Combined health impact assessment of noise and air quality in urban agglomerations

An explorative study

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Danny Houthuijs, Frank de Leeuw

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Author affiliation:
Danny Houthuijs, Frank de Leeuw (RIVM, NL)

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ETC/ACM Technical Paper 2017/12
European Topic Centre on Air Pollution and Climate Change Mitigation
PO Box 1
3720 BA Bilthoven
The Netherlands
Phone +31 30 2748562
Fax +31 30 2744433
Email etcacm@rivm.nl
Website http://acm.eionet.europa.eu/

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Executive summary

Noise and air pollution are two (closely related) important environmental and social issues. Elevated levels of noise and air pollution are found in densely populated areas, like agglomerations. To give insight in the combined risks of noise and air pollution, we carried out a health assessment in urban areas in Europe that builds upon data reported by the EU Member States under the Environmental Noise Directive (END) (EU, 2002) and the Air Quality Directive (EU, 2008). An urban agglomeration was defined as an agglomeration that was included in the second round of noise mapping in 2012 according to the END. We included in the assessment road traffic noise, particulate matter, ozone and nitrogen dioxide.

The key messages from this report are:

- We were successful in ranking 497 urban agglomerations in Europe on their combined health impact of road traffic noise and air pollution.

- The combined health impact of road traffic noise and air pollution in agglomerations is, on average, 1,745 DALYs per year per 100.000 inhabitants. This corresponds with 6.2% of the total burden of disease for all causes per year.

- Particulate air pollution contributes, on average, for 45% to the total impact, followed by nitrogen dioxide with 33%. Road traffic noise is associated with 21% and ozone with 2% of the combined burden of disease by road traffic noise and air pollution.

- There is a 7 to 9 fold difference in the health impact between the highest and lowest ranked agglomeration. This difference indicate that there is ample room for improvement of the health of citizens in European agglomerations by policy measures aiming at further reduction of road traffic noise and air pollution.

- Several points of improvements were identified in the combined health impact assessment for road traffic noise and air pollution. Therefore the results of this report should be considered as explorative.

1 Introduction

It is well documented that the exposure to noise and air pollution may lead to adverse health effects. Noise exposure can lead to annoyance, sleep disturbance and increases the risk of hypertension and cardiovascular disease. Hypertension and cardiovascular disease are important risk factors for premature mortality; noise exposure can indirectly reduce life expectancy as well. Exposure to air pollution may lead to premature mortality and morbidity, mainly related to respiratory and cardiovascular diseases.

Transport, in particular road transport, and industry are important sources both for noise and air pollution. High exposure levels can be found in densely populated areas. Reporting under the Environmental Noise Directive (END) (EU, 2002) shows that 42 million urban residents are exposed to noise levels exceeding 55 dB L_{den} (Houthuijs et al., 2014). According to the EEA (2017) 82-85 % of the urban population in the EU-28 are exposed to $PM_{2.5}$ concentration exceeding the air quality guidelines (AQG) as recommended by the WHO (2006). However, the difference in the definition of “urban areas” as applied in both studies disables an estimation of the cumulative risk of noise and air pollution in urban regions.

The objective of this study is to assess the combined burden of disease attributable to the exposure to environmental noise and ambient air pollution in urban agglomerations in Europe. A second objective was to explore if urban agglomerations in Europe could be ranked based on the impact of the combined exposure.

In this report the definition of an urban agglomeration is according to the one given in the END. The reason to use the END definition as a starting point is that countries report on the aggregated noise exposure distribution which cannot be unravelled to assess the distribution for other areas. The health assessment builds upon data reported by the EU Member States under the Environmental Noise Directive (EU, 2002) and the Air Quality Directive (EU, 2008). Environmental noise is limited in this report to road traffic noise, since in the Noise in Europe 2014 report (EEA, 2014) it was identified that about 90% of the health impact of transport and industrial noise may be related to road traffic noise.

Chapter 2 discusses the methodology of the combined assessment. The sources of the exposure distribution for road traffic noise and air pollution, the exposure/concentration – response relations and additional data to quantify the health impacts for noise and air pollution are described. The concept of burden of disease that was used to integrate the health impact of road traffic noise and air pollution is introduced.

The results of the health impact assessment are given in Chapter 3. The contribution of the health risks to the overall burden of disease is given. Regions in Europe with a similar pattern for the combined exposure and for the combined health impact of road traffic noise and air pollution in agglomerations are identified. A summary of the health impact in European capitals and in agglomerations with more than 2 million inhabitants is presented. Also, the top 15 of the best and the worst agglomerations is given. The report ends with a discussion (Chapter 4) and conclusions (Chapter 5).

2 Data and methodology

2.1 Introduction

In section 2.2 the source of the exposure distribution and the relevant exposure response relations for road traffic noise are described. This is done for air quality in section 2.3. To be able to express the health impact of air pollution and road traffic noise in the same units, we expressed their burden of disease into disability-adjusted life years section (2.4). Chapter 2 ends with a section on the applied statistical methods.

2.2 Road traffic noise

2.2.1 Exposure

The noise exposure distribution in the agglomerations is based upon information from the EEA's member countries for 2012 obtained using noise modelling and measurement methods, and reported to the EEA up to 15 April 2016. The exposure data related to major roads and major railways in agglomerations is not complete (Blanes et al., 2016). Gap filling has been applied in order to estimate the noise exposure distribution in all agglomerations under study. The methodology is described in a working paper (ETC/ACM, 2015) based on previous work done by Extrium (2013).

Health and well-being risks due to noise are also present at noise levels below the lowest values of L_{den} (55 dB) and L_{night} (50 dB) that are commonly applied in the END assessments. The END results were extrapolated to lower noise levels using a statistical approach to estimate the complete population exposure distribution of road traffic noise during the 24 hour (L_{den}) and the night period (L_{night}). Also the noise exposure distribution was refined from 5 to 1 dB (Houthuijs et al., 2015).

2.2.2 Health effects

The health impact assessment was carried out with the exposure response relations that were used for the health impacts assessment reported in the Noise in Europe 2014 report (EEA, 2014). A summary of the relevant health outcomes and exposure-response relations for road traffic noise is given in Table 1.

Details of the method of the health impact assessment are described in Houthuijs et al. (2014). For this report, we updated some of the baseline data and we applied a slightly different calculation method for cardiovascular endpoints (for details, see Annex 1).

The exposure-response relations for severe annoyance and highly sleep disturbed only depend on the noise level (L_{den} and L_{night} respectively), so they can be applied directly when the noise level is known. Often the relations are applied as a polynomial with a forced starting point for the exposure-response functions (L_0 is Table 1) at 42 dB L_{den} or 40 dB L_{night}. In this report we applied the original derived exposure-response relations from the papers (Miedema and Oudshoorn, 2001; Miedema and Vos, 2007) which makes extrapolation of the exposure-response relation to lower noise levels than 42 dB L_{den} or 40 dB L_{night} possible.

Table 1. Summary of health outcome, exposure-response relation for road traffic noise.

Health outcome	Reference of exposure response relation	Baseline rate (a)	L_0 (dB)
(Severe) annoyance, age 18+	Miedema & Oudshoorn (2001)	Not applicable	0
(Higly) sleep disturbed, age 18+	Miedema & Vos (2007)	Not applicable	0
Hypertension, age 18+	Van Kempen & Babisch (2012)	WHO Global Health Observatory (WHO,2017a)	50
Coronary heart disease(incidence and mortality), all ages	Vinneau et al. (2013)	Global Health Estimates 2015 (WHO, 2016a) , European Cardiovascular Disease Statistics 2017 (Wilkens et al., 2017)	50
Cerebrovascular disease (incidence and mortality), all ages	Houthuijs et al., 2014	Global Health Estimates 2015 (WHO, 2016c) , European Cardiovascular Disease Statistics 2017 (Wilkens et al., 2017)	50

(a) The baseline rate refers to the prevalence of a health status or the incidence of a disease in the population

In the absence of noise, there is a baseline prevalence of hypertension and/or incidence of the cardiovascular endpoints present in the population. The contribution of the noise exposure to the prevalence of incidence depends on the exposure level as well as the baseline rate.

For hypertension new (baseline) data on the prevalence of raised blood pressure had become available. We used country specific data on the crude prevalence of raised blood pressure among males and females in 2015 in the population 18 years and older (WHO Global Health Observatory, WHO, 2017a).

For the cardiovascular endpoints coronary heart disease and stroke, the data on baseline mortality was updated to country-specific data using more recent data (Global Health Estimates 2015, WHO 2016d, 2016e). In the Noise in Europe 2014 report (EEA, 2014) hospital discharge data was used as indicator for the incidence of coronary heart disease and stroke. European wide incidence data has become recently available in the 2017 European Cardiovascular Disease Statistics (Wilkens et al., 2017) and is preferred above the hospital discharge data.

The calculation of the population attributable fraction (PAF) for cardiovascular endpoints was brought in line with the calculation for the air pollution endpoints (see also Annex 1). The PAF fraction is the relative reduction in cardiovascular disease or mortality in a population that would occur if the exposures to road traffic were reduced to an alternative exposure distribution (in this report to levels without an additional health risk).

Assuming a log-linear relationship between the noise exposure level (L) and relative risk (RR), the relative risk is given by:

$$RR = e^{-\beta\,\Delta L} \qquad [1]$$

where β is the exposure-response factor and $\Delta L = (L - L_0)$ is the difference between the actual noise exposure level L and a threshold level L_0 below which no additional health risk is expected.

Once the relative risks have been estimated, the fraction of the baseline incidence attributable to the given exposure difference (PAF) is:

$$\mathrm{PAF} = \frac{\sum_i (f_i \cdot RR_i) - 1}{\sum_i (f_i \cdot RR_i)} \qquad [2]$$

where the summation is over all the 1 dB noise exposure classes in an agglomeration, f_i is the fraction of the population in the exposure class.

For each agglomeration the expected burden of disease attributable to road traffic noise is given by:

$$E = PAF \cdot MR_a \cdot Pop_a \qquad [3]$$

where *E* is the expected burden of disease (e.g. number of new cases with coronary heart disease per year); MR_a is the baseline incidence of the given health effect (e.g. the total number of new cases of coronary heart disease per 100,000 people per year); Pop_a is the size of population in each agglomeration.

The health impact assessment for road traffic noise in this report is based on the combination of reported data, ‘gap filled’ data and the extrapolation to lower noise levels.

