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LETTER

Impact of excess NO_X emissions from diesel cars on air quality, public health and eutrophication in Europe

J E Jonson[1,5], J Borken-Kleefeld[2], D Simpson[1,3], A Nyíri[1], M Posch[4] and C Heyes[2]

[1] Norwegian Meteorological Institute, NO-0313, Oslo, Norway
[2] International Institute for Applied Systems Analysis (IIASA), Schlossplatz 1, 2361 Laxenburg, Austria
[3] Department of Space, Earth and the Environment, Chalmers University of Technology, Gothenburg, Sweden
[4] Coordination Centre for Effects (CCE), RIVM, PO Box 1, NL-3720 BA Bilthoven, The Netherlands
[5] Author to whom any correspondence should be addressed.

E-mail: j.e.jonson@met.no

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Abstract

Diesel cars have been emitting four to seven times more NO_x in on-road driving than in type approval tests. These 'excess emissions' are a consequence of deliberate design of the vehicle's after-treatment system, as investigations during the 'Dieselgate' scandal have revealed. Here we calculate health and environmental impacts of these excess NO_x emissions in all European countries for the year 2013. We use national emissions reported officially under the UNECE Convention for Long-range Transport of Atmospheric Pollutants and employ the EMEP MSC-W Chemistry Transport Model and the GAINS Integrated Assessment Model to determine atmospheric concentrations and resulting impacts. We compare with impacts from hypothetical emissions where light duty diesel vehicles are assumed to emit only as much as their respective type approval limit value or as little as petrol cars of the same age. Excess NO_2 concentrations can also have direct health impacts, but these overlap with the impacts from particulate matter (PM) and are not included here. We estimate that almost 10 000 premature deaths from $PM_{2.5}$ and ozone in the adult population (age >30 years) can be attributed to the NO_x emissions from diesel cars and light commercial vehicles in EU28 plus Norway and Switzerland in 2013. About 50% of these could have been avoided if diesel limits had been achieved also in on-road driving; and had diesel cars emitted as little NO_x as petrol cars, 80% of these premature deaths could have been avoided. Ecosystem eutrophication impacts (critical load exceedances) from the same diesel vehicles would also have been reduced at similar rates as for the health effects.

1. Introduction

Since the late 1990s the sales share of LDDVs (light duty diesel vehicles) has risen sharply at the expense of petrol in most European countries (see the online supplementary data available at stacks.iop.org/ERL/12/094017/mmedia). This development has been intentional, as diesel engines emit less CO_2 than comparable petrol engines. However, they emit significantly more NO_x ($NO + NO_2$). Road transport contributes about 40% of the land based NO_x emissions in the EU28+ countries (EU28 plus Norway and Switzerland), and is one of the main reasons why several countries, including Germany, France, Austria, Belgium, and Ireland, consistently exceed their internationally agreed NO_x emission caps (Wankmüller *et al* 2015). Exceedances of Europe's ambient NO_2 air quality limit values have to a large part been attributed to the emissions from diesel cars (Kiesewetter *et al* 2014, Carslaw *et al* 2016, Degraeuwe *et al* 2016).

Euro 3 and newer LDDVs should emit less NO_x than the older vehicles, both petrol and diesel, that they are replacing. However, unlike petrol cars, on the road these cars do not comply with Euro limit values. (Chen and Borken-Kleefeld 2014, Fontaras *et al* 2014, Hagman *et al* 2015, Yang *et al* 2015). In fact, measurements indicate that Euro 3, 4 and 5 LDDVs may have higher emissions under actual driving conditions than older diesel vehicles (Chen and Borken-Kleefeld 2014). Furthermore Euro 5 LDDVs may have higher emissions than previous Euro class vehicles under actual driving conditions (Ligterink et al 2013).

The difference between expected and actual emissions of NO_x has gained a lot of attention following the discovery of Volkswagen's (VW) diesel 'defeat devices' (US-EPA 2015, BMVI 2016, UK Department for Transport 2016), where software is installed in the vehicles which reduces NO_x emissions to comply with emissions tests in the US and Europe. In Europe about 8.5 million VW diesel cars have been sold with defeat devices installed, a far greater number than in the US where the defeat device was first discovered. As a result, the effects on health in Europe should be much larger than that calculated for the US (Hou *et al* 2016). Chossière *et al* (2017) have calculated the effects of excess NO_x emissions from VW-only diesel passenger vehicles in Germany, and found that these vehicles have resulted in about 1200 premature deaths integrated over the sales period 2008–2015 in Germany and neighbouring countries. However, large discrepancies between test cycle NO_x emissions and on-road emissions have been found across all manufacturers (UK Department for Transport 2016, Transport and Environment 2016). Therefore, in this study, we analyse the consequences of excess NO_x emissions from the whole fleet of LDDVs (and not just a single brand) across all of Europe.

Here we calculate concentrations and depositions of air pollutants in Europe with emissions and meteorology from 2013 focusing on the effects of excess NO_x emissions from LDDVs. NO_x emissions from traffic assumed to reflect real driving conditions are used as a reference case. Additional model calculations are made assuming emissions from LDDVs are compliant with the EU and national regulations and assuming all LDDVs emit as petrol vehicles. Finally we also include a model calculation without LDDVs. Based on these scenarios, the effects on the concentrations of the atmospheric pollutants NO_2, $PM_{2.5}$ and ozone, as well as the deposition of nitrogen, are estimated.

