Public Health Impacts of Excess NO_x Emissions from Volkswagen Diesel Passenger Vehicles in Germany

by

Guillaume P. Chossière

Submitted to the Department of Aeronautics and Astronautics and to the Institute for Data, Systems, and Society in partial fulfillment of the requirements for the degrees of

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Abstract

In September 2015, the Volkswagen Group (VW) admitted the use of "defeat devices" designed to lower emissions measured during VW vehicle testing for regulatory purposes. Globally, 11 million cars sold between 2008 and 2015 are affected, including about 2.6 million in Germany. On-road emissions tests have yielded mean on-road NO_x emissions for these cars of 0.85 $g.km^{-1}$, over four times the applicable European limit of 0.18 $g.km^{-1}$. This thesis estimates the human health impacts and costs associated with excess emissions from VW cars driven in Germany. A distribution of on-road emissions factors is derived from existing measurements and combined with sales data and a vehicle fleet model to estimate total excess NO_x emissions. These emissions are distributed on a 25 by 28 km grid covering Europe, using the German Environmental Protection Agency's (UBA) estimate of the spatial distribution of NO_x emissions from passenger cars in Germany. I use the GEOS-Chem chemistry-transport model to predict the corresponding increase in population exposure to fine particulate matter and ozone in the European Union, Switzerland, and Norway, and a set of concentration-response functions to estimate mortality outcomes in terms of early deaths and of life-years lost. Integrated over the sales period (2008 - 2015), I estimate median premature mortality impacts from VW excess emissions in Germany to be 1,200 premature deaths in Europe, corresponding to 13,000 life-years lost and 1.9 billion EUR in costs associated with life-years lost. Approximately 60 % of mortality costs occur outside Germany. For the current fleet, I estimate that if on-road emissions for all affected VW vehicles in Germany are reduced to the applicable European emission standard by the end of 2017, this would avert 29,000 life-years lost and 4.1 billion 2015 EUR in health costs (median estimates) relative to a counterfactual case with no recall.

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Contents

List of Figures

List of Tables

Chapter 1

Introduction

Public health is significantly and globally affected by outdoor air pollution (WHO 2006 [72]), with the European Environment Agency (EEA) estimating that outdoor air pollution is responsible for more than 400,000 premature deaths annually in Europe [5]. These impacts are driven primarily by population exposure to fine particulate matter with an aerodynamic diameter of 2.5 µm or less ($PM_{2.5}$) and, to a lesser extent, ozone, both of which have been associated with an increased risk of premature mortality by epidemiological studies (see the review study by Hoek et al [41] and the multi-city analysis in the WHO 2013 report [64]). Following the OECD, road transportation emissions account for approximately 50% of the total health impacts from ambient air pollution in Europe [58].

Vehicle emissions standards initially introduced in 1991 (the Euro standards) aimed to reduce the air quality impacts of road transportation [24]. However, observed reductions in ambient pollution have not been as significant as expected, and the EEA noted that "emissions in real-life driving conditions are often higher, especially for diesel vehicles, than those measured during the approval test" [5]. A partial explanation for this discrepancy came to light in September 2015 when an investigation by Thompson et al [66] quantified excess emissions from Volkswagen group vehicles, ultimately leading to allegations by the US Environmental Protection Agency (EPA) and admission by VW that certain vehicle models had engines equipped with software designed to reduce emissions during approval testing significantly below levels achieved under actual on-road operation [31].

Consequently, in October 2015 the German Federal Motor Transport Authority (KBA) ordered a mandatory recall of all 2.4 million affected passenger vehicles still in service and stated that they must be modified to comply with European regulations [46]. Following the KBA order, VW announced that the recall would begin in 2016 with completion originally scheduled for the end of the same year [37]. The first vehicles were recalled at the end of January 2016 [38]. For all three engine sizes affected (1.2 liter, 1.6 liter and 2.0 liter), Volkswagen will update the EA 189 engine software to ensure that on-road emissions stay below the permitted limit. In addition, for the 1.6 liter engine, VW will fit a flow rectifier to the intake duct.

In this study, I estimate excess NO_x emissions from affected cars commercialized by the brands of the Volkswagen Group that used the EA 189 1.2, 1.6 and 2.0 liter engines (Audi, Seat, Skoda, Volkswagen) within Germany, calculating health impacts and health costs throughout Europe which have already occurred between 2008 and 2015, in addition to future outcomes under different recall scenarios. I seek to estimate the number of early deaths as well as the total number of life-years lost which are at risk of resulting from excess emissions. Morbidity impacts are not evaluated or estimated.

Chapter 2

Methods

Excess emissions in this study are defined as the difference between vehicle on-road NO_x emissions and the limit value set by the applicable European emissions standard, Euro 5. Even though European Union regulations only require vehicles to meet this standard during the type approval test conducted in the laboratory using the New European Driving Cycle, the intent of the regulation is to limit on-road emissions and achieve real-world air quality improvements (European Commission [25]). In the absence of an official factor to account for the difference between laboratory and on-road testing, I use the regulatory limit as the point of reference for our calculations of excess emissions.

For effects occurring in the future, I first estimate impacts assuming all affected VW vehicles in Germany are modified to bring on-road emissions down to Euro 5 standards by the end of 2016, with a fleet recall occurring at a fixed rate over the course of 2016. I then calculate effects for the counterfactual case that affected vehicles remain in the fleet until their retirement without on-road emissions being reduced to Euro 5 levels. The difference between the counterfactual and recall cases yields the benefits of the recall in terms of avoided excess emissions, health impacts and health costs. The study also quantifies the impact for a case where the recall is performed at a slower rate and completed by the end of 2017 with the same start date. It should be noted here that future health impacts under one of the "recall" scenarios correspond to a reduction of on-road emissions to the Euro 5 limit value. While the absolute decrease in NO_x emissions which would result from a recall is difficult to estimate, any recall, if carried out, would reduce the average NO_x emissions rate from the current estimated value of approximately 0.85 $g.km^{-1}$ to some lower value. I take the regulatory limit (0.18 $g.km^{-1}$) as a reasonable minimum value. It is possible to calculate the benefits of smaller reductions as a linear combination of the "no policy" scenario and the "recall" scenario. As detailed later in this study, the atmospheric response to excess emissions is approximately linear for the range of perturbations considered.

This study augments and extends the modeling capabilities used in Barrett et al [11] to estimate the impact of excess VW NO_x emissions in the US. They used an adjoint-sensitivity-based air quality modeling approach to capture the impacts of spatially disaggregate excess NO_x emissions on aggregate $PM_{2.5}$ and ozone population exposure in the US. Due to nonlinearity associated with differences in population density and distribution, background atmospheric composition and local meteorology, linear scaling of the results calculated by Barrett et al for the US will not necessarily yield an accurate or reliable answer for other countries. In this study I therefore use a chemistry-transport model to capture the impacts of spatially disaggregate excess NO_x emissions in Germany on spatially disaggregate $PM_{2.5}$ and ozone population exposure in Germany and other European countries. The calculated increase in population exposure due to these excess emissions is then converted into policy-relevant metrics of health impacts and costs.

The air quality and health impacts of the NO_x excess emissions in Germany are computed using spatially-resolved excess NO_x emissions estimates. Unless otherwise specified, NO_x emissions are reported on an NO_2 mass basis, consistent with the standards used in emissions inventories. Excess emissions are estimated from on-road measurements of the affected models, as well as the number of affected cars and the vehicle kilometers traveled. Their spatial distribution is inferred from the distribution of overall NO_x emissions from passenger cars in Germany, obtained from the German Environmental Protection Agency [68]. A regional chemistry- transport model is used to relate emissions to pollutant concentrations. Concentration-response functions are applied to convert the resultant population exposure to $PM_{2.5}$ and ozone into health impacts. Given the significant uncertainties in estimating health impacts of air pollution and the associated costs, I use a probabilistic framework which propagates uncertainty through all aspects of the calculation. Public health outcomes are monetized using both the Value of Statistical Life (VSL) and Value of Life Year (VOLY) approach. In this section I describe the uncertainty estimation approach, the excess emissions calculation, the population exposure estimation, health impacts estimation and the monetary valuation of the impacts.

2.1 Uncertainty quantification

Uncertainty in input variables such as vehicle activity and emissions factors (mass of NO_x emitted per kilometer driven) is propagated through the analysis by performing a Monte Carlo simulation. For each sample, a random draw is taken for each of the parameters listed in table 2.1, according to the distributions shown. The total excess emissions from German light-duty VW vehicles is calculated for each sample based on the randomly drawn emissions factor and on the vehicle-kilometers traveled (VKT) growth factor. These two variables and their distributions are discussed in section 2.3. Total population exposure to $PM_{2.5}$ and ozone is then calculated by interpolating results from the chemistry-transport model GEOS-Chem, described in section 2.4. This includes compensation for bias in the modeled response gradient for both ozone and $PM_{2.5}$, each drawn as a random variable.

The health impacts of variations in exposure to $PM_{2.5}$ and ozone are calculated by applying concentration response functions (CRFs), as described in section 2.5. For each CRF, the rate of increase in relative risk per unit of exposure is again drawn as a random variable for each sample. The total mortality resulting from the excess emissions for that sample is calculated by summing the premature mortalities calculated for $PM_{2.5}$ and ozone, assuming independent effects. One thousand independent values are drawn from the distribution of each variable to yield one thousand independent estimates of the number of premature mortalities and Years of Life Lost (YLL). The

Table 2.1: Uncertain parameters used in this study when calculating premature mortalities. Uncertainty in monetization is calculated separately.

Parameter	**Central value [95% CI]**	**Distribution type**
Emissions factor (g $NO_x.L^{-1}$)	16 [5.0, 28]	Truncated normal
VKT growth rate (2008-2020)	0.005 [0.003, 0.007]	Triangular
VKT growth rate (2020-2040)	0.003 [0.001, 0.004]	Triangular
$PM_{2.5}$ response bias	+11 % [-15 %, +91 %]	Uniform
Ozone response bias	-0.050 % [-18 %, +35 %]	Uniform
$PM_{2.5}$ CRF (risk per 10 μg.m^{-3})	1.11 [1.05, 1.16]	Triangular
Ozone CRF (risk per 10 ppbv)	1.040 [1.013, 1.067]	Triangular

rationale for the choice of sample size is presented in the following section. The input variables and their distributions are discussed in the following sections and summarized in table 2.1. In monetizing health impacts, ten additional values are drawn from a distribution of the economic value of statistical life (VSL) and ten draws of a distribution of the Value of Life Year (VOLY), discussed in section 2.6.