2.3 Air quality

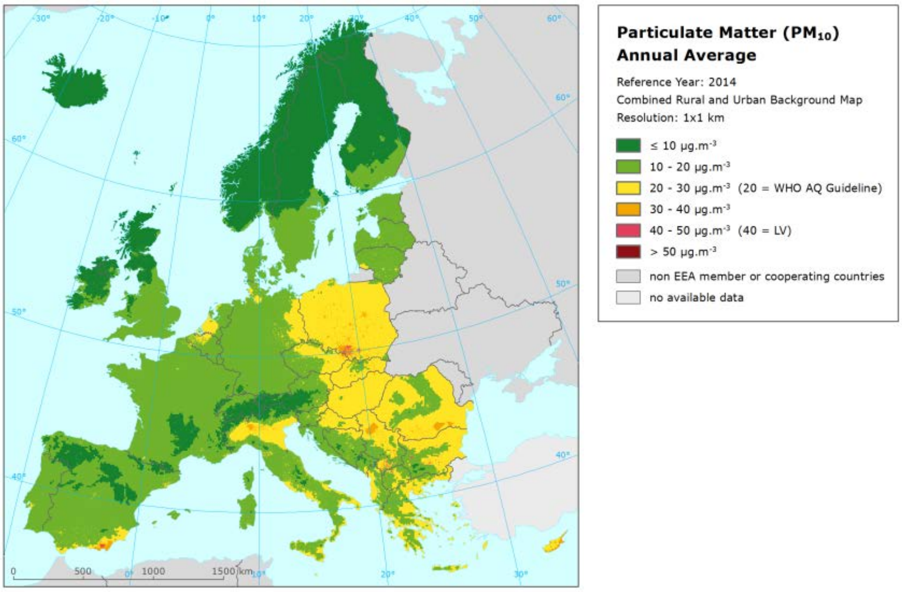
2.3.1 Exposure

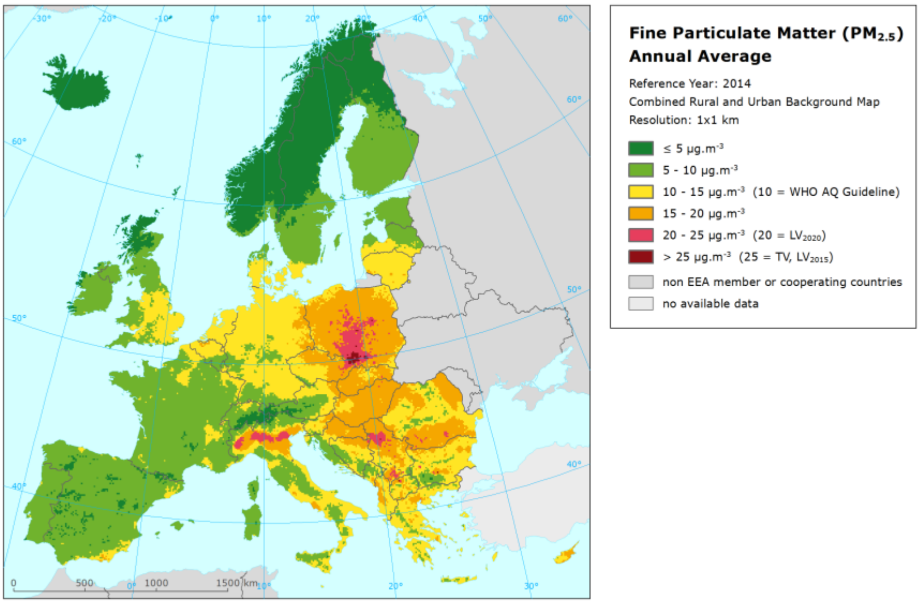
Air quality concentration maps (annual mean concentration for $PM_{2.5}$, PM_{10} and NO2; SOMO35 for ozone) prepared by the European Topic Centre on Air and Climate Change (ETC/ACM, Horálek et al. (2016, 2017).) have been used as input to the health impact assessment.

The maps are based on the interpolation of annual statistics of monitoring data from 2014, reported by EEA member and cooperating countries. The mapping method is based primarily on air quality measurements. It combines concentrations observed at background stations with additional spatial information (such as the results from chemical transport models, meteorological data, altitude, population density map). The method is a linear regression model followed by kriging of the residuals produced from that model (residual kriging). A computational spatial resolution of 1*1 km is used; this fine resolution enables to account for the smaller urbanisations.

To increase the spatial coverage of the $PM_{2.5}$ measurements, so-called pseudo $PM_{2.5}$ stations data were used in addition to measured $PM_{2.5}$ data. Pseudo $PM_{2.5}$ stations data are estimated using PM_{10} measured data, surface solar radiation, latitude and longitude.
The mapping of NO_2 concentrations includes measurement data from traffic stations in combination with land cover data and road data. Details of the mapping procedures and the data used are described by Horálek et al. (2016, 2017).

Annual concentrations (Figure 1) are used for $PM_{2.5}$, PM_{10} and NO_2; for ozone the SOMO35, that is, the sum of maximum daily 8-hour means over 35 ppb (70 µg/m^3) is used.





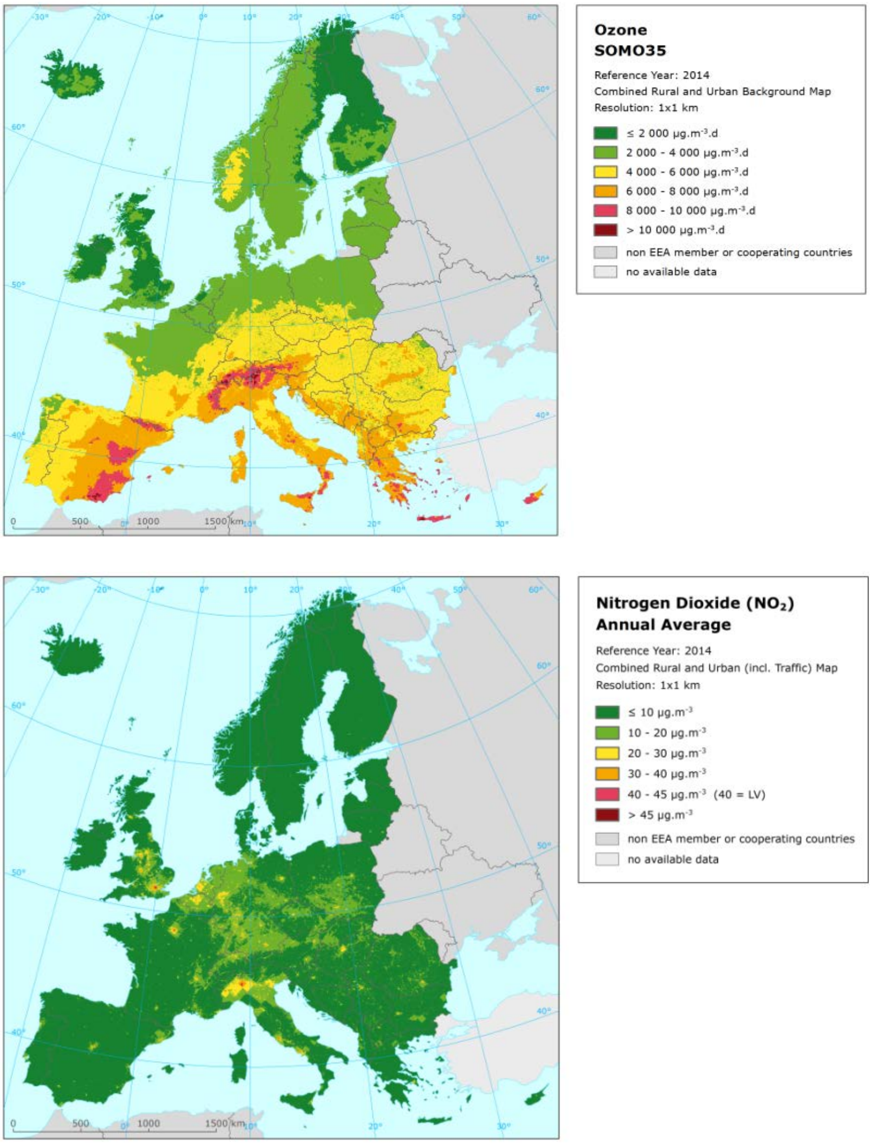


Figure 1. Concentration maps of $PM_{2.5}$ (annual mean) , PM_{10} (annual mean), NO_2 (annual mean) and ozone (SOMO35), reference year 2014. Source: Horálek et al. (2016, 2017).

2.3.2 Health effects

For the selection of the health endpoints and concentration-response functions, the recommendations of the HRAPIE-project (WHO, 2013) were followed. From the concentration-response function discussed in this report we selected those endpoints which have been used in the cost-benefit analysis of the EU Clean Air Package (EC, 2013; Holland, 2014). A summary is given in Table 2.

Table 2. Summary of health outcome, concentration-response relation for air pollution.

Pollutant	Health outcome	RR (95% CI) per 10 µg/m^3	Baseline (a)	C_0 (µg/m^3)
$PM_{2.5}$	Mortality, all cause (natural); age 30+	1.062 (1.040- 1.083)	European mortality database (WHO,2016b)	2.5
PM_{10}	Postneonatal infant mortality, all causes age 1-12 months	1.04 (1.02 – 1.07)	European Health for all database (WHO, 2016c)	3.8
O_3	Mortality, all cause (natural) all ages	1.0029 (1.0014 – 1.0043)	European mortality database (WHO,2016b)	70
NO_2	Mortality, all cause (natural); age 30+	1.039 (1.022-1.056)	European mortality database (WHO,2016b)	10
$PM_{2.5}$	cardiovascular hospital admissions, all ages	1.0091 (1.0017-1.-166	Eurostat (2017a,b)	2.5
$PM_{2.5}$	Respiratory hospital admissions, all ages	1.019 (0.9982 -1.0402)	Eurostat (2017a,b)	2.5
$PM_{2.5}$	Restricted activity days (RAD) all ages	1.047 (1.042-1.053)	HRAPIE project(WHO,2013)	2.5
$PM_{2.5}$	Work days lost; age 20-64	1.046 (1.039 – 1.053)	European Health for all database (WHO, 2016c). Eurostat(2017c)	2.5
PM_{10}	Prevalence of bronchitis in children (age 6-12)	1.08 (0.98 – 1.19)	HRAPIE project(WHO,2013)	3.8
PM_{10}	Incidence of chronic bronchitis in adults (age 18+)	1.117 (1.040 – 1.189)	HRAPIE project(WHO,2013)	3.8
PM_{10}	Incidence of asthma symptoms in asthmatic children, aged 5-19	1.04 (1.02-1.07)	European Health for all database (WHO, 2016c).	3.8
O_3	Respiratory hospital admissions, age 65+	1.0044 (1.0007- 1.0083)	Eurostat (2017a,b)	70
O_3	cardiovascular hospital admissions (excluding stroke), age 65+,	1.0089 (1.005 – 1.0127)	Eurostat (2017a,b)	70
O_3	Minor restricted activity days (MRAD)	1.0154 (1.006 – 1.0249)	HRAPIE project(WHO,2013)	70

(a) see main text and de Leeuw and Horálek (2017) for further details.

The estimated health impact is sensitive for the assumption of the counterfactual concentration (C_0 in Table 2). For $PM_{2.5}$ we have adopted a value of 2.5 µg/m^3. In the health impact assessments made for the Clean Air Package (EC, 2013), impacts are estimated for the (modelled) anthropogenic contribution to $PM_{2.5}$ which implies that a (natural) background contribution is not considered. In the Global Burden of Disease 2013-study (Burnett et al., 2014), impacts are estimated only above a counterfactual concentration of 5.8 to 8.8 µg/m^3. In the GBD-2015 study (GBD 2015 risk factor collaborators, 2016) the range was lowered to 2.4 – 5.9 µg/m^3. A C_0- value of 2.5 µg/m^3 corresponds to the lower boundaries of the updated GBD range. This level of 2.5 µg/m^3 also corresponds to the lowest value found in populated areas in the interpolated concentration map and is therefore seen as a realistic estimate of the European background concentration. Given a European averaged $PM_{2.5}/PM_{10}$ ratio of 0.65 (de Leeuw and Horálek, 2009), the PM_{10} counterfactual concentrations was set to 3.8 µg/m^3.

Restricted activity days (RAD) includes days when people reduce their normal activities, including absenteeism from work or school. The effects of PM on work days lost, hospitalization, asthmatic symptoms and the prevalence of bronchitis in children are estimated according to specific concentration-response function (Table 2). To avoid double counting the total number of RAD has been corrected for these more specific outcomes.

The ozone exposure estimates are based on SOMO35 levels. SOMO35 is defined as the yearly sum of excess of daily maximum 8-h running average over a cut-off of 70 µg/m^3 (35 ppb).

To estimate the effects of long-term NO_2 exposure on all-cause mortality, a relative risk of 1.055 (95% CI 1.031 to 1.08) has been recommended (WHO, 2013); the NO_2 impact has to be calculated for levels above 20 µg/m^3. The material as available in the HRAPIE review did not suggest that there is no effect below 20 µg/m^3; the studies rather showed that the size of the effect is less certain below 20 µg/m^3. However, this recommendation might be too conservative, as indicated by Héroux et al. (2015). A Danish study (Raaschou-Nielsen et al., 2012) showed a significant correlation between NO_2 concentrations and health outcomes throughout the full range of observed concentrations (10.5 to 59.6 µg/m^3 with a median of 15.1 µg/m^3). Also in a large Dutch cohort study a linear relation between NO_2 and mortality risk was seen for concentrations between 10 and 20 µg/m^3 (Fischer et al., 2015).

It should be noted that, as concentrations are (sometimes strongly) correlated, the impacts estimated for $PM_{2.5}$ and NO_2 may not be added to determine the total impact attributable to exposure to air pollution. This may lead to a double counting of up to 30 % of the effects of $PM_{2.5}$ and NO_2 (WHO, 2013). To account for this overlap, in this paper a relative risk downward adjusted by 30% (1.039 (95% CI 1.022- 1.056) is used in combination with a counterfactual concentration of 10 µg/m^3.

