# RIVM
# Road traffic measures: impact on air quality and public health

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At the request of the Netherlands Court of Audit, the National Institute for Public Health and the Environment (RIVM) has calculated the estimated impact of certain road traffic measures on air quality and public health in the Netherlands.

Exposure to atmospheric pollution is a serious risk to public health. Atmospheric pollution consists of a mixture of substances that can influence health in a variety of ways and to various degrees. Particulate matter (PM) is by far the most relevant substance in this risk to public health, followed by nitrogen dioxide. Ozone has far less of an impact on the disease burden.

An improvement in air quality improves public health and the quality of life. This study first estimates the changes in atmospheric concentrations following the implementation of certain road traffic measures and then the improvement in health. Best estimates were calculated of the impact of eight road traffic measures on reducing emissions of particulates and nitrogen dioxides ($NO_x$).

## Concentrations

To plot the Large-Scale Air Quality Concentration Maps of the Netherlands (GCN) (Velders et al., 2017) concentrations of $NO_2$, particulates ($PM_{10}$) and fine particulates ($PM_{2.5}$) are calculated at a resolution of 1x1 km every year. The GCN maps are based on separate emission maps for each of a large number of underlying emission sectors. Road traffic is divided into 15 sectors, depending on the type of vehicle and type of road.

With the aid of an atmospheric distribution model, a concentration map is calculated and saved for each of the emission sectors. The calculation is based on an Operational Model for Priority Substances (OPS model) (Jaarsveld, 2004; Sauter et al., 2015). Given the emission and meteorological conditions, the OPS model calculates the distribution of a substance across the Netherlands, taking account of its removal from the atmosphere by means of dry deposition (pollutant deposited on the earth's surface by means of gravitational settlement) and wet deposition (pollutant deposited by means of precipitation). The calculation also allows for the chemical conversion of primary emitted compounds (sulphur dioxide, nitrogen dioxide and ammonia) into secondary aerosols (ammonium nitrate and ammonium sulphate).

The results have a linear relationship with the emissions entered in the OPS model. If the emission entered in the model increases or decreases by x%, the resultant concentration will also increase or decrease by x%. A measure's impact on the increase or decrease in emissions can thus be calculated for each sector and expressed as a concentration.

A scaling factor is calculated for each sector to express the difference between the emission in the benchmark calculation and the emission in the different scenarios ( i.e. the difference between the emission after measure(s) have been taken and the emission in the benchmark calculation). The original concentration grid (the concentration map produced for the benchmark calculation with a resolution of 1x1 km for the whole of the Netherlands) for the sector concerned is multiplied by this scaling factor in order to identify a measure's impact on a concentration. Repeating this procedure for all sectors in which measures are implemented produces the cumulative difference relative to the benchmark situation. These calculations are made each year using current meteorological information.

Our study looked at the measures' impacts on concentrations of $PM_{2.5}$ and $PM_{10}$. PM is a mixture of emitted primary substances and secondary aerosols formed in the atmosphere. $NO_x$ emissions play a role in the formation of secondary aerosols. Lower $NO_x$ emissions reduce the formation of secondary aerosols and therefore reduce the PM concentration.

Population-weighted annual average concentrations are shown in figure 5. The chart on the left shows the concentrations calculated for the benchmark situation and the chart on the right the difference between the benchmark calculation and the scenarios before correction for free riders and after correction for free riders.

An example of a calculated concentration map is shown in figure 4 (in this case the difference between the benchmark situation and the scenario before correction for free riders for the year 2011).