Unless explicitly stated otherwise, I report the median value of the output distribution. Data sources and the uncertainties associated with the different parts of our study are detailed in the following subsections.

2.2 Monte Carlo simulation

To address uncertainties about the input parameters, I perform a Monte Carlo simulation with 1,000 draws. In order to characterize the convergence of our solution, I re-sample the output distribution of premature mortalities while varying the size of the sub-samples (applying the bootstrap technique).

For each sample size between 100 and 1,000, I perform 100 independent resamplings by drawing values from the original distribution using a normal random number generator. I then compare the distribution of the variances of the obtained distributions for each sample size. Figure 2-1 below presents the variance of the distribution of variances as a function of the sample size.

As the number of samples increases, the variance of the distribution of variances decreases, which means that the distribution of estimates of early deaths becomes

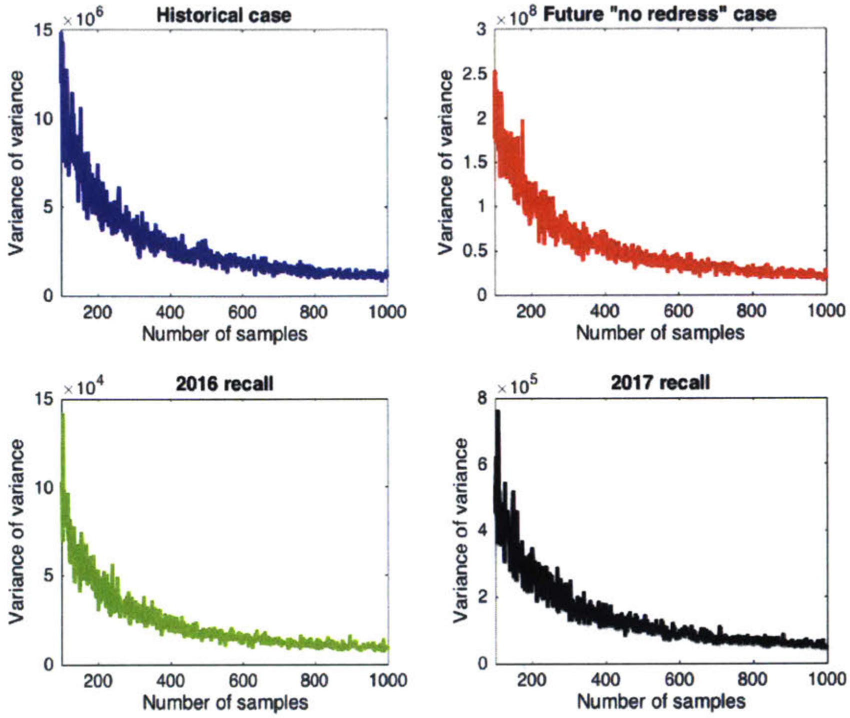

Figure 2-1: Variance of the distribution of variances of the distribution of mortality estimates obtained by re-sampling the original 1,000-sample distribution of estimates of premature mortalities, as a function of the size of the sub-sample

sample size represents a minor change in variance, and the results presented in the main paper assume the convergence of the results for a sample size of 1,000 samples.

2.3 Affected vehicle fleet and geographic distribution

An inventory of affected vehicle models in Germany is constructed from publicly available sources [10, 6]. It comprises cars marketed by the brands of the Volkswagen Group that used the EA 189 engine in its 1.2, 1.6, and 2.0 liter versions (Audi, Seat, Skoda, Volkswagen). I compute the number of affected cars for each relevant model that entered into service each year using annual new registration data published by the

German Federal Motor Transport Authority (KBA). In order to obtain estimates for vehicle-kilometers traveled (VKT) by all affected vehicles in each year, the registration data are used as inputs to the Stochastic Transport Emissions Policy (STEP) light-duty vehicle fleet model, developed by Bastani et al [12]. I further calibrate the model using Germany-specific data gathered by the TRACCS project [62]. A logistic function is used to estimate the vehicle retirement rate.

The total activity of the affected vehicles is expressed in vehicles-kilometers traveled (VKT) per year. The growth rate of this quantity over time is treated as a random variable and drawn from two triangular distributions, one applicable to the years prior to 2020 and one to the years between 2020 and 2040. The total activity for each year and each sample is estimated for each 25 km by 28 km grid cell within Germany using a spatially resolved emissions dataset of NO_x emissions from passenger cars, as reported by the German Federal Environmental Protection Agency [68].

2.4 Emissions factors for affected vehicles

The estimation of NO_x emissions factors (mass of NO_x emitted per kilometer driven) for the affected vehicles relies on two sets of on-road measurements. The first set of measurements is from KBA [36] using the Real-world Driving Emissions test cycle, as announced by the European Commission's Technical Committee of Motor Vehicles [26]. NO_x emission estimates are available for four vehicle models (VW Beetle 2.0L EA 189 Euro 5, VW Golf Plus 1.6L EA 189 Euro 5, VW Passat 2.0L EA 189 Euro 5 and VW Polo 1.2L EA 189 Euro 5). KBA published one measurement per model, yielding four on-road results.

The second set of measurements is from Thompson et al [66], who tested two vehicles - a 2012 Jetta (Vehicle A) and a 2013 Passat (Vehicle B) - on several drive cycles characteristic of different types of driving (urban, rural, highway). Each cycle was driven one or two times for each vehicle, yielding a total of 17 samples. I combined the results from both sources into a single distribution, composed of 21 samples. Each measured emissions factor expressed in grams of NO_x per kilometer is normalized by the fuel economy of the tested car (in liters of fuel per 100 kilometers). A truncated normal distribution of these factors is used to perform the draws for the Monte Carlo simulation method. The lower bound of the truncation corresponds to the Euro 5 limit of 0.18 $g.km^{-1}$ for the average fuel economy of the fleet of affected vehicles (namely 3.7 $g.L^{-1}$). The upper bound of the truncation is chosen to preserver the mean value of the distribution (and equals 29 $g.L^{-1}$). I obtain the emissions factor of each affected model by multiplying its fuel economy (in L/100 km) by the drawn value in 100 $g.L^{-1}$, yielding a value in $g.km^{-1}$, from which I compute the excess emissions factor. This approach assumes that the emissions factors scale linearly with fuel economy, and that the cars that were tested on-road constitute a representative sample of the fleet of affected vehicles. The excess emissions factor for each vehicle is multiplied by the corresponding VKT sample for this vehicle for a given year, and the sum over all vehicles yields the total amount of excess NO_x emitted each year.

Although VW have published technical details regarding the modifications to be made to each vehicle as part of the recall, at the time of publication there is no reliable data regarding the effect that these modifications have on the NO_x emissions of the affected vehicles. As such, I assume that the emissions of a recalled vehicle are brought down to the Euro 5 standard of 0.18 $g_{NO_x}.km^{-1}$ driven.

2.5 Air quality modeling

The GEOS-Chem chemistry-transport model, originally developed by Bey et al [14] and since continuously developed and updated, is used to calculate the $PM_{2.5}$ and ozone concentrations. The model domain encompasses most of Europe, covering 15° W-40° E and 33° N-61° N.The resolution is 0.25° in latitude and 0.3125° in longitude (approximately 25 km × 28 km). This resolution corresponds to 759 grid cells over Germany.

The GEOS-Chem model has been extensively used at comparable resolutions to capture $PM_{2.5}$ and ozone impacts at the ground level. Numerous studies have evaluated predicted $PM_{2.5}$ concentrations against observations in the US [34, 39, 40, 50, 77], and in Asia [16, 17, 45, 47, 70, 69]. Barrett et al [11] used the model over the US domain to predict $PM_{2.5}$ and ozone concentrations. Protonotariou et al [63] used the model over the European domain and evaluated the predicted ozone concentrations in Greece.

I use meteorological data from GEOS-FP, provided by the Global Modeling and Assimilation Office (GMAO) at NASA's Goddard Space Flight Center. Boundary conditions for this nested domain are obtained from a global GEOS-Chem run at 4° × 5° resolution, using the same meteorological source. I use the 2013 GEOS-FP data for all simulations. 2013 was a climatologically average year in Europe, with the average temperature 0.08 °C warmer than the 1995-2015 mean, and 0.16 °C below the mean of the period of interest 2008-2015 [55]. I use the EMEP anthropogenic emissions inventory for 2012 and note that emissions reductions between 2012 and 2013 associated with other anthropogenic sources are less than 3% [54]. Each simulation is run for a 15-month period with the first 3 months used as model spin up, during which the model is run but the output is not included in the analysis, to ensure that initial conditions do not impact the results.

A model simulation using baseline anthropogenic emissions (excluding excess emissions associated with the VW defeat device) is performed to assess the accuracy of the model in calculating $PM_{2.5}$ and ozone concentrations. Two scenario runs are then performed where a total of 72 and 130 kilotonnes (10^6 kg) excess NO_x emissions are added to the baseline anthropogenic emissions using the spatial distribution described in section 2.3. Excess NO_x emissions are input in the air quality model as NO. This is consistent with the fact that NO accounts for more than 80% of NO_x emissions from diesel vehicles [22, 75], and that the NO_x steady-state in the atmosphere is expected to be reached quickly with respect to the timescale of this study (3 to 30 minutes for NO concentrations of 10 to 1 ppb, following Seinfeld and Pandis [65]). The uncertainty associated with this hypothesis is further discussed in section 4. Differences in the $PM_{2.5}$ and ozone concentrations between the baseline and scenario simulations are attributed to the computed excess VW emissions. Given that the highest estimate for excess VW NO_x emissions is less than 0.8% of the background anthropogenic NO_x

excess VW NO_x emissions is less than 0.8% of the background anthropogenic NO_x emissions in Germany, I use a linear approach in estimating the $PM_{2.5}$ and ozone impacts of intermediate amounts of excess emissions. One additional scenario run with 520 kilotonnes of annual excess NO_x emissions was conducted in order to verify the validity of this linear approach. This is further discussed in the next section.

