A Reevaluation of the Literature Regarding the Health Assessment of Diesel Engine Exhaust

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While the International Agency for Research on Cancer (IARC) classified diesel exhaust (DE) as a “probable” carcinogen in 1989 based primarily on “sufficient” animal data, other investigators have since concluded that the lung tumors found in the rat studies were a result of particle overloading. Subsequent health risk assessments of DE have not used the rat cancer data. The U.S. Environmental Protection Agency (EPA), in developing its 2002 Health Assessment Document (HAD) for DE, primarily considered the epidemiology studies of railroad workers and truck drivers to develop health risk assessments of DE. However, both sets of epidemiology studies have serious weaknesses that make them unsuitable for cancer risk assessment. Major shortcomings were the lack of contemporaneous measurements of exposures to DE, difficulties with exposure history reconstruction, and adequately accounting for other exposures such as gasoline exhaust and cigarette smoke. To compound these problems, there was not, and there is still not, a specific exposure marker for DE. Interestingly, in the underground mining industry, where diesel exposures are much higher than observed in railroad workers and truck drivers, there was no increase in lung cancer. These problems and concerns led the U.S. EPA to conclude that while DE was a “likely” carcinogen, a unit risk value or range of risk cannot be calculated from existing data and that the risk could be zero. In addition, the DE emissions have changed and continue to change with the implementation of new emission control technologies. The HAD recognized this fact and noted that further studies are needed to assess new diesel engine emissions. Recent chemical characterization studies on low-emitting diesel engines with catalyzed particulate filters have shown emissions rates for several chemicals of concern that are even lower than comparable compressed natural gas (CNG)-fueled engines. With lower emissions, better fire safety, and improved cost-effectiveness of new low-emitting diesels compared to CNG, current efforts to restrict use of low-emitting diesels seems misguided.
The HAD selected a reference dose (daily inhalation exposure that is not likely to cause harmful effects during a lifetime) of 5 \( \mu g/m^3 \) for DE. However, it stated that there is no known method for separating DE from other airborne fine particulate matter (PM); PM 2.5 is currently regulated at 15 \( \mu g/m^3 \). The document noted that diesel was a portion of PM, and PM would be evaluated separately.

**EXPOSURE MEASUREMENT**

One of the major challenges in conducting diesel epidemiology research is measuring exposures to DE. The variability of DE (due to differences in types of engines and fuels as well as variability in the work cycle over time) makes consistent exposure measurement specific to DE difficult in the occupational or environmental setting. Various measurement techniques and surrogates have been used to estimate DE exposure concentrations, including nitrogen oxides, total carbon, specific PAHs, particle counts, size-selective gravimetric techniques (e.g., PM10 or PM1.0), and gravimetric techniques that adjust for certain nondiesel species (e.g., nicotine, combustible organics). All of these techniques require critical assumptions about the characteristics of DE and of the nature of the exposure environment. Size-selective gravimetric measurement techniques assume that all particles below the separation size are from DE. Particle counting methods make size, shape, and density assumptions and assume all small particles are DE.

Recent exposure measurements have favored elemental carbon (EC) as the leading candidate for separating out the DE component of air particulate for several reasons:

1. EC is always present in diesel engine combustion, and at significant levels.
2. EC analysis is specific for combustion products, with minor interference from tobacco smoke, road dust, pollen, and other common organic and inorganic particulates.
3. EC analysis is more sensitive than gravimetric techniques; the limit of detection is on the order of 100 times lower for EC than for gravimetric techniques.

A comparison of DE particulate matter measurement methods found that EC analysis was more precise, sensitive, and free from interferences than other methods currently in use for occupational exposure limits in the United States, Canada, and Germany (Ramachandran & Watts, 2003).

Although EC has been the leading candidate for DE exposure determination, serious limitations remain. The EC content of DE is not consistent over engine size or technology. Also, the range of EC content can be three times higher during some parts of the duty cycle than at idle. And EC, while fairly specific to combustion sources, is not exclusive to diesel. Table 1 demonstrates these limitations but also confirms that EC is a substantial component of current DE.

As diesel technologies improve and the diesel fleet becomes cleaner, the amount and fraction of EC in DE are becoming...
smaller. A recent study showed that particulate traps with low-sulfur fuel can reduce the levels of EC emitted from a diesel engine to below the levels from a compressed natural gas (CNG) engine (Ayala et al., 2001, 2002, 2003). Other combustion processes such as fireplaces, cooking, forest fires, gasoline engines, and power plants also emit EC, and the relative contributions of these other sources are becoming more significant.

