6 Exposure to Carbon Monoxide

Peter G. Flachsbart
University of Hawai'i at Manoa

CONTENTS

6.1 SYNOPSIS

Incomplete combustion of carbonaceous fuels (i.e., fuels with carbon atoms) can produce significant quantities of carbon monoxide (CO). Exposure to CO occurs during a variety of daily

activities such as traveling by motor vehicle in traffic or cooking food over an unvented gas range. Fortunately, reducing CO exposures has been one of the "… greatest success stories in air-pollution control," according to a report published by the National Research Council in 2003. Much of that success is due to the adoption in 1968 of nationwide emission controls on new cars, and to promulgation in 1970 of the National Ambient Air Quality Standards (NAAQS) for CO and several other "criteria" air pollutants. In spite of that success, many people die or suffer the ill effects of high CO exposure every year. In fact, CO is the only regulated air pollutant that appears on death certificates. Accordingly, this chapter first summarizes the principal sources and health effects of CO. It then describes key studies of CO exposure over the last 40 years to show how the goals and methods of these studies have evolved over time. Studies of CO exposure in the 1960s and 1970s essentially pioneered the field of exposure analysis. The earliest studies found that CO concentrations on congested roadways and busy intersections in downtown areas typically exceeded ambient CO levels measured at fixed-site monitors. The U.S. Environmental Protection Agency (USEPA) relies on these monitors to determine compliance with the NAAQS. The chapter reveals typical concentrations of CO that people encounter in their daily lives and identifies factors that affect or contribute to CO exposures as a person performs his or her daily activities. The chapter shows how policies and programs of the Clean Air Act have affected trends in CO exposure over time. The chapter concludes that CO exposure studies are essential for identifying health risks to human populations, for setting and reviewing air quality standards, and for evaluating emission control policies and programs. The chapter recommends that studies of CO exposure are particularly applicable to developing countries that have rapidly growing motor vehicle populations, congested streets and confined spaces in urban areas, and nascent motor vehicle emission control programs.

6.2 INTRODUCTION

The National Research Council (NRC) in Washington, DC recently issued a report titled *Managing Carbon Monoxide Pollution in Meteorological and Topographical Problem Areas*. The report concluded: "CO control has been one of the greatest success stories in air-pollution control. As a result, the focus of United States air quality management has shifted to characterizing and controlling other pollutants, such as tropospheric ozone, fine particulate matter ($PM_{2.5}$), and air toxics." (NRC 2003, p. 149) As evidence for this conclusion, the NRC acknowledged that the number of monitoring stations showing violations of the National Ambient Air Quality Standards (NAAQS) for CO had fallen significantly from the early 1970s when CO monitoring became widespread. Many of the remaining violations occur in areas with meteorological or topographical handicaps. For example, CO violations in Fairbanks, Alaska, have been attributed to stagnant air masses during winter. The atmosphere is more likely to be stable during winter, because there is less solar heating and more frequent ground-level temperature inversions. Other contributing factors are low wind speeds and mountains that hinder dispersion of air pollutants. However, violations are occurring less frequently even in areas with these natural handicaps (NRC 2003).

If most Americans are no longer exposed to unhealthy CO levels, then why study CO exposure? One reason is that CO studies pioneered the field of exposure analysis. The earliest CO exposure studies, which date back to the mid-1960s, focused on tailpipe emissions on urban expressways, because motor vehicles represented the highest percentage of total CO emissions. These studies found that CO exposures on congested roadways and busy intersections typically exceeded ambient levels of CO measured at fixed-site monitors. This problem receded when automakers equipped motor vehicles with catalytic converters to satisfy tailpipe emission standards. Nevertheless, many Americans are still exposed to hazardous and sometimes fatal CO concentrations in their daily activities. Second, CO can be viewed as an indicator of other types of roadway emissions that are relatively stable in the atmosphere. For example, CO concentrations are highly correlated with concentrations of air pollutants such as benzene (a known carcinogen),

black carbon, and certain ultra-fine particles (NRC 2003). CO is not an indicator of reactive air pollutants, such as hydrocarbons and nitrogen oxides, which are also emitted by motor vehicles. Third, personal exposure to CO can easily be measured using relatively inexpensive and reliable portable monitors that run on batteries. Certain monitors are capable of very precise and accurate CO measurements, and can store data electronically for later analysis. Finally, CO exposure studies are relevant to developing countries that have seen rapid growth in the use of motor vehicles. Many developing countries mandate less stringent vehicular emission standards than are found in North America, Japan, or Europe. Consequently, these countries have motor vehicle fleets with outdated emission controls.

6.3 SOURCES OF CARBON MONOXIDE

Carbon monoxide is a gaseous by-product that results from incomplete combustion of fuels (e.g., oil, natural gas, coal, kerosene, and wood) and other materials (e.g., tobacco products) that contain carbon atoms. According to the national inventory of air pollutant emissions compiled annually by the U.S. Environmental Protection Agency (USEPA), transportation sources in the United States accounted for nearly 70% of total CO emissions in 2000. Fuel combustion, industrial processes, and miscellaneous sources comprised the remaining 30%. Transportation sources include both on-road motor vehicles (e.g., cars and trucks) and non-road engines and vehicles (e.g., aircraft, boats, locomotives, recreational vehicles, and gasoline-powered lawnmowers). Of particular importance are those sources (such as cars, trucks, and lawnmowers) that release their emissions in close proximity to human receptors (Colvile et al. 2001).

The USEPA's annual emission inventory shows that the relative shares of on-road and non-road sources have shifted during the last two decades. The share of total emissions from on-road vehicles fell from 66.5% in 1980 to 44.3% in 2000, while the share from non-road vehicles increased from 12.3% in 1980 to 25.6% in 2000. This shift can be attributed to tailpipe exhaust emission standards, which have affected all new cars sold in the United States since 1968, and the fact that non-road sources are largely unregulated (USEPA 2003). Although CO emission rates of new cars have fallen over time, due to tighter emission standards imposed by the Clean Air Act (CAA), the number of vehicles and miles driven per vehicle have both been increasing due to population growth and urban sprawl (TRB 1995). Total miles of travel by all types of motor vehicles increased three times faster than population growth in the United States between 1980 and 2000 (Downs 2004).

The USEPA and the State of California have separate models to inventory motor vehicle emissions, because California is allowed to set its own motor vehicle emission standards under the CAA. Both models show that CO emission rates climb substantially when average speeds fall below 15 miles per hour. Since these low speeds often occur during periods of severe traffic congestion, many CO exposure studies focus on commuting activities, particularly during peak periods of travel. Other studies focus on "cold starts" that occur after vehicles have been parked for several hours. Cold starts may elevate CO exposures in homes (Akland et al. 1985) and office buildings (Flachsbart and Ott 1986) with attached garages. After several minutes the engine reaches higher temperatures and CO emissions begin to subside. Higher tailpipe emissions also occur when the driver accelerates the vehicle, runs its air conditioning system, or climbs a hill, because more fuel is needed to achieve extra power. By comparison, diesels emit less CO because excess air is used in the combustion process. CO emissions during conditions of severe fuel enrichment, which are essentially unregulated, can account for 40% of a typical trip's total CO emissions, even though these conditions prevail for only 2% of trip time (Faiz, Weaver, and Walsh 1996). Higher CO emissions also occur if the driver defers vehicle maintenance and repairs, tampers with the catalytic converter, or uses leaded fuel, which renders the converter ineffective (NRC 2000). These actions can spike the exposure of pedestrians, cyclists, and motorists if they are exposed to malfunctioning vehicles on city streets and roads. In 1973, the United States began to phase out lead from gasoline and banned

lead additives in commercial gasoline after December 31, 1995. The remaining use of leaded gasoline in U.S. motor vehicles occurs predominantly in rural areas (Walsh 1996). Last but not least, defective exhaust systems can contaminate the passenger compartments of motor vehicles (Amiro 1969) and sustained-use vehicles such as buses, taxicabs, and police cars (Ziskind et al. 1981), and lead to accidental CO poisoning of passengers in the back of pickup trucks (Hampson and Norkool 1992).

6.4 HEALTH EFFECTS OF CARBON MONOXIDE

CO molecules, which have no color, odor, or taste, enter the body through normal breathing (i.e., inhalation exposure). In the lung, the CO molecule passes into the bloodstream through the alveolar and capillary membranes of the lung and blood vessels, respectively. Once in the blood, CO competes with oxygen for attachment to iron sites in red blood cells (hemoglobin). The attraction of hemoglobin (Hb) to CO is about 250 times stronger than it is for oxygen (Burr 2000). The chemical bond between CO and Hb is known as carboxyhemoglobin (COHb). COHb not only reduces the amount of oxygen that can be delivered to organs and tissues, a condition known as hypoxia, it also interferes with the release of oxygen from the blood. This interference occurs because COHb strengthens the bond between hemoglobin and oxygen in the blood. The percentage of COHb in the blood is thus a dosage indicator of CO exposure and a physiological marker that can be linked to various health effects of CO exposure. The percentage of COHb in a person's blood depends not only on the duration of one's exposure to CO concentrations in the air, but also on one's breathing rate, lung capacity, health status, and metabolism. Because of the high affinity of CO and Hb, the elimination of COHb from the body can take between 2 and 6.5 hours depending on the initial level of COHb in the blood (USEPA 2000). Because the elimination of COHb from the body is a slow process, continuous exposure to even low concentrations of CO may increase COHb (Godish 2004).

Everyone on Earth is exposed to background CO concentrations in the ambient air on the order of 120 parts per billion (ppb) by volume in the Northern Hemisphere and about 40 ppb in the Southern Hemisphere. This difference occurs because the Northern Hemisphere is more developed than the Southern Hemisphere and their respective atmospheres are not completely mixed. In addition, metabolism of heme in the blood produces an endogenous level of CO that occurs naturally in the body. As a result, the body of a nonsmoker has a baseline or residual COHb level in the range of 0.3–0.7% and an endogenous breath CO level of 1–2 ppm. This level varies from one person to another due to human variation in basal metabolisms and other metabolic factors (USEPA 2000). Besides exogenous sources of CO, metabolism of many drugs, solvents (e.g., methylene chloride), and other compounds can also elevate COHb levels above baseline levels through endogenous production of CO. If exposure to drugs and solvents continues for several hours, it can prolong cardiovascular stress caused by excess COHb in the blood. The maximum COHb level from endogenous CO production can last up to twice as long as comparable COHb levels caused by exposures to exogenous CO (Wilcosky and Simonsen 1991; ATSDR 1993).

The percentage of COHb in blood can be related to the breath concentration of CO by simultaneously sampling a person's blood for COHb and his or her end-tidal breath for CO concentration. Coburn, Forster, and Kane (1965) developed an equation to predict the percentage of blood COHb in nonsmokers, based on external CO exposure and assumptions about breathing rate, altitude, blood volume, hemoglobin level, lung diffusivity, and endogenous rate of CO production. For example, a nonsmoking adult engaged in light exercise can expect to have COHb levels below 2–3% if exposed to CO levels of less than 25–50 ppm for 1 hour or 4–7% if the same exposure lasted for 8 hours. Since endogenous COHb leads to a breath CO of about 1–2 ppm, a measured breath CO level of 10 ppm corresponds roughly to an exogenous exposure of 9 ppm (under steady-state conditions).

Burr (2000) describes acute, subacute, chronic, and long-term cardiovascular effects of CO exposure in healthy and diseased populations. Severe oxygen deprivation first affects the brain and then the heart. Patients with heart disease, anemia, emphysema or other lung disease are more susceptible to the harmful effects of CO because their bodies are unable to compensate for oxygen deficiencies. Healthy pregnant women, young children, the elderly, and tobacco smokers are more likely to be adversely affected by CO exposure than are other people. COHb levels of 2.4% or higher can induce chest pain in patients with angina, and levels of 2.3–4.3% can affect the performance of people competing in athletic events (USEPA 2000). COHb levels below 5% can result from exposure to high CO concentrations in the ambient air. People working in certain occupations (e.g., chainsaw gas tool operators, firefighters, garage mechanics, forklift operators) can have COHb levels above 5%, which can affect visual perception and learning ability. Baseline COHb concentrations in smokers average around 4% and range from 3–8% for people who smoke one to two packs per day. COHb levels between 5% and 20% can affect vigilance and diminish hand-eye coordination, which can affect a person's ability to drive a vehicle in traffic. Dizziness, fainting and fatigue can occur at COHb levels of 20% (USEPA 2000). Coma, convulsions and death may occur if COHb levels exceed 60% (Burr 2000). A more complete discussion of the health effects of CO appears in reports by Jain (1990), Penney (1996), and Ernst and Zibrak (1998).

6.5 EARLY STUDIES OF CO EXPOSURE

The commercial districts of cities generate large volumes of motor vehicle traffic during business hours. Vehicles often circulate at low speeds with frequent stops and starts at intersections. This traffic pattern can produce relatively high CO emissions particularly during peak travel periods. Tailpipe exhaust gases rise in the atmosphere, because they are warmer and less dense than air. CO spreads through the atmosphere very easily, because it has a lighter molecular weight than air. In open areas, CO concentrations fall rapidly with greater wind speed and distance from sources. Higher CO concentrations may occur in street canyons, however, because tall buildings affect wind patterns. These facts may explain why early studies of exposure focused on activities such as driving in traffic, while other studies measured roadside concentrations attributable to different levels of traffic in urban areas.

6.5.1 Surveys of Exposure while Driving in Traffic

Until the early 1950s, most automotive engineers thought that motor vehicle emissions played a minor role in air pollution. That thinking began to change in November 1950, when Professor Arie J. Haagen-Smit announced results of his laboratory experiments at the California Institute of Technology (Cal Tech). Haagen-Smit's experiments showed how sunlight converted certain gases emitted by motor vehicles and oil refineries, namely oxides of nitrogen and volatile hydrocarbons, into a secondary air pollutant known as ozone (O_3) (Doyle 2000).

Compared to Haagen-Smit's now famous laboratory experiments on ozone formation, his field surveys of CO concentrations while driving in Los Angeles are not as well known. In these surveys, he equipped the passenger cabin of his car with a prototype, continuously recording CO analyzer developed by Dr. P. Hersch. Haagen-Smit placed the instrument next to the dashboard of his car and ran a glass tube from the instrument's CO sensor to the *outside* air through the front window. (The outside measurement can be a good approximation of exposure inside the car, if there is a rapid exchange of air between the passenger cabin and exterior environment.) He made eight 30-mile round-trips, including travel on suburban streets in Pasadena, portions of two interstate freeways, and surface streets near downtown Los Angeles. CO concentrations outside the vehicle averaged 37 ppm and ranged from 23–58 ppm for trips of 40–115 minutes. Average CO concentrations ranged from 38–72 ppm when he drove under 20 miles per hour (mph) in heavy traffic. Ambient CO levels at fixed-site monitors were above 20 ppm during summer and above 30 ppm

during the winter season on 50% of days monitored between 1960 and 1964. Thus, Haagen-Smit appears to be the first analyst to observe that CO concentrations on freeways exceeded urban ambient levels, and that these concentrations rose in heavy traffic moving at slow speeds (Haagen-Smit 1966).

