Swedish MK1 diesel versus European EN 590 diesel

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Comparing the impact on emissions and health risks in the metropolitan area of Stockholm

A report for the Swedish Transport Administration by

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Abbreviations

Abbreviation (in alphabetic order)	**Explanation**
BaP	Benzo[a]pyrene. A polycyclic aromatic hydrocarbon.
CO	Carbon monoxide.
DB[a,l]P	Dibenzo[a,l]pyrene. A polycyclic aromatic hydrocarbon.
DPF	Diesel particle filter. Exhaust aftertreatment, reduces particle emissions.
EGR	Exhaust gas recirculation. Exhaust after treatment technique; reduces NOx formation due to lower combustion temperature and lower oxygen concentration.
EN590	A European diesel fuel quality introduced in 1993. It has undergone several revisions (1999, 2004 and 2009) that have reduced the sulphur content from 2000 mg/kg to 10 mg/kg.
HBEFA	Handbook Emission Factors for Road Transport. Provides emission factors for all current vehicle categories (cars, LDV, HDV, urban buses, coaches and motor cycles), each divided into different categories, for a wide variety of traffic situations. Emission factors for all regulated and the most important non-regulated pollutants as well as fuel consumption and CO_2 are included.
HC	Hydrocarbons.
HDV	Heavy duty vehicles. Includes trucks and busses.
LDV	Light duty vehicles. Includes cars and light duty vans and busses.
MK1 diesel	“Miljöklass 1 diesel”. Environmental Class 1 diesel, introduced 1991 according to the requirements defined in SS 15 54 35 (e g max 10 mg sulphur per kg diesel and 0.02 vol% PAH). It has undergone several revisions.
NOx	Nitrogen oxides. Sum of nitric oxide (NO) and nitrogen dioxide (NO_2).
PAH	Polycyclic aromatic hydrocarbons. In this report PAH is the sum of 18 different compounds.
SCR	Selective Catalytic Reduction. Exhaust after treatment technique; reduces NOx to nitrogen (N_2).
TEF	Toxic equivalence factor. In this report TEF values for individual PAH are relative to BaP (with a TEF value of 1).
TEQ	Toxic equivalence. In this report it is the product of a TEF factor for a specific compound and its concentration. The toxicity of a mixture of PAH compounds is expressed in a single number, TEQ for PAH, as the sum of all products of the concentration and TEF values for all individual PAHs.

Summary

The objective of this project was to determine the effect on the exposure and health of the population in the metropolitan area of Stockholm of a change in diesel fuel quality of the road traffic, from Swedish Environmental Class 1 (MK1) to European EN 590 (MK3, called EN590 below). The project has been financed by the Swedish Transport Administration and is part of an assignment by the Swedish Government to assess the effects of an introduction of European EN 590 diesel in Sweden.

The air pollution exposure of the population in Greater Stockholm area (1 628 528 people) due to road traffic emissions have been calculated for a scenario with all diesel being MK1 and a scenario with all diesel being EN590. The models and the emission database of the Stockholm and Uppsala Air Quality Management Association were used. Emission factors were obtained from i) a literature study, ii) The Handbook Emission Factors for Road Transport (HBEFA version 3.1), iii) chassis dynamometer measurements on two trucks by AVL MTC Motor Test Center AB (Haninge) and, iv) chassis dynamometer measurements on cars by Ecotraffic ERD AB (Nacka). Total emissions from road traffic (including gasoline fuelled vehicles emissions) and population exposures were calculated for 2010, 2015 and 2020, with vehicle composition obtained from the Swedish Transport Administration.

For NOx, exhaust-PM, sum of 18 PAHs, benzo[a]pyrene, dibenzo[a,l]pyrene, acrolein and 1,3-butadiene, total road traffic emissions will decrease from 2010 to 2020. Replacing Swedish MK1 diesel with European EN590 will lead to higher emissions for all these species during all years (2010, 2015 and 2020). For some compounds both heavy duty diesel vehicles and light duty diesel vehicles have higher emissions with EN590 as compared to MK1. In some cases differences in emissions are insignificant or there is not enough information to judge if there is a difference. An example of the latter is benzene, for which results from different tests show very small differences compared to the variability between measurements using the same diesel quality.

In most cases, the difference in population exposure with MK1 and EN590 diesel is small compared to the total concentrations due to all road traffic emissions; a few percent. Highest impact is seen for exhaust-PM with an increase of 17%, 15% and 12% of the total local road traffic contribution when EN590 is used in 2010, 2015 and 2020, respectively.

The impact of the increased exposure on cancer risk and mortality among the population of Greater Stockholm, is estimated using different methodologies to illustrate the uncertainties in such estimates. With NOx as indicator of mortality due to road traffic exhaust, and relative mortality risks (RR) from an epidemiological study in Oslo, the number of premature deaths is estimated to increase by 36, 23 and 15 per year for 2010, 2015 and 2020, respectively. Using exhaust-PM as indicator for exposures and RR for PM2.5 from a study in Los Angeles, the number of deaths would increase by 6, 3 and 2

for 2010, 2015 and 2020, respectively. The total number of cancer incidents due to increased PAH exposures is estimated to be 53, 39 and 29 in 2010, 2015 and 2020.

The estimates using NOx and PM2.5 as indicators of mortality should not be added, but considered as different ways to estimate the impact of increased traffic exhaust exposure on mortality. The number of premature deaths based on the study in Oslo (using NOx as indicator) is based on all-cause mortality, including deaths in lung cancer that may have been due to exposure to PAH from road traffic emissions. If all of the cancer incidents due to PAH would be lung cancer (with low rate of survival), one would expect that the estimated number of deaths based on NOx to be much higher, since there are many more cancer incidents due PAH than premature deaths with NOx as indicator. The reason for the difference is not known, but it may be that NOx is not a good indicator in this case or simply because of uncertainties involved in the different methodologies of estimating mortality. But, despite these uncertainties, it may safely be concluded that the replacement of all MK1 diesel with European EN 590 is expected to have significant impact on the health of the population, with effects on both cancer incidence and cardiovascular mortality. In addition, there may be significant effects on respiratory and cardiovascular morbidity, but this has not been assessed in this study.

Background and objectives

Sweden has introduced environmental classification of diesel fuels and from early 1990s the cleanest of those diesel fuels, Swedish Environmental Class 1 (MK1) – according to SS 15 54 35, have dominated the Swedish market. Sales of MK1 diesel has benefited from lower tax compared to the European diesel EN 590. But the situation may change drastically if the differentiation in taxation is removed. Even though, the European diesel quality has improved dramatically over the years a replacement of the Swedish MK1 diesel with European diesel may lead to higher emissions of some air pollutants. In Sweden typical aromatic contents in diesel are 3 vol% to 5 vol%, whereas in European diesel the aromatic content is in the range of 15 vol% to 30 vol% (Danielsson and Erlandsson, 2010a). For 2-ringed PAH and larger in MK1 is 0.1% - 0.5% (EN590 specifies a maximum of 8%), while 3-ringed PAH and larger in EN 590 is around 0.2% – 0.7% (MK1 specifies maximum 0.02%).

The objective of the report is to determine the effect on the exposure and health of the population in the metropolitan area of Stockholm of a change of the diesel fuel quality from MK1 to En 590. This work is financed by the Swedish Transport Administration and is part of an assignment by the Swedish Government to assess the effects of an introduction of European EN 590 diesel in Sweden.

