Environmental Exposure to Benzene: An Update

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During the 1990s, several large-scale studies of benzene concentrations in air, food, and blood have added to our knowledge of its environmental occurrence. In general, the new studies have confirmed the earlier findings of the U.S. Environmental Protection Agency Total Exposure Assessment Methodology (TEAM) studies and other large-scale studies in Germany and the Netherlands concerning the levels of exposure and major sources. For example, the new studies found that personal exposures exceeded indoor concentrations of benzene, which in turn exceeded outdoor concentrations. The new studies of food concentrations have confirmed earlier indications that food is not an important pathway for benzene exposure. The results of the National Health and Nutrition Examination Survey on blood levels in a nationwide sample of 883 persons are in good agreement with the concentrations in exhaled breath measured in about 800 persons a decade earlier in the TEAM studies. Major sources of exposure continue to be active and passive smoking, auto exhaust, and driving or riding in automobiles. New methods in breath and blood sampling and analysis offer opportunities to investigate short-term peak exposures and resulting body burden under almost any conceivable field conditions. — Environ Health Perspect 104(Suppl 6):1129–1136 (1996)

Key words: benzene, exposure, indoor air, outdoor air, personal monitors, TEAM study, body burden, breath, blood

Introduction

Much of our knowledge of nonoccupational exposure to benzene was supplied throughout the 1980s by the U.S. Environmental Protection Agency (U.S. EPA) Total Exposure Assessment Methodology (TEAM) studies of volatile organic compounds (VOCs) (1). These studies employed personal air quality monitors to measure direct personal exposures of approximately 800 persons in about eight areas in the United States between 1980 and 1987 (2–13). The participants were selected on a strict probability sampling basis to represent about 800,000 persons in these areas. Measurements of indoor and outdoor air, drinking water, and exhaled breath were made to supplement the personal air measurements. In a pilot study (2,14), measurements were also made in food and beverages; since few VOCs and no benzene was detected, those measurements were not repeated in the main study. The basic results of the TEAM study as they apply to benzene may be summarized as follows (15–20):

- Benzene was not found, or was found in insignificant amounts, in water, food, and beverages. More than 99% of the total personal exposure was through air.
- Mean personal air exposures exceeded indoor air concentrations, which in turn exceeded outdoor air concentrations. A global average personal exposure was about 15 μg/m³ (range 7–29 μg/m³). Indoor concentrations were measured only in the 1987 TEAM studies in Los Angeles, CA, Baltimore, MD, and Bayonne, NJ, and appeared to be on the order of 10 μg/m³. Outdoor concentrations had a global average of 6 μg/m³ (range 2–19 μg/m³).
- No effect on personal exposure of living close to major fixed sources of benzene (oil refineries, storage tanks, chemical plants) could be detected in Beaumont, TX (2,3); Bayonne and Elizabeth, NJ (5–8); or Los Angeles, Antioch, and Pittsburg, CA (9–11).
- The overwhelming source of benzene exposure for smokers was mainstream cigarette smoke (15). Smokers had an average benzene body burden about 6 to 10 times that of nonsmokers, and received about 90% of their benzene exposure from smoking (Figure 1). Roughly half the total benzene exposure in the United States was borne by smokers.
- For nonsmokers, most benzene exposure ultimately is derived from auto exhaust or gasoline vapor emissions. This includes most of the benzene exposure due to outdoor air, indoor exposures due to intrusion of evaporative gasoline fumes from attached garages (21), and personal activities such as driving (Figure 2). A portion of the exposure is due to environmental tobacco smoke (15). A small portion (about 6%) of the exposure is due to major point sources of benzene, such as petrochemical plants or refineries.

Two large-scale European studies (22,23) confirmed the TEAM study results for indoor and outdoor benzene concentrations. The study in Germany (23) also confirmed the effect of environmental tobacco smoke (15). A small portion (about 6%) of the exposure is due to major point sources of benzene, such as petrochemical plants or refineries.