Assuming a log-linear relationship between air pollutant concentration (C) and relative risk (RR), the relative risk is given by:

$$\mathrm{RR} = \mathrm{e}^{-\beta\,\Delta \mathrm{C}} \qquad [4]$$

where β is the concentration-response factor and ΔC= ($C - C_0$) is the difference between the actual concentration and the above described the counterfactual concentration.

Once the relative risks have been estimated, the fraction of the baseline incidence attributable to the given concentration difference (population attributable factor, PAF) is:

$$\mathrm{PAF} = \frac{\sum_i (f_i \cdot RR_i) - 1}{\sum_i (f_i \cdot RR_i)} \qquad [5]$$

where the summation is over all the 1x1 km grid cells in an agglomeration, f_i is the fraction of the population in the grid cell.

For each agglomeration the expected burden of disease attributable to air pollution is given by:

$$E = PAF \cdot MR_a \cdot Pop_a \qquad [6]$$

where E is the expected burden of disease (e.g. number of premature deaths or number of hospital admissions due to outdoor air pollution); MR_a is the baseline incidence of the given health effect (e.g. the total or cause specific number of deaths per 100,000 people per year); Pop_a is the size of population in each agglomeration.

The number of years of life lost due to premature mortality is calculated according to:

$$\mathrm{YLL} = \sum(\mathrm{E_i} \cdot \mathrm{L_i}) \qquad [7]$$

where E_i is the number of deaths in age class i attributable to air pollution and L_i is the life expectancy at age of death (in years). The summation is done for each 5-year age class

2.4 Disability-adjusted life years

2.4.1 Introduction

The disability-adjusted life year (DALY) is a measure of overall disease burden, expressed as the number of years lost due to ill-health, disability or early death. The DALY was developed by the World Bank as a way of comparing the overall health and life expectancy of different countries.
DALYs for a disease or health outcome are calculated as the sum of the years of life lost due to premature mortality (YLL) and the years lost due to disability (YLD) for morbidity outcome:

$$\mathrm{DALY} = \mathrm{YLL} + \mathrm{YLD} \qquad [8]$$

To estimate YLD for a particular health outcome the number of attributable cases is multiplied by the average duration of the disease and a weight factor that reflects the severity of the disease on a scale from 0 (perfect health) to 1 (death) ((WHO, 2017b):

$$\mathrm{YLD} = \mathrm{E} \cdot \mathrm{DW} \cdot \mathrm{L} \qquad [9]$$

Where E is the number of attributable cases, DW is a disability weight and L is the average duration.

An alternative for the calculation is to multiple the population attributable fraction of a risk factor with the estimated total YLLs and YLDs for the specific disease.

2.4.2 Road traffic noise

Highly sleep disturbed and cardiovascular morbidity and mortality were taken into account as health endpoints related to road traffic noise in the calculation of DALYs. Annoyance is a clear effect of noise, but was not recommended by a working group of WHO (2012) as an endpoint to be quantify in burden of disease calculations for environmental noise. We did not quantify the impact of raised blood pressure. We consider raised blood pressure as an important risk factor for cardiovascular disease and mortality. There is a large probability of overlap in road traffic noise attributable burden of disease when DALYs for raised blood pressure as well as cardiovascular endpoints are calculated separately: adding them together will lead to an overestimation.

The cases of highly sleep disturbed among adults (duration one year) were multiplied with a disability weight of 0.06 to obtain the YLD. WHO (2011) derived a mean weight of 0.07 from 4 studies. We looked again into the original studies, since the distribution of the DWs is rather skewed. Van Kempen (1998) reported a mean disability weight of 0.10 for severe sleep disturbance due to noise, based on a pilot study in the Netherlands among 13 medical experts. De Hollander (2004) expanded Van Kempen’s study to 35 public health professionals and assessed a median disability weight of 0.08 (mean: 0.10; range 0-0.45). Müller-Wenk (2002) assessed a median disability of 0.04 (mean: 0.055; range: 0.02-0.31) for those highly sleep-disturbed by night-time road noise, in on a survey of 42 Swiss physicians. In a later study of Knoblauch and Müller-Wenk (2005) the mean judgement of 14 general practitioners was that noise-related sleep disturbance has a mean severity of 0.9 times the severity of primary insomnia (range: 0-2.1), which resulted in a disability weight of 0.09. Like in the other studies, the distribution was rather skewed; the median severity ratio was 0.63 which corresponds with to a median DW of 0.06. Given the rather skewed distributions of the reported disability weights, we used the median (0.06) of the medians of De Hollander (0.08), Müller-Wenk (0.04) and Knoblauch & Müller-Wenk (0.06).

Previous used (Houthuijs et al., 2014) information on YLLs and YLDs for coronary heart disease and stroke for the Worldbank/WHO regions “high-income countries” and “European Region (low- and middle-income countries)” were updated to estimates of the burden of disease by cause per WHO member state for 2015 using WHO Global Health Estimates (WHO, 2016d, 2016e). We extracted DALYs for all causes and YLLs and YLDs for ischaemic heart disease (similar to coronary heart disease) and stroke (similar to cerebrovascular disease). The population attributable fraction related to the noise exposure (Eq [2]) was used to calculate the health impact for coronary heart and cerebrovascular disease in YLLs and YLDs per agglomeration.

The DALY for all causes was used to express the road traffic noise (and air quality) impact as percentage of the total burden of disease.

2.4.3 Air quality

Table 3 summarises the disability weights and durations of the selected health outcomes that were used for the calculation of the DALYs related to air quality. The disability weight and duration for the calculation of YLD according to Eq[9] are based on the results reported by Bachmann and van der Kamp (2017), HEIMSTA/INTARESE (2011) and WHO (2017b).

The YLL due to $PM_{2.5}$, O_3 and NO_2 corresponds to the number of deaths multiplied by the life expectancy at the age of death (Eq [7]).

Table 3. Health outcome and corresponding disability weights and durations

Health outcome	unit	Disability weight	Duration (year)	reference
Mortality due to $PM_{2.5}$, O_3 , NO_2	Years of life lost (YLL)	1	1	(a)
Postneonatal infant mortality, all causes,age 1-12 months; PM_{10}	cases	1	80	(a)
cardiovascular hospital admissions, PM_{10}	cases	0.588	0.038	(a)
cardiovascular hospital admissions, ozone	cases	0.179	0.038	(a)
Respiratory hospital admissions, PM_{10}, O_3	cases	0.408	0.038	(a)
Restricted activity days (RAD) all ages	days	0.099	0.0027	(b)
Work days lost; age 20-64	days	0.099	0.0027	(b)
Prevalence of bronchitis in children (age 6-12)	days	0.225	0.0027	(c)
Incidence of chronic bronchitis in adults (age 18+)	cases	0.099	10	(b)
Incidence of asthma symptoms in asthmatic children, aged 5-19	days	0.099	0.0027	(b)
Minor restricted activity days (MRAD)	days	0.07	0.0027	(b)

(a) Bachmann and van der Kamp, 2017
(b) HEIMSTA/INTARESE, 2011.
(c) WHO, 2017b.

In the health impact assessment the computations are made on the 1x1 km grid; in this way the known correlation between air pollution levels and population density is (partly) taken into account. It also allows for the introduction of a threshold value, e.g. an assumed threshold value below which no health impacts are to be expected or a “correction” for natural contributions.

2.5 Statistical analysis

A cluster analysis has been applied in order to groups the agglomerations with similar characteristics. The cluster analysis was performed on air pollution and noise levels as well and on the health impacts (expressed in DALYs per 100,000 inhabitants or as percentage of the total burden of disease) attributable to noise exposure and to each of the air pollutants components (PM, O_3 and NO_2). Before clustering variables were normalized (mean=0, standard deviation = 1). The selection of the number of clusters was based on the majority rule from a set of 30 indices. Clustering is performed using the *k*-means algorithm; all calculations were made using the NbClust-package in R (Charrad et al., 2014).

We explored whether the results have sufficient quality to rank the impact in the agglomerations. We limited our exploration to the influence of the applied exposure response relations.

These relations have statistical uncertainty, expressed in the 95% confidence intervals of the relations. If we ignore this statistical uncertainty, the total impact has no uncertainty and the ranking of agglomerations is unquestionable. If we include uncertainty in the calculations, the ranking may change, depending on the contribution of the various health endpoints and the uncertainty of the exposure response relations. A health endpoint with a large uncertain exposure response relation might hardly affect the ranking if the average contribution to the total burden is very low. Vice versa, an endpoint with a relative small uncertainty could influence the rank substantially if the contribution of this endpoint to the overall impact is large.
We calculated the standard deviation of the impact of each of the separate health endpoints per agglomeration (expressed in DALYs per 100.000 inhabitants). Subsequently, we simulated 100 times the uncertainty in the calculation of the total impact per agglomeration. Per simulation, we took a different set of random values from the normal probability distribution for the various health endpoints and calculated in combination with their standard deviation a combined impact per agglomeration. Subsequently, we calculated the reliability of the ranking (rankability) over the 100 simulated datasets by relating heterogeneity to uncertainty within (σ^2) and between agglomerations (τ^2) (Van Dishoek et al., 2011).

$$\text{Rankability} = \tau 2/(\tau 2 + \text{median}(\sigma 2)) \qquad [10]$$

Lingsma et al. (2010) suggested that any ranking is meaningless when rankability is low (<0.50). Simple integer ranks are appropriate when the rankabilty is high (>0.75).

3 Results

3.1 Assessment in 479 agglomerations

The total END-dataset contains 482 agglomerations. Three agglomerations, all located on the Canarias, could not be included due to missing air pollution data, so the assessment was carried out for 479 agglomerations. The total number of inhabitants is 177.6 million.

3.2 Exposure to road traffic noise and air quality

The quality indicators (annual mean of $PM_{2.5}$, PM_{10} and NO_2, O_3-SOMO35 , L_{den} and L_{night}) do not show a strong correlation except – as expected – between $PM_{2.5}/PM_{10}$ (r = 0.94) and L_{den}/L_{night} (r = 0.94). Although road traffic is an important source for NO_2 and $PM_{2.5}$, the population weighted concentrations show no correlation with the road traffic related L_{den} and L_{night} levels. At the urban scale the traffic emissions gives only a relatively small contribution to the total concentration; main contributors are domestic heating, industry and from sources outside the urban area (Kiesewetter and Amann, 2014). At street level, noise and air pollution levels are highly correlated (Can et al., 2011; Morelli et al., 2015; Kim et al., 2012). However, the END reporting mechanism gives a noise exposure distribution for the whole agglomeration. The variability within a city and the possible increase in environmental risk due to the co-occurrence of high noise and air pollution levels can therefore not be considered.

A summary of the population weighted averaged air pollutant concentrations and noise levels is given in Figure 2. In 26 agglomerations with in total 9.1 million residents, the population weighted $PM_{2.5}$ concentration exceeds the limit value of 25 µg/m^3; 21.9 M residents are exposed to averaged concentrations below the WHO AQG. For PM_{10} it is estimated that 4.5 M residents are exposed to concentration above the limit value of 40 µg/m^3, and 105.9 M to concentrations below the WHO-AQG of 20 µg/m^3.