**EXPOSURE MEASUREMENT IN CURRENT EPIDEMIOLOGY STUDIES**

In November 2002, the Health Effects Institute (HEI) held a workshop with a panel of more than 50 experts on environmental and exposure monitoring of DE. The HEI review concluded that “Current studies have concurrent measures of exposure, such as EC (and sometimes other measures such as distance from traffic), but these are not sufficiently specific to provide exposure-response information for diesel alone in an environment containing other combustion particles” (Health Effects Institute, 2003, p. 12). Therefore, EC is not likely to be a useful marker for DE in the environment and is probably not useful in most occupational settings where there are multiple exposures. An exception would be in areas where there are no other potential sources of EC (e.g., hard rock mining).

**RAILROAD INDUSTRY STUDIES**

Epidemiology studies of railroad workers are frequently cited in diesel quantitative health risk assessment. Since exposures were not measured contemporaneously for these study populations, DE exposure classifications for the workers were based on industrial hygiene measurements of sample populations in similar, though later, settings. Several aspects of emission exposure measurement in these studies are noteworthy and suggest that these studies do not provide a sound basis for quantitative DE health risk assessment.

Garshick et al. (1987, 1988) studied mortality from lung cancer and DE exposure in a cohort of railroad workers from 1959 to 1980. They considered DE exposure to be a dichotomous (yes/no) variable. In this study, railroad worker DE exposure categories were based on industrial hygiene evaluations of 39 job categories in 4 railroads as reported by Woskie et al. (1988a, 1988b). Garshick et al. found “a small but significantly elevated risk for lung cancer” in this population associated with adjusted respirable particulate (ARP) exposure, which they assumed was equivalent to DE exposure. However, some aspects of the workers’ exposures to DE as reported in the Woskie study are noteworthy. Woskie et al. found significant amounts of cigarette smoke (up to 67% of total respirable particulate) in many of the respirable particulate samples. Samples were adjusted for cigarette smoke by measuring nicotine and subtracting a weight based on a calculated ratio of nicotine in cigarette smoke. The remaining particulate called ARP, was assumed to represent exposure to DE (Table 2). The authors also acknowledged some possible contribution of sand, dirt, and other particulate matter to the calculated ARP. While speculating that these “background” contributions might be in the range of 30 µg/m³, they did not correct for them.

Current DE air concentration measurements commonly use EC as a marker to separate DE from other airborne particulates. DE particles are primarily carbon, and the EC method was developed subsequent to the Woskie et al. (1988a, 1988b) study to reduce interference from other sources. A more recent industrial hygiene study of exposure to DE in a railroad work environment by Verma et al. (1999) found total respirable particulate matter concentrations in the same range as reported by Woskie et al. (1988a). However, Verma et al. (1999) observed corresponding mean EC levels from 4.4 to 17.8 µg/m³ for the different exposure groups, which are much lower than the ARP levels reported by Woskie et al. (1988a). Importantly, Verma et al. (1999) showed that the exposure to EC was a minor part (3 to 16%) of railroad worker respirable particulate matter exposures. Results are summarized in Table 3.

The results of the more recent Verma et al. (1999) study and the lack of a specific and sensitive indicator of diesel exposure in the earlier Woskie et al. (1988a, 1988b) studies raise the possibility that the earlier study overestimated the DE content of the respirable particulate exposure. The EC content of DE varies as shown earlier, but even if DE mass of the samples were twice the measured EC mass in the Verma study, DE would still be a small fraction of the respirable particulate exposure.

**TRUCKING INDUSTRY STUDIES**

Epidemiology studies of DE exposure in the trucking industry have also been cited as a basis for quantitative risk assessment of DE. However, concerns have been expressed about the appropriateness of the exposure measurements used in these studies, too.