The relationship between NO_2 and health is scientifically not as well founded as for $PM_{2.5}$ and ozone (WMO 2013). In this paper we therefore assess changes in $PM_{2.5}$ and ozone as the most important indicators of human health impacts. Furthermore the effects of increased nitrogen deposition are quantified by the computation of exceedances of critical loads for eutrophication of semi-natural soils.

2. Emissions from diesel cars— Reference vs. WhatIf scenarios

Emissions as officially reported by European countries under the Convention on Long-range Transport of Air Pollutants for the year 2013 are used as input for the calculations (Wankmüller *et al* 2015) (see also WebDab (www.ceip.at/ms/ceip_home1/ceip_home/webdab_emepdatabase/). As the discrepancy between Euro standards and real-world emissions for LDDVs has been known for some time already, the emission factors used in European emission inventories are not based upon Euro standards, but rather already reflect the high NO_x emissions expected in real-world conditions (Ntziachristos and Samaras 2009, HBEFA 2010, Katsis *et al* 2012). According to ERMES (European Research Group on Mobile Emission Sources, www.hbefa.net/d/pdf/ERMES_NOX_EF_V20151009.pdf), the emissions from passenger cars estimated with such tools should be at least a factor of four larger than emissions would have been if the vehicles had been complying with the Euro emissions standards.

Here we conduct four runs:

- 'Reference case': Here the officially reported emissions are used (including real driving emissions from LDDVs) in the calculation of the health and environmental impacts of the current situation. It is thus the backdrop against which the other scenarios are to be evaluated.
  Reported emissions are however only given for all LDDVs together. They are neither disaggregated by fuel type nor by emission control stage (i.e. Euro class) as needed for the scenario calculations. Therefore we employ the GAINS model that is calibrated to the reported emissions and represents the necessary details for all countries (Amann *et al* 2015), notably how much NO_x is emitted by diesel cars of a certain emission control stage[6].

For the scenario calculations we leave emissions from all other sectors unchanged and only adjust NO_x emissions from LDDVs as follows:

- Scenario 'WhatIf-Dlim': This scenario assumes that average on-road NO_x emissions of LDDVs are at the level of the respective diesel limit values. This could represent emissions as intended by the legislation under 'normal conditions of use'. We apply the respective emission rates to the whole fleet of passenger cars and light duty diesel vehicles, as measurements show that high on-road NO_x emissions are not limited to a certain manufacturer (BMVI 2016, UK Department for Transport 2016).
- Scenario 'WhatIf-Petrol': This scenario assumes that average on-road NO_x emissions of diesel cars are as low as the respective on-road emissions of petrol cars of the same age and emission standard. Petrol powered cars comply with stricter NO_x emission limits in the European Union than diesel cars. Here we assume that no discrimination would be made by propulsion type as is the case in the US legislation.

[6] GAINS is set-up in five year intervals. Therefore emission shares used here are actually interpolated between 2010 and 2015.