Recent Studies

About half-a-dozen large-scale studies of personal or indoor air levels of benzene

Figure 1. Sources of benzene exposure: smokers. A typical smoker takes in roughly 2 mg benzene/day; about 1.8 mg is delivered by mainstream smoke (55 μg/cigarette × 32 cigarettes per day). Source: U.S. EPA TEAM studies.
A 1991 study (24) have been conducted since 1990. They are briefly described below.

**Personal Exposure Studies**

A 1991 study (24) took place in 128 homes in Woodland, California, a community in a largely agricultural region. Personal, indoor, and outdoor benzene concentrations were measured using both Tenax (Eina Research Institute, Arnhem, the Netherlands) and evacuated canister samplers. Good agreement was noted between the side-by-side Tenax and canisters. Mean concentrations were 5.0, 4.0, and 1.2 μg/m³ for the personal, indoor, and outdoor samples.

Day and night 12-hr average concentrations of benzene were measured for 58 residents of Valdez, Alaska (25). The mean benzene concentrations in the personal, indoor, and outdoor samples were 20, 16, and 5 μg/m³ during the summer, and 28, 25, and 11 μg/m³ during the winter, respectively.

Personal exposures to benzene were measured over a 3-hr period in the evening for 49 nonsmoking females in Columbus, Ohio (26). The median value in 25 homes with a smoker was 4.0 μg/m³ compared to 2.4 μg/m³ in 24 homes without smokers. The difference was statistically significant.

Personal exposures to benzene as measured in the TEAM studies and in the Valdez and Woodland studies are summarized in Table 1. Outdoor and indoor benzene values in the TEAM, Valdez, and Woodland studies are summarized in Tables 2 and 3.

**Indoor Air Studies**

A nationwide Canadian study (27) measured 24-hr indoor air concentrations of benzene in 754 randomly selected homes. Benzene mean indoor concentrations were 6.39, 5.60, 2.72, and 6.98 μg/m³ in the winter, spring, summer, and fall seasons, respectively.

Indoor and outdoor 48-hr average concentrations of benzene were measured at 161 homes throughout much of California (28). The Pro-Tek charcoal badges formerly manufactured by E.I. duPont (Newark, DE) were used. Indoor mean concentrations were 8.3 μg/m³ compared to 6.1 μg/m³ outdoors.

Seventeen volunteers in Windsor, Canada, wore 3-stage adsorbent tubes with pumps in three microenvironments: at home, at work, and during commuting (29). Benzene concentrations were 3.5, 4.7, and 15.7 μg/m³ in these three locations during summer 1991 and 2.7, 2.7, and 15.1 μg/m³ during winter 1992. Outdoor levels near homes were 3.8 and 2.0 μg/m³ during summer and winter, respectively. A later study (summer 1992) considered various microenvironments. Benzene levels averaged 2.2 μg/m³ in homes of 26 asthmatics, 4.6 μg/m³ in 13 samples from home rooms, 6.0 μg/m³ in 17 samples collected during commuting, 20.8 μg/m³ in 39 samples from four bingo halls, and 34.5 μg/m³ in two taverns.

Brown and Crump (30) reported on a study of 173 homes in Avon, England. Passive Tenax tubes (Perkin-Elmer) collected

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**Table 1.** Personal air concentrations (μg/m³) of benzene measured in the TEAM, Valdez, and Woodland studies.