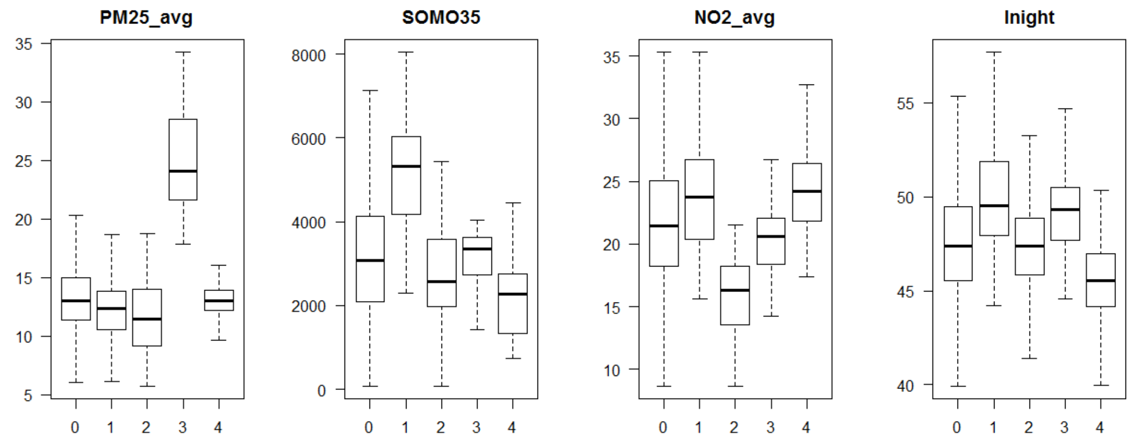


Figure 2. Percentiles (5-25-50-75-95) of population weighted concentrations (annual mean data for $PM_{2.5}$, PM_{10}, NO_2, SOMO35 data for O_3) and noise levels (L_{night}) for the total set of 479 agglomerations (label = 0) and for the each of the four clusters (see section 3.3).

3.3 Clusters of agglomerations based on environmental quality

A cluster analysis was performed on the levels of $PM_{2.5}$, NO_2, O_3 and L_{night}. PM_{10} and L_{den} were not included as they are highly correlated with $PM_{2.5}$ and L_{night}, respectively. The dataset was separated into four clusters (Figure 3).

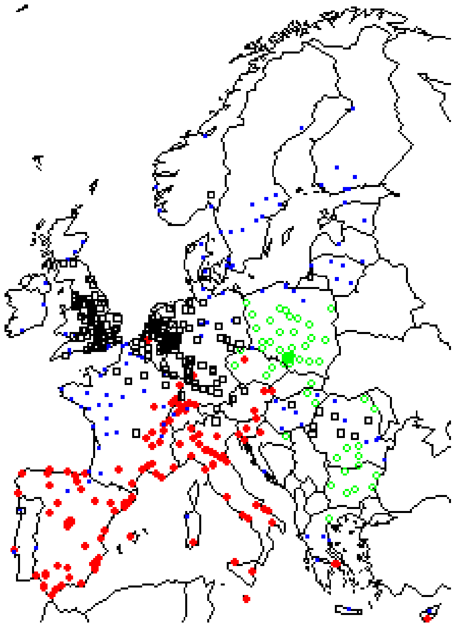


Figure 3. Map of agglomerations clustered in four group, based on environmental quality (cluster codes 1 to 4: red, blue, green, black)

The first cluster (the red dots in Figure 3) is characterized by high ozone, NO_2 and noise levels (see Figure 2 in section 3.2), the agglomerations are mainly located in Spain, Italy and the eastern and south-eastern part of France (Figure 3).
Cluster 2 (blue dots) is mainly located in west and northwest Europe along the coasts of the Atlantic Ocean, North Sea and Baltic Sea. Cluster 2 has the best air quality, in particular low NO_2 concentrations and moderated noise levels. The averaged population density for cluster 2 is the lowest among the four clusters.
Cluster 3 (green dots), mainly located in the eastern Member States, has by far the highest $PM_{2.5}$ concentrations together with high noise levels.
Cluster 4 (black dots) is mainly located in Germany, Benelux-area and England; NO_2 concentrations are relatively high; the levels of the other air quality and noise indicators are moderate to low.

3.4 Burden of disease due to road traffic noise and air quality

For all 497 agglomerations, the combined burden of disease due to air quality and road traffic noise is 3.10 million DALYs per year which corresponds with 1,745 DALYs per year per 100,000 inhabitants and to 6.2% of the total burden of disease (total number of DALYs per year per 100.000 inhabitants due to all causes).

In Figure 4, the contribution of the air quality components and road traffic noise to the combined burden of disease given (left hand pie of Figure 4); the contribution is also split

into YLL and YLD for road traffic noise and the combined air quality components (right hand pie).
Road traffic noise exposure contributes 21% to this combined burden of disease; exposure to particulate matter accounts for 45% (left hand pie in Figure 4).

From the right hand pie of Figure 4 it appears that for the combined air pollution components the effects on mortality (YLL) are most important whereas road traffic noise exposure contributes for the most part to the years lost due to disability (YLD). The population that is highly sleep disturbed due to road traffic noise is responsible for the large contribution to YLD; exposure to levels below 50 dB L_{night} give a substantial contribution (32%) to the total burden of disease due to road traffic noise (see Figure 5).

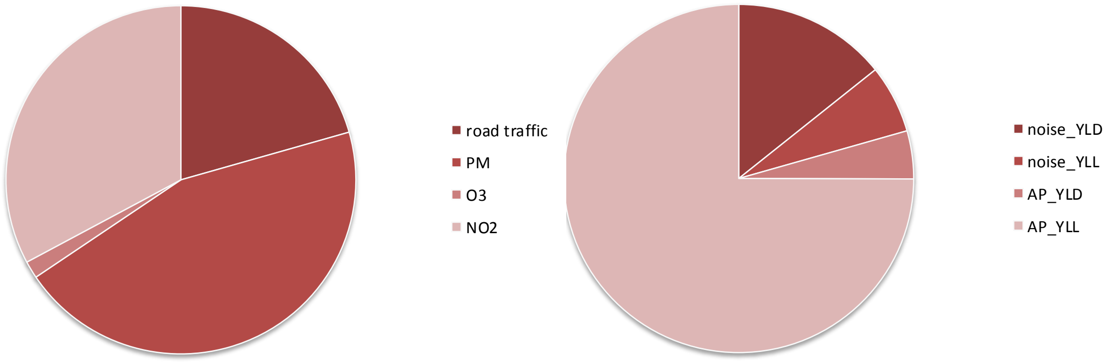


Figure 4. Fractions of burden of disease related to air quality components and road traffic noise (left) and to years of life lost (YLL) and years lost due to disability (YLD) for road traffic noise and the combined air pollution components; totals for 479 agglomerations.

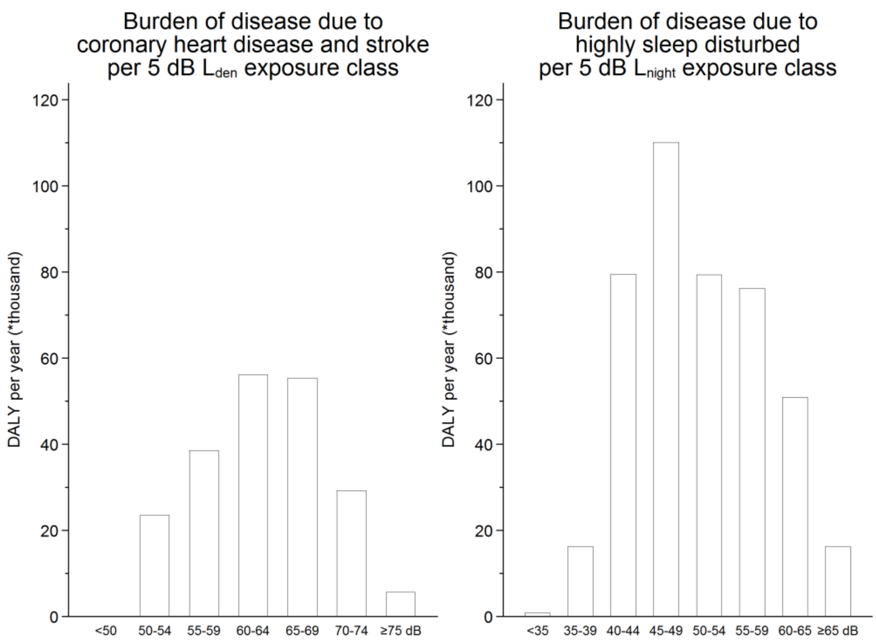


Figure 5. Burden of disease related to 24 hour (L_{den} left) and night time (L_{night} right) road traffic noise levels per 5 dB exposure class.

3.5 *Clusters of agglomerations based on burden of disease per 100,000 inhabitants*

A cluster analysis performed on the DALYs (expressed per 100,000 inhabitants) attributable to PM, O_3, NO_2, and road traffic noise, resulted in three distinctive clusters (Figure 6).

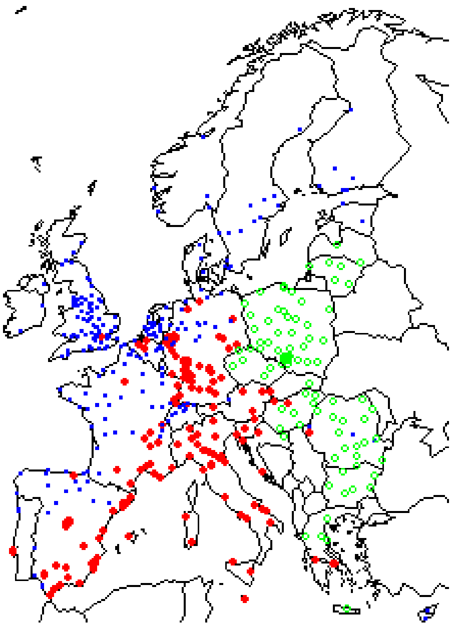


Figure 6. Map of agglomerations clustered in three groups, based on combined burden of disease per 100,000 inhabitants from road traffic noise and air quality (cluster codes 1 to 3: red, blue, green)

The distribution of the contribution of PM, O_3, NO_2, and road traffic noise of these three clusters to the burden of disease expressed as DALYs per 100,000 inhabitants is shown in Figure 7.

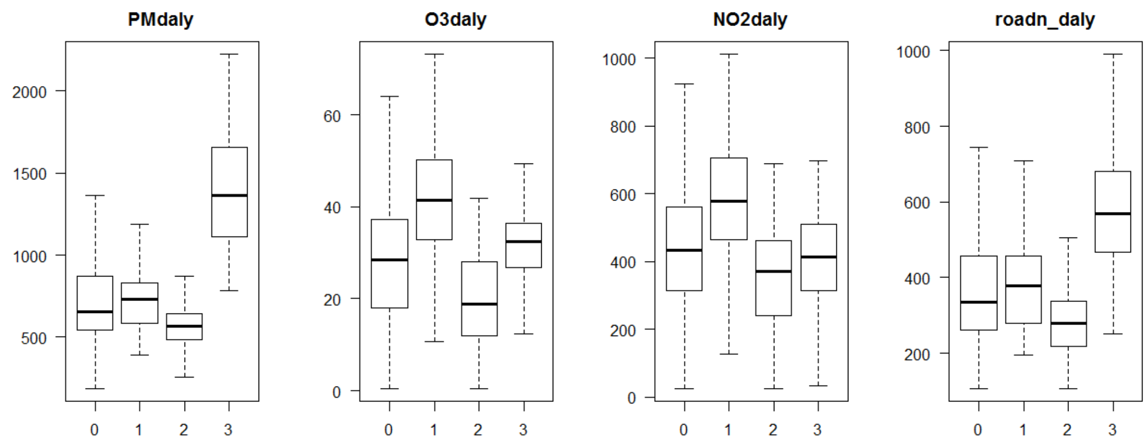


Figure 7. Percentiles (5-25-50-75-95) of DALYs per 100,000 inhabitants for PM, O_3, NO_2 and road traffic noise for the total set of 479 agglomerations (label = 0) and for the each of the three clusters.

Relative low contributions for all four environmental exposures are found in the (extended) coastal regions of Atlantic and North Sea and in the Baltic countries (cluster 2, blue dots in Figure 5).
The largest contributions from road traffic noise and PM are observed in the eastern part of Europe (cluster 3, green dots in Figure 6).
In the central and southern part of Europe a larger contribution from O_3 and NO_2 is seen (cluster 1, red dots in Figure 6)

The health impact of air quality and road traffic noise is unequally distributed in Europe (see Table 4). In cluster 2, the average environmental burden is almost1,300 DALYs per 100,000 inhabitants. 38% of the total population in the 479 agglomerations lives in cities of cluster 2 where 28% of the attributable burden of disease can be found.
In cluster 3, the health impact of air quality and road traffic noise per capita is twice as high: on average, 2,650 DALYs per 100,000 inhabitants. 21% of the combined burden of disease precipitates among 14% of the population.