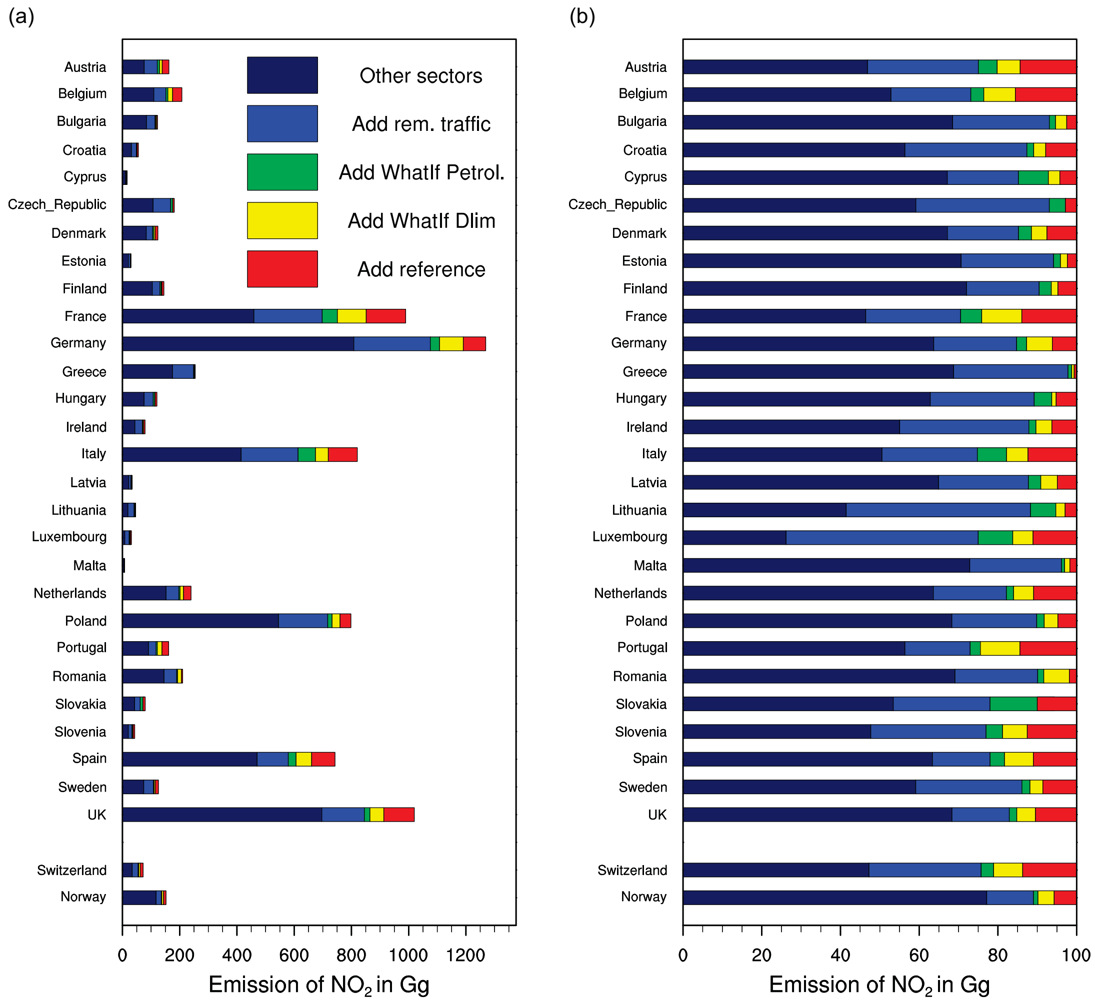

Figure 1. NO_x emissions from the individual EU28+ countries in Gg NO_2 (left), and in percent (right). 'Other sources' are the emissions from all other sources, including off-road mobile machinery, and 'add rem. traffic' is emissions from road traffic other than LDDVs. Emissions from LDDVs are shown as the cumulative sums of 'Add WhatIf-Petrol', 'Add WhatIf-Dlim' and 'Add Reference'. See text for definitions and further explanation.

- Scenario 'WhatIf-NoLDDV': Lastly, we set NO_x emissions from LDDVs to zero. This model run is used to calculate the impacts of NO_x emissions from all other sources except LDDVs. The difference between the Reference case and NoLDDVs can then be interpreted as the health and environmental burden from these vehicles, and as such represents a scale against which the effects of WhatIf-Petrol and WhatIf-Dlim can be measured.

Figure 1 illustrates the distributions between non traffic emissions, and traffic emissions split between reference driving emissions and the scenario emissions of NO_x for the EU28+ countries, both in Gg and in percent. The contributions from the different sources/scenarios are stacked so that the total emissions from LDDVs from the individual countries are shown as the sums of the 'WhatIf Petrol', 'WhatIf Dlim' and 'Reference' emissions. Thus, emissions exceeding 'WhatIf Dlim' are shown in red labelled 'Add reference' in the figure. Further reductions that could have been achieved with 'WhatIf Petrol' are shown in yellow labelled 'Add WhatIf-Dlim'. In the EU28+ countries we find that excess NO_x emissions from LDDVs account for up to 15% of national emissions. Remaining traffic emissions are essentially trucks and petrol cars.

We calculate the 'WhatIf' NO_x emissions accounting for different shares, ages and annual mileages of diesel cars in each European country. These 'WhatIf' emissions replace the LDDV emissions in the Reference case (data documented in table 1). Their spatial and temporal distribution is equal to the Reference case, even though in reality this percentage is likely to differ from one region to another within the countries. Emissions from all other sources are left unchanged, i.e. as officially reported by the countries. Concentrations and impacts are calculated through for these counterfactual scenarios and results compared to the reference case with actual driving emissions. (It can be noted that the GAINS and EMEP LDDV emissions differ slightly, but the discrepancy of about 4% has little impact on our calculations.)

Table 1. Average on-road NO_x emission factors in g NO_x MJ^{-1} (where 1 MJ = 10^6 J (Joule)) fuel consumed and NO_x emissions in Gg in the EU28 countries for the Reference case (i.e. with current high NO_x emissions) and for the WhatIf-Dlim and WhatIf-Petrol scenarios. For the WhatIf scenarios the percentage reductions in LDDV emissions compared to the Reference are also listed.

	Reference		WhatIf Dlim			WhatIf petrol		
NO_x emissions passenger cars	Emission factors	Gg	Emission factors	Gg	in %	Emission factors	Gg	in %
Pre Euro	0.252	11	0.252	11	0%	0.792	36	214%
Euro 1	0.291	34	0.291	34	0%	0.170	20	−41%
Euro 2	0.298	96	0.226	73	−24%	0.095	31	−68%
Euro 3	0.337	249	0.198	146	−41%	0.039	29	−88%
Euro 4	0.261	390	0.109	163	−58%	0.023	34	−91%
Euro 5	0.316	392	0.083	87	−78%	0.020	24	−94%
Subtotal	0.296	1173	0.130	515	−56%	0.044	174	−85%
NO_x emissions light commercial vehicles	Emission factors	Gg	Emission factors	Gg	in %	Emission factors	Gg	in %
Pre Euro	0.443	30	0.443	30	0%	0.774	53	75%
Euro 1	0.373	22	0.277	17	−26%	0.118	7	−69%
Euro 2	0.371	51	0.296	41	−20%	0.053	7	−86%
Euro 3	0.309	93	0.190	57	−39%	0.032	10	−90%
Euro 4	0.233	110	0.095	45	−59%	0.018	9	−92%
Euro 5	0.241	60	0.073	18	−70%	0.015	4	−94%
Subtotal	0.285	366	0.162	208	−43%	0.069	89	−76%
Total LDDVs		1539		723	−53%		263	−83%
Petrol cars	0.070	237	0.070	237	0%	0.070	237	0%
Other PV and LCV	0.118	50	0.118	50	0%	0.118	50	0%

3. Model calculations

In this study we use the EMEP/MSC-W chemical transport model, version rv4.8. This model, available as open source (www.emep.int), has been described in detail in Simpson *et al* (2012), with various updates, see (Simpson *et al* 2016) and references within.

For Europe the model is regularly evaluated against measurements in the EMEP annual reports, available at www.emep.int. In addition the EMEP model has been included in model intercomparisons and model validations in a number of peer reviewed publications (Jonson *et al* 2006, Jonson *et al* 2010, Simpson *et al* 2006, Simpson *et al* 2006, Colette *et al* 2011, Colette *et al* 2012, Angelbratt *et al* 2011, Dore *et al* 2015, Stjern *et al* 2016).