| Site   | Household, estimated no. | Year, season | Time  | n | Mean | SE  | Geom mean | 25  | 50  | 75  | 90  | 95  | Max  |
|--------|--------------------------|--------------|-------|---|------|-----|-----------|-----|-----|-----|-----|-----|------|
| NJ1    | 130,000                  | 1981, fall   | Day   | 340 | 26.2 | 2   | 11        | 7   | 17  | 32  | 65  | 81  | 270  |
| NJ3    | Unweighted data          | 1983, winter | Day   | 47  | 21.0 | 2   | 16        | 9   | 16  | 26  | 46  | 62  | 64   |
| GNC    | 130,000                  | 1982, spring | Day   | 24  | 7.9  | 2   | 8         | –   | 8   | 13  | –   | –   | 36   |
| A-P    | 91,000                   | 1984, spring | Day   | 67  | 8.5  | 1   | 7         | 5   | 6   | 11  | 17  | 21  | 25   |
| LA1    | 360,000                  | 1984, winter | Day   | 112 | 19.1 | 2   | 15        | 10  | 15  | 23  | 35  | 51  | 86   |
| LA2    | 330,000                  | 1984, summer | Day   | 50  | 10.5 | 2   | 7         | 3   | 7   | 12  | 25  | 34  | 54   |
| LA3    | Unweighted data          | 1987, winter | Day   | 33  | 21.6 | 6   | 13        | 7   | 13  | 21  | 40  | 139 | 163  |
| LA4    | Unweighted data          | 1987, summer | Day   | 32  | 13.6 | 2   | 10        | 6   | 12  | 19  | 22  | 32  | 42   |
| BAL    | 70,000                   | 1987, spring | Day   | 40  | 13.7 | 3   | 9         | 5   | 7   | 13  | 26  | 84  | 98   |
| VAL    | Unweighted data          | 1990, summer | Day   | 70  | 16.4 | 2   | 9         | –   | 11  | 22  | 32  | 45  | 129  |
| VAL    | Unweighted data          | 1991, winter | Day   | 55  | 25.4 | 5   | 14        | 7   | 13  | 23  | 70  | 130 | 210  |
| WDL    | 30,000                   | 1990, spring | Day   | 58  | 23.6 | 4   | 13        | 7   | 12  | 26  | 65  | 100 | 170  |

Abbreviations: n, number; SE, standard error; geom mean, geometric mean. NJ1, Bayonne–Elizabeth, NJ; NJ3, Bayonne–Elizabeth, NJ; GNC, Greensboro, NC; A-P, Antioch–Pittsburg, CA; LA1, Los Angeles, CA; LA2, Los Angeles, CA; LA3, Los Angeles, CA; LA4, Los Angeles, CA; BAL, Baltimore, MD (Dundalk); VAL, Valdez, AK; WDL, Woodland, CA.
**Table 2. Household-weighted outdoor air concentrations (µg/m³).**

| Site   | Households, estimated no. | Year, season | Time | n  | Mean | SE  | Geom mean | Percentile |
|--------|---------------------------|--------------|------|----|------|-----|-----------|------------|
|        |                           |              |      |    |      |     |           |            |
| NJ1    | 40,000                    | 1981, fall   | Day  | 88 | 9.5  | 0.9 | 3.8       | 12 50 75 90 95 Max |
|        |                           |              | Night| 64 | 8.6  | 1.1 | 4.1       | 2.2 6.7 11 15 24 91 |
| NJ3    | Unweighted data           | 1983, winter | Day  | 8  | 3.9  | 0.8 | -         | 2.7 4 5.4 7.3 - 7.3 |
| A-P    | 25,000                    | 1984, spring | Day  | 10 | 2    | 0.6 | 1.5       | 0.9 13 1.6 6.3 - 6.3 |
| LA1    | 120,000                   | 1984, winter | Day  | 24 | 13   | 1.3 | 11        | 8.1 14 18 21 22 35 |
| LA2    | 110,000                   | 1984, summer | Day  | 24 | 4.2  | 0.8 | 3.2       | 2.3 3.1 4.8 8.7 12 15 |
| LA3    | Unweighted data           | 1987, winter | Day  | 41 | 4.7  | 0.5 | 3.8       | 2.6 3.8 6.2 8.7 12 14 |
| LA4    | Unweighted data           | 1987, summer | Day  | 38 | 3.4  | 0.4 | 2.8       | 1.9 2.6 4.8 6.6 8.7 12 |
| VAL    | Unweighted data           | 1990, summer | Day  | 30 | 4.8  | 0.4 | 4.1       | 3.1 5 7 8 9 10  |
| VAL    | Unweighted data           | 1991, winter | Day  | 26 | 5    | 1   | 3.6       | 2 5 7 10 11 14 |
| WDL    | 10,000                    | 1990, spring | 24 hr| 48 | 1.2  | 0.9 | 1.1       | 0.8 1.1 1.4 1.9 - 3 |