Table 4. Burden of disease for PM, O_3, NO_2, road traffic noise and combined expressed as DALYs per 100,000 inhabitants for three clusters

cluster	Burden of disease expressed as DALYs per 100,000 inhabitants					Population (*1,000)
	PM	O_3	NO_2	Road traffic noise	Total	
1	753	36	723	344	1,8567	84,945
2	565	18	410	280	1,273	67,784
3	1,490	29	503	626	2,649	24,910
Total	785	28	573	359	1,745	177,640

3.6 Clusters of agglomerations based on burden of disease as percentage of total burden of disease

We also expressed the attributable burden of disease PM, O_3, NO_2, and road traffic noise as percentage of the total burden of disease, since there is a large variation in total disease within Europe. Again a cluster analysis was carried out; this led to four distinctive clusters (Figure 8).

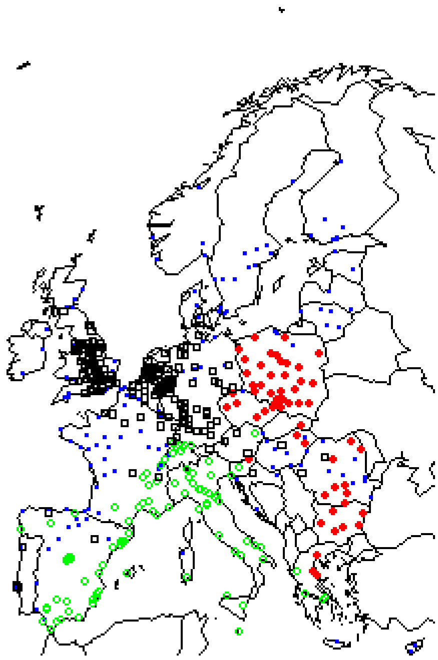


Figure 8. Map of agglomerations clustered in four group, based on combined burden of disease from road traffic noise and air quality expressed as percentage of the total burden of disease (cluster codes 1 to 4: red, blue, green, black)

The first cluster (the red dots in Figure 8) is characterized by a high contribution of PM to the total burden of disease (see Figure 9). Also road traffic noise adds to the burden of disease. The agglomerations are mainly located in the eastern Member States. The cluster corresponds with cluster 4 in Figure 3 based on environmental quality. The average contribution of the combined exposure to the total burden of disease is 8.3%: three quarter of the burden is due to air pollution (see Table 5).

Cluster 2 (blue dots) corresponds with cluster 2 in Figure 3 and has the best air quality, in particular low NO_2 concentrations and moderated noise levels. It is mainly located in west and northwest Europe along the coasts of the Atlantic Ocean, North Sea and Baltic Sea. The environmental burden of disease in cluster 2 (4.4%) is almost half of cluster 1. Road traffic noise is responsible for about 30% of the burden in cluster 2 (see Table 5).

Cluster 3 (green dots) is characterized by a relative large contribution of NO_2 and road traffic noise to the total burden of disease. The cluster corresponds with cluster 1 in Figure 3. The agglomerations are mainly located in Spain, Italy and the eastern and south-eastern part of France (Figure 8). Although high ozone concentrations are a common characteristic in this cluster, the contribution of ozone to the combined burden of 7.1% is limited (<3%).

Cluster 4 (black dots) is mainly located in Germany, Benelux-area and England. NO_2 has a relative large contribution (40%) to the total burden of disease of 6.0%. The burden of disease due to the other pollutants is moderate to low in comparison to the other clusters. Almost half of the population in the 479 agglomerations lives in this cluster.

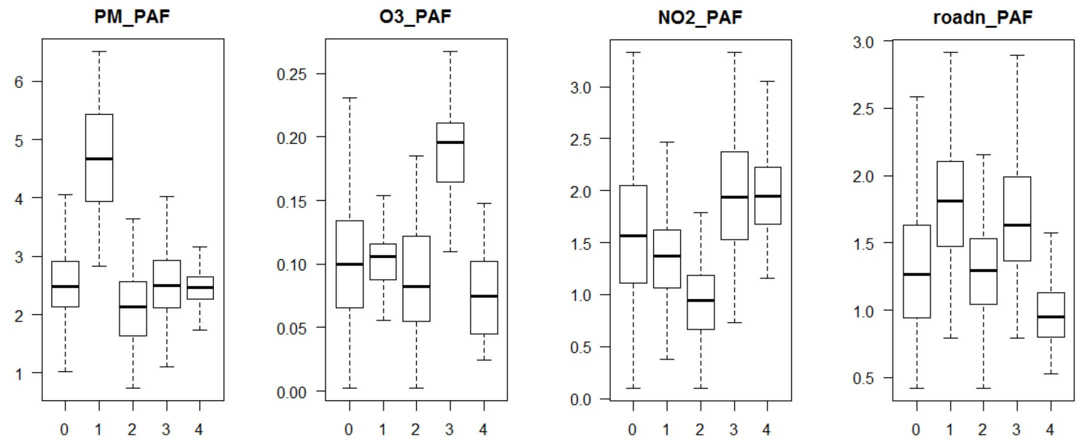


Figure 9. Percentiles (5-25-50-75-95) of attributable DALYs as percentage of total burden of disease for PM, O_3, NO_2 and road traffic noise for the total set of 479 agglomerations (label = 0) and for the each of the four clusters.

The clusters based on the environmental burden of disease expressed as percentage of the total burden of disease are almost identical to the clusters based on environmental quality.

Table 5. Burden of disease for PM, O_3, NO_2, road traffic noise and combined expressed as percentage of the total burden of disease for four clusters

cluster	Burden of disease expressed as percentage of the total burden of disease					Population (*1,000)
	PM	O_3	NO_2	Road traffic noise	Total	
1	4.73	0.09	1.59	1.84	8.25	19,475
2	2.07	0.08	0.96	1.30	4.41	32,128
3	2.75	0.18	2.58	1.60	7.12	40,375
4	2.54	0.07	2.39	0.98	5.99	85,663
Total	2.74	0.10	2.08	1.27	6.20	177,640

3.7 Combined burden of disease per agglomeration

In Table 6, a summary of the health impact assessment of road traffic noise and air quality is given for capitals in Europe and for agglomerations with more than 2 million inhabitants. Data for all 479 agglomerations is available in an excel spreadsheet.

Table 6. Summary of health impact assessment for road traffic noise and air quality in captials in Europe and agglomerations with more than 2 million inhabitants (sorted by country).

Agglomeration	**Country**	**Population Size**	**Road traffic noise**		**Air quality**		**Combined road traffic noise and air quality**		
			YLL/yr	**YLD/yr**	**YLL/yr**	**YLD/yr**	**Total DALY/yr**	**DALY/yr per 100,000 inhabitants**	**% of total disease burden**
All 479 agglomerations	-	177,640,000	442,101	196,044	138,275	2,323,709	3,100,130	1,745	6,2
Wien	AT	1,741,330	1,915	4,476	23,389	1,419	31,199	1,792	6.7
Brussels	BE	999,896	752	2,386	14,973	784	18,894	1,890	6.9
Sofia	BG	1,377,532	8,557	5,367	40,576	2,231	56,732	4,118	10
Bern	CH	350,791	290	915	3,145	226	4,576	1,305	5.6
Nicosia	CY	204,029	201	527	1,296	216	2,239	1,097	5.4
Praha agglomeration	CZ	1,234,005	3,506	3,917	18,427	1,228	27,078	2,194	7.1
Berlin	DE	3,331,248	1,850	6,739	54,160	3,193	65,941	1,979	6.6
Copenhagen	DK	1,163,001	921	3,197	10,872	745	15,734	1,353	5.1
Tallinn	EE	401,142	1,328	1,287	2,556	194	5,365	1,337	3.9
Madrid	ES	3,269,860	2,184	8,561	39,870	2,148	52,764	1,614	6.6
Helsinki	FI	570,577	711	1,469	3,470	261	5,911	1,036	3.7
Paris	FR	9,665,898	3,193	19,133	140,868	6,677	169,870	1,757	6.9
Athens	GR	745,517	1,187	1,813	14,019	761	17,780	2,385	8.3
Zagreb	HR	790,015	1,570	1,942	13,866	755	18,133	2,295	6.6
Budapest agglomeration	HU	2,158,869	4,146	5,174	46,142	2,261	57,723	2,674	7.2
Dublin	IE	1,273,101	1,068	3,378	4,927	585	9,957	782	3.6
The Great Reykjavik Area	IS	191,999	80	388	526	67	1,061	552	2.8
Roma	IT	2,546,802	1,161	5,831	50,130	2,542	59,664	2,343	8.6
Vilnius	LT	554,097	2,066	1,427	8,018	488	11,999	2,165	5.2
Luxembourg	LU	133,962	132	390	1,068	86	1,676	1,251	5.7
Riga agglomeration	LV	659,416	1,868	1,664	10,235	575	14,343	2,175	5.1

Table 6 (Continued). Summary of health impact assessment for road traffic noise and air quality in captials in Europe and agglomerations with more than 2 million inhabitants.

Agglomeration	**Country**	**Population Size**	**Road traffic noise**		**Air quality**		**Combined road traffic noise and air quality**		
			YLL/yr	**YLD/yr**	**YLL/yr**	**YLD/yr**	**Total DALY/yr**	**DALY/yr per 100,000 inhabitants**	**% of total disease burden**
Greater Valletta (Zabbar to Naxxar)	MT	270,004	191	556	1,947	254	2,949	1,092	4.4
Amsterdam / Haarlem	NL	1,615,332	995	3,673	19,282	1,230	25,180	1,559	6.1
Oslo	NO	906,319	730	2,549	5,938	461	9,678	1,068	4.6
Warszawa	PL	1,714,447	3,266	4,762	35,633	2,317	45,978	2,682	8.4
Lisboa	PT	547,737	409	1,324	6,038	361	8,132	1,485	5.2
Bucharest	RO	1,677,987	6,856	5,390	39,612	2,038	53,897	3,212	8.4
Stockholm	SE	795,164	480	1,601	3,707	323	6,110	768	3.0
Ljubljana	SI	274,464	372	738	3,792	240	5,141	1,873	6.7
Bratislava, agglomeration	SK	528,124	1,042	1,461	6,153	468	9,123	1,727	5.6
Greater London Urban Area	UK	9,302,002	4,772	19,967	137,802	5,869	168,410	1,810	7.0
West Midlands Urban Area	UK	2,431,998	1,256	5,282	30,377	1,518	38,434	1,580	6.1
Greater Manchester Urban Area	UK	2,403,000	1,488	5,511	29,390	1,474	37,862	1,576	6.1

In Figure 10, the impact based on the exposure to road traffic noise and to air pollutants in the capitals of the EU-28 is compared with the perceived quality of the noise levels and the air pollution using data from the survey on Quality of Life in European Cities 2015.

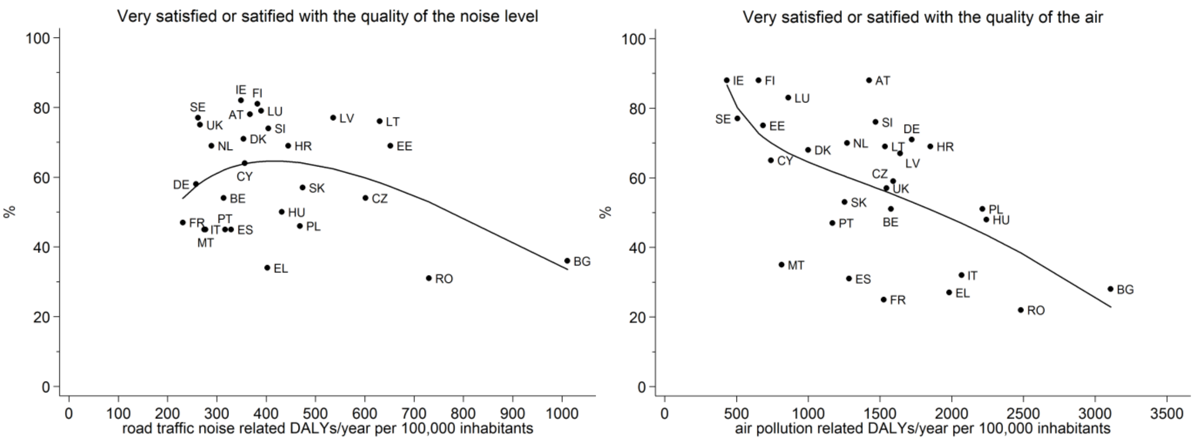


Figure 10. Perceived quality of the noise level (left hand side) and of the air pollution (right hand side) in European capitals Percentiles related to the impact of road traffic noise and air pollution expressed in DALYs/year per 100,000 inhabitants. The capitals are label with the ISO code for country.