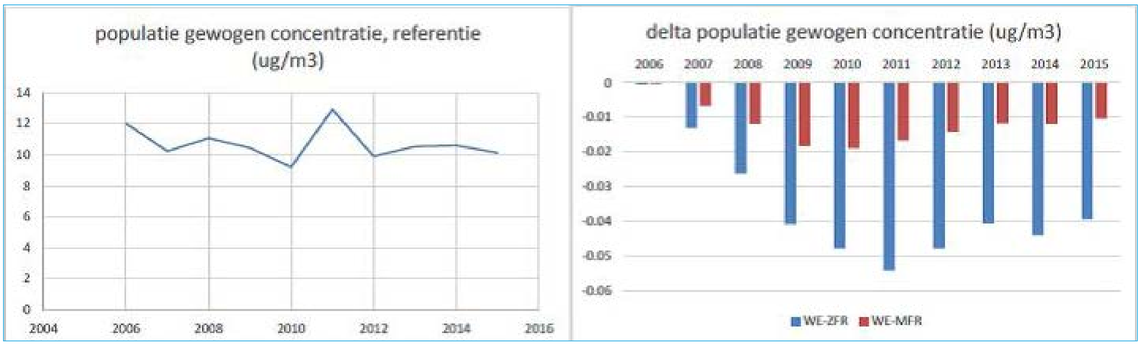

**Figure 1** *Population-weighted annual average PM$_{2.5}$ concentration*

### Change in PM$_{2.5}$ concentration (benchmark versus scenario)

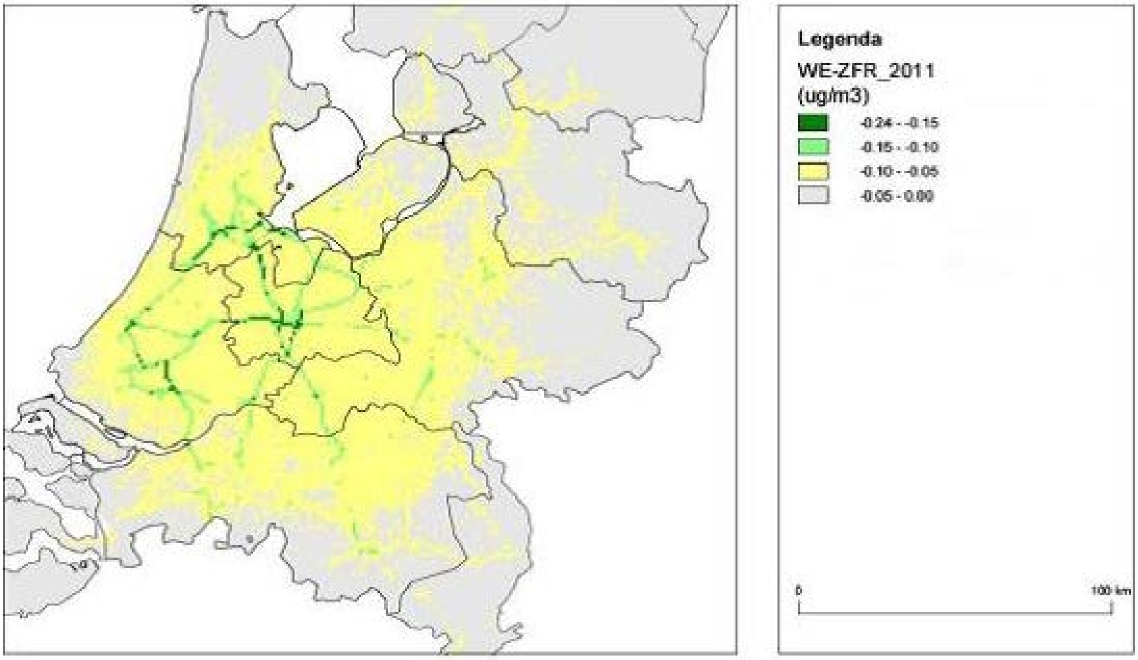

**Figure 2** *Model results: change in concentrations between benchmark calculation and the scenario before correction for free riders; emission and meteorological data for 2011*

## Health impact

The impact on public health was calculated on the basis of relative risks, as recommended by the WHO (2013). Only health effects due to exposure to PM were considered. The difference in exposure to PM$_{2.5}$ was determined as the difference between the concentration grid calculated for the benchmark situation and the calculation grid calculated for a particular scenario before or after correction for free riders. The calculation method is explained in Leeuw and Horalek (2016).

Ozone makes a considerably smaller contribution to the disease burden. Ozone concentrations are determined chiefly by emissions of $NO_x$ and volatile organic matter and methane in the entire northern hemisphere. Local road traffic emissions have only a modest influence. Furthermore, grid-specific information on the emission of volatile organic matter that is necessary to calculate changes in ozone concentrations is not available.