Methods

The emission and dispersion module of the Aiviro air quality management system (SMHI, Norrköping, Sweden; http://airviro.smhi.se) of the Stockholm Uppsala Air Quality Management Association (www.slb.nu/elvf) has been used for calculating the emissions and exposure concentrations for the Metropolitan area of Stockholm (greater Stockholm).

Air quality modeling

The annual mean concentrations were calculated using a wind model and a Gaussian air quality dispersion model part of the Airviro system. Meteorological conditions were based on a climatology that was created from 15 years of meteorological measurements (15 minute averages) in a 50 meters high mast located in the southern part of Stockholm. The climatology consists of a list of hourly events, each of them with a certain frequency of occurrence, which together will yield a distribution of different weather conditions that is similar to the distribution of the full scenario period. We have used a scenario that consist of 60 wind direction classes with 6 stability classes within each wind sector, making a total of 360 hourly events. The wind field for the whole model domain was calculated based on the concept first described by Danard (1976). This concept assumes that small scale winds can be seen as a local adaptation of large scale winds (free winds) due to local fluxes of heat and momentum from the sea or earth surface. Any non-linear interaction between the scales is neglected. It is also assumed that the adaptation process is very fast and that horizontal processes can be described by non-linear equations while the vertical processes can be parameterized as linear functions. The large scale winds as well as vertical fluxes of momentum and temperature are estimated from profile measurements in one or several meteorological masts (called principal masts). For the model domain analyzed in this study (35x35 km^2) only one principal mast is used. This is located in the southern part of the city. Topography and land use data for the Danard model are given by 500 meter resolution. Since the topography of Stockholm is relatively smooth, without dominating ridges or valleys, the free wind can be assumed to be horizontally uniform in the whole domain.

The dispersion calculations were performed on a 100 meter resolution (122 500 receptor points). Individual buildings and street canyons are not resolved but treated using a roughness parameter (similar to the treatment used by Gidhagen et al., 2005). In an open area the calculation height is 2 m above ground level. Over a city the simulation will reflect the concentrations at 2 meters above roof height. A special treatment of the Gauss model plume length is introduced to avoid unrealistically long plumes. This length depends on the stability and persistency of weather conditions. A detailed description of the model is given in the Airviro User Documentation (SMHI, 1993). Chemical and physical transformation processes of particles as well as dry and wet deposition were neglected in the model calculations of annual mean PM10 and number concentrations. See also Gidhagen et al. (2005) regarding the influence of dry deposition and coagulation on particle number concentrations over the urban scale of Stockholm.

Population exposure

Population weighted exposure concentrations are obtained by multiplying the calculated concentrations with the number of people in each 100x100 m grid square, summing all products and dividing by the total population. This procedure has been used in several earlier studies (Johansson, 2007; Johansson et al., 2009). Population data for 2010 at 100x100 m resolution (exactly the same grid cell areas as for the dispersion model calculation) has been obtained from Statistics Sweden. The data refer to the residential addresses and the total number of people in the area is 1 628 528.

Emission scenarios

Temporally and spatially resolved emissions are calculated for each hour based on the emission inventory of the Stockholm and Uppsala Air Quality Management Association (http://www.slb.nu/lvf) (Johansson et al., 1999). Vehicle composition and emission factors for MK1 and EN 590 diesel (hereafter called EN590 diesel) are tabulated in Appendix 1.

Emission from diesel passenger cars and light duty vans

There are no measurements (at least found within this project) on light duty diesel vans. In Stockholm light duty vans comprise around 15% of the vehicle transports and most of them are diesel, so switching from MK1 to EN590 in these vehicles may also be important to consider. In this report we have simply assumed the same effect on emissions from light duty vans as due to emissions from diesel passenger cars. The emissions due to MK 1 or EN590 diesel fuel for light duty vehicles (LDVs, passenger cars and light duty vans) is based on a literature review by Eriksson (2012) and measurements on some passenger cars as described in detail by Eriksson et al. (2012). In summary we conclude the following.

Pre-Euro through Euro 3

- ca 16% higher exhaust PM emissions with EN590
- ca 5% higher 1,3-butadiene emissions with EN590
- ca 250% higher B(a)P emissions with EN590

Euro 3 (DPF[1])

- ca 26% higher exhaust PM emissions with EN590
- 11% higher 1,3-butadiene emissions with EN590
- 16% lower B(a)P emissions with EN590

and Euro 4 (DPF)

- ca 26% higher exhaust PM emissions with EN590
- ca 25% higher emissions of 1,3-butadiene with EN590
- 16% lower B(a)P emissions with EN590

Euro 5 (DPF) and Euro 6 (DPF)

- ca 35% higher exhaust PM emissions with EN590
- ca 63% higher 1,3-butadiene emissions with EN590
- ca 54% lower B(a)P emissions with EN590

For NOx, CO and HC there are no significant differences from light duty diesel vehicles. DB(a,l)P emissions with EN590 diesel in light duty vehicles has not been measured. There are no acrolein measurements, only total aldehyde emissions have been reported. The total aldehyde emissions increased with 2% to 12 % with EN590 (the higher number for measurements with an oxidation catalyst). For benzene different tests show very small differences compared to the variability between measurements using the same diesel quality.

Emissions from heavy duty vehicles (HDV)

For differences in emissions due to MK 1 or EN590 diesel fuel for HDVs the following conclusions have been made:

Pre-Euro through Euro III (Danielsson and Erlandsson, 2010a and 2010b)

- No significant differences for CO and hydrocarbon emissions
- Ca 10% lower emission of NOx with MK1 diesel compared to EN590

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[1] DPF = Diesel particle filter.

- Between 20%-30% lower emissions of exhaust particles with MK1 compared to EN590
- Between 40%-90% lower emissions of PAH with MK1 diesel compared to EN590

Euro V (Almén, 2012)

- No significant differences for CO and hydrocarbon emissions
- For NOx and exhaust-PM there are significant differences depending on after-treatment (EGR[2] or SCR[3]).
- No significant differences for aldehydes, olefines and particle number
- Significant differences in PAH emissions

Euro IV och Euro VI

There are no data at hand for these vehicles. Euro VI correspond to Euro V with SCR, but Euro VI also has particle filter.

For all HDV cold start and hot engine emission factors are weighted as 10% cold start and 90% hot engine.

Vehicle composition

Emissions are calculated for 2010, 2015 and 2020 by weighting the vehicle fleet composition and emission factors for different Euro classes with or without different diesel particle filters (Appendix 1 and Figure 1). The vehicle fleet composition is based on information from the Swedish Transport Administration and HBEFA 3.1.

For heavy duty vehicles Euro III engines dominate in 2010 with about one third of the HDV fleet. In 2015 Euro V with SCR is expected to dominate with ca 44 % and in 2020 Euro VI with diesel particle filter (DPF) dominate the transports in the HDV fleet.

For light diesel passenger cars Euro 4 with DPF dominate in 2010 with 55% of the vehicle fleet transports. In 2015 Euro 5 with DPF is expected to account for about 60% of the fleet and in 2020 Euro 6 with DPF is expected to account for about 60% of the fleet.

For light duty vans and busses Euro 3 and Euro 4 vehicles account for ca 60% of the vehicle transports in 2010. For 2015 and 2020 ca 60% of the transport is made by Euro 5 and Euro 6 vehicles, respectively. In both cases equipped with particle filters.

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[2] EGR = Exhaust Gas Recirculation.