Steenland et al. (1990, 1992, 1998) studied lung cancer in the trucking industry and related lung cancer experience to DE exposures (Zaebst et al., 1991). To develop exposure estimates for the 1950 to 1990 exposure period, industrial hygiene measurements of EC were collected in 1990 in occupational settings in the trucking industry using an impactor that was adjusted to select particles below 1 µm. The EC content of these samples

<table>
<thead>
<tr>
<th>Engine type</th>
<th>EC content range (%)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heavy-duty diesel</td>
<td>65–85</td>
<td>Fujita et al. (1998), Watson et al. (1998)</td>
</tr>
<tr>
<td>Light-duty diesel</td>
<td>45–75</td>
<td>Norbeck et al. (1998b)</td>
</tr>
</tbody>
</table>
TABLE 2
Railroad worker personal exposures by job group

<table>
<thead>
<tr>
<th>Career exposure group</th>
<th>Job group</th>
<th>Environmental tobacco, geometric mean (µg/m³)</th>
<th>Adjusted respirable particulate (ARP), geometric mean (µg/m³)</th>
<th>Total respirable particulate, geometric mean (µg/m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clerks</td>
<td>Clerks/station agent</td>
<td>64</td>
<td>17</td>
<td>99</td>
</tr>
<tr>
<td>Signal maintainers</td>
<td>Signal maintainers</td>
<td>9</td>
<td>49</td>
<td>58</td>
</tr>
<tr>
<td>Engineer/firer</td>
<td>Freight</td>
<td>17</td>
<td>73</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>Yard</td>
<td>28</td>
<td>49</td>
<td>78</td>
</tr>
<tr>
<td></td>
<td>Passenger</td>
<td>18</td>
<td>39</td>
<td>57</td>
</tr>
<tr>
<td>Braker/conductor</td>
<td>Freight conductor</td>
<td>43</td>
<td>52</td>
<td>113</td>
</tr>
<tr>
<td></td>
<td>Freight braker</td>
<td>34</td>
<td>88</td>
<td>127</td>
</tr>
<tr>
<td></td>
<td>Passenger</td>
<td>5</td>
<td>85</td>
<td>90</td>
</tr>
<tr>
<td></td>
<td>Yard</td>
<td>54</td>
<td>92</td>
<td>146</td>
</tr>
<tr>
<td></td>
<td>Hostler</td>
<td>6</td>
<td>191</td>
<td>197</td>
</tr>
<tr>
<td>Shop</td>
<td>Electrician</td>
<td>41</td>
<td>134</td>
<td>177</td>
</tr>
<tr>
<td></td>
<td>Machinist</td>
<td>34</td>
<td>114</td>
<td>152</td>
</tr>
<tr>
<td></td>
<td>Supervisor, laborer, and</td>
<td>54</td>
<td>130</td>
<td>199</td>
</tr>
<tr>
<td></td>
<td>other shop workers</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Note. From Woskie et al. (1988a).

‘The sum of tobacco and ARP masses equals total respirable particulate mass for each individual sample, but unlike arithmetic means, the geometric means of the tobacco and ARP results do not necessarily add together to equal the geometric mean of the total.

was used as an indicator of DE concentrations. Overall geometric mean exposures to submicrometer-sized EC ranged from 3.8 µg/m³ for over-the-road (highway) drivers to 13.8 µg/m³ for dock workers.

In addition, to establish a relationship between EC and respirable dust in various work areas, the Steenland studies collected 56 paired samples; one of each pair was for respirable dust and the other was for submicrometer-sized EC. EC content averaged slightly less than 20% of the respirable dust concentrations for a subset of 18 pairs. In order to improve the regression analysis at higher levels, 38 pairs were excluded if the elemental carbon concentration was less than 10 µg/m³. If the 38 sample pairs with little EC content were included, the average percentage of EC in the complete set of 56 samples would be much less than 20%.

Several aspects of the Steenland et al. study are noteworthy. First, in these results it is important to recognize that the relationship between EC and total respirable particulate only applies to a subset of samples that were predetermined to contain at least 10 µg/m³ EC. Second, the Steenland analysis assumed that the EC level outside the truck correlated with exposure of the driver to DE from his truck. However, a 2003 paper by Borak et al. reported that exposure is minimal for the driver from the vehicle he is driving. Third, Steenland assumed that dieselization (switch from gasoline to diesel engines) occurred earlier than it actually did in the Class 8 heavy duty trucking fleet. Furthermore, there was little dieselization of the delivery and short-haul fleets during the study period. Therefore, the primary particle and EC exposures in this study were from gasoline engines. In fact, in the Steenland study, gasoline exhaust exposure and vehicle miles traveled correlate as well with the reported increased risk as their diesel counterparts (Bunn & Slavin, 2001). Steenland et al. (1998) concluded that “Our results depend on estimates about unknown past exposures, and should be viewed as exploratory. They conform reasonably well to recent estimates for diesel-exposed railroad workers done by the California EPA, although those results themselves have been disputed” (p. 220).