*Abbreviations: n, number; SE, standard error; geom mean, geometric mean; NJ1, Bayonne–Elizabeth; NJ; NJ3, Bayonne–Elizabeth; NJ; A-P, Antioch–Pittsburg, CA; LA1, Los Angeles, CA; LA2, Los Angeles, CA; LA3, Los Angeles, CA; LA4, Los Angeles, CA; VAL, Valdez, AK; WDL, Woodland, CA.*

**Table 3. Indoor air concentrations (µg/m³).**

| Site   | Households, no. | Year, season | Time | Room | n  | Mean | SE  | Geom mean | Percentile |
|--------|-----------------|--------------|------|------|----|------|-----|-----------|------------|
|        |                 |              |      |      |    |      |     |           |            |
| LA3    | Unweighted data | 1987, winter | Day  | LR  | 36 | 9.9 | 1.4 | 7.3       | 4.2 7.1 13 19 32 44 |
|        |                 |              | Day  | Kit | 38 | 11  | 2.6 | 6.5       | 3.7 7.5 12 21 30 97 |
|        |                 |              | Night| Kit | 36 | 15  | 2.2 | 9.7       | 5 11 18 40 46 53 |
| LA4    | Unweighted data | 1987, summer | Day  | LR  | 40 | 6.5 | 0.9 | 4.9       | 2.8 4.6 9.4 15 20 24 |
|        |                 |              | Day  | Kit | 38 | 5.5 | 0.8 | 4.4       | 2.5 4.7 6.7 9.9 14 28 |
|        |                 |              | Night| Kit | 37 | 6.5 | 1.2 | 4.4       | 2.3 4.5 7.4 13 23 44 |
| VAL    | Unweighted data | 1990, summer | Day  | LR  | 30 | 13  | 4   | 8.1       | 4 8 14 19 41 96 |
|        |                 |              | Night| LR  | 30 | 18  | 5.2 | 7.6       | 4 8 22 29 120 120 |
| VAL    | Unweighted data | 1991, winter | Day  | LR  | 29 | 26  | 4.4 | 17        | 7 16 34 62 81 86 |
|        |                 |              | Night| LR  | 27 | 24  | 4.6 | 14        | 7 16 28 74 79 83 |
| WDL    | 10,000          | 1990, spring | 24 hr| LR  | 104| 4.7 | 1.1 | 2.5       | 1.3 2.2 5.1 8.3 - 130 |

*Abbreviations: n, number; SE, standard error; geom mean, geometric mean; LR, living room; Kit, kitchen. LA, Los Angeles, CA; VAL, Valdez, AK; WDL, Woodland, CA.*

28-day samples in the living room and main bedroom of the home for 1 year. Thirteen sets of 12-month outdoor samples were also collected over the course of the study (November 1990–February 1993). The mean indoor concentration was 8 µg/m³ (n = 3000 samples) compared to an outdoor mean of 5 µg/m³ (n = 125).

**Ambient Concentrations**

Benzene concentrations were reported for 586 ambient air samples collected from 10 Canadian cities (T Dann, unpublished data). The overall mean was 4.4 µg/m³, with Ottawa and Montreal ranging between 5.1 and 7.6 µg/m³. A more recent survey (T Dann and D Wang, unpublished data) found similar levels, with three rural sites ranging from 0.6 to 1.2 µg/m³.

