Short report

How serious are we about protecting workers health? The case of diesel engine exhaust

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ABSTRACT

Objectives Regulators frequently deviate from health-based recommendations when setting occupational exposure limits, but the impact on workers' health is rarely made explicit. We present a quantitative evaluation of the expected impact of recently proposed regulatory limits for occupational diesel engine exhaust (DEE) exposure on the excess burden of lung cancer (LC) in Europe.

Methods We used a lifetable approach, basing our analyses on the DEE exposure distribution in a large general population study, as well as the 5% prevalence used in earlier DEE burden calculations. We evaluated the effects of intervention on DEE exposures according to a health based limit (1 ug/m^3 of elemental carbon (EC)) and both Dutch (10 ug/m^3) and European (50 ug/m^3) proposed regulatory limit values. Results were expressed as individual excess lifetime risks (ELR), total excess number of cases and population attributable fraction of LC.

Results The ELR for the EU working population was estimated to be 341/10 000 workers based on our empirical exposure distribution and 46/10 000 workers based on the 5% prevalence. Implementing the proposed health based DEE limit would reduce the ELR by approximately 93%, while the proposed regulatory limits of 10 and 50 ug/m^3 EC would reduce the ELR by 51% and 21%, respectively.

Discussion Although the proposed regulatory limits are expected to reduce the number of DEE related LC deaths, the residual ELRs are still significantly higher than the targets used for deriving health-based risk limits. The number of additional cases of LC in Europe due to DEE exposure, therefore, remains significant.

Key messages

What is already known about this subject?

⇒ While setting occupational exposure limits, regulators frequently deviate from health-based limit recommendations, but the impact on workers' health is rarely explicitly evaluated.

⇒ Recently, several regulatory limits for diesel engine exhaust (DEE) were proposed in Europe that deviate significantly (a factor 10–50) from the health-based recommended limit.

What are the new findings?

⇒ These regulatory limits, when successfully enforced, are expected to reduce the estimated lifetime risk of DEE-related lung cancer, but the residual excess risk remains high.

How might this impact on policy or clinical practice in the foreseeable future?

⇒ Deviations from purely health-based limit recommendations should, in our view, be accompanied by an impact assessment describing the residual disease burden.

INTRODUCTION

One way to protect workers health is by establishing exposure limits in the workplace. Several international and national bodies have been charged with establishing these limits, and the process often starts with estimating exposure limit values based on either determining a No-Observed-Adverse-Effect-Level or Lowest-Observed-Adverse-Effect-Level or, when safe levels may not exist as for genotoxic carcinogens, by determining the exposure below which any remaining excess risks are considered acceptable risk (AR) or maximum tolerable risk, MTR. Curiously, for the later (stochastic) approach different cutoffs are used for environmental than for occupational exposures. For example, in Europe, for environmental exposures, a one in a million excess lifetime risk (ELR) is considered acceptable, while this is four in a one-hundred-thousand for occupational exposures (a 40-fold difference). This begs the question why, as a society, we accept that exposures in the work environment can bring more risk than exposures encountered in our general lives?

However, even if one accepts the premise that working life may be riskier, we need to make sure that exposure limits are set appropriately. In the case of occupational exposures the choice is often made to use the MTR, instead of the AR, which (in the Netherlands) is defined as an ELR of four in a thousand (note, US OSHA uses 1/1000 typically), so 100 times the AR and no less than 4000 times the risk considered acceptable for the general population.[1] The argument often provided for favouring an exposure limit value based on the MTR rather than AR, is that the latter is so low that certain activities or industries would need to be discontinued. At times even the MTR is considered to results in exposure limit values that are too strict, often because of concerns regarding the (economic or technical) feasibility to control exposures at these levels.[2] These considerations may be defensible, but the impact of ignoring the health-based recommended limit values on excess risk is rarely evaluated. Recently, the Health Council of the Netherlands in an advice to the state secretary of social affairs and employment, derived a health-based limit for occupational exposure to diesel engine exhaust (DEE), using the diesel lung cancer (LC) exposure–response relation of Vermeulen *et al*,[3] of 1 ug/m^3 elemental carbon (EC) based on the MTR for LC (4/1000 extra LC deaths due to 40 years of occupational exposure to DEE based on an 8-hour time-weighted average).[4] After further deliberation regarding the social and economic impacts, the exposure limit was eventually set at 10 ug/m^3 EC. An even higher limit of 50 ug/m^3 EC was recently set in the EU, which will become effective in general occupational environments in 2023 and in underground mining and tunnel construction in 2026.[5] Here, we evaluate the expected impact of using these higher exposure limit values instead of the MTR on the burden of LC due to occupational DEE exposure in the EU population.