Simulated $PM_{2.5}$ and ozone model output concentrations over Germany are validated against observations from the European Environment Agency Air Quality e-Reporting dataset for Germany [4], as well as the entire European domain. Similar to Caiazzo et al [21], the model normalized mean biases are used to account for the uncertainty in predicting $PM_{2.5}$ and ozone concentrations. They are obtained from the point-to-point comparison between available measurements and the model predictions. The resulting distributions have means of 0.11 and 5.0×10^{-4} (95% CI: -0.15 to 0.91 and -0.18 to 0.35) for $PM_{2.5}$ and ozone, respectively. This implies that the model typically overpredicts concen- trations of $PM_{2.5}$ by 11%, and underpredicts ozone by 0.05%. The reciprocals of the biases are used as multiplicative factors to correct the GEOS-Chem model predictions in the uncertainty calculations. Only the results from the comparison with the observations in Germany are used in the uncertainty calculations.

2.6 Validity of the linear hypothesis

The baseline and the two excess emissions scenarios that I ran yielded three different maps of concentrations of $PM_{2.5}$ and ozone. The assumption that I use is that for any total mass of excess emissions occurring between two of these points (0, 72 kilotonnes and 130 kilotonnes), the output maps of concentrations ($PM_{2.5}$ and ozone) would scale linearly.

In order to test this assumption, I compute for each pair of point, the slope of the concentrations, obtained by dividing the point-to-point difference of the maps of concentrations by the difference of the masses of NO_x that were initially added to the baseline. Then, I characterize how far off the third point (that was not used to define the slope) is, by computing the distribution of the relative differences in concentrations between the linear prediction and the result of the simulation. I do this for both $PM_{2.5}$ and ozone, which results in six different distributions of relative differences. Their mean values range from $1.8 \times 10^{-4}\%$ to $2.4 \times 10^{-3}\%$ and their standard deviations from $6.0 \times 10^{-4}\%$ to $1.7 \times 10^{-2}\%$.

We note that no matter the pair of points chosen to compute the local slope, the characteristics of the distributions of premature mortalities obtained further downstream do not change significantly. For instance, the average number of early deaths in Germany because of excess emissions between 2008 and 2015 shifts by less than 3.4% and the median number by less than 4.2%.

We added to this analysis the results of a fourth scenario that was run with 520 kilotonnes of annual excess NO_x emissions in Germany. For each grid cell, I performed a linear regression of the predicted concentrations versus the annual excess emissions (0, 70, 130, and 520 kilotonnes). Three metrics of concentrations were assessed: annual average of $PM_{2.5}$ concentrations, annual average of daily one-hour maxima of ozone concentrations during the ozone season, and annual average of daily maxima of eight-hour average ozone concentrations. The corresponding maps of R^2 coefficients were set to zero over grid cells where there was no population from the countries of interest for this study. The results are shown below. These maps represent the validity of the linearity assumption over populated areas in the countries of interest.

Even though the linearity assumption is weaker at certain locations for the annual average of daily one-hour maxima of ozone concentrations, I note that the main driver of health impacts resulting from excess NO_x emissions in Germany is exposure to $PM_{2.5}$.

Finally, the hypothesis of local linearity for the levels of excess NO_x estimated is valid, and I use, for each estimate of the mass of excess NO_x, the linear interpolation defined by its two nearest neighbors among the points used for the original simulation. For instance, the health impacts of 30 kilotonnes of NO_x is derived using the slope computed from 0 and 72 kilotonnes, whereas for 100 kilotonnes, I use the slope computed from 72 and 130 kilotonnes.

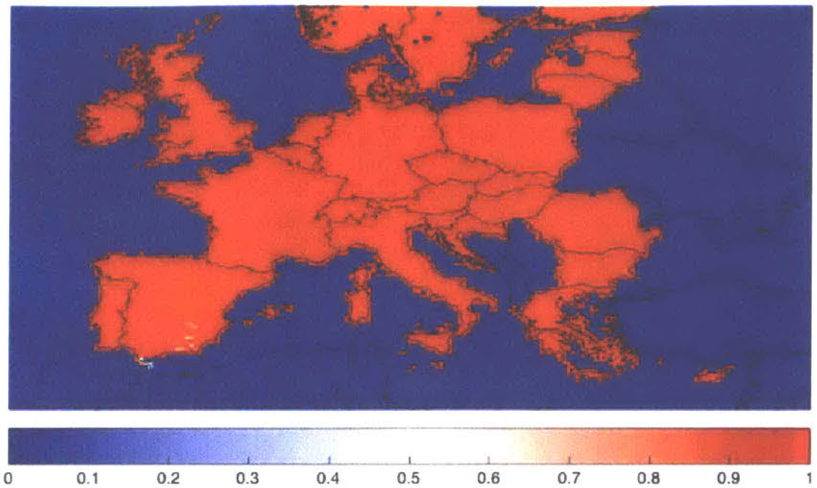

Figure 2-2: R^2 coefficients resulting from the linear regression of annual average $PM_{2.5}$ concentrations versus annual excess NO_x emissions in Germany. Only the coefficients over populated areas in the countries of interest for this study are shown here, the others were set to zero. This distribution is characterized by a mean value of 0.9957, a standard deviation of 0.0239, and a median value of 0.994

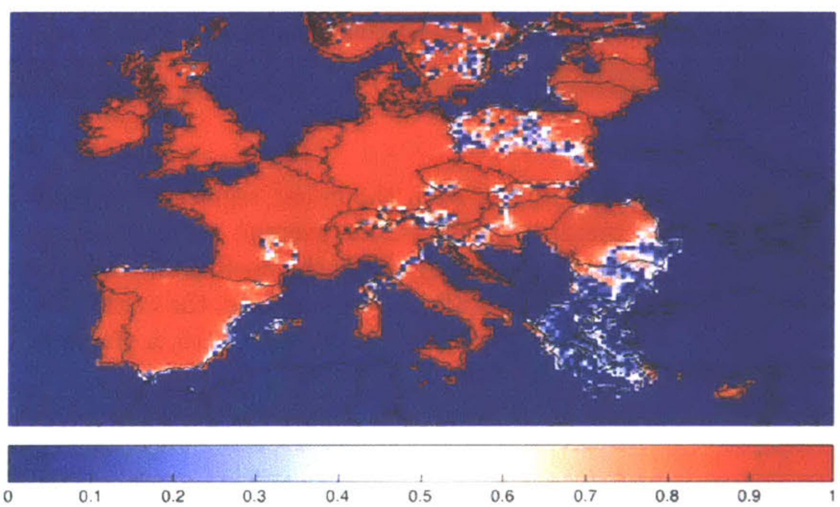

Figure 2-3: R^2 coefficients resulting from the linear regression of annual average of daily one-hour maxima of ozone concentrations versus annual excess NO_x emissions in Germany. Only the coefficients over populated areas in the countries of interest for this study are shown here, the others were set to zero. This distribution is characterized by a mean value of 0.8461, a standard deviation of 0.2620, and a median value of 0.9621.

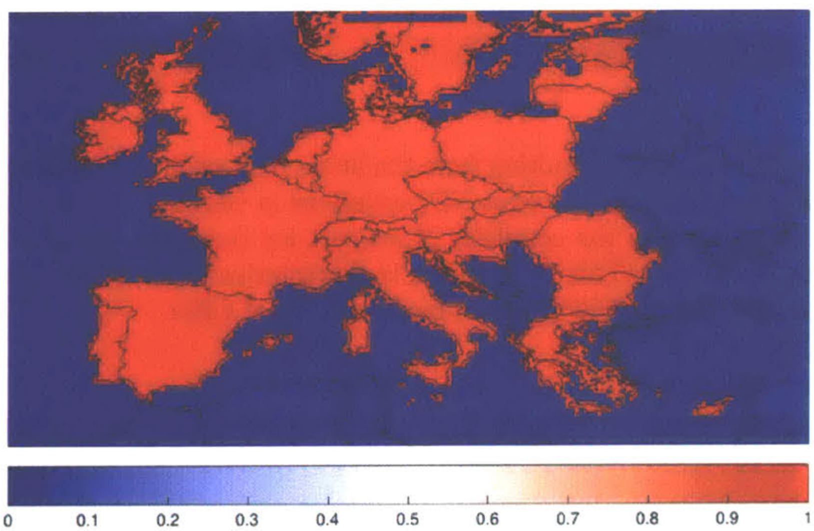

Figure 2-4: R^2 coefficients resulting from the linear regression of annual average of annual average of daily maxima of eight-hour average ozone concentrations versus annual excess NO_x emissions in Germany. Only the coefficients over populated areas in the countries of interest for this study are shown here, the others were set to zero. This distribution is characterized by a mean value of 0.9918, a standard deviation of 0.0138, and a median value of 0.9993.

2.7 Health impacts

Population exposure to both annual $PM_{2.5}$ and one- hour daily maximum ozone concentrations attributable to excess NO_x emissions from Volkswagen cars within Germany are calculated using the outputs of the GEOS-Chem air quality model described in section 2.4. The annual average $PM_{2.5}$ concentrations are obtained from the hourly output of GEOS-Chem for the species that constitute $PM_{2.5}$ (the GEOS-Chem tracers NH4, NIT, SO4, BPCI, BPCO, OCPI, and OCPO). One-hour daily maximum ozone concentrations during ozone season are obtained from the hourly output for the tracer O3. Population exposure is calculated by multiplying population counts in each grid cell by the annual-average $PM_{2.5}$ or one-hour daily maximum ozone concentration within that grid cell. The spatial distribution of population in Europe is taken from the LandScan database for 2013 [18], which is aggregated from a 1 km resolution to the model grid cells. Country-specific population counts are obtained from the UN World Population Prospects Division for each year of the study [28] and are used to scale the population density distribution. I take into account the current (2016) member states of the European Union and, due to their geographic proximity, Switzerland and Norway. I use the term "Europe" to refer to these states and note that the United Kingdom is still included in future year analyses irrespective of the date the state leaves the EU.