Twenty-four-hour average benzene levels have been measured every 12th day at about 20 sites throughout California since 1986 (31). Statewide average annual values fluctuated between 5 and 7 µg/m³ until 1993 and 1994, when they dropped to about 4 µg/m³ (Figure 3). This decline appears to be real and may be due to one or more of several factors: a) the 50% reduction in hydrocarbon emissions mandated for new cars; b) the Stage II vapor recovery controls recently in effect; c) a reduction in benzene content in gasoline down to the 1% mandated in the 1990 Clean Air Act Amendments.

The California database also allows analysis of seasonal variation. A clear sinusoidal curve is apparent, with winter values about twice summer values (Figure 4). This may be due to changes in the blend of the gasoline toward greater volatility in the winter or to increased likelihood of inversions during the winter.

The mean personal, indoor, and outdoor values of benzene measured in these more recent studies are compared in Table 4.

**In-Vehicle Studies**

The largest study of in-vehicle benzene exposure continues to be the 200-trip study (32) of Los Angeles commuters carried out in the summer and winter seasons. This study found an average benzene exposure of 13 ppb (40 µg/m³) for commuters during rush hour, on the order of 5 times the concentration measured at a fixed outdoor site.

A small study in North Carolina (33) also showed in-vehicle concentrations to 8 times background ambient levels. A second small study in Boston (34) resulted in passenger levels 1.5 times roadway levels on an interstate highway.

More recently, a study (35) of benzene levels in two-passenger vehicles during...
typical commutes in the New Jersey--New York area resulted in measured exposures of 9 to 12 μg/m³ in suburban and turnpike conditions, and 26 μg/m³ in the Lincoln Tunnel. The author stated that the concentrations during the commutes to New York City were about 10 times the ambient background concentration measured the same day in suburban New Jersey.

Unfortunately, none of the studies measured the benzene concentration in the gasoline used, so it is not possible to determine whether the lower concentrations in the later studies might be due to lower amounts of benzene in gasoline.

**Gasoline Spill Study**

A study of exposure to benzene while showering with gasoline-contaminated groundwater (36) was carried out in a home in North Carolina. The ground water had a measured benzene concentration of 292 μg/liter, well above the U. S. EPA's Maximum Contaminant Level of 5 μg/liter. Three 20-min showers on consecutive days resulted in peak shower-stall concentrations of 800 to 1670 μg/m³. Bathroom concentrations reached 370 to 500 μg/m³, and concentrations in the remainder of the house peaked (0.5–1 hr later) at 40 to 140 μg/m³. The inhalation dose during the 20-min shower ranged from 80 to 100 μg. A dermal dose of 160 μg was also calculated, using measured breath concentrations. The combined dose of about 250 μg from the 20-min shower is roughly equal to the mean total daily inhalation dose of about 200 μg for all nonsmokers in the TEAM study (assuming 15 μg/m³ × 14 m³/day alveolar inspiration).

**Body Burden**

Benzene in the blood of 883 persons was measured (37) as part of the National Health and Nutrition Examination Survey (NHANES III). These blood concentrations were compared with the breath concentrations measured in about 800 persons in the TEAM studies of the 1980s. Since the TEAM study measurements were made using mixed breath, the breath values were multiplied by 10/7 to account for a dead space estimated at 30% of the volume of an inhaled breath. Theoretically, one might expect that if the two populations are comparable, the ratio of blood to alveolar air concentrations for corresponding percentiles should remain constant at the magnitude of the blood/air partition coefficient for benzene, for which several estimates ranging between 7 and 10 have been

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**Figure 3.** Ambient benzene in California, annual averages across all sites 1986–1994. Annual average outdoor benzene concentrations at about 20 sites in California. At each site, a 24-hr average is taken every 12 days. The decline in 1993 to 1994 may be due to reduced emissions from automobiles. Source: California Air Resources Board, data from 20+ cities.