The direct impact of $NO_2$ on public health has not yet been included in the calculations as there are still mathematical uncertainties about double counting in the relative risks of $NO_2$ and particulates and uncertainties regarding $NO_2$ concentrations below which there is no observable impact (Heroux et al., 2015). The contribution of NOX to secondary PM is included in the calculations. The omission of $NO_2$ could lead to the study underestimating the health impact of road traffic measures by 10–50%.

Table A.1 in the tables appendix summarises the selected impacts and concentration response functions.[1] To arrive at an estimate of the total health gain, the impacts are aggregated both in DALYs (Disability Adjusted Life Years) and on the basis of monetised health loss (table A.2).

## Results

The cumulative health gains (aggregated over all years in the period 2006 to 2015) is summarised for each of the scenarios (before correction for free riders (BCFR) and after correction for free riders (ACFR)) in table 1.

The loss avoided (or gain) of each health impact is shown in tables A.3a and A.3b in the tables appendix. The number of years of life lost makes the highest contribution to both the DALYs avoided and the monetised loss avoided, higher than the disease burden.

Table 1 presents two monetised loss calculations: the first uses the lower and upper limits for a year of life lost (YLL) due to premature death as used in the European Commission's Impact Assessments (Holland, 2014) and the second uses the valuation of a year of life lost as used in the Environmental Prices Handbook of CE Delft. CE Delft uses a central estimate with lower and upper values.

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[1] In some cases the concentration response function is based on the $PM_{10}$ concentration. This concentration has not been modelled directly but is estimated from the empirical relation: $c(PM_{10}) = 1.52\ c(PM_{2.5})$.

On average, the measures reduced the concentration by 0.3% (BCFR) and 0.1% (ACFR) in the period 2006–2015. The measures reduced the overall disease burden by 0.02% (BCFR) and 0.01% (ACFR), equal to 1,150-3,378 healthy years of life.

**Table 1** Aggregate DALYs avoided and health loss avoided over the period (2015 price levels)

| Scenario | DALYs avoided | Loss avoided (€ millions) * | Loss avoided (€ millions) ** |
|---|---:|---:|---:|
| ACFR | 1,150 | 94–185 | 96 (76–138) |
| BCFR | 3,378 | 275–542 | 283 (222–404) |

* lower and upper values of years of life lost (YLL) based on table A.2.

** based on central value of € 70,000 per year of life lost (lower and upper values of € 50,000 and € 110,000 respectively) according to CE Delft (2017a, b).

On the basis of the valuation figures in table A.2, the minimum total gain in the Netherlands (aggregated over the period 2006–2015) is estimated at € 275 million in the BCFR scenario and at € 94 million in the ACFR scenario. If the central valuation for a lost year of life in the Environmental Prices Handbook (CE Delft 2017a, b) is used, the totals are slightly higher.

On the basis of the number of DALYs avoided, premature death represents about 90% of the reduction in the overall disease burden (including premature death). The reduction in premature death was responsible for 74% of the health loss avoided.
For a number of reasons, these gains are probably underestimated:

- The gains relate solely to the Netherlands. Emissions from the Netherlands, however, are carried to neighbouring countries and so contribute to exposure concentrations throughout Europe. Measures taken in the Netherlands will also lead to health gains in the rest of Europe. European models (EMEP, 2016) estimate that the health gains in the other EU member states of the road traffic measures taken in the Netherlands are two to three times greater than in the Netherlands itself. Conversely, of course, the Netherlands benefits from measures taken in other countries.
- Not all health impacts are taken into account because they cannot be quantified or are surrounded by great uncertainty. One example of this is premature death due to exposure to $NO_2$. A concentration response function is available but there are uncertainties regarding the relative risk, the double counting with particulates and the no-effect value (Heroux et al., 2015).
- Particulates are made up of a large number of different components; the composition can differ significantly depending on the place and time. The health effects, however, are based on the total mass of $PM_{2.5}$. Equal concentrations are assumed to have equal impacts regardless of their composition. There are indications that combustion

aerosols (soot) are more significant than other components (Janssen et al., 2011). Emissions from diesel vehicles contain a relatively high soot fraction. Using the same risk factor for all particulate components may underestimate the health gains. If soot is indeed more harmful than other components, the measures targeting diesel vehicles (such as particulate filters) will make a bigger contribution to the health impact.