Field surveys similar to Haagen-Smit's pioneering effort in Los Angeles were performed shortly thereafter in many U.S. cities. For example, Brice and Roesler (1966) used Mylar™ bags to measure CO, as well as hydrocarbon concentrations, *inside* vehicles moving in traffic in five U.S. cities (Chicago, Cincinnati, Denver, St. Louis, and Washington, DC) between 7 A.M. and 7 P.M. Air samples were also collected at points alongside traffic routes in Chicago, Washington, DC, and Philadelphia. The average CO concentration measured for trips of 20–30 minutes on arterial streets and expressways ranged from 21 ppm in Cincinnati to 40 ppm in Denver. The average CO levels on high-density traffic routes were 1.3–6.8 times the corresponding CO concentrations measured at fixed monitoring stations. The study concluded that ambient monitoring stations significantly underestimated the pollutant exposures of commuters and those working long hours in traffic (e.g., bus drivers, taxicab drivers, policemen, etc.). Besides revealing inadequacies of ambient monitoring, the study provided a significant baseline for comparing the results of later studies of commuter exposure.

At about the same time as the Brice and Roesler study, Lynn et al. (1967) measured commuter CO and hydrocarbon exposures in 14 American cities between April 1966 and June 1967. They used a mobile sampling van and trailer to collect exposures during 30-minute trips. Lynn et al. (1967) attributed variation in the ratio of commuter exposure to ambient concentrations to variation in the location of monitoring stations. After combining and reanalyzing the data for all 14 cities, Ott, Switzer, and Willits (1993a) reported that the average CO concentrations inside test vehicles varied from 28 ppm on routes through city centers to 22 ppm on arterials and 18 ppm on expressways. The variation in exposure by route could be explained by variation in traffic volume and vehicle speed on each route.

6.5.2 Surveys of CO Concentrations on Streets and Sidewalks

Early studies showed that CO emissions and roadside concentrations can increase dramatically whenever motor vehicles form a queue at street intersections. Therefore, the severity of concentrations may partly depend on how much traffic is handled by an intersection and one's distance from it. To test this hypothesis, Ramsey (1966) surveyed 50 intersections over a 6-month period in Dayton, Ohio. Concentrations were 56.1 ± 18.4 ppm (mean ± one standard deviation) for heavy traffic, 31.4 ± 31.5 ppm for moderate traffic, and 15.3 ± 10.2 ppm for light traffic. Ramsey also reported that concentrations were greater at intersections along major arteries somewhat removed from downtown Dayton, and that their mean concentration was 3.4 times the mean concentration of intersections a block away and perpendicular to the axis of the arterial. In a later study, Claggett, Shrock, and Noll (1981) found that CO concentrations at intersections with signals were higher than those measured near freeways that had two to three times greater traffic volumes.

Colucci and Begeman (1969) found that outdoor mean CO concentrations were usually the highest but varied the most (3.5–10 ppm) in commercial areas of Detroit, New York, and Los Angeles. By comparison, outdoor CO levels varied less near freeways (6–8 ppm) and were lowest in residential areas (2.5–5.5 ppm). They also found that outdoor CO concentrations in New York and Los Angeles tended to be higher during summer and autumn when average wind speeds were generally lower. Later studies looked at how CO concentrations varied with distance from sources for a given location. For example, Besner and Atkins (1970) reported that CO concentrations declined with greater distance from an expressway in an open area of Austin, Texas. At 16 feet from the road, CO concentrations ranged from 3.4–6.0 ppm, while at 95 feet concentrations ranged from 2.4 to 3.9 ppm.

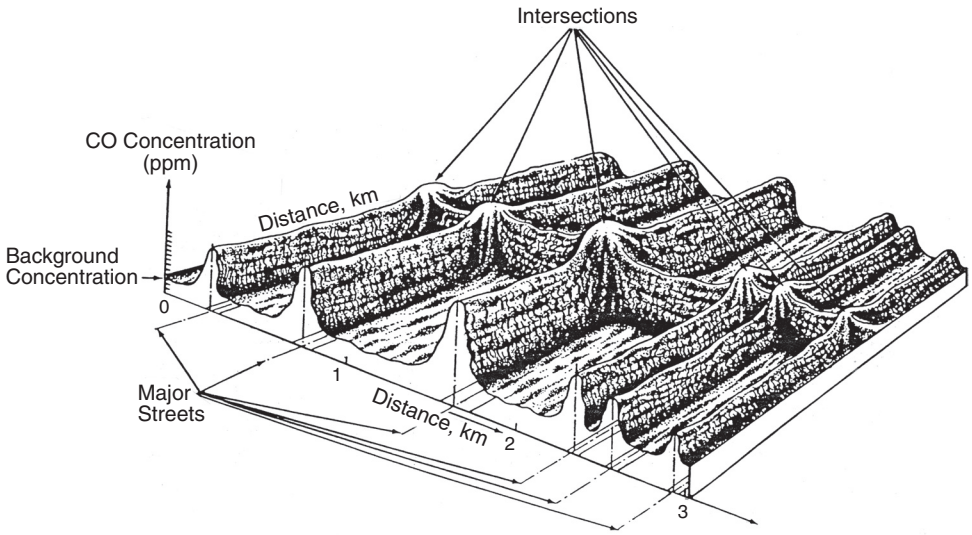


FIGURE 6.1 Model of the spatial variation of CO concentrations at breathing level in an urban area. (From Ott, 1982. With permission from Elsevier.)

These early studies supported the view that CO concentrations at breathing levels were higher in commercial districts of cities, and at intersections and along city streets, but were lower as one moved away from traffic. Figure 6.1 depicts this view of CO concentrations for a portion of a city, based on what was known about the spatial distribution of CO concentrations in the 1970s (Ott 1982). The vertical scale of this figure, which represents CO concentration, would have to be divided by three or four to make the concentrations shown in the figure relevant to the present. The figure also illustrates the superposition principle of CO exposure measurement. This principle holds that the observed CO concentration at a given point in time and space consists of the sum of microenvironmental and background components.

6.6 THE CLEAN AIR ACT AMENDMENTS OF 1970

Most of the early exposure studies were cited in a document titled *Air Quality Criteria for Carbon Monoxide*, published in March 1970 by the National Air Pollution Control Administration (NAPCA) of the U.S. Department of Health, Education, and Welfare (NAPCA 1970). NAPCA, along with several other governmental agencies, became the U.S. Environmental Protection Agency (USEPA) on July 9, 1970, when President Richard Nixon issued an executive order creating the agency. Another significant event of 1970 was congressional approval of amendments to the Clean Air Act (CAA), which required the USEPA to promulgate National Ambient Air Quality Standards (NAAQS) for several air pollutants including CO. Exposure studies frequently refer to the NAAQS for guidance on allowable limits of exposure. The NAAQS include a set of primary standards to protect public health and secondary standards to protect public welfare, such as crop damage from ozone. The NAAQS apply to "criteria" pollutants, because the USEPA must issue air quality criteria for pollutants that may reasonably endanger public health or welfare. Accordingly, the NAAQS set maximum permissible concentrations in ambient air for specified averaging times. The standards include a safety margin to reflect uncertainties in the science of effects of air pollution.

On April 30, 1971, the USEPA promulgated identical primary and secondary NAAQS for CO. In 1985, the USEPA rescinded the secondary standards, because there was no evidence of adverse effects on public welfare due to ambient CO levels. However, the USEPA retained the primary

standards, which have remained since 1971 at 9 ppm for an 8-hour average and 35 ppm for a 1-hour average. These standards are designed to keep COHb levels below 2% in the blood of the general public, including probable high-risk groups. These groups include the elderly; pregnant women; fetuses; young infants; and those suffering from anemia or certain other blood, cardiovascular, or respiratory diseases. People at greatest risk from exposures to ambient CO levels are those with coronary artery disease. These people may suffer chest pain during exercise when exposed to COHb levels ≥2.4% (USEPA 2000). Although annual death rates from heart disease have been declining since 1980, heart disease is still the nation's leading cause of death (Arias et al. 2003). Coronary artery disease reduces a person's circulatory capacity, which is particularly critical during exercise when muscles need more oxygen.

In accordance with the CAA, the USEPA must determine whether or not a community complies with the NAAQS based on measurements of ambient air quality made by a nationwide network of fixed-site monitoring stations. This network consists of state and local air monitoring stations (SLAMS), which send data to USEPA's Aerometric Information Retrieval System (now Air Quality System) within 6 months of acquisition (Blumenthal 2005). Several stations within the SLAMS network belong to a network of national air monitoring stations (NAMS) to enable national assessments of air quality. A station is in non-attainment of the NAAQS for CO if it records an ambient concentration that exceeds either the 1-hour or 8-hour standard more than once per year. These stations use the non-dispersive infrared (NDIR) method to measure ambient CO concentrations. Monitoring instruments based on NDIR are large, complex, and expensive, and require an air-conditioned facility for the production of accurate and reliable data. Because NDIR monitors are not portable, they cannot be used to measure CO exposure as a person performs routine daily activities.

The CAA amendments also mandated stringent automobile emission standards to assist in attainment of the NAAQS. When the NAAQS were adopted, highway vehicles accounted for substantial percentages of total national emissions of CO, hydrocarbons, and nitrogen oxides (NO_x). Compared to emissions from new cars sold during the 1970 model year, the CAA amendments required automakers to produce passenger cars that achieved 90% reductions in CO and hydrocarbon emissions by the 1975 model year. By the 1976 model year, manufacturers had to achieve a 90% rollback in NO_x emissions over 1971 levels (Ortolano 1984). Studies persuaded Congress that these emission standards would accomplish ambient air quality goals by 1990 in those areas that had the worst air pollution in the nation (Grad et al. 1975).

Automobile manufacturers viewed the emission standards as "technology forcing," because the technology to achieve them did not exist when the standards were adopted (Ortolano 1984). During the 1970s, the industry's efforts to reduce vehicle emissions were achieved through the use of increasingly elaborate and sophisticated technologies (e.g., the three-way catalytic converter). By the 1981 model year, the CO emission rate of new passenger cars was below the pre-control level (prior to 1968) by 96% (Johnson 1988).

6.7 LIMITATIONS OF FIXED-SITE MONITORS

Several pioneering studies during the 1970s revealed the inability of fixed-site monitors to represent human exposure to CO in certain situations. In one study, Yocom, Clink, and Cote (1971) reported that when make-up air was introduced into an air-conditioned building during morning rush hours (when outdoor CO levels were high), indoor CO concentrations exceeded outdoor levels for the remainder of the day. This finding took on added significance when social scientists reported during the early 1970s that many Americans spent most of their time indoors (Szalai 1972; Chapin 1974). In another study, Wayne Ott collected "walking samples" of CO concentrations on sidewalks along congested streets in downtown San Jose, California, for his doctoral dissertation in civil engineering at Stanford University. He collected samples in large Tedlar™ bags filled by a constant flow pump over a 5-minute period at various times over an 8-hour period. Ott, together with his faculty adviser,

Professor Rolf Eliassen, reported average CO levels ranging from 5.2–14.2 ppm on San Jose's sidewalks. Concurrent CO levels, reported as 1-hour averages at nearby fixed-site monitors, were only 2.4–6.2 ppm (Ott and Eliassen 1973).

A few years later, Cortese and Spengler (1976) did the first survey to determine the CO exposure of "real" people who commuted to and from work. This type of study was made possible by the development of portable electrochemical CO monitors in the early 1970s (USEPA 1991). The research team recruited 66 nonsmoking volunteers who lived in different parts of the metropolitan area of Boston, Massachusetts. The study focused on several travel corridors serving the city's central business district. Each volunteer carried an Ecolyzer monitor attached to a Simpson recorder for 3–5 days between October 1974 and February 1975. The study also estimated COHb levels in the blood based on samples of air in the alveolar sacs of the lung before and after each trip. The study's simultaneous measurement of CO exposure and body burden (% COHb) set a precedent for subsequent studies that involved human participants.

The study reported that the mean of all commuter exposures (11.9 ppm) was about twice the mean concentration measured concurrently at six fixed-site monitors (6 ppm). That was similar to the ratio observed in five cities by Brice and Roesler in the mid-1960s. However, the net mean in-vehicle exposure in Boston was about 42% of the net value reported by Brice and Roesler (1966). Excluding commuters whose cars had "faulty exhaust systems," only 0.5% of 346 sampled CO exposures in the Boston study exceeded the 1-hour CO NAAQS of 35 ppm. Automobile commuters had exposures nearly twice that of transit users, and about 1.6 times that of people who did "split-mode" commuting, which involved both auto and transit. Based on the Boston study, Cortese and Spengler recommended a mobile monitoring program to supplement data from fixed-site monitors.

6.8 ESTIMATES OF NATIONWIDE POPULATION EXPOSURE

The USEPA inherited a fixed-site monitoring program when the agency was established in 1970. Moreover, the Clean Air Act amendments of 1970 did not require measurements of personal exposure to supplement air quality monitoring at fixed sites. There were several proposals to estimate *potential* population exposure to air pollutants during the 1970s (Ott 1982). For example, one estimate simply multiplies the number of days that violations of the NAAQS are observed at county monitoring stations times the county's population. Estimates of exposure using this method are expressed in units of person-days (CEQ 1980).

Knowing that crude estimates of population exposure to CO were potentially inaccurate, the U.S. Public Health Service (PHS) measured the percentage of COHb in the blood of a nationwide sample of 8,405 people between 1976 and 1980. The National Health and Nutrition Examination Survey (NHANES) estimated that 6.4% of those people who never smoked had COHb levels above 2% (Radford and Drizd 1982). This estimate is based on data from a random selection of 3,141 people ranging in age from 12–74 years living in 65 geographic areas of the United States. The estimate was made when ambient CO concentrations were much higher than they are today. As shown by Figure 6.2, the estimated probability distribution of COHb levels appears to be lognormal (Apte 1997). The curve is based on data with a geometric mean (GM) of 0.725% and a geometric standard deviation (GSD) of 2.15%.