**Figure 4.** California benzene monthly averages. Seasonal variation in outdoor benzene at about 20 sites in California. Higher values in winter may be due to seasonally varying gasoline formulations, and perhaps to increased frequency of atmospheric inversions. The decline in benzene concentrations over the past few years is consistent over all seasons. Source: California Air Resources Board, data from all (about 20) sites.

| Table 4. Mean benzene concentrations (μg/m³) reported in recent studies. |
|---------------------------------------------------------------|
| Reference | Location | n   | Personal | Indoor | Outdoor |
|-----------|----------|-----|----------|--------|---------|
| Goldstein et al., 1992 (25) | Alaska | 112 | 24 | 20 | 8 |
| Sheldon et al., 1991 (24) | California | 120 | 5.0 | 4.0 | 1 |
| Heavner et al., 1994 (26) | Ohio | 49 | 3.2 | | |
| Brown and Crump 1996 (30) | England | 173 | | 8 | 5 |
| Wilson et al., 1993 (28) | California | 161 | | 8.3 | 6 |
| Fellin and Otson, 1993 (27) | Canada | 754 | | 5.4 | |
| Dann (unpublished data) | Canada | 586 | | | 4 |
| CARB 1988–1992 (37) | California | 3000 | | | 7 |
| CARB 1993–1994 (37) | California | 1000 | | | 4 |
made. However, the actual observed ratio of blood to alveolar air concentrations appears to decrease with increasing concentrations, from 24 and 33 at the 16th and 25th percentiles through a range of 11 to 8 at progressively higher percentiles (Table 5). This is similar to the observation (38) of a blood/breath ratio of about 20/38 for an unexposed population of nonsmoking nurses, while the ratio for an occupationally exposed cohort of smokers was about 7.7. Both these findings may be explained by the possibility suggested by Travis and Bowers (39) that at low concentrations, a saturable blood component (e.g., proteins) binds a limited amount of benzene, making it unavailable for distribution throughout the body or elimination in breath. Travis and Bowers estimated the capacity of the blood proteins to be 90 ng/liter, based on the observations of Perbellini et al. (38). They also estimated the plasma partition coefficient to be 9.0. Adding the NHANES/TEAM data to those of Perbellini et al. and adjusting the Travis/Bowers model to fit all the data, one arrives at a lower estimate of blood capacity of 30 ng/liter, and a slightly lower plasma partition coefficient of 8 (Figure 5).

Concentrations in Food

There were reports in the 1970s of benzene being found at ppm levels in some foods such as eggs (40). However, in a special study that was part of the TEAM pilot study in 1980, breath measurements before and after eating eggs showed no increase in benzene. Also, no effect on benzene levels in breath from eating eggs or any other food item could be discerned from regressions on all participants in the main TEAM study, using the participants' responses to a detailed questionnaire on food intake. It is possible that minor levels of benzene in foodstuffs could still have been present and not detected in breath due to efficient metabolism by the liver, which receives materials from the gut directly before they enter the blood stream. However, it was thought that major concentrations in food would be detectable in breath; since they were not, it was concluded that food and beverages were an unimportant pathway for benzene exposure.

Two recent studies of benzene levels in foods have confirmed that conclusion by finding negligible quantities in nearly all foods measured. In one study by the U.S. Food and Drug Administration (FDA), more than 50 foods were analyzed for benzene (41). Most of these were under 2 ng/g ppbw (parts per billion by weight) benzene. Exceptions included strawberry preserves (38 ng/g), taco sauce (9 and 22 ng/g), duck sauce (7 ng/g), and barbecue sauce (5 ng/g). The authors speculated that the added benzoates and ascorbates in these foods might react to form benzene; thus, if either one or the other were removed, the benzene might no longer be formed. In a second study (42), 57 foods were measured, with only shelled peanuts and fried eggs giving positive results, each at 30 ng/g, again far below the parts per million (ppm) levels previously reported. A recent Canadian review of benzene exposures (43) concluded that food and drinking water each contributed only about 0.02 µg/kg benzene per day compared to a total intake of 2.4 µg/kg per day from airborne exposures (3.3 µg/kg/day if exposed to cigarette smoke). Thus, airborne exposure accounts for 98 to 99% of total benzene intake for Canadian nonsmokers.