[3] SCR = Selective Catalytic Reduction

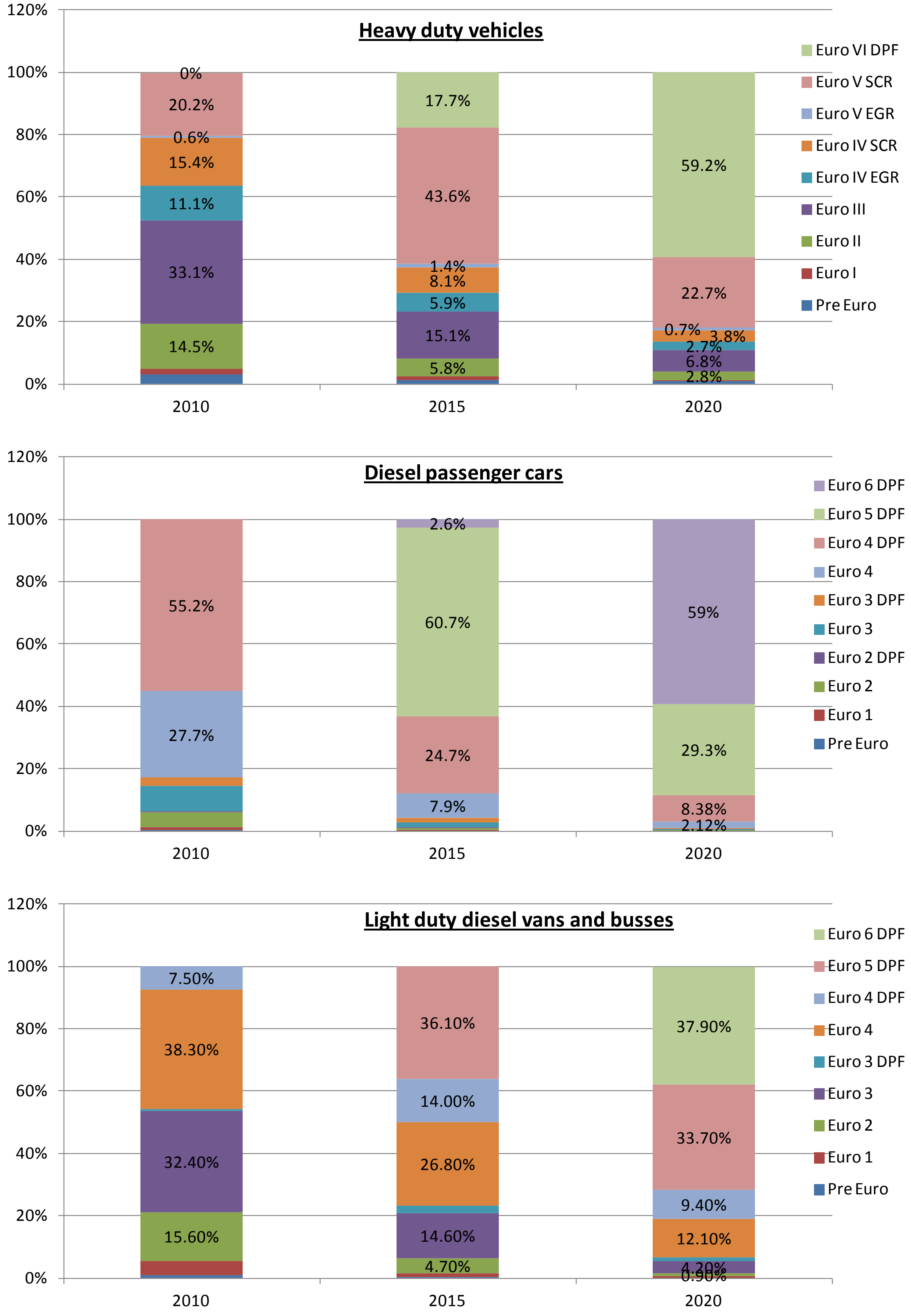

Figure 1. Heavy duty and light duty vehicle fleet according to European emission standards in 2010, 2015 and 2020. Percentages correspond to share of total transports in the Greater Stockholm area. DPF = diesel particle filter; SCR = selective catalytic reduction; EGR = exhaust gas recirculation.

Results

Emissions

The road traffic exhaust emissions from HDV and LDV vehicles of PM, NOx, PAH (sum of 17 different PAH[4]), benzo[a]pyrene (BaP), dibenzo[a,l]pyrene (DB(al)P) in the Greater Stockholm area in 2010, 2015 and 2020 with only MK1 or EN590 diesel are shown in Figure 2. All data is also provided in tables in Appendix 2.

For most compounds the emissions increase with European EN590 diesel compared to MK1 diesel. Largest relative increase is seen for PAH from HDV's, with almost a factor of 4 higher emissions with EN590 compared to MK1 diesel. The most potent carcinogens, BaP and DB[a,l]P, increase with up to 145%.

Figure 2 also shows that the total emissions decrease from 2010 to 2020. This is due to the higher emissions from old vehicles that are being replaced by new vehicles. But for all years, using EN590 diesel lead to increased emissions compared to MK1 diesel, despite cleaner vehicle fleet. But since the absolute emissions are decreasing, the impact on exposure and health of the population is decreasing.

In some cases the difference in emissions with EN590 and MK1 is not significant, for example for NOx emissions from LDV's. The measurements by AVL/MTC on two diesel engines showed very low emissions of 1,3-butadiene and the data was not considered to be reliable. For DB[a,l]P there are no data on differences in emissions from MK1 and EN590 from LDV's.

For PM exhaust, NOx and total PAH, the total diesel vehicle emissions from LDV and HDV comprise a large fraction of the total emissions from road traffic, i. e. emissions including all fuels. However, for BaP and DB[a,l]P, gasoline vehicle emissions may be important[5] (se Westerholm et al., 2012). This means that the impact on the total emissions of changing from MK1 to EN590 will be large for PM exhaust, NOx and total PAH, but relatively small for BaP and DB[a,l]P.

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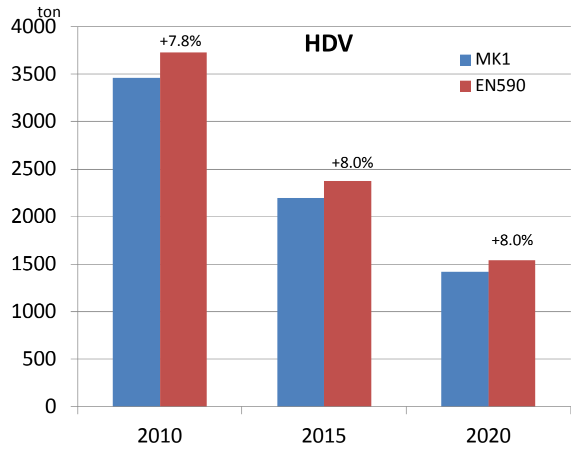
[4] All PAHs are the sum of the amount in particulate and gaseous phase and it includes phenanthrene, 3-methylphenanthrene, 2-methylphenanthrene, 2-methylanthracene, 9-methylphenanthrene, 1-methylphenanthrene, 9-methylanthracene, fluoranthene, pyrene, 1-methylfluoranthene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo[a]pyrene, dibenzo[a,l]pyrene, dibenzo(a,e)pyrene, dibenzo(a,i)pyrene and dibenzo(a,h)pyrene