A more recent study of electric utility industry workers using diesel trucks and equipment (Whittaker et al., 1999) found that worker exposure to the EC fraction of DE was approximately

TABLE 3
Occupational exposures to railroad DE

<table>
<thead>
<tr>
<th>Location</th>
<th>Elemental carbon, mean (µg/m³)</th>
<th>Total respirable particulate, mean (µg/m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Turnaround all yards, personal</td>
<td>8.4</td>
<td>250</td>
</tr>
<tr>
<td>Turnaround yard 1 area</td>
<td>6.2</td>
<td>150</td>
</tr>
<tr>
<td>Heavy repair yard 1 area</td>
<td>4.4</td>
<td>160</td>
</tr>
<tr>
<td>On board locomotives area</td>
<td>17.8</td>
<td>110</td>
</tr>
</tbody>
</table>

Note. From Verma et al. (1999).
3 to 5 µg/m³ and the percent of EC in total airborne carbon was 5%. The total respirable particulate concentration was not determined. These results plus those of the Steenland study indicate that DE is a small part of the overall respirable particulate exposure in the trucking industry.

OTHER EXPOSURE MEASUREMENTS

In 2002, International Truck and Engine Corporation published data for airborne carbon particulate in several diesel truck and engine plants (Bunn et al., 2002). Results, summarized in Table 4, show total carbon concentrations of 42 to 194 µg/m³ and mean EC concentrations of 1.7 to 11 µg/m³ in several different occupational settings. The study included truck plants that manufacture diesel-powered medium or heavy-duty trucks and buses, engine plants that manufacture diesel engines, and a foundry with no known diesel sources. The EC exposure in the foundry may be from carbon volatilized from molten iron or carbonaceous materials added to sand molds.

These results show substantial nondiesel carbon exposures in occupations that involve work with diesel equipment. In particular, test cells, in which a diesel engine is kept running in an enclosed area, have the highest levels of EC (11 µg/m³) but still have much higher exposures to other carbon sources (139 µg/m³ total carbon). These other sources may include burn off of oil or paint from hot surfaces as new trucks are run and tested for the first time.

Thus, the EC results show that separating out DE particulate from non-DE particulate is more complex than was assumed in earlier measurements with less specific techniques. Moreover, in occupations where workers are considered to be exposed to diesel, DE is, in fact, only a small portion of the overall particulate exposure. The lack of exposure data therefore means that the occupational studies of diesel are more appropriately characterized as studies of combustion sources or fine particles than DE per se.

MINING INDUSTRY STUDIES

Miners have been extensively studied because of their exposure to numerous agents capable of affecting the respiratory tract including coal dust, crystalline silica, and radon. During the mining of various natural resources, diesel-engine equipment has in some cases been used for production and transportation. Therefore, miners are also a useful population in which to evaluate the relationship between lung cancer and DE exposure. As shown in Table 5, estimated exposures to DE in mines using diesel equipment (640 to 830 µg/m³) are more than an order of magnitude greater than in other occupations, such as truckers or railroad workers (10 to 70 µg/m³). The DE exposures in miners are proportionately higher and less confounded by other combustion aerosol exposures such as gasoline engine exhaust, cigarette smoke, and ambient combustion products. Another advantage to using health data from miners is that diesel-fueled equipment has been used in some mines for more than 60 yr, which allows sufficient time for the latency period of any potential lung cancer effects.

Using data from the International Agency for Research on Cancer (IARC, 1997) and additional studies recently published, Slavin (2001) separated epidemiologic studies of miners into

<table>
<thead>
<tr>
<th>Exposure type</th>
<th>Mean (µg/m³)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Truckers</td>
<td>10</td>
<td>Zaebst et al. (1991)</td>
</tr>
<tr>
<td>Railroad workers</td>
<td>70a</td>
<td>Woskie et al. (1988a), Hammond et al. (1988)</td>
</tr>
<tr>
<td>U.S. surface miners</td>
<td>88</td>
<td>MSHA (1998)</td>
</tr>
<tr>
<td>U.S. underground coal miners</td>
<td>640</td>
<td>MSHA (1998)</td>
</tr>
</tbody>
</table>

aTotal particulate minus tobacco smoke.
### TABLE 6
Lung cancer risk of miners exposed to diesel engine exhaust