The model is run with a 0.1 × 0.1 degrees resolution driven by ECMWF-IFS meteorology and emissions as described in section 2. A comprehensive description, including model evaluations, of the model results with the 0.1 × 0.1 degrees application of the EMEP model for 2013 can be found in Tsyro *et al* (2015). All model runs have been made for a single year, 2013. More details about the model, including specifications of the model validation, is included in the supplementary data.

3.1. Health impacts from $PM_{2.5}$

Following the methodology recommended for European health impact assessments by the HRAPIE (Health risks of air pollution in Europe) project (HRAPIE 2013) of the World Health Organization, the assessment of $PM_{2.5}$ health effects is based on estimates of the impact of long-term (annual average) exposure to $PM_{2.5}$ on all-cause (natural) mortality in adult populations (age >30 years). For the $PM_{2.5}$ concentrations prevailing in Europe, a linear concentration–response function is employed, with a relative risk coefficient of 1.062 (95% confidence interval (CI) 1.040−1.083) per $10\mu g\,m^{-3}$. This relationship is applied to $PM_{2.5}$ concentrations arising from anthropogenic emission sources, while impacts from natural sources of $PM_{2.5}$ remain unquantified.

In calculating the population exposure to $PM_{2.5}$ it is recognized that the resolution of the atmospheric dispersion calculation is insufficient to reproduce accurately measured urban background $PM_{2.5}$ concentrations. To account for this, an urban increment related to primary PM emissions from local low-level sources is estimated by applying a downscaling scheme based on a redistribution of primary PM concentrations according to the primary PM emission densities from the domestic and transport sectors. A detailed description of the methodology is given by Kiesewetter *et al* (2015). The annual mean population-weighted $PM_{2.5}$ concentrations are used to calculate premature mortality attributable to $PM_{2.5}$ in country i as:

$$\mathrm{mort}_i = \mathrm{PM2.5}_i \times \mathrm{RR}_{\mathrm{PM}} \times \mathrm{deaths30}_i$$

where:

mort_i cases of premature mortality per year in country i;

$\mathrm{deaths30}_i$ baseline mortality (number of natural adult deaths per year) in country i;

RRPM relative risk per $\mu g\,m^{-3}$ annual mean $PM_{2.5}$;

$PM_{2.5i}$ annual mean population-weighted $PM_{2.5}$ in country i.

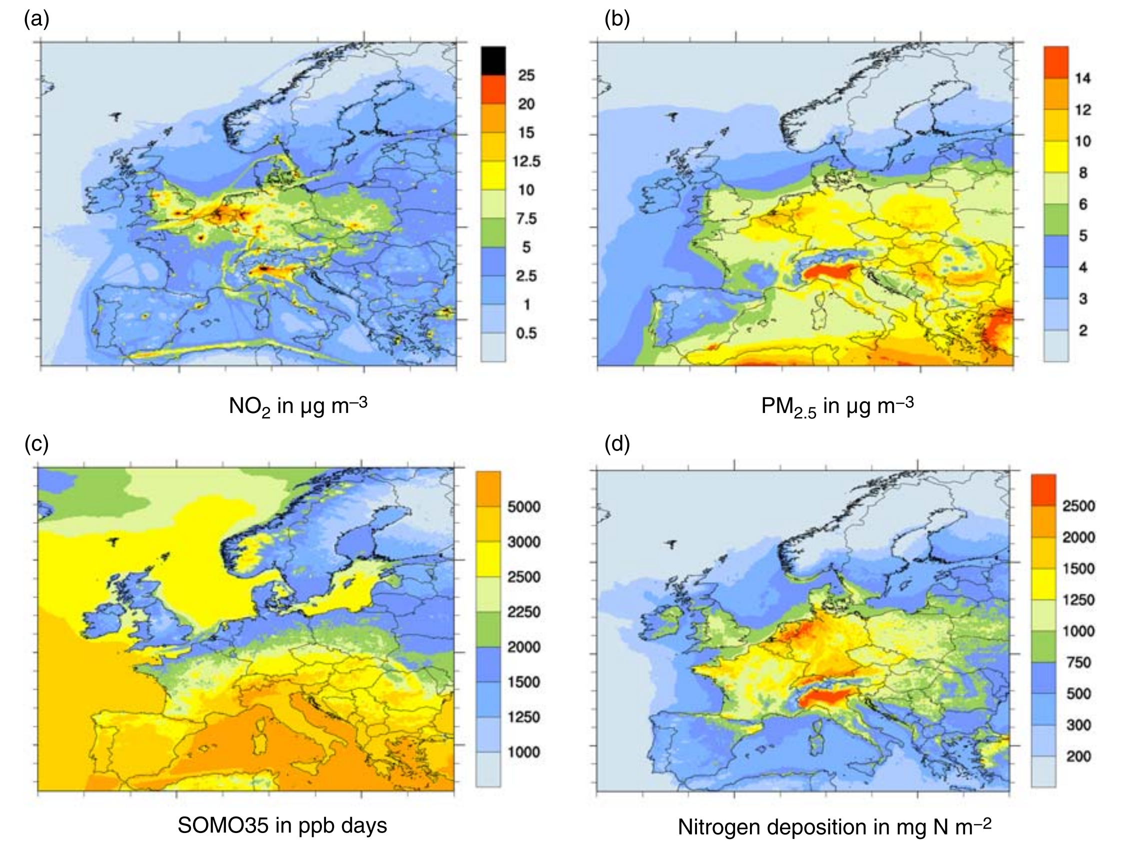

Figure 2. Concentrations of NO_2 and $PM_{2.5}$, SOMO35 and deposition of nitrogen calculated for 2013 (assuming Reference case emissions of NO_x).