The USEPA continues to report the number of Americans who live in areas of the country that are in non-attainment of the NAAQS on an annual basis. The agency's Office of Air Quality Planning and Standards (OAQPS) estimated that 19.130 million Americans residing in 13 counties as of September 2002 (roughly 6.6% of the resident U.S. population) were exposed to ambient CO concentrations that exceeded the 1-hour NAAQS of 35 ppm. Two major metropolitan areas (Los Angeles and Phoenix) accounted for 88.2% of that population at risk (USEPA 2003).

As indicated above, crude estimates of population exposure to CO are made by combining census data on county populations with data on violations of the CO NAAQS recorded by stationary

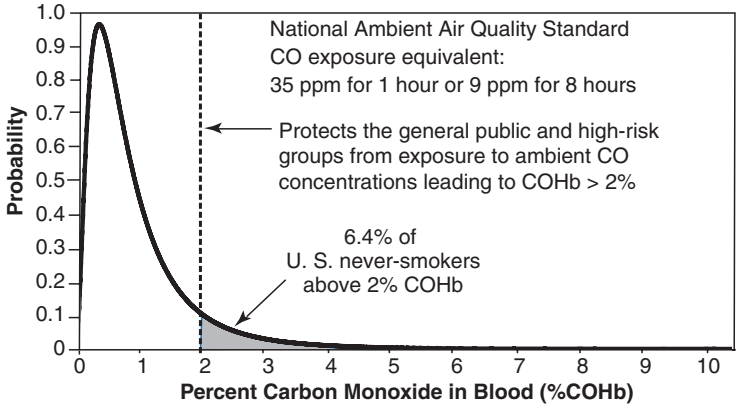


FIGURE 6.2 Estimated probability distribution of carboxyhemoglobin (COHb) levels in blood samples from never-smokers in the United States, 1976–1980.

monitors in each county. Crude estimates of population exposure are based on four assumptions (CEQ 1980):

1. The population does not travel outside the area represented by the fixed-site monitor
2. Air pollutant concentrations measured by fixed-site monitors are representative of the concentrations inhaled by the population throughout the area represented by the monitor
3. The air quality in any one area is only as good as that at the location that had the worst recorded air quality
4. There are no violations in areas of the country (e.g., rural areas) that are not monitored

The early exposure studies (cited previously) challenged the validity of the second assumption regarding the ability of fixed-site monitors to represent the actual CO exposures of people living in cities. Recognizing these studies, the OAQPS developed a risk-analysis framework to support periodic reviews of the NAAQS for CO (Padgett and Richmond 1983; Jordan, Richmond, and McCurdy 1983). This framework gave purpose to subsequent CO exposure studies and stimulated the development of methods and models to estimate total exposure to CO, which is the topic of discussion below.

6.9 ESTIMATING TOTAL CO EXPOSURE

Technical improvements in personal exposure monitors during the 1970s stimulated scholarly interest in how to use and apply them. Fugas (1975) and Duan (1982) advocated that a person's total air pollution exposure could be estimated indirectly based on the following mathematical model:

$$E_i = \sum_{k=1}^{K} c_k(t_{ik}) \tag{6.1}$$

where

E_i = the total integrated exposure of person i over some time period of interest (e.g., 24 hours)

c_k = the air pollutant concentration in microenvironment type k

t_{ik} = the amount of time spent by person i in microenvironment type k
K = the number of microenvironment types encountered by person i over the period of interest

This model states that an individual's total exposure over a given time period can be estimated as the sum of a series of separate exposures resulting from time spent in different types of microenvironments. According to Sexton and Ryan (1988), this model rests on four assumptions: (1) air pollutant concentrations in each microenvironment remain constant for the duration of a person's visit to that microenvironment; (2) the air pollutant concentration and time spent in a microenvironment are independent of each other (i.e., a person does not avoid or leave a microenvironment simply because it is polluted); (3) an adequate number of microenvironments can be identified to characterize a person's total exposure; and (4) the total integrated exposure for a given time period can be related to health effects.

Duan (1982) defined a microenvironment as "a chunk of air space with homogeneous pollutant concentration" (p. 305). Identifying microenvironments is a challenge for analysts who study CO exposure, because CO concentrations not only vary in time and space but can be affected by many factors. For example, CO concentrations in a kitchen will vary depending on whether the kitchen has a gas range, whether it is vented or not, and the duration of its use. In population exposure studies, analysts may ask study participants to keep a diary of time spent in various indoor and outdoor locations and to record certain information about activities (e.g., cooking, smoking behavior) that occur there. Figure 6.3 shows a page from a diary used for this purpose in a study of population exposure in Denver, Colorado. The response categories shown on this page indicate microenvironments relevant to CO exposure.

The concept of integrated air pollutant exposure inspired Ott to develop a computer simulation model to estimate the CO exposure of an urban population. Since Ott's simulation model (described later) needed typical CO concentrations for different microenvironments, he launched two major studies to acquire the data in the late 1970s. These were (1) the field surveys of commercial microenvironments in four California cities (described below), and (2) the El Camino Real commuter exposure surveys on the San Francisco Peninsula (described toward the end of the chapter). At the same time, Ott promoted the emerging science of exposure analysis, which focused on humans as receptors of environmental pollution, to the scientific community (Ott 1985).

6.10 FIELD SURVEYS OF COMMERCIAL MICROENVIRONMENTS

Milton Feldstein, who directed the Bay Area Air Quality Management District in San Francisco during the 1970s, was familiar with Ott's "walking sampling" survey on the streets of downtown San Jose. At the conclusion of that study, Feldstein encouraged Ott to monitor the personal exposures of people, not only as they walked on sidewalks, but also while they shopped in retail stores, ate in restaurants, and did similar activities in the commercial areas of cities. In 1979, Ott proposed this study to Peter Flachsbart at Stanford University's Department of Civil Engineering. The study not only contributed valuable data to Ott's simulation model of population exposure, it also provided insights to the design of two large-scale field surveys of urban population exposure. These larger scale surveys (described later), took place in Denver, Colorado, and Washington, DC.

Knowing the potential for CO exposure along busy streets, Ott and Flachsbart (1982) surveyed locations in the Westwood district of Los Angeles, the financial and Union Square districts of San Francisco, and the central business districts of two suburbs (Palo Alto and Mountain View) of the Bay Area. Using a General Electric electrochemical monitor, they collected 5,000 instantaneous measurements of CO concentrations at 1-minute intervals as they walked the streets of each commercial district. Data collection was cumbersome, because the CO concentration had to be read and recorded from the monitor's digital display, and the time and location of each CO reading

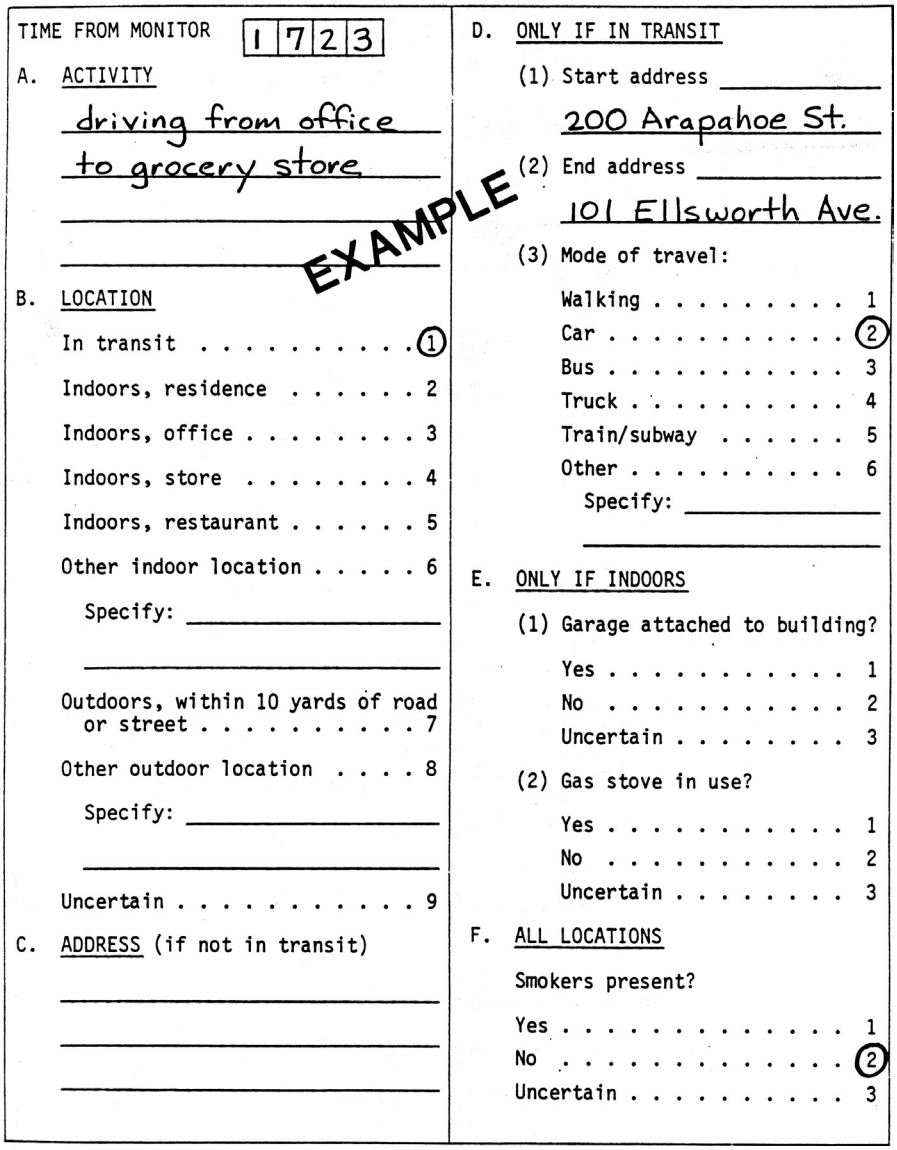


FIGURE 6.3 Example of a completed page from a diary for an exposure study. (From Johnson 1984.)

had to be recorded manually on a clipboard. During 15 field surveys between November 1979 and July 1980, Ott and Flachsbart visited 588 "commercial settings": 220 indoor settings (e.g., department stores, hotels, office buildings, parking garages, retail stores, restaurants, banks, travel agencies, and theaters); and 368 outdoor settings (e.g., street intersections, sidewalks, arcades, parks, plazas, and parking lots).

The results of their study supported the hypothesis that CO emissions from traffic in commercial areas tend to diffuse into adjacent buildings and stores. Although indoor levels were above zero, they seldom were very high — except for parking garages and certain buildings attached to garages. They found that indoor CO concentrations were relatively stable for nearly 2 hours. This finding was very important, because it suggested that estimates of 1-hour indoor CO exposures could be made based on short visits of only a few minutes to each setting. Thus many settings could be visited during each survey. Excluding 10 parking garages, they found that CO concentrations at 6

of 210 settings (2.9%) equaled or exceeded 9 ppm during brief visits (Ott and Flachsbart 1982; Flachsbart and Ott 1984).

On their first survey of downtown Palo Alto, they observed high CO concentrations (above 9 ppm) on 5 floors of a 15-story office building on University Avenue. They traced the source of the problem to an underground parking garage, where CO levels often exceeded 20 ppm. Based on this observation, they developed a "rapid survey technique" to measure CO concentrations in the building and parking garage during nine visits to the building in 1980. High CO levels accumulated in the garage, because its ventilation system had been shut down periodically to save energy in the wake of high electricity costs triggered by a national oil crisis. A survey of the entire building revealed that CO concentrations diffused into the building from the garage through a stairwell. The door between the garage and stairwell was often kept open. Having identified a potential problem, Flachsbart and Ott told the building manager that the building was "hot." When several tenants threatened legal action, the manager hired an air quality consultant to confirm the problem. In 1984, Flachsbart and Ott resurveyed the building on five different dates. By then, the manager was operating the building's ventilation system on a continuous basis and had installed heavier door closers to make sure the door to the garage would shut automatically after opening. A comparison of average CO concentrations, before and after these actions, showed that levels in the garage had fallen from 40.6 ppm in 1980 to 7.9 ppm in 1984, while typical CO levels in the building fell from 11–12 ppm in 1980 to 1–2 ppm in 1984 (Flachsbart and Ott 1986).

Since the detection of the "hot" building in Palo Alto was accidental, the study could not show how to detect "hot" buildings from among the thousands of office buildings that exist in urban areas. As a result, the prevalence of "hot" buildings has never been determined on an urban, state, or national scale. Such a survey would probably require an amendment of the Clean Air Act to enable routine monitoring of *indoor* air quality in the United States. Until then, the Palo Alto study is noteworthy, because it illustrates how portable monitors can be used to survey a high-rise office building quickly once elevated CO concentrations are detected, to determine the source of high concentrations, and to evaluate solutions to reduce those concentrations.

6.11 DIRECT AND INDIRECT APPROACHES TO MEASURE EXPOSURE

The advent of microelectronics during the 1970s enabled the initial development of reliable, compact personal exposure monitors. This development stimulated new thinking about how to measure exposure in the field. Most exposure analysts recognize direct and indirect methods for measuring an individual's exposure to an air pollutant (Sexton and Ryan 1988; Mage 1991).

In the direct approach, personal monitors are distributed ideally to a representative, probability sample of a human population. Population exposure parameters cannot be estimated from a "convenience sample," such as the Boston study of volunteer commuters (described earlier), because such a sample may not represent the population from which it was drawn. Personal monitors must be calibrated with gases of known concentration, before and after their distribution to the public, to assure that the CO measurements are accurate. As one can imagine, distributing calibrated monitors to a large sample of participants on a daily basis creates logistical problems. Participants must be provided with instructions on how to record their exposures as they perform their daily activities during the next day or two. Subjects may be asked to use a diary (Figure 6.3) to record pertinent information about these activities. This information (e.g., presence of a wood-burning fireplace or people smoking) may shed light on circumstances that affect high exposures at certain times of the day. Many exposure analysts believe that the direct approach provides the most accurate estimate of population exposure, because it surveys the actual exposures of people doing their daily activities. However, the direct approach is expensive, because it requires substantial amounts of time and labor to gather data from large samples of urban populations.

The intrusiveness and expense of the direct approach motivated some researchers to favor the indirect approach of exposure measurement. In that approach, trained technicians use calibrated portable monitors to measure pollutant concentrations in specific microenvironments. That information is then combined in mathematical models with separate estimates of time spent in those microenvironments from activity surveys of a population. This approach can lead to errors in estimates of population exposure, because measurements of CO concentrations for different microenvironments and time spent in them are made separately.