- There are significant uncertainties in the risk figures. The total number of DALYs and the loss avoided are based largely on premature death. Apart from the uncertainty in the valuation of a year of lost life, the additional uncertainty margin in relation to exposure and impact is about 30% (with a 95% reliability interval).

Social cost-benefit analyses compare the cost of implementing a measure with the monetised social benefits, with each extra year of life being valued at between € 50,000 and € 110,000 in accordance with the Environmental Prices Handbook (CE Delft, 2017a, b).

In comparison with these benefits the cost of the road traffic measures studied was relatively high per DALY gained. The national cost amounted to more than € 400 million,[2] which is equal to nearly € 350,000 per DALY gained (after correction for free riders).

It should be noted that the measures' impact was two to three times greater outside the Netherlands. If these effects are taken into account the cost per DALY gained comes to € 87,000 – € 116,000. The figures might have been even lower if the direct health impact of $NO_2$ and the specific risks of soot had also been taken into account.

It should also be noted that the measures were not taken solely to improve public health. A more important reason for introducing them was to stop breaches of EU limits so that construction projects could be given the green light.

Atmospheric concentrations fell during the period 2006–2015 not only because of the road traffic measures taken but chiefly because of the measures agreed at EU level (such as the NEC directive, and the directives concerning industrial sources, vehicles, boilers and products). According to Smeets et al. (2015) the cost of implementing EU air policy to the Netherlands was about € 4,000 per year of life gained.

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[2] Source: Ministry of Infrastructure and the Environment, with an estimate of implementation costs incurred by the Netherlands Court of Audit and an estimate of the Euro 6 discount by CE Delft.

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# Tables RIVM study

**Table A.1** Health impact and concentration response functions (CI: 95% reliability interval). The relative risk indicates how large the change in impact is for every 10 $\mu g/m^3$ increase in concentration; for example, a 10 $\mu g/m^3$ increase in $PM_{2.5}$ concentrations leads to a 6.2% increase in premature deaths.

| Component | Health impact | Relative risk (a) (95% CI) per 10 $\mu g/m^3$ | Incidence / prevalence | Ref |
|---|---|---|---|---|
| $PM_{10}$ | Days with bronchitis among children aged 6-12 | 1.08 (0.98 – 1.19) | 7.9% | WHO (2013) |
| $PM_{10}$ | Incidence of chronic bronchitis among adults aged 27 and older | 1.117 (1.040 – 1.189) | 0.39% | WHO (2013) |
| $PM_{2.5}$ | Emergency hospital admissions for cardiovascular disease, all age groups | 1.0091 (1.0017 – 1.0166) | 1,692 per 100,000 (2012) | CBS (2014) |
| $PM_{2.5}$ | Emergency hospital admissions for bronchial complaints, all age groups | 1.0190 (0.9982 – 1.0402) | 813 per 100,000 (2012) | CBS (2014) |
| $PM_{2.5}$ | Days with limited activity, all age groups* | 1.047 (1.042 – 1.053) | 19 days per person per annum | WHO (2013) |
| $PM_{2.5}$ | Number of days sickness absence (20-65 year olds) | 1.046 (1.039 – 1.053) | 10.6 per working person | WHO (2016a) |
| $PM_{10}$ | Days with asthma complaints among children with asthma (5 – 18 year olds) | 1.028 (1.006 – 1.051) | 4.9% of the children, on average 62 days per annum | WHO (2013) |
| $PM_{2.5}$ | Premature death | 1.062 (1.04 – 1.083) | 834.4 per 100,000** | WHO (2016b) |

* To prevent double counting, estimates of the cost of the number of days with limited activity are adjusted for absenteeism, hospital days for cardiovascular disease and bronchial complaints and days of acute bronchitis among children (WHO, 2013).