3.2. Health impacts from ozone

Following the methodology recommended for European health impact assessments by the HRAPIE project (HRAPIE 2013), the assessment of effects due to ozone is based on estimates of the impact of short-term (daily maximum eight-hour mean) exposure to ozone on all-cause mortality for all ages. This is calculated by applying a linear function with a relative risk coefficient of 1.0029 (95% CI 1.0014–1.0043) per 10 $\mu g\,m^{-3}$. Using the modelled estimates of the SOMO35 indicator, which quantifies the yearly sum of the daily maximum eight-hour ozone concentrations exceeding a 35 ppb threshold, the annual cases of premature mortality in country i attributable to ozone are then calculated as:

$$\mathrm{mort}_i = (\mathrm{SOMO35}_i/365) \times \mathrm{RR}_{O3} \times \mathrm{deaths}_i$$

where mort_i and $\mathrm{deaths30}_i$ are as given above, and:

RR_{O3} relative risk per ppb 8 h maximum ozone concentration per day;

$\mathrm{SOMO35}_i$ population-weighted SOMO35 in country i.

3.3. Exceedances of critical loads for eutrophication

A critical load (CL) is defined as 'a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson and Grennfelt 1988). CLs are calculated for different receptors (e.g. terrestrial ecosystems, aquatic ecosystems), and a sensitive element can be any part (or the whole) of an ecosystem or ecosystem process. CLs have been derived for several pollutants. Here we restrict ourselves to CLs defined to avoid the eutrophying effects of N deposition (critical load of eutrophying N, $CL_{eut}N$). Nitrogen also influences the acidity status of soils, but CLs of acidity are no longer exceeded much in Europe (Slootweg *et al* 2015).

The $CL_{eut}N$ for a site is either derived empirically or calculated from a simple steady-state mass balance equation linking a chemical criterion (e.g. an acceptable N concentration in soil solution that should not be exceeded) with the corresponding deposition value. Methods to compute CLs are summarised in the so-called Mapping Manual UNECE (2004), see also De Vries *et al* (2015), which is used within the Convention on Long-range Transboundary Air Pollution (www.unece.org/env/lrtap).

If deposition is higher than the CL at a site, the CL is said to be exceeded. The single exceedance number computed for a grid cell (or any other region) is the so-called average accumulated exceedance (AAE), defined as the weighted mean of the exceedances of all ecosystems within the grid cell, with the weights being the respective ecosystem areas (Posch *et al* 2001).

Exceedances of $CL_{eut}N$ are calculated using the current CL database held at the Coordination Centre for

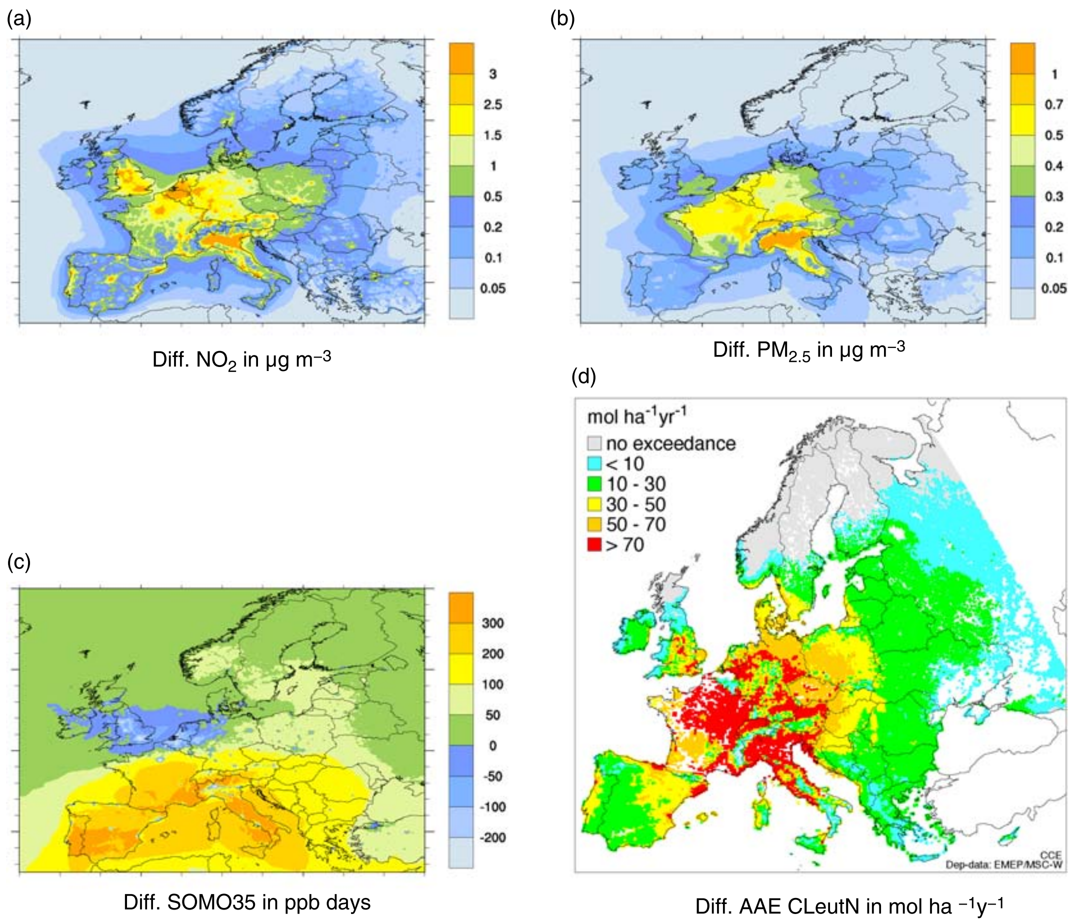

Figure 3. Differences in concentrations of NO_2(*a*) and $PM_{2.5}$ (*b*), SOMO35 (*c*) and exceedance (AAE) of the critical loads of eutrophication (CLeutN) (*d*) calculated with Reference—WhatIf NoLDDV NO_x emissions.