Besides choosing between the direct and indirect approaches, the exposure analyst must consider several types of personal monitors. Small, passive CO monitors are usually placed near a person's oral/nasal cavity close to where inhalation exposure actually occurs. Larger monitors with air pumps usually come with a shoulder strap so they can be carried. In commuter studies, monitors are set or mounted somewhere inside the vehicle. Using data from portable monitors, one can plot CO concentrations as a function of time and location for a particular activity, such as commuting. From this information, one can determine the average CO concentration to which a person has been exposed for a given time period.

6.11.1 Studies Using the Direct Approach

In the early 1980s, the USEPA funded a pilot study and two large-scale surveys of urban population exposure based on the direct approach. In the pilot study, Ziskind, Fite, and Mage (1982) asked nine residents of Los Angeles to record their exposures and corresponding activities manually using diaries. Because this process was cumbersome and potentially affected the amount of time doing the activity, the USEPA funded the development in the early 1980s of automated data-logging personal exposure monitors (PEMs). These instruments measured and stored CO concentrations, as well as the time spent doing activities associated with those concentrations (Ott et al. 1986). Figure 6.4 shows an example of a CO monitor carried by a woman shopping in a grocery store.

The Los Angeles pilot study provided useful insights on how to design field surveys of population exposure using the direct approach. For example, the study showed that there was greater variation in CO exposure from person to person than from day to day for any one person. This suggested that subsequent studies should try to sample more people rather than to sample the same person over many days. Akland et al. (1985) described how the USEPA used this insight in designing two large-scale field surveys of population exposure that occurred in the fall of 1982 and winter of 1983. Study participants consisted of 454 residents of Denver, Colorado, and 712 residents of Washington, DC. In each city, the target population consisted of non-institutionalized, nonsmoking residents ages 18–70 who lived in the metropolitan area. Akland et al. (1985) estimated the size of the target population to be 500,000 people in Denver and about 1.2 million people in Washington. Study participants carried a CO PEM and diary for 48 hours in Denver and for 24 hours in Washington.

The goal of these studies was to estimate the percentage of the urban population that was exposed to CO concentrations in excess of the NAAQS for CO. The studies showed that over 10% of the daily maximum 8-hour personal exposures in Denver exceeded the NAAQS of 9 ppm, and about 4% did so in Washington. By comparison, the CO concentrations at fixed-site monitors exceeded the 8-hour NAAQS for CO (9 ppm) only 3% of the time in Denver, and never exceeded the 9 ppm standard in Washington, during the survey period (Akland et al. 1985). These results raised further doubts as to the ability of fixed-site monitors to represent the total CO exposure of urban populations. The studies also showed that the end-expired breath CO levels (after correcting the measured breath concentrations for the influence of room air CO concentrations) were in excess of 10 ppm, which was roughly equivalent to about 2% COHb, in about 12.5% of the Denver participants and about 10% of the Washington participants.

The two studies also looked at factors that contributed to higher levels of exposure. Of 10 different microenvironments, both studies found that parking garages had the highest average CO

FIGURE 6.4 Lady carrying a personal exposure monitor while shopping in a grocery store. (Courtesy of U.S. Environmental Protection Agency.)

concentrations, but the shortest duration of exposure. The Denver study found that the average PEM indoor mean exposure (unadjusted for cofactors) was increased 2.59 ppm (134%) by gas stove operation, 1.59 ppm (84%) by tobacco use from smokers other than study participants, and 0.41 ppm (22%) by attached garages. Table 6.1 shows CO concentrations for selected microenvironments of the Denver study. The table indicates that higher CO exposures occurred for travel by motor vehicle (motorcycle, bus, car, and truck) than for pedestrian or bicycle modes of travel. It also shows that high indoor concentrations (above the 8-hour NAAQS of 9 ppm) occurred in public garages, service stations or vehicle repair facilities. In the Washington study, those who commuted 6 hours or more per week had higher average CO exposures than those who commuted fewer hours per week. Also, certain occupations increased one's CO exposure in the Washington study. People who drove trucks, buses or taxis, as well as people who worked as automobile mechanics, garage workers, and policemen had a mean CO exposure (22.1 ppm) that was three times greater than those who did not work with gasoline-powered automobiles (6.3 ppm). Of the two cities, Denver had higher average CO concentrations than did Washington for all microenvironments. This difference was later attributed to Denver's higher altitude and colder winter climate (Ott, Mage, and Thomas 1992).

Since the Denver–Washington surveys of 1982–1983, there have been only a few small-scale studies of CO exposure using the direct approach. Recognizing the higher exposures of certain types of commuters, Shikiya et al. (1989) measured in-vehicle concentrations of CO and several air toxics (e.g., benzene) in the Los Angeles metropolitan area. The researchers selected a random sample of 140 nonsmokers who commuted from home to work in privately owned vehicles during peak commuting hours during the summer of 1987 and winter of 1988. Trips averaged 33 minutes and driving patterns and ventilation conditions were not controlled. Based on 192 samples, in-vehicle CO concentrations averaged 6.5 ppm in summer and 10.1 ppm in winter.

TABLE 6.1
CO Concentrations of Selected Microenvironments in Denver, CO, 1982–1983 (Listed in Descending Order of Mean CO Concentration)

Microenvironment	*n*	Mean[a] (ppm)	Standard Error (ppm)
In-Transit			
Motorcycle	22	9.79	1.74
Bus	76	8.52	0.81
Car	3,632	8.10	0.16
Truck	405	7.03	0.49
Walking	619	3.88	0.27
Bicycling	9	1.34	1.20
Outdoor			
Public garages	29	8.20	0.99
Residential garages or carports	22	7.53	1.90
Service stations or vehicle repair facilities	12	3.68	1.10
Parking lots	61	3.45	0.54
Other locations	126	3.17	0.49
School grounds	16	1.99	0.85
Residential grounds	74	1.36	0.26
Sports arenas, amphitheaters	29	0.97	0.52
Parks, golf courses	21	0.69	0.24
Indoor			
Public garages	116	13.46	1.68
Service stations or vehicle repair facilities	125	9.17	0.83
Other locations	427	7.40	0.87
Other repair shops	55	5.64	1.03
Shopping malls	58	4.90	0.85
Residential garages	66	4.35	0.87
Restaurants	524	3.71	0.19
Offices	2,287	3.59	0.002
Auditoriums, sports arenas, concert halls	100	3.37	0.48
Stores	734	3.23	0.21
Healthcare facilities	351	2.22	0.23
Other public buildings	115	2.15	0.30
Manufacturing facilities	42	2.04	0.39
Homes	21,543	2.04	0.02
Schools	426	1.64	0.13
Churches	179	1.56	0.25

[a] An observation was recorded whenever a person changed a microenvironment, and on every clock hour; thus each observation had an averaging time of 60 min or less.

Source: Johnson (1984) as reported in U.S. Environmental Protection Agency (1991).

The studies by Shikiya et al. (1989) and Cortese and Spengler (1976) involved real commuters. A comparison of these studies, which were performed 13 years apart, shows the effect of stricter CO regulations. To factor out the effect of ambient CO concentrations, an analyst compares their net mean in-vehicle CO concentrations, which is the estimated mean in-vehicle CO concentration during commuting minus the mean ambient CO concentration, as recorded concurrently at an

appropriate fixed-site monitor. The net value in Boston (7.4 ppm) in 1974–1975 was 51% higher than the net value in Los Angeles (4.9 ppm) in 1987–1988, which are the respective years in which these studies were performed. The difference in net values can be attributed to the prevalence of emission controls on motor vehicles, which differed for the two studies. By the 1980 model year, half of all passenger cars in use in the United States had catalytic and other types of emission controls (MVMA 1990). Hence, most cars in the Boston study probably lacked these emission controls while most cars in the Los Angeles study probably had them.

6.11.2 A Study Using the Indirect Approach

In the early 1980s, a significant CO exposure study occurred in Honolulu, which is located on the Island of Oahu in Hawai'i. Honolulu has no major smokestack industries except for a few refineries located in an industrial area on the southwest corner of the island. Generally, prevailing "trade winds" from the northeast blow most air pollutants in Honolulu out to sea. Occasionally, southerly winds trap air pollutants against the Ko'olau Mountains and create a stagnant air mass. The city is thus an ideal location for doing microenvironmental exposure studies, because ambient CO levels are generally low and satisfy the NAAQS. Several Honolulu "walking surveys" revealed high CO concentrations on the street level of the Ala Moana Shopping Center, less than a mile west of Waikiki Beach (Flachsbart and Brown 1985, 1986).

Several factors contributed to what appeared to be a significant CO exposure problem at the Ala Moana Center. First, it had 155 business outlets that attracted 40 million people including many tourists each year. Second, it had an attached structure with 7,800 parking spaces on several decks. CO emission rates were high, because the posted speed limit for driveways in the structure was 15 mph for the safety of pedestrians. Third, one deck of the parking structure functioned as a lid on the exhaust emissions of cars at the street level of the structure. Fourth, many of the 94 outlets at street level kept their doors open during business hours to attract customers. This allowed CO concentrations from the parking area and internal driveways to diffuse into many retail outlets.

Flachsbart and Brown (1989) devised an indirect method to estimate the CO exposures of employees working in retail stores at the center's street (semi-enclosed) level. Using a portable monitor, they measured CO concentrations and counted employees who worked at 25 retail outlets. Data collection occurred during three periods of the day (10 A.M.–12 noon, 2–4 P.M., and 6–8 P.M.) at 5-day intervals between early November 1981 and late March 1982. Based on 30 days of sampling, they estimated that between 24.5% and 36.1% of employee exposures could have exceeded the 8-hour NAAQS of 9 ppm, and between 2.2% and 2.4% of employee exposures could have exceeded the 1-hour NAAQS of 35 ppm. By comparison, the vast majority (88.5%) of the 8-hour CO averages at the nearest fixed-site monitor, located 3 miles east of the shopping center, were less than or equal to 1 ppm during the study period. The survey on December 21, 1981, showed that CO concentrations in 10 of 25 stores (40%) exceeded the 1-hour NAAQS of 35 ppm. The average CO levels during visits to these 10 stores on that date ranged from a low of 36.3 ppm to a high of 86.7 ppm (Flachsbart and Brown 1985, 1989). CO concentrations on that scale are sufficient to trigger actions by public health officials if they have a mandate to act.

The Ala Moana study showed the potential of portable monitors to reveal CO exposure problems in a specific population, much like the study of the 15-story office building in downtown Palo Alto by Flachsbart and Ott (1986). Both studies also revealed a gap in existing environmental laws and regulations, which do not protect the public from high CO exposures on private property. The Honolulu study also revealed a gap in occupational standards. The Pollution Investigation and Enforcement branch of the Hawai'i State Department of Health (DOH) acknowledged that the Ala Moana study revealed a potential CO exposure problem for retail workers. However, the DOH branch of Occupational Safety and Health found no technical violation of occupational standards for CO, because the occupational CO standards were 200 ppm for 1 hour and 50 ppm for 8 hours at that time. Unlike the NAAQS, which are designed to protect the most sensitive class of the

general population, occupational standards for CO assume a healthy young male employee working in an industrial setting. However, it is not clear that these standards fully protect everyone who works at the Ala Moana Shopping Center, because many of its employees are women and older adults.

6.12 OCCUPATIONAL EXPOSURES

The direct studies of urban populations in Denver and Washington, and the indirect study of shopping center employees in Honolulu, both found higher exposures among occupations that involved use of or proximity to gasoline-powered motor vehicles. To put these studies in perspective, a national survey found that 3.5 million workers in the private sector potentially are exposed to CO from motor exhaust, a figure greater than that for any other physical or chemical agent (Pedersen and Sieber 1988). In 1992, there were 900 work-related CO poisonings resulting in death or illness in private industry as reported by the U.S. Bureau of Labor Statistics in a publication by the National Institute for Occupational Safety and Health (NIOSH 1996). Three risk factors affect industrial occupational exposure: (1) the work environment is located in a densely populated area that has high background (i.e., ambient CO concentrations); (2) the work environment produces CO as a product or by-product of an industrial process, or the work environment tends to accumulate CO concentrations that may result in occupational exposures; and (3) the work environment involves exposure to methylene chloride, which is metabolized to CO in the body. Proximity to fuel combustion of all types elevates CO exposure for certain occupations: airport employees; auto mechanics; small gasoline-powered tool operators (e.g., users of chainsaws); parking garage or gasoline station attendants; policemen; taxi, bus, and truck drivers; tollbooth and roadside workers; warehouse workers and forklift operators (USEPA 1991, 2000).

6.13 RESIDENTIAL EXPOSURES

Residential exposures are an important component of total daily exposure to air pollution, because people spend a substantial portion of their day at home often in close proximity to significant sources of CO emissions. Wilson, Colome, and Tian (1993a,b) and Colome, Wilson, and Tian (1994) reported CO exposures for a random sample of California homes with gas appliances. Of surveyed homes, 13 of 286 homes (4.5%) had indoor CO concentrations above the NAAQS of 9 ppm for 8 hours, and 8 of 282 homes (2.8%) had outdoor CO concentrations above the standard. They found that a small percentage of California homes would still have indoor CO problems even if outdoor CO levels complied with federal ambient standards. They traced higher net indoor CO levels (indoor minus outdoor CO concentrations) to space heating with gas ranges and gas-fired wall furnaces, use of gas ranges with continuous gas pilot lights, small home volumes, and cigarette smoke. Other factors contributing to high CO concentrations included malfunctioning gas furnaces; automobile exhausts leaking into homes from attached garages and carports; improper use of gas appliances (e.g., gas fireplaces); and improper installation of gas appliances (e.g., forced-air unit ducts).

6.14 RECREATIONAL EXPOSURES

Potentially dangerous CO exposures also occur in both outdoor and indoor recreational settings. The CO exposure of cycling as a travel mode has been studied and compared to the exposure of motorists in two European countries. In England, Bevan et al. (1991) reported that the mean CO exposure of cyclists in Southhampton was 10.5 ppm, based on 16 runs over two 6-mile routes that took an average of 35 minutes to complete. Note that the CO exposures of European cyclists may not be comparable to cyclists' exposures in the United States, because installation of catalytic

converters on new cars in Europe occurred in 1988, about 13 years after their introduction in the United States (Faiz, Weaver and Walsh 1996). In the Netherlands, Van Wijnen et al. (1995) compared exposures of volunteers serving as both car drivers and cyclists on several routes in Amsterdam during winter and spring. For a given route, the mean personal 1-hour CO concentrations were always higher for car drivers than for cyclists regardless of when sampling occurred during the year. However, the analysts reported that a volunteer inhaled 2.3 times more air per minute on average as a cyclist than as a car driver. When adjusted for variation in breathing rate, the range in median 1-hour averaged uptakes of CO of cyclists (2.4–3.2 mg) approached that of car drivers (2.7–3.4 mg).