[5] It should be noted that there are significant uncertainties regarding the emission factors for BaP and DB(al)P from gasoline vehicles. We have validated our estimates based on comparison with total emission factors calculated for Hornsgatan in Stockholm (Westerholm et al., 2012)

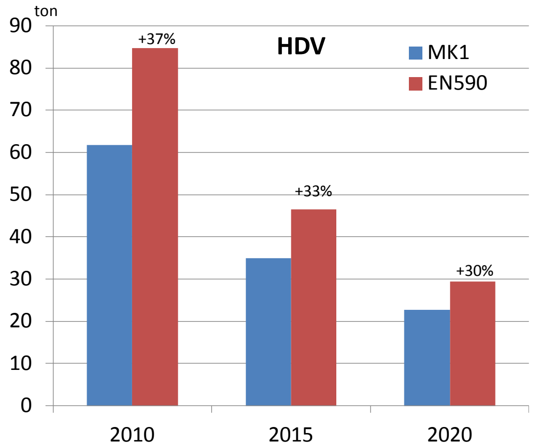
NOx

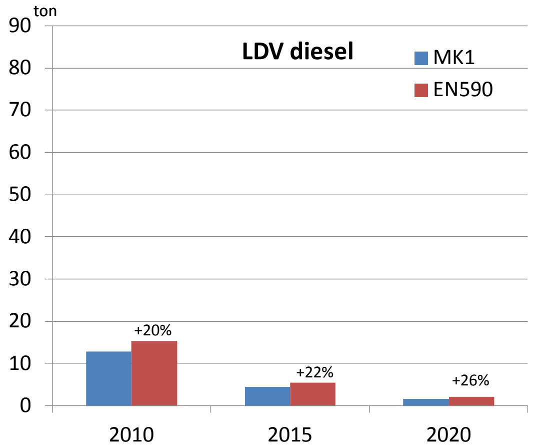


LDV:

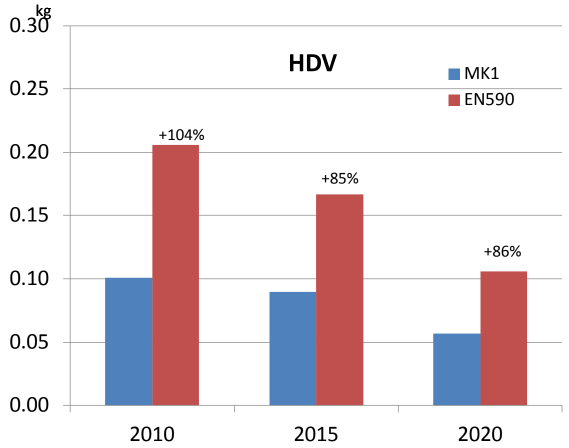
No significant difference

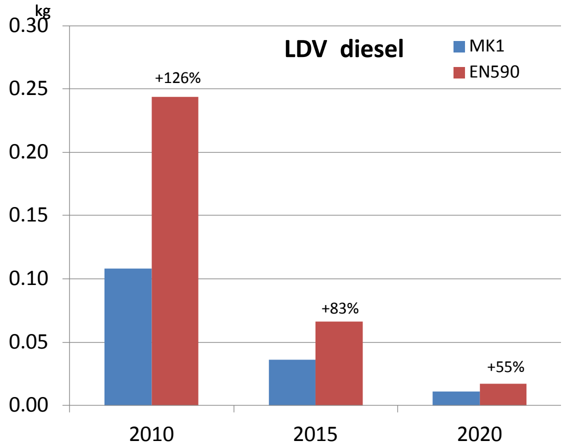
Exhaust PM



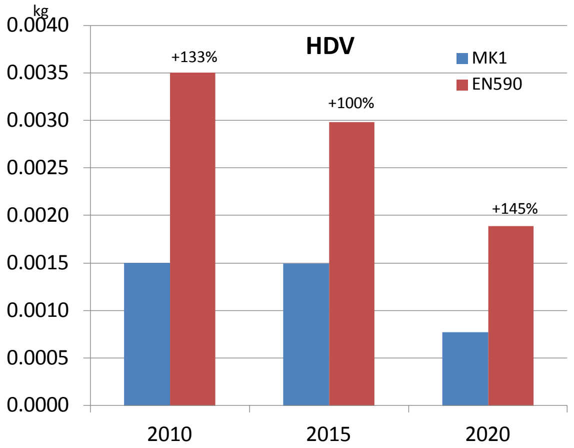


Benzo[a]pyrene





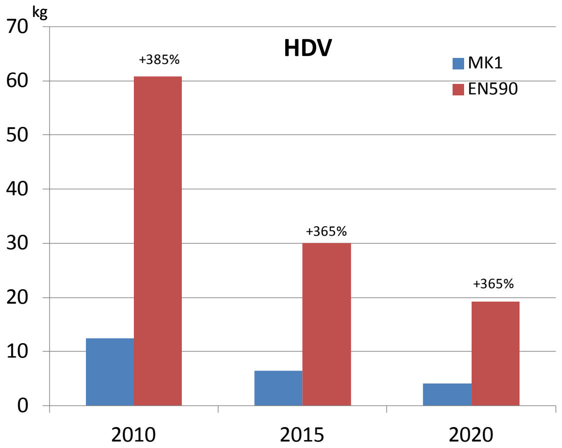
Dibenzo[a,l]pyrene



LDV:

No data

Sum of 17 PAH



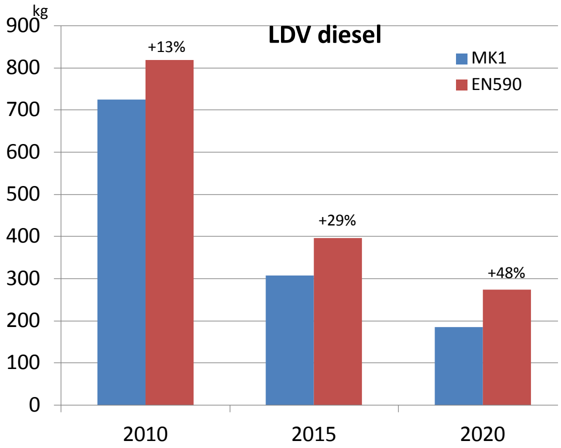
LDV:

No significant difference or lack of data

1,3-butadien

HDV:

No significant difference (lack of data)