Effects (Slootweg *et al* 2015) and used in supporting European assessments and negotiations on emission reductions (Hettelingh *et al* 2001, Reis *et al* 2012, EEA 2014)

4. Results

Here we briefly describe the Reference situation, i.e. pollution and their impacts from our best estimate of emissions for the year 2013. Then we identify what part of this pollution can be attributed to emissions from light duty diesel vehicles as the difference between the Reference and WhatIf scenarios. Finally, we estimate the additional effects on human health and the environment from LDDV NO_x emissions.

4.1. LDDVs: effects on ambient concentrations

Figure 2 shows the year 2013 model calculated levels of NO_2, $PM_{2.5}$, SOMO35 and depositions of nitrogen for the Reference case.

The spatial pattern of the NO_2 concentrations (figure 2(*a*)) is very similar to the NO_x emissions as the residence time of NO_2 is short (often just hours in daytime). Concentration hot-spots are seen in the countries close to the English Channel and in the Po Valley in northern Italy.

$PM_{2.5}$ concentrations (figure 2(*b*)) are in particular high in the Po Valley. Ozone concentrations, and subsequently SOMO35, increase from north to south as temperature and sunshine levels increase towards the Mediterranean (figure 2(*c*)). High nitrogen depositions are seen in the Po Valley and in and around the English Channel (figure 2(*d*)). A detailed description of these concentrations, and depositions of nitrogen, is provided in the supplementary data.

The geographic pattern is quite different for the pollutants in figure 2. Therefore we can also expect different outcomes for each of the impact indicators calculated here. Figure 3 presents the differences in air pollutant concentrations and eutrophication that we can allocate to NO_x emissions from LDDVs. More material illustrating the effects of LDDVs for the individual EU28+ countries (also in percent) for the three WhatIf scenarios is included in the supplementary material.

In several regions contributions corresponding to 30% and more of NO_2 can be attributed to the LDDV NO_x emissions, particularly in northern Italy, France, and Belgium (figure 3(*a*)).

Major contributions to $PM_{2.5}$ levels from LDDV NO_x emissions are calculated for parts of the Alpine region and the whole of France, with the Po Valley

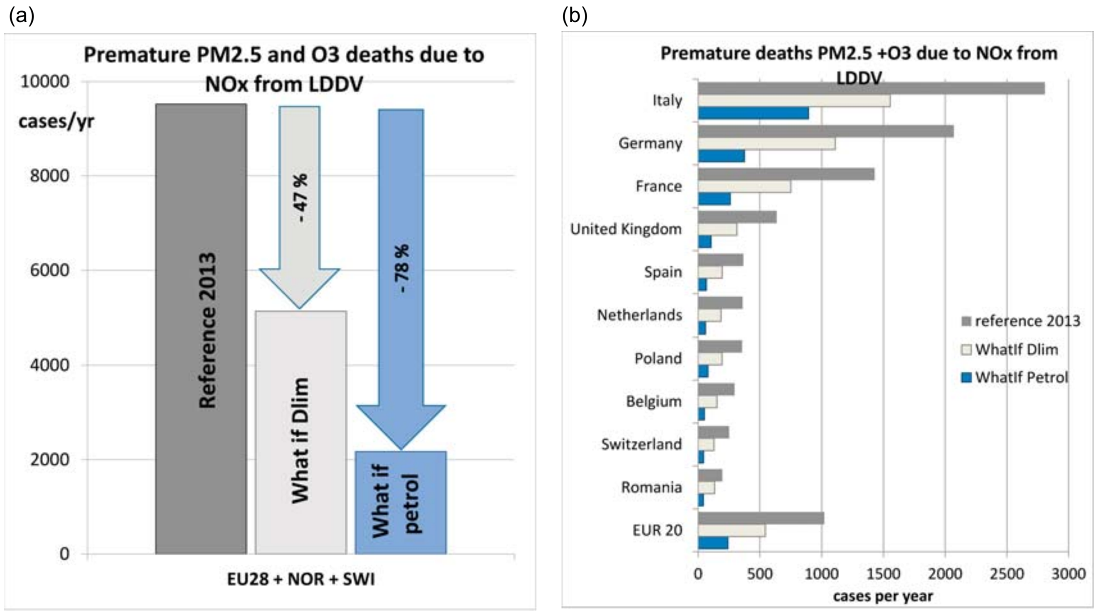

Figure 4. Premature deaths as caused by LDDVs in EU28+ as a whole, left, and for the 10 countries with the highest numbers of premature deaths in EU28 plus Switzerland, right. EUR 20 is the sum for the remaining 19 EU countries plus Norway.

Table 2. Average pollutant impacts for the EU28+ region from LDDVs in the Reference case and percentage reductions for the counter-factual scenarios WhatIf-Dlim and WhatIf-Petrol.

