increased risk; workers may bring home clothing and shoes contaminated with mercury in the workplace (ATSDR, 1999b). Elemental mercury is also used in some cultural practices, either carried as a good luck charm, or sprinkled in cars and homes or over infants (ATSDR, 1999b). Table 2.11 lists the sources of mercury in homes.

Table 2.11. Sources of Mercury in the Residential Environment

Type of Mercury	Sources
Metallic	Thermometers, electric switches, fluorescent lights, thermostats, barometers, batteries, ethnic remedies, spiritual practices
Inorganic and organic	Latex paints, medicines, disinfectants, detergents, skin bleaching creams, antiseptics

2.3.7.3 Mercury Concentrations

There is limited information on airborne exposures in the non-occupational environment. From June 2000 through March 2001, Carpi and Chen (2001) measured mercury concentrations in indoor air for nine residences and three businesses in the New York metropolitan area. Nearly all of the sites showed higher indoor levels than outdoor levels. The average indoor mercury concentration for all buildings was 69 ng/m³, and was highly dependent on season.

Mercury is found in many household items, but unless the mercury-containing device is broken or disturbed, mercury does not pollute the air. Once released, however, mercury persists in the indoor environment for months or years after its first release, especially if the spill is not properly cleaned (ATSDR, 1999b). According to the New Jersey Mercury Task Force (2002): "Exposure to Hg⁰ (such as in certain cultural practices) can be significant with respect to health effects. As little as one drop (0.05 ml) of liquid Hg⁰ in a sealed bedroom-sized room (assuming a room volume of about 33 m³ and no air exchange) can result in an air concentration equal to the U.S. EPA Reference Concentration."

Several nationwide incidences of non-occupational exposure to mercury have been documented. The Ohio Department of Health, in cooperation with the federal ATSDR, analyzed the indoor mercury concentrations in nine school gymnasiums (ATSDR, 2002). The highest detectable level of mercury in indoor air was 1.6 µg/m³. A *Morbidity and Mortality Weekly Report* summarized findings from the contamination of homes and schools with Hg⁰ in Palm Beach County, Florida (CDC, 1995). The report illustrated how quickly mercury can spread and how difficult it is to contain. The local authorities evacuated 50 homes, and closed schools until mercury levels dropped below 10 µg/m³. Seventeen homes contained vapor concentrations greater than 15 µg/m³. Table 2.12 shows the guideline levels for mercury.

Table 2.12. Recommended Airborne Concentration of Mercury

Values	Mercury Vapor
ATSDR Minimum Risk Level (MRL)	0.2 µg/m ³
ATSDR recommended screening level	1 µg/m ³
ATSDR action level	10 µg/m ³
U.S. EPA reference concentration (RfC)	0.3 µg/m ³

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2.3.8 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are a complex class of semi-volatile organic compounds and consist of two or more fused benzene rings. The larger PAH species are associated with fine or ultrafine particles; humans become exposed to a complex mixture of PAHs, not individual compounds (IARC, 1983; U.S. EPA, 1987c; Atkinson and Arey, 1989). Benzo(a)pyrene, a PAH with five fused benzene rings, is often considered to be the representative compound for the PAHs (U.S. EPA, 1987c; IARC, 1983). PAHs are produced from incomplete combustion; indoor sources are primarily tobacco smoking, woodburning, and infiltration of outdoor air.

2.3.8.1 Health Effects of PAHs

Several individual PAHs are believed to be carcinogenic to humans. The U.S. Environmental Protection Agency (U.S. EPA) classifies benzo(a)pyrene as a possible human carcinogen; the International Agency for Research on Cancer (IARC) classifies benzo(a)pyrene as a probable human carcinogen (IARC, 1983; U.S. EPA, 1984). Other PAHs characterized by either IARC or U.S. EPA as possible human carcinogens include: chrysene, certain benzofluoranthenes, and indeno(1,2,3-cd)pyrene (IARC, 1983; U.S. EPA, 1984, ARB/OEHHA, 1994; ARB, 1994a). For most of the PAH species, there is insufficient evidence to classify these compounds as to their carcinogenicity.

2.3.8.2 Sources of PAHs

Tobacco smoking, when present, was identified as the strongest indoor source of benzo(a)pyrene and other PAHs indoors in a study conducted in California (Sheldon *et al.*, 1993; Sheldon *et al.*, 1992b). Investigators have measured as many as 56 PAH species and derivatives in tobacco smoke; hundreds more may exist (Vu-Duc and Huynh, 1989; Wynder and Hoffman, 1964).

Cooking activities were found to be the greatest contributor to PAH concentrations, second to infiltration of traffic -polluted outdoor air in a small study conducted by Dubowski *et al.* (1999). These investigators also determined candle- and incense burning contribute to indoor levels of PAHs.

Woodburning appliances contribute to indoor PAH levels. Fireplaces can emit soot and PAHs directly into the indoor air environment (Traynor *et al.*, 1987). While the newer, more efficient airtight stoves appear to emit less than the older, leakier woodstove models, poor maintenance and certain practices, such as operating the woodstove with the door open, can raise PAH levels substantially (Traynor *et al.*, 1987).

Transport of outdoor air into the home can introduce PAHs from outdoor sources such as traffic, diesel engines, power plants and agricultural burning (IARC, 1983; U.S. EPA, 1987c; Dubowski, 1999).

2.3.8.3 PAH Concentrations

The Air Resources Board funded two large studies to characterize indoor exposures to PAHs in California residences. In a northern California study, investigators measured PAHs inside 280 homes during the winter of 1992, evaluating the relationship between PAH levels and common

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indoor combustion sources (Sheldon *et al.*, 1993). Investigators reported average concentrations of 2.2 ng/m³ of benzo(a)pyrene in smoker's homes, but only 0.83 ng/m³ in nonsmoker's homes. Investigators reported that cigarette smoking significantly raised levels of 12 of 13 PAH species when compared to levels in nonsmokers' homes.

In the same study, fireplace and woodstove use raised average benzo(a)pyrene levels to about twice the levels found in homes with no obvious combustion sources. Researchers reported average benzo(a)pyrene concentrations of 1.2 ng/m³ compared to 0.83 ng/m³ measured in "no source" homes. The benzo(a)pyrene level of 0.83 ng/m³ was primarily attributed to infiltration of smoke from community woodburning, a value well above the typical average outdoor levels in California (Atkinson and Arey, 1989).

The second ARB-funded study was conducted in Southern California homes. In this study, Sheldon *et al.* (1992b) found polluted outdoor air to be a major source of indoor PAHs when other combustion sources were absent in the home. Other investigators have reported similar findings (Waldman *et al.*, 1989; Lioy and Greenberg, 1990; Naumova *et al.*, 2002). In two Los Angeles area communities, Naumova *et al.*, (2002) measured PAHs in homes located near busy traffic areas and reported significantly lower levels of particle-bound PAHs indoors than outdoors. Investigators stated that these data suggested that indoor concentrations were driven by outdoor PAH pollutant levels. Concentrations were dominated by coronene, an indicator of motor vehicle-related pollution (*ibid*). Cigarette smoking was also a major contributor to indoor PAH levels in the study of southern California homes (Sheldon *et al.*, 1992b). In smokers' homes, investigators reported statistically significant increased concentrations relative to nonsmoking homes of nine PAH species, including benzo(a)pyrene during the daytime, and 6 PAH species during the night.

Investigators in other states have reported substantial increases in benzo(a)pyrene indoors as a result of tobacco smoking in the home (Kanarek *et al.*, 1985; Chuang *et al.*, 1988; Turk *et al.*, 1987). In a study of PAH levels in 15 public buildings in the Pacific Northwest, Turk (1987) reported average levels of benzo(a)pyrene of 1.07 ng/m³ smoking areas compared to 0.39 ng/m³ in nonsmoking areas. The mean indoor-to-outdoor ratio was 7.6.

2.3.9 Radon

Radon is a naturally occurring radioactive gas (radon-222) that emanates from the decay of uranium-238, which is present in small amounts in soils, rocks and water. Radon is an inert gas, non-reactive, odorless, and colorless. It is a carcinogenic agent because it emits radioactivity in the form of alpha particles. Radon in the lungs can have a very destructive effect by releasing alpha particles in close proximity to lung epithelial cells (Cothorn, 1987). A statewide survey indicates that only 0.8% of California residences have annual radon levels above 4.0 pCi/L.

2.3.9.1 Health Effects of Radon

Causal associations between radon exposure and lung cancer have been demonstrated in many epidemiological studies of underground miners. The U.S. EPA estimated that 7,000 to 30,000 lung cancer deaths per year in the U.S. are due to indoor radon exposure, the second leading cause of lung cancer and a serious public health concern. Concurrent exposure to radon and smoking show synergistic reaction, that is, the risk of lung cancer is higher than predicted by adding the individual risks (NRC, 1999a).

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While the majority of radon risk is associated with inhalation exposure from air, naturally occurring radon in water may also pose risk. Recent estimates for radon in drinking water are that ingested radon causes about 168 cancer deaths per year: 89 percent from lung cancer caused by breathing radon released from water (e.g., showering or flushing the toilet), and 11 percent from stomach cancer caused by drinking radon-containing water (NRC, 1999b).

2.3.9.2 Sources of Radon

Indoor radon derives from various sources in the vicinity of a building: (a) release from soils underneath the building (generally the most important source), (b) building materials enriched with U-238 (e.g., granite or concrete blocks used in the foundation or walls), (c) use of radon-enriched local well water (especially by bathing and appliance use), and (d) outdoor air. Elevated indoor radon levels are generally due to the combination of a rich source, a driving force into the building, and relatively low dilution rate for fresh air (Eichholz, 1987).

2.3.9.3 Radon Concentrations

Samplers are available to test indoor radon levels at reasonable cost. Measurement techniques include charcoal canisters, alpha track detectors, electret ion chamber detectors or scintillation flasks. Radon concentrations are reported as the number of radioactive decays per time in a volume of either air or water, or picoCuries per liter (pCi/L). The U.S. EPA has issued recommendations to reduce the health risks from radon exposure in homes. Immediate corrective action should be pursued when indoor air radon levels are above 200 pCi/L. For radon levels between 20 and 200 pCi/L, pursue corrective action within a few months. For radon levels between 4 and 20 pCi/L, pursue corrective action within a few years (U.S. EPA, 2002a).

A statewide survey in California found that only 0.8% of California residences have annual radon concentrations exceeding 4 pCi/L (Liu *et al.*, 1991). Areas with higher than average radon levels include the Central Valley, the Sierra Nevada mountains and foothills, the valleys east of the Sierra Nevada, and Ventura and Santa Barbara counties. Surveys of California public elementary schools demonstrated similar geographic patterns for areas with elevated indoor radon (Churchill and Youngs, 1993; Zhou *et al.*, 1998).

The National Residential Radon Survey was conducted by the U.S. EPA in 1989-90 to assess annual average radon concentrations in U.S. residences (Marcinowski *et al.*, 1994). The survey estimated an arithmetic annual average radon concentration in U.S. homes of 1.25 ± 0.12 pCi/L, and about 6.0% of homes had radon levels greater than the U.S. EPA action level, 4 pCi/L.

While elevated homes are more commonly found in areas with higher natural levels of soil radioactivity (i.e., uranium), complex geology, as well as home construction and operation factors, make predictions of residential radon concentrations difficult (Churchill, 1997). Nonetheless, building factors, such as ventilated crawl space (and no basement), rooms above the first level, and higher frequency of open windows are known to reduce indoor radon (Tsai and Waldman, 2003).

2.3.10 Asbestos

Asbestos is a naturally occurring mineral fiber with strong commercial appeal due to its physical properties. It is a poor conductor, but a good insulator, and is strong, flexible, non-corrosive, and flame-resistant. Asbestos is a commercial term for a number of naturally-occurring

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minerals: chrysotile belongs to the serpentine group, while amosite, crocidolite, tremolite, actinolite, and anthophyllite are in a group called amphiboles. When inhaled, asbestos fibers penetrate deep into lung tissues where they cannot be expelled or destroyed by the body. Federal and International agencies recognize that asbestos is a carcinogen. In 1986, asbestos was identified as a toxic air contaminant under California's Toxic Air Contaminants Program (AB 1807). Exposure to asbestos is associated with increases in non-malignant respiratory symptoms, and may cause asbestosis, lung cancer, and mesothelioma (ATSDR, 2001). Indoor asbestos is primarily found in older homes in pipe and furnace insulation, shingles, millboard, textured paints and other coating materials, and floor tiles. Due to remediation efforts, asbestos concentrations are generally quite low in most buildings today.

2.3.10.1 Health Effects of Asbestos

Lung cancer is the predominant asbestos-related disease, accounting for the majority of deaths from asbestos exposure. Asbestos workers who smoke are at greater risk of developing lung cancer than workers who do not smoke. Asbestosis, a diffuse fibrous scarring of the lungs characterized by shortness of breath, mainly arises after long-term heavy exposure to asbestos. Mesothelioma, a rare cancer of the lining of the chest and abdomen, does not appear to be influenced by smoking. A study of vermiculite miners in Libby, Montana revealed a 4.2% mortality rate due to mesothelioma (McDonald *et al.*, 2004). Based on the high-end estimate of the potency of mixed asbestos fibers, breathing air that contains 100 fibers/m³ (0.0001 fibers/ml) poses a one to two in ten thousand excess lifetime risk of cancer (ARB, 1986). Others (U.S. EPA, 1994) have made similar estimates.

Non-occupational exposures are generally low compared to occupational exposures. Workers' families have developed asbestos-related pulmonary disease (Whitehouse, 2000; Peipins *et al.*, 2003), lung cancer and mesothelioma (Magnani *et al.*, 2001) from para-occupational exposure to asbestos, such as inhalation of asbestos fibers released by clothing and equipment brought home from job sites. People living in the vicinity of asbestos mines and factories may be at risk from neighborhood exposure to asbestos. Although the risk to the general population is minimal, no safe exposure level for asbestos has been established.

2.3.10.2 Sources of Asbestos

For 30 years following World War II, asbestos was extensively used in the renovation and construction of homes, schools, and public buildings. Asbestos was once found in nearly 3,000 different types of commercial products, including older plastics, paper products, brake linings, floor tiles, textiles, sealants, cement pipe, cement sheets, and insulation (NTP, 2002). It is now prohibited in the manufacture, processing, and importation of most products by the 1989 Asbestos Ban and Phase Out Rule (40 CFR 763 Subpart I, Sec. 762.160 - 763.179). Some asbestos paper products and asbestos-cement products remain in use today.

Friable asbestos-containing material (ACM) can be crumbled by hand pressure when dry, releasing fibers to air for potential exposure by individuals. Non-friable ACM can become friable during abatement and maintenance activities, and with age. The three most common asbestos forms used in construction are amosite, crocidolite, and chrysotile (DHS, 2000). When the minerals are crushed, they break into tiny fibers invisible to the naked eye, but distinguishable under light microscope. Chrysotile fragments into curly fibers; the other types release tiny, needle-like fibers.

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Homes built in areas of naturally occurring asbestos may become contaminated with tracked-in asbestos and fibers entering in suspended outdoor air. Once such fibers are indoors, they can be resuspended by normal household activities, such as vacuuming (OEHHA, 2000b).

2.3.10.3 Asbestos Concentrations

Asbestos concentrations in most buildings are quite low. Crump and Farrar (1989) examined indoor asbestos concentrations for 49 public buildings located in six cities across the U.S. The concentration of all fibers was 0.00073 f/mL and for fibers $\geq 5 \mu\text{m}$ the average concentration was 0.00007 f/mL. The mean indoor concentration was 0.00020 f/mL for buildings with no known asbestos, 0.00059 f/mL for buildings with asbestos in good condition, and 0.00073 f/mL for buildings with damaged asbestos. There were no statistically significant differences in airborne asbestos levels among the three categories of buildings. For the 43 buildings that contained ACM, the average indoor levels for fibers $\geq 5 \mu\text{m}$ were 0.00005 f/mL; these levels were indistinguishable from outdoor levels (Lee *et al.*, 1992).

In a nationwide study conducted by Lee *et al.* (1992), airborne asbestos concentrations were made for 315 public, commercial, residential, school, and university buildings. The mean indoor level was 0.02485 f/mL for total asbestos structures and 0.00013 f/mL for fibers longer than 5 μm . Indoor total asbestos was significantly higher than outdoor total asbestos for all building types. In comparing indoor and outdoor levels of fibers $\geq 5 \mu\text{m}$ for building types, indoor levels were higher than outdoor levels only for schools ($p=0.003$). Only fifty-two percent of all indoor samples contained asbestos fibers. Approximately 92% of fibers were shorter than 5 μm in all buildings, and 2% of fibers were amphiboles. Ninety-percent of the samples for fibers longer than 5 μm had levels less than 0.00071 f/mL and 0.00054 f/mL in schools and public buildings, respectively.

Corn *et al.* (1991) reported asbestos concentrations for a portion of the study conducted by Lee *et al.* (1992) involving 71 occupied schools in 8 different states. Schools from different states did not differ in their levels of airborne asbestos. The type of asbestos-containing materials (ACM) present (i.e. chrysotile or amphibole) and the condition of ACM were not significantly correlated with levels of indoor asbestos. Most of the asbestos fibers were chrysotile (95%) with an average of 0.018 f/mL, and few were longer than 5 μm (7.9%). The mean indoor concentration of fibers $> 5 \mu\text{m}$ long was 0.00024 f/mL. Total asbestos structures found indoors averaged 0.20 f/mL.

2.3.11 Polybrominated Diphenyl Ethers and Other Endocrine Disruptors

Endocrine disruptors are substances that alter the normal function(s) of the endocrine systems of animals and humans and adversely affect growth, development or reproduction. They can act like a natural hormone, bind to a receptor and prevent a normal response, or interfere with the way natural hormones and receptors are synthesized or controlled. Public attention has been drawn to endocrine disruptors that mimic or block the natural effects of female sex hormones (estrogens), but they can also affect male sex hormones, development and behavior. The range of substances that cause endocrine disruption is wide and varied, and includes both natural and synthetic chemicals.

Polybrominated diphenyl ethers (PBDE), a chemical class of endocrine disruptors, are added to plastics and textiles to meet fire safety regulations. The PBDE class is comprised of 209 possible congeners, differentiated by the position and number of bromine atoms on the two phenyl rings. U.S. EPA has classified deca-BDE as a possible human carcinogen since it has

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been linked to development of liver tumors in rats and mice. PBDEs have been identified as federal Hazardous Air Pollutants (HAP) and California Toxic Air Contaminants (TAC).

2.3.11.1 Health effects of PBDE

Research has shown that PBDE compounds are toxic and bioaccumulate similarly to polychlorinated biphenyls (PCBs) and dioxins and have genotoxicity profiles similar to PCBs. PBDEs have similar chemical structures to PCBs and the human thyroid hormone, Thyroxine (T4). Toxicological endpoints of PBDEs are suspected to be thyroid hormone disruption, neurodevelopmental deficits and cancer (McDonald, 2002).

Neurodevelopmental deficits have been linked to PBDE exposure (Eriksson *et al.*, 1998, 1999; Viberg *et al.*, 2000, 2001). Other results indicated deficits in brain development and possibly causing changes in the cholinergic system. Changes in the cholinergic system interfere with choline acetyltransferase activity and contribute to the motor and mental impairments in animals. A no-observable-effects level for neurobehavioral effects has not been established for PBDEs (McDonald, 2002).

Evidence shows that exposure to these substances can lead to disruption of endocrine function in a number of wildlife species. Effects suggesting endocrine disruption have been reported in mollusks, crustaceans, fish, reptiles, birds and mammals in various parts of the world. There is limited evidence in humans that adverse endocrine-mediated effects have resulted from either intentional or accidental exposure to chemicals. Diethylstilbestrol (DES), a synthetic estrogen prescribed in the 1960s to pregnant women to prevent spontaneous abortion, was found to cause developmental abnormalities in some children exposed *in utero*, and an unusual form of vaginal cancer in some girls when they reached puberty. In addition, adverse health effects were reported in a population living near a chemical plant in Italy as a result of accidental release of dioxin, a suspected endocrine disrupter. Scientists have now focused attention on endocrine disrupting substances that are not so highly persistent, but are widespread in the environment. Many of these chemicals, even at relatively low levels, are known to affect growth, reproduction and development of organisms in the ecosystems. The impact of these substances on human health is still under investigations.

2.3.11.2 Sources of PBDE

The production of PBDEs has steadily increased since the 1970s. Commercial forms of PBDEs include deca-, octa- and penta-BDE. In 1999, its use in the United States was estimated at 25,000 tons or 44% of the total global use (Hale *et al.*, 2002). California is a major user of fire retardants due to strict regulations concerning consumer safety.

Commercial penta-BDE is added to soft polyurethane foam cushions, upholstery textiles and mattresses. Commercial octa-BDE is incorporated into plastics used for hard casings of office equipment, fax machines, computers, telephone handsets and car trim. Commercial deca-BDE is utilized in high-impact plastics found in televisions, computers, stereos and other electronic equipment.