<table>
<thead>
<tr>
<th>Type of mine/location</th>
<th>Study population</th>
<th>Lung cancer risk (95% CI)</th>
<th>Reference</th>
</tr>
</thead>
</table>
| Coal/United Kingdom   | 1674 Face workers, 1225 other underground workers, and 811 surface workers, who died in 1961, were evaluated for cause of death | SMR (face): 0.49  
SMR (other underground): 0.53  
SMR (surface): 0.82 | Liddell (1973) |
| Coal/Australia        | 213 Miners of which 205 worked underground; 83% had worked >10 yr underground | O/E: 0.2 (0–2.2)* | Armstrong et al. (1979) |
| Sulfide ore/Finland    | 597 Miners, who were first employed between 1954 and 1973 and worked underground for at least 3 yr, were followed until 1986 | O/E (local male population): 1.45 | Ahlman et al. (1991) |
| Coal/Poland           | 7065 Coal miner with pneumoconiosis; mortality from 1970–1991 assessed | SMR: 1.04 (0.88–1.22) | Starzynski et al. (1996) |
| Coal/Germany          | 4578 Miners were followed, 1980–1991 | SMR: 0.70 (0.5–1.0) | Morfeld et al. (1997) |
| Coal/Australia        | 23,630 Coal industry workers followed, 1973–1992; cohort included underground and surface miners and office personal | SIR: 0.74 (0.50–1.06) | Brown et al. (1997) |
| Potash/Germany        | 5536 Underground miners followed, 1970–1994; 3258 had worked >10 yr, held only one job >80% of the time, and never held >3 jobs; exposure to diesel was grouped by job category: production (high), maintenance (medium); workshop (low) | SMR (whole cohort): 0.78 (0.55–1.07)  
RR (subcohort, production vs. workshop): 2.17 (0.79–5.99) | Saverin et al. (1999) |
| Coal/Australia        | Extension of Brown et al. (1997): 24,139 coal industry workers followed, 1973–1997 | SIR: 0.65 (0.48–0.86) | Kirby et al. (2000) |

*CI calculated by IARC (1997), not in original publication.

cancer risk was less for diesel-exposed miners than for other workers in most of these studies. The risk of lung cancer in miners exposed to DE is similar to that found in miners working in mines without diesel equipment (Tables 6 and 7). Various measures of risk—standard mortality ratio (SMR), standard incidence ratio (SIR), odds ratio (OR), or relative risk (RR)—do not reveal significant increases in having or dying from lung cancer among either population of miners.

For the majority of studies given in Table 6, exposure to DE particulate was presumed. In some of these studies, efforts were made to evaluate only underground miners, where exposures to dust materials, including diesel, would have been significantly...


**TABLE 7**

Lung cancer risk of miners not exposed to diesel engine exhaust

<table>
<thead>
<tr>
<th>Type of mine/location</th>
<th>Study population</th>
<th>Lung cancer risk (95% CI)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coal/United Kingdom</td>
<td>Miners and ex-miners (number not given) aged 20–65 yr were evaluated for cause of death</td>
<td>SMR (underground): 0.70 (0.61–0.80)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Goldman (1965)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>SMR (surface): 0.92 (0.69–1.19)&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Coal/United Kingdom</td>
<td>Miners (number for coal miners not given) aged &gt;15 yr, who died between 1948 and 1967, were evaluated for cause of death</td>
<td>O/E (underground/local population): 0.79 (0.53–1.15)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Boyd et al. (1970)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>O/E (surface/local population): 0.99 (0.49–1.77)&lt;sup&gt;a&lt;/sup&gt;</td>
<td></td>
</tr>
<tr>
<td>Coal/United Kingdom</td>
<td>1003 Deaths among miners recorded 1974–1976 were evaluated for cause of death</td>
<td>O/E (local male population): 1.17 (0.69–1.47)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Rooke et al. (1979)</td>
</tr>
<tr>
<td>Coal/United States</td>
<td>553 Miners were followed, 1938–1966</td>
<td>SMR: 1.11 (0.3–2.85)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Enterline (1972)</td>
</tr>
<tr>
<td>Coal/United States</td>
<td>3726 Miners were followed, 1962–1972</td>
<td>SMR: 0.67 (0.43–0.99)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Costello et al. (1974)</td>
</tr>
<tr>
<td>Coal/United States</td>
<td>23,232 Miners were followed, 1959–1971</td>
<td>SMR: 1.13 (1.02–1.26)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Rockette (1977)</td>
</tr>
</tbody>
</table>

<sup>a</sup>CI calculated by IARC (1997), not in original publication.