Discussion

The general finding from previous studies that personal exposures to benzene exceed indoor air concentrations, which in turn exceed outdoor air concentrations, has been confirmed by the more recent studies. Two of the three personal monitoring studies mentioned above had somewhat lower mean personal exposures to benzene than had previously been reported. One such study (24) was in a small rural community in California, which also had a lower mean outdoor benzene value (1.2 µg/m³) than has been previously reported. The second study (26) included only 3-hr exposures in the evening at home; to the extent that all other personal exposure studies included time spent in vehicles, where benzene exposures have been shown to range up to 40 µg/m³, such a study limited to the home microenvironment might be expected to produce smaller personal exposures. Therefore, both of these studies would be expected to be at the low end of benzene exposures.

On the other hand, the outdoor concentrations of about 5 µg/m³ in Valdez were similar to outdoor concentrations in the various TEAM study sites, but the indoor and personal concentrations (20 and 24 µg/m³) were considerably greater than in all TEAM study sites except for Los Angeles in the winter. It may be speculated that persons in frontier-type situations make more use of gasoline-powered instruments such as chain saws, snow blowers, and snowmobiles than persons in urban communities. It may also be that the requirements for warming up automobiles for extended periods, and the larger amounts of benzene that are found in Alaskan and Canadian gasoline blends, led to higher exposures from attached garages and driving.

Considering that the TEAM Studies showed a range of benzene exposures from 7 to 26 µg/m³ (16), the range observed since 1990 of 3.2 to 24 µg/m³ provides no firm evidence as yet for a downward trend in benzene exposures.

A second finding from previous studies, that benzene levels were increased in homes with smokers, was also replicated. The new study (26) found a significant increase of 1.6 µg/m³, which is less than

![Figure 5](image-url)
the increases of 3.5 and 4.5 μg/m³ found in the TEAM and West German studies (15,23) but represents about the same percentage increase of 50 to 67% compared to nonsmoking homes. The Windsor study (29) that found increased benzene concentrations in bingo halls and taverns, where smoking is prevalent, might also be viewed as confirming the effect of smoking on indoor benzene concentrations.

Several small studies replicated the findings of an earlier major study in Los Angeles that showed increased benzene exposures while driving. The later studies appeared to involve much smaller exposures but also had much smaller outdoor concentrations, so the ratio of personal exposure to outdoor concentration continued to be in the neighborhood of 5 to 10. The smaller concentrations could be due to differences in location (Los Angeles vs North Carolina and New Jersey—New York) but could also reflect reductions in the amount of benzene in the gasoline. The results of the national fuel survey carried out by the American Automobile Manufacturers Association (44) indicate that the goal of 1% benzene in gasoline set by the 1990 Clean Air Act Amendments has been very nearly met, with the average for premium, intermediate, and regular gasoline for the winter of 1994 to 1995 being 0.9, 0.9, and 1.1% by volume, respectively. This is a considerable reduction compared to the 2 to 3% levels that were probably common during the large California in-vehicle study. However, it should also be noted that the amount of benzene in the exhaust may be related only weakly to the amount of benzene in the gasoline. One study (45) indicated that exhaust benzene remained unchanged at about 5% of total hydrocarbon emissions whether the gasoline burned contained 1 or 3% benzene by volume.

A large number of food groups were tested but found to contain negligible amounts of benzene. This corroborated the conclusions of the TEAM studies, which found no evidence of food contributions to body burden of participants.