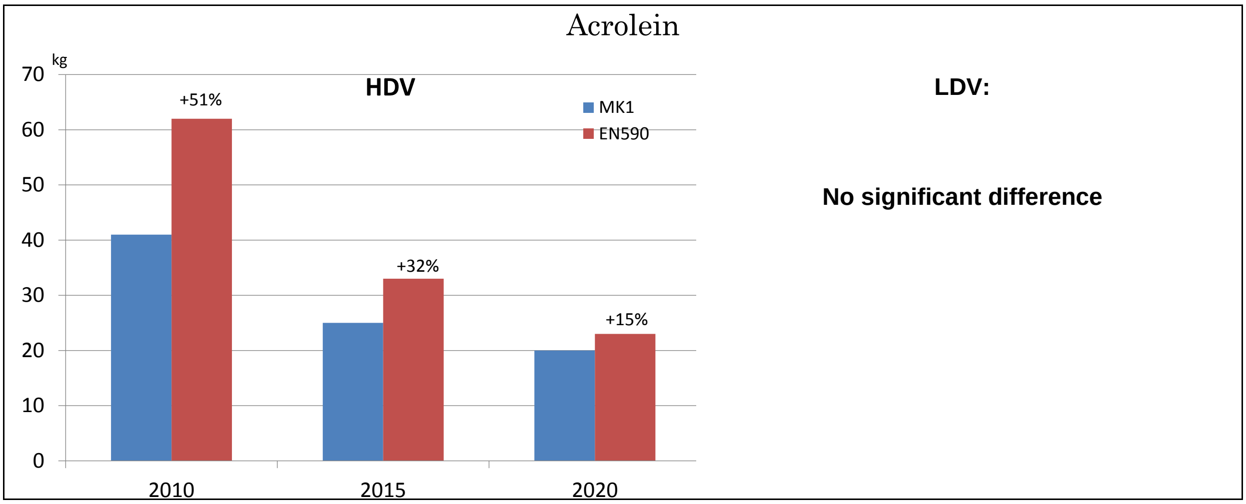


Figure 2. Total emissions from HDV and LDV diesel vehicles in the greater Stockholm region 2010, 2015 and 2020 with diesel as MK1 or EN590. The percentage increase with EN590 relative to MK1 is indicated above each bar.

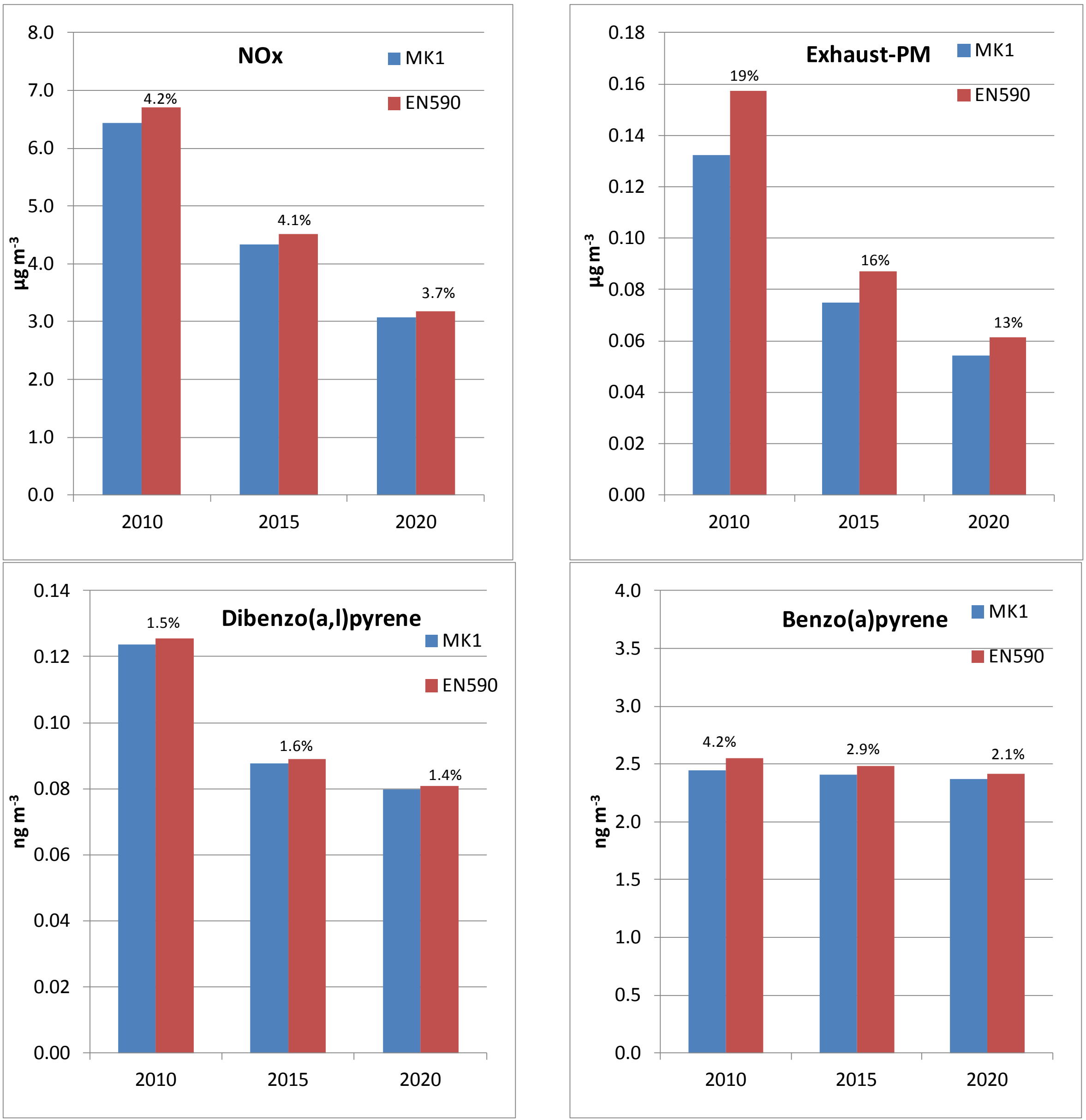
Population weighted exposure

In this section we present population exposure concentrations due to the emissions from road traffic in the Greater Stockholm area. The differences in population weighted exposure concentrations between the situation with diesel fuel being MK1 or EN590 are generally small, <5% difference (Figure 3). The concentrations are due to emissions from all fuels (mainly diesel and gasoline), but the differences are due to the diesel being either MK1 or EN590 diesel.

For **NOx** the exposure concentrations increase with 0.27, 0.18 and 0.11 μg m^{-3} for 2010, 2015 and 2020, respectively. This corresponds to an increase of between 3.7% and 4.2% of the total contribution from road traffic emissions with EN590 compared to MK1. The increased exposure is due to increased emissions from HDV's.

Largest relative increase is seen for **exhaust PM** with 19%, 16% and 13% higher exposure concentrations in 2010, 2015 and 2020, respectively. But as can be seen from Figure 3 the absolute concentration increases are very small; 0.025, 0.012 and 0.07 μg m^{-3} for 2010, 2015 and 2020, respectively. Most of this increase in exposure is due to increased emissions from HDV's.

For **DB[a,l]P** and **B(a)P** the increases in exposure concentrations are very small both in relative and absolute terms. For DB(a,l)P, only HDV diesel emissions contribute to the increase with EN590, but it cannot be ruled out that also LDV diesels may have higher emissions with EN590 compared to MK1 diesel. For B(a)P both HDV and LDV diesel emissions contribute to the increase using EN590 compared to MK1 diesel. For **PAH** we have calculated a TEQ weighted exposure concentration (the exposure concentrations are multiplied with toxic equivalence factors (TEF) for different substances to obtain the toxic equivalents (TEQs) that are then summed to get a total PAH-TEQ concentration). The increases in PAH-TEQ exposure concentrations include only the contribution from HDV and they are small both in relative and absolute terms (see Figure 3).