Greater. Only one study correlated lung cancer risk with airborne dust levels (Saverin et al., 1999). However, the Saverin study had up to 70% smokers in the population studied and no controls for pack-years or environmental tobacco smoke, which seriously undermines the validity of the correlation reported. Waxweiler et al. (1973) (not shown in Table 6) evaluated underground miners from potash mines, but took into consideration when diesel equipment was introduced into the mines. The authors compared mortality rates in one mine that had been using diesel engines since 1949 with another mine that had not started using diesel engines until 1957. Although the SMRs for lung cancer were not provided, the authors noted that there were no significant differences in any causes of death between the miners from the two mines.

Because age-specific total mortality is often lower in miners than in the general population, the potential for confounding by the “healthy worker effect” needs to be considered. For lung cancer, however, one would not expect that the healthy worker effect to be a factor. Lung cancer is not foreshadowed by predisposing symptoms, like cardiovascular disease, in which workers might remove themselves from the workplace because of early signs or symptoms of heart or circulatory health problems. In addition, the finding of similar cancer rates in diesel and non-diesel mines makes a healthy worker effect less likely.

Overall, if there is a causal role for DE in the lung cancer risks reported in occupations such as railroad workers and truckers, the lung cancer “signal” should be overwhelming for underground mining populations where DE exposure is more than an order of magnitude higher. However, the data for underground miners does not demonstrate a lung cancer excess associated with DE, and in fact no clear difference can be seen between highly exposed underground miners and lightly exposed aboveground miners.

**EMISSIONS STUDIES IN DIESEL BUSES**

Borak et al. (2003) measured airborne diesel particulate inside school buses on a test track simulating pickup and delivery of students. Levels inside the buses were similar to background levels. The study used two methods to monitor diesel particulate matter: NIOSH 5040 (thermal/optical measure of elemental carbon) and Aethalometer (direct reading of black carbon). While the two methods gave comparable results, the Aethalometer demonstrated sensitivity to vibrations, which makes it unsuitable for routine monitoring in moving vehicles.

A recent study compared the emissions profiles of three currently available school bus configurations: conventional diesel,
FIG. 1. Relative toxicity of potency-weighted emissions. Chronic toxicity calculations adapted from OEHHAA (1999) and cancer potency calculations adapted from SCAQMD (2000).

low-emitting diesel, and natural gas (Ullman et al., 2003). The low-emitting diesel engine used Green Diesel Technology (catalyzed diesel particulate filter, a low-nitrogen oxides engine control module, and ultra-low-sulfur fuel). Twenty-one of the 41 TACs listed by the California Air Resources Board as being present in DE were not found in the exhaust of any of the vehicles tested, despite very low detection limits. The conventional diesel had the highest emissions for five TACs. Natural gas exhaust had higher levels of six TACs (acetaldehyde, acrolein, benzene, formaldehyde, methyl ethyl ketone, propionaldehyde) compared to low-emitting diesel. None of the TAC emissions of the natural gas vehicle were lower than the low-emitting diesel. For 8 of the 11 air quality emissions, low-emitting diesel was lower than natural gas, including nitrogen oxides, nitrogen oxide, particulate matter, soluble organic fraction of particulate matter, total hydrocarbons, nonmethane hydrocarbons, methane, and carbon monoxide. Both cancer and noncancer potency-weighted emissions were lowest for the low-emitting diesel school bus (see Figure 1) (Lapin et al., 2003). Overall, the low-emitting diesel technology had the lowest risk from emissions associated with EPA criteria pollutants and TACs.

Similarly, two recent studies by the California Air Resources Board (CARB) and BP compared emissions in transit buses using either diesel or CNG as fuel (CARB study: Ayala et al., 2002, 2003; Holmen & Ayala, 2002, Okamoto et al., 2002; Kado & Kuzmicky, 2003; BP study: Lev-On et al., 2002). These studies presented the same picture as in the Ullman study— the low-emitting diesel bus had the lowest air pollutants and TAC emissions. The cancer potency-weighted emissions in each of the three engine categories of school and transit buses presented in Figure 2 show remarkable comparability and consistency. In addition, mutagenic potency of emissions was highest for the CNG buses (Figure 3). Taken together, the transit and school bus studies challenge the notion that CNG-fueled vehicles are necessary to meet air quality goals.