METHODS

We previously described a lifetable approach to evaluate excess risks of LC due to DEE, comparing several different risk functions that had been proposed.[1] In a lifetable approach, excess risk is usually calculated by subtracting the estimated cumulative risk of LC in a hypothetical population that is unexposed from that in the same population that is exposed to DEE according to some specific exposure scenario (combination of exposure level and exposure duration) that is to be evaluated. To estimate the excess risk in a real-life population that is exposed according to multiple different exposure scenarios, we performed lifetable calculations for a representative sample of scenarios and averaged the resulting estimated ELRs. The effects of an exposure intervention (eg, enforcing some specific exposure limit) can then be evaluated by modifying the scenarios according to this intervention and repeating the calculations. A full description of the methods used and a worked example can be found in the online supplemental.

In brief, we selected relevant exposure scenarios from job histories that were obtained for the control population of the Synergy lung cancer case–control study.[6 7] This study, which comprises 14 hospital-based and population-based LC case–control studies from 13 European countries and Canada, was designed to investigate exposure–response relations for several potential occupational carcinogens. DEE in the Synergy study was estimated using a recently developed quantitative DEE Job Exposure Matrix (JEM)[6] that was linked to study participant job histories using the International Standard Classification of Occupations (ISCO)-68 coding classification.

We excluded information from subjects that lived outside the EU (ie, in either Canada or Russia; n=2862), were born before 1930 (n=6143), or were not old enough to provide complete job history information (ie, were younger than 65; n=8612). Of the remaining 3188 subjects, 2004 (63%) were never occupationally exposed to diesel, leaving 1184 exposure scenarios available for the lifetable analysis.

The average duration of DEE exposure among those that were exposed was 22 years. Yearly averaged DEE exposure exceeded 50 ug/m^3 in only 10 (0.8%) of scenarios and exceeded 10 ug/m^3 EC in 385 scenarios (33%). The prevalence of DEE exposure in the selected set of subjects used to derive these scenarios was notably higher than that used in published risk and/or impact assessments for occupational DEE exposure (37% vs 3.3%–8.4%),[8–10] in part due to the latter more focusing on high exposure occupations in high risk industries. To evaluate the impact of these differences, we present results assuming either that 95% of the population is never occupationally exposed or that occupational exposure is distributed as observed in the Synergy control population.

Table 1 Excess risks of lung cancer (LC), number of LC cases and population attributable fraction (PAF) according to different regulatory standards (ie, 1, 10 and 50 ug/m^3) in the EU

	No limit	50 ug/m^3	10 ug/m^3	1 ug/m^3
Prev (ever diesel exposure)=37%				
Excess lifetime risk of LC (per 10 000)	341	268	166	26
Expected excess cases of LC in EU#	779 891	614 567	380 099	59 524
PAF of LC	8.8%	7.1%	4.5%	0.73%
Prev (ever diesel exposure)=5%				
Excess lifetime risk of LC (per 10 000)	46	36	22	3
Expected excess cases of LC in EU*	104 995	82 738	51 172	8014
PAF of LC	1.3%	1.0%	0.63%	0.10%

*The number of subjects out of the present EU working population (229 million) that is expected to ever die from LC due to diesel exposure.
EU, European Union; LC, lung cancer.