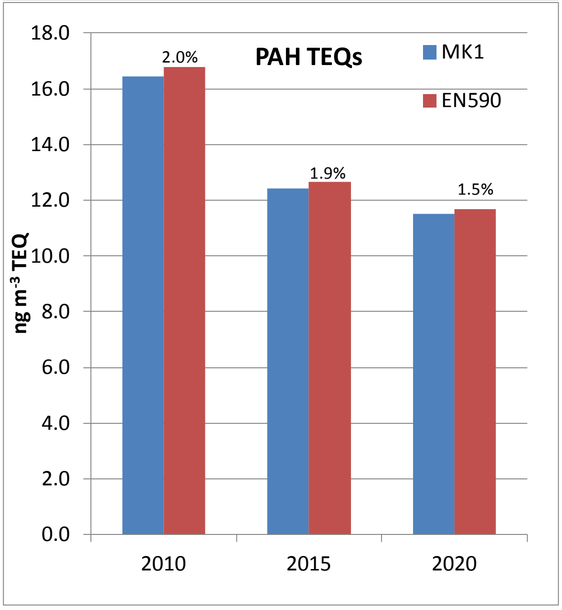


Figure 3. Population weighted exposure concentrations NOx, exhaust-PM, dibenzon(a,l)pyrene, benzo(a)pyrene and PAH-TEQ due to the road traffic emissions (all fuels) for the total population in the Greater Stockholm area in 2010, 2015 and 2020 with only MK1 or only EN590 diesel fuel. The percentages indicate the increase in exposure with EN590 compared to MK1.

For **acrolein** the impact on the exposure due to higher emissions from EN590 decrease from around 40 pg m^{-3} in 2010 to 8 pg m^{-3} in 2020 (Figure 4). All increase calculated is due to increased emissions from HDV's.

For **1,3-butadiene** we have estimated the increased exposure due to increased emissions from LDV's when EN590 is used instead of MK1 diesel. Exposure concentrations are estimated to increase with 103, 98 and 97 pg m^{-3} for 2010, 2015 and 2020 (Figure 4). All increase is due to LDV's. For HDV's, emission measurements found are not reliable or show large variability.

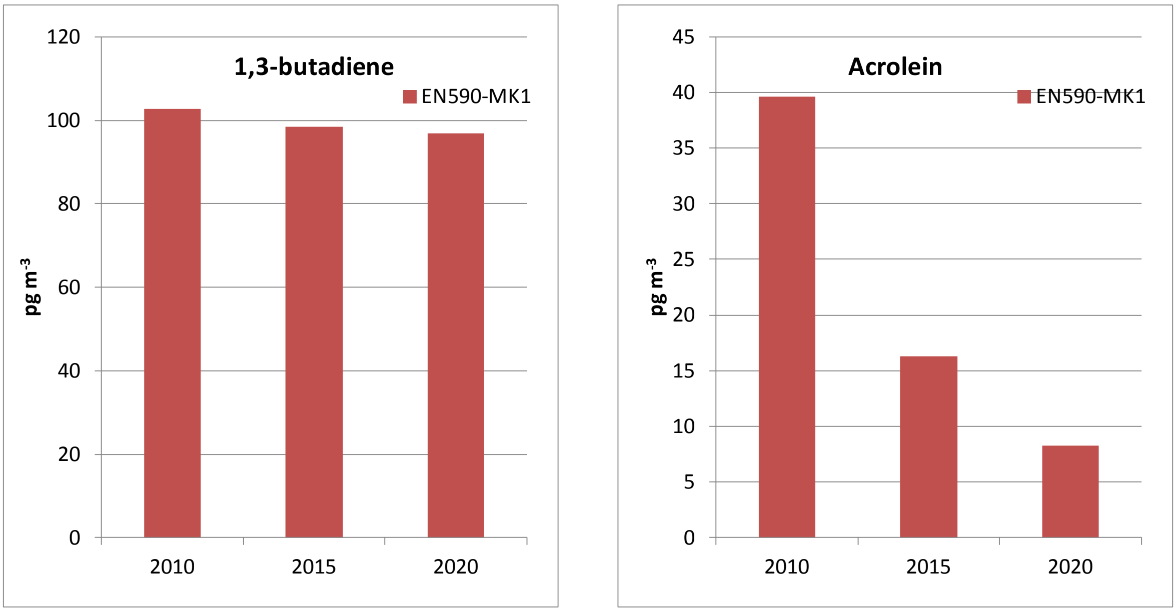


Figure 4. Increases in population weighted exposure concentrations with only EN590 compared to MK1 diesel fuel for 1,3-butadiene and acrolein. In the case of acrolein the increase is due to emissions from HDV's and in the case of 1,3-butadiene the increase is due to LDV's. The exposure refers to the total population in the Greater Stockholm area in 2010, 2015 and 2020.

In Table 1 measured total concentrations in central Stockholm (Torkel Knutssonsgatan) are compared with the calculated contribution to the concentration due to the emissions from road traffic only, with MK1 diesel. The comparison is made for the year 2010. It is seen that the calculated absolute differences in concentrations between the case with MK1 and EN590 diesel are very small compared to the total measured concentrations. The difference in NOx concentration is only 3% of the total concentration and only 0.3% and 0.10% for DB[a,l]P and BaP, respectively.

Table 1. Comparison between the measured total concentrations in central Stockholm (Torkel Knutssonsgatan) and the calculated contribution to the concentration due to the emissions from road traffic only, with MK1 diesel. The comparison is for the year 2010.

	NOx µg/m^3	**DB[a,l]P pg/m^3**	**BaP pg/m^3**
Measured concentration in rural background	2.8	0.70	48.3
Measured total concentration in central Stockholm (Torkel Knutssonsgatan)	15.5	0.96	56.8
Calculated contribution due to road traffic emissions with MK1 diesel	10.9	0.23	4.6
Absolute difference in concentration between road traffic contributions with MK1 and EN590	0.40	0.030	0.059
Difference in concentration between road traffic contributions with MK1 and EN590 relative to the total measured concentrations	3 %	0.3 %	0.10 %

Health impacts

Exposure response factors (relative risks per µg m^{-3} increase in exposure concentration, RR) from different epidemiological studies were used to estimate the health impacts. The studies present effects on mortality among large populations due long term (decades) exposure to road traffic using different exposure indicators. Table 2 shows calculated effects on mortality in the population of the greater Stockholm area due to increased exposure when EN590 diesel is used instead of MK1 diesel using NOx as indicator based on a Norwegian study by Nafstad et al. (2004). The number of deaths is estimated to increase by 36, 23 and 15 per year for 2010, 2015 and 2020, respectively. The problem with using NOx as indicator for exposures to diesel exhaust emission is that the relation between the emissions of exhaust particles (which may be the real contributing factor to the cause of death) and NOx has changed during recent years, and is not the same as it was during the exposure of the people included in Nafstad's study. Exhaust particle emissions has decreased much more rapidly than NOx emissions, especially for diesel vehicles. The relation between NOx and PM in diesel exhaust of the vehicle fleet in Stockholm is changing as more diesel vehicles with particle filters are introduced. This makes the use of the RR for NOx from the study of Nafstad et al. uncertain for estimating effects on the mortality of the population.

For particulate matter, PM2.5 or PM10 are often used as exposure indicator, but it has not been possible to separate the effect of different sources (Anderson, 2009). Applying a RR of 17% per 10 µg m^{-3} PM2.5 as suggested by Jerret et al. (2004), the number of deaths would increase by 6, 3 and 2 for 2010, 2015 and 2020, respectively.

The study by Jerret et al. (2004), where PM2.5 was used as indicator, do not reflect purely exhaust particle exposure, but also other sources and to some extent secondary particle components. It is therefore likely that the effects are underestimated using the RR of Jerret et al., but they may be overestimated using the RR of Nafstad et al. The ranges found for number of premature deaths; 6-36, 3-23 and 2-15 for 2010, 2015 and 2020, respectively, may be interpreted as reflecting the uncertainty of the estimated effects.