Significant quantities of CO can accumulate over short periods of time in poorly ventilated indoor arenas used for sporting events. The CO is emitted by several sources including ice resurfacing machines and ice-edgers during skating events; gas-powered radiant heaters used to heat viewing stands; and motor vehicles at motocross, monster-truck, and tractor-pull competitions. Some of these sporting events and competitions involve large numbers of spectators and motor vehicles with no emission controls. Several U.S. studies are briefly discussed here to show the scale of the problem.

Lee et al. (1994) reported that CO concentrations measured inside six enclosed rinks in the Boston area during a 2-hour hockey game ranged from 4–117 ppm, whereas outdoor levels were about 2–3 ppm. The alveolar CO concentrations of hockey players increased by an average of 0.53 ppm per 1 ppm of CO exposure over 2 hours. Fifteen years earlier, Spengler, Stone, and Lilley (1978) found CO levels ranging from 23 to 100 ppm in eight enclosed rinks in the Boston area, which suggested that CO levels in ice arenas had not changed very much.

Boudreau et al. (1994) reported CO levels for three indoor sporting events (i.e., monster-truck competitions, tractor pulls) in Cincinnati, Ohio. The CO measurements were taken before and during each event at different elevations in the public seating area of each arena with most readings obtained at the midpoint elevation where most people were seated. Average CO concentrations over 1–2 hours ranged from 13 to 23 ppm before the event and from 79 to 140 ppm during the event. Measured CO concentrations were lower at higher seating levels. The ventilation system was operated maximally, and ground-level entrances were completely open.

6.15 POPULATION EXPOSURE MODELS

Under the Clean Air Act, the USEPA has a statutory requirement to perform a periodic review of the criteria that support the NAAQS. Each review must be based on the latest information published in scientific, peer-reviewed literature. Following the 1991 review, the USEPA's Office of Air Quality Planning and Standards (OAQPS) estimated that fewer than 0.1% of the nonsmoking population with cardiovascular diseases would experience a COHb level of ≥ 2.1% when exposed to CO levels at current ambient standards in the absence of indoor sources (USEPA 1992). That estimate influenced Carol Browner, who was the USEPA administrator under President Clinton, to retain the existing NAAQS for CO in 1994. The next review of the criteria in 2000 supported the conclusions of the review in 1991. The review in 2000 also found that "there is not a good estimate of CO exposure distribution for the current population" (USEPA 2000, p. 7-10).

The OAQPS estimate of population exposure in 1992 was derived from an exposure model. Such models are important because it is impossible to measure the exposure of every person in a population on a real-time basis. Models of human exposure are empirically derived mathematical relationships, theoretical algorithms, or hybrids of these two. To support policy decisions related to the setting of ambient and emission standards, the USEPA supported development of four general population exposure models: (1) the Simulation of Human Activity and Pollutant Exposure (SHAPE) model, (2) the NAAQS Exposure Model (NEM), (3) the probabilistic NEM for CO (pNEM/CO), and (4) the Air Pollutants Exposure Model (APEX). These models rest on Duan's (1982) theory for estimating total human exposure to air pollution as previously discussed.

The SHAPE model used a stochastic approach to simulate the exposure of an individual over a 24-hour period (Ott 1983–1984). The model replicates a person's daily activity pattern by sampling from probability distributions representing the chance of entry, time of entry, and time spent in 22 different microenvironments. Transition probabilities determine a person's movement from one microenvironment to another. The model assumes that microenvironmental concentrations reflect the contribution of an ambient concentration and a component representing CO sources within each microenvironment. Because SHAPE relies on field surveys of representative populations, the data requirements of the model are fairly extensive. The SHAPE model can estimate the frequency distribution of maximum standardized exposures to CO for an urban population and the cumulative frequency distribution of maximum exposures for both 1-hour and 8-hour periods, thereby allowing estimates of the proportion of the population that is exposed to CO concentrations above the NAAQS. An evaluation of SHAPE by Ott et al. (1988), using survey data from the aforementioned Denver population study, showed that the observed and predicted arithmetic means of the 1-hour and 8-hour maximum average CO exposures were in close agreement; however, SHAPE over-predicted low-level exposures and under-predicted high-level exposures.

Unlike SHAPE, which uses diary data from a probability sample of a population, NEM aggregates people into cohorts. The NEM has evolved over time from deterministic to probabilistic versions. As described elsewhere (Johnson and Paul 1983; Paul and Johnson 1985), the deterministic version of NEM simulates movements of selected groups (cohorts) of an urban population through a set of exposure districts or neighborhoods and through different microenvironments. Cohorts are identified by district of residence and, if applicable, district of employment, as well as by age-occupation group and activity pattern subgroup. The NEM uses empirical adjustment factors for indoor and in-transit microenvironments, and accumulates exposure over 1 year. Although the deterministic NEM was able to estimate central tendencies in total exposure accurately, it did less well estimating the associated uncertainty caused by variation in time spent in various microenvironments (Sexton and Ryan 1988) or variation in microenvironmental concentrations (Akland et al. 1985). Paul, Johnson, and McCurdy (1988) discussed improvements in the deterministic version of NEM.

Subsequently, the USEPA developed the probabilistic NEM for CO (pNEM/CO). The model enables the USEPA to evaluate alternative ambient standards for CO by establishing distributions of personal exposures to CO when the alternative CO standard is met. The USEPA evaluated the predictions of pNEM/CO against observed CO exposure data for subjects of the Denver study. That evaluation concluded that there was relatively close agreement between simulated and observed exposures for CO concentrations near the average exposure, within the range of 6–13 ppm for the 1-hour standard and within 5.5–7 ppm for the 8-hour standard. However, the model over-predicted lower exposures and under-predicted higher exposures for both standards (Law et al. 1997).

More recently, the USEPA developed the Air Pollutants Exposure Model (APEX). Like its predecessors, APEX includes a dosimetry module to estimate the percentage of COHb in the blood of an individual person for a calendar year. Richmond et al. (2003) describe an application of APEX to estimate adult exposure to CO in the Los Angeles, Orange, and San Bernardino counties of California. Fixed-site monitors in the region had recorded ambient CO concentrations in excess of the 1-hour standard of 35 ppm in 2002 (USEPA 2003). Model inputs included ambient air quality data for the metropolitan region in 1997, data on two major indoor sources of CO (i.e., passive smoking and gas stoves), demographic data from the 2000 census, and diary data from the USEPA's Consolidated Human Activity Database (CHAD). The ability of APEX to make accurate estimates of population exposure to CO must still be determined.

6.16 ACTIVITY PATTERNS

Population exposure models require extensive data on human activity patterns. To supply that data, the USEPA authorized and supported the National Human Activity Pattern Survey (NHAPS). This

survey collected 24-hour diary data on human activities and their locations from a sample of 9,386 U.S. residents between October 1992 and September 1994 (Klepeis, Tsang, and Behar 1996; Klepeis et al. 2001). To enable projections to the larger U.S. population, the sample was weighted by the 1990 U.S. census data to account for disproportionate sampling of certain population groups defined by age and gender. Results were analyzed across a dozen subgroups defined by various characteristics of respondents: gender, age, race, education, employment, census region, day of week, season, asthma, angina, and bronchitis/emphysema. The weighted results showed that, on average, 86.9% of a person's day was spent indoors (68.7% at residential locations), 7.2% of the day was spent in or near vehicles, and 5.9% of the day was spent in outdoor locations.

The study also reported unweighted descriptive statistics and percentiles for both the full population and various subpopulations (i.e., people who actually did certain activities or who spent time in certain microenvironments) (Tsang and Klepeis 1996). The findings on activities that contribute to elevated CO exposures are of particular relevance. Of all respondents, 38.3% reported having a gas range or oven at home, and 23.7% said that the range or oven had a burning pilot light. When asked about motor vehicle use, 10% of 6,560 people (7.0% of the total sample) spent more than 175 minutes per day inside a car, and 10% of 1,172 people (1.2% of the total sample) spent more than 180 minutes inside a truck or van. Of those who were inside a car and knew they had angina (n = 154 respondents), 10% of them spent more than 162 minutes per day inside a car. The survey also asked about sources of household pollutants. Of 4,723 respondents to this question, 10.5% were exposed to solvents, 10.4% to open flames, and 8.4% to "gasoline-diesel"-powered equipment; 6.3% of these respondents were in a garage or indoor parking lot; and 5.7% reported that someone smoked cigarettes at home. Only 1.8% of 4,663 respondents reported having a kerosene space heater at home.

Workers have adjusted their commuting behavior during the past 25 years in response to growing traffic congestion and social trends. The decennial census collected travel time data for the first time in 1980. Census reports showed that the nation's average commuting time of 21.7 minutes in 1980 increased only 40 seconds to 22.4 minutes in 1990 (Pisarski 1992), but then jumped to 25.5 minutes in 2000 (U.S. Census Bureau 2003). Although the number of workers who commuted 45 minutes or more increased from 10.9 million in 1980 to 13.9 million in 1990, the mean travel time of this commuter cohort actually decreased slightly from 59.6 minutes in 1980 to 58.5 minutes in 1990. One reason for this is that more people were taking their morning commute from home to work during the "shoulder hours" from 6–7 A.M. or from 8–9 A.M. than during the "peak hour" from 7–8 A.M. In 1990, the "shoulder hours" accounted for about 37% of worker trip starts, whereas the "peak hour" accounted for only 32% of trip starts (Pisarski 1992). Flachsbart (1999c) showed that a commuter's travel time and CO exposure for a trip from home to work was related to trip departure time. In his study of a coastal artery in Honolulu, travel during off-peak hours reduced both travel time and CO exposure compared to travel during peak hours.

Increasingly, more people are working, shopping, and entertaining themselves at home to avoid traffic. This trend is possible due to the advent of the "information superhighway" (i.e., broadband, two-way communications facilitated by personal computers and the Internet). The percentage of people working at home increased from 2.3% in 1980 (Pisarski 1992) to 3.3% in 2000 (U.S. Census Bureau 2003). The overall impact of this trend on commuter CO exposures has not been studied.

6.17 PUBLIC POLICIES AFFECTING EXPOSURE TO VEHICLE EMISSIONS

The Clean Air Act (CAA) amendments of 1970, 1977, and 1990 articulated three approaches for regulating motor vehicle emissions. The foremost approach requires companies that sell cars and trucks for the U.S. market to produce vehicles that emit fewer air pollutants, and oil companies to sell fuels that achieve the same goal. The second approach relies on state governments to implement

programs that maintain the effectiveness of motor vehicle emission control systems and reduce the use of motor vehicles. The third approach requires that new investments in transportation systems support achievement of the NAAQS (Howitt and Altshuler 1999). This section discusses the effect of the Clean Air Act on CO exposure.

6.17.1 Effects of Motor Vehicle Emission Standards on Unintentional Deaths Attributed to Exposure

Cobb and Etzel (1991) reported that the unintentional, CO-related, annual death rate per 100,000 people in the United States declined from 0.67 in 1979 to 0.39 in 1988, based on death certificate reports compiled by the National Center for Health Statistics. Motor vehicle exhaust gas accounted for 6,552 deaths, or 56.7% of the total 11,547 unintentional, CO-related deaths that occurred during the 10-year period. The highest death rates occurred among males, blacks, the elderly, and residents of northern states. Monthly variation in death rates indicated a seasonal pattern, with January fatalities routinely about 2–5 times higher than in July. The study speculated that declining death rates could be attributed to industry compliance with motor vehicle CO emission standards of the CAA amendments. Table 6.2 shows data to support the study's conclusion. The investigators argued that tighter CO emission standards enabled cars to emit exhaust into an enclosed space for a longer period of time before CO concentrations could accumulate to toxic levels. The findings of the study by Cobb and Etzel (1991) are noteworthy even though mortality is not a health effect used by USEPA to set the NAAQS for CO.

6.17.2 Effects of Transportation Investments on Commuter Exposure

Since the mid-1960s, major construction projects intended to expand highway capacities have been opposed in some metropolitan areas in the United States. Opponents claimed that these projects promoted urban sprawl and induced motor vehicle travel that raised regional air pollutant emissions. To address these concerns, the 1990 CAA amendments stipulated that transportation actions (plans, programs, and projects) cannot create new NAAQS violations, increase the frequency or severity of existing NAAQS violations, nor delay attainment of the NAAQS (U.S. Code 1990). Pursuant thereto, the USEPA promulgated its Transportation Conformity Rule (TCR). Complementary provisions of the 1991 Intermodal Surface Transportation Efficiency Act (ISTEA) offered financial incentives to improve air quality under the Congestion Management and Air Quality (CMAQ) improvement program (Ortolano 1997). Under CMAQ, metropolitan planning organizations (MPOs) were offered federal funds to improve air quality by implementing TCMs. Examples of TCMs include programs to promote car- and van-pooling, flexible working schedules, special lanes for high occupancy vehicles (HOVs), and parking restrictions. USEPA's Office of Mobile Sources (OMS) did an extensive study of how TCMs have changed travel activity, including the number of trips, vehicle miles of travel, vehicle speed, travel time, and the extent to which commuters have shifted travel from peak to off-peak periods. Using an emission factors model (i.e., MOBILE5), OMS estimated how much TCMs would change the average speeds and CO emissions of motor vehicles (USEPA 1994).