For the lifetable calculations, background mortality rates (total and LC-specific) for men and women combined for EU member states were obtained from the Eurostat website for the year 2008 (the latest year of data collection in the Synergy study). We evaluated the excess risk of LC at age 80 by weighing the lifetable results for each exposure scenario according to the estimated current exposure distribution and after capping exposure levels according to different regulatory standards (ie, 1, 10 and 50 ug/m^3 EC). The exposure–response association used in these calculations was derived from Vermeulen *et al*.[3] Results are expressed as the estimated ELR and number of LC cases in the EU, assuming a working population of 229 million people, and as a population attributable fraction.

RESULTS

The lifetime excess risk of LC due to occupational DEE exposure in the EU working population is estimated at 341/10 000 workers based on the exposure levels and durations derived from the Synergy control population (table 1). Excess risks would be reduced to 268, 166 and 26 per 10 000 workers, after reducing maximum exposure levels to 50, 10 or 1 ug/m^3 EC, respectively. Because there were only few jobs in our sample where either past or current DEE exposure levels exceeded 50 ug/m^3 EC (0.8%), establishing a maximum exposure limit at this level had only little effect on estimated excess risk. Lifetime excess risk would be reduced by approximately 50% from 341 down to 166 per 10 000 workers when setting a maximum exposure limit of 10 ug/m^3 EC. Note, however, that the worker population to which this calculation applies includes many workers that are not occupationally exposed to DEE and that the excess risk is still much higher than that used for deriving the MTR. Using 10 ug/m^3 EC as a maximum exposure limit, the number of subjects out of the present EU working population (229 million) that is expected to ever die from LC due to DEE amounts to 380 000 LC cases. Using a more conservative prevalence estimate of only 5% of the working population being occupationally exposed to DEE (the number that was used in previous burden of disease calculations)[8–10] and assuming the stricter Dutch exposure limit of 10 ug/m^3 EC would apply, we still expect more than 50 000 workers to ever die of LC from DEE exposure.

DISCUSSION

The press release by the European Parliament that accompanied the new EU limit stated that: ‘In order to protect some 3.6 million workers in the EU potentially exposed to DEE, the parliament succeeded in including diesel fumes in the scope of the new rules…… The new rules should further lower the risk for workers of getting cancer, which remains the primary cause of work-related deaths across Europe’. The results from our calculations suggest that the new EU exposure limit for DEE will have only marginal impact on workers’ health and leaves much of the excess risk due to DEE exposure in place, thereby failing to protect the lives of many.

Our presented calculations are inherently uncertain with respect to exposure scenarios and how these exposure scenarios develop towards the future and what impact, where and when can be expected. However, as we noted before[1] due to the long latency of DEE-induced LC combined with the longevity of diesel engines such an impact may only been seen many years from now.

So how serious are we about workers’ health? The process of setting occupational exposure limits runs the risk of becoming a mere mathematical exercise, with little (normative) discussion or sense of what these numbers actually imply. Stakeholders that evaluate the feasibility of proposed standards and negotiate with governmental bodies rarely have access to detailed impact data as we present here. We may, therefore, insufficiently realise what the impact is of ignoring the health-based recommendations when choosing higher exposure limits for regulation. We believe that the process used to set exposure limits could be made more transparent by including an explicit social cost–benefit assessment under different regulatory limit scenarios. Only by having these numbers on the table the impact of (political) decisions can become apparent, thereby allowing us to answer the question whether we are serious about workers health and at what costs.

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Online Supplement: Additional methods and worked example

Excess risks, expected number of excess deaths, and population attributable fractions

We used a lifetable approach to estimate the impact of exposure interventions. This is a counterfactual approach where we estimate the attributable (excess) cumulative risk of death from lung cancer that is the result from exposure to diesel engine exhaust (DEE), while accounting for competing causes of death using information on (age-specific) cancer-specific and all-cause mortality rates.