Epidemiological concentration-response functions (CRFs) are applied to estimate premature mortality resulting from population exposure to each species. Cardiovascular premature mortality from long-term exposure to $PM_{2.5}$ is calculated using a log-linear CRF, with a relative risk distribution taken from Hoek et al [41]. They find a central relative risk for cardiovascular disease mortality of 1.11 (95% CI: 1.05-1.16) per 10 $\mu g.m^{-3}$ increase in $PM_{2.5}$ exposure. Premature mortality due to exacerbation of both asthma and chronic obstructive pulmonary disease (COPD) as a result of exposure to ozone is also calculated using a log-linear CRF, this time applying the relative risk distribution from an American Cancer Society study [43]. This CRF has been widely used by previous studies to assess air quality impacts [11, 44]. Jerrett et al find a central relative risk of 1.04 (95% CI: 1.013-1.067) per 10 ppb increase in one-hour daily maximum ozone exposure during local ozone season. Country-specific baseline mortality rates for each disease are taken from the World Health Organization global burden of disease database [74] and are assumed to remain constant over time. I estimate premature mortality for adults aged 30 and older given that the epidemiological studies are based on a cohort of participants aged 30 years or greater [43, 49]. The age fraction for each country and each year is taken from UN population forecasts [28]. Alternative $PM_{2.5}$ CRF shapes and parameters, including the integrated exposure-response (IER) function applied in the 2010 Global Burden of Disease study [20, 51], a log-linear CRF from an American Cancer Society study [49] and an all- cause log-linear CRF reported in Hoek et al [41], are evaluated to determine the sensitivity of the result to the choice of CRF. I also implemented a log-linear ozone CRF derived from the findings of Turner et al [67] that link changes in exposure to the annual average of the daily maximum 8-hour average ozone concentrations to changes in mortality from respira- tory and circulatory diseases. These results are presented in the appendix. In all cases, a cessation lag structure is applied where 30% of the mortalities due to exposure in a given year occur in the first year, 50% occur equally in years 2 through 5, and the remaining 20% occur equally over years 6 through 20, based on US EPA [29] recommendations.

The relative risk factors for each of the CRFs are treated as independent, uncertain variables. I assume a triangular distribution with mode and 95% confidence interval taken from the corresponding epidemiological study.

We also compute the number of life-years lost for each estimated number of premature mortalities. The number of life-years lost is the product of mortalities in each age group (for adults over 30) with the age group's corresponding standard life expectancy, obtained from UN population forecasts for the appropriate year [28].

2.8 Monetization of health impacts

Mortality effects are valued using two monetization methods that have been widely used in the literature and by government agencies. A detailed overview is given by OECD [56]. The first method relies on estimates for the monetary value of changes in mortality risk due to $PM_{2.5}$ and ozone exposure, calculated as the Value of a Statistical Life (VSL). The second uses estimates for the Value Of a Life Year lost (VOLY) due to exposure to $PM_{2.5}$ and ozone. VSL and VOLY estimates are not additive but describe two different approaches for placing a monetary value on the health impacts of air pollution.

A distribution of VSL estimates for Europe is only available in year 2010 USD [56], therefore this distribution is converted to 2015 EUR by accounting for the USD/EUR purchasing power parity, changes in economic growth and inflation. The supplementary material contains detailed information on the conversion. I estimate a lower-bound VSL for 2015 of 1.82 million EUR and a higher bound of 5.48 million EUR, to which I fit a triangular distribution with an adjusted OECD base value of 3.65 million EUR as mode. For other years, I adjust VSL distributions for forecast changes in GDP per capita compared to 2015. The supplementary material shows the VSL distributions for all years considered in the analysis. Annual health costs from changes in premature mortality are calculated by multiplying the annual incidences of premature mortality due to excess NO_x emissions with a year-specific VSL estimate, assuming the mortality lag structure recommended by the EPA for air-quality impacts [60].

For VOLY, I use a year-2015 mean value of 133,000 EUR per year of life lost, with a standard deviation of 16,000 EUR derived from recommendations by European agencies [3, 52]. For other years, I adjust VOLY distributions for forecast changes in GDP per capita compared to 2015 in the same fashion as for the VSL. Annual health costs from life-years lost due to exposure to $PM_{2.5}$ and ozone in each year and each sample are calculated by multiplying the number of life-years lost by a year-specific estimate for the VOLY drawn from a year- specific normal distribution. Monetization with the VOLY method is therefore sensitive to age, with early deaths in younger age groups resulting in greater estimated costs than if they occur in older age groups.

Total health costs from historical excess emissions during the years 2008 to 2015 are expressed in year-2015 EUR using a social rate of time preference of 3 percent (discount rate), as recommended by the EU ExternE methodology [15] and by the US EPA [30] Total health costs because of additional incidences of premature mortality or life-years lost occurring in future years are expressed in year-2015 EUR with the same social rate of time preference as for historical costs. I follow Bickel and Friedrich [15] and also calculate results for lower and upper bounds of discount rates as sensitivity analyses (0% and 6%, respectively). These alternative discount scenarios are presented in appendix.

Chapter 3

Results

I report results for excess emissions, health impacts and health costs for historical excess emissions from the affected fleet as well as for future emissions from this fleet.

3.1 Excess emissions

Based on the drive cycle tests by KBA [36] and Thompson et al [66], I estimate the overall NO_x emissions indices for the affected vehicles to be 16 grams per liter fuel consumed. The associated 95% confidence interval (95% CI) is 5.0 to 28 $g.L^{-1}$. The central estimate is equivalent to 0.85 $g.km^{-1}$ for the given fleet distribution, compared to the Euro 5 standard of 0.18 $g.km^{-1}$. The total excess emissions for the affected fleet over time are shown in figure 3-1. The emissions factor is multiplied by the fuel economy of each model affected in order to obtain the mass of NO_x emitted per kilometer driven.

The total vehicle-kilometers traveled (VKT) by the affected vehicles between 2008 and 2015 is 354 billion km (95% CI: 349 to 360), corresponding to emissions of 240 kilotonnes of NO_x (95% CI: 28 to 440) above the total emissions that would have occurred if on-road emissions of all affected vehicles were equivalent to the Euro 5 NO_x limit of 0.18 $g.km^{-1}$. In the absence of a recall that brings on-road emissions down to Euro 5 limit values, future emissions from 2016 onward would be 560 kilotonnes of excess NO_x emissions (95% CI: 67 to 1100), as a result of total expected

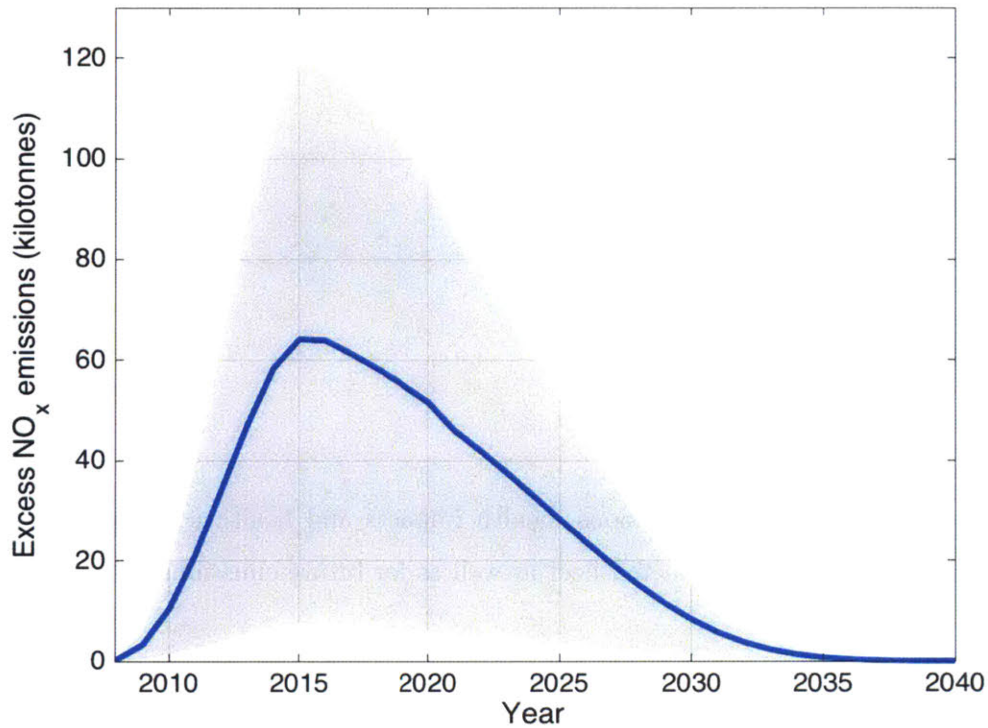

Figure 3-1: Annual excess NO_x emissions for the affected cars in kilotonnes. The results up to 2015 are estimates of the historical excess NO_x emissions and the results from 2016 onward assume no sales of new vehicles from September 2015 on and no return to recall of affected vehicles. The blue curve represents the median value, and the shaded area the 95% confidence interval.

future VKT for the affected vehicles from 2016 onward of 838 billion km (95% CI: 823 to 854). If, instead, on-road emissions from all affected vehicles are reduced to Euro 5 standards by the end of 2016, future excess NO_x emissions would be reduced to 29 kilotonnes (95% CI: 3.4 to 54). If the recall takes longer and is not complete until the end of 2017, future excess NO_x emissions increase to 59 kilotonnes (95% CI: 7.0 to 110).

Figure 3-2 represents the spatial distribution of the aggregated excess NO_x emissions between 2008 and 2015. It is assumed to follow the same spatial pattern as passenger cars NO_x emissions, as reported and gridded by UBA for the year 2010 [68].