By comparison, the direct effect of TCMs on commuter exposure to CO has received only limited study. Flachsbart (1989) hypothesized that priority (with-flow and contra-flow) lanes could be effective in reducing exposure to CO, because these lanes enable commuters to travel at higher speeds than commuters in unrestricted lanes. In theory, priority lanes provide a speed advantage during the line-haul phase of one's trip to compensate for the extra time needed to collect passengers at the origin of the trip. This speed advantage could translate to lower CO exposure for people using priority lanes. To test this hypothesis, Flachsbart recruited several volunteers who used the Kalaniana'ole Highway, a coastal artery in Honolulu. All volunteers commuted from their homes in East Honolulu to the Manoa campus of the University of Hawai'i during the 1981–1982 academic

TABLE 6.2
Motor Vehicle CO Emission Standards, In-Vehicle CO Exposures, and Unintentional CO-Related Annual Death Rates

Year	New Passenger Car CO Emission Standard Federal (g/mile)	California (g/mile)	Net Mean In-Vehicle CO Concentration (ppm)	CO Exposure Study Location	United States CO-Related Annual Death Rates per 10^5 People
Pre-control	84.0	84.0			
~1965	84.0	84.0	12	Los Angeles, CA	
1966	84.0	51.0	17.5	5 U.S. cities	
1968	51.0	51.0			
1970	34.0	34.0			
1972	28.0	34.0			
1973	28.0	34.0	11.5	Los Angeles, CA	
1974	28.0	34.0			
1974–75	15.0	9.0	7.4	Boston, MA	
1975–77	15.0	9.0			
1978	15.0	9.0	10.3	Washington, DC	
1979	15.0	9.0	9.7	Los Angeles, CA	0.67
1980–81	7.0	9.0	8.3	Santa Clara Co., CA	0.55
1981	3.4	7.0	5.2	Denver, CO	0.58
1981	3.4	7.0	4.3	Los Angeles, CA	0.58
1981	3.4	7.0	2.9	Phoenix, AZ	0.58
1981	3.4	7.0	2.9	Stamford, CT	0.58
1981–82	3.4	7.0	9.5	Honolulu, HI	
1982	3.4	7.0			0.56
1982–83	3.4	7.0	1.4	Denver, CO	
1982–83	3.4	7.0	1.8	Washington, DC	
1983	3.4	7.0	9.4	Washington, DC	0.53
1984	3.4	7.0			0.49
1985	3.4	7.0			0.49
1986	3.4	7.0			0.44
1987	3.4	7.0			0.39
1987–88	3.4	7.0	4.9	Los Angeles, CA	
1988	3.4	7.0	8.4	Raleigh, NC	0.39
1989–90	3.4	7.0			
1991–92	3.4	7.0	~3.6	Santa Clara Co., CA	
1992	3.4	7.0	<3.0	New Jersey suburbs of New York City	

Source: From Flachsbart, 1999. With permission from Elsevier.

year. During weekdays a crew from the state Department of Transportation coned off priority lanes on a 4-mile segment of the artery for the exclusive use of express buses, carpools, and vanpools between 6 and 8 A.M. Volunteers using these modes of travel entered designated lanes within 15 minutes of each other to facilitate valid comparisons of their travel times and CO exposures.

The results of this study show the potential of express lanes to reduce commuter exposure to CO. Compared with average CO concentrations inside passengers cars on the highway's unrestricted

lanes, average CO concentrations were 61% less for passengers of express buses (n = 28 trips), about 28% less for those using non-bus, high-occupancy vehicles (HOVs) (n = 20 trips), and about 18% less for people in carpools (n = 52 trips). One explanation for these results is that faster vehicles in priority lanes created more air turbulence, which dispersed air pollutants surrounding the vehicle. Of course, users of priority lanes also reduced their total trip exposure to CO, because these lanes were very effective in reducing total travel time. The differences in CO exposure between lanes existed even though the priority lanes were frequently downwind of large numbers of slower vehicles emitting high levels of CO in congested lanes. However, differences in CO exposure by travel lane also could have been caused by differences in vehicle type and ventilation, both of which could not be controlled by study design.

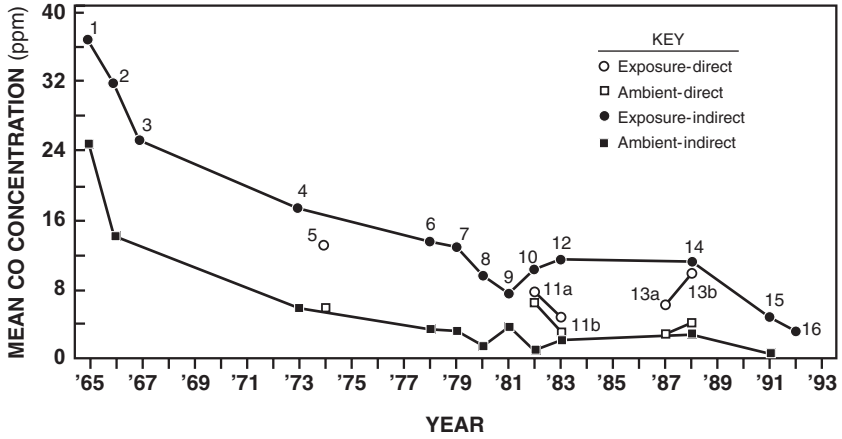
6.17.3 Effects of Motor Vehicle Emission Standards on Commuter Exposure

The percentage of in-use passenger cars in the United States with catalytic and other types of emission controls increased from 50.0% in 1980 to 90.3% by 1989 (MVMA 1990). To study the effect of this trend on the CO exposure of commuters, Flachsbart (1995) reviewed 16 in-vehicle studies performed in various cities in the United States between the mid-1960s and early 1990s. In each study, trips lasted for an hour or less. Table 6.2 summarizes the results of these studies, and Figure 6.5 shows evidence of downward trends in both the mean in-vehicle CO concentrations (top line) and the concurrently measured ambient CO concentrations (bottom line) of these studies. These lines do not imply that CO concentrations can be inferred from points on the lines themselves, or that relationships exist between results for different cities. If one assumes that the results of this "meta-analysis" (i.e., study of studies) are representative of typical CO exposures for commuters in large U.S. cities, then CO exposures fell approximately 90% for commuters over a 27-year period.

In comparing the methodology of the 16 studies, Flachsbart (1995) found that typical in-vehicle CO exposures varied by study approach (direct vs. indirect) and even by researcher. For the more common type of study (i.e., the indirect approach), typical exposures varied by city, season of the year, roadway type and location, travel mode, and the ventilation settings of the test vehicle. The two studies of an arterial highway (El Camino Real) in Northern California by Ott, Switzer, and Willits (1993b) were noteworthy exceptions to this observation, because they adhered to a standardized protocol to facilitate comparisons in exposure over one decade. (Figure 6.5 identifies these two studies as #8 in 1980, and #15 in 1991.) Flachsbart (1995) recommended that future studies use standard methods similar to those used by Ott, Switzer, and Willits (1993b) to "improve their potential to assess the effectiveness of public policies in curtailing automotive emissions over the long term" (Flachsbart 1995, p. 493). Based on that recommendation, Ott resurveyed the study site on El Camino Real in 2001–2002, and recruited Flachsbart to analyze and compare the data from all three studies. The next section discusses the results of that comparison.

6.18 THE EL CAMINO REAL COMMUTER EXPOSURE SURVEYS

The three studies of El Camino Real, which together span the period between 1980 and 2002, show compelling evidence of the effectiveness of California's motor vehicle emission control program. This program includes a set of tailpipe exhaust emission standards, which have affected all new cars sold in the state since 1966, and an inspection and maintenance (I/M) program (known as Smog Check), which was implemented in 1984 to regulate emissions of in-use vehicles. The El Camino Real study differs from many other exposure studies, because it adopted a longitudinal design to observe the cumulative effect of progressively tighter emission standards and the I/M program over two decades. In this study, field surveys of the highway occurred during 15-month periods in 1980–1981, 1991–1992, and 2001–2002. During each period, Wayne Ott measured his personal CO exposure inside an automobile during round-trips on a 5.9-mile segment of the highway



1. Los Angeles, CA
2. Chicago, IL; Cincinnati, OH; Denver, CO; St. Louis, MO; Washington, DC
3. 14 cities
4. Los Angeles, CA
5. Boston, MA
6. Washington, DC
7. Los Angeles, CA
8. Menlo Park, Palo Alto, and Los Altos, CA
9. Denver, CO; Los Angeles, CA; Phoenix, AZ; Stamford, CT
10. Honolulu, HI
11a. Denver, CO
11b. Washington, DC
12. Washington, DC
13a. Los Angeles, CA (summer)
13b. Los Angeles, CA (winter)
14. Raleigh, NC
15. Menlo Park, Palo Alto, and Los Altos, CA
16. New Jersey Turnpike and Route 18, NJ

FIGURE 6.5 Trends in ambient CO concentrations and in-vehicle CO exposures in the United States, 1965–1992. (From Flachsbart, 1995. With permission from Macmillan Publ.)

using a standardized protocol to control for confounding variables. Fifty trips from each period — for a total of 150 trips — were matched by date, day of the week, and starting time to facilitate comparisons over two decades (Flachsbart, Ott, and Switzer 2003).

The net CO concentration of each trip was obtained by subtracting the background CO level from the average CO concentration for the entire trip. The mean net CO concentration (1.7 ppm) for 2001–2002 was 35.4% of the corresponding value (4.8 ppm) for 1991–1992 and 17.5% of the value (9.7 ppm) for 1980–1981. Figure 6.6 shows that the median (1.5 ppm) of the observed CO concentrations for the 50 trips in 2001–2002 was only slightly below the predicted range in median values (1.55–1.78 ppm) for 2002–2003. This prediction was based on the roadway emission estimate of the STREET model, as reported by Yu, Hildemann, and Ott (1996).

Flachsbart, Ott, and Switzer (2003) attributed the results of their study to the adoption of progressively tighter tailpipe CO emission standards on new motor vehicles sold in the state since 1966, and to the gradual replacement of older cars with newer models. California's cold-temperature CO standard implemented in 1996 appeared to reduce high CO exposures that were observed during the late fall and winter of 1980–1981. The lack of sharp peaks in CO measurements taken during trips in 2001–2002 indicated fewer "high-emitting" vehicles on the highway relative to comparable measurements taken during previous surveys. This result was attributed to both the I/M program and tougher "durability standards" on emission controls that were phased in on new cars sold in 1993 and 1994.

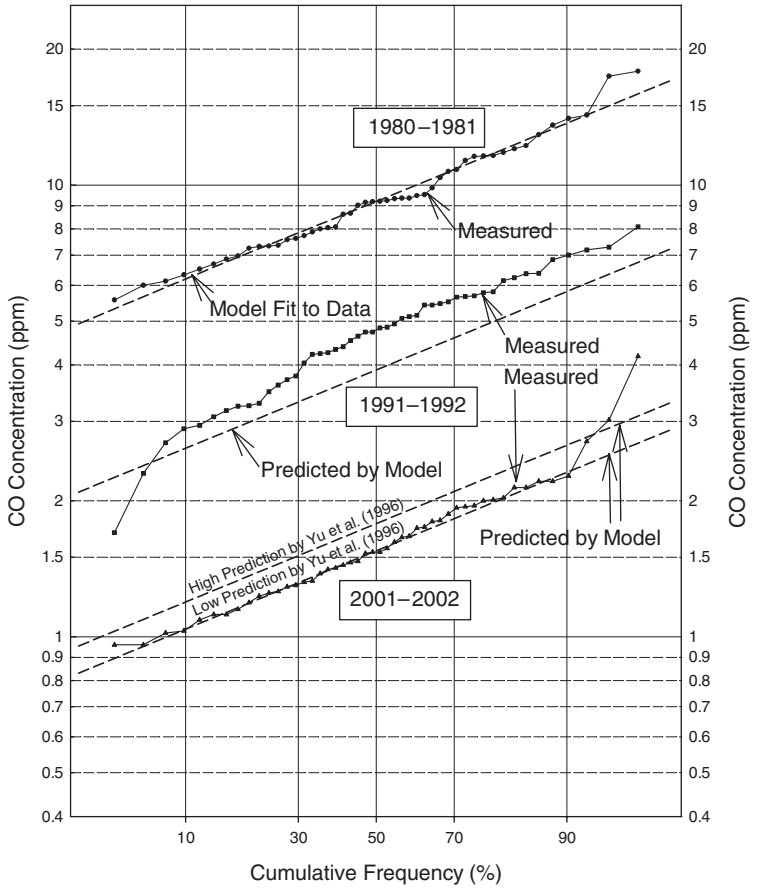


FIGURE 6.6 Logarithmic probability plots of net in-vehicle CO concentrations for three time periods. (From Flachsbart, Ott, and Switzer 2003.)

6.19 INTERNATIONAL COMPARISONS OF COMMUTER EXPOSURE

By comparing studies in different countries, Flachsbart (1999a,b) suggested that commuter exposure to motor vehicle emissions may be an indicator of a country's level of economic development. Similar studies of CO exposure for three modes of travel (i.e., automobile, diesel bus, and rail transit) were performed in Washington, DC (Flachsbart et al. 1987) and in Mexico City (Fernandez-Bremauntz and Ashmore, 1995a,b) about 8 years apart. Each study used the indirect approach to measure commuter CO exposure during peak periods of travel and gathered concurrent ambient CO levels from fixed-site monitors. Table 6.3 reports the net mean CO concentrations (i.e., ambient CO levels subtracted from in-vehicle exposure) by travel mode for each study. Notice that net mean CO concentrations varied significantly, not only by travel mode but also by country. Variation by travel mode could have occurred due to variation in the traffic volume of sampled routes, to variation in the height of the portable monitor above the roadway, and to variation in the vehicle ventilation system of each mode.

The variation in exposure by country can partly be explained by comparing the history of automotive emission standards in each country. The United States initiated nationwide emission standards on new passenger cars in 1968, adopted progressively tighter controls throughout the 1970s (see Table 6.2), and required catalytic converters on all new cars beginning in the 1975

TABLE 6.3
Typical Net Mean CO Concentrations by Travel Mode in Washington, DC, and Mexico City[a,b]

	Washington, DC, U.S. (1983)		Mexico City, Mexico (1991)	
Travel Mode	Net Mean CO Concentrations (ppm)	Averaging Times (minutes)	Net Mean CO Concentrations (ppm)	Averaging Times (minutes)
Automobile	7–12	34–69	37–47	35–63
Diesel bus	2–6	82–115	14–27	40–99
Rail transit	0–3	27–48	9–13	39–59

[a] "Typical" means do not include outlier values that can be attributed to unusual circumstances.
[b] Net mean CO concentration = mean in-vehicle CO concentration – mean ambient CO concentration.

Source: From Flachsbart, 1999. With permission from Elsevier.

model year (Johnson 1988). Mexico adopted a tailpipe CO emission standard of 47.0 grams/mile for the 1975 model year, when the U.S. standard was 15.0 grams/mile, and reached parity with the 1981 U.S. standard of 3.4 grams/mile by the 1993 model year (Faiz, Weaver, and Walsh 1996). The Washington exposure study occurred in early 1983, about 2 years after the 3.4 grams/mile standard took effect in the United States, and the Mexico City exposure study occurred in 1991, 2 years before the same standard took effect in Mexico. Hence, the vehicle fleet in Washington, DC, at the time of its exposure study in 1983 was under tighter emission standards than the fleet in Mexico City at the time of its exposure study in 1991.

6.20 CONCLUSIONS

In accordance with the Clean Air Act, the USEPA must determine whether or not a community complies with National Ambient Air Quality Standards (NAAQS). This determination is based on measurements of ambient air quality made by a nationwide network of fixed-site monitoring stations. Using personal exposure monitors, early studies revealed that measurements at fixed-site monitors were not sufficient to estimate the total CO exposure of urban populations in U.S. cities. In response to these findings, the USEPA's Office of Air Quality Planning and Standards (OAQPS) developed a risk-analysis framework to enable more formal treatment of the uncertainties associated with scientific estimates of population exposure. This framework benefited from progress made in the science and technology of personal exposure monitoring devices and in the development of new methods for measuring exposure as people performed their daily activities.