2.3.11.3 PBDE Concentrations

The air and dust inside U.S. homes are likely to contain a wide variety of chemicals and pesticides, many of which are identified as endocrine disrupting substances. A recent study

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(Rudel et. al., 2003) found numerous endocrine disrupting compounds in indoor air and dust obtained from 120 homes in Cape Cod, Massachusetts. The most abundant compounds in air included bis(2-ethylhexyl) phthalates (DEHP)(a plasticizer used in children's toys, shower curtains, raincoats, shoes, and floor tiles); o-phenylphenol (disinfectant); 4-nonylphenol (detergent metabolite); and 4-tert-butylphenol (adhesive). The penta- and tetrabrominated diphenyl ethers used as flame retardants were frequently detected in dust. Numerous pesticides were detected in air and dusts from these homes, the most abundant being permethrins and the synergist piperonyl butoxide. The banned pesticides heptachlor, chlordane, methoxychlor, and DDT were also frequently detected.

PBDE congeners were quantified in samples of human breast milk collected in Sweden during the period from 1972 to 1997 (Noren and Meironyte, 1998). This Swedish study reported that levels of PBDEs in breast milk had increased 40 fold since 1972, indicating a doubling every 5 years. In a 2003 U.S. study by Schechter, *et al.*, levels of PBDEs in women's breast milk measured in Texas were 10 to 100 times higher compared to levels found in Europe (Schechter *et al.*, 2003). Two recent U.S. studies in California (Petreas *et al.*, 2003) and Indiana (Mazdai *et al.*, 2003) also report similar results with levels from 10 to 100 times higher than levels reported in Europe (Schechter *et al.*, 2003).

PBDEs have been found in a wide variety of environmental samples, including fish, birds, soil sediments, air, marine mammals, and human blood (Strandberg *et al.*, 2001, Darnerud *et al.*, 2001). Penta-BDE has been detected among 89 percent of the fish collected from two large Virginia watersheds (Hale *et al.*, 2001a).

An air study found that PBDE compounds were widely distributed and that PBDEs with lower molecular weight, such as penta-BDE (which is found mostly in the vapor phase), can be transported through the atmosphere to remote areas (Strandberg *et al.*, 2001).

3. COSTS OF INDOOR AIR POLLUTION

The health effects from indoor air pollution impose large costs on society, both economic and personal. The loss of human life, or increased mortality, due to indoor air pollution constitutes a large economic and social cost. In addition, the medical costs of increased illness or morbidity due to indoor air pollution, such as for cancer treatments, hospitalizations, chronic respiratory disease, and emergency room visits for asthma attacks and CO poisoning, are substantial. Student absenteeism, reduced worker productivity, and associated costs can also result from indoor air pollution. However, only very limited quantitative estimates on the costs of indoor pollution are available to date.

Estimates of the economic costs of some aspects of indoor air pollution have been made for the United States (Fisk, 2000; Mendell *et al.*, 2002). These studies allow order-of-magnitude estimates of the economic costs for California due to increased asthma symptoms and lost worker productivity in schools and office buildings. In addition, California-specific information for the magnitude of health impacts due to indoor air pollutants is presented in Section 2, Health Effects, in this report. Cost or value estimates can be derived for some of these health impacts.

This information is used below, along with national and California-specific cost estimates for medical treatment and other costs (where available), to estimate the economic costs or value of specific health impacts from indoor pollutants in California (U.S. EPA, 2002b; Thayer *et al.*, 2003). However, only the costs for selected indoor air pollutant exposures that have been reasonably well characterized in California are used for the final estimate. To account for inflation, cost estimates are adjusted to 2000 dollars using the relative increase in the urban consumer costs of the Consumer Price Index; for adjusting health care costs, the changes in medical care costs are used instead (USCB 2002). Table 3.1 presents the unit medical costs (direct and indirect) and economic valuations of premature deaths used in the following analyses.

3.1 MORTALITY (PREMATURE DEATH)

Exposure to some air pollutants, primarily carbon monoxide, ETS, other carcinogens, and outdoor PM, can lead to near-term death or significantly increase the risk of premature death. The economic impact of premature death, or the value of a human life, has been estimated in the scientific literature and in environmental regulatory settings. The estimation methods include those based on: 1) an individual's willingness to pay to avoid a health risk, 2) the additional compensation demanded in the labor market for riskier jobs, and 3) society's willingness to pay to avoid a health or safety risk.

U.S. EPA (2002b) reviewed 26 value-of-life studies that used either the first or second method above, and confirmed the finding of a previous literature review that "most of the reasonable estimates of the value of life are clustered in the \$3.7 to \$8.6 million range." Based on its review and input from the scientific community, U.S. EPA selected \$6 million (1999 dollars) as a point estimate for the "value of a statistical life," along with caveats for applying the study results to the general population. U.S. EPA (2000c) has used this point estimate in assessing the impact of regulations for diesel fuel and other pollutant sources. This point estimate is equivalent to \$6.33 million in 2000 dollars, and is used in the following sections to estimate the economic valuation of premature deaths due to indoor air pollution. The estimated valuation of mortality from indoor air pollution in California is discussed below and summarized in Table 3.2.

Table 3.1. Unit Costs for Health Effects.

Outcome	Method¹	Unit Cost in year 2000 dollars ²
Premature death	WTP ³	\$6,330,000
Asthma, chronic	WTP	\$33,000
Asthma exacerbation (attack, moderate or worse asthma day)	WTP	\$42
Bronchitis, acute	WTP	\$59
Cancer medical costs (various types, survivor only)	COI	\$94,600
Lung cancer medical costs (weighted average for all cases)	COI	\$64,900
Low birth weight, lifetime	COI	\$118,000
Asthma emergency room visit	COI	\$310
Cardiovascular hospitalization	COI ⁴	\$15,200
Respiratory hospitalization (US)	COI	\$11,000 ⁵
Acute respiratory hospitalization (CA) Age < 18 Age ≥ 18	COI ⁴	\$11,800 ⁵ \$23,500 ⁵

1. WTP: willingness to pay method; obtained from surveys. COI: cost of illness method; based from medical cost data and estimates of indirect costs such as lost workdays.
2. Unless noted otherwise, all values are averages from *Cost of Illness Handbook* (U.S. EPA, 2002b), after adjustment to 2000 dollars using the Consumer Price Index for medical care cost.
3. Midpoint value, adjusted for inflation using urban consumer costs.
4. From a California study by Thayer *et al.* (2003). Cost adjusted for inflation using the Consumer Price Index for medical care costs.
5. These values not used in this analysis due to inability to separate hospitalization cases from doctor's visits in literature. However, some cases are hospitalized, and this would increase the cost of the morbidity estimates notably.

Another approach to estimating the economic cost from deaths due to indoor air pollution is to base the estimate on court decisions and insurance awards, such as those from CO poisoning. However, nearly all cases of fatal and non-fatal CO poisoning are settled out of court, and utility companies are not required to report such costs to the California Public Utility Commission; thus, there is little reliable data available on these legal settlements.

3.1.1 Deaths From CO Poisonings

Based on the DHS review of coroners' reports in the 1980's, an average of 30-40 Californians die each year from accidental CO poisoning (Girman *et al.*, 1998; Liu *et al.*, 1993a, 2000). On average, about 2/3 of those deaths (about 20-26, or 23 average) documented in the DHS study were due to appliance-related causes such as faulty furnaces, gas ovens used for space heating, and charcoal grills used indoors. This estimate has a high degree of certainty relative to

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other pollutant risk estimates because the CO death estimates are based on coroners' reports that clearly identify CO poisoning deaths. The economic value of these lost lives in California is estimated to be \$130-160 million per year (2000 dollars), or about \$150 million on average, as shown in Table 3.2.

Table 3.2. Estimated Valuation of Mortality Attributed to Indoor Air Pollution in California

End Point	Premature Deaths (cases/yr) ¹			Valuation per case ² (\$ Million)	Cost per Year ³ (\$ Billion)		
	Low	Average	High		Low	Average	High
CO: Poisoning ⁴	20	23	26	6.33	0.13	0.15	0.16
VOCs: Cancer ⁵	115	115	115	6.33	0.73	0.73	0.73
ETS: Lung cancer ⁶	380	380	380	6.33	2.4	2.4	2.4
ETS: Heart disease ⁷	1,700	3,600	5,500	6.33	11	23	35
Mold and Moisture: Asthma and allergy ⁸	NA	NA	NA	NA	0.031	0.031	0.031
TOTAL⁹					14	26	38

1. Low and high estimates are based on a range or confidence intervals, where available. Average estimates are based on mean values from available estimates or ranges.
2. From *Cost of Illness Handbook* (USEPA, 2002b), adjusted to 2000 dollars.
3. Costs per year are rounded to 2 significant figures.
4. Case estimate from coroner's data in California (Liu *et al.*, 2000).
5. Case estimate from California Comparative Risk Project (CCRP, 1994), updated. See Section 2 and Appendix II of this report.
6. From OEHHA (1997). Adjusted for 1996-2000 population growth of 5.8%. This estimate is probably an overestimate because it does not account for reductions in ETS exposure in California.
7. From OEHHA (2003b), based on 1999 data. OEHHA is currently updating these estimates.
8. Cost estimates taken from national cost estimates by Weiss and Sullivan (2001), inflated to \$2000 dollars, scaled to 12% for California estimate, and multiplied by fraction of asthma risk attributed to mold and moisture in three large epidemiology studies (0.13).
9. Rounded to 2 significant figures. These totals do not include mortality from exposures to other indoor PM, which could be substantial but is not currently quantifiable due to lack of appropriate studies.

Although the rate of CO deaths in the U.S. population has declined over the years (CPSC, 1997), the actual numbers of CO poisoning cases in California may have increased since 1990. Based on California's population growth alone – from 30.38 million in 1990 (USCB, 1990) to

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33.87 million in 2000 (DOF, 2002a,b), a 14% increase – one might expect the number of deaths to have increased by a similar percentage. In addition, much of this population growth has occurred in the inland and foothill regions of California, which have longer, colder winters than coastal areas, and thus, would have a greater likelihood of CO poisoning due to more frequent use of combustion appliances for space heating.

However, trends toward increased use of CO alarms and testing of combustion appliances could potentially have reduced the number of appliance-related CO poisonings in California since 1990. CO poisonings at the national level have declined (CPSC, 1997), which may be due to lower pollution emissions from motor vehicles and the increased heavy marketing of inexpensive CO alarms for use in homes, motor homes, and boats. The state's program for weatherizing existing, low-income homes has required installation of CO alarms and combustion appliance safety testing, as discussed in Sec. 4.3.2, Appliances.

Overall, based on the factors in both directions discussed above, we believe the annual number of fatal CO poisonings in California is similar to that in 1990. It is likely that the risk of CO poisoning remains higher than average in older homes with older gas appliances, in homes with propane appliances, and in some socioeconomic groups that use unvented gas stoves or charcoal grills indoors more frequently.

3.1.2 Deaths From Volatile Organic Compounds (VOC) Exposure

As discussed in Section 2 and Appendix II, about 230 excess cancer cases due to VOCs from indoor exposures are estimated to occur in California each year. This estimate is based on the results of the California Comparative Risk Project (CCRP, 1994) that have been updated to reflect reductions in formaldehyde exposures and increased population growth as of the year 2000. These are conservative estimates relative to the total cancer burden from indoor carcinogens, because they do not include:

- The additional, significant cancer risks from radon.
- The risk from many other carcinogens also found in indoor air and house dust, such as acetaldehyde, PAHs other than B(a)P, phthalates other than DEHP, and asbestos.

Half of the VOC-related cancer cases attributed to indoor air (115) per year are estimated to result in premature death. U.S. EPA (2002b) assumed this same fraction of fatal cases within five years of diagnosis when estimating the costs of typical cancer treatment. Actual premature death rates could be higher than one-half of the cases because cancers caused by these air pollutants typically result in lung, organ, or blood cancers, which are more difficult to detect and treat, and thus, are more deadly. The estimated value of premature deaths from the 115 excess cancer deaths in California totals \$730 million per year, as shown in Table 3.2. This estimate does not explicitly include the much smaller costs of cancer treatment, which are discussed below.

3.1.3 Deaths from Environmental Tobacco Smoke Exposure

OEHHA (1997) estimated that exposure to ETS in California in 1996 produced 360 premature deaths due to lung cancer. These estimates were based on 1996 smoking prevalence rates in the U.S., and were scaled down to reflect that California contained 12% of the U.S. population. OEHHA (2003b) is in the process of updating these estimates; the updated estimates will undergo external scientific review and additional public review later this year. In the absence of other more recent risk estimates that have completed public review, OEHHA's 1997 estimates are used here to estimate the costs of ETS-related lung cancer in California.

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To update the OEHHA 1996 estimate, the number of lung cancer deaths is multiplied by 1.058 to adjust for population growth in California from 1996-2000. This yields an estimated number of 380 excess lung cancer deaths cases per year from ETS exposures. The estimated value of premature deaths from the 380 excess lung cancer deaths in California totals \$2.4 billion per year, as shown in Table 3.2. This estimate does not explicitly include the much smaller costs of cancer treatment, which are discussed below

For ETS effects on ischemic heart disease, OEHHA (2003b) has published more recent estimates. Using 1999 mortality and exposure data, OEHHA estimated that ETS exposure resulted in 1,700-5,500 (average = 3,600) premature deaths from heart disease. The estimated value of 3,600 deaths is \$23 billion, as shown in Table 3.2. This is probably an overestimate because of major reductions in ETS exposure by the year 2000.

These estimated numbers of premature deaths, and the morbidity effects discussed below probably overestimate current risk levels due to major reductions in ETS exposure since 1996. These estimates are based on U.S. smoking prevalence rates in 1996, or on 1999 data that included some workplace ETS exposure, but ETS exposure in California had already declined greatly by 2000. For example, smoke-free homes in California increased from 69.2% of all homes in 1996 to 79.5% in 2000 (DHS, 2002). In addition, California's workplace smoking ban was expected to virtually eliminate ETS exposure in enclosed workplaces by 2000 – by 1999 over 93% of indoor workers reported having a smoke-free workplace (DHS, 2002). On the other hand, newer data on the health effects of ETS may increase the unit risk of ETS for mortality and morbidity. OEHHA is currently revising the risk estimates for ETS in California to reflect current data on ETS exposure data and health effects.

These costs of ETS-related health effects could be prevented if people smoked only outdoors or stopped smoking. However, to achieve these potential cost savings, further efforts would be needed to change peoples' smoking behaviors in California.

3.1.4 Deaths from Mold and Moisture-related Problems in Homes

Weiss and Sullivan (2001) estimated that the costs of excess mortality in children and adults due to asthma in the U.S. was \$2 billion, in year 2000 dollars. Scaling this to reflect that 12% of the U.S. population resides in California gives an estimated cost for asthma deaths in California of \$0.24 billion. As discussed below in the Medical Costs section, the fraction of these health effects attributed to mold and moisture problems in homes is 0.13. This fraction is an average of four large, well-conducted epidemiology studies of adults and/or children (shown in Table 3.4). Multiplying the estimated cost of asthma in California by this fraction yields an estimated cost of \$31 million attributable to excess mortality due to residential mold and moisture problems. This estimate does not include other indoor allergen sources such as dust mites, pets, cockroaches, and chemical emissions.

3.2 MEDICAL COSTS

Illness and disease caused by indoor air pollution include the production of new asthma cases (induction), exacerbation of asthma symptoms, development of other respiratory disease and symptoms, and induction or exacerbation of allergies. These impacts of indoor air pollution also affect the quality of a person's life in terms of reduced or limited activities, limited employment opportunities, and reduced productivity.

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The cost of illness and disease (morbidity) has been estimated in various ways. These include methods based on medical costs, work-related costs, education-related costs, and willingness of individuals to pay to avoid the anxiety, pain, suffering, and other health risks (U.S. EPA, 2002b). The following section uses medical costs shown in Table 3.1 to estimate the medical costs of indoor air pollution in California, but generally excludes the indirect costs of reduced productivity of individuals and employees. The estimated medical costs are summarized in Table 3.3.

3.2.1 CO-Related Hospitalization

About 175-700 emergency room visits and hospitalizations (midpoint of 438) due to non-fatal CO poisoning are estimated to occur in California each year, on average (see Section 2). Emergency room visits cost about \$442 per visit, based on costs for asthma room visits to emergency rooms and adjustment to 2000 dollars (U.S. EPA, 2002b). Therefore, the annual cost of emergency room visits due to CO poisoning in California ranges from \$77,000-310,000, or an average of about \$190,000, as shown in Table 3.3. This cost estimate is conservative relative to the total morbidity impacts of CO, because it does not include the lost school and work days, the cost of hyperbaric oxygen treatment for severe poisoning, the long term neurological and developmental damage to many victims of sub-lethal CO poisoning, or the estimated hundreds to thousands more CO poisoning cases per year that do not result in hospitalization.

3.2.2 VOC-Related Cancer Treatment

The number of excess cancer cases from VOC exposures was estimated above to be 230 cases per year, and one-half of those cases (115) were assumed to survive, as discussed above. The average cost of medical treatment in the U.S. for the 13 most common types of cancer, when assuming a 50% mortality rate at five years and no discounting, is \$82,581 in 1996 dollars (U.S. EPA, 2002b) or \$94,619 in 2000 dollars. This treatment cost estimate is conservative because it does not consider recent changes in cancer treatment that can be very expensive, such as the bone marrow transplant procedure and new pharmaceutical treatments. At this treatment cost per cancer case, the total cost for medical treatment for 115 cases from VOC exposure was estimated to be \$11 million, as shown in Table 3.3.

3.2.3 ETS-related Illness and Hospitalization

As discussed above, ETS exposure was estimated to result in 380 excess deaths per year from lung cancer. To estimate the medical costs of this impact, the approach used above for VOCs was used, except that the average cost of medical treatment for lung cancer alone was used instead: \$56,624 in 1996 dollars (U.S. EPA, 2002b) or \$64,900 in 2000 dollars. Costs for lung cancer only were used because ETS exposure is primarily associated causally with lung cancer, while VOCs are causally associated with several other types of cancer. This cost is based on the total costs for non-survivors and survivors over 10 years, weighted annually for survival rates and discounted at 5%. The total cost of cancer treatment for the 380 cancer cases from ETS exposures was estimated to be \$25 million (in 2000 dollars), as shown in Table 3.3.

Table 3.3. Estimated Annual Medical Costs of Indoor Air Pollution in California.

Indoor Pollutant and Health End Point	Average Cases per Year ¹ (Range)	Medical Cost per Case ^{1,2} (\$)	Average Cost per Year ³ (\$ millions)
CO: poisoning	438 (175-700)	442	0.19
VOCs: cancer	115	94,600	11
ETS: lung cancer ^{4,5}	380	64,900	25
ETS: heart disease ^{5,6}	3,600 (1,700-5,500)	15,200	55
ETS: asthma induction ⁴	2,200 (1,000-3,300)	33,000	73
ETS: asthma exacerbation ⁶	38,000 (29,000-47,000)	42	1.6
ETS: bronchitis & pneumonia ⁴	29,000 (19,000-38,000)	59	1.7
ETS: low birth weight ⁶	1,600	118,000	190
Mold & moisture: asthma and allergy ⁷	NA	NA	190
Total⁸			550

1. Original data were adjusted to 2000 dollars and 2000 population. NA = not available.
2. Medical cost values were taken from Table 3.1 and do not include indirect costs, unless noted otherwise. Cancer treatment cost does not include costs for outpatient prescription medications and nursing home care below the skilled level.
3. Rounded to two significant figures. These totals do not include morbidity from indoor sources of PM, which could be substantial.
4. From OEHHA (1997). Adjusted for 1996-2000 population growth of 5.8%. These estimates are probably overestimates because they do not account for reductions in ETS exposure and the workplace smoking ban in California.
5. Medical costs for fatal cancer cases also are implicitly included in mortality valuations, but are generally an insignificant amount relative to the total valuations.
6. From OEHHA (2003b), based on 1999 data.
7. Cost estimates taken from national estimates for medical and indirect costs by Weiss and Sullivan (2001), adjusted to 2000 dollars, scaled to 12% for California estimate, and multiplied by fraction of asthma risk attributed to mold and moisture in four large epidemiology studies (0.13). Includes estimate of \$60 million for indirect costs, such as lost workdays, lost school days, reduced housekeeping.

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For estimating costs due to hospitalization for heart disease due to ETS exposure, an approach similar to that used above for VOCs was used. However, in this case, the average cost of medical treatment for heart disease (\$15,200) from Table 3.1 was used. The total cost of hospitalization for 3,600 heart disease cases attributed to ETS exposures was estimated to be \$55 million (in 2000 dollars), as shown in Table 3.3.

ETS exposure in California is estimated to have resulted in 960-3,120 new cases of asthma (OEHHA, 1997), 29,000 to 47,000 cases of asthma exacerbation (OEHHA, 2003b), and 18,000-36,000 cases of lower respiratory illness, i.e., bronchitis or pneumonia (OEHHA, 1997). These health effects were primarily observed in infants and children, but they may also increase the risk of health effects later in life. Using the same approach as that used above for heart disease, the 1997 case estimates were updated to 2000 population levels to yield the ranges shown in Table 3.3. The midpoint of these ranges was then multiplied by the relevant unit medical costs shown in Table 3.3. The actual costs for asthma exacerbation, bronchitis, and pneumonia are likely to be substantially higher because some fraction of those cases would require emergency room treatment or hospitalization, which have much higher costs than those assumed here, as shown in Table 3.1.