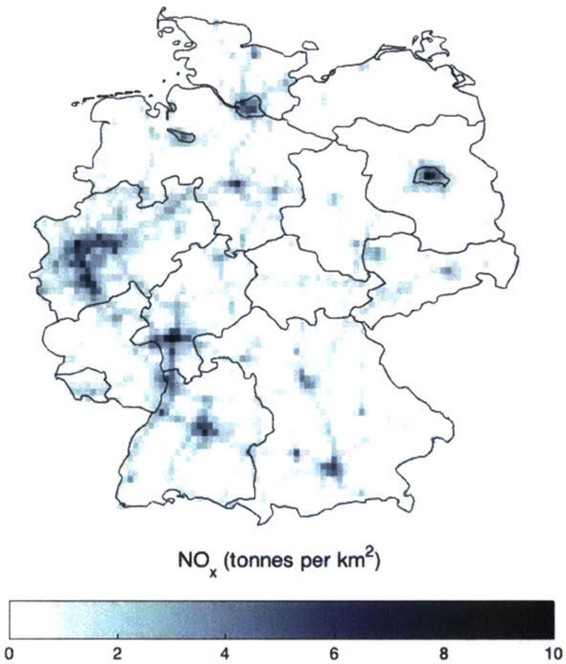

Figure 3-2: Spatial distribution of the aggregated excess NO_x emissions over 8 years (2008 to 2015) in tonnes per km^2 based on data provided by UBA [68]. This figure is at the resolution of the original UBA data, $0.1° \times 0.1°$. It shows the median value of the estimated aggregate excess emissions, namely 240 kilotonnes. Emission density peaks at 11.4 tonnes.km^{-2} . For comparison, the UBA inventory for the year 2010 only reports a total of 230 kilotonnes of NO_x emitted from passenger cars, with a peak emission density of 11 tonnes.km^{-2}.

3.2 Exposure and health impacts

For each year within the scope of our analysis and each draw of the Monte Carlo simulation, the linearity assumption allows us to compute the increase in the concentrations of $PM_{2.5}$ and ozone over Europe, as well as the corresponding increase in population exposure. Figures 3-3 (a) and (b) below present an example of the variation of the concentration of $PM_{2.5}$ and ozone, respectively, correspond- ing to annual excess emissions of 48 kilotonnes of NO_x, the approximate median value for 2013. Figures 3-3 (c) and (d) show the corresponding population exposure to $PM_{2.5}$

and ozone respectively, and are obtained by multiplying the concentration of $PM_{2.5}$ or ozone in each grid cell by the population count in the cell.

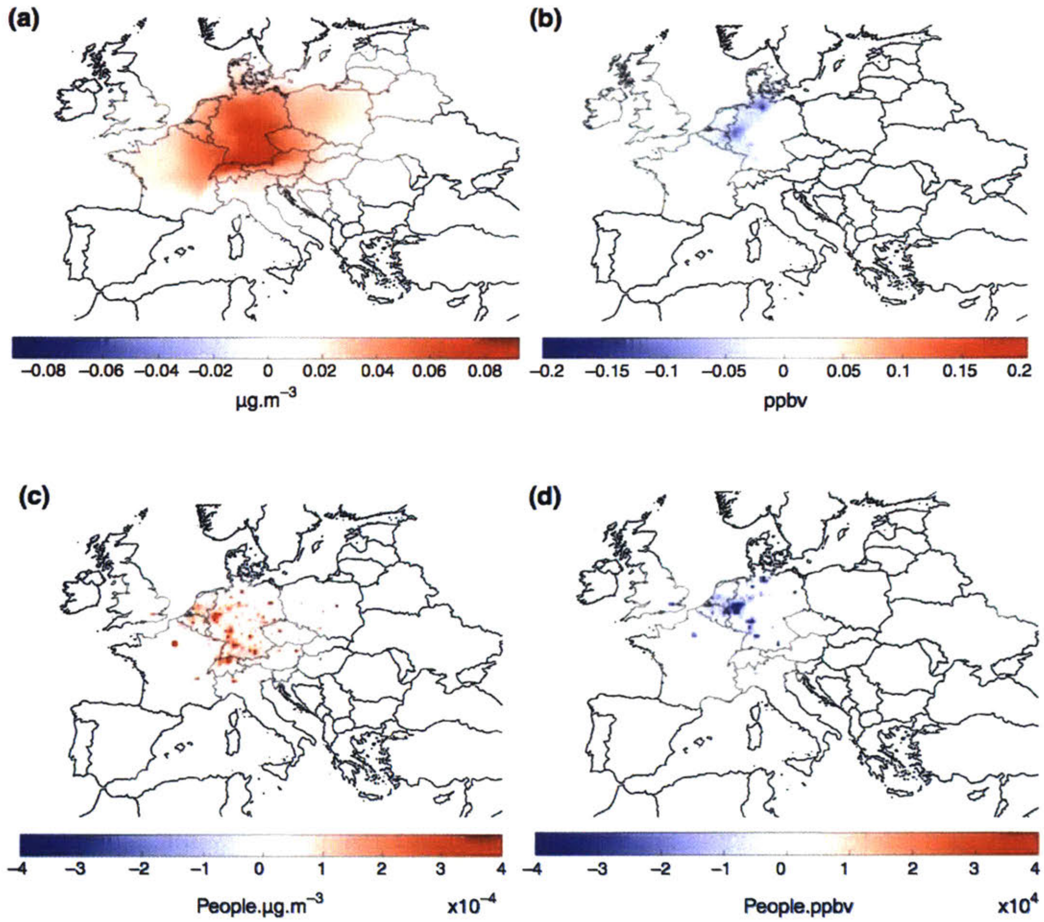

Figure 3-3: (a) and (b) Increase in concentration of $PM_{2.5}$ (3a, left) and ozone (3b, right) resulting from the emission of 48 kilotonnes of NO_x in a year. (c) and (d) Corresponding increase in population exposure expressed in people (over 30 years old) micrograms per cubic meter for $PM_{2.5}$ (3c, left) and people (over 30 years old) particles per billion for ozone (3d, right).

The excess NO_x emissions result in reduced ozone concentrations in almost all countries impacted by the excess emissions (with the exception of the region of the Alps between Switzerland and Austria). This is attributed to the high concentration of NO_x relative to volatile organic compound (VOC) concentrations at these locations. Under these conditions, an increase in NO_x emissions inhibits the production of ozone due to the removal of oxidants from the atmosphere via chemical reaction with NO_x [65]. These atmospheric conditions are denoted as VOC-limited chemical regimes and have been established for Europe by previous studies [13, 53]. In other locations with higher VOC concentrations, it appears that long-range transported NO_x from Germany dominates ozone production.

After applying the concentration response functions to the computed population exposures to $PM_{2.5}$ and ozone, I are able to evaluate the number of premature mortalities that resulted from the estimated excess NO_x emissions.

Figure 3-4 summarizes the health impacts associated with excess emissions in Germany and in Europe. It features a retrospective analysis for the period 2008 to 2015 and three prospective scenarios, the first one without a recall of the affected cars, the second one with a recall completed by the end of 2016 and the third one with a recall completed by the end of 2017.

The benefits of a recall completed at a constant rate over the course of 2016 are obtained by comparing the scenario assuming that all affected cars are driven during their whole lifecycle without any modification to the scenario where all cars are brought down to the Euro 5 standard by the end of 2016. I also computed the benefit of a recall completed by the end of 2017. The mortality reductions from reduced ozone exposure amount to approximately 10% of the aggregate estimates shown in this table.

Figure 3-5 presents detailed estimates for the number of early deaths occurring in the ten countries that were most affected by excess emissions released between 2008 and 2015. The results for other European countries, as well as the distribution of the number of life-years lost per country, are presented in appendix.

		Germany	Europe (excluding Germany)	Europe
2008 to 2015 estimated impacts	Exposure to $PM_{2.5}$ [a]	14 (1.8; 27)	17 (0.23; 35)	31 (0.41; 62)
	Exposure to ozone [a]	−11 (−13; −2.1)	−4.2 (8.1; −0.50)	−16 (−30; −1.8)
	Early deaths	500 (54; 1200)	660 (71; 1600)	1200 (130; 2800)
	Number of life-years lost	5600 (610; 14 000)	7700 (830; 19 000)	13 000 (1400; 32 000)
	Mortality costs (VSL) [b]	1.5 (0.17; 3.7)	2.0 (0.23; 4.8)	3.5 (0.40; 8.5)
	Mortality costs (VOLY) [b]	0.80 (0.088; 1.9)	1.1 (0.12; 2.6)	1.9 (0.21; 4.9)
Forecast impacts from 2016 assuming no recall	Exposure to $PM_{2.5}$ [a]	34 (4.4; 67)	44 (0.57; 87)	77 (10; 150)
	Exposure to ozone [a]	−27 (−52; 3.2)	−11 (−21; −1.3)	−38 (−73; −4.5)
	Early deaths	1200 (130; 3000)	1600 (180; 4000)	2900 (310; 7000)
	Number of life-years lost	14 000 (1500; 34 000)	19 000 (2100; 46 000)	33 000 (3600; 80 000)
	Mortality costs (VSL) [b]	3.8 (0.44; 9.4.)	5.1 (0.58; 12)	8.9 (1.0; 22)
	Mortality costs (VOLY) [b]	2.0 (0.22; 4.8)	2.7 (0.30; 6.6)	4.6 (0.52; 11)
Benefit of a recall completed by the end of 2016	Exposure to $PM_{2.5}$ [a]	32 (4.2; 63)	41 (0.54; 82)	74 (9.6; 150)
	Exposure to ozone [a]	−26 (−49; −3.1)	−10 (−20; −1.2)	−36 (−69; −4.3)
	Early deaths	1200 (130; 2800)	1600 (170; 3800)	2700 (300; 6600)
	Number of life-years lost	13 000 (1400; 32 000)	18 000 (2000; 44 000)	31 000 (3400; 76 000)
	Mortality costs (VSL) [b]	3.6 (0.42; 8.8)	4.8 (0.55; 12)	8.4 (0.97; 21)
	Mortality costs (VOLY) [b]	1.9 (0.20; 4.5)	2.5 (0.28; 6.2)	4.4 (0.49; 11)
Benefit of a recall completed by the end of 2017	Exposure to $PM_{2.5}$ [a]	30 (4.0; 60)	39 (0.51; 78)	69 (9.1; 140)
	Exposure to ozone [a]	−25 (−47; −2.9)	−9.7 (−18; −1.2)	−34 (−65; −4.1)
	Early deaths	1100 (120; 2700)	1500 (160; 3600)	2600 (280; 6200)
	Number of life-years lost	12 000 (1400; 30 000)	17 000 (1900; 42 000)	29 000 (3200; 72 000)
	Mortality costs (VSL) [b]	3.4 (0.39; 8.3)	4.5 (0.52; 11)	7.9 (0.91; 19)
	Mortality costs (VOLY) [b]	1.7 (0.19; 4.2)	2.4 (0.27; 5.8)	4.1 (0.46; 10)

[a] Exposure is in million of people over 30 years old times $\mu g\, m^{-3}$ (or times ppbv for ozone).
[b] Mortality costs are expressed in billion 2015 EUR.