Over the years, the USEPA also supported the development of several large-scale, population exposure models (e.g., SHAPE, NEM, pNEM/CO, and APEX) at considerable expense to assist the agency in its periodic review and evaluation of the NAAQS for CO. Each review required an accurate estimate and assessment of population exposure to CO. To provide data for these models, the USEPA supported several direct and indirect studies of population exposure to CO at considerable expense. These studies included the landmark Denver–Washington population exposure studies of 1982–1983. Compared to crude methods, these models improved the accuracy of the

USEPA's estimates of population exposure. Even so, the NAAQS for CO have not changed since their promulgation in April 1971. Future adjustments to the NAAQS for CO may depend on more refined estimates of exposure in sensitive populations.

It is not likely that we shall see another large-scale survey in the United States of urban population exposure to CO. The surveys of population exposure in Denver and Washington were undertaken when ambient CO levels were higher than they are today and the ability of fixed-site CO monitors to represent population exposure was in question. Noting the steady decline in ambient CO concentrations in many cities nationwide, the National Research Council declared in 2003 that the control of CO has been a success and that air quality management in the United States has shifted to other air pollutants such as tropospheric ozone and fine particulate matter. In fact, CO exposure has not been the sole focus of most commuter exposure studies that have been performed in the United States since the mid-1980s. The trend has been to measure exposure to CO, together with exposures to air toxics (e.g., benzene) and particles. Given these trends, the purpose and methods of CO exposure studies will likely change in the future. At best, we may see a few indirect studies of exposure in the United States that build on existing knowledge of the sources of CO emissions and human activity patterns, and take advantage of the portability of personal monitors. These studies could discover new microenvironments with elevated CO concentrations or revisit known microenvironments (e.g., buildings attached to parking garages, indoor sport arenas) to establish trends in exposure.

Over the years there has been a series of unrelated commuter exposure studies in the United States undertaken by many researchers using different methodologies. Viewed collectively, these studies suggest that substantial reductions in commuter CO exposure can be attributed to the Clean Air Act amendments of 1970, 1977, and 1990. The strongest evidence of this trend comes from a long-term study of commuter exposure on an arterial highway (El Camino Real) in northern California. The study quantified a significant reduction in commuter CO exposure over two decades, which could be attributed to California's motor vehicle emission control program. This implies that all past measurements of commuter CO exposure (i.e., those made prior to 2000) may no longer be indicative of current CO exposures. It also implies that large-scale models (e.g., the new APEX model) of urban population exposure are trying to provide estimates of a moving target.

Studies of exposure are particularly applicable to mega-cities in developing countries that have rapidly growing motor vehicle populations, congested streets and confined spaces in urban areas, and nascent motor vehicle emission control programs. Studies of population exposure would enable these countries to determine what fraction of total exposure to CO and other mobile source pollutants can be attributed to commuting activities. Microenvironmental studies of commuter exposure would enable them to evaluate the effectiveness of emission control programs and transportation measures designed to reduce exposure to mobile source emissions. These countries should first establish baseline information on commuter exposure, as the United States did in the mid-1960s before implementing tailpipe emission standards for new passenger cars in 1968. Periodically, exposure should then be remeasured using a standard data collection protocol to assess progress and facilitate comparisons over time. Ideally, the measurement protocol should take advantage of modern personal monitors, which are capable of measuring CO and other vehicle-related air pollutants continuously over time in the field, storing these measurements automatically using portable data loggers, and transferring data directly to computer files for analysis. Moreover, the protocol should represent typical travel patterns, recognize that exposure can vary over time and space, and adhere to good principles of research design.

6.21 ACKNOWLEDGMENTS

Several people assisted the author in preparing this chapter. Lance Wallace and Doug Eisinger provided thoughtful advice and comments on a previous draft. Wayne Ott and Michael Apte rendered

invaluable assistance in preparing figures. The author expresses his gratitude to each of them but assumes responsibility for any factual errors that may still remain.

Portions of text of this chapter are reprinted, with permission, from: *Chemosphere—Global Change Science,* Vol. 1, M.A.K. Khalil, J.P. Pinto, and M.J. Shearer, Eds., P.G. Flachsbart, "Human Exposure to Carbon Monoxide from Mobile Sources," pages 301–329, copyright (1999b), published by Elsevier; *Urban Traffic Pollution*, D. Schwela and O. Zali, Eds., P.G. Flachsbart, "Chapter 4: Exposure to Exhaust and Evaporative Emissions from Motor Vehicles," pages 89–132, copyright (1999a), published by the World Health Organization; and *Journal of Exposure Analysis and Environmental Epidemiology*, **5**(4), Peter G. Flachsbart, "Long-Term Trends in United States Highway Emissions, Ambient Concentrations, and In-Vehicle Exposure to Carbon Monoxide in Traffic," pages 473–495, copyright (1995), published by Macmillan Publishers Ltd.

6.22 QUESTIONS FOR REVIEW

1. What are the principal sources of CO emissions? Under what conditions or circumstances do these motor vehicles emit higher amounts of CO?
2. What constitutes exposure to CO for human beings? What groups of people are at greater risk from exposure to ambient CO concentrations in urban areas?
3. What are the NAAQS for CO? How does the USEPA determine whether a community is in compliance with the NAAQS? What carboxyhemoglobin (COHb) level in the blood of the general population are the NAAQS for CO designed to prevent?
4. What assumptions are necessary to estimate population exposure based on fixed-site monitors? Explain how CO exposure studies have challenged the validity of some of these assumptions.
5. If ambient CO concentrations for a community comply with the NAAQS, give several reasons to study CO exposure in that community.
6. How do direct studies differ from indirect studies of CO exposure? Give examples of microenvironmental studies. What are the pros and cons of each type of study?
7. What factors contribute to elevated CO exposures as a person performs his or her daily activities? What types of microenvironments, travel modes, and daily activities increase CO exposure?
8. Describe gaps in the ability of the Clean Air Act as amended to protect public health based on your knowledge of CO exposure studies.
9. What is net CO exposure? Why is it necessary to compute net exposures in making comparisons of exposure over time and space?
10. Why study trends in CO exposure over time? What can a trend study of CO exposure reveal about motor vehicle emission control programs?
11. Explain why commuter CO exposure varies among developed and developing countries. Assess the potential for a commuter CO exposure study in a mega-city of a developing country.

REFERENCES

Akland, G.G., Hartwell, T.D., Johnson, T.R., and Whitmore, R.W. (1985) Monitoring Human Exposure to Carbon Monoxide in Washington, D.C., and Denver, Colorado, during the Winter of 1982–1983, *Environmental Science and Technology*, **19**(10): 911–918.

Amiro, A. (1969) CO Presents Public Health Problems, *Journal of Environmental Health*, **32**: 83–88.

Apte, M.G. (1997) A Population-Based Exposure Assessment Methodology for Carbon Monoxide: Development of a Carbon Monoxide Passive Sampler and Occupational Dosimeter, Ph.D. diss., Department of Environmental Health Sciences, University of California, Berkeley, CA.

Arias, E., Anderson, R.N., Kung, H.-C., Murphy, S.L., and Kochanek, K.D. (2003) Deaths: Final Data for 2001, *National Vital Statistics Reports*, Center for Health Statistics, Hyattsville, MD, **52**(3).

ATSDR (1993) Methylene Chloride Toxicity, Agency for Toxic Substances and Disease Registry, *American Family Physician*, **47**(5): 1159–1166.

Besner, D. and Atkins, P. (1970) The Dispersion of Lead and Carbon Monoxide from a Heavily Travelled Expressway, presented at the 63rd Annual Meeting of the Air Pollution Control Association, St. Louis, MO.

Bevan, M.A.J., Proctor, C.J., Baker-Rogers, J., and Warren, N.D. (1991) Exposure to Carbon Monoxide, Respirable Suspended Particulates, and Volatile Organic Compounds while Commuting by Bicycle, *Environmental Science & Technology*, **25**(4): 788–791.

Blumenthal, D. (2005) The Use of Real-Time Air Quality Data in Daily Forecasting and Decision-Making, *EM: The Magazine for Environmental Managers,* September: 18.

Boudreau, D.R., Spadafora, M.P., Wolf, L.R., and Siegel, E. (1994) Carbon Monoxide Levels during Indoor Sporting Events — Cincinnati, 1992–1993, [reprint] from *MMWR* 43(2): 21–23, *JAMA, Journal of the American Medical Association*, **271**(6): 419.

Brice, R.M. and Roesler, J.F. (1966) The Exposure to Carbon Monoxide of Occupants of Vehicles Moving in Heavy Traffic, *Journal of the Air Pollution Control Association,* **16**(11): 597–600.

Burr, M.L. (2000) Combustion Products, in *Indoor Air Quality Handbook*, Spengler, J.D., Samet, J.M., and McCarthy, J.F., Eds., McGraw-Hill, New York, NY, 29.3–29.25.

CEQ (1980) *Environmental Quality — 1980: The Eleventh Annual Report of the Council on Environmental Quality*, Council on Environmental Quality, U.S. Government Printing Office, Washington, DC.

Chapin, F.S. Jr. (1974) *Human Activity Patterns in the City*, Wiley-Interscience, New York, NY.

Claggett, M., Shrock, J., and Noll, K. (1981) Carbon Monoxide near an Urban Intersection, *Atmospheric Environment*, **15**(9): 1633–1642.

Cobb, N. and Etzel, R.A. (1991) Unintentional Carbon Monoxide-Related Deaths in the United States, 1979 through 1988, *JAMA: Journal of the American Medical Association*, **266**(5): 659–663.

Coburn, R.F., Forster, R.E., and Kane, P.B. (1965) Considerations of the Physiological Variables That Determine the Blood Carboxyhemoglobin Concentration in Man, *Journal of Clinical Investigation,* **44**: 1899–1910.

Colome, S.D., Wilson, A.L., and Tian, Y. (1994) *California Residential Indoor Air Quality Study, Volume 2: Carbon Monoxide and Air Exchange Rate, an Univariate and Multivariate Analysis*, GRI-93/0224.3, Gas Research Institute, Chicago, IL.

Colucci, J. and Begeman, C. (1969) Carbon Monoxide in Detroit, New York, and Los Angeles Air, *Environmental Science and Technology,* **3**(1): 41–47.

Colvile, R.N., Hutchinson, E.J., Mindell, J.S., and Warren, R.F. (2001) The Transport Sector as a Source of Air Pollution, *Atmospheric Environment,* **35**(9): 1537–1565.

Cortese, A.D. and Spengler, J.D. (1976) Ability of Fixed Monitoring Stations to Represent Personal Carbon Monoxide Exposure, *Journal of the Air Pollution Control Association,* **26**(12): 1144–1150.

Downs, A. (2004) *Still Stuck in Traffic: Coping with Peak-Hour Traffic Congestion*, Brookings Institution Press, Washington, DC.

Doyle, J. (2000) *Taken for a Ride: Detroit's Big Three and the Politics of Pollution*, Four Walls Eight Windows, New York, NY.

Duan, N. (1982) Models for Human Exposure to Air Pollution, *Environment International,* **8**: 305–309.

Ernst, A. and Zibrak, J.D. (1998) Carbon Monoxide Poisoning, *New England Journal of Medicine*, **339**(22): 1603–1608.

Faiz, A., Weaver, C.S., and Walsh, M.P. (1996) *Air Pollution from Motor Vehicles: Standards and Technologies for Controlling Emissions*, The World Bank, Washington, DC.

Fernandez-Bremauntz, A.A. and Ashmore, M.R. (1995a) Exposure of Commuters to Carbon Monoxide in Mexico City, I. Measurement of In-Vehicle Concentrations, *Atmospheric Environment*, **29**(4): 525–532.

Fernandez-Bremauntz, A.A. and Ashmore, M.R. (1995b) Exposure of Commuters to Carbon Monoxide in Mexico City, II. Comparison of In-Vehicle and Fixed-Site Concentrations, *Journal of Exposure Analysis and Environmental Epidemiology*, **5**(4): 497–510.

Flachsbart, P.G. (1989) Effectiveness of Priority Lanes in Reducing Travel Time and Carbon Monoxide Exposure, *Institute of Transportation Engineers Journal*, **59**(1): 41–45.

Flachsbart, P.G. (1995) Long-Term Trends in United States Highway Emissions, Ambient Concentrations, and In-Vehicle Exposure to Carbon Monoxide in Traffic, *Journal of Exposure Analysis and Environmental Epidemiology*, **5**(4): 473–495.

Flachsbart, P.G. (1999a) Exposure to Exhaust and Evaporative Emissions from Motor Vehicles, in *Urban Traffic Pollution*, Schwela, D. and Zali, O., Eds., World Health Organization, Geneva, Switzerland, 89–132.

Flachsbart, P.G. (1999b) Human Exposure to Carbon Monoxide from Mobile Sources, *Chemosphere: Global Change Science*, **1**: 301–329.

Flachsbart, P.G. (1999c) Models of Exposure to Carbon Monoxide inside a Vehicle on a Honolulu Highway, *Journal of Exposure Analysis and Environmental Epidemiology*, **9**(3): 245–260.

Flachsbart, P.G. and Brown, D.E. (1985) *Surveys of Personal Exposure to Vehicle Exhaust in Honolulu Microenvironments*, Department of Urban and Regional Planning, University of Hawai'i at Manoa, Honolulu, HI.

Flachsbart, P.G. and Brown, D.E. (1986) A Seasonal Study of Personal Exposure to CO in Indoor and Outdoor Microenvironments of Honolulu, presented at the 79th Annual Meeting of the Air Pollution Control Association, Minneapolis, MN.

Flachsbart, P.G. and Brown, D.E. (1989) Employee Exposure to Motor Vehicle Exhaust at a Honolulu Shopping Center, *The Journal of Architectural and Planning Research*, **6**(1): 19–33.

Flachsbart, P.G. and Ott, W.R. (1984) *Field Surveys of Carbon Monoxide in Commercial Settings Using Personal Exposure Monitors,* Report No. EPA 600/4-84-019, Office of Monitoring Systems and Quality Assurance, U.S. Environmental Protection Agency, Washington, DC.

Flachsbart, P.G. and Ott, W.R. (1986) A Rapid Method for Surveying CO Concentrations in High-Rise Buildings, *Environment International,* **12**: 255–264.

Flachsbart, P.G., Mack, G.A, Howes, J.E., and Rodes, C.E. (1987) Carbon Monoxide Exposures of Washington Commuters, *Journal of the Air Pollution Control Association*, **37**(2): 135–142.