The perceived quality of the air in European capitals seems more closely related to the health impact of air pollution than is the case for noise. This could be explained by that only road traffic noise was included in the health impact assessment for noise and that the perceived noise level may also include noise from other transport sources, shops and neighbours.

3.8 Ranking agglomerations on the combined burden of disease

The reliability of the ranking of the agglomerations on the combined burden of disease of road traffic noise and air pollution expressed as DALYs per 100.000 inhabitants was 0.92. The standard deviation was, on average, 10% of the mean of the combined burden of disease (standard deviation is 177 with an average of 1,745 DALYS per 100.000 inhabitants). The rankability based on the impact of road traffic noise and air pollution expressed as percentage of the total burden of disease was 0.87. Given the high rankability of these two indicators (>0.75), ranking of the agglomerations is possible.

In Table 7, the 15 agglomeration with the lowest burden of disease due to combined exposure to road traffic noise and air quality are ranked according to DALYs/year per 100.000 inhabitants. In Table 8, the 15 agglomerations with the highest burden of disease are found. We choose to show the top 15, since the information of 15 agglomerations fits nicely on a page. In the Table 9 and 10 the top 15 agglomerations with the lowest and highest burden of disease are shown when the ranking is based on impact of road traffic noise and air pollution as percentage of total burden of disease

The top 15's of agglomerations with the lowest combined burden are dominated by agglomerations from Finland, Iceland, Norway and Sweden. The top 15's of agglomerations with the highest combined burden consist of agglomerations from Bulgaria, Italy, Poland and Romania.

There is more than a 9 fold difference in combined impact of road traffic noise and air quality between the agglomeration with the highest burden of disease (4,540 DALY/year per 100,000 inhabitants) and the lowest burden of disease (481 DALY/year per 100,000 inhabitants). For the impact expressed in percentage of the total burden of disease, the difference between the highest and the lowest is slightly smaller (7 fold difference).

Table 7. Top 15 agglomeration with lowest burden of disease due to combined exposure to road traffic noise and air quality (based on DALYs/year per 100.000 inhabitants)

Agglomeration	Country	Population Size	Road traffic noise		Air quality		Combined road traffic noise and air quality		
			YLL/yr	YLD/yr	YLL/yr	YLD/yr	Total DALY/yr	DALY/yr per 100,000 inhabitants	% of total disease burden
Umeå	SE	111,770	64	219	225	30	538	481	1.9
TheGreatReykjavikArea	IS	191,999	80	388	526	67	1,061	552	2.8
Uppsala	SE	187,539	160	399	445	64	1,068	569	2.2
Orebro	SE	130,430	51	233	426	50	759	582	2.3
Oulu	FI	131,587	71	240	449	44	805	611	2.2
Vasteras	SE	133,728	85	274	432	49	840	628	2.5
Linkoping	SE	140,367	68	265	493	59	885	631	2.5
Boras	SE	100,983	47	192	358	43	639	633	2.5
Saint_Nazaire	FR	136,899	17	204	577	69	867	633	2.5
Jonkoping	SE	123,708	95	281	384	50	810	655	2.6
Norrkoping	SE	126,680	70	263	471	53	857	677	2.7
Fredrikstad	NO	129,317	65	276	480	56	878	679	2.9
Trondheim	NO	176,344	110	411	632	58	1,210	686	2.9
Tampere	FI	207,867	184	451	767	77	1,480	712	2.6
Stavanger	NO	230,167	104	460	1,002	100	1,666	724	3.1

Table 8. Top 15 agglomeration with highest burden of disease due to combined exposure to road traffic noise and air quality (based on DALYs/year per 100.000 inhabitants)

Agglomeration	Country	Population Size	Road traffic noise		Air quality		Combined road traffic noise and air quality		
			YLL/yr	YLD/yr	YLL/yr	YLD/yr	Total DALY/yr	DALY/yr per 100,000 inhabitants	% of total disease burden
Plovdiv	BG	342,200	2,933	1,519	10,447	638	15,536	4,540	11.0
Sofia	BG	1,377,532	8,557	5,367	40,576	2,231	56,732	4,118	10.0
Varna	BG	320,300	2,772	1,600	7,091	433	11,896	3,714	9.0
Milano	IT	1,256,211	2,534	5,286	36,611	1,688	46,119	3,671	13.4
Craiova	RO	243,767	1,523	1,034	5,827	355	8,739	3,585	9.4
Ruda Sl|eska	PL	143,395	520	566	3,447	241	4,775	3,330	10.5
Zabrze	PL	187,676	598	669	4,593	318	6,178	3,292	10.4
Pleven	BG	140,966	539	316	3,546	226	4,627	3,283	7.9
Chorzow	PL	113,008	332	360	2,799	189	3,679	3,256	10.2
Bucharest	RO	1,677,987	6,856	5,390	39,612	2,038	53,897	3,212	8.4
Torino agglomeration	IT	1,278,521	1,081	3,084	34,880	1,612	40,657	3,180	11.6
Gliwice	PL	196,171	476	571	4,752	336	6,135	3,127	9.8
Rybnik	PL	141,372	354	432	3,371	244	4,402	3,114	9.8
Bytom	PL	182,748	439	596	4,345	303	5,682	3,109	9.8
Katowice	PL	311,179	564	819	7,579	515	9,476	3,045	9.6

Table 9. Top 15 agglomeration with lowest burden of disease due to combined exposure to road traffic noise and air quality (based on percentage of total burden of disease)

Agglomeration	Country	Population Size	Road traffic noise		Air quality		Combined road traffic noise and air quality		
			YLL/yr	YLD/yr	YLL/yr	YLD/yr	Total DALY/yr	DALY/yr per 100,000 inhabitants	% of total disease burden
Umeå	SE	111,770	64	219	225	30	538	481	1.9
Oulu	FI	131,587	71	240	449	44	805	611	2.2
Uppsala	SE	187,539	160	399	445	64	1,068	569	2.2
Orebro	SE	130,430	51	233	426	50	759	582	2.3
Vasteras	SE	133,728	85	274	432	49	840	628	2.5
Linkoping	SE	140,367	68	265	493	59	885	631	2.5
Saint_Nazaire	FR	136,899	17	204	577	69	867	633	2.5
Boras	SE	100,983	47	192	358	43	639	633	2.5
Tampere	FI	207,867	184	451	767	77	1,480	712	2.6
Jonkoping	SE	123,708	95	281	384	50	810	655	2.6
Lahti	FI	99,998	83	215	399	41	738	738	2.6
Norrkoping	SE	126,680	70	263	471	53	857	677	2.7
The Great Reykjavik Area	IS	191,999	80	388	526	67	1,061	552	2.8
Turku	FI	175,285	176	414	761	64	1,415	807	2.9
Fredrikstad	NO	129,317	65	276	480	56	878	679	2.9

Table 10. Top 15 agglomeration with highest burden of disease due to combined exposure to road traffic noise and air quality (based on percentage of total burden of disease)

Agglomeration	Country	Population Size	Road traffic noise		Air quality		Combined road traffic noise and air quality		
			YLL/yr	YLD/yr	YLL/yr	YLD/yr	Total DALY/yr	DALY/yr per 100,000 inhabitants	% of total disease burden
Milano	IT	1,256,211	2,534	5,286	36,611	1,688	46,119	3,671	13.4
Torino agglomeration	IT	1,278,521	1,081	3,084	34,880	1,612	40,657	3,180	11.6
Plovdiv	BG	342,200	2,933	1,519	10,447	638	15,536	4,540	11.0
Ruda Sl|eska	PL	143,395	520	566	3,447	241	4,775	3,330	10.5
Zabrze	PL	187,676	598	669	4,593	318	6,178	3,292	10.4
Chorzow	PL	113,008	332	360	2,799	189	3,679	3,256	10.2
Sofia	BG	1,377,532	8,557	5,367	40,576	2,231	56,732	4,118	10.0
Gliwice	PL	196,171	476	571	4,752	336	6,135	3,127	9.8
Rybnik	PL	141,372	354	432	3,371	244	4,402	3,114	9.8
Bytom	PL	182,748	439	596	4,345	303	5,682	3,109	9.8
Katowice	PL	311,179	564	819	7,579	515	9,476	3,045	9.6
Lodz	PL	742,385	2,150	2,784	16,460	1,097	22,490	3,029	9.5
Krakow	PL	754,996	1,224	1,844	18,423	1,178	22,670	3,003	9.4
Craiova	RO	243,767	1,523	1,034	5,827	355	8,739	3,585	9.4
Vicenza	IT	115,927	166	351	2,318	140	2,975	2,566	9.4

4 Discussion

4.1 Introduction

To our knowledge this is the first time that a combined health impact assessment of air quality and noise on a European level has been carried out that is built upon data reported by the EU Member States under the Environmental Noise Directive (END) (EU, 2002) and the Air Quality Directive (EU, 2008).

The objectives were to assess the combined burden of disease attributable to the exposure to environmental noise and ambient air pollution in urban agglomerations in Europe and to explore if the agglomerations could be ranked based on the impact of the combined exposure.

The disability-adjusted life year (DALY) was chosen as the indicator for the integrated health impact assessment. The concept was already applied in the 90s to describe the environmental burden of disease (de Hollander et al., 1999). More recently, the environmental burden of disease in six European countries was compared using DALYs (Hänninen and Knol, 2011; Hänninen et al., 2014).

We considered quantifying the combined impact of noise and air pollution by estimating the total cost of these externalities. To our knowledge there is not an accepted method in Europe how to quantify the combined impact of both factors. In cost benefit analyses of environmental noise often a benefit of 25 Euro per household per decibel per year above noise levels of 50–55 dB L_{den} is used. This value is based on a meta-analysis of willingness to pay studies by Navrud (2002). However willingness to pay studies were not included in the cost benefit study that was carried out for the Clean Air Policy Package (Holland, 2014) so it is difficult to apply a common method that does do justice to the methods already applied in the fields of noise and air pollution.

In this chapter we discuss the uncertainties in the assessment and we compare our finding with results from other studies.

4.2 Uncertainties

We discuss the uncertainties in the results following the steps that can be distinguished in the process of calculation DALYs.

4.2.1 Selection of health endpoints with sufficient proof of a causal relation

It was expected that the WHO regional office for Europe (WHO/Europe) would have completed their Environmental Noise Guidelines for the European Region so their findings and recommendations could be used for the health impact assessment. Unfortunately this was not the case when this report was prepared. The review of WHO focuses on sleep disturbance, annoyance, cognitive impairment, mental health and wellbeing, cardiovascular diseases, hearing impairment and tinnitus and adverse birth outcomes. Sleep disturbance and cardiovascular diseases (coronary heart disease and stroke) were taken into account is this report. It is unknown if WHO will judge whether the evidence for stoke is sufficient. This endpoint was not part of the previous WHO evaluation (WHO, 2011) and the relation between noise and stroke has been evaluated in a limited number of studies. If not, the consequence for the health impact assessment is limited since the contribution of stroke to the overall burden in the 479 agglomerations is less than 10% (see Annex 1).

For the selection of the health endpoints related to air pollution results from the recent HRAPIE-project (WHO, 2013) were used. Since then, a number of new studies on the health effects of air pollution have been published. It is not expected that the addition of new health endpoints like Parkinson disease or a re-evaluation of the used health endpoints would lead to substantial differences in the health impacts assessment. Years of life lost due to premature mortality have the largest contribution to the burden of disease. In this study we used total mortality as endpoint, since the information about the base-line mortality risk is easily available and the calculation of the YYL is straightforward. If the impact had been calculated for cause specific mortality, the estimated impact is probably lower. In the EBODE study (Hänninen and Knol, 2011) the difference was about 25%.