Figure 3-4: Estimated historical and forecast impacts of excess NO_x emissions. The median estimates are reported with 95% CI in parentheses. Results after 2016 assume no new sales of affected vehicles. Note that the sums of medians may not equal the median of sums. All values are rounded consistently with the order of magnitude of the standard deviation of the distribution considered.

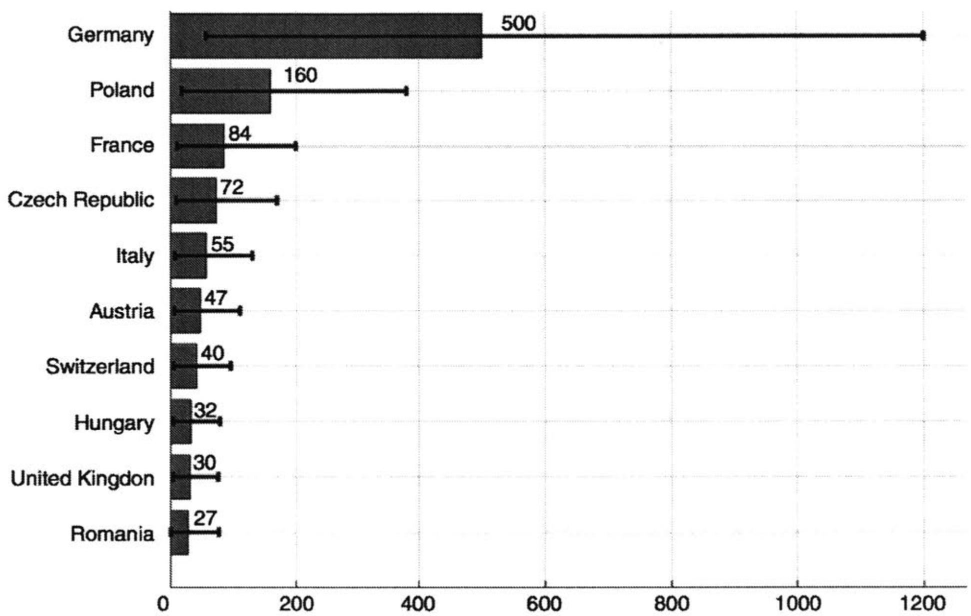

Figure 3-5: Number of premature mortalities due to excess NO_x emissions released in Germany between 2008 and 2015 in the ten most affected countries. The numerical values correspond to the median estimate in each case. The solid bars represent the 95% confidence intervals.

3.3 Health costs

Monetized health impacts of the excess NO_x emissions are shown in figure 3-4. Costs calculated using the VOLY method are approximately 50% lower on average than those calculated using the VSL, with an average estimate of 11 life-years lost per premature mortality. This difference is due to the prevalence of cardiovascular and respiratory disease in the older population, resulting in the majority of air-quality-related mortalities occurring in higher age brackets with lower life expectancies (in terms of life-years remaining).

Using the VOLY method, I find that past excess emissions have already caused median health costs of 1.9 billion EUR, and that median additional costs of 4.6 billion EUR are expected under a "no recall" scenario. 95% of these costs are avoided if the recall is completed by the end of 2016; this is reduced to 90% if the recall is not completed until the end of 2017. The costs presented here do not take into account non-mortality impacts such as new cases of chronic bronchitis or increased hospital admissions, but these are expected to be small relative to costs associated with premature mortality [11].

Regardless of the valuation method used, I find that the majority of monetary damages are incurred outside of Germany, with approximately 60% of all mortality costs exported to other countries within the area of interest. This implies that there might also be significant health costs within Germany due to diesel cars operating outside of Germany which are not accounted for in this study.

Chapter 4

Discussion and conclusion

I compare our results to an existing study by Oldenkamp et al [61] which estimates Europe-wide health impacts of VW excess emissions using a simplified modeling approach. The authors calculate point estimates for health impacts from spatially unresolved, constant relationships between excess emissions and mortality impacts. Using this approach, the authors link estimated historical excess NO_x emissions of 491.7 kilotonnes to 5,000 early deaths, and estimated future emissions in the absence of vehicle modifications of 802.2 kilotonnes to an additional 8,200 early deaths. I use a chemistry-transport model to estimate spatially resolved population exposure in each European country as a result of the additional NO_x emissions in Germany. Using this approach yields an estimate for premature mortality per unit of excess emissions that is 50% lower than the value used in Oldenkamp et al [61], i.e. 5.1 vs. 10.2 early deaths per kilotonne of excess emissions.

In the case of the US, Barrett et al [11] report median excess NO_x emissions of 36.7 kilotonnes for 2008 from 2015 from 482,000 affected cars sold, leading to 59 additional early deaths at the median. Similarly, Oldenkamp et al [61] found 33.8 kilotonnes of excess NO_x over the same period, yielding 59 premature mortalities. Holland et al [42], found 45.1 kilotonnes of NO_x and 46.1 mortalities. In our study, I find that while the number of VW cars sold in Germany is 440% higher than the number sold in the US, excess emissions in Germany are 540% higher than in the US, which is a result of annual kilometer driven per vehicle in Germany being 19 percent higher than in the US. I note however that our estimate is for the average diesel vehicle in Germany, whereas Barrett et al [11] compute the annual mileage of an average American vehicle.

Further comparing the results of these studies to our results, I find that each unit of NO_x emitted in Europe results in 5 times as many premature mortalities (per capita) as in the US. This difference can be explained by the combination of Europe's greater population density and its more NO_x-sensitive background conditions. A study by Koo et al [48] found that in Europe, a combination of high ammonia emissions and relatively low sulfur emissions results in a higher sensitivity of population-weighted $PM_{2.5}$ to NO_x emissions than any other region in the world apart from Eastern China, where population densities are comparable to or greater than Europe. This aspect is discussed further in appendix.

4.1 Limitations

Some sources of uncertainty in this study have not been quantified. The fleet model does not account for possible differences in distance traveled by vehicle model considered. Vehicle kilometers traveled for a given year are assumed to depend only on the year and the age of the vehicle. However, I note that the TRACCS report [62] distinguished between four categories of diesel passenger cars (small, lower-medium, upper-medium, executive) and found no significant differences among these categories in terms of vehicle kilometers traveled, when controlling for year and vehicle age (less than 5% for the average vehicle). I also assume that the spatial distribution of the excess NO_x emissions and population remains constant over time.

The excess NO_x emissions are input into the air quality model as pure NO. While it is usually considered that NO accounts for more than 80% of NO_x emissions from diesel vehicles [22, 75], it appears that the NO_2 to NO_x ratio in primary emissions from diesel vehicles is increasing over time [7]. However, the NO_x steady-state is reached in about 3 to 30 minutes for ambient NO concentrations of 10 ppb to 1 ppb respectively [65], which means that the initial NO_2 to NO_x ratio is not expected influence $PM_{2.5}$ production. Ashok et al [9] have shown in the case of aviation that the maximum sensitivity of population $PM_{2.5}$ exposure to the NO_2 to NO_x ratio in emissions was 2.75%. However, $PM_{2.5}$ is the main driver of the health impacts estimated in this study, suggesting that our results would not vary significantly if the NO_2 to NO_x ratio in the primary emissions were modified. As for ozone, the fact that excess emissions are input as pure NO results in increased nighttime titration of ozone. However, I find that this does not occur during daylight hours due to the high background ozone concentrations. Since the ozone health impact metrics are based on exposure to 1-hour daily maximum and 8-hour daily maximum ozone concentrations which occur during the daylight hours, the calculated ozone impacts are insensitive to the primary NO_2 to NO_x ratio.

I assume the toxicity of different PM species to be equal, consistent with standard practice. However, it is worth noting that the particle composition of PM2.5 is suspected to influence the related health impacts, as pointed out by Hoek et al [41] and WHO [64]. Since NO_x emissions impact ammonium nitrate most strongly, any differential toxicity of ammonium nitrate relative to the basket of urban PM for which CRFs are derived is not captured. The standard GEOS-Chem chemical mechanism does not include a detailed secondary organic aerosol model, and these compounds are therefore not included in our health impact assessment.

Although some epidemiological studies have been published which find a link between increased exposure to NO_2 and premature mortality independent of exposure to other pollutants, there remains substantial disagreement with regard to the specific health outcomes affected and their magnitude. If I include the health impacts of changes in exposure to NO_2 due to VW excess emissions following the recommendations of the HRAPIE group [73], I find that excess NO_x emissions in Germany released between 2008 and 2015 caused 1,300 (95% CI: 160 to 2800) additional premature mortalities due to exposure to NO_2, including 1,200 in Germany. The associated health costs are estimated to be 2 billion EUR (median VOLY estimate). Future emissions in the absence of modification of affected cars are expected to cause 3,200 additional premature mortalities from exposure to NO_2 in Europe (including 2,800 in Germany) and 5.2 billion EUR in associated health costs. Including NO_2 impacts would approximately double the total impacts estimated earlier, with Germany bearing the great majority of this additional burden. It should also be noted that NO_2 is a precursor to ozone and secondary particulate matter (these effects are accounted for in our study).

When using a CRF derived from the findings of Turner et al [67], I observed greater reductions in mortality due to reduced ozone exposure than our central estimate (derived from Jerrett et al [43]). Integrated excess emissions over the period 2008 to 2015 are estimated to have averted 28 premature mortalities (95% CI: 1.7 to 85) with the Jerrett et al [43] CRF, while the Turner CRF predicts 78 avoided mortalities (95% CI: 0.28 to 640). The total impacts in Europe of past excess emissions (from exposure to both $PM_{2.5}$ and O_3) using the Turner CRF as the central estimate would then be 1000 premature mortalities (95% CI: 93 to 2,700). This result is to be compared to the 1,200 mortalities obtained with our central CRF (95% CI: 130 to 2,800). Using the Turner CRF as the central ozone estimate, I find that future emissions in the absence of any modification to the affected cars are predicted to cause 2,600 (95% CI: 230 to 6,700) additional premature mortalities in Europe. This is to be compared to 2,900 (95% CI: 310 to 7,000) additional premature mortalities estimated with the Jerrett CRF as our central ozone estimate. In conclusion, changing our central estimate for ozone-related health impacts from Jerrett et al [43] to Turner et al [67] yields 10% lower mortality estimates. The appendix provides results obtained with different $PM_{2.5}$ CRFs.