Flachsbart, P., Ott, W., and Switzer, P. (2003) Long-Term Trends in Exposure to Carbon Monoxide on a California Arterial Highway, presented at the 96th Annual Meeting and Exhibition of the Air and Waste Management Association, San Diego, CA.

Fugas, M. (1975) Assessment of Total Exposure to an Air Pollutant, presented at the International Symposium on Environmental Sensing and Assessment, Las Vegas, NV.

Godish, T. (2004) *Air Quality*, 4th ed., CRC Press, Boca Raton, FL.

Grad, F.P., Rosenthal, A.J., Ruckett, L.R., Fay, J.A., Heywood, J., Kain, J.F., Ingram, G.K., Harrison, D. Jr., and Tietenberg, T. (1975) *The Automobile and the Regulation of Its Impact on the Environment*, University of Oklahoma Press, Norman, OK.

Haagen-Smit, A.J. (1966) Carbon Monoxide Levels in City Driving, *Archives of Environmental Health*, **12**(5): 548–551.

Hampson, N.B. and Norkool, D.M. (1992) Carbon Monoxide Poisoning in Children Riding in the Back of Pickup Trucks, *JAMA: Journal of the American Medical Association,* **267**: 538–540.

Howitt, A.M. and Altshuler, A. (1999) The Politics of Controlling Auto Air Pollution, in *Essays in Transportation Economics and Policy: A Handbook in Honor of John R. Meyer*, Gomez-Ibanez, J., Tye, W.B., and Winston, C., Eds., Brookings Institution Press, Washington, DC, 223–255.

Jain, K.K. (1990) *Carbon Monoxide Poisoning*, Warren H. Green, Inc., St. Louis, MO.

Johnson, J.H. (1988) Automotive Emissions, in *Air Pollution, the Automobile, and Public Health*, Watson, A.Y., Bates, R.R., and Kennedy, D., Eds., National Academy Press, Washington, DC., 39–75.

Johnson, T. (1984) *A Study of Personal Exposure to Carbon Monoxide in Denver, Colorado*, Report No. EPA-600/4/84-014, U.S. Environmental Protection Agency, Environmental Monitoring Systems Laboratory, Research Triangle Park, NC.

Johnson, T. and Paul, R.A. (1983) *The NAAQS Exposure Model (NEM) Applied to Carbon Monoxide,* Report No. EPA 450/5-83-003, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Jordan, B.C., Richmond, H.M., and McCurdy, T. (1983) The Use of Scientific Information in Setting Ambient Air Standards, *Environmental Health Perspectives*, **52**: 233–240.

Klepeis, N.E., Tsang, A.M., and Behar, J.V. (1996) *Analysis of the National Human Activity Pattern Survey (NHAPS) Respondents from a Standpoint of Exposure Assessment,* Report No. EPA 600/R-96/074, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.

Klepeis, N.E., Nelson, W.C., Ott, W.R., Robinson, J.P., Tsang, A.M., Switzer, P., Behar, J.V., Hern, S.C., and Engelmann, W.H. (2001) The National Human Activity Pattern Survey (NHAPS): A Resource for Assessing Exposure to Environmental Pollutants, *Journal of Exposure Analysis and Environmental Epidemiology*, **11**(3): 231–252.

Law, P.L, Lioy, P.J., Zelenka, M.P., Huber, A.H., and McCurdy, T.R. (1997) Evaluation of a Probabilistic Exposure Model Applied to Carbon Monoxide (pNEM/CO) using Denver Personal Exposure Monitoring Data, *Journal of the Air and Waste Management Association*, **47**(4): 491–500.

Lee, K., Yanagisawa, Y., Spengler, J.D., and Nakai, S. (1994) Carbon Monoxide and Nitrogen Dioxide Exposures in Indoor Ice Skating Rinks, *Journal of Sport Science*, **12**: 279–283.

Lynn, D.A., Ott, W.R., Tabor, E.C., and Smith, R. (1967) Present and Future Commuter Exposures to Carbon Monoxide, presented at the 60th Annual Meeting of the Air Pollution Control Association, Cleveland, OH.

Mage, D. (1991) A Comparison of the Direct and Indirect Methods of Human Exposure, in *New Horizons in Biological Dosimetry: Proceedings of the International Symposium on Trends in Biological Dosimetry*, Gledhill, B.L. and Mauro, F., Eds., Wiley-Liss, New York, NY, 443–454.

MVMA (1990) *MVMA Motor Vehicle Facts & Figures '90*, Motor Vehicle Manufacturers Association of the United States, Detroit, MI.

NAPCA (1970) *Air Quality Criteria for Carbon Monoxide*, National Air Pollution Control Administration, Public Health Service, Environmental Service, Report No. AP-62, U.S. Department of Health, Education, and Welfare, Washington, DC.

NIOSH (1996) *Preventing Carbon Monoxide Poisoning from Small Gasoline-Powered Engines and Tools [NIOSH Alert],* Report No. 96-118, Public Health Service, National Institute for Occupational Safety and Health, U.S. Department of Health and Human Services, Cincinnati, OH.

NRC (2000) *Modeling Mobile-Source Emissions*, National Research Council, National Academies Press, Washington, DC.

NRC (2003) *Managing Carbon Monoxide Pollution in Meteorological and Topographical Problem Areas*, National Research Council, National Academies Press, Washington, DC.

Ortolano, L. (1984) *Environmental Planning and Decision Making*, John Wiley & Sons, New York, NY.

Ortolano, L. (1997) *Environmental Regulation and Impact Assessment,* John Wiley & Sons, New York, NY.

Ott, W.R. (1982) Concepts of Human Exposure to Air Pollution, *Environment International*, **7**: 179–196.

Ott, W.R. (1983–1984) Exposure Estimates Based on Computer Generated Activity Patterns, *Journal of Toxicology — Clinical Toxicology,* **21**(1&2): 97–128.

Ott, W.R. (1985) Total Human Exposure: An Emerging Science Focuses on Humans as Receptors of Environmental Pollution, *Environmental Science and Technology,* **19**(10): 880–886.

Ott, W.R. and Eliassen, R. (1973) A Survey Technique for Determining the Representativeness of Urban Air Monitoring Stations with Respect to Carbon Monoxide, *Journal of the Air Pollution Control Association*, **23**(8): 685–690.

Ott, W.R. and Flachsbart, P.G. (1982) Measurement of Carbon Monoxide Concentrations in Indoor and Outdoor Locations Using Personal Exposure Monitors, *Environment International,* **8**: 295–304.

Ott, W.R., Mage, D.T., and Thomas, J. (1992) Comparison of Microenvironmental CO Concentrations in Two Cities for Human Exposure Modeling, *Journal of Exposure Analysis and Environmental Epidemiology*, **2**(2): 249–267.

Ott, W.R., Rodes, C.E., Drago, R.J., Williams, C., and Burmann, F.J. (1986) Automated Data-Logging Personal Exposure Monitors for Carbon Monoxide, *Journal of the Air Pollution Control Association*, **36**(8): 883–887.

Ott, W.R., Switzer, P., and Willits, N. (1993a) *Carbon Monoxide Exposures inside an Automobile Traveling on an Urban Arterial Highway: Final Report on the 1980-81 Field Study*, SIMS Technical Report No. 150, Department of Statistics, Stanford University, Stanford, CA.

Ott, W.R., Switzer, P., and Willits, N. (1993b) Trends of In-Vehicle CO Exposures on a California Arterial Highway over One Decade, presented at the 86th Annual Meeting and Exhibition of the Air and Waste Management Association, Denver, CO.

Ott, W.R., Thomas, J., Mage, D.T., and Wallace, L. (1988) Validation of the Simulation of Human Activity and Pollutant Exposure (SHAPE) Model Using Paired Days from the Denver, CO, Carbon Monoxide Field Study, *Atmospheric Environment*, **22**(10): 2101–2113.

Padgett, J. and Richmond, H. (1983) The Process of Establishing and Revising National Ambient Air Quality Standards, *Journal of the Air Pollution Control Association*, **33**(1): 13–16.

Paul, R.A. and Johnson, T. (1985) *The NAAQS Exposure Model (NEM) Applied to Carbon Monoxide: Addendum,* Report No. EPA 450/5-85-004, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC.

Paul, R.A., Johnson, T., and McCurdy, T. (1988) Advancements in Estimating Urban Population Exposure, presented at the 81st Annual Meeting of the Air Pollution Control Association, Dallas, TX.

Pedersen, D.H. and Sieber, W.K. (1988) *National Occupational Exposure Survey. Volume III: Analysis of Management Interview Responses,* Publication No. 89-103, National Institute of Occupational Safety and Health, U.S. Department of Health and Human Services, Cincinnati, OH.

Penney, D.G., Ed. (1996) *Carbon Monoxide*, CRC Press, Boca Raton, FL.

Pisarski, A.E. (1992) *New Perspectives in Commuting Based on Early Data from the 1990 Decennial Census and the 1990 Nationwide Personal Transportation Study,* Report No. FHWA-PL/92-026, Federal Highway Administration, U.S. Department of Transportation, Washington, DC.

Radford, E.P. and Drizd, T.A. (1982) *Blood Carbon Monoxide Levels in Persons 3–74 Years of Age: United States, 1976-80,* PHS 82-1250, Public Health Service, National Center for Health Statistics, U.S. Department of Health and Human Services, Hyattsville, MD.

Ramsey, J. (1966) Concentrations of Carbon Monoxide at Traffic Intersections in Dayton, Ohio, *Archives of Environmental Health,* **13**(1): 44–46.

Richmond, H., Langstaff, J., Johnson, T., Capel, J., Rosenbaum, A., and McCurdy, T. (2003) EPA's Air Pollutants Exposure Model (APEX): Estimation of Carbon Monoxide Population Exposure in Los Angeles — A Case Study, presented at the 13th Annual Conference of the International Society of Exposure Analysis, Stresa on Lago Maggiore, Italy.

Sexton, K. and Ryan, P.B. (1988) Assessment of Human Exposure to Air Pollution: Methods, Measurements, and Models, in *Air Pollution, the Automobile, and Public Health*, Watson, A.Y., Bates, R.R., and Kennedy, D., Eds., National Academy Press, Washington, DC, 207–238.

Shikiya, D.C., Liu, C.S., Kahn, M.I., Juarros, J., and Barcikowski, W. (1989) *In-Vehicle Air Toxics Characterization Study in the South Coast Air Basin,* Office of Planning and Rules, South Coast Air Quality Management District, El Monte, CA.

Spengler, J.D., Stone, K.R., and Lilley, F.W. (1978) High Carbon Monoxide Levels Measured in Enclosed Skating Rinks, *Journal of the Air Pollution Control Association*, **28**(8): 776–779.

Szalai, A., Ed. (1972) *The Use of Time: Daily Activities of Urban and Suburban Populations in Twelve Countries*, Mouton and Co., The Hague, The Netherlands.

TRB (1995) *Expanding Metropolitan Highways: Implications for Air Quality and Energy Use*, Transportation Research Board, National Academy Press, Washington, DC.

Tsang, A.M. and Klepeis, N.E. (1996) *Descriptive Statistics Tables from a Detailed Analysis of the National Human Activity Pattern Survey (NHAPS) Data,* Report No. EPA 600/R-96/148, Office of Research and Development, U.S. Environmental Protection Agency, Washington, DC.

U.S. Census Bureau (2003) *Statistical Abstract of the United States: 2003*, U.S. Department of Commerce, Washington, DC.

U.S. Code (1990) Clean Air Act, as amended by PL 101-549, November 15, 1990. U. S. C. §§42 7401-7671q.

USEPA (1991) *Air Quality Criteria for Carbon Monoxide*, Report No. 600/8-90/045F, Office of Health and Environmental Assessment, Environmental Criteria and Assessment Office, U.S. Environmental Protection Agency, Research Triangle Park, NC.

USEPA (1992) *Review of the National Ambient Air Quality Standards for Carbon Monoxide: 1992 Reassessment of Scientific and Technical Information*, Report No. EPA 452/R-92-004, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC.

USEPA (1994) *Methodologies for Estimating Emission and Travel Activity Effects of TCMs*, Report No. EPA 420-R-94-002, Office of Mobile Sources, U.S. Environmental Protection Agency, Washington, DC.

USEPA (2000) *Air Quality Criteria for Carbon Monoxide,* Report No. EPA 600/P-99/001F, Office of Research and Development, National Center for Environmental Assessment, U.S. Environmental Protection Agency, Washington, D.C, http://www.epa.gov/NCEA/pdfs/coaqcd.pdf (accessed March 22, 2004).

USEPA (2003) *National Air Quality and Emissions Trends Report, 2003 Special Studies Edition*, Report No. EPA 454/R-03-005, Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency, Research Triangle Park, NC, http://www.epa.gov/oar/aqtrnd03 (accessed May 24, 2004).

Van Wijnen, J.H., Verhoeff, A.P., Jans, H.W.A., and Van Bruggen, M. (1995) The Exposure of Cyclists, Car Drivers and Pedestrians to Traffic-Related Air Pollutants, *International Archives of Occupational and Environmental Health*, **67**: 187–193.

Walsh, M. (1996) EPA Bans Leaded Gasoline, *Car Lines*, **96**(2): 17–18.

Wilcosky, T.C. and Simonsen, N.R. (1991) Solvent Exposure and Cardiovascular Disease, *American Journal of Industrial Medicine*, **19**: 569–586.

Wilson, A.L., Colome, S.D., and Tian, Y. (1993a) *California Residential Indoor Air Quality Study, Volume 1: Methodology and Descriptive Statistics,* GRI-93/0224.1, Gas Research Institute, Chicago, IL.

Wilson, A.L., Colome, S.D., and Tian, Y. (1993b) *California Residential Indoor Air Quality Study, Volume 1: Methodology and Descriptive Statistics, Appendices,* GRI-93/0224.2, Gas Research Institute, Chicago, IL.

Yocom, J.E., Clink, W.L., and Cote, W.A. (1971) Indoor/Outdoor Air Quality Relationships, *Journal of the Air Pollution Control Association*, **21**(5): 251–259.

Yu, L., Hildemann, L., and Ott, W. (1996) A Mathematical Model for Predicting Trends in Carbon Monoxide Emissions and Exposures on Urban Arterial Highways, *Journal of the Air and Waste Management Association*, **46**(5): 430–440.

Ziskind, R.A., Fite, K., and Mage, D.T. (1982) Pilot Field Study: Carbon Monoxide Exposure Monitoring in the General Population, *Environment International*, **8**: 283–293.

Ziskind, R.A., Rogozen, M., Carlin, T., and Drago, R. (1981) Carbon Monoxide Intrusion into Sustained-Use Vehicles, *Environment International*, **5**: 109–123.