	Attributed to LDDV	What If Dlim	What If Petrol
Concentration of NO_2	$0.60\ \mu g\ m^{-3}$	−52%	−81%
Concentration of $PM_{2.5}$	$0.19\ \mu g\ m^{-3}$	−50%	−79%
Premature deaths ($PM_{2.5}$)	9390	−47%	−78%
SOMO35	96 ppb days	−45%	−76%
Premature deaths (ozone)	392	−27%	−64%
Deposition of N	42.1 mg(N) m^{-2}	−51%	−80%
Exceedance of $CL_{eut}H$	27.8 mol $ha^{-1}\ yr^{-1}$	−53%	−81%
Exceedance of ecosystem area ($CL_{eut}N$)	2.16%	−47%	−79%

being the most affected by contributions corresponding to 10% or more (figure 3(*b*)).

Given the nonlinear behaviour in the response of ozone to changes in NO_x emissions, the effects of the WhatIf scenarios varies across Europe. The contributions to SOMO35 are shown in figure 3(*c*). In high NO_x emitting areas, such as the countries bordering the southern parts of the North Sea, parts of the Po Valley and for several large cities, the SOMO35 levels are reduced due to titration effects following additional NO_x emissions. Ozone levels are increased by up to 10% in large parts of France, Spain and Portugal, and somewhat less in the rest of southern and central Europe. In particular in southern Europe reductions in SOMO35 could have been achieved with lower NO_x emissions.

Before being deposited, nitrogen species can be transported over relatively large distances (Hertel *et al* 2012). As shown in the supplementary material the additional NO_x emissions from LDDVs result in increased nitrogen depositions all over Europe, with notable contributions of up to 10% again in the Po Valley, but also large areas from Scandinavia south to the Mediterranean.

The effects of these scenarios on EU28+ as a whole are given in table 2, suggesting that concentrations and depositions would be reduced by about 50% if emission factors from LDDVs complied with diesel limit values (WhatIf-Dlim). If emission factors could be made comparable to petrol vehicles (WhatIf-Petrol), this would reduce concentrations and depositions from LDDVs by about 80%.

4.2. Health effects

More than 400 000 and 15 500 premature deaths have been associated with the current $PM_{2.5}$ and ozone concentrations respectively in EU28+ (EEA 2016). We estimate that about 3.5% of $PM_{2.5}$ and 2.3% of ozone premature deaths can be attributed to the NO_x emissions from diesel LDDVs (difference reference to no LDDV emissions). We calculate that about 9390 cases of premature deaths can be attributed to $PM_{2.5}$ from LDDVs in EU28+[7], and 392 to ozone (see table 2). As seen in figure 4(*a*) almost 50% (about 4500) of

[7] Our own bottom-up impact assessment estimates only some 260 000 premature deaths attributable to $PM_{2.5}$ in EU28. The lower value compared to the EEA assessment is largely due to lower average $PM_{2.5}$ concentrations that we calculated. Based on the EEA figures, 3.5% of $PM_{2.5}$ related premature deaths would imply about 13300 deaths. For ozone we calculate about 16 250 premature deaths, close to the EEA result.

the premature deaths could have been avoided if NO_x emissions from LDDVs on the road would have been as in the test laboratory, i.e. no more than the emission limit value. Had diesel cars emitted as little NO_x as petrol cars, almost four out of five of these premature deaths could have been avoided. Almost all countries would have benefitted in roughly equal proportions, see figure 4(*b*). In a recent study Anenberg *et al* (2017) attribute almost 6900 premature deaths to excess NO_x emissions from LDDVs in EU28+ in the year 2015. Given the different modelling set-up and uncertainties in fleet emissions, atmospheric dispersion and health impacts, this independent estimate is in good agreement. Furthermore, they attribute about an additional 4650 premature deaths from heavy duty trucks and buses. According to our calculations, the countries with the highest number of premature deaths attributable to $PM_{2.5}$ induced from LDDVs are Italy, Germany and France (figure 4(*b*)) resulting from their large populations and high share of diesel cars in their national fleets. These three countries comprise two thirds of premature deaths from excess NO_x emissions of LDDV in EU28+ in the year 2013. The highest number of premature deaths per inhabitant attributable to LDDV emissions occurs in Italy, Switzerland and Belgium. With 2.85 to 4.4 cases per 100 000 inhabitants the risk in these countries is 40% to 140% higher than the EU28+ average (1.8 cases per 100 000 inhabitants). The lowest risk for premature death because of LDDV related NO_x emissions is in Norway, Finland and Cyprus where the risk is at least 14 times lower than the EU28+ average.

There are some 392 cases of premature deaths attributable to ozone exposure in EU28+ as a consequence of excess LDDV NO_x emissions. This value is more than 30 times lower than the $PM_{2.5}$ attributed premature deaths. A compliance with diesel emission limits or even petrol emission limits would have resulted in one quarter and almost two thirds fewer premature deaths in EU28+. This is lower than the corresponding change in NO_x emissions and also lower than the induced $PM_{2.5}$ concentrations because ground level ozone formation scales in a nonlinear way with the NO_x emissions. In some countries (Belgium, Netherlands, United Kingdom) the extra NO_x emissions from LDDVs have actually led to less ozone being formed. Thus, for these countries a reduction in NO_x emissions from diesel cars, as simulated, would result in 70 to 90 more cases of ozone related premature deaths.