OEHHA (1997) estimated that ETS exposure resulted in 1,200-2,200 (average = 1,600) cases of low birth weight. Again the case numbers were updated for population growth and multiplied by the unit medical cost shown (\$118,000) in Table 3.1. The estimated cost of the 1,600 cases of low birth weight infants was \$190 million. This unit medical cost is likely to be an underestimate of actual costs because credible data are lacking for the first two years of life when intensive medical care and additional hospitalization are most likely, and for some age groups where costs were extrapolated from others (U.S. EPA, 2002b). In addition, cost data for non-medical expenses over a lifetime were lacking; these costs could be substantial as well.

OEHHA (1997) also estimated that 76,600-188,700 cases of otitis media were associated with ETS exposure in California. However, cost estimates were not available yet for otitis media, so California costs for this effect of ETS exposure were not estimated. The potential costs of otitis media attributed to ETS exposure in California could be substantial, considering the potential costs of numerous hospital visits, surgery, lost workdays, and time spent by caretakers.

3.2.4 Medical and Indirect Costs Associated with Indoor Mold and Moisture-related Problems in Homes

Available data demonstrate a relatively strong and consistent association of dampness and mold in buildings with an increase in lower respiratory symptoms that are often considered evidence of asthma exacerbation (IOM, 2004). While the health-relevant exposures from dampness and mold are not fully understood and may not be exclusively particles, available data suggest that biological particles are the dominant source of the related adverse health effects. Few large studies of mold, allergens, and other biological contaminants have been conducted in office buildings, schools, and other non-residential buildings, so this section focuses on residential exposures.

To estimate the cost of asthma attributable to dampness or mold, the fractions of asthma attributable to dampness or mold in apartments and homes in the U.S. and California are multiplied by estimates of the cost of asthma. Four large studies of the health risks of building dampness and mold for adults and/or children have been conducted in the U.S. and Canada, as shown in Table 3.4. Each of these studies controlled for the effects of many factors other than

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dampness and mold. Equation 1 was used to estimate the fraction of asthma attributable (AF) to dampness and mold, using odds ratios (OR) and prevalence rates (p) specific to each study population:

$$AF = p(OR-1) / p(OR-1)+1 \quad [Equation 1; Lilienfeld and Lilienfeld, 1980]$$

Table 3.4. Fraction of Asthma Attributable to Mold or Dampness in Houses

Study and Condition	Study Population	Prevalence of Condition (%)	Asthma Symptoms	
			Adjusted Odds Ratio	Estimated Attributable Fraction
Spengler <i>et al.</i> (1994), mold or mildew	12,842 U.S. children ages 9-11	36	1.39 (1.23-1.57)	0.12
Dales <i>et al.</i> (1991a), dampness or mold	13,495 Canadian children ages 5-8	38	1.45 (1.23-1.71)	0.15
Dales <i>et al.</i> (1991b), dampness or mold	14,700 Canadian adults	38	1.56 (1.25 -1.95)	0.18
Zock <i>et al.</i> (2002), mold or mildew last year	19,218 adults, 38 centers in U.S., Europe, Australia, India, New Zealand	22	1.28 (1.13-1.46)	0.06
AVERAGE				0.13

Table 3.4 shows the calculations of attributable fraction¹ using odds ratios, rather than relative risks, from the results of these four major studies. The average attributable fraction of asthma due to dampness or mold in homes is estimated to be 0.13.

To estimate the national cost of asthma, two estimates of asthma costs (Weiss and Sullivan, 2001; Smith *et al.*, 1997) were reviewed. The estimate of direct costs from the two studies are similar. However, the estimate of medical and indirect costs based on Weiss and Sullivan (2001) are more recent than those from Smith *et al.* (1997), and they do not exclude persons in the military, schools, and other institutions. Thus, the Weiss and Sullivan results may better reflect the current costs of treatment and medications, and are used here.

The Weiss and Sullivan estimates were adjusted to 2000 dollars and 2000 population using the data on population growth and medical care and general inflation. No adjustment was made for changes in asthma prevalence because it is not clear that asthma prevalence has changed since 1994 (Mannino *et al.*, 2002). It was also assumed that the prevalence of asthma in California is similar to the average prevalence of asthma in the U.S. Data in IOM (2000) indicate

¹ The attributable fraction is normally calculated using the relative risk value, but odds ratios can be used instead where the outcome prevalences are relatively low. For example, with prevalence rates of 11.4% for asthma symptoms and 16.9% for lower respiratory symptoms in the study by Spengler *et al.* (1994), the relative risks are approximately equal to the odds ratios. Thus, the attributable fraction can be estimated using odds ratios (ORs) in place of relative risks (RRs).

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that self-reported asthma prevalence in California was 7.1% versus 6.4% for the U.S.; however, more recent data (Rhodes *et al.*, 2002) indicate that the prevalence of current asthma in 2002 was 6.4% in California versus 7.5% for the full U.S. Thus, it is not clear that asthma prevalence in California differs from that for the U.S. at this time.

The updated costs for the full U.S. were then multiplied by the 0.12, the percentage of U.S. population that resided in California in 2000. As shown in Table 3.5, the estimated mortality costs are \$24 million, as discussed in the mortality section above. The estimated medical costs and indirect costs (such as lost work days and school days) of asthma in California are \$980 million and \$460 million, respectively.

To estimate the asthma costs attributed to indoor mold and moisture problems, these costs of asthma in California were multiplied by 0.13, the attributable fraction of asthma from mold and dampness that was calculated above. This yields the estimated medical and indirect costs of \$130 million and \$60 million, respectively, for a total of \$190 million, shown in Table 3.3. This estimate does not include the costs of other indoor allergen sources such as dust mites, pets, cockroaches, and chemical emissions.

Table 3.5. Updated Estimates of Annual Cost of Asthma.

Source and Population	Cost in U.S. (\$ billions) ¹				Cost in California (\$ billions) ¹			
	Mortality	Medical	Indirect ²	Total	Mortality	Medical	Indirect ²	Total
Weiss and Sullivan (2001)								
Children			1.2				0.14	
Adults			2.6				0.31	
Both	2.0	8.2	3.8	13.9	0.24	0.98	0.46	1.7

1. In 2000 dollars. Row and column totals may not add precisely due to rounding of numbers.

2. Indirect cost elements include loss of work, loss of school, reduced housekeeping.

Extensive data on are not available on the costs of asthma that are preventable, but a large portion would appear to be readily preventable. The particle sources considered in this analysis are tobacco smoking, pets, use of gas stoves for heat, dust mites, cockroaches, and dampness/mold. Behavioral changes can eliminate all indoor tobacco smoking, indoor pets in homes of people with pet allergies, and use of gas stoves for heat in the homes of asthmatics. Dust mite allergen levels can be diminished by reducing indoor humidity and by surface cleaning; however, studies of dust mite remediation measures have had only moderate success. Cockroach infestations can be reduced substantially using pest management methods. Mold contamination in buildings can be reduced by preventing and remediating dampness problems.

Many dampness problems, probably a majority of serious problems, result from water leaks that could be prevented through better building maintenance and improved design and construction. These measures would also reduce the costs of dampness-caused mold contamination and degradation of building materials. Better ventilation and use of dehumidifiers could reduce dampness problems that result from high indoor humidity. Thus, with proper measures, it is probably feasible to eliminate at least 50% of the particle exposures that contribute to asthma exacerbation, and likely more.

3.3 PRODUCTIVITY COSTS OF INDOOR AIR POLLUTION

Sick building syndrome is a collection of non-specific symptoms such as eye, nose, skin, and throat irritation; headache; fatigue; and skin rash that have no known cause. Inadequate building ventilation, as indicated by elevated indoor CO₂ concentrations, as well as elevated indoor levels of VOCs, elevated levels of biological contaminants, and other environmental stressors have been implicated as potential causes of sick building syndrome.

Fisk (2000) estimated the economic impacts of sick building syndrome in the U.S. due to reductions in worker productivity. Several field and laboratory studies of office buildings and school buildings were reviewed; the performance reduction for specific tasks ranged from 3-5%. The midpoint value of 4% was reduced to 2% to estimate the overall productivity reduction throughout the day. Using this conservative estimate of a 2% preventable reduction in worker productivity due to sick building syndrome, the estimated cost savings for the U.S. in 1996 dollars were \$10-30 billion (\$11-33 billion in 2000 dollars). An independent group of scientists who reviewed the literature and assessed the impacts of indoor air quality on worker health and productivity for the National Institute of Occupational Safety and Health estimated the same costs (Mendell *et al.*, 2002).

To estimate the costs of sick building syndrome in California, this \$11-33 billion U.S. estimate by Fisk can be scaled to reflect California's portion of the US population, or 12%. This would yield a California estimate of \$1.3 - 4.0 billion per year, for an average of \$2.6 billion per year.

Kats (2003) used the 2% preventable reduction in worker productivity estimate by Fisk (2000) to estimate an avoidable cost of sick building syndrome in California. Using California-specific data where available, Kats estimated this avoidable cost to be \$9 billion per year (\$6 billion in wages and \$3 billion in worker benefits). Using only a 1% lost productivity value, Kats calculated that sick building syndrome reductions in State of California buildings were equivalent to a present value of \$37 - \$55 per square foot of building space over 20 years, compared to average energy costs of \$1.47 per square foot for state buildings. The avoidable cost estimate of \$9 billion for California is much larger than the estimate derived from Fisk because it includes employee benefits to reflect the workers' full market value, and it reflects the higher percentage of office workers and the higher salaries in California compared to the U.S.

For estimating the costs of sick building syndrome in California's school and office buildings, we use the more comprehensive and California-specific approach of Kats. We modified his estimate by using more recent data and conservative estimates of worker benefits costs, as follows. California has 7 million workers in offices or schools, and their average salary is \$43,000 per year in 1998 dollars (Kats, 2003). Sick building syndrome symptoms are again conservatively estimated to cause a 2% decrease in worker productivity. Multiplying these values together yields a cost of about \$6 billion per year. For U.S. white collar workers in 1999, salary accounted for 72.8% of their total compensation on average, while benefits such as health insurance and retirement accounted for 27.2% (USCB 2002). Dividing this benefit percentage by the salary percentage yields a ratio of 0.37 as the additional fraction for compensation as benefits. Multiplying 0.37 by the salary estimate of \$6 billion yields an estimated overhead cost of \$2.2 billion. This is a conservative estimate because it does not include other overhead costs to the employer such as training, equipment, and travel. Combining these salary and benefit cost estimates yields a total cost estimate of \$8.2 billion, or \$8.5 billion in 2000 dollars, due to the avoidable impacts of sick building syndrome on worker productivity in California.

3.4 SUMMARY OF ESTIMATED COSTS

The combined cost of both fatal and non-fatal impacts due to indoor air pollution in California homes, schools, and non-industrial workplaces is substantial: it is estimated at **\$35 billion per year**, as shown in Table 3.6 below. The annual valuation of mortality

Table 3.6. Summary of Estimated Costs of Indoor Air Pollution in California

Health End Point	Health Valuation: Mortality ^{1,2} (\$ Billions/yr)	Medical Cost ^{1,2} (\$ Billions/yr)	Lost Productivity Cost ^{1,2} (\$ Billions/yr)	Total Cost ¹ (\$ Billions/yr)
CO: poisoning	0.15	< 0.01	NA	0.15
VOCs: cancer	0.73	0.011	NA	0.74
ETS: lung cancer ³	2.4	0.025	NA	2.4
ETS: heart disease	23	0.055	NA	23
ETS: respiratory disease ^{3,4}	NA	0.076	NA	0.076
ETS: low birth weight	NA	0.19	NA	0.19
Mold and moisture: asthma and allergies	0.031	0.19	NA	0.22
Sick building syndrome	NA	NA	8.5	8.5
TOTAL⁵	26	0.6	8.5	35

1. Estimates are based on average or mid-point of incidence rates of mortality and morbidity from previous tables, and estimates of productivity discussed in the text. Values are rounded to two significant figures.
2. Original data were adjusted to year 2000 dollars and year 2000 population, except where noted otherwise in previous tables.
3. These estimates are probably overestimates because they are based on 1996 U.S. smoking prevalence, which does not account for major reductions in smoking prevalence among Californians and the virtual elimination of workplace smoking in California by 2000. OEHHA is currently developing new risk estimates.
4. Combined total of asthma induction and exacerbation, bronchitis, and pneumonia.
5. Totals are rounded to 2 significant figures. These totals are likely low because conservative cost estimates were used, and quantitative information is not readily available for many known impacts of indoor air pollution, such as for indoor PM and many indirect costs of health effects. The actual impact on the California economy may be several times this total amount.

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attributable to indoor air pollution is estimated to total about \$26 billion, with most stemming from ETS. The actual total valuation of mortality is likely to be even higher because these estimates do not include other pollutants that can increase the risk of premature death. Examples of these other pollutants include: PM from wood smoke; other carcinogens emitted from materials and products; and radon. The medical costs (direct and some indirect) due to indoor air pollution total more than \$0.6 billion per year, with a large portion of the costs attributable to mold and other moisture-related allergens. Again, this is certainly a very low estimate: the cost estimates for morbidity do not include of the potential losses due to other indoor allergens, CO poisoning's long-term effects, reduced student performance, lost earnings opportunity, unpaid caregivers, and human suffering. Finally, the cost of reduced worker productivity due to indoor air pollution (sick building syndrome) that could be prevented is estimated to be \$8.5 billion per year. As discussed in the next section, case studies have documented that measures to reduce indoor air pollution in homes and schools can have immediate and cost-effective benefits on human health and student performance.

3.5 BENEFITS OF IMPROVING INDOOR AIR QUALITY

Several case studies and demonstration programs have documented the economic, productivity, and health benefits that can accrue by improving IAQ. The following examples and other available examples provide empirical data on the benefits achieved in homes:

The Seattle Healthy Home Program has found that home visits to low-income households by a trained community health worker can reduce asthma symptom significantly and cost-effectively (Takaro *et al.*, 2004). This program reduced medical costs for asthma treatment significantly, in both the low-intensity version (one visit over the year) and the high-intensity version (multiple visits over one year). The observed marginal cost savings were \$1,316 to \$1,849 per patient over four years. These savings do not reflect the reduced number of emergency room visits and the reduced risk of asthma-related deaths.

The "Healthy Neighborhoods Program" of New York State inspected over 45,000 homes for health and safety problems, and intervened where necessary (HUD, 1999; NYS, 1999). Interventions addressed lead-based paint hazards, fire safety, sources and detection of carbon monoxide, and asthma-related conditions. Interventions were generally simple and low cost, such as providing working smoke detectors and batteries, CO detectors, and furnace filters. The program results showed that the visits to each home cost \$132 per unit. However, the benefits for lead poisoning prevention, asthma reduction, and burn prevention alone were worth at least \$285 per unit, excluding estimated benefits associated with reduced injury, carbon monoxide poisoning and fire.

The following case studies in schools also illustrate the benefits of improving indoor air quality:

Office visits for asthma inhaler use dropped by 50% at two elementary schools in San Francisco Unified School District (USD) after the *IAQ Tools For Schools* program was pilot tested (U.S. EPA, 2000a). In addition, fewer asthma episodes occurred, and fewer students brought asthma medications or inhalers to school.

A study of Chicago and Washington, DC schools found that better school facilities can add three to four percentage points to a school's standardized test scores, even after controlling for demographic factors (Schneider, 2002). This and other studies reviewed by Kats (2003) and

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Fisk (2000) confirm a widely held, common sense perception that the physical quality of the classroom environment greatly affects how well children learn.

An analysis of two school districts in Illinois, one small and one large, found that student attendance improved by 5% after incorporating cost effective indoor air quality improvements – regardless of school district size (Healthy Schools Campaign, 2003).

Clovis USD near Fresno, and Everett USD near Seattle, have used the *IAQ Tools for Schools* program and a rapid complaint response approach using a portable indoor air monitoring kit. Additional staff were not needed to implement the program. This program has quickly paid for equipment by reducing the number of complaints by up to 60%, and by reducing the cost for environmental consultants (Poytress, 2003; Jefferis, 2004). It has also greatly improved the credibility of the school maintenance program with school staff and parents of students.

Indoor air quality is one of several important indoor environmental factors affecting student performance. In a study of over 8,000 students in third through sixth grade in Fresno, Heschong (2003) investigated the effect of daylight and other indoor environmental quality effects on student test scores over an academic year. As expected, commonly recognized factors such as teacher characteristics, number of computers, or attendance rates were found to be significant in predicting student performance. However, physical characteristics such as indoor air quality, ventilation, acoustics, and especially daylighting were found to be equally significant, if not more significant.

4. EXISTING REGULATIONS, GUIDELINES, AND PRACTICES

Despite the ubiquitous presence of toxic pollutants in the indoor environment, there are no government standards for emissions from common sources of indoor pollutants, and no air quality standards for residences, schools, or public buildings. However, a variety of government agencies and private organizations have established standards and guidelines that can be applied to indoor environments to assist in the assessment and control of health hazards from air pollutants. Foremost among these are workplace standards; however, those standards are designed for 8-hour exposures of healthy adults, and are generally not protective of the more sensitive subgroups of the population. Others, such as ambient air quality standards, Proposition 65, and AB 13, which prohibits cigarette smoking in workplaces, are applicable to indoor air quality only in a limited way, or are only advisory, and do not prevent emissions and exposures.

4.1 STANDARDS AND REGULATIONS

4.1.1 Workplace Standards and Regulations

The California Occupational Safety and Health Program (Cal/OSHA) in the Department of Industrial Relations (DIR) has jurisdiction over most private and public employers and employees in California, with the exception of U.S. government employees. Cal/OSHA has regulatory authority to develop, promulgate, and enforce air pollutant exposure limits, ventilation regulations, and other standards for the workplace that directly impact indoor air quality. The California Occupational Safety and Health Standards Board is the unit within the Cal/OSHA program with authority to adopt standards and regulations to protect workers. Labor Code section 144.6 requires the Standards Board to adopt standards that “most adequately assures, to the extent feasible, that no employee will suffer material impairment of health or functional capacity even if such employee has regular exposure to a hazard regulated by such standard for the period of his working life.”

Under CCR Title 8, Section 5155, the Standards Board sets permissible exposure limits (PELs) and other limits for airborne contaminants. The PELs are generally protective of the health of most workers, though they may not be sufficient to prevent health impacts for all individuals, or to prevent exacerbations of asthma or other reactions by sensitive individuals. The Cal/OSHA PELs are 8-hour exposure limits designed to protect healthy working adults, and may be based in part on technological and economic feasibility considerations (non-health related criteria). These standards are not established with the intent to protect infants, the elderly, or individuals with pre-existing heart or respiratory disease, all of whom are generally more susceptible to the effects of pollutants, nor are they intended to be protective for exposures greater than eight hours per day, five days a week. Generally, lower exposure limits would be necessary to protect the general population.

Cal/OSHA also enforces a number of requirements for workplace ventilation and sanitation in Title 8 that directly impact indoor air quality. Section 5142 requires employers to maintain and operate ventilation systems to provide at least the quantity of outdoor air required by the State Building Code (Title 24) at the time the building permit was issued. This section also requires that systems be operated continuously during work hours, except during times of repair or emergency, and that annual inspections be conducted and maintenance records kept. Section

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5143 has additional requirements related to local exhaust ventilation, replacement of filters, and provision of clean make-up (supply) air.

Section 3362 requires that workplaces be maintained in a sanitary condition, and subsection (g) requires employers to correct all types of water intrusion or leakage, to reduce the potential for mold growth. Other subsections address insects, vermin, garbage and waste handling, and other aspects of a sanitary environment. Section 3203, Illness and Injury Prevention Program, requires employers to have written plans for hazard identification, communication, prevention, and training. Other Cal/OSHA requirements reduce or prevent employee exposures to asbestos (Sections 5208, 1529), lead (Section 1532.1), and environmental tobacco smoke (Labor Code 6404.5).

4.1.2 Ambient Air Quality Standards

National and state ambient air quality standards (AAQS), established by the U.S. EPA and the ARB, respectively, are developed to protect the general public from the harmful effects of “traditional pollutants” in outdoor air, for specified averaging times (exposure times). California’s AAQS are often more protective than the national AAQS. Currently, the state AAQS are under review to ensure that they are protective of sensitive populations, especially infants and children (ARB/OEHHA, 2000). In the absence of indoor air quality standards or guidelines, the AAQS serve as useful guidelines for indoor air quality, because they are based on specified averaging times and incorporate a margin of safety. Outdoor standards for PM are often exceeded in indoor environments, and standards for CO, NO₂ and ozone are sometimes exceeded. National and state AAQS are available at <http://www.arb.ca.gov/research/aaqs/aaqs.htm>.

4.1.3 Proposition 65

In 1986, California voters approved Proposition 65, the Safe Drinking Water and Toxic Enforcement Act of 1986, an initiative to address concerns about exposure to toxic chemicals. Proposition 65 requires the State to publish a list of chemicals known to cause cancer, birth defects, or other reproductive harm. The list includes approximately 750 chemicals, many of which are additives or ingredients in pesticides, common household products, food, drugs, dyes, solvents, building materials, and other sources found indoors. Businesses are required to provide a “clear and reasonable” warning when their products or actions may result in a release of chemicals above a specified threshold level, so that members of the public are aware they may be exposed to harmful chemicals. Warnings have evolved to include labeling of consumer products, posting signs at the workplace or on new housing, and publishing notices in a newspaper. OEHHA develops numerical guidance levels, known as “safe harbor” levels, for determining whether a warning is necessary. For potential carcinogens, the Proposition 65 “no significant risk level” is one excess case of cancer per 100,000 individuals exposed over a 70-year lifetime; for reproductive toxicants, the “no significant risk level” is one-thousandth of the no observable effect level (NOEL). Proposition 65 pollutants and safe harbor levels are available on the OEHHA website at <http://oehha.ca.gov/prop65.html>.