Table 2. Calculated population weighted annual mean concentrations of NOx and exhaust particles in 2010, 2015 and 2020 for the situation with only MK1 and EN590 diesel, respectively. The table also shows the difference in exposure with MK1 and EN590 and the number of extra premature deaths per year that would result if MK1 diesel would be replaced by EN590 diesel in the whole vehicle fleet. The annual death rate is 1013 per 100 000 inhabitants.

	NOx			**Exhaust-PM**		
Year	2010	2015	2020	2010	2015	2020
Population weighted concentration with MK1 diesel (µg/m^3)	6.43	4.33	3.07	0.132	0.075	0.054
Population weighted concentration with EN590 diesel (µg/m^3)	6.71	4.51	3.18	0.154	0.086	0.061
Difference in population weighted concentration (with EN590 minus MK1) (µg/m^3)	0.27	0.18	0.11	0.025	0.012	0.007
Relative risk of mortality	8 % per 10 µg NOx/m^3 (Nafstad et al., 2004)			17 % per 10 µg PM/m^3 (Jerret et al., 2004)		
Number of extra deaths per year with EN590 diesel compared to MK1	36	23	15	6	3	2

Effect of exposures on cancer

The estimates using NOx and PM2.5 as indicators should reflect the overall effect on the mortality of the population after long-term exposure to the complex mixture of road traffic exhaust. There are also some specific compounds for which cancer risk estimates have been calculated (Hanberg et al. (2012).

To estimate number of cancer cases (mainly lung cancer) due to exposure to PAH we multiplied the exposure concentrations with toxic equivalence factors (TEF) for different substances to obtain the toxic equivalents (TEQs). The total TEQ for all 10 PAH is obtained by summing the TEQs of the individual compounds. TEFs for different compounds are taken from the report by Hanberg et al. (2012). Table 3 presents calculated TEQs for dibenzo[a,l]pyrene, benzo[a]pyrene and the sum of 10 PAH compounds due to road traffic emissions with MK1 or EN590 diesel fuel.

Table 3 shows the number of extra cancer cases due to increased long term exposure if MK1 diesel is replaced by EN290 diesel. For the sum of all PAHs we estimate 53, 39 and 29 cancer cases for 2010, 2015 and 2020, respectively. The estimation shows that the majority of the cancer cases are due to exposure to the highly carcinogenic PAH dibenzo[a,l]pyrene (Table 3). Butadiene exposure does not contribute to any extra cancer cases. Acrolein may be irritating to the nasal and bronchiolar epithelia, but is not classified as carcinogenic to humans (see Hanberg et al., 2012).

Table 3. Exposure concentrations (ng/m^3) and number of cancer cases due to exposure of road traffic emissions with MK1 and EN590 diesel in the Greater Stockholm area 2010, 2015 and 2030. The toxic equivalents (TEQs) for the PAHs are obtained by multiplying calculated population weighted annual mean concentrations with toxic equivalent factors (TEFs) (see Hanberg et al., 2012).

	Year	TEQ for MK1 case	TEQ for EN590 case	Difference between EN590 and MK1	Number of extra cancer cases with EN590 compared to MK1
Dibenzo[a,l]pyrene	2010	12.4	12.6	0.19	31[2]
	2015	8.77	8.91	0.14	23[2]
	2020	7.98	8.08	0.11	18[2]
Benzo[a]pyrene	2010	2.44	2.55	0.1	17[2]
	2015	2.41	2.48	0.07	11[2]
	2020	2.37	2.42	0.05	8[2]
Sum of 10 PAH[1]	2010	16.5	16.8	0.33	53[2]
	2015	12.4	12.6	0.24	39[2]
	2020	11.5	11.7	0.18	29[2]
1,3-butadiene	2010	na	na	0.10	<0.01[3]
	2015	na	na	0.098	<0.01[3]
	2020	na	na	0.097	<0.01[3]
Acrolein	2010	na	na	40	na[4]
	2015	na	na	16	na[4]
	2020	na	na	8.3	na[4]

[1] All PAHs are the sum of the amount in particulate and gaseous phase and it includes phenanthrene, fluoranthene, pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo[a]pyrene, dibenzo[a,l]pyrene, dibenzo(a,e)pyrene, dibenzo(a,i)pyrene and dibenzo(a,h)pyrene.

[2] Based on a cancer risk of 1 out of 100 000 when exposed for 0.1 ng m^{-3} during a life time and a total population of 1 628 528 people.

[3] Based on a cancer risk of 1 out of 100 000 when exposed for 0.3 µg m^{-3} during a life time and a total population of 1 628 528 people.

[4] Not carcinogenic.

Discussion of health impacts estimates and uncertainties

As shown above the resulting effects of using EN590 instead of MK1 diesel on the mortality among the population in Greater Stockholm differ depending on the methodology used. The effects on mortality due to long term exposures are based on different risk estimates with different indicators of exposure .

NOx has been considered a very good indicator of road traffic exhaust emissions and it has been used to estimate the health benefits in Stockholm of the congestion charging (Johansson et al., 2009). The contributing factor in vehicle exhaust to the increased risk of mortality is not known but the exhaust particles, with its associated toxic compounds, are often considered to be a likely candidate. High correlation between NOx and PM exhaust emissions have previously been a strong argument for using NOx (Gidhagen et al., 2004) as a surrogate for health effects. Based on an epidemiological study in Oslo, Nafstad et al. (2004) found significant associations between calculated long term NOx exposures and total deaths as well as cause-specific death (lung cancer, other respiratory diseases, ischemic heart disease and cerebrovascular disease). The total mortality was found to increase with 8% per 10 µg m^{-3} with a 95% confidence interval of 6% to 11%.

Based on a study in Los Angeles, Jerret et al. (2004), found an increased all-cause mortality of 17% per 10 µg m^{-3} of PM2.5 with a 95% confidence interval of 5%-30%. Jerret et al. (2004) also noted associations between PM2.5 exposure and specific causes of death; ischemic heart disease showing stronger associations than all-cause mortality, and lung cancer showing higher RR than other cancers.

Thus, the studies from Oslo and Los Angeles, estimate the overall effects on mortality of the population. Using the confidence intervals from the study by Nafstad et al. (2004), NOx exposures in Stockholm result in 27 to 50 deaths per year in 2010, and for exhaust PM the confidence intervals of Jerret et al. gives 2-11 deaths per year. The study by Jerret et al. (2004) used PM2.5 as indicator, is likely underestimating the effects on mortality, since chemical analyses showed that a large fraction of the PM was secondary inorganic compounds, expected to be less toxic than fresh exhaust. Also, the estimate based on NOx levels may overestimate the mortality, since the amount of exhaust PM per gram of NOx has decreased.

The cancer risk estimates due to increased exposure to PAHs are based on somewhat different studies, including studies where the specific cancer risk of individual compounds has been estimated. There are large uncertainties in these risk estimates, especially for dibenzopyrenes.

It should also be mentioned that there are substantial uncertainties in the calculated exposures and emissions due to limited information on the emissions from the two diesel fuels, especially for older light duty and heavy duty vehicles. The contributions to the population exposure of emissions from light duty vans, that mainly run on diesel and comprise ca 15% of the vehicle transports, have not been included.