4.2.2 Assessment of population exposure distributions

As indicated before the assessment is built upon data reported by the EU Member States under the Environmental Noise Directive (END) (EU, 2002) and the Air Quality Directive (EU, 2008). It was necessary to generate, from the reported data, exposure distributions for road traffic noise, PM, O_3 and NO_2 that cover the entire population in the considered agglomerations. Health risks of noise and air quality are also present in situations that are not considered as hotspots. This study could make use of the datasets produced by separate and unrelated studies on the exposure assessment of road traffic noise and of air quality (Houthuijs et al., 2015; Horálek et al, 2016 and Horálek et al, 2017). Since noise data is in the framework of the END only reported for hotspots like agglomerations with a population in excess of 100,000 inhabitants, the combined assessment was limited to the almost 500 END agglomerations in Europe.

The uncertainties in the air pollution exposure data are described in Horálek et al, 2016 and Horálek et al, 2017. In general, the precision of the interpolated data is within plus or minus 20% of the concentrations measured. The mapping methodology fulfilled the model data quality objectives as set by the Air Quality Directive (EC, 2008).

For the reported noise exposure data, there is concern about the comparability of the noise assessment in the various member states. Recently, Houthuijs et al. (2018) compared the reported fraction above 55 dB L_{den} for END agglomerations in 2012 with the modelled fraction based on population and road network density grids of 1 by 1 km. Systematic differences were found between countries. The correlation between the reported and modelled fraction improved from 0.30 to 0.70 when systematic differences between countries were taken into account. Non comparability of results was also identified for the first round of noise mapping in Europe (De Vos and Licitra, 2013). The non-comparability can have a substantial influence on the contribution of health endpoints with an assumed threshold, like the cardiovascular endpoints. It is not possible to quantify the magnitude of the uncertainties introduced by systematic differences between countries, nor is it possible to quality the consequences for the ranking of agglomerations. In addition, the noise exposure distribution for road traffic noise was extrapolated below 55 dB L_{den} and 50 dB L_{night}. The uncertainty in this part of the exposure distribution is unknown. The results from the health impact assessment suggest that the impact of the uncertainty could be limited. The exposure levels below 50 dB L_{night} contribute for 32% to the health impact of noise, so the uncertainty of the extrapolation is confined to this part. For cardiovascular endpoints the contribution of noise level below 55 dB L_{den} is very limited (see Figure 5).

This study estimates the attributable burden due to air pollution and road traffic noise in major European agglomerations. We are hesitant to extrapolate noise levels from other noise sources to lower levels. For example, a common policy of airports is to avoid or reduce aircraft flights over densely populated areas like agglomerations. It is therefore difficult to assess for individual agglomerations how the reported population sizes per 5 dB exposure category relate to the population size at lower noise levels. Road traffic noise is for the endpoints evaluated the most important contributor within agglomerations. Railway noise within agglomerations accounts for about 10% (Houthuijs et al, 2015). The contribution of aircraft noise (within and outside agglomerations) to the size of the (highly) sleep disturbed population is about a factor 4 smaller than the contribution of road traffic in agglomerations. The contribution of aircraft noise to coronary heart disease and cerebrovascular disease is small compared to the other sources (Houthuijs et al., 2015; Vienneau et al., 2015). By including only road traffic noise in the health assessment, the total burden of disease from noise might be underestimated by 20-25%.

4.2.3 Identification of exposure-response relations in dimensions compatible with the exposure distributions

The exposure response relations used for road traffic noise were all based on meta-analyses or pooled analyses of multiple studies (see Houthuijs et al., 2014). Exposure indicators were transformed to L_{den} or L_{night} where necessary. The exposure response relation for highly sleep disturbed was extrapolated to lower night-time noise levels (<40 dB L_{night}). The uncertainty related to the extrapolation is limited since these lower levels only contribute less than 20% of the burden of noise. We applied a threshold of 50 dB L_{den} for the cardiovascular effect of noise. It is unknown if WHO will declare a threshold for cardiovascular effect of noise and, if yes, what the value of this threshold will be.

The exposure response relations for air pollution were based on the HRAPIE study (WHO, 2013). We discussed in the methods section already that the recommendations from HRAPIE for NO_2 about a possible threshold value of 20 µg/m^3 might be too conservative, so we applied a counterfactual concentration of 10 µg/m^3. To avoid a possible overlap in risk with particulate matter, we adjusted the relative risk for NO_2 downwards to 1.039 per 10 µg/m^3, although this risk estimate is still higher than the interim value (1.025 per 10 µg/m^3) suggested by COMEAP (2015) or the for PM10 adjusted NO_2 estimate (1.02 per 10 µg/m^3) in Fischer et al. (2015) For the other endpoints we followed the HRAPIE study. We acknowledge that there is discussion about the use of the association between particulate matter and chronic bronchitis. Recently, COMEAP (2016) recommended not to include an association between long-term exposure to ambient air pollution and chronic bronchitis in core health impact assessments, but only in sensitivity analysis. The committee indicated that the considered evidence does not sufficiently establish causality. Chronic bronchitis contributes about 3.5% to the total DALYs attributable to air pollution in this report.

4.2.4 Estimation of the number of cases with the specific health endpoint

We used country specific data on population characteristics like the age distribution. Specific data for agglomerations is not easily available; we do not expect that using country instead of city baseline data will have a large influence on the results.

Also recent baseline country specific data for the incidence, mortality, hospital admissions and the burden of disease of cardiovascular or respiratory disease was used. Exceptions are the prevalence and incidence of (chronic) bronchitis and of (minor) restricted activity days for which one baseline value for all countries was used. Recent comparable European wide data on bronchitis or restricted activities days is missing, so it is difficult to evaluate the uncertainty introduced by adopting the same prevalence or incidence in all agglomerations. For sleep disturbance it is assumed that the same exposure response relation is applicable in all agglomerations. The relation is based on the L_{night} assessed at the highest façade of the building. It is expected that there are difference between the indoor and outdoor night-time level between the agglomerations, given the variation in climate and housing standards in Europe. So far as we know, there is no literature describing the validity of the assumption that the relation between night-time noise and sleep disturbance is generally applicable.

4.2.5 Estimation of the duration and of selection of appropriate weighing factors

The duration and weighing factors for the majority of the air pollution endpoints (mortality, cardiovascular and respiratory hospital admissions) were derived from the recent study of Bachmann and van der Kamp (2017). They compared and analysed the duration and weights from various studies and derived up-to-date values for them. In addition, we calculated DALYs for restricted activity days, works days lost, bronchitis and asthma. The uncertainties related to the duration and severity will have limited impact, since the contribution of all health endpoints of air pollution to the YLDs is less than 6% of the combined burden of disease of road traffic noise and air pollution.

For road traffic noise a different approach was followed. For the cardiovascular endpoints, the population attributable fraction was directly applied on the total YLD and YLL that was estimated by WHO for each of the countries. For sleep disturbance we applied a weighting factor of 0.06 which is slightly less than applied in other assessments (WHO, 2011; Hänninen et al., 2014). We derived the value from the same studies but, instead of the mean, we used the median given the skewed distributions. In recent Global Burden of Disease studies the assessment of the severity weights was improved by using paired comparison (Haagsma et al., 2015). It is recommended to update the severity weight of sleep disturbance, since the underlying studies applied different methods.

4.2.6 Computation of the total health burden

For this report, we updated for the health impact of noise some of the baseline data and we applied a slightly different calculation method for cardiovascular endpoints than in previous assessment (for details see Annex 1). The update of the baseline data and refinement of the method did not lead, on a European level, to major deviations of previous reported results. The methodology to assess the health impact of air pollution used here corresponds to the methodology used in the impact assessments presented in the annual Air Quality reports published by the EEA (2015, 2016, 2017). The year-to-year changes noted in the EEA reports largely stems from changes in the concentration fields. The sensitivity on the assumption of counterfactual concentration has been shown in the 2017-report (EEA, 2017). In comparison to the EEA results, we have adjusted the relative risk factor for NO2 as discussed above.

Ranking agglomerations on all kinds of aspect is very popular, especially in the lay press. We carried out a quantitative assessment of the uncertainty in the parameters of the exposure response relations and their consequence in the uncertainty in the total impact. The lower and upper 95% confidence limits of the combined burden of disease in each of the agglomerations are, on average, plus and minus 20% of the calculated total impact. It appears

that this size of uncertainty is small in comparison with the variation in combined impact between the agglomerations. Therefore the rankability of the agglomerations is good.
For future assessments it is recommended to expand the analysis to other sources of uncertainty (exposure distribution, variation in thresholds, baseline health risks, etc.). It might be possible to assess which source of uncertainty is the most relevant for the total impact, so improvements in the assessment can be target to the most relevant factor(s).

We identified in the paragraphs above various points of improvements in the health impact assessment for road traffic noise and air pollution. Although not quantified, we see as the most important points of attention the incomparability of the road traffic noise exposure data between agglomerations, the uncertainty about a threshold value for NO_2 and the potential overlap in health risks between NO_2 and particulate matter.

4.3 Comparison with results from other studies

In 11 we summarise the results of European studies into the health impact of environmental noise expressed as DALYs per 100,000 inhabitants.

Table 11. Overview of European studies into the health impact of noise.

Source	Country or city	DALYs per 100,000 inhabitants	Endpoints included
De Hollander et al., 1999	Netherlands, transportation noise	~180	Annoyance and sleep disturbance (YLD), and coronary heart disease (YLD and YLL) from 60 (?) dB
De Hollander, 2004	Netherlands, transportation noise	20-58	Cardiovascular endpoints (YLD + YLL) from 55 dB L_{den}
Knol and Staatsen, 2005	Netherlands, transportation noise	218	Annoyance and sleep disturbance (YLD), and hypertension related mortality (YLL) from 55 dB L_{den}
Stassen et al., 2008	Flanders (Belgium), transportion noise	342	Annoyance and sleep disturbance (YLD), and hypertension and coronary heart disease (YLD) from 55 dB L_{den}
Vienneu et al., 2015	Switzerland, transport noise	77	Cardiovascular disease (YLL) from 48 dB L_{den}.
Haninnen et al., 2015	Belgium, Finland, France, Germany, Italy, Netherlands, END data from 55 dB L_{den} and 50 dB L_{night}	37-148	Sleep disturbance (YLD), and coronary heart disease (YLD and YLL) from 55 dB
Tainio, 2015	Warsaw, Poland, END data from 55 dB L_{den} and 50 dB L_{night}	1,558	Annoyance, sleep disturbance (YLD), and coronary heart disease (YLD and YLL) from 55 dB
Muller et al., 2017	Barcelona, Spain, road traffic noise	1,184	Annoyance and sleep disturbance (YLD), hypertension and cardiovascular diseases (YLD) from 45 dB and mortality (YLL) from 60 dB
This study	479 agglomerations in Europe, road traffic noise	359	Sleep disturbance (YLD), coronary heart disease and stroke (YLD+YLL) from 50 dB L_{den}

The estimated impact in Table 11 varies from 37 to 1,558 DALYs per 100,000 inhabitants, a 40 fold difference. The most important explanation for the differences are whether annoyance is included in the calculation of the burden of disease and the applied severity weights for annoyance and sleep disturbance. In the (oldest) studies in the Netherlands and Belgium annoyance and sleep disturbance were included in the calculation of the burden of disease, but relative low severity weights were applied (0.01 or 0.02). In later studies higher disability weights (0.02 and 0.07) were used referring to the report on the Burden of Disease of Environmental Noise report (WHO, 2011). In later report of WHO (2012), it was recommended not to use annoyance in burden of disease calculation. This was implemented in the EBODE study (Hanninen et al., 2015) and in this report, but not in the assessments in Warsaw and Barcelona (Taino, 2015 and Muller et al., 2017). In Barcelona annoyance contributed 37% to the total health impact of noise; in Warsaw 49%.

The evidence about the association between noise and cardiovascular disease has changed over time, since results from new studies have become available. In the older health impact assessment studies on noise a threshold of 55 or 60 dB was applied for the risk of hypertension or cardiovascular disease which leads to smaller impacts.

There are differences in the location of the assessments in the studies. In a country wide assessment (the first 6 references in Table 11) the average exposure level will be lower than in assessments in agglomerations. Also in some studies, the full exposure distribution for noise is not available (Haninnen et al., 2015 and Tainio, 2015), or only one source of transportation noise is quantified (Muller et al. 2017 and this study).