We make the simplifying assumption that both exposures and mortality rates are constant over a period of one year from each age, leading to piecewise exponential survival, and define *x(t)* to be a function that returns age-specific exposure levels, which we will call an exposure scenario in what follows. We define S(t,*x(t)*) as the proportion of subjects with exposure scenario *x(t)* that remain at risk of dying from lung cancer at the beginning of the interval starting at age t, with S(0,*x(0)*) = 100% by definition. Under these assumptions and with age- and exposure-scenario specific all-cause and lung-cancer mortality rates equal to m(t,*x(t)*) and l(t,*x(t)*) respectively, the proportion of subjects dying in the 1 year interval from age t is:

$ND(t,x(t)) = S(t,x(t)) * (1-\exp(-m(t,x(t))))$

and the proportion dying from lung cancer:

$NC(t,x(t)) = l(t,x(t))) / m(t,x(t))) * ND(t,x(t))$

The cumulative proportion of the population that has died from lung cancer up until age T, i.e. the cumulative mortality risk at age T, is then calculated as:

$R(T,x(T)) = \sum_{t=1}^{T} \mathrm{NC}(t, x(t))$

For risk assessment purposes, typically T = 80 years is used. We define the “unexposed” exposure scenario as $X_0(t) = 0$ for $t \in N_0$ and assume that mortality rates obtained from general population registries can be used to estimate age-specific mortality rates for this “unexposed” scenario (which holds for agents with a relatively low exposure prevalence or weak effect), indicating these with $m_0(t,X_0(t))$ and $l_0(t,X_0(t))$ for all-cause and lung-cancer mortality respectively. To estimate mortality rates for an exposed population we assume a relative rate model, where an exposure-response function is available that allows estimation of age- and exposure scenario-specific lung cancer mortality rate ratios, which we indicate by h(t,X(t)). For any specific exposure scenario X(t) we then define:

$l(t,X(t)) = h(t,X(t)) * l_0(t,X_0(t))$ (for lung cancer mortality)

$m(t,X(t) = m_0(t,X_0(t)) + (h(t,X(t)) – 1) * l_0(t,X_0(t))$ (for all-cause mortality)

For the (common) situation where a cox-regression model is used to estimate the log-hazard ratio (β) as a (linear) function of cumulative DEE exposure, the lung cancer mortality rate ratio function could be defined as:

$h(t,X(t)) = \exp(\beta * \sum_{i=1}^{t} \mathrm{X(i)})$

To estimate the excess risk due to DEE exposure at age T, we subtract the cumulative lung cancer risk in an unexposed population from that in an exposed population:

$ER(T,X(T)) = R(T,X(T)) - R(T,X_0(T))$

To estimate the average excess risk at age T in a population with *p* varying exposure scenarios (i.e. different exposure levels and occupational histories) with optional weights $w_1,..,w_p$, we calculate the weighted average of scenario-specific excess risks:

$$\overline{\mathrm{ER}_{\mathrm{pop}}}(T) = \sum_{i=1}^{p} \mathrm{w}_i * \mathrm{ER}(\mathrm{T}, \mathrm{X}_\mathrm{i}) \,/ \sum_{i=1}^{p} \mathrm{w}_i$$

The expected number of excess deaths at age T in a population of size N_{pop} is then $\overline{\mathrm{ERpop}}(T) * \mathrm{N}_{\mathrm{pop}}$.

Finally, we estimate the population-attributable fraction (PAF) of lung cancer deaths due to DEE at age T as:

$$\overline{\mathrm{PAF}_{\mathrm{pop}}}(\mathrm{T}) = \sum_{\mathrm{i}=1}^{\mathrm{p}} \mathrm{w}_\mathrm{i} * \frac{\mathrm{ER}(\mathrm{T}, \mathrm{X}_\mathrm{i})}{\mathrm{R}(\mathrm{T}, \mathrm{X}_\mathrm{i})} / \sum_{\mathrm{i}=1}^{\mathrm{p}} \mathrm{w}_\mathrm{i}$$

Selection of exposure scenarios

We selected relevant exposure scenarios from job histories that were obtained for the control population of the Synergy lung cancer case-control study. Age-specific occupational DEE exposure was estimated using a recently developed quantitative JEM (DEE-JEM)[1] that was linked to study participant job histories using the ISCO-68 occupational coding classification.