4.1.4 Reference Exposure Levels for Air Toxics

The Air Toxics “Hot Spots” Information and Assessment Act of 1987 (Assembly Bill 2588 as codified in Health and Safety Code Section 44300 *et seq*) requires stationary sources of air pollutants (e.g., industrial plants) to report the types and quantities of substances their facilities routinely release into the air, ascertain health risks associated with the release, and notify

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nearby residents of significant risks. An amendment to the statute in 1992, SB 1731 (Calderon), requires OEHHA to prepare and adopt risk assessment guidelines, and requires facilities with a significant risk to prepare and implement risk reduction plans. Under this act, OEHHA develops acute and chronic reference exposure levels (RELs) as guidelines to prevent harm from toxic air pollution. <http://www.arb.ca.gov/toxics/toxics.htm>.

RELs are used by Cal/EPA agencies as indicators of potential adverse health effects other than cancer. RELs are generally based on the most sensitive adverse health effect reported in the medical and toxicological literature. They are designed to protect the most sensitive subgroups of the population by the inclusion of margins of safety. Because uncertainty factors are incorporated into the REL, an air concentration greater than the REL does not necessarily mean that the exposed public will suffer adverse health impacts. The methodology and studies used to develop the REL health standards are detailed in two documents available on the OEHHA website (www.OEHHA.ca.gov): the *Air Toxics Hot Spots Program Risk Assessment Guidelines, Part 1: The Determination of Acute Reference Exposure Levels for Airborne Toxicants*, March 1999 and the *Air Toxics Hot Spots Program Risk Assessment Guidelines, Part III: The Determination of Chronic Reference Exposure Levels for Airborne Toxicants*, February 2003.

OEHHA has established chronic RELs for approximately 71 air pollutants. Chronic RELs are indicators of healthy versus potentially unsafe levels resulting from long-term exposure, exposure lasting at least 8% of a lifetime or 12 years and possibly longer. Chronic RELs are intended to indicate levels that will not cause adverse health effects in individuals with high susceptibility for chemical injury as well as identifiable sensitive subpopulations (high-risk individuals). However, chronic RELs may not necessarily be indicative of safe levels in hypersensitive individuals.

Acute RELs are indicators of healthy versus potentially unsafe levels of exposure for short periods of time, such as one hour. Because exposure is for a short time period, acute RELs are always higher than chronic RELs. OEHHA has established acute RELs for 51 chemicals (OEHHA, 2000a). OEHHA recommends that acute RELs be used to evaluate exposures that occur no more frequently than every two weeks in a given year.

In addition to providing an indication of healthful versus potentially unsafe levels outdoors near stationary sources, the acute and chronic RELs are also being used to identify healthful versus unhealthy pollutant levels in indoor air. Formaldehyde is a very common indoor air contaminant that has both cancer and non-cancer health effects. Because of the frequent use of RELs for indoor application, OEHHA developed an additional REL for formaldehyde, an interim REL (IREL) based on an 8-hour exposure period. The IREL established for formaldehyde is 27 ppb. The IREL identifies the level below which irritant effects such as eye, nose, and throat irritation would not be expected to occur during typical daytime (8-hour) occupancy of buildings. Other 8-hour IRELs are not yet available. Chronic RELs have been used in developing building material emission limits for use by DGS in specifying requirements for new state buildings and furnishings. These are known as Section 01350 requirements, and are discussed further below.

4.1.5 Tobacco Control

Cigarette smoking is a major source of indoor pollution that is now prohibited in most public buildings in California. Passage of a statewide smoke-free workplace law in 1995 (Labor Code 6404.5) led to a reduction in smoking by the California population and eliminated smoking at nearly all California indoor workplaces, including restaurants, bars and gaming clubs. This statewide prohibition is primarily enforced at the local level. Cal/OSHA responds to complaints

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of workplace cigarette smoke only after the employer has been found guilty at the local level three times in the previous year. Prior to passage of the statewide law, numerous city and county ordinances had been implemented to restrict cigarette smoking.

The smoke-free workplace law contains fourteen exceptions to the smoking ban, each with additional explanations and stipulations. Exclusions include some hotel/motel guest rooms, hotel/motel lobbies, private meeting rooms, tobacco shops, truck cabs, warehouse facilities, gaming and bingo clubs, bars and taverns until January, 1997, theatrical productions, medical research sites, private residences, patient smoking areas, employee breakrooms, and small businesses. Despite the exclusions, the ban has been very successful in reducing worker exposure to cigarette smoke. In 1999, 93.4% of California's indoor workers reported working in a smoke-free environment, compared to only 45% in 1990 (Gilpin *et al.*, 2001).

The workplace prohibition of smoking has had far reaching benefits. The percentage of Californians with children under the age of 18 who do not allow smoking in the household has increased substantially. In 1994, 63.0% of Californians with children did not allow smoking in the house. By 2001, 77.9% did not allow it (Gilpin *et al.*, 2001). Fewer Californians are smoking as a result of this legislation and the DHS Tobacco Control Program. Smoking rates among California adults have declined from 26% in 1984 to 17% in 2001 (BRFSS, 2001). Californians who still smoke are smoking fewer cigarettes than they did in the past (Gilpin *et al.*, 2001).

4.1.6 Radon In Drinking Water

Radon levels in typical groundwater concentrations pose higher risks than those posed by the other drinking water contaminants that have been subjected to regulation (e.g., disinfection by-products). U.S. EPA was directed under the Safe Drinking Act (as amended in 1996) to attempt to regulate radon in drinking water. Though it has no authority to regulate indoor air radon, which is dominated by the soil gas infiltration, the U.S. EPA recognized that it would be far more cost effective to mitigate indoor air radon. The Safe Drinking Act gave U.S. EPA the latitude to allow higher concentrations of radon in drinking water if efforts were established to reduced indoor air radon risk *equal to or greater than* that would be achieved by reducing the concentration of radon in drinking water (U.S. EPA, 1996).

The U.S. EPA is in the process of promulgating a maximum contaminant level (MCL) for radon, which is the allowable concentration based only on the contaminant risk in drinking water (U.S. EPA, 2000e). The proposed MCL is 300 pCi/L. An alternative MCL, 4000 pCi/L could also be used as a target concentration. This is the concentration of radon in water "necessary to reduce the contribution of radon in indoor air from drinking water to a concentration that is equivalent to the national average concentration of radon in outdoor air." The AMCL would apply for states that had *multimedia radon mitigation programs*, and would allow utilities not to treat radon concentrations between the MCL and AMCL. Multimedia programs would aim to reduce indoor air radon risk, using a combination of approaches allowed in the regulations: public education; testing; training; technical assistance; remediation grants, loan or incentive programs; or other regulatory or non-regulatory measures.

4.2 GUIDELINES

Various governmental and private organizations issue guidelines to promote healthful environments. Guidelines cannot be enforced; compliance with the recommended levels is voluntary. Guidelines mentioned here are based on health endpoints and when followed, are

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protective of different segments of the population, depending on the establishing agency and purpose of the guideline.

4.2.1 ARB Indoor Air Quality Guidelines

The ARB has published three indoor air quality guidelines – for formaldehyde, combustion pollutants, and chlorinated hydrocarbons. ARB's indoor air quality guidelines discuss the

Table 4.1. ARB Indoor Air Quality Guidelines

Pollutant	Measurement Period	Outdoor Air Quality Standards		ARB Recommended Maximum Indoor Levels
		ARB	U. S. EPA	
Formaldehyde (HCHO)	24-hour	---	---	Lowest level feasible to reduce cancer
	8-hour	---	---	< 27 ppb to avoid acute irritant effects
Carbon Monoxide (CO)	8 hours	9 ppm	9 ppm	9 ppm
	1 hour	20 ppm	35 ppm	20 ppm
Nitrogen Dioxide (NO ₂)	24 hours	---	---	0.08 ppm avoid repeated high exposures
	1 hour	0.25 ppm	---	0.25 ppm
Particles – (PM ₁₀)*	24 hours	50 µg/m ³	150 µg/m ³	50 µg/m ³
Particles – (PM _{2.5})*	24 hours	---	65 µg/m ³	65 µg/m ³
Polycyclic Aromatic Hydrocarbons (PAHs)	---	---	---	Lowest levels feasible to avoid cancer risk. Avoid or minimize exposure.
Chlorinated Hydrocarbons, e.g., - chloroform - trichloroethylene - <i>p</i> -dichlorobenzene - methylene chloride - perchloroethylene - methyl chloroform	---	---	---	Lowest levels feasible to avoid cancer risk. Avoid or minimize personal exposures.

* ARB's indoor PM guidelines may be updated soon to be consistent with the revisions to the outdoor air quality standards.

indoor sources and potential health effects of various pollutants, and provide information to the public on ways to limit or reduce their exposure to those pollutants. The guideline levels were developed in consultation with DHS and OEHHA, and are set to protect sensitive subgroups of the population. Because a number of the chemicals covered in the guidelines may cause cancer or other significant health problems, and no absolutely safe levels have been identified,

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recommendations are made to take action to prevent emissions of those chemicals in the home and to reduce exposure to the greatest extent feasible (ARB, 1991; ARB, 1994b; ARB, 2001b; ARB, 2004). Guidelines for traditional pollutants are based on the ambient air quality standards for California, because the basis of those standards is the health impacts seen at different air concentrations for specified exposure durations, which would be the same regardless of whether the individual is standing indoors or outdoors.

4.2.2 American Society of Heating, Refrigerating, And Air Conditioning Engineers

The American Society of Heating, Refrigerating, and Air Conditioning Engineers (ASHRAE) provides professional guidance on minimum building ventilation rates for human health and comfort. While not regulatory, ASHRAE Standards, especially Standard 62, *Ventilation for Acceptable Indoor Air Quality* (ASHRAE, 2003b), and Standard 55-2004 *Thermal Environmental Conditions for Human Occupancy* (ASHRAE, 2004) are important references for California's ventilation codes and recommended comfort levels. ASHRAE and other groups have also developed guidelines using indoor carbon dioxide concentrations as an indoor air quality indicator, ranging from about 800 to 1200 parts per million (ppm) as a "not to exceed" level.

4.2.3 Guidelines For Europe

"Air Quality Guidelines for Europe" were developed by the World Health Organization's (WHO) Regional Office for Europe, a specialized agency of the United Nations. These guidelines were first published in 1987, then revised in 1997. The guideline value indicates a concentration of air pollutant below which no adverse effect to human health is expected, based on consideration of both carcinogenic and other health effects. They are intended to be applicable to both indoor and outdoor air exposures. Guideline values have been established for 38 non-carcinogenic compounds and some carcinogens for various exposure periods ranging from 15 minutes to one year. They address public health, not occupational health, and so are intended to be protective of the entire population (<http://www.who.int/inf-fs/fact187.html>).

4.2.4 DHS Mold Guidelines

DHS has developed several fact sheets that provide guidance to the public on mold assessment and remediation. In 2001, the Toxic Mold Protection Act (HSC Section 26100) was approved. The Act requires DHS to determine the feasibility of adopting permissible exposure limits for indoor molds and develop new standards or guidelines to

- Assess the health threat posed by the presence of indoor molds,
- Determine valid methods for fungal sampling and identification, provide practical guidance for mold removal and abatement of water intrusion,
- Disclose the presence of mold growth in real property at rental or sale, and
- Assess the need for standards for mold assessment and remediation professionals.

However, the implementation of this statute depends on the provision of funding which has not occurred. Thus, the statute has not been implemented.

4.2.5 U.S. EPA's *IAQ Tools For Schools Program*

Although it does not provide indoor pollutant guideline levels, U.S. EPA's *IAQ Tools for Schools Program* (<http://www.epa.gov/iaq/schools/>) provides schools with information they need to understand IAQ issues, prevent IAQ problems, and assure healthful indoor air quality. The

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program uses a team approach to school IEQ management and emphasizes staff and occupant training, communication, and improved routine operation and a district.

In California, U.S. EPA has trained more than 2000 individuals from districts throughout the state. Despite the outreach, awareness and use of the *IAQ Tools for Schools* program among California schools is still relatively low: about 11% of school districts use all or part of the program (ARB/DHS, 2003). U.S. EPA is partnering with organizations of school officials (e.g., ACSA, the Association of California School Administrators, and CASBO, the California Association of School Business Officers) to promote increased use of this program.

California Assembly Concurrent Resolution No. 75 (Chan), enrolled in June 2003, recognizes the significance of school indoor environments to the childhood asthma problem, and “encourages California school districts to implement the *Indoor Air Quality Tools for Schools Program* for the benefit of asthmatic children and for the health, well-being, learning, and productivity of the entire school population”.

4.3 EMISSION LIMITS

Prevention or reduction of emissions from pollutant sources is the preferred control technique for most pollutants. This section will discuss known regulations, guidelines, and practices for limiting emissions from consumer products, appliances, and building materials. Since no single federal or state agency has comprehensive regulatory authority over indoor air quality, regulations are limited to a few market sectors, and the driving force for the regulation may not be improved indoor air quality. Only a few indoor pollutant sources are regulated for reducing indoor emissions.

4.3.1 Consumer Products

4.3.1.1 Government Regulations For Consumer Products

Consumer Product Safety Commission (CPSC)

The federal Consumer Product Safety Commission (CPSC) has broad jurisdiction over consumer products, which are defined as any articles which are produced or distributed for sale to, or use by, consumers in or around the home or in schools, recreation areas, or other non-occupational settings. Certain products are exempted from CPSC authority, including pesticides, cosmetics, tobacco and cigarettes, food, drugs, automobiles, airplanes and firearms. In addition, houses and other buildings are specifically excluded by legislative history and judicial review. However, CPSC does have jurisdiction over home building materials that are sold or distributed as separate products to consumers.

CPSC has the authority under certain conditions to: (1) ban a product; (2) establish mandatory safety standards for products; (3) recall products for repair, replacement or refund; (4) mandate warning labels for products; and (5) cooperate with manufacturers in the development of voluntary product standards. CPSC has made a few notable contributions in areas related to IAQ regulation and research. Indoor pollutants regulated to some extent by CPSC through product bans or labeling requirements include asbestos, vinyl chloride, and combustion pollutants from appliances. The CPSC has recently begun to assess the health risks associated with chlorocarbons such as methylene chloride found in various consumer products; airborne biological contaminants that may be associated with humidifiers and vaporizers, and exposure to lead by paint removal tools.

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Voluntary product standards, rather than product bans and mandatory standards, are generally preferred by CPSC for several reasons. Voluntary standards take less time to develop and implement and are less likely to be challenged legally, so that public protection should occur sooner; they tend to be less intrusive, more responsive to technological change, and cheaper for all concerned.

The CPSC has focused on regulations for safety more than for health. For example, a policy on methylene chloride led to labeling regulations rather than a ban on its use in household products (CPSC, 1987a). The mandate of the California Department of Consumer (CD) affairs differs from that of the CPSC in that the DCA does not regulate consumer products directly, but rather regulates services provided to consumers and those who provide them, and addresses consumer complaints.

ARB Consumer Products Program

The focus of ARB's Consumer Products and Architectural Coating Programs is to reduce the amount of reactive volatile organic compounds (VOCs) that are emitted from the use of consumer products. The driving force behind ARB's Consumer Product Regulations is not reducing exposure to VOCs in indoor air, but limiting the release of VOCs that have the potential to react with other pollutants under sunlight to form ground-level ozone and particulate matter (PM10), the main ingredients in smog. "Consumer product" means a chemically formulated product used by household and institutional consumers, including, but not limited to, detergents; cleaning compounds; polishes; floor finishes; cosmetics; personal care products; home, lawn, and garden products; disinfectants; sanitizers; aerosol paints; and automotive specialty products; but does not include other paint products, furniture coatings, or architectural coatings. The VOC compositions of these latter products are regulated by the local air pollution control and air quality management districts (districts), although the ARB provides considerable technical assistance to the districts and develops Architectural Coatings Suggested Control Measures (i.e., model rules) for adoption by the districts.

4.3.1.2 Government Guidelines For Consumer Products

A statewide sustainable building program can promote sustainable building concepts and challenge manufacturers to produce more environmentally friendly products. Recognizing this opportunity, Governor Davis issued Executive order D-16-00 in August 2000. This Executive Order directed the Secretary of the State and Consumer Services agency to develop a strategy for how the State could design, build, operate, and maintain buildings that are models of efficiency while providing healthy indoor environments. As part of that roadmap, the Department of General Services (DGS) directed the development of specifications for environmentally preferable janitorial products for the cleaning and maintenance of state-owned buildings. Green Seal, a product certification organization, has recently developed a standard for cleaning materials (GS-37) that relies primarily on chemical content. The specification reduces emissions to the general work area as well as reduces exposures to the individuals using the products. The specification is currently under revision to provide greater health protection. These specifications can result in reduced exposure to cleaning chemicals; however, the use of these products in state-owned buildings is voluntary.

4.3.2 Appliances

Indoor appliances that burn fuel (combustion appliances) or heat food but lack a venting system to remove combustion by-products, moisture, and odors, such as kerosene heaters, unvented

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gas logs, and unvented gas stoves, can be a major source of indoor air pollution. Wood stoves and fireplaces that burn gas or wood are always vented by design, but they can also be a source of indoor air pollution due to leakage, re-entrainment of air pollutant emissions, or blockage of the flue. Building standards also require gas furnaces and gas water heaters to be vented, but the emissions of these appliances can sometimes leak or be drawn into a building due to blocked or malfunctioning vent systems.

4.3.2.1 Government Regulations For Appliances

The design and installation of combustion appliances in California is usually regulated through building standards that are enforced by local building officials, and through local air quality regulations that are implemented by local air pollution control districts. Therefore, compliance inspections for combustion appliances in homes occurs only during home construction, remodeling, and resale, unless inspections are made in response to complaints.

Unvented Appliances

Since the 1980's, state regulations have banned the offering for sale of unvented combustion heaters designed for residential use (HSC Sec. 19881[a]). These appliances are still marketed for other uses such as workshops and barns, and are now marketed by out-of-state sellers via mail order and the Internet. It is not clear how effective this regulation has been in preventing the residential use of unvented kerosene heaters, since one to three percent of California households use them as primary or secondary heat sources (Phillips *et al.*, 1990).

These regulations were amended in 1997 to allow the sale and indoor use of "natural gas fueled unvented decorative gas logs and fireplaces," provided a series of steps were completed (SB 798, Haynes, HSC 19881[b-c]). The first step, the development of recommended standards by DHS and the Department of Housing and Community Development, was begun by funding a contractor study of the indoor air quality and health impacts of unvented gas logs. The contractor (Traynor, 1999) reviewed the scientific literature and other pertinent information on the topic, conducted limited modeling of indoor air quality impacts, and recommended 21 different options for California standards. Indoor NO₂ emissions and perhaps water vapor emission were found to be the most restrictive limitations for avoiding indoor air quality impacts.

The contractor recommendations for unvented gas log standards included restrictions on emission rates for NO₂ and CO with adjustments for different house volumes. DHS recommended not setting a California standard for these appliances, and the manufacturers of unvented gas logs agreed with this approach. The sale of these devices is still banned under current regulations.

Despite the potential of unvented cooking appliances to trigger serious health effects such as CO poisoning or asthma attacks, indoor emissions of appliances are generally not regulated by any government agency. California regulations do not require cook stoves to have exhaust ventilation or limit the flow rate of the range hood. The State Building Standard only requires that the range hood supply the ventilation that is required by the stove manufacturer.

Some local municipalities in the U.S. and California may now be requiring range hoods to exhaust cooking emissions to the outside (Repair Clinic, 2004). In California, local building Departments may adopt different building standards for the design and installation of appliances, as long as the local standards are at least equivalent to the state building standard code in performance, safety, and the protection of life and health (HSC 17951[d]2). The City of

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Hayward (2002) requires range hoods exhausted to the outside for their housing rehabilitation program.

Vented Appliances

Most of the 35 California Air Districts have current rules governing outdoor emissions from residential appliances (ARB, 2003). Most of these rules require certification that natural gas-fired water heaters sold or installed in the district must meet the 2002 emission limitations for NO_x in South Coast Air Quality Management District's Rule 1121 – NO_x emission no more than 40 ng/J (SCAQMD, 1999). However, the SCAQMD emission limits for gas water heaters in homes will be lowered in 2005 to 10 ng/J of NO_x (mobile home standards will not change). This reduction in NO_x limits reduces the potential magnitude of indoor pollutant exposure when water heater emissions enter a home due to backdrafting.

Wood burning devices have come under increasing control by local air pollution control districts for their outdoor emissions. For example, in the Northern Sonoma County and San Luis Obispo County Air Pollution Control Districts, PM emissions from non-catalytic and catalytic wood fired appliances must be less than or equal to 7.5 g/hr and 4.1 g/hr, respectively. The districts also require U.S. EPA-Certified Phase II wood burning devices for new sales and installations (40 CFR, Part 60, Subpart AAA, Section 60.530 through 60.539b).