** 30 years and older, natural death.

**Table A.2** DALY units and monetary loss per health impact.

| Com-ponent | Health impact | Unit | DALY per unit | Source | Loss per unit (a) |
|---|---|---|---:|---:|---:|
| $PM_{10}$ | Days with bronchitis among children aged 6-12 | Days per annum | 0.00062 | (b) | € 49 |
| $PM_{10}$ | Incidence of chronic bronchitis among adults aged 27 and older | Number per annum | 0.99 | (c) | € 62,712 |
| $PM_{2.5}$ | Emergency hospital admissions for cardiovascular disease, all age groups | Number per annum | 0.02255 | (d) | € 2,574 |
| $PM_{2.5}$ | Emergency hospital admissions for bronchial complaints, all age groups | Number per annum | 0.01565 | (d) | € 2,574 |
| $PM_{2.5}$ | Days with limited activity, all age groups | Days per annum | 0.00027 | (c) | € 108 |
| $PM_{2.5}$ | Number of days sickness absence (20-65 year olds) | Days per annum | 0.00027 | (c) | € 152 |
| $PM_{10}$ | Days with asthma complaints among children with asthma (5 – 18 year olds) | Days per annum | 0.00019 | (c) | € 49 |
| $PM_{2.5}$ | Premature death (YLL) | Number per annum | 1 | (d) | € 67,500 (e) |

(a) Holland (2014), adjusted for 2015 price levels. Amounts based on lower limit.

(b) WHO (2017)

(c) HEIMSTA/INTARE (2011)

(d) Bachmann and Van der Kamp (2017)

(e) Holland (2014) adopts an upper limit of €133,000 to value a year of lost life.

**Table A.3a** Numbers and DALYs avoided and monetary benefits, BCFR scenario, aggregated over 2006–2015. The benefits are based on the cost per health impact, as shown in table A.2.

| Component | Health impact | Unit | Number | DALY | Benefit (€ thousands) |
|---|---|---|---:|---:|---:|
| $PM_{10}$ | Days with bronchitis among children aged 6–12 | Days per annum | 5,539 | 3 | 271 |
| $PM_{10}$ | Incidence of chronic bronchitis among adults aged 27 and older | Number per annum | 220 | 218 | 13,795 |
| $PM_{2.5}$ | Emergency hospital admissions for cardiovascular disease, all age groups | Number per annum | 90 | 2 | 232 |
| $PM_{2.5}$ | Emergency hospital admissions for bronchial complaints, all age groups | Number per annum | 89 | 1 | 229 |
| $PM_{2.5}$ | Days with limited activity, all age groups (a) | Days per annum | 349,614 | 95 | 37,758 |
| $PM_{2.5}$ | Number of days sickness absence (20-65 year olds) | Days per annum | 122,638 | 33 | 18,641 |
| $PM_{10}$ | Days with asthma complaints among children with asthma (5–18 year olds) | Days per annum | 12,722 | 2 | 623 |
| $PM_{2.5}$ | Premature death (YLL) | Number per annum | 3,023 | 3,023 | 204,101 |
| | | **Total** | | 3,378 | 275,650 |

**Table A.3b** Numbers avoided, DALYs and costs, ACFR scenario, aggregated over 2006–2015. The benefits are based on the cost per health impact, as shown in table A.2.

| Component | Health impact | Unit | Number | DALY | Benefit (€ thousands) |
|---|---|---|---:|---:|---:|
| $PM_{10}$ | Days with bronchitis among children aged 6–12 | Days per annum | 1,886 | 1 | 92 |
| $PM_{10}$ | Incidence of chronic bronchitis among adults aged 27 and older | Number per annum | 75 | 74 | 4,698 |
| $PM_{2.5}$ | Emergency hospital admissions for cardiovascular disease, all age groups | Number per annum | 31 | 1 | 79 |
| $PM_{2.5}$ | Emergency hospital admissions for bronchial complaints, all age groups | Number per annum | 30 | 0 | 78 |
| $PM_{2.5}$ | Days with limited activity, all age groups (a) | Days per annum | 119,050 | 32 | 12,857 |
| $PM_{2.5}$ | Number of days sickness absence (20-65 year olds) | Days per annum | 41,760 | 11 | 6,348 |
| $PM_{10}$ | Days with asthma complaints among children with asthma (5–18 year olds) | Days per annum | 4,332 | 1 | 212 |
| $PM_{2.5}$ | Premature death (YLL) | Number per annum | 1,030 | 1,030 | 69,502 |
|  |  | **Total** |  | 1,150 | 93,867 |