We excluded information from subjects that lived outside the EU (i.e. in either Canada or Russia; n=2,862), were born before 1930 (n=6,143), or were not old enough to provide complete job history information (i.e. were younger than 65; n=8,612). Of the remaining 3,188 subjects, 2,004 (63%) were never occupationally exposed to DEE, leaving 1,184 exposure scenarios available for the lifetable analysis. All exposed scenarios were assigned the same weight, with the summed weight for these scenarios equal to the empirically derived prevalence of diesel exposure (i.e. 0.37).

The prevalence of DEE exposure in the selected set of subjects used to derive these scenarios was notably higher than that used in several recently published risk and/or impact assessments for occupational diesel exposure (37% versus 3.3%-8.4%). Although the difference is likely at least partly due to these latter studies focusing more on high exposure occupations in high risk industries, we evaluated excess risks and population attributable fractions also using a much lower summed weight of 0.05 for the exposed scenarios.

Estimation of age-specific background all-cause and lung cancer mortality rates

Mortality rate information was obtained for men and women combined from the Eurostat website for the year 2008 and was available as the number of deaths and size of the population at the start of the year in approximate 5-year age-categories. To estimate age-specific mortality rates, we fitted a penalized poisson regression spline model, assigning the midpoint for each age-category and transforming estimated probabilities of dying into mortality rates assuming these were constant within a single year.

DEE exposure-response relations with lung cancer mortality

We used the previously published exposure-response relation by Vermeulen et al.[2] that assumes a linear relation between log(RelativeRate) and cumulative exposure (using a 5-year lag) as follows:

log(RelativeRate) = 0.000982 * cumulative DEE exposure (in µg/m^3-years)

We accounted for the lag in the life-table calculations by discarding the last 5 years of exposure in calculating cumulative exposures at each age.

Example calculation

We provide a detailed example to allow interpretation of the calculations used to derive the estimated effect of limiting DEE exposure levels to 1 µg/m^3, assuming that the lifetime prevalence of DEE exposure is as observed in the Synergy study (i.e. 37%).

The risk of having died from lung cancer by the age of 80 in an unexposed population is estimated to be approximately 3.53%. Across the exposure scenarios, and without any of the new regulatory standards in place, the same risk ranges from 3.53% to 91.2%, with a mean of 4.45% and a median of 3.86%. The scenario for which the very high risk was estimated entailed an estimated exposure of 150 µg/m^3 DEE for 41 years, resulting in a cumulative exposure of 6,160 µg/m^3-years at age 80 and thus a relative rate of 420 for lung cancer mortality at that age. The average excess risk can be directly estimated from these numbers as (4.45%-3.53%)*37% = 340 per 100,000, which is within rounding error of what we report in the paper (341 per 100,000). With a working population of 229 million workers, the additional number of deaths from lung cancer in the EU can be estimated as 3400 * 229 = 778,600. The population attributable fraction can be estimated by noting that the 340 additional cases per 100,000 subjects would occur among a total number of cases of 4.45%*37% + 3.35*63% = 3,757 cases per 100,000 subjects, the ratio of which is 9.0% (we report 8.8%).

After regulatory standards have been applied, the maximum exposure levels to DEE are reduced to 1 µg/m^3. Risks across the intervened exposure scenarios range from 3.53% to 3.67%, with a mean and median of 3.60%. Following the same logic and calculations as before, the average excess risk now is (3.60%-3.53%)*37% = 26 per 100,000, the additional number of deaths in the EU is 260 * 229 = 59,540, and the PAF is 26/3443 = 0.8%.

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