Although nearly all the studies reviewed here have been more in the nature of confirmatory studies rather than breaking new ground, the study of benzene exposures while showering in gasoline-contaminated water presented new data of considerable value. The 20-min exposure from this source was the same order of magnitude as a full day’s exposure to benzene for a typical nonsmoker. However, a smoker (of more than five cigarettes a day) using the same gasoline-contaminated water would still get most of his or her exposure through smoking—an indication of the extensive exposure encountered by some 43 million U.S. citizens. Since the number of persons affected by such spills is very small, the effect on the national exposure budget for benzene is also very small.

Apart from these presumably very rare gasoline spill situations, there may be a larger number of cases where well water is contaminated by benzene at low concentrations. A number of studies have reported finding benzene at levels on the order of 5 ng/liter (ppb) in surface and well waters. However, these levels correspond to a daily intake of <10 ng benzene, assuming 2 liters of water drunk daily. This amount is only 0.5% of the average daily intake for nonsmokers of 200 ng from air. Thus, it is concluded that the effect of contaminated water on total benzene intake is negligible.

It may fairly be asked whether any of the differences observed in various studies at different locations and times are dependent on the different methods employed. The TEAM studies employed Tenax-GC with active pumping, as did the later Woodland and Valdez studies; thus, all the studies using personal monitors used very similar or identical methods.

The indoor air studies in England and Canada employed passive (diffusive) samplers with extended monitoring periods. The English investigators performed a number of tests on the effect of extended sampling on the net uptake of different VOCs by the Perkin-Elmer sorbent tubes containing Tenax-TA. They found that the more volatile VOCs such as benzene and toluene had net diffusive uptake that declined over time, probably because of back diffusion off the tubes. For sorbent tubes exposed to a concentration of 2500 μg/m³ toluene, the diffusive uptake rate declined to 71% of the ideal after 7 days and 54% after 28 days. For benzene, the net diffusive uptake was 30% after 28 days. For the less volatile compounds such as xylenes, decane, and trimethylbenzenes, the sampling rate stayed nearly constant over the 28-day period. Therefore, given a month-long sampling period, an average monthly exposure can be chosen for any given chemical; however, for chemicals more volatile than the xylenes, this rate will only be an average value from a declining curve. This means that the early part of the sampling period for these volatile compounds may be underrepresented because of back diffusion losses from the substrate. However, if the average sampling rate is correctly chosen, this would not cause a bias, only greater variability than exists in fact.

The Canadian investigators used commercial samplers with a charcoal sorbent (the 3M organic vapor badge), followed by solvent desorption using carbon disulfide (CS₂). Carbon disulfide is well known to have a contamination problem with benzene; however, the Canadian investigators developed their own methods for cleaning the CS₂, and report no serious problems with contamination. It is not clear whether they used the ideal sampling rate for benzene or determined an effective sampling rate for the 1-week sampling period.

Both Tenax and charcoal have problems with nonzero background benzene concentrations. The Tenax must be carefully cleaned to avoid such problems. The early TEAM studies had high and variable backgrounds of benzene equivalent to about 5 ± 3 μg/m³ on the Tenax cartridges. This would lead to decreased precision, although since average backgrounds were subtracted from each raw datum, it is not clear whether any bias remained. The later TEAM studies reduced backgrounds to the equivalent of about ± 0.5 μg/m³, reducing the uncertainty in the estimated exposures considerably. The charcoal badges also have high backgrounds of benzene; however, the extended sampling period should have provided sufficient benzene to reduce the background effect. Therefore it is unlikely that a significant bias or lack of precision has affected the personal, indoor, or outdoor air concentrations of benzene.

The initial breath measurements in the TEAM New Jersey study of 1981 employed Tedlar (Nutech Corp., Durham, NC) bags stored in a van. Because of the possibility that exhaust vapors had penetrated the Tedlar bags, future studies passed pure helium over the bags at positive pressure to remove this possible source of contamination. Both higher exposures and higher breath concentrations were noted in this period, but about 45% of the New Jersey participants were smokers compared to 22% of the California participants; thus the higher breath concentrations observed in the New Jersey participants may well have been due to the higher smoking rate.
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