Recently the San Joaquin Valley Unified Air Pollution Control District adopted fairly stringent regulations for fireplaces and woodstoves. These regulations: 1) ban the installation of fireplaces in most new residential development, 2) require U.S. EPA Phase II wood stoves in any sale, supply, transfer, and installation, and 3) require the removal of older woodstoves, with the option to install U.S. EPA Phase II or pellet-burning wood stoves, in any real estate sale or transfer (SJVUAPCD, 2003). The district can also curtail wood burning in the region when outdoor PM levels are predicted to exceed an Air Quality Index level of 150.

California's energy efficiency standards for new home construction require vented fireplaces, decorative gas appliances, and gas logs to be installed with closable doors over the opening, a combustion air intake for outside air, and a flue damper with a readily accessible control (CCR, Title 24, Pt. 6, Sec. 150[e] http://www.energy.ca.gov/title24/residential_manual/res_manual_chapter2.PDF). In addition, gas-fired logs, lighters, or decorative appliances must have a flue damper that is permanently blocked open as required by the California Mechanical Code or the manufacturer.

4.3.2.2 Government Guidelines For Appliances

ARB's (1994b) Indoor Air Quality Guideline, "Combustion Pollutants in Your Home", recommends actions to reduce combustion pollutants in homes, including emissions from combustion appliances and motor vehicles. These actions include removing unvented combustion sources from the home, using and maintaining vented combustion appliances properly, and ensuring adequate and balanced ventilation for the whole house. Also recommended are annual inspections and adjustments to combustion appliances, installation of CO alarms, and testing of combustion appliance safety. This guideline has been widely distributed in California and other states, especially through low-income weatherization groups, of which many are community-based organizations.

ARB (1997b) has also published the "Woodburning Handbook", which recommends measures for reducing emissions and exposures from residential woodburning. These measures include using a gas-fired furnace or gas fireplace insert instead, increasing the efficiency of the house

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and wood stove, and improving the operation and maintenance of the wood stove. This handbook has been widely distributed in California since 1995, mainly through local air pollution control districts. There is no evidence available to show how effective this guideline has been in reducing the health impacts of woodburning.

The Combustion Appliance Safety (CAS) test protocol is used by the low-income weatherization program in California (DCSD, 2003) and many other states. These programs include appliance cleaning and adjustment, insulation; and sealing. This protocol was adopted to improve the health and safety of program participants, and to avoid unnecessary liability. It requires that home appliances and venting systems be inspected and tested before and after weatherization is completed. The protocol specifications include the following:

- CO levels in the flue of the furnace and water heater cannot exceed 100 ppm, 100 ppm above the cooktop, or 225 ppm above the oven vent for both the oven and broiler burner. Cleaning, adjustment, and repair or replacement are required to bring the appliances within these limits.
- Clothes dryers must be vented outside with metal ductwork, and CO must not exceed 100 ppm in the exhaust vent.
- The flue draft of furnaces and water heaters appliances must meet certain minimums, and house depressurization cannot exceed limits that can backdraft vented combustion appliances.
- Indoor CO cannot exceed 9 ppm in the room.
- An operable exhaust fan is required in the cooking area.

4.3.2.3 *Professional and Industry Guidelines and Practices For Appliances*

The CAS test protocol discussed above is also used by many high performance home builders and remodelers, such as those in the Energy Efficient Building Association (EEBA, 2003) and the California Building Performance Contractor Association (CBPCA, 2003a,b). Some, but not all, California utilities are using the CAS test protocol (Hosler, 1998). The California Public Utilities Commission (CPUC, 1998, 2003) is currently developing standard testing protocols of natural-gas appliances for investor-owned utilities in California.

The home builder groups above, along with the U.S. Department of Energy's Build America program for high performance homes (BSC, 2003), recommend additional measures to avoid combustion safety and IAQ problems. These measures include:

- Provide exhaust ventilation for gas cooking appliances.
- Use sealed combustion or power-vented gas appliances that are immune to backdrafting.
- Test to ensure house depressurization is not excessive.

However, according to the American Gas Association (AGA, undated): "No industry standards or uniform governmental requirements currently exist for evaluating acceptability of range emissions." Manufacturers currently test gas stoves, cooktops, and other gas appliances using the industry-promulgated test standards and CO emission limits of the American National Standards Institute (ANSI). A test method for NO₂ emissions is under ANSI development (Traynor, 1999). The ANSI standards for gas stoves would allow excessive levels of indoor air pollution to build up in homes when gas stoves are used for extended periods without adequate venting to the outside.

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4.3.3 Building Materials

Conventional building materials such as vinyl flooring, carpet, paint, cabinets, and composite wood products can be strong indoor sources of VOCs with a major impact on human health. In the absence of clear regulatory authority, governmental purchasing guidelines are being pursued to recommend low-emitting products in new construction. Also, certain segments of industry have initiated voluntary programs to reduce product emissions, and label their products accordingly.

4.3.3.1 Government Regulations for Building Materials

HUD Formaldehyde Emissions in Mobile Homes

The U.S. Department of Housing and Urban Development (HUD) has set limits for formaldehyde emissions from plywood and particleboard used in mobile homes. HUD has several mandated responsibilities that directly involve indoor air quality, although its basic mission is to provide adequate housing, promote community and economic development of urban areas, and eliminate discrimination in housing markets. In 1984, HUD established formaldehyde emission standards for plywood and particleboard used in mobile homes: test chamber concentrations are not to exceed 0.2 ppm and 0.3 ppm, respectively, to maintain indoor air concentrations of formaldehyde in mobile homes below 0.4 ppm [24 CFR 3280.309(a)]. CFR Title 24 Section 3280.309 also states that each new manufactured home shall have a health notice on formaldehyde emissions prominently displayed in the kitchen. The HUD mobile home standards preempt any existing or future standards of state or local governments that apply to the same aspect of mobile home performance (Sec. 604 [d]).

ARB Composite Wood Control Measures

An air toxic control measure (ATCM) is under development by the ARB to reduce the release of formaldehyde from composite wood products such as particleboard, medium density fiberboard, hardwood plywood, and composite veneer. The adhesive binding systems used to manufacture these composite products contain urea-formaldehyde resins, which release formaldehyde to the air. As discussed earlier, this can be a strong source of formaldehyde emissions (Kelly *et al.*, 1999; Hodgson *et al.*, 2002). It is estimated that formaldehyde emissions from composite building materials could exceed 400 tons per year in California (ARB, 2001c). Staff from ARB are working with representatives in the composite wood industry to develop an ATCM that will reduce formaldehyde emissions from these sources.

4.3.3.2 Government Guidelines for Building Materials

DHS Non-Binding Guidelines

Reducing Occupant Exposure to Volatile Organic Compounds from Office Building Construction Materials: Non-Binding Guidelines was developed by DHS (1996), in response to California legislation (Ch. 1229, Statutes of 1990). The guidelines present a simple technical approach for evaluating, selecting, and installing building materials in order to minimize occupant exposures to VOCs emitted from the materials in newly constructed or remodeled office buildings. The guidelines recommend a five-step approach to reducing exposure to VOCs:

1. Evaluate and select low-VOC impact building materials and products.
2. Pre-condition certain materials to minimize VOC emissions after installation.
3. Install building materials and products based on their VOC emission decay rates.
4. Ventilate a building during and after installation of new materials and products.
5. Delay occupancy until VOC concentrations have been reduced adequately.

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The basic principles in these guidelines have been expanded more recently in manuals that describe “best practices” for assuring good indoor environmental quality.

Sustainable Buildings and Section 01350

Several state agencies collaborate on task forces to develop guidelines for “green” buildings, also called sustainable buildings. Components of sustainable buildings include materials for interior surfaces and furnishings that are protective of indoor air quality and health, cost-effective, durable, recyclable, and include some recycled content. Under the direction of the Secretary of the State and Consumer Services Agency (SCSA), interagency task forces have been established for Sustainable Buildings (Executive Order D-16-00) and Environmentally Preferable Purchasing (AB 498, Chan, 2002), to develop and implement guidelines for construction, renovation, and operation of state buildings, and procurement of state purchases. Stakeholders include members from 40 governmental agencies including ARB, DHS, OEHHA, Integrated Waste Management Board (IWMB), Department of General Services (DGS), and Division of State Architect (DSA).

In 1999, the Legislature directed DGS to incorporate sustainable building measures into the design and construction of a \$392 million State office building complex in Sacramento known as the Capitol Area East End Complex. A product of the effort is *Special Environmental Requirements* (Section 01350; State of California, 2002), which requires product emissions testing to protect human health. A test protocol was developed for laboratory testing of building materials by manufacturers. Under Section 01350, emissions from a single building material cannot exceed one half the chronic REL (developed by OEHHA) for toxic air contaminants (<http://ciwmb.ca.gov/GreenBuilding/Specs/Section01350/>). Formaldehyde is an exception; emissions from a single product cannot exceed one half the interim 8-hour REL for formaldehyde, as discussed elsewhere in this document. The RELs have undergone extensive public comments and peer review by the State’s Scientific Review Panel, so they are appropriate for use as an indoor guideline for this purpose.

Section 01350 has been incorporated into the DGS standard agreement for engineering and architectural services. It is also used in *Reference Specifications for Energy and Resource Efficiency* (CEC, 2004) and the *Collaborative for High Performance Schools (CHPS) Best Practices Manual* (CHPS, 2001). Section 01350 emission test requirements are also being considered for use on materials used in California’s new portable classrooms, building cleaning and maintenance products, carpets, and others.

Collaborative For High Performance Schools

The Collaborative for High Performance Schools (CHPS) is a California consortium of public agencies and energy utilities working to facilitate the design and construction of “high performance” schools. These are school facilities that aim to be models of energy and resource efficiency, as well as healthy and comfortable settings supporting quality education. CHPS uses a whole-building design approach, as well as providing designers with specific guidance on component systems, that incorporates the best of current knowledge and technologies. CHPS developed their own grading criteria using a point system, similar to the U.S. Green Building Council’s Leadership in Energy & Environmental Design (LEED™) scoring (U.S. Green Building Council, 2004).

The CHPS certification criteria include prerequisites and optional measures for low- or no-emission building materials, furnishings, and cleaning practices. Construction practices to minimize indoor pollutant buildup or dispersion are also required. The building materials

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requirements are based on the Section 01350 specifications discussed above, with adaptations for classroom conditions.

4.3.3.3 *Professional and Industry Guidelines and Practices for Building Materials*

Several building material industries have developed voluntary emission standards and programs to certify products that meet those standards. The most notable are the composite wood products industry and the carpet and rug industry. These programs have been successful in reducing emissions from their products over the last few decades.

The Composite Panel Association (CPA) is a professional organization for manufacturers of composite wood products. Composite wood products include pressed wood products such as plywood, particleboard, oriented strand board, and medium density fiberboard. The Association, founded in 1960, is dedicated to promoting the benefits of their products and producing products that meet a variety of rigid standards. Of particular note is the Grademark Certification Program that involves third-party certification of formaldehyde emission levels from wood panel products. This program uses the American National Standard Institute (ANSI) test procedures and limits for formaldehyde emissions from wood products.

The ANSI standard for particleboard and medium density fiberboard (MDF) establishes product specifications for dimensional criteria, physical and mechanical criteria, and formaldehyde emissions. Formaldehyde emissions criteria are identical to those established by HUD for mobile homes. Standard test method ASTM E 1333-96, *Determining Formaldehyde Concentrations in Air and Emission Rates from Wood Products Using a Large Chamber*, is used to simulate and measure emissions during normal product use. Products from each manufacturer must pass this test to bear the CPA Formaldehyde Grademark Program stamp. The Hardwood Plywood and Veneer Association (HPVA) also uses the ASTM E 1333 Large Chamber test for product evaluation.

The Carpet and Rug Institute (CRI) initiated a voluntary testing program in 1994 as a result of the U.S. EPA's Carpet Policy Dialogue. Carpet samples are tested for chemical emissions by a third party laboratory according to ASTM D 5116 – *Guide for Small-Scale Environmental Chamber Determinations of Organic Emissions from Indoor Materials/Products*. For carpet, emissions limits are established at 0.5 mg/m² for TVOC; 0.05 mg/m² for 4-PC (4-Phenylcyclohexene, the compound most associated with “new carpet odor”); 0.05 mg/m² for formaldehyde; and 0.4 mg/m² for styrene. CRI has similar emission standards and labeling programs for carpet cushion and adhesive. Products are tested quarterly to assure continued compliance with the test program requirements. Carpets that meet the emission test criteria bear a green and white CRI Indoor Air Quality Carpet Test Program logo in carpet showrooms so that consumers can identify the lower emitting carpet. Despite this labeling program, new carpet may still have a substantial “new carpet” odor, and extensive ventilation is recommended during installation and immediately after installation. Concerns remain regarding carpets' contribution to population exposures to formaldehyde and other potentially carcinogenic VOCs.

4.4 BUILDING DESIGN AND CONSTRUCTION STANDARDS AND GUIDELINES

Unlike some states, California does not have specific building design and construction standards to address IAQ. Standards for building design and construction that address IAQ in California are located mainly within the State's building energy and ventilation regulations. California Energy Commission (Commission) develops building energy and ventilation

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standards for nonresidential and residential building design and construction in California. These standards are enforced by local building officials, or in the case of public schools, by certified inspectors and the California Division of the State Architect. For existing homes, the home weatherization programs operated by the State and some utilities address the indoor emissions from combustion appliances and the building ventilation system. In addition, the Commission is beginning to consider energy efficiency standards for existing buildings.

Guidelines for IAQ in building design and construction are available for the major building types, usually as an integral part of “green”, “healthy”, and “sustainable” building programs. In recent years, ARB and other State agencies have worked together, through the State Sustainable Building Task Force and the Collaborative for High Performance Schools, to develop IAQ measures to be used in sustainable buildings for the State and public schools. However, these IAQ measures have not yet been fully incorporated into all state building requirements, and are not currently required in non-State buildings, in schools, or homes.

4.4.1 State Regulation of Design and Construction

The general mandate of the Commission is to ensure a reliable energy supply in a manner consistent with protecting the State's environment and enhancing its economy. The Commission's main impact on IAQ results from its energy conservation standards for new buildings, appliances, and insulation materials. The Commission does not directly regulate the operation and maintenance of buildings and appliances, factors that are among the most important determinants of IAQ. However, some aspects of the Title 24 standards deal with HVAC control systems, operation modes, and providing building operation, maintenance, and design ventilation rate information to building owners and building managers upon occupancy (Sec. 10-03[b]). In addition, the Title 24 standards provide the basis for HVAC operation and maintenance standards developed and enforced by Cal/OSHA. The Commission's standards and other activities pertinent to building design and construction standards are summarized below.

4.4.1.1 Design Ventilation Standards: Nonresidential, High-Rise Residential, And Hotel Buildings

The California Energy Commission sets minimum ventilation standards for the design and construction of new buildings and building additions in nonresidential buildings. The standards also apply to high-rise residential and hotels. These standards are published under CCR, Title 24, Part 6, and are required to be cost-effective. Local building officials enforce these State standards, but for public school buildings, the California Division of the State Architect (DSA, 2003) reviews the building plans and certifies building inspectors, thereby replacing the local building official function.

Currently, the 2001 edition of the Title 24 standards for nonresidential buildings require that newly constructed nonresidential buildings, and new HVAC systems in additions, be capable of supplying specific minimum ventilation rates for outdoor air. These minimum ventilation rates are typically met by a mechanical ventilation system that includes duct systems, control systems, and air filters. The standards require air balancing of airflows in the HVAC system, or airflow measurements, before occupancy in order to document that minimum ventilation rates are supplied.

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The minimum ventilation rate is calculated by two alternative criteria. New HVAC equipment must be designed and installed to be capable of providing no less than the larger of the following:

- The minimum ventilation rate based on the size of the building, expressed in cubic feet per minute (cfm) per square foot of conditioned building floor area. Rates required for different types of building use are shown in Table 4.2.
- The minimum ventilation rate based on the number of occupants: “15 cfm per person times the expected number of occupants.” The expected number of occupants may be specified by the designer. However, the expected number of occupants may not be assumed to be less than one half the maximum occupant load for existing purposes in Chapter 10 of the California Building Code.

Table 4.2. Minimum Design Ventilation Rates For Nonresidential, High-Rise Residential, and Hotel Buildings, 2001 Title 24 Standards¹

Type of Use	Cubic Feet per Minute per Square Foot of Conditioned Floor Area
Auto repair workshops	1.50
Barber shops	0.40
Bars, cocktail lounges, and casinos	1.50
Beauty shops	0.40
Coin-operated dry cleaning	0.30
Commercial dry cleaning	0.45
High-rise residential (5 stories or more)	Per CBC Section 1203 ²
Hotel guest rooms (less than 500 sq. ft.)	30 cfm/guest room
Hotel guest rooms (500 sq. ft. or greater)	0.15
Retail stores	0.20
Smoking Lounges	1.50
All others	0.15

1. Adapted from CCR, Title 24, Part 6, Sec. 121(b) 2, <http://www.energy.ca.gov/title24/standards/index.html>.
2. Specifies minimum opening areas for operable windows, but does not require actual operation of window or maintenance of operability.

These Title 24 standards also address the control of the HVAC system, which can markedly affect IAQ. The system and its controls must be capable of: 1) supplying the minimum ventilation rate at all times the building is occupied, and 2) providing one-hour pre-occupancy

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flushing at full occupancy ventilation rates, or at 3 air changes per hour or more. For spaces with high occupant density and at least 3,000 cfm capacity, such as auditoriums and large meeting rooms, the standards require “demand controlled ventilation” using a CO₂ sensor to achieve at least 0.15 cfm per square foot and maintain indoor CO₂ levels at or below 800 ppm. Such control systems have not been widely used in California, and it is not clear how well this requirement has been implemented. For the upcoming 2005 nonresidential Title 24 standards, the Commission required “acceptance testing” of demand control ventilation systems before occupancy, to ensure that the system performs as designed.

The Title 24 nonresidential standards also include an option for natural ventilation rather than mechanical ventilation. Natural ventilation by windows or roof openings is allowed if the openings are operable, accessible to the occupants, within 20 feet (25 feet in the 2005 standards) of the room’s outer edge, and sized to be greater than 5% of the conditioned floor area. Because there is no requirement for anyone to actually open the windows or to avoid nearby sources of outdoor pollutants or noise, the actual ventilation and IAQ provided under this option is highly variable, difficult to quantify, and difficult for the building owner or Cal/OSHA to verify.

4.4.1.2 *Design Ventilation Standards: Low-Rise Residential Buildings*

Unlike nonresidential buildings, most homes in California do not have outside air provided by mechanical ventilation – outside air is provided instead via open windows and doors and by unintentional air leakage in the building shell and the HVAC ductwork. In 1978 the Commission adopted residential building design standards that required extensive caulking, weather-stripping, and sealing to reduce air infiltration in new low-rise homes. To avoid IAQ problems for very tight homes in a few climate zones of California, heat recovery ventilators (air-to-air heat exchangers) were required to provide outdoor air ventilation for certain home designs (Maeda, 2004), but very few builders used this option and the requirement was subsequently dropped.

More recent Title 24 standards for low-rise residential buildings require additional reduction of air leakage through the sealing of HVAC ductwork. The sealing of duct work not only reduces natural air infiltration, but also unintentional infiltration of pollutant sources from spaces adjacent to the home, such as crawlspaces, garages, vehicle traffic, and local woodburning. Compliance credit for additional reductions in building air leakage reduction can be obtained by wrapping the building shell with an infiltration barrier, but this measure is optional.

To help ensure adequate IAQ, the 2001 Title 24 standards require mechanical ventilation with outdoor air when the target infiltration is below a certain limit. Energy credits for “unusually tight” construction are not allowed unless 1) air leakage is measured by an approved home energy rater using a blower door test method, 2) additional outdoor air is provided continuously by mechanical ventilation, and 3) combustion appliances such as furnaces and water heaters that use indoor air for combustion are not installed. The mechanical system may be either an independent system with its own fan, filter, and ductwork, or it may be integrated with a larger central air system. This approach is entitled “Compliance Through Quality Construction” (http://www.energy.ca.gov/title24/residential_manual/res_manual_2001.PDF, p. 129).

4.4.1.3 *Design and Performance Standards For Existing Buildings*

The Commission has begun to investigate strategies to achieve peak load reduction in existing buildings (mandated by AB 549, Longville). Such strategies may involve weatherization and

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duct sealing in existing buildings, which would affect levels of indoor air pollutants produced by combustion appliances in homes. Strategies under consideration for non-residential buildings could affect and perhaps improve the operation of existing HVAC systems, e.g., retro-commissioning (performance testing) and the increased use of control systems to obtain optimum building system performance. The Commission's recommendations to the Legislature are expected in October 2005. It is not clear at this time how IAQ will be addressed.

4.4.1.4 *Standards For Home Weatherization Programs*

The California Department of Community Services Development administers the federal Low-Income Weatherization Assistance Program through trained and certified contractors (DCSD, 2003; U.S. DOE, 2003a,b). This no-fee program installs insulation, weather-stripping, caulking, duct sealing, furnaces, and air conditioners as needed in low-income homes, both renter- and owner-occupied. To avoid IAQ and ventilation problems, the program contractors use a standard protocol across the state to:

- Limit the tightening of the building shell. This minimum leakage rate is based on modeling of the amount of leakage typically needed to provide 0.35 air changes per hour when averaged over the heating season.
- Inspect and test combustion appliances, venting systems, and ductwork, as discussed above under Appliances. Testing includes measurements of CO emissions, vent draft pressure, and building depressurization using the Combustion Appliance Safety (CAS) test.