In addition, our assessment does not include morbidity impacts, which would increase the health costs of the estimated excess emissions but typically account for less than 10% of total health costs associated with air quality impacts [60]. It should also be mentioned here that although the epidemiological studies used here and in the supplementary material are widely used in the literature (Barrett et al [11] for instance), they represent only a small portion of the health literature on $PM_{2.5}$ and ozone and mortality.

I also note that no potential environmental benefits of excess NO_x emissions are included in the analysis. An example could be the reduced use of diesel exhaust fluid in the selective catalytic reduction, which could lead to a lower risk of ammonia leakage, a phenomenon which has been known to contribute to the $PM_{2.5}$ health impacts of the road transportation sector at approximately equal magnitude as NO_x [27].

Finally, I note that the assessment of model performance and the corresponding correction by reciprocal biases use the baseline scenario and the officially reported emissions data. These might not fully account for real on-road emissions, and this issue might introduce a second-order effect that is not accounted for. However, given the magnitude of the perturbation (less than 0.8% of total NO_x emissions), this effect is not expected to alter our results significantly.

4.2 Wider implications

In January 2016, and following orders by the KBA [46], VW started to recall affected vehicles with the aim of reducing their NO_x emissions to an as-yet unspecified level. The modification costs per car have been reported to amount to 60 EUR, excluding the value added tax [76]. I assume that the vehicle modification can be completed within 1 hour, including time required to travel to and from the modification site. Applying an average value-of-time of 40 EUR per hour [1], modification of one vehicle incurs a total cost of approximately 100 EUR. The overall cost of the recall would then be 240 million EUR. This estimate does not include other costs such as increased fuel expenses or social costs of additional carbon emissions if the vehicle modification results in a reduction in fuel economy. Assuming that the modification reduces on-road emissions to the Euro 5 standard, the total cost of the recall amounts to 5% of the median 4.4 billion EUR health costs (using the VOLY approach) which I estimate will be avoided if the recall is completed by the end of 2016. Our analysis suggests that the recall will remain net beneficial as long as the per-vehicle cost is below 1,800 EUR.

Although this analysis was centered on Volkswagen-manufactured cars operated in Germany, NO_x emissions from diesel vehicles in excess of the existing standards constitute a broader and ongoing issue in the European Union. On-road measurements have shown that vehicles from several other manufacturers emit more NO than the applicable limits [23, 35, 36], despite the absence of any identified measures intended to circumvent emissions tests. These discrepancies have motivated the development of on-road emissions testing procedures [71, 26] which were approved by the European Council in February 2016. The VW vehicles modeled in this study comprise 2.6 million of the 13 million diesel passenger vehicles operating in Germany in 2012 [32]. Consequently, there will be additional health impacts due to excess emissions from vehicles produced by other manufacturers that are not accounted for in our study. In addition, there were a total of 86 million diesel passenger vehicles operating in the EU28 in 2010 [62], making excess diesel NO_x emissions a public health concern for all of Europe. On the modeling side, current road transport NO_x inventories are also likely to be affected, if they are based on measurements from laboratory test cycles (e.g. EEA 2013 [2]), given that emissions from real-world driving conditions could be significantly different than those measured during a laboratory test cycle [35].

Appendix A

Appendix

A.1 Sensitivity of $PM_{2.5}$ to NO_x in Europe compared to the US

I find that, per unit of NO_x emitted, the increase in mortality rate (per capita) is 5 times greater for Germany than the US (Barrett et al [11]). This difference is partially attributable to greater population density in Europe than in the US. A second factor is the greater sensitivity of European $PM_{2.5}$ to NO_x conditions. Previous studies, such as Koo et al [48], have suggested that the background atmospheric composition in the European surface layer makes for efficient $PM_{2.5}$ from NO_x emissions. The quantity of secondary formation of $PM_{2.5}$ associated with NO_x emissions is related to the "gas ratio" (GR) [8], approximated as

$$GR = \frac{[NH_3] - 2 \times [SO_4]}{[NO_3]} \tag{A.1}$$

where NO_3 is total nitrate (nitric acid and nitrate aerosol), NH_3 is total ammonia (ammonia gas and ammonium aerosol) and SO_4 is sulfate aerosol. Where the gas ratio is greater than 1, the system is NO_x-limited and there is ammonia available to be neutralized; this means that NO_x emissions are likely to result in the formation of ammonium nitrate aerosol. Where the gas ratio is less than 1, the system is ammonia-limited, and there is insufficient ammonia to neutralize all available sulfate. Under

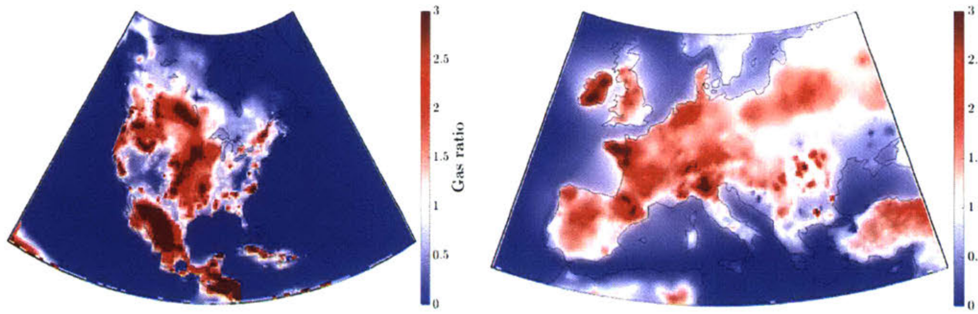

Figure A-1: Annual-average gas ratio calculated by the forward component of the GEOS-Chem adjoint in the US (left) and by the GEOS-Chem forward model in Europe (right).

these conditions, additional NO_x will be forced to compete with sulfate to form an ammoniated aerosol, reducing the yield of $PM_{2.5}$ expected from a NO_x emission.

Figure A-1 shows the gas ratio as calculated in the US and Europe for the Barrett et al [11] study and this study, respectively. The gas ratio is consistently larger than 1 throughout most of Europe, with pronounced peaks in northern Italy and western France. As a result, NO_x emitted anywhere in this region is likely to result in efficient formation of $PM_{2.5}$. By comparison, the US is more heterogeneous. Gas ratios on the densely-populated East coast are mostly less than 1, with the exception of North Carolina. Although high gas ratios are observed further west, they are mostly isolated to the sparsely-populated Midwest, while the coastal population centers on the west coast such as in California are protected by westerly winds. I therefore propose that both chemical and demographic factors are necessary to explaining the difference between the response of US and European mortality rates to NO_x emissions.

A.2 Calculation of the value of a statistical life

VSL estimates are taken from OECD recommendations for Europe [56]. We convert the high, low and base values in 2010 USD provided by OECD to year 2015 EUR by first converting 2010 USD into 2010 EUR using a purchasing power parity of 0.796 [59]. Secondly, VSL changes with real income changes and we therefore follow the proposed approach by OECD [56] to adjust VSL for changes real Gross Domestic Product per capita between 2010 and 2015, using the recommended income elasticity of VSL of 0.8. Alternatively adjusted VSL values derived from an income elasticity of 0.08 and 1.0, respectively, are calculated as a sensitivity as proposed by US EPA [30] and the results are presented in this supplementary material. Finally, the purchasing power and income change adjusted VSL values are also adjusted for changes in prices between 2010 and 2015 by employing a Europe-specific consumer price index change during this period of 6.9% [19]. This yields an adjusted 2015 low VSL of 1.82 million EUR and a high VSL of 5.48 million EUR, to which we fit a triangular distribution with an adjusted OECD base value of 3.65 million EUR as mode. For other years, we adjust VSL distributions for differences in GDP per capita compared to 2015 using an income elasticity of VSL of 0.8 as outlined before. Alternative VSL using different income elasticities are presented in table A.2. Historical real GDP per capita changes in Europe are taken from Eurostat [33] and long-term annual GDP per capita growth for Europe is assumed to be 1.5% as projected by OECD [57]. The VSL distributions for the years considered in the main analysis are shown in table A.1 below.

Health costs are calculated for each year by multiplying the annual incidences of premature mortality due to excess NO_x emissions with a year-specific VSL estimate, assuming the mortality lag structure recommended by the EPA for air-quality impacts [60]. Total health costs from historical excess emissions during the years 2008 to 2015 are expressed in year-2015 EUR using a social rate of time preference of 3 percent, as recommended by the ExternE methodology developed in the EU [15] and by the US EPA [30]. Total health costs because of additional incidences of premature mortality occurring in future years are expressed in year-2015 EUR with the same social rate of time preference as for historical costs. We follow Bickel and Friedrich [15] and also calculate results for lower and upper bounds for discount rates as sensitivity analysis (0% and 6%, respectively) presented below.

Table A.2 summarizes our results using three different values for the income elasticity of VSL (IE), and three different values for the discount rate. The values presented in the table below aggregate estimated health costs from historical excess

Table A.1: Estimated parameters of the triangular distribution used for the Value of Statistical Life. The parameters are expressed in million 2015 EUR.