4.3. Eutrophication effects

Reference model calculations of exceedance (AAE) of the critical loads of eutrophication in Europe are shown in the supplementary material. The critical loads are exceeded in about 60% of the European ecosystem area, but with particularly large exceedances in the Po Valley in Italy and along the Dutch–German border. In addition to NO_x, a large portion of the nitrogen deposition is from ammonia emissions. As a result, the contribution from LDDVs to the levels of exceedances are moderate, increasing the area exceeded by about 2% in the EU28+, see table 2. However, there are marked contributions from LDDVs in the Po Valley and in western parts of France. In table 2 we show that had the LDDVs emitted NO_x according to the diesel limit about half of the exceedance could have been avoided, and if they had emitted as petrol vehicles most of the additional exceedances could have been avoided.

5. Uncertainties

Model calculations are of course uncertain, but the EMEP model has been extensively evaluated for many pollutants in many different climates (see references in section 3), and is among the best models operating in Europe today, e.g. Dore *et al* (2015). With a 0.1×0.1 degrees resolution we are not able to fully resolve pollution levels at locations close to major sources of traffic (Schaap *et al* 2015). Dispersion of primary compounds such as NO_x or primary $PM_{2.5}$ could in principle be modelled with urban-scale models. However, such models are usually not capable of modelling the medium to long range transport and chemical formation of oxidants and aerosols. In any case, fine-scale meteorology and emissions are not generally available for European-scale assessments such as ours.

Methods extrapolating regional model results, as the CITY-DELTA methods (http://ies-webarchive.jrc.ec.europa.eu/citydelta/) or the Air Quality Regridder Model (Theobald *et al* 2016), could be an alternative, but such methods require an individual treatment of each city, beyond the scope of this study.

Further, model uncertainties tend to be systematic rather than random, with small perturbations in for instance emission inputs, leading to well-behaved changes in output concentrations. In the scenario calculations several additional assumptions are made that may affect the uncertainty. As already discussed, the emissions from diesel vehicles differ substantially depending on make and model (Transport and Environment 2016). The European emission reporting guidelines are the same for all countries, but the vehicle model mix is not. Therefore the exact emissions in the individual countries will depend on the local fleet mix. Subsequently the calculated effects are likely to be a lower estimate in countries where the most polluting brands and makes are popular. In addition NO_x emissions as officially reported (i.e. as used in our Reference calculation) are likely to be revised upwards in the light of recent findings (Keller *et al* 2017). Thus excess NO_x emissions may be higher than reported here. As a result we believe that the calculated impacts are likely to be a lower estimate.

As noted above, we are not able to resolve locations close to heavy traffic. However, the formation of secondary PM and ozone happens on spatial scales of several tens of kilometers, and concentrations are relatively uniform across larger regions. Therefore a finer spatial resolution would likely not change our result much.

The link between air pollution and health is well established through projects such as the HRAPIE project (HRAPIE 2013), but additional processing and use of the model results will inevitably increase the uncertainty to some extent.

In this paper health effects from LDDVs are only considered for ozone and secondary particles from NO_x. The effects of primary emitted particles are not included, as these emissions are within the emission limits. Health effects from increased NO_2 concentrations are not included for lack of agreed risk factors and uncertain overlap with PM health impacts.

Furthermore we assume that all populations have the same health response to air pollution and that there exists no interaction between ozone, $PM_{2.5}$ and other air pollutants. The paper does not account for other health effects beyond adult mortality.

6. Conclusions

The calculations in this paper demonstrate how excess NO_x emissions from LDDVs lead to increased levels of NO_2, $PM_{2.5}$, ozone and depositions of eutrophying nitrogen in Europe. This increase in pollution levels affects human health and the environment. The effects as summed up for the EU28 countries as a whole are as follows:

- Up to 10 000 premature deaths due to $PM_{2.5}$ and ozone formation can be attributed to high NO_x emissions from LDDVs in Europe in the year 2013.
- About half of these cases could have been avoided if these vehicles would in real driving emit no more than the EU limit value (as possibly will be achieved with forthcoming legislation).
- About 80% of the impacts could have been avoided if these vehicles would emit no more than petrol vehicles (as achieved in the US with technology neutral emission standards).
- Excess premature deaths will continue into the future until LDDVs with high on-road NO_x emissions have been replaced, possibly from 2021 and onwards when the final stage of the Euro 6 legislation is intended to close the gap between the test cycle and on-road emissions.
- NO_x emissions from LDDVs contribute to the exceedance of the critical loads of eutrophication in Europe. About half of this additional exceedance could have been avoided if LDDVs would emit no more than the EU limit value, and about 80% if they would emit no more than petrol cars.

There are signs that diesel penetration in Europe is now going down, partly as a result of the 'Dieselgate' scandal. Dieselgate may also have the effect of expediting the transition to electric vehicles and other alternative fuel modes of transport. Several economic studies, such as Bloomberg (2016) and the Fitch report (www.fitchratings.com/site/pr/1013282), predict a strong growth in the electrification of the car industry in the next decade that will inevitably reduce all types of direct traffic emissions.

It is interesting to note that even though the calculation chain, from emissions to pollution levels to effects on human health and the environment, involve several potentially nonlinear steps, the calculations are close to linear. The 50%–80% changes in the emissions of NO_x from LDDVs studied here result in similar percentage changes in concentrations of NO_2 and $PM_{2.5}$, and depositions of nitrogen, and furthermore in the number of premature deaths and in the exceedances of critical loads of eutrophication (both in amount and ecosystem area). The exception is ozone, where the percentage changes in ozone (as SOMO35) and in premature deaths are smaller than the changes in NO_x emissions as result of NO_x titration in some highly polluted regions.

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ORCID iDS

C Heyes https://orcid.org/0000-0001-5254-493X

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