FIRE SAFETY STUDY

A recent University of Maryland study (Chamberlain et al., 2002) investigated the fire safety of CNG school buses compared to their diesel counterparts. The researchers concluded that risk of death is 230 times greater for the occupants of CNG school buses compared to diesel buses. In addition, the worst-case fire scenarios in CNG buses are expected to be far more serious than diesel buses.

COST-EFFECTIVENESS ANALYSES

The incremental cost-effectiveness (CE) of diesel buses equipped with catalyzed diesel particulate filters (low-emitting diesel or LED) was compared to CNG-fueled buses (Cohen et al., 2003; Cohen, 2004). The CE ratio numerator reflected the costs of acquisition and operation, while the denominator reflected health costs of mortality and morbidity due primarily to particulate matter and ozone exposure. The health costs, expressed as quality adjusted life years (QALYs), were lower in the CNG-fueled transit bus compared to the LED transit, while the QALYs were similar between the CNG and LED school buses. However, the operating costs of the CNG buses were 6 to 10 times higher, leading to CE ratios that were much more favorable for the LED transit and school buses compared to the CNG-fueled buses.

RECENT ANIMAL STUDIES AND ONGOING RESEARCH

Seagrave et al. (2002) investigated the mutagenicity and toxicity of combined particulate and semivolatile organic fractions of either gasoline or diesel vehicle emissions in rats exposed by intratracheal instillation. Endpoints included cytotoxicity (enzyme and protein levels in lung fluid, ratio of lung/body weights), inflammation (lung fluid cell counts and cytokines), and lung parenchymal changes. Emissions from normal-running diesel and gasoline vehicles were equally potent on a per unit mass basis and produced similar transient elevations in toxic indicators. In contrast, the high-emitter diesel and gasoline engine exhausts were both more potent than exhaust from the normal-running engines. However, lung fluid and lung cell indicators basically returned to normal levels within 1 wk postexposure.

Harrod et al. (2003) found that mice exposed via inhalation to whole DE (30 or 1000 µg/m³ for 6 h/day for 7 days) had increased susceptibility to respiratory syncytial virus, a common pediatric respiratory pathogen. The relevance of these findings to ambient human exposures, which are about 10 times lower, needs to be further investigated.

Continued interest in the potential health effects of DE and the rapid changes in diesel emission control technology call for continued toxicological investigations (Mauderly, 2001). An industry-wide initiative lead by the HEI and Engine Manufacturers Association is underway to conduct further studies of the new low-emitting diesel technologies over the next 3 to 4 yr.

CONCLUSIONS

In conclusion, current studies do not provide an adequate basis for a quantitative cancer risk assessment of DE. However, there is continued scrutiny of a possible relationship between cancer and diesel exposure. The study of populations exposed to diesel is hampered by the lack of a marker that is specific to DE. Studies on diesel vehicles with the latest emission control technology show significant reductions in emission substances that are of concern for potential health effects. In this regard, these emissions are well below those observed with CNG-fueled vehicles. In addition, the fire safety risk is much lower with diesel-fueled vehicles compared to CNG. Diesel vehicles are also less costly to operate than CNG-fueled vehicles. Finally, changes in emissions, both in composition and particle size distribution, observed with the new emission control technology argue...
strongly that today’s diesels should be considered separately from those evaluated in the 2002 U.S. EPA DE HAD.

New research with better exposure measurements is needed before risk assessment of DE can be performed. Studies of miners by the National Institute of Occupational Safety and Health and the National Cancer Institute and of truckers by Harvard may provide useful data. In the meantime, reductions of particles and oxides of nitrogen by 99% from pre-1988 levels in diesel trucks will be accomplished with the current 2004, 2007, and 2010 U.S. and California regulations. Diesel particulate regulatory standards will be similar or stricter than gasoline or natural gas regulations by 2007 for heavy-duty vehicles and by 2010 for light-duty vehicles. In fact, fleets of “green diesel” buses using low-sulfur fuel have already achieved the 2007 and 2010 particulate emission levels.

REFERENCES


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