The California Public Utilities Commission (CPUC) funds investor-owned utilities to provide rebates to homeowners for weatherization work. Some utilities are using the same CAS protocol as the state's program, some are using different protocols, and some are not doing any combustion safety testing. After holding public meetings and funding a study of the issue, the CPUC is drafting a standard CAS protocol for all investor-owned utilities and will hold adoption hearings in 2004 (O'Bannon, 2004).

4.4.2 Federal Regulations for Building Design and Construction

Federal standards do not currently exist for building design standards, with the exception of: new federal buildings, manufactured housing (mobile homes), and federal public housing. New federal buildings such as office buildings and residential buildings must meet federal design standards for energy efficient design, but federal policy has been to comply with local building standards where they are more stringent. Although California energy efficiency standards have long been more stringent than federal standards, the state and federal building ventilation standards are similar except that new federal standards for new single family homes include mechanical ventilation. Standards for manufactured housing and HUD-financed public housing are discussed below.

4.4.2.1 *Manufactured Housing*

Manufactured housing is regulated by HUD, which has developed requirements for low-formaldehyde plywood and particle board (but not medium density plywood), moisture control, exhaust venting, and fresh-air ventilation systems (HUD, 2002). These standards are enforced in California by the Manufactured Housing Section of the Department of Housing and Community Development. The formaldehyde emission requirements for wood products are discussed in Section 4.3.3.

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Earlier versions of the HUD standards gave homebuyers the option of the fresh-air ventilation system, but very few buyers in California selected that option. The current standard requires such a whole-house ventilation system and exhaust ventilation for kitchens and bathrooms. The whole-house ventilation system must provide 0.10 air changes per hour (ach), plus an assumed natural infiltration rate of 0.25 ach, for a total of 0.35 ach. However, a large study of newer manufactured homes in the Pacific Northwest found that many homes received much less than 0.35 ach, and that occupants did not use the ventilation systems as they are designed to be used (Lubliner and Gordon, 1990). In addition, some homes with wood fireplaces that have loose operable access doors have been observed to "backdraft" smoke into the homes, due to excess negative pressures caused by the HVAC equipment (Boe, 1999). The U.S. Department of Energy is funding research to improve the energy and indoor air quality performance of manufactured housing (Lubliner and Gordon, 1990).

4.4.2.2 *Public Housing*

Federal regulations for public housing financed by HUD require that the site characteristics, including the neighborhood and surrounding properties, undergo environmental review. HUD (1996) has provided a guide for this review, which addresses site hazards such as lead, PCBs from transformers, fuels, and solvents that could contaminate the homes during and after construction. At least three public housing sites in the U.S. have been built on known hazardous waste sites. Other housing built by the public or private sector may undergo a similar review through the California Environmental Quality Assurance (CEQA) review of environmental impacts, but the extent of the review for CEQA appears to be much less.

4.4.3 Professional and Industry Guidelines for Design and Construction

Several private and public sector organizations develop recommendations that serve as "consensus standards" or guidelines for achieving good indoor air quality through building design and construction. Many of these guidelines go beyond current regulatory requirements, but some are eventually incorporated into the regulations of government agencies. Concern has sometimes been expressed regarding the validity and effectiveness of such standards in protecting public health and welfare. However, although they may not meet all recognized needs, consensus standards serve an important function by providing guidelines for professionals such as building designers, managers and consultants, often long before government rule-making procedures could provide similar guidance. Government agency staff often actively participate in the development of such consensus standards by serving on standards-development committees of the major standards organizations.

ASHRAE has adopted a number of standards related to IAQ. These standards include those for ventilation and IAQ (Standard 62), air filter performance, thermal comfort, air-to-air heat exchanger performance, and building commissioning. These ASHRAE standards, especially Standard 62, have provided part of the basis for some building standards adopted by agencies such as HUD, CEC, and local governments. Thus, ASHRAE standards and their revisions have played a significant role in the previous and current State regulations for ventilation system design and energy-efficient building design as they relate to indoor air quality. Key guidelines from the public and private sector that address indoor air quality through building design, construction, and commissioning are summarized below.

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4.4.3.1 Guidelines for Nonresidential Buildings

ASHRAE's Standard 62, Ventilation for Acceptable Indoor Air Quality, was first adopted in 1973. It was based on criteria for odor perception and indoor carbon dioxide concentrations, with a prerequisite that all outdoor and re-circulated air meet State or federal outdoor air quality standards. The Standard included minimum ventilation rates for various types of room types in nonresidential buildings, and specified higher recommended ventilation rates for areas with smoking or other indoor sources of pollutants present. In 1981, ASHRAE Standard 62-1981 permitted lower ventilation rates in non-smoking areas, and included additional indoor air quality guidelines for several non-criteria pollutants such as formaldehyde and radon. The current version of Standard 62 for nonresidential buildings (ASHRAE, 2003a) specifies ventilation rates for outdoor air to adequately dilute pollutants emitted by occupants, occupant activities, and building materials, furnishings, and systems. For example, 15 cubic feet per minute (cfm) of outdoor air per person is required for most types of building occupants, and increased ventilation rates are specified for building areas with significant sources of indoor air pollution, such as bathrooms, printing and copying rooms, cooking areas, and garages. In addition, an appendix to the standard addresses the amount of ventilation required to reach recommended levels of specific indoor air pollutants. Standard 62-2001 is currently being modified under the continuous maintenance process; addenda to the standard are considered and undergo public review several times yearly.

4.4.3.2 Guidelines for School Buildings

Design guidelines for environmentally sustainable, healthy schools in California schools have been developed by the Collaborative for High Performance Schools (CHPS). CHPS members include state agencies, utilities, and building professionals. CHPS (2001) has published the *Best Practices Manual* that describes such design features as selection of low-emitting materials, ventilation system specifications, and IAQ management plans. The *Manual* provides a self-rating certification system and the CHPS criteria for the indoor environment and related building topics. School designers have latitude to incorporate practices in the manner that best fits the district's application. CHPS also provides ongoing training for school facility staff, building designers, engineers, consultants, and manufacturers. LAUSD, San Francisco USD, and a few other school districts have already adopted CHPS criteria for new school construction planning. A national version of CHPS has been adopted by a few other states recently. The U.S. EPA (2003b) has recently published "IAQ Design Tools for Schools," which is largely based on the CHPS *Best Practices Manual*.

CHPS plans to update its *Best Practices Manual* in 2004, and to add an Operations and Maintenance volume. Recently, the U.S. Department of Energy adapted the CHPS *Best Practices Manual* Volume II (Design) for a national audience (U.S. DOE, 2003). Several school districts, notably Los Angeles Unified, have established policies to require all new facilities to meet the CHPS criteria, including those for emission testing of building materials. CHPS manuals are available free of charge on the Internet at <http://www.chps.net>.

4.4.3.3 Guidelines for Residential Buildings

For residences, ASHRAE Standard 62-73 and subsequent versions minimally addressed ventilation and IAQ. However, ASHRAE recently expanded Standard 62 to include ASHRAE Standard 62.2, which specifically addresses IAQ in low-rise residential buildings (Sherman, 2003a; ASHRAE 2003a). This standard now includes several IAQ features, such as whole-

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house mechanical ventilation with outdoor air, “local exhaust” fans in the kitchen and bathrooms, backdraft testing, and carbon monoxide alarms (Sherman 2003b). Ventilation rate requirements are 7.5 cfm/person, based on the need to remove pollutants generated by occupants’ indoor activities, plus 3 cfm/100 square feet of floor area, based on the need to remove pollutants generated by background or building-related sources. The standard also refers to Standard 62.1 for specific guidance in providing sufficient ventilation based on the pollutant emissions from the occupant activities and from the building materials and furnishings. This standard is for new home design, but can also be applied to existing homes. It is not a regulatory requirement in California, but it may help provide the basis for future State standards.

Professional home builder and public health groups in California have developed recommended practices for improving the performance of new and existing homes that take a “whole-building approach” to address building materials, ventilation, combustion safety, moisture control, thermal comfort, and energy efficiency. Examples of such groups are the Energy Efficient Building Association (EEBA, 2003), and the California Building Performance Contractors Association (CBPCA, 2003a.b). In addition, the National Association of Home Builders (NAHB, 2004) is beginning to develop guidelines for “green” home building that consider the whole building and the various environmental concerns. The American Lung Association (ALA, 2004) has implemented the Health House Program, a whole-building approach that includes builder guidelines, training, and educational information developed with a focus on preventing and reducing asthma and other indoor pollutant hazards. The HUD (1999) Healthy Neighborhood Program has funded several groups across the U.S. to use inspection and intervention to address asthma triggers, CO poisoning, lead poisoning, and other health and safety hazards for children in low-income populations. Numerous similar guidelines and programs have come into existence.

The recommendations from these groups generally include measures such as low-VOC building materials, whole house mechanical ventilation, exhaust fans for high-moisture areas such as bathrooms and kitchens, combustion appliance safety testing, and combustion appliances that are resistant to backdrafting and other venting problems. However, these approaches have not been widely used in California – only a few custom homes or demonstration projects have used these approaches in California.

4.4.3.4 *Building Commissioning*

Building commissioning is the process of verifying and documenting that construction, operation, and maintenance of a building meet the design specifications of the project. Ideally, commissioning is conducted by a trained, third-party inspector, begins during the design phase of a project, and includes the development and training for operation and maintenance procedures. Through the commissioning process, potential problems are detected early on, HVAC systems are precisely tuned, change orders are reduced, building operators are trained, utility and other costs are reduced, and a healthful and comfortable workplace is produced (ODOE, 2004).

ASHRAE (1993, 1996) has published guidelines for commissioning HVAC and control systems in large buildings. The California Commissioning Collaborative (CCC, 2004), which includes government, utility and building services professionals, has been promoting building commissioning through incentives, training, research, and the development of model Request for Proposals. Research funded by CCC members have found that in order to achieve building performance goals over the lifetime of the building, it is necessary to conduct enhanced training

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of building operators, to tracking building performance, and to conduct ongoing commissioning to meet current needs of building occupants (Friedman *et al.*, 2003).

Building commissioning has proven to be very cost-effective in achieving reduced use of energy and other resources, while providing health and safety benefits through improved indoor air quality. The Commission included basic commissioning requirements in earlier Title 24 standards – air balance and flow testing of HVAC systems in nonresidential buildings (residential buildings do not have such requirements). The State of California has recently become more active in the building commissioning area.

- With guidance from the State Sustainable Building Task Force, the Department of General Services (DGS) has begun to incorporate building commissioning in new projects for large office buildings, starting with the Capitol East End Office Building Project that was recently completed. DGS is also beginning to “re-commission” existing state buildings.
- The Commission included HVAC commissioning requirements for HVAC control systems and major components in the 2001 Title 24 standards.

However, despite these and some related activities, formal and complete building commissioning is still relatively rare in California’s nonresidential buildings, and it is still in the methods development stage for residential applications (LBNL, 2003).

4.5 BUILDING OPERATION AND MAINTENANCE STANDARDS

4.5.1 State Standards

The primary building operation and maintenance regulations in California are those of Cal/OSHA, which apply to most industrial and non-industrial workplaces. Non-industrial workplaces include classrooms, beauty salons, hospitals, dry cleansers, medical laboratories, retail shops, copy shops, and many other workplaces that can at times have elevated levels of air pollutants associated with specific activities.

Residential buildings and home offices are generally not subject to regulations or other standards for operation and maintenance. The exceptions are residential buildings used for special purposes: nursing and rest homes; daycare centers; and public housing. School buildings are also subject to Cal/OSHA regulations, although school maintenance staff are not always aware that these workplace regulations apply.

The Cal/OSHA regulations most applicable to indoor air quality are Sections 5142 and 5143 (CCR, Title 8, <http://www.dir.ca.gov/samples/search/query.htm>). These regulations address the proper operation and maintenance of mechanical ventilation systems. Cal/OSHA enforcement of these standards mainly occurs when a worker files a complaint, and is therefore very limited. Other sections of Title 8, such as those addressing asbestos and lead, take a pro-active approach by requiring worker training, building assessment, and occupant notification.

4.5.1.1 Ventilation Regulations for Workplaces

Employers are required to maintain and operate HVAC systems to provide at least the minimum quantity of outdoor air required by the State Building Code (Title 24) at the time the building permit was issued. Section 5142, *Mechanically Driven Heating, Ventilating and Air Conditioning*

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(HVAC) *Systems to Provide Minimum Building Ventilation*, was adopted in 1987 in response to a labor union lawsuit concerning IAQ. With limited exceptions, systems must be operated during working hours, unless the employer can document that the outdoor air requirements are being met by non-mechanical means. Employers must perform at least annual inspections, and correct problems found during those inspections. This section also includes record-keeping requirements; employees can obtain these records on request.

Section 5143, *General Requirements of Mechanical Ventilation Systems*, was adopted in 1976. This section requires annual performance measurement of ventilation systems that are used to prevent harmful exposures, such as local exhaust ventilation systems and laboratory hoods. This section requires contaminant-free make-up air (supply air), regular filter maintenance, and pressure gauge installation to indicate when filters must be cleaned or replaced. This section also has record-keeping requirements.

4.5.1.2 *Mold in Workplaces*

As mentioned in Section 4.1 of this report, Cal/OSHA requires that workplaces be maintained in a sanitary condition. Section 3362 in CCR Title 8, *Sanitation – General Requirements*, was amended in 2002 to include new subsection (g) that requires employers to correct water intrusion, leakage or other uncontrolled accumulation because of the potential to cause mold growth. Other subsections require the employer to maintain the workplace in a clean, orderly, and sanitary condition; to prevent the entrance of insects, rodents or other vermin; to store putrescible wastes and garbage in covered cans; and to remove wastes as frequently as necessary to prevent harmful exposure.

4.5.1.3 *Construction Related Indoor Air Quality Investigations in Workplaces*

Many complaints to Cal/OSHA result from remodeling or other construction activities conducted in or adjacent to areas in which employees are working. In these situations, employers are required under Section 3203 to inspect the new operations, to evaluate hazards found, and to take measures to correct them in a timely manner. Section 5194 further requires employers to inform employees about hazardous substances in their work area, including substances being applied by contractors. In addition, the following Cal/OSHA sections require employers to control specific construction related hazards, such as asbestos, lead, and dusts.

Section 5208, *Asbestos*, was filed in 1996 and amended subsequently. It requires building and facility owners to determine the presence, location, and quantity of installed asbestos-containing materials (ACM) or presumed asbestos-containing materials (PACM), and to inform employers or employees in that building or facility of those results. Employers are required to provide that information to their employees. This section further requires training for employees who perform janitorial or maintenance activities in areas containing ACM or PACM, and the posting of labels or signs on installed ACM or PACM.

Section 1529, *Asbestos*, requires employers engaged in construction activities involving ACM or PACM to use certain control measures to protect employees in adjacent areas. Subsection (d) requires that asbestos contractors inform employers of employees in adjacent areas of those control measures, and it requires those employers to take measures to protect their employees from asbestos hazards and to take steps on a daily basis to ensure that the control measures are adequately protecting their employees.

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Section 1532.1, *Lead*, requires employers engaged in construction activities involving lead to use specified control measures, to establish a regulated area where exposures exceed the PEL, and to inform adjacent contractors of those control measures.

4.5.1.4 *Environmental Tobacco Smoke in Workplaces*

Environmental tobacco smoke has historically been strongly correlated with poor indoor air quality. Since 1995, California regulations (Labor Code 6404.5) prohibit smoking in most enclosed workplaces (also discussed in Section 4.1.5). This statewide prohibition is primarily enforced at the local level. Cal/OSHA is not required to respond to complaints until the employer has been found guilty at a local level three times in the previous year (http://www.dir.ca.gov/dosh/dosh_publications/smoking.html). On May 29, 2003, in a Decision After Reconsideration (Robert D. Schultz and James A. Noll (OSHAB 01-125), the Appeals Board held that the Division does not have authority to take action to enforce the Labor Code in the absence of a regulation promulgated by the Standards Board. Subsequently, in February 2004, the Standards Board adopted a workplace regulation for enforcing the tobacco smoking ban.

4.5.1.5 *Indoor Air Quality Enforcement Policy & Procedures for Workplaces*

Policy and Procedure C-48 (<http://www.dir.ca.gov/doshpol/p%26pc%2D48.htm>) describes Cal/OSHA enforcement procedures for indoor air quality investigations. In general, it defines “serious indoor air quality complaints” as those in which there is a known or suspected building-related illness, or a cluster of cancer cases or adverse birth outcomes. Non-specific complaints related to building IAQ are also considered serious when their magnitude, frequency, and their severity are above certain levels. “Non-serious indoor air quality complaints” are defined as those involving particles, HVAC systems, thermal stress, or symptoms that are not specific to a particular disease or pollutant. Investigations of building related illness might involve the Cal/OSHA medical unit, as well as support from state and local health agencies. Interview, inspection, monitoring, and citation procedures are also outlined.

4.5.1.6 *Standards for Nursing Homes and Public Housing*

Nursing homes, group homes, and community care facilities in California are regulated by the DHS Licensing and Certification Program (HSC 1265 et seq., HSC 1520 et seq.; HSC 1725 et seq.). The licensing requirements include provisions to protect the health and safety of clients. State inspectors conduct annual inspections or “surveys” and investigate complaints regarding the condition of the building, food preparation, and health care. They also make follow-up visits to assure that problems have been identified and corrected. However, it is not clear that IAQ is a high priority item for inspections unless there are several complaints by clients or an outbreak of a building related illness.

Public and Indian housing in California are funded by federal and state agencies, often through local public housing authorities and redevelopment agencies. The U.S. Department of Housing and Urban Development (HUD) also provides funding for five percent of U.S. homeowners and renters through programs for public, assisted, and insured housing as well as housing financed through HUD's formula for competitive grant programs (<http://www.hud.gov/news/release.cfm?content=pr02-105.cfm>). State agencies often provide additional funding. HUD has adopted regulations regarding the operation and maintenance of public and Indian housing that receives federal funds. Public housing authorities are required to

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annually set aside funds for building operation and maintenance. Some operation and maintenance activities are required to undergo environmental review (24CFR990.111). However, except for lead and asbestos contamination problems, public and Indian housing programs have not routinely addressed IAQ in their operation and maintenance programs.

4.5.2 Guidelines for Building Operation and Maintenance

Various guidelines for providing good indoor environmental quality through building operation and maintenance are available from the public and private sectors. Examples from California and national organizations are discussed below. Many other guidelines are available from other states and nations that have active programs to address indoor environmental quality.

4.5.2.1 *IAQ Tools For Schools*

The U.S. EPA's *IAQ (Indoor Air Quality) Tools for Schools* Program is a program developed to help schools identify and prevent indoor air quality problems, using a team approach to school IEQ management. The program provides educational materials and tools for evaluating the impact of school maintenance functions and occupants' daily activities on indoor air quality. U.S. EPA makes their *IAQ Tools for Schools* action kits available at no cost, and has funded numerous training workshops, including many in California. Implementation of this program by school districts in California and other states has not taken a significant amount of staff or funding, but it has yielded substantial health and economic benefits (see Section 3, Costs of Indoor Air Pollution). As a prerequisite for obtaining state funding for school operation and maintenance, the State of Minnesota has required schools to adopt an IAQ management plan based on the *IAQ Tools For Schools* program.

Despite the outreach efforts for the *IAQ Tools For Schools* program, awareness and use of the program among California schools are still low: 35% of schools recently reported that they were familiar with the program, but only 11% of California schools reported that they use all or part of the program (ARB/DHS, 2003). These low levels may be due to a misperception regarding the level of effort required, although the program is adaptable to any level of resources.

4.5.2.2 *Large Building Air Quality Guidance*

U.S. EPA (1991) and the National Institute of Occupational Safety and Health have published "Building Air Quality: A Guide for Building and Facility Managers." This document describes the common principles of good facility management, provides background information on topics related to IAQ, and recommends strategies for problem identification and resolution in large buildings such as office buildings. The guidance emphasizes changing how building managers operate and maintain their building, not increasing the amount of work or cost of maintaining the building. The companion document, the "Building Air Quality Action Plan", is an easy-to-understand, 8-step implementation path for building owners and managers (U.S. EPA, 1998). The U.S. EPA (1997b) has also published guidelines for occupants of office guidelines, entitled "An Office Building Occupant's Guide to Indoor Air Quality".

DHS (1995) has published guidance for occupants of large buildings with HVAC systems. Entitled "A 'Do-It-Yourself' Inspection of a Ventilation System," this document provides background information and guidance to help occupants identify potential causes of IAQ problems in their building, and to help them ask the right questions of building managers and investigators.

5. METHODS TO PREVENT AND REDUCE INDOOR AIR POLLUTION

There are many actions that can be taken to reduce indoor air pollution and, in some cases, completely eliminate sources. The best approach will depend on the specific problem and will vary accordingly.