Year	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**
low	1.69	1.60	1.66	1.69	1.71	1.72	1.76	1.83
mode	3.37	3.21	3.31	3.38	3.42	3.45	3.53	3.65
high	5.06	4.81	4.97	5.08	5.14	5.17	5.29	5.48
Year	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**2023**
low	1.85	1.87	1.89	1.91	1.94	1.96	1.98	2.01
mode	3.69	3.74	3.78	3.83	3.87	3.92	3.97	4.02
high	5.54	5.61	5.67	5.74	5.81	5.88	5.95	6.02
Year	**2024**	**2025**	**2026**	**2027**	**2028**	**2029**	**2030**	**2031**
low	2.03	2.06	2.08	2.11	2.13	2.16	2.18	2.21
mode	4.06	4.11	4.16	4.21	4.26	4.31	4.36	4.42
high	6.10	6.17	6.24	6.32	6.39	6.47	6.55	6.63
Year	**2032**	**2033**	**2034**	**2035**	**2036**	**2037**	**2038**	**2039**
low	2.23	2.26	2.29	2.32	2.34	2.37	2.40	2.43
mode	4.47	4.52	4.58	4.63	4.69	4.74	4.80	4.86
high	6.70	6.79	6.87	6.95	7.03	7.12	7.20	7.29
Year	**2040**	**2041**	**2042**	**2043**	**2044**	**2045**	**2046**	**2047**
low	2.46	2.49	2.52	2.55	2.58	2.61	2.64	2.67
mode	4.92	4.98	5.04	5.10	5.16	5.22	5.28	5.34
high	7.37	7.46	7.55	7.64	7.73	7.83	7.92	8.02
Year	**2048**	**2049**	**2050**	**2051**	**2052**	**2053**	**2054**	**2055**
low	2.70	2.74	2.77	2.80	2.84	2.87	2.90	2.94
mode	5.41	5.47	5.54	5.60	5.67	5.74	5.81	5.88
high	8.11	8.21	8.31	8.41	8.51	8.61	8.71	8.82
Year	**2056**	**2057**	**2058**	**2059**	**2060**			
low	2.97	3.01	3.05	3.08	3.12			
mode	5.95	6.02	6.09	6.17	6.24			
high	8.92	9.03	9.14	9.25	9.36			

Table A.2: Overall median health costs in Europe obtained by the VSL method for different discount rates. ÊValues shown are aggregate estimated health costs from historical excess emissions as well as future excess emissions in the absence of a recall scenario. 95% CI are in brackets.

Billion 2015 EUR	**0%**	**3%**	**6%**
IE 0.8	15 (0; 41)	12 (0; 33)	10 (0; 28)
IE 0.08	13 (0; 35)	10 (0; 29)	8.9 (0; 25)
IE 1	16 (0; 43)	13 (0; 34)	11 (0; 29)

emissions as well as future excess emissions in the absence of a recall scenario.

We note that lowering the income elasticity of VSL from 0.8 to 0.08 lowers the median of the distribution of estimated health costs by approximately 14%. An increase of the IE to 1 increases the median estimated costs by 8%.

A.3 Results obtained with other CRFs

Applying different concentration-response functions to the simulated concentrations of $PM_{2.5}$ and ozone yields different results. The log-linear CRF for $PM_{2.5}$ from the American Cancer Society (Krewski et al [49], mentioned in the methods section) leads to results that are on average 80% higher than those reported in the main paper. Results published for the United States [11] use the ACS CRF. The all-cause CRF obtained from Hoek et al [41] yields results that are on average 108% greater than those obtained in this study. Finally, the integrated exposure-response function taken from Burnett et al [20] predicts on average 25% more premature mortalities than the results based on the methods we describe in the main paper.

Additionally, we implemented a CRF derived from the findings of Turner et al [67]. It yielded greater reductions in mortality due to reduced ozone exposure than our central estimate (derived from Jerrett et al [43]). Integrated excess emissions over the period 2008 to 2015 are estimated to have averted 28 premature mortalities (95% CI: 1.7 to 85) with the Jerrett et al [43] CRF, while the Turner CRF predicts 78 avoided mortalities (95% CI: 0.28 to 640). The total impacts in Europe of past excess emissions (from exposure to both $PM_{2.5}$ and O_3) using the Turner CRF as the central estimate would then be 1,000 premature mortalities (95% CI: 93 to 2,700). This result is to be compared to the 1,200 mortalities obtained with our central CRF (95% CI: 130 to 2,800). Using the Turner CRF as the central ozone estimate, we find that future emissions in the absence of any modification to the affected cars are predicted to cause 2,600 (95% CI: 230 to 6,700) additional premature mortalities in Europe. This is to be compared to 2,900 (95% CI: 310 to 7,000) additional premature mortalities estimated with the Jerrett CRF as our central ozone estimate. In conclusion, changing our central estimate for ozone-related health impacts from Jerrett et al [43] tu Turner et al [67] yields 10% lower mortality estimates.

A.4 Detailed results for health impacts estimates

The estimates of the health impacts of excess emissions were obtained for each country separately, taking into account country-specific population data and baseline mortality incidences. The estimates were then added sample by sample and year by year to produce the aggregate results that are presented in the main study. Table A.3 presents our estimates of the number of premature mortalities due to excess emissions released between 2008 and 2015 (mean and median values, 2.5 and 97.5 percentiles). Table A.4 presents the corresponding number of life-years lost. Table A.5 shows our estimate of the number of premature mortalities due to excess emissions in a prospective scenario where the affected cars stay on the road until retirement. Table A.6 presents the corresponding number of life-years lost.

Table A.3: Mean, median, 2.5 and 97.5 percentiles of the distribution of premature mortalities caused by historical excess emissions in Germany.

Country	2.5 percentile	Median	97.5 percentile	Mean
Germany	55	500	1200	540
Poland	17	160	380	170
France	9.1	84	200	89
Czech Republic	7.8	72	170	76
Italy	5.9	55	140	59
Austria	5.1	47	110	50
Switzerland	4.3	40	95	42
Hungary	3.4	32	78	34
UK	3	30	75	32
Romania	3	28	69	30
Netherlands	2.5	25	63	27
Belgium	2.4	23	58	25
Slovakia	1.9	17	42	18
Croatia	0.96	8.8	21	9.3
Lithuania	0.89	8.3	20	8.7
Bulgaria	0.69	6.4	15	6.7
Spain	0.64	5.8	14	6.1
Denmark	0.52	4.8	12	5.1
Sweden	0.45	4.6	12	5.2
Latvia	0.45	4.2	10	4.4
Slovenia	0.4	3.7	8.9	3.9
Greece	0.17	1.5	3.7	1.6
Luxembourg	0.12	1.1	2.6	1.2
Estonia	0.11	1	2.4	1.1
Ireland	0.1	0.99	2.5	1.1
Portugal	0.08	0.7	1.7	0.74
Finland	0.05	0.42	1.1	0.45
Norway	0.04	0.35	0.9	0.38
Malta	0	0.02	0.05	0.02
Cyprus	0	0.01	0.02	0.01

Table A.4: Mean, median, 2.5 and 97.5 percentiles of the distribution of the number of life-years lost caused by historical excess emissions in Germany.

Country	**2.5 percentile**	**Median**	**97.5 percentile**	**Mean**
Germany	610	5600	14000	6000
Poland	210	1900	4700	2000
France	110	1000	2500	1100
Czech Republic	87	810	2000	860
Italy	67	630	1500	660
Austria	56	520	1300	550
Switzerland	49	460	1100	480
Hungary	37	360	910	390
UK	36	340	840	360
Romania	31	310	780	330
Netherlands	31	290	710	310
Belgium	28	280	700	300
Slovakia	21	190	460	200
Croatia	9.8	91	220	96
Lithuania	8.6	80	190	85
Bulgaria	7.7	70	170	73
Spain	6.7	62	150	65
Sweden	5.8	54	130	57
Denmark	5	53	140	59
Latvia	4.8	44	110	47
Slovenia	4.5	42	100	45
Greece	2	17	42	18
Luxembourg	1.2	11	27	12
Ireland	1.2	12	30	13
Estonia	1	8	20	8.9
Portugal	0.84	7.8	19	8.2
Finland	0.6	5	13	5.5
Norway	0.39	3.9	10	4.3
Malta	0.05	0.3	0.57	0.31
Cyprus	0	0	0	0.1

Table A.5: Mean, median, 2.5 and 97.5 percentiles of the distribution of premature mortalities caused by excess emissions released from 2016 onward in the absence of redress scenario.

Country	2.5 percentile	Median	97.5 percentile	Mean
Germany	130	1200	3000	1300
Poland	43	390	940	410
France	23	210	510	220
Czech Republic	19	180	420	190
Italy	14	130	320	140
Austria	13	120	280	120
Switzerland	11	100	250	110
Hungary	8	75	180	80
UK	8	75	190	83
Romania	7.2	66	160	70
Netherlands	6	62	160	68
Belgium	6	58	150	64
Slovakia	5	43	100	46
Croatia	2	21	50	22
Lithuania	2.1	19	45	20
Bulgaria	2	15	35	15
Spain	2	14	34	15
Sweden	1	12	29	13
Denmark	1	12	30	13
Latvia	1.1	10	24	10
Slovenia	1	9	22	10
Greece	0.4	4	9	4
Luxembourg	0	3	7	3
Ireland	0.3	3	6	3
Estonia	0	2	6	3
Portugal	0.2	2	4	2
Finland	0	1	3	1
Norway	0.1	1	2	1
Malta	0	0	0	0
Cyprus	0	0	0	0

Table A.6: Mean, median, 2.5 and 97.5 percentiles of the distribution of the number of life-years lost by excess emissions released from 2016 onward in the absence of redress scenario.

Country	2.5 percentile	Median	97.5 percentile	Mean
Germany	1500	14000	34000	15000
Poland	520	4800	11000	5100
France	290	2600	6300	2800
Czech Republic	210	2000	4700	2100
Italy	160	1500	3600	1600
Austria	140	1300	3200	1400
Switzerland	130	1200	2900	1300
Hungary	95	910	2300	1000
UK	88	800	1900	850
Romania	77	760	1900	830
Netherlands	73	700	1800	770
Belgium	73	670	1600	710
Slovakia	52	470	1100	500
Croatia	24	220	520	230
Lithuania	20	190	450	200
Bulgaria	19	170	410	180
Spain	16	140	330	150
Sweden	15	140	330	140
Denmark	13	130	340	150
Latvia	11	100	250	110
Slovenia	11	100	250	110
Greece	4.7	40	100	43
Luxembourg	3.2	31	78	34
Ireland	2.9	27	65	28
Estonia	2.5	23	56	25
Portugal	2	18	44	19
Finland	1.4	13	32	14
Norway	1.1	10	26	11
Malta	0.12	0.8	1.5	1
Cyprus	0	0	1	0.2

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