These observations illustrate that it is not possible to compare the results of this report with the outcomes of European studies on the health impact of environmental noise, given the various differences in the methods applied.

The largest contributor to the health impact of air pollution is premature mortality. In earlier health impact studies only particulate matter was linked to mortality. For example the EBODE study in 6 European countries estimated an impact of 600-1,000 DALYs per year per 100,000 inhabitants for the exposure to $PM_{2.5}$. Our results (on average, 785 DALYs per year per 100,000 inhabitants) are in line with these results from EBODE (Hanninen et al., 2015). Particulate matter is therefore the most important environmental contributor to the overall burden of disease.

The HRAPIE study of WHO (2013) indicated that there is an independent association of nitrogen dioxide and mortality. In this report the mortality of NO_2 contributes to 40% of the total health impact of air pollution in agglomerations (on average, 573 DALYs per year per 100,000 inhabitants). From the studies in Table 11, only in the health impact assessment for Warsaw the impact of NO_2 was quantified (20% contribution to the total impact of air pollution, about 300 DALYs per year per 100,000 inhabitants). We already mentioned earlier the uncertainty about the threshold value for NO_2 and the potential overlap in health risks between NO_2 and particulate matter. Differences in the appreciation of these uncertainties could easily lead to different implementations of exposure response relations for NO_2 in future health impact assessments.

Several health impact assessments concluded in the past that in the prioritising of environmental risks factor it is particulate matter “first”, and noise “second”. The results of this report suggest that for agglomerations the health impact of NO_2 is, on average, larger than the health impact of road traffic noise. In 292 of the 479 agglomerations (61%) the impact of NO_2 was larger than the impact of road traffic noise. We have identified in the report several improvements for the health impact assessment of NO_2 and for road traffic noise, so we do not conclude that NO_2 is of more importance than road traffic noise.

The comparison of the results of this report had led to the notion that for some of the 479 agglomerations results of health impact assessments on environmental noise and air pollution

separately or combined are already available. It is likely that the results from these studies will differ from the results in this report. We emphasize that an important objective of this report is to get insight in the "average" burden of disease due to noise and air pollution in agglomerations in Europe and to explore if ranking of the agglomerations is possible. For this reason, we applied a common methodology for all agglomerations.
We acknowledge that other choices in the quantification of the health impacts are possible. We expect that in local or country specific studies the selection of the health endpoints, exposure response relations and other relevant aspects have been discussed and agreed with stakeholders so that the methodology will be based on more detailed information and is therefore more tailored to the local situation than is possible in a European wide assessment. If results of the health impact assessment from this report differ from the results described in local or country reports, we recommend valuing the results from the local or country reports as leading.

4.4 The use of DALYs for comparison of noise and air pollution impacts

It should be noted that DALYs give, at the best, an indication of the potential, relative order of the magnitude of different environmental health impacts in the agglomerations or about the relative order of the combined health impact in the agglomerations. The burden of disease cannot be interpreted as absolute or completely representative numbers in the light of the discussion about the representative of the exposure distributions and the exposure response relations for individual agglomerations, the selection of the health endpoints, disability weights and durations, etc. Prioritisation should therefore not be done on DALYs solely (Knol et al., 2009).
DALYs are only useful, and not more than a tool, in situations where environmental factors with different and incomparable health states have to be compared or summed in the same unit to be used for a ranking of the combined impact. Comparability in the calculation of the impact of different environmental factors and assessment of the exposure in similar ways are crucial otherwise the use of DALYs as an integrated indicator cannot be justified (Knol et al., 2009). The concept of DALYs is particularly useful when comparing the effectiveness of abatement measures to reduce the environmental health impacts.

Overall, we are confident that for the majority of the agglomerations the ranking of the agglomerations is robust. Unfortunately, it is difficult to quantify how comparable noise exposure distributions reported in the framework of the END are between agglomerations. There are agglomerations with a relative small impact for road traffic noise. In some cases, we have serious doubts about the quality of the reported data. Second, there are differences between the calculation of the impact of noise and the calculation of air pollution and in the appreciation of the health endpoints. We choose to make use of authoritative reviews or literature (WHO, CAFÉ, EEA, etc.). However, it should be noted that most of these reviews are orientated on or noise or air pollution, and not on the combination of both. The results of the relative small number of papers where both environmental factors are quantified indicate that study findings are difficult to compare between papers given the different methods that were used. This does not mean that the applied methods are not appropriate. As pointed out above, DALYs should be used for relative and not for absolute comparisons which could lead to different choices. Nevertheless, the field of integrated health impact assessment would benefit if the methods for health impact assessment of environmental noise and air pollution are more harmonised. For this reason, we added as sub title to this paper "An explorative study".

5 Conclusions

It is possible to rank large agglomerations in Europe on their combined health impact of road traffic noise and air pollution. Rankings based on the percentage of the total burden of disease or based on DALYs per 100,000 inhabitants may slightly differ. Thirteen agglomerations can be found in both top15's with the lowest burden. These agglomerations are mainly located in the Nordic countries; population numbers are below 250,000. Agglomerations with the highest burden are located in eastern Europe and Italy. A wide range in size is seen: population numbers range from 113,000 to 1.4 million. Twelve agglomerations are included in the two listings of the top15 having the highest burden.

The combined health impact is 1,745 DALYs per year per 100.000 inhabitants. This corresponds with 6.2% of the total burden of disease for all causes per year. 1.3% Of the burden is by noise and 5.0% by air pollution.

There is a 7 to 9 fold difference in the health impact between the highest and lowest ranked agglomeration, so there is ample room for improvement of the health impact of road traffic noise and air pollution in a large number of European agglomerations.

Particulate air pollution contributes, on average, for 45% to the total impact, followed by nitrogen dioxide with 33%. Road traffic noise is associated with 21% and ozone with 2% of the combined burden of disease by road traffic noise and air pollution.

The impact of air pollution is for 94% related to premature mortality and for 6% to years lost with a disability. For road traffic noise, 30% is related to premature mortality and 70% to years lost with a disability.

The health impact is not homogenous distributed over Europe. Three clusters of agglomerations can be spatially distinguished.

- Relative low contributions from particulate matter, ozone, nitrogen dioxide and road traffic noise are found in the (extended) coastal regions of the Atlantic and North Sea and in the Baltic countries (total impact is, on average, 1,273 DALYs per year per 100.000 inhabitants);
- In the central and southern part of Europe, relative large contributions of nitrogen dioxide and ozone are seen (total impact is, on average, 1,857 DALYs per year per 100.000 inhabitants), and
- In the eastern part of the EU the impact is almost twice as high as in the coastal areas. Particulate matter and road traffic noise contribute to the relative high total impact (on average, 2,649 DALYs per year per 100.000 inhabitants).

Within these clusters, there can be substantial differences between agglomerations.

This report identified various points of improvements in the health impact assessment for road traffic noise and air pollution itself and for the combined assessment. Important points of attention are the incomparability of the road traffic noise exposure data between agglomerations, the lack of information about noise exposures levels below 55 dB L_{den} and 50 dB L_{night}, and the uncertainty about a threshold value for NO_2 and the potential overlap in health risks between NO_2 and particulate matter.

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Annex 1 Update of methodology for health impact assessment of noise

Introduction

The methodology for health impact assessment of noise has been described in more detail in Houthuijs et al (2014). At that time, the number of additional cases for cardiovascular disease was calculated in line with the calculation of additional cases for reading impairment and hypertension. The consequence was that the results slightly overestimates the attributable number of cases and slightly underestimate the population attributable fraction. However, an adequate calculation of the population attributable fraction was not possible, since a part of the relevant exposure (between 50 and 55 dB L_{den}) was missed.

Population attributable fraction

Since in this report the full noise exposure distribution in agglomerations was known, we used the common approach to calculate the population attributable fraction (PAF). The PAF is 'the proportional reduction in population disease or mortality that would occur if exposure to a risk factor were reduced to an alternative ideal exposure scenario'. The PAF per agglomeration can be calculated with the following equation:

$$PAF(c) = \frac{\sum_{Lden=0}^{75} f_{inhab}(Lden, c) * RR(Lden) - \sum_{Lden=0}^{75} f_{inhab}(Lden_{alt}, c) * RR(Lden)}{\sum_{Lden=0}^{75} f_{inhab}\,(Lden, c) * RR(Lden)}$$

with:

$f_{inhab}(L_{den}, c)$	fraction of inhabitants per dB L_{den} per agglomeration
$f_{inhab}(L_{den,alt}, c)$	fraction of inhabitants per dB L_{den} per agglomeration in alternative ideal exposure scenario
$RR(L_{den})$	Relative Risk at exposure level L_{den}

If we assume that an ideal exposure scenario does not lead to an excess risk (RR=1), the equation can be written as:

$$PAF(c) = \frac{\sum_{Lden=0}^{75} f_{inhab}(Lden, c) * (RR(Lden) - 1)}{(\sum_{Lden=0}^{75} f_{inhab}\,(Lden, c) * (RR(Lden) - 1)) + 1}$$

In the next step, the number of attributable cases per year can be calculated:

PAF(c) * incidence(c)

With:

incidence(c)	incidence of disease characteristic per year per agglomeration (country estimate was used)

Baseline disease estimates

In the earlier assessment we used (crude) country-specific incidence data on hospital discharges and mortality from the European Cardiovascular Disease Statistics 2012 to assess the 'base-line' risks for coronary heart disease and stroke. For the new assessment, we used incidence data for coronary heart disease and stroke from 2015 for male and females separately (European Cardiovascular Disease Statistics 2017). Incidence was not known earlier, so we changed from hospital discharges data to incidence data. We extracted deaths by cause from the Global Health Estimates 2015 (data from March 2017).

Information about the disease burden associated with coronary heart disease and stroke (years lived with disability and years life lost) was obtained from the WHO Global Health Estimates for 2000-2011 for the Worldbank/WHO regions "high-income countries" and "European Region (low- and middle-income countries)". For the new assessment, we used DALY by cause and Burden of disease by cause from the Global Health Estimates 2015 (data from December 2016). An important difference is that the baseline data is now available per country instead of WHO region.

Comparison of results

We compared the results of the previous applied method of health impact assessment with the (updated) methodology that was applied in this report. The table below gives the main results for all agglomerations.

Endpoint	**Previous method**	**Updated method**	**Remark**
Incidence of coronary heart disease per year	31,378	29,678	Hospital discharges, versus incidence, update of data, change of calculation method
Incidence of cerebrovascular disease per year	15,168	5,033	
Mortality due to coronary heart disease per year	6,947	9,606	Update of data, change of calculation method
Mortality due to cerebrovascular disease per year	3,910	3,606	
YLD due to coronary heart disease per year	11,920	8,001	
YLL due to coronary heart disease per year	128,860	144,026	
YLD due to cerebrovascular disease per year	6,969	4,567	
YLL due to cerebrovascular disease per year	53,454	52,018	

The results indicate that a substantial decrease in the incidence of cerebrovascular disease (from 15,168 to 5,033 cases per year: -67%) is introduced by the switch from hospital discharges to incidence data. The total incidence of cardiovascular disease decreases from 46,546 to 34,711 cases per year (-25%).

The incidence of mortality due to coronary heart disease has the largest increase (from 6,947 to 9,606 cases per year: +38%) due the change of data and methodology. The total number of deaths per year changes from 10,857 to 13,212 (+22%).

In the report of 2014, we did not report the burden of disease expressed as DALYs. In the table below we compare the results of both methods for the calculation of the burden of disease.

Burden of disease	Previous method	Updated method	Remark
Highly sleep disturbed (YLD)	429,638	429,533	Change of rounding in process of calculation (more significant digits in new method)
Coronary heart disease (YLD+YLL)	140,780	152,026	Update of data, change of calculation method
Cerebrovascular disease (YLD+YLL)	60,423	56,585	
Total DALYs	630,841	638,145	

The update of the data sources and the change in methodology leads to different results. However, the overall effect is limited (630,841 to 638,145 DALY per year: +1.2%).

The changes for individual agglomerations can be larger than described for the total of all agglomerations. However, results for individual agglomerations have not been reported before.