Five basic approaches to indoor pollution prevention and control are discussed below. The most effective approach to solving existing problems is usually to remove or reduce the sources of indoor pollution. The most effective preventive measure is to use building materials, consumer products, and appliances that emit little or no air pollutants. Ventilation, necessary for healthful indoor air quality and comfort, also helps reduce exposure to indoor air pollutants, but not as effectively as source reduction. Public education is an important part of pollution prevention; it can lead to a better understanding of indoor pollution and promote informed choices. Air cleaning devices (air filters and air cleaners) can also be helpful in certain situations when used along with source reduction and ventilation. Finally, proper operation and regular maintenance of buildings is key to preventing indoor air quality problems.

5.1 SOURCE CONTROL

Preventing or reducing emissions at the source is the most effective and reliable approach to reducing indoor pollution and risk (ARB, 1989; NRC, 1981). Source control can be accomplished through source substitution, source removal, and source modification.

Source substitution involves using an alternative product that emits little or no pollutants of concern. For example, alternative building materials that emit little or no formaldehyde are generally available and can be used in place of a building material that emits formaldehyde. Source removal involves eliminating the source. For example, properly disposing of old cans of paint or aerosol spray products stored in a closet or under the kitchen sink removes these sources from the home. Source modification involves reducing the rate at which a pollutant is emitted into the indoor environment (NRC, 1981). Source modification could involve a change in design, formulation, or usage of a consumer product (i.e., a liquid cleaner with a reduced rate of pollutant emissions). Another example of source modification would be sealing all surfaces of a particleboard bookshelf to prevent the formaldehyde emissions (ATS, 1997; NRC, 1981).

Source control typically can be accomplished at the manufacturing level without direct impact on people's activities and homes. Consumer products can often be reformulated by the manufacturer to reduce or eliminate any potentially harmful emissions, with little or no impact on the consumer. Extremely reactive constituents (i.e. terpenes, such as limonene or pinene) that contribute to the formation of toxic pollutants indoors can be removed from individual products that are prevalent in the home, thereby reducing related health risks. Appliance standards can also be improved to increase combustion efficiency as well as energy efficiency, thereby lowering indoor and outdoor air emissions.

Minimizing indoor emissions is generally more effective than removing them after emission has occurred.

T.J. Kelly, Battelle, Indoor Air Quality Symposium: Risk Reduction in the 21st Century, Sacramento, May, 2000

5.2 VENTILATION

ASHRAE Standard 62 defines ventilation as “air delivered to a space to dilute airborne contaminants: the use of outdoor (makeup) air and appropriately cleaned recirculated air.” (ASHRAE 1981/2003b). Ventilation is a standard engineering approach to achieving good indoor air quality (ATS, 1997). Ventilation can be intentional such as opening doors or windows, or unintentional, namely infiltration of air through the cracks and gaps of the building envelope. Intentional ventilation can be natural or mechanical. Natural ventilation is achieved by passive airflow through open windows and doors. Mechanical ventilation is achieved through the use of an appliance such as heating, ventilation, and air conditioning (HVAC) systems commonly used in large buildings, or through kitchen and bathroom exhaust fans in homes.

There are two types of mechanical ventilation – mixing and displacement. Mixing ventilation mixes “fresh” air and existing room air to uniformly dilute pollutants; this is the most common type of mechanical ventilation. The quality of outdoor supply air plays an important role in actual and perceived air quality, Sick Building Syndrome (SBS) symptoms, and productivity (Wargocki *et al.*, 2000). Displacement ventilation is the introduction of “fresh” air so that the air around the occupants (breathing zone) is separated from the polluting sources. Displacement ventilation requires precision design and operation (Liddament, 2000). The airtightness of a building is important to the energy performance of the ventilation system. A relatively airtight building is required for some heat recovery systems to function correctly and can also provide a barrier to transient outdoor pollutants (Liddament, 2000).

What is needed to improve ventilation in California buildings?

- ◆ Better commissioning and annual operational checks
- ◆ Quieter systems, especially for schools
- ◆ Control of building space pressures
- ◆ Use of higher efficiency air filters
- ◆ Better control of thermal comfort parameters
(temperature and humidity)

F.J. Offermann III, Indoor Environmental Engineering, Indoor Air Quality Symposium: Risk Reduction in the 21st Century, Sacramento, May, 2000

Localized ventilation is often used to remove pollutants and increase ventilation in special use areas, such as the bathroom and kitchen. Local exhaust ventilation can be very effective when airflow is sufficient and exhaust ventilation is used. The use of exhaust hoods ducted to the outdoors is especially critical when using gas stoves or ovens, in order to remove the emissions from the appliances and those generated during cooking, which can be substantial.

Adequate and effective ventilation is necessary for acceptable indoor air quality, even when known air contaminants are minimized. For instance the air may have very low pollutant levels, but without ventilation, exhaled carbon dioxide and other body effluents will build up, resulting in a stuffy, uncomfortable environment. Ventilation not only removes and dilutes contaminants, it also removes moisture from the air which helps to prevent mold growth. However, ventilation is not a complete solution to indoor pollution. Ventilation consumes energy, and some pollutants,

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such as formaldehyde from building materials, require years to off-gas and are not completely removed by ventilation. Also, the near-source exposures that occur when people use individual products are not completely eliminated through ventilation, just reduced. It is generally preferable to remove highly toxic compounds from these products all together (ARB, 2000b).

5.3 PUBLIC EDUCATION

People's choices and behavior have a major impact on their exposures to air pollution. Human activities are probably the most important factor in determining the condition of the indoor environment. Cooking, the use of various consumer products, cigarette smoking, and other activities can result in immediate, significant releases of pollutants indoors which are immediately inhaled (ARB, 1987). Thus, public education is a key step for reducing Californians' exposures to indoor air pollution (NRC, 1981; ATS, 1997). People are often not aware of the risks associated with indoor pollution and what they can do to protect their health. Sometimes activity pattern changes are needed, e.g., people must use some types of consumer products properly in order to reduce their exposure and risk.

Increasing people's awareness can be accomplished through public education. However, public education will never be a complete solution. Some groups of the population cannot respond appropriately to take needed action. For example, children cannot read or understand all written information that is provided, elderly people living in group settings cannot control the products used in the facility, and low income families may not be able to afford safer alternatives, even when they are fully aware of them.

Additionally, public education will not always reach all members of society. The many non-English speaking ethnic groups that enrich the California population pose an increased communication challenge. The California population contains a large number of people who speak a language other than English at home, as well as a large number of foreign-born individuals. According to the U.S. Census Bureau, the year 2000 census reported about 18% of the U.S. population 5 years and older spoke a language other than English at home. In California, that number is higher: about 40% of Californians speak a language other than English at home (DOF, 2002b). Also, the proportion of the California population that reported being foreign-born has increased from about 21% in 1990 to over 26% in 2000 (Malone *et al.*, 2003). Because of these reasons, increased public education and information are not adequate as a sole approach to preventing indoor pollution.

5.4 AIR CLEANING DEVICES

Air cleaning devices are available in a variety of types and sizes. Most air cleaners remove particles, a few remove gases, and some do both. Air cleaners can remove particles from the air using a mechanical or physical barrier, or electronically. Mechanical air cleaners draw air through a filter with different sized pores that trap the particles. Electronic air cleaners are available in three basic types: electrostatic precipitators (ESPs), ionizers, and mechanical-electronic hybrids. ESPs use a small electrical charge to collect particles from air drawn through the device. Ionizers, or negative ion generators, cause particles to stick to materials near the ionizer (such as the carpet and walls). Hybrid devices use both mechanical and electronic devices for pollutant removal. Both ESPs and ionizers produce ozone as a by-product.

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Air cleaners that remove gases and odors are less common, and relatively more expensive to purchase and maintain. Gaseous pollutants are typically trapped or destroyed as the air is drawn through materials such as activated charcoal or alumina coated with potassium permanganate. However, the filter material can become quickly overloaded and may need to be replaced often.

Portable air cleaners are sometimes practical for rooms in existing homes where addition of a central air cleaner is too costly. Proper size, installation, and maintenance are critical for portable air cleaners to be effective. Test standards for particle removal by air cleaning devices have been developed by trade and engineering groups. The effectiveness of some portable air cleaners in removing particles is usually rated in terms of pollutant removal efficiency or Clean Air Delivery Rate (CADR), measured in cubic feet per minute (cfm). The CADR equals the airflow (cfm) multiplied by the efficiency of particle removal; a larger CADR is better. The CADR ratings are given separately for the removal of dust, pollen, and environmental tobacco smoke. Standards for gas removal have not been developed.

Portable air cleaning devices have very limited utility, are often not very effective (ARB, 2000c; ATS, 1997; Consumers Union, October 2003), and can incur high energy costs relative to the benefit gained. Air cleaners may be useful for controlling airborne particles for some individuals with special sensitivities, such as those with asthma or allergies who use them in their bedrooms at night. Based on the limited scientific evidence that is currently available, the health benefits of air cleaners are not clear. However, it is clear that an ozone generating air cleaner should not be used (DHS, 1997; ALA, 1997). Generally, it is more effective to prevent emissions rather than to try to remove them from the air once they are there.

For new homes or major remodels, "whole-house" or "fresh-air" ventilation systems can be installed. Installed costs depend on the system size and the type of air cleaning device. Fresh-air ventilation systems are recommended in new, tightly built energy-efficient houses, and for situations where the outdoor air is the major source of indoor pollution. To design and install any central system air cleaner, a company or contractor that is well experienced in designing and installing central filtration systems is important (ARB, 2000c).

Ozone Does Not Clean the Air

Independent studies by the U.S. Environmental Protection Agency, the Consumers Union, and others have shown that ozone-generating air cleaners do not effectively destroy microbes, remove odor sources, or reduce indoor pollutants enough to provide any health benefits. Ozone masks the odor of other indoor pollutants by deadening the sense of smell. It also reacts with certain indoor pollutants to produce toxic by-products, such as formaldehyde. Ozone in the air must reach extremely hazardous levels (50-100 times the outdoor air quality standards) to effectively kill microbes (Foarde *et al.*, 1997).

The California Department of Health Services advises the public not to use so-called "air purifiers" that are specifically designed to generate ozone indoors. These ozone generators are sometimes marketed as emitting "trivalent" oxygen, "activated" oxygen, "saturated" oxygen, "super-oxygen," or "mountain-fresh air." These devices may also be combined with a negative ion generator. However, these devices are actually emitting ozone, and they are currently unregulated. Ozone is a harmful air pollutant that is the main component of ground-level smog. Breathing ozone can be harmful, especially for children, the elderly, and people with asthma, emphysema, bronchitis, or other respiratory diseases. Ozone also irritates the eyes, nose, and throat. Long term exposure to ozone may permanently reduce a person's breathing ability.

Some devices, primarily ESPs and ionizers, emit ozone as a by-product. To minimize ozone emissions, these devices need to be cleaned and maintained regularly, and operated only according to the manufacturer's instructions.

Ozone-generating Air Cleaners

"People should avoid using indoor air cleaning devices that produce ozone...These devices can quickly produce enough ozone in a confined space to exceed the California Stage 2 and 3 smog alert levels as well as worker health and safety standards."

Jim Stratton, M.D., M.P.H., State Health Officer, Department of Health Services, Press Release 27-97, Sacramento, April 1997.

5.5 BUILDING OPERATIONS AND MAINTENANCE

A well-operated building on regular maintenance is fundamental to the provision of good indoor air and a comfortable and productive indoor environment (Spengler, 2001; ATS, 1997). Basic building elements interact to affect the indoor environment. The building shell shelters the occupants from the elements and at the same time may contain contaminants. The HVAC system conditions the air for comfort, but if not properly maintained can add to the discomfort of occupants as well as increase the concentration of indoor pollutants. The outdoor environment is the source of fresh air for the occupants and often needs to be conditioned (e.g., warmed in the winter, cooled in the summer, and moisture level adjusted) prior to delivery for occupant use.

Maintaining high quality indoor air requires a solid understanding of how buildings operate and the relationship between the HVAC equipment and indoor air quality (Bearn, 1993). For example, without adequate scheduled maintenance, dust can accumulate in ductwork systems and provide a site for microbiological growth, such as fungi and bacteria (Liddament, 2000). Proper operation and maintenance begins with thorough training of personnel who monitor building operations and perform maintenance.

6. PRIORITIZATION OF SOURCES AND POLLUTANTS BASED ON EXPOSURE AND ADVERSE IMPACTS

Reduction of public exposure to the many indoor air pollutants is most effectively achieved by reducing pollution at the source. Because exposure is dependent on the presence and use of sources, pollutants are best prioritized for action by source categories. Mitigation approaches focused on source categories would be most effective for reducing pollutant exposure, and would be feasible to implement. Table 6.1 suggests a prioritization scheme for implementation of mitigation measures, by source categories, based generally on estimated exposure and risk, with the highest priority categories listed first. The primary criteria used in prioritizing the source categories included the extent of the population's exposure to the sources and their emissions, and the relative reduction in health impacts that could be achieved with action.

Table 6.1 also suggests potential approaches for mitigating the pollutants and sources listed. Emission reductions should be accomplished at the manufacturing, distribution, or construction stage. Alternatives or mitigation options are currently available for most of the sources listed. Emission limitations achieved at the manufacturing stage, such as reducing toxic contaminants in building materials, would be effectively invisible to the consumer and assure exposure reduction. Similarly, new appliance standards would reduce combustion emissions without reducing the range of product choices in the market place. With this approach, no changes in individual behavior would be anticipated. As a result of public awareness and demand, there are an increasing number of "green" or low-emitting building materials, furnishings, and consumer products already available. Low-emitting carpets, no-formaldehyde furniture, and non-toxic cleaning products are currently available, and would grow in market share.

- **Building materials and furnishings** are the highest priority for mitigation because they often emit multiple toxic air pollutants, especially when new, and have a high loading level in indoor environments. A reduction in pollutants from this category (formaldehyde being the most predominant) would benefit all indoor environments and has potential for significant health benefits due to reduced incidence of asthma exacerbation, cancer, and eye, nose and throat irritation. Composite wood products made with phenol-formaldehyde resin or methyl diisocyanate (MDI) have much lower formaldehyde emissions than those made with urea-formaldehyde resin. In cabinets or furniture, all surfaces of these products can be coated or laminated to substantially reduce formaldehyde emissions. Building materials are currently available that meet Section 01350 emission requirements for formaldehyde and other TACs (Alevantis, 2003). Low-emitting modular office furniture was used in the Capitol East End Project, and should be used in all future state and private offices. Public demand derived from educational efforts and sensitive groups has resulted in many alternative low-emitting products. Numerous governmental groups focused on sustainable building and environmentally preferable purchasing are compiling lists of acceptable products that have a low impact on indoor environments. These low- and no-emitting alternatives are available and should be widely used.
- **Environmental tobacco smoke** has been greatly reduced in California, primarily due to legislation that bans smoking at the workplace. However, children's exposures remain a concern, because they can be highly exposed when smoking occurs in their home or in vehicles driven by family or friends who smoke. Actions to reduce children's exposure—such as an increased focus of public education on smoking parents, and reduction of smoking inside vehicles and homes with children—remain a high priority.

Table 6.1. Prioritization of Pollutant Sources for Mitigation¹

Sources of Pollutant	Examples of Toxic Air Pollutants² Emitted	Potential Approach to Mitigation³
Building Materials & Furnishings (particle board, plywood, paneling, flooring, caulk, adhesives, carpet, furniture)	Formaldehyde, acetaldehyde, benzene derivatives, acrylates, naphthalene, phenol, other VOCs	Emission limitations, product use restrictions, market incentives
Environmental Tobacco Smoke	Particles, polycyclic aromatic hydrocarbons, benzene, carbon monoxide, VOCs	Focused parent education, restrictions in vehicles and outdoor locations
Combustion Appliances (gas & propane stoves, ovens, furnaces, heaters; woodstoves and fireplaces)	Carbon monoxide, nitrogen oxides, particles, soot, polycyclic aromatic hydrocarbons	Emission limitations, active exhaust ventilation, safety devices, product use restrictions, product re-design, improved venting
Air Cleaners	Ozone, particles	Emission limitations, efficiency ratings
Consumer Products (e.g. household cleansers, furniture- and floor-care products, air fresheners, stain removers, detergents)	Methylene chloride, para-dichlorobenzene, perchloroethylene, terpenes, toluene, benzene, naphthalene	Emission limitations, chemical reformulations, and product use restrictions to reduce TACs and nonreactive VOCs with health impacts; labeling program
Architectural Coatings (e.g. paints, sealants, lacquers, varnishes)	Formaldehyde, acetaldehyde, ethylene glycol, metals, others	Emission limitations, chemical reformulations, use restrictions to reduce TACs & nonreactive VOCs with health impacts;
Personal Care Products (e.g. products used for hair and skin care)	Formaldehyde, acetaldehyde, toluene, metals, others	Emission limitations, chemical reformulations to reduce TACs and nonreactive VOCs
Household & Office Equipment and Appliances (computers, photocopy machines, vacuum cleaners)	Particles, styrene, VOCs, phthalates, ozone, PBDE	Emission limitations, local exhaust requirements

1. Source categories are presented in general order of prioritization. However, individual sources in a lower priority category may supersede some sources in “higher” priority categories.
2. Toxic Air Pollutants: pollutants identified as Toxic Air Contaminants (TACs) by the California Air Resources Board, and/or identified as Proposition 65 chemicals; or, criteria (traditional) air pollutants.
3. Public education, economic incentives, and non-regulatory approaches should also be used where appropriate. The actual approach taken would be determined after extensive discussions with the relevant industries, in consideration of costs, feasibility, and effectiveness.

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- **Combustion appliances** are also a high priority for mitigation. They emit carbon monoxide, nitrogen dioxide, polycyclic aromatic hydrocarbons, particles, and other pollutants, depending on the appliance fuel. These pollutants can have severe acute health effects, exacerbate asthma, and cause cancer. Reduced exposure to pollutants from gas and propane appliances, whether it be through emission limitations, active exhaust ventilation, or both, could have immediate widespread benefits for occupants in environments with such appliances. Precedents for mitigation of appliance emissions has been set in the low-income weatherization program (DCSD, 2003). Guidelines for CO levels and venting requirements in this program can be expanded to statewide regulations.

Statewide measures to reduce emissions from woodstoves and fireplaces are also highly desirable; these could have a major impact on improving both community-wide indoor and outdoor air quality in many areas of the state. Emission limitations, product re-design, product use restrictions, and improved venting can be used for reducing this type of pollution. A number of local government entities have recently approved regulations restricting the use of woodstoves and fireplaces: in the San Francisco Bay area, 24 cities have ordinances that prohibit conventional fireplaces in new construction, and the mountain town of Truckee has a more aggressive policy that requires that existing unapproved wood burning appliances be removed by July 15, 2006. The San Joaquin Valley (www.valleyair.org/Burn_Programs) implemented a daily advisory for restrictions on residential fireplace or wood stove use on January 1, 2004. Woodsmoke especially impacts those with asthma and other respiratory disease.

- **Indoor air cleaning devices** should be restricted to eliminate ozone emissions. Some devices marketed as air cleaners release ozone which can directly harm sensitive occupants. These air cleaners should be prohibited in occupied spaces, because they are ineffective at safe levels and produce potentially harmful levels of indoor ozone, and effective alternatives are available. Consumers often seek advice about the type of air cleaner they should buy to meet their specific needs. There are few criteria available to guide consumers in their purchase. Both emission limitations and pollutant removal efficiency ratings are needed for these devices, which are currently unregulated. Private publications such as *Consumer Reports* have rated air cleaners (Consumers Union, October 2003). This could be expanded to a governmental program.
- **Consumer products, architectural coatings, and personal care products** have been regulated to reduce emissions of reactive VOCs in order to reduce smog formation. Further restrictions to assure reduction of toxic air contaminants and nonreactive VOCs with potential health implications appear warranted. Due to the nature of these products (household cleansers, air fresheners, stain removers, etc.), the user is in close proximity to the release of chemicals during use and experiences greater pollutant exposure than individuals who may be elsewhere in the room or building where the product is used. Implementation of chemical reformulations, emission limitations, and/or product use restrictions of consumer products could result in significant risk reduction for the user (and others in the building).

A few such measures have already been taken: for example, in the 1990s, trichloroethylene was eliminated from a brand of typewriter correction fluid, thereby reducing indoor exposures to this chemical. Also, the ARB previously required that chlorinated solvents be removed from aerosol adhesives by January 1, 2002. Many of the architectural coatings are now available in "low VOC" versions due to formulation changes targeted toward reducing outdoor ozone. Reformulation of other products, such as cleaning agents to remove

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terpenes, could go far to reduce irritant and carcinogenic effects. Wider usage of the products with low emissions could be encouraged or enforced. Efforts focused on these source categories are a medium priority.

- **Household appliances and office equipment** such as computers, copy machines, and vacuum cleaners can emit a variety of pollutants such as particles, styrene, phthalates, ozone, particles, and PBDEs. Emissions from each type of appliance should be addressed through emission limitations and/or local exhaust requirements. Examples of a specification for local exhaust would be the placement of a large copy machine under a negative pressure ventilation hood, or specifying a level of exhaust ventilation for a room. In most cases, these pollutants are emitted directly into the living area. Mitigation efforts for these products are a medium priority. As further research is conducted on these sources, some products may become a high priority for emission reductions.

Within each of the categories shown in Table 6.1, there are high-risk emissions that can and should be addressed through some form of government action. Options for mitigation approaches are suggested below.

7. OPTIONS TO MITIGATE INDOOR AIR POLLUTION

This report has shown that there are many sources of indoor air pollution that produce substantial adverse health effects, result in lost productivity, and require considerable expenditures for health care. Despite these facts, there is no systematic program to improve indoor air quality, there are relatively few regulations or standards to address individual indoor air quality problems, and few resources focused on effectively addressing problems and promoting improvements. Current efforts to address indoor pollution are not commensurate with the scope of the risk to health it poses to Californians.

7.1 GENERAL MITIGATION OPTIONS

In California and under federal law, ambient (mainly outdoor) air quality is protected through a comprehensive system that requires that air quality standards be set and attained for selected pollutants. Pollutants identified as toxic air contaminants must be reduced to the maximum extent feasible. The approach used to reduce toxic air contaminants in ambient air, in which source emissions are reduced without setting enforceable air quality levels, seems most applicable to indoor air. While regulatory action to reduce emissions and exposures would assure reduction of exposure and risk from key sources and be a major component of a new effort to address indoor air, other approaches including public education, product testing and labeling, and setting of maximum exposure guideline levels, should also be part of the mix. The following elements of an indoor air pollution reduction program are recommended for consideration:

1. Create a management system for indoor air quality that establishes and assigns authority and responsibility for assessing indoor health problems, identifies the actions needed to reduce the most significant problems, and sets emissions standards or other requirements that will be effective in reducing the health impacts of indoor sources.
2. Authorize the appropriate state agencies to establish emission limits for building materials, furnishings, combustion appliances, air cleaners, and other indoor pollutant sources that pose excessive risks due to their indoor emissions.
3. Require emissions testing by manufacturers of building materials, furnishings, combustion appliances, consumer products, and other significant source categories, and labeling in language consumers can understand. Implementation of a required test program could prove to be an effective approach, at least in new buildings. A prototype emissions testing program has already been developed for state sustainable building projects: Section 01350 (State of California, 2002) is a testing and assessment protocol developed for building materials and furnishings, and is designed to protect human health. A partial list of products that meet this specification is available at <http://ciwmb.ca.gov/GreenBuilding/Specs/EastEnd/>. However, there is currently no requirement for state agencies or others to use these guideline emission specifications.
4. Make children's health in schools, homes, and care institutions the top priority. Implement the recommendations for schools in the section below.
5. Develop indoor air quality standards or guidelines for homes, schools, offices, and institutional living quarters. Establish a state policy of using indoor air quality "Best Practices" in the design, construction, operation and maintenance of public, commercial,

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school, and institutional buildings. Require IAQ to be fully addressed in government procurement and construction activities. Full commissioning should be required for all new public, commercial, and institutional residential buildings, to assure that they are constructed and operate as intended, and that they provide acceptable indoor air quality. ASHRAE Guidelines (1993, 1996) provide basic guidance for building commissioning. Training and technical assistance should be provided for the private sector to develop the skills and services needed for high-quality building commissioning.

6. Increase efforts to reduce children's exposure to environmental tobacco smoke. Substantial education and outreach efforts to smoking parents and caretakers are needed to inform them of the health dangers of second-hand smoke, and the actions they should be taking to protect children under their care from these dangers.
7. Amend building codes to address indoor air quality. For example, unvented cook stoves, ovens, and combustion appliances should not be allowed in residences.
8. Fund an outreach and education program, especially focused on health professionals, teachers, school facility managers, and related professionals who must be able to identify and remedy indoor air quality problems.
9. Conduct more research on indoor air quality. Many pollutants have not been studied, and synergistic and cumulative health effects are not understood.
10. Fund an Innovative Clean Air Technology program (ICAT) for indoor air quality to foster the development and commercialization of legitimate, cost effective technologies that can improve IAQ. For example, improved ventilation technologies, improved air monitors and assessment tools, and effective low-noise air cleaners are needed.

7.2 SOLUTIONS FOR SCHOOLS

The ARB and DHS recently completed a report on a statewide study entitled *Environmental Health Conditions in California's Portable Classrooms* (ARB/DHS, 2003). The study was conducted at the request of Governor Gray Davis and the State Legislature (AB 2872 Shelley; HSC § 39619.6). It was prompted by concerns regarding reports of mold contamination, inadequate ventilation, poor temperature control, elevated levels of volatile chemicals, and excessive use of some pesticides. The study was funded to help understand the extent of these problems and to determine whether those problems warranted response by the state and/or schools and school districts.

In this study of kindergarten through 12th grade public schools, the major fieldwork was conducted by Research Triangle Institute under contract to ARB. A large, representative sample of classrooms was studied throughout the state, with two portable classrooms and one traditional classroom studied at each school included in the study. Because of the stratified random sample selection process used, and the number of classrooms studied, the results of this study are representative of California classrooms. Stakeholder input was obtained throughout the study.

The results of this comprehensive study have been condensed into a report to the Legislature, and provide important information for state and local decision-makers regarding the degree to which California classrooms provide a safe, healthful, and productive learning environment for

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children. The report summarizes serious conditions identified in the study that need to be addressed at the State and local levels, and discusses options for improving conditions in both portable and traditional classrooms. The key results and recommendations included in the report are summarized below. The recommendations were developed in consultation with relevant state agencies, industries, school officials, and other interested stakeholders.

7.2.1 Problems Identified

The report identifies and addresses a number of environmental problems that were frequently found in classrooms throughout California. These problems were found in both portable (relocatable) and traditional (site-built) classrooms; however, some of the problems were found more frequently in portable classrooms. Government standards and guidelines that are designed to protect children in classrooms and other buildings are essentially lacking. Thus, in this study, results were compared to the most relevant environmental health guidelines and standards available, primarily from professional societies (such as those for ventilation and lighting) and government agencies, such as the Office of Environmental Health Hazard Assessment (OEHHA) and California Department of Industrial Relations (Cal/OSHA). The primary problems found include:

- Inadequate ventilation with outdoor air during 40 percent of class hours, and seriously deficient ventilation 10 percent of the time. This is due largely to teachers turning off HVAC (heating, ventilating, and air-conditioning) systems because of excessive noise and to other factors such as closed outdoor air dampers and inadequate HVAC capacity.
- Temperature and humidity levels outside of professional standards for thermal comfort in about one-fourth of the classrooms.
- Formaldehyde air concentrations above guidelines for preventing acute eye, nose, and throat irritation in about 4 percent of the classrooms; and, nearly all classrooms exceeded formaldehyde guidelines for preventing long-term health effects, including cancer. Elevated formaldehyde is due primarily to the use of formaldehyde-containing building materials and furnishings.
- Noise levels in all classrooms exceeded the national voluntary acoustic standard for unoccupied classrooms of 35 decibels, a somewhat controversial standard that has not been adopted by any California agency. About one-half of the classrooms also exceeded 55 decibels, the level used by many communities in the state for their outdoor nuisance regulations. Excess noise was due primarily to noisy HVAC systems. Noisy lighting and noise from nearby outdoor activities also contributed. Excess noise directly impacts indoor air quality in the classrooms: when teachers turn off the systems due to noise, classrooms become stuffy and indoor pollutant levels rise.
- Lead, arsenic, and numerous pesticide residues in classroom floor dust; these residues are a concern because they can be inhaled, ingested, or absorbed through the skin (pesticides) by children, especially very young children who sit on the floor and put their hands in their mouths.
- Obvious mold in about 3 percent of classrooms; water stains, excess wall moisture, and other potential mold indicators in about one-third of classrooms; musty odors reported by

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69 percent of teachers. These conditions are often attributable to inadequate maintenance.

- Lighting was inadequate in about one-third of the classrooms.

7.2.2 Report Recommendations

To address the breadth of problems identified and the many actions needed at all levels to resolve them, a total of 16 recommendations are discussed in the report to the Legislature. These are presented in two groups in the report: Group 1 includes high priority, high benefit actions that can be achieved at relatively low cost and should be accomplished in the near term, while Group 2 recommendations, also a priority, will require a longer timeframe and/or more substantial resources to accomplish. The recommendations fall into four general approaches needed to remedy and prevent the problems seen. The state should:

- Direct and assist schools to comply with state regulations, especially workplace regulations (Cal/OSHA) related to building operation and maintenance.
- Develop and promote “Best Practices” for design, construction, operation, and maintenance of school facilities.
- Improve support (both funding and training) for school facilities and staff.
- Establish guidelines and standards for school environmental health that are protective of children.

The specific Group I recommendations are:

- Schools, districts, and the state should assure that all school buildings meet all relevant state regulations, especially the Cal/OSHA workplace regulations regarding ventilation, sanitation and water intrusion, and illness and injury prevention. Many schools do not meet Cal/OSHA environmental health requirements.
- Schools and school districts should conduct “self-assessments of basic health and safety conditions, similar to the Facility Self-inspection Program undertaken by the Los Angeles Unified School District (LAUSD). Checklists are available on the web.
- The state should require schools to develop indoor environmental quality management plans. The U.S. Environmental Protection Agency’s *IAQ Tools for Schools Program* provides guidance and free kits to accomplish this. The kit is available free of charge at <http://www.U.S. EPA.gov/iaq/schools/>.
- The state should establish a policy to incorporate “Best Practices” into the design, construction, operations, and maintenance of California schools, especially the measures developed by the Collaborative for High Performance Schools (CHPS). The CHPS *Best Practices Manual* at <http://www.chps.net/> provides broad guidance for measures that will improve schools while also saving energy and reducing long-term costs.

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- State-level review by the Division of the State Architect of the designs for new schools should be expanded to include elements such as ventilation systems and building materials in addition to current elements such as fire and life-safety provisions.
- Portable classrooms (and traditional classrooms) should be sited correctly, away from busy roadways and with proper drainage.
- The state should implement an interim requirement for new classrooms of a maximum noise level of 45 decibels, unoccupied, until a specially-convened task force can determine an appropriate level for California schools (see below).

Group 2 recommendations specify that:

- The state and school districts should assure stable, long-term funding mechanisms and sources for both construction and preventive maintenance; currently, funding fluctuates from year to year, especially for the Deferred Maintenance Program.
- The state should develop and offer focused training programs for school facility managers, custodial staff, and teachers, in cooperation with interested organizations; those closest to the classrooms often are not aware of current “best practices” for operation and maintenance of classrooms. A concerted, ongoing training program could go far to improve conditions in classrooms.
- Integrated Pest Management Programs should be implemented at all schools.
- Older portable classrooms should be retired when they become unserviceable or do not provide an adequate learning environment for children.
- The state and school districts should develop and require full new building commissioning procedures.
- The state should improve its database of school facilities: currently, there is no complete database on the condition, location, or even number, of school buildings.
- The state should convene a task force of experts to develop a California indoor noise guideline or standard for K-12 schools.
- The state should develop chemical exposure guidelines or standards for classrooms that are protective of children and teachers.
- Portable classrooms should be re-designed from the ground up. Several groups are producing new prototypes that use an integrated “whole building” approach; these should be supported through the demonstration phase to evaluate design changes that provide substantive improvements over older portables.

Some actions have already been taken to begin to address these problems; however, they constitute only a first step toward realizing actual improvements in school conditions. Only a small percentage of schools and districts have actively pursued the many tools that are readily available to them to help improve the school environment. The CHPS' *Best Practices Manual*, U.S. EPA's *IAQ Tools for Schools Kits*, and the LAUSD's “Safe School Inspection Guidebook”

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are all available on the Internet free of charge, yet the number of California schools utilizing each of these tools is small. A proactive effort to implement the recommendations of the report is needed.

The complete *Report to the Legislature on Environmental Health Conditions in California's Portable Classrooms* is available at ARB's website at <http://www.arb.ca.gov/research/indoor/pcs/pcs.htm>.

8. SUMMARY

Indoor air pollution poses a significant health risk to Californians, in addition to the known risks from outdoor pollution. Indoor pollutant sources emit gases and particles known to trigger asthma attacks and cause cancer, heart and lung disease, and immediate irritant and neurological effects such as eye and throat irritation and headache. Indoor pollution has repeatedly been ranked in the “High Risk” categories in both federal and state comparative risk projects. Some of the known risks in California include:

- ◆ Indoor air pollution includes many asthma triggers. It has been implicated by national scientists as a factor in the serious increase in asthma observed in recent decades.
- ◆ At least 230 excess cancers per year are estimated to occur in California due to indoor carcinogens from residential and consumer sources, such as formaldehyde. This approaches the 260 estimated excess cancer cases per year from diesel exhaust and equals about two-thirds of the total outdoor cancer burden. Environmental tobacco smoke also adds a significant cancer risk.
- ◆ Each year, accidental carbon monoxide poisoning from indoor sources causes about 20-26 accidental deaths, hundreds of avoidable emergency room visits, and hundreds to thousands of cases of avoidable illness.
- ◆ Many VOCs, especially formaldehyde, are typically found at elevated levels indoors, and those levels often exceed health-based guideline levels, such as acceptable cancer risk levels.
- ◆ Indoor sources of PM may be partly responsible for the large numbers of premature deaths, hospitalizations, emergency room visits, and increased respiratory disease associated with PM exposures in California each year, and may contribute to these effects beyond the levels quantified in the outdoor epidemiology studies.

Indoor pollutants significantly impact people’s health because there are many indoor sources of pollutants, which often result in elevated indoor concentrations. Additionally, Californians, like others, spend about 87% of their time indoors, on average. Infants and young children spend the most time indoors, most of it in their homes. Based on these factors, one scientist has calculated that indoor emissions are about 1000 times more likely to be breathed than outdoor emissions.

Indoor pollution is estimated to cost California’s economy \$35 billion each year due to medical costs, lost worker productivity, loss of life, and related factors. This estimate is derived from only partial costs of cancer, respiratory disease, cardiovascular disease, and sick building symptoms. This is believed to be an underestimate; the total cost is likely much higher.

Despite the high health and economic costs of indoor pollution, no state or federal agency has explicit authority to regulate indoor sources of pollution to protect building occupants from harmful exposures. Indoor pollution remains the only major environmental health problem that does not have the benefit of a focused risk reduction program.

Many actions could be taken to reduce indoor pollution at relatively low cost that would reap large health benefits. Options for mitigation include the development of emission limitations for building materials, furnishings, consumer products, and appliances; improvements in building codes; focused outreach and education programs; improved technologies; and adherence to current “Best Practices” and other guidelines. Substitute products are available for many products and materials that currently emit high levels of pollutants; thus, emission reductions

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should be achievable in many products with little impact on consumers and homeowners. This would go far to improve indoor air quality in schools and homes, which require a focused effort because little has been done to improve those indoor environments, and the most sensitive members of the population—children, the elderly, and the infirm—spend most of their time there. Focused public education, improved appliance standards, and modified building codes would also foster improved indoor air quality in homes and schools. Following current “Best Practices” in design, construction, operation and maintenance is a non-regulatory approach that could yield large gains in indoor health in all types of buildings. Increased effort to publicize and enforce existing Cal/OSHA regulations in schools and non-industrial workplaces would also go far to reduce indoor pollution in those environments; those regulations are in place, but are often not followed.

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10. GLOSSARY

TERM	DEFINITION
Active/Passive Sampling	Active sampling depends on a mechanical process like pumping to collect the sample at a known rate; this is used for VOC and aldehyde sample collection. Passive sampling involves non-mechanical processes, usually diffusion, in which the air is sampled at whatever rate it passes across a badge surface or tube opening.
Air Changes per Hour	ACH, the volume of air moved in one hour. One air change per hour in a room, home, or building means that the equivalent of the volume of air in that space will be replaced in one hour.
Air Cleaners	These are devices designed to remove pollutants from a room. Air cleaners can be portable, or part of a central air system. Air cleaners can be mechanical, employing a filter to remove pollutants, or electronic using a small electrical charge to collect particles from air pulled through a device.
Air Fresheners	These devices are promoted to neutralize odors rather than remove pollutants. Products often emit a fragrance which diffuses into the air. Some styles can spray fragrances and be programmed for fragrance dispersion.
Air Flow Rate	The rate at which air moves into a space. Expressed in units of air changes per hour or cubic feet per minute.
Air Handling Unit	HVAC (heating, ventilation and air conditioning) unit. Refers to equipment that includes a blower or fan, heating and/or cooling coils, and related equipment such as controls, condensate drain pans, and air filters. Does not include ductwork, registers, or grilles, or boilers and chillers.
Allergen	A chemical or biological substance (e.g., pollen, animal dander, or house dust mite proteins) that induces an allergic response, characterized by hypersensitivity.
Ambient Air Quality Standards (AAQS)	State (ARB) and federal (U.S. EPA) enforceable regulations designed to protect the public from the harmful effects of traditional pollutants in outdoor air.
Asthma	A chronic disease of lung tissue which involves inflamed airways, breathing difficulty, and an increased sensitivity to allergens and contaminants in the air.
Biological Contaminants	Agents derived from or that are living organisms (e.g., viruses, bacteria, fungi, and mammal, arthropod, and bird antigens) that

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	can be inhaled and can cause many types of health effects including allergic reactions, respiratory disorders, hypersensitivity diseases, and infectious diseases. Also referred to as biological agents.
Comfort Measures	Factors that determine human perception of thermal comfort, including temperature, relative humidity, and draft
Commissioning	Testing the important building systems before occupancy to ensure that they operate the way the designers expect and that they serve the needs of the building occupants. Systems that can be tested include HVAC, electrical, energy management, plumbing, etc.
Fungi	A group of organisms that lack chlorophyll, including molds, mildews, yeasts, mushrooms.
Integrated Pest Management (IPM)	A pest management strategy that focuses on long-term prevention or suppression of pest problems through a combination of techniques such as monitoring for pest presence and establishing treatment threshold levels, using non-chemical practices to make the habitat less conducive to pest development, improving sanitation, and employing mechanical and physical controls. Pesticides that pose the least possible hazard and are effective in a manner that minimizes risks to people, property, and the environment, are used only after careful monitoring indicates they are needed according to pre-established guidelines and treatment thresholds.
Micron	A unit of length equal to one millionth of a meter; a micrometer.
Microorganism	A microscopic organism, usually a bacterium, fungus, or protozoan.
Natural Ventilation	The movement of outdoor air into a space through intentionally provided openings, such as windows and doors, or through non-mechanical ventilators, by wind, air pressure differences, or other natural, non-mechanical means.
Permissible Exposure Limits (PELs)	Enforceable pollutant exposure limits determined by OSHA that are designed to protect healthy adult workers in industrial environments from adverse health effects associated with pollutant exposure. None of these limits are targeted toward protecting children.
Pesticides	A pesticide is any substance or mixture of substances intended to prevent, destroy, repel, or mitigate any pest. Though often misunderstood to refer only to insecticides, the term pesticide also applies to herbicides, fungicides, disinfectants, and antimicrobials. Under U. S. law, a pesticide is also any substance or mixture of

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	substances intended for use as a plant regulator, defoliant, or desiccant.
Polycyclic Aromatic Hydrocarbons (PAHs)	A class of stable organic molecules comprised of only carbon and hydrogen. They are a common product of combustion from automobiles, airplanes, woodburning, cigarettes, and some types of cooking. Many of these molecules are highly carcinogenic and very common.
Portable Classrooms	Classrooms that are designed and constructed to be moveable and transportable over public streets, also known as temporary or relocatable classrooms.
Quality Control (QC)	Internal checks on the operation of sample collection and/or sample analysis. Methods for determining the operation include blanks, spiked samples, flow checks, and duplicate samples. QC measures can be used to determine accuracy, bias, and precision of the data reported.
Real-time Monitoring	This type of environmental measurement gives instantaneous information at the point of sampling; measurements are recorded as often as every minute, every second, or in fractions of a second.
Reference Exposure Level (REL)	The concentration level at or below which no adverse health effects are anticipated for a specified exposure duration. RELs are based on the most relevant, adverse health effect reported in the medical and toxicological literature for the population group known to be most sensitive to the chemical. RELs are designed to protect the most sensitive individuals in the population by the inclusion of margins of safety. Since margins of safety are incorporated to address data gaps and uncertainties, exceeding the REL does not automatically indicate an adverse health impact will occur. OEHHA provides acute (1-hour) and chronic (lifetime, non-cancer), RELs for a number of chemicals, and has developed an 8-hour "indoor" REL for formaldehyde.
Relative Humidity	The measure of moisture in the atmosphere, expressed as a percent of the maximum moisture the air can hold at a given temperature.
Return Air	Air removed from a space by the HVAC system to be recirculated or exhausted.
Sick Building Syndrome	A set of symptoms (including headache, fatigue, and eye irritation) typically affecting workers in modern airtight office buildings, believed to be caused by indoor pollutants (such as formaldehyde fumes or microorganisms).
Supply Air	Air delivered to the conditioned space by the HVAC system and used for ventilation, heating, cooling, humidification, or

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	dehumidification. It is usually a combination of outdoor air and return air.
Traditional Classrooms	Classrooms in permanent, site-built school buildings.
Variable Air Volume System	Air handling system that conditions the air to a temperature using a varying amount of outside airflow based essentially on the outdoor temperature.
Ventilation	The process of intentionally supplying and removing air by natural or mechanical means to and from any space.
Volatile Organic Compounds (VOCs)	Compounds that evaporate quickly from the many housekeeping, maintenance, and building products made with organic chemicals. These compounds are released from products that are being used and that are in storage. Many are carcinogenic, neurotoxins, or mucous membrane irritants.