Despite these uncertainties, it may be safely concluded that the replacement of all MK1 diesel with European EN 590 is expected to have significant negative impact on the health of the population. In this report we have focused on mortality effects and cancer incidence, but there may also be significant effects on respiratory and cardiovascular morbidity. In addition, there are susceptible subgroups of the population. For carcinogens (PAH) the risk assessment does not include susceptibility from early-life (fetal and childhood) exposure and may thus underestimate the risk. Cancer risks are generally higher from early-life exposure than from similar exposure durations later in life (US-EPA, 2005). Some of these susceptible subgroups have been included in the estimations of health risks in the present study, but some may not have been included.

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Appendix 1. Vehicle composition and emission factors

Vehicle share trucks and buses

Trucks			
	2010	**2015**	**2020**
Pre Euro	3.1%	1.4%	0.8%
Euro 1	1.9%	1.0%	0.4%
Euro 2	14.5%	5.8%	2.8%
Euro 3	33.1%	15.1%	6.8%
Euro 4 EGR 1)	11.1%	5.9%	2.7%
Euro 4 SCR 2)	15.4%	8.1%	3.8%
Euro 5 EGR 1)	0.6%	1.4%	0.7%
Euro 5 SCR 2)	20.2%	43.6%	22.7%
Euro 6 DPF 3)	0%	17.7%	59.2%

Buses			
	2010	**2015**	**2020**
Pre Euro	0.5%	0.2%	0.2%
Euro 1	2.2%	0.3%	0.3%
Euro 2	24.8%	4.3%	4.3%
Euro 3	36.0%	18.9%	18.9%
Euro 4 EGR	11.5%	8.7%	8.7%
Euro 4 SCR	10.2%	7.7%	7.7%
Euro 5 EGR	2.2%	6.6%	6.6%
Euro 5 SCR	12.6%	37.3%	37.3%
Euro 6 DPF	0%	16.0%	16.0%

1) *EGR - Exhaust Gas Recirculation*
2) *SCR - Selectiv Catalytic Reduction*
3) *DPF – Diesel Particle Filter*

Emission factors trucks and buses

Trucks	MK1								
	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
Pre Euro	0.497	12.42	0.900	2.619	36114	187	3.08	0.112	0.00125
Euro 1	0.358	8.48	0.580	1.873	36114	187	3.08	0.112	0.00125
Euro 2	0.250	11.74	0.453	2.196	36114	187	3.08	0.112	0.00125
Euro 3	0.200	9.42	0.397	2.410	36114	187	3.08	0.112	0.00125
Euro 4 EGR	0.03175	5.73	0.0195	0.1425	36114	187	3.08	0.112	0.00125
Euro 4 SCR	0.03175	5.73	0.0195	0.1425	2319	239	3.80	*0.020*[1]	*0.1485*[1]
Euro 5 EGR	0.03184	3.24	0.0198	0.1420	36114	187	3.08	0.112	0.00125
Euro 5 SCR	0.03184	3.24	0.0198	0.1420	2319	239	3.80	*0.020*[1]	*0.1485*[1]
Euro 6 DPF	0.010	0.61	0.006	0.137	7.47	0.64	0.0043	*0.020*[1]	*0.1485*[1]

[1] *Very uncertain value.*

Trucks	EN590								
	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
Pre Euro	0.695	13.66	0.900	2.619	178877	427	6.83	0.179	0.00182
Euro 1	0.501	9.33	0.580	1.873	178877	427	6.83	0.179	0.00182
Euro 2	0.350	12.91	0.453	2.196	178877	427	6.83	0.179	0.00182
Euro 3	0.280	10.36	0.397	2.410	178877	427	6.83	0.179	0.00182
Euro 4 EGR	0.044	5.16	0.020	0.142	178877	427	6.83	0.179	0.00182
Euro 4 SCR	0.037	6.30	0.0195104	0.1425	3651	391	6.88	*0.013*[1]	*0.1665*[1]
Euro 5 EGR	0.045	2.91	0.0197776	0.14198	178877	427	6.83	0.179	0.00182
Euro 5 SCR	0.037	3.56	0.0197776	0.14198	3651	391	6.88	*0.013*[1]	*0.1665*[1]
Euro 6 DPF	0.010	0.67	0.006	0.137	7.47	0.64	0.0043	*0.013*[1]	*0.1665*[1]

[1] *Very uncertain value.*

Buses	MK1								
	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
Pre Euro	0.681	8.498	2.3116	3.8157	36114	187	3.08	0.112	0.00125
Euro 1	0.278	6.530	0.5244	1.4916	36114	187	3.08	0.112	0.00125
Euro 2	0.139	7.471	0.3596	1.4750	36114	187	3.08	0.112	0.00125
Euro 3	0.143	6.763	0.3362	1.7909	36114	187	3.08	0.112	0.00125
Euro 4 EGR	0.029	4.000	0.0168	0.1472	36114	187	3.08	0.112	0.00125
Euro 4 SCR	0.029	4.000	0.0168	0.1472	2319	239	3.80	*0.020*[1]	*0.1485*[1]
Euro 5 EGR	0.033	2.801	0.0190	0.1741	36114	187	3.08	0.112	0.00125
Euro 5 SCR	0.033	2.801	0.0190	0.1741	2319	239	3.80	*0.020*[1]	*0.1485*[1]
Euro 6 DPF	0.011	0.553	0.0055	0.1719	7.47	0.64	0.0043	*0.020*[1]	*0.1485*[1]

[1] *Very uncertain value.*

Buses	EN590								
	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
Pre Euro	0.953	9.348	2.3116	3.8157	178877	427	6.83	0.179	0.00182
Euro 1	0.389	7.183	0.5244	1.4916	178877	427	6.83	0.179	0.00182
Euro 2	0.195	8.219	0.3596	1.4750	178877	427	6.83	0.179	0.00182
Euro 3	0.201	7.439	0.3362	1.7909	178877	427	6.83	0.179	0.00182
Euro 4 EGR	0.040	3.600	0.0168	0.1472	178877	427	6.83	0.179	0.00182
Euro 4 SCR	0.033	4.400	0.0168	0.1472	3651	391	6.88	*0.013*[1]	*0.1665*[1]
Euro 5 EGR	0.046	2.520	0.0190	0.1741	178877	427	6.83	0.179	0.00182
Euro 5 SCR	0.038	3.081	0.0190	0.1741	3651	391	6.88	*0.013*[1]	*0.1665*[1]
Euro 6 DPF	0.011	0.553	0.0055	0.1719	7.47	0.64	0.0043	*0.013*[1]	*0.1665*[1]

[1] *Very uncertain value.*

EN590/MK1							
	PM	**Nox**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
Pre Euro	1.40	1.10	4.95	2.28	2.22	1.60	1.46
Euro 1	1.40	1.10	4.95	2.28	2.22	1.60	1.46
Euro 2	1.40	1.10	4.95	2.28	2.22	1.60	1.46
Euro 3	1.40	1.10	4.95	2.28	2.22	1.60	1.46
Euro 4 EGR	1.40	0.90	4.95	2.28	2.22	1.60	1.46
Euro 4 SCR	1.15	1.10	1.57	1.64	1.81	*0.63*[1]	*1.12*[1]
Euro 5 EGR	1.40	0.90	4.95	2.28	2.22	1.60	1.46
Euro 5 SCR	1.15	1.10	1.57	1.64	1.81	*0.63*[1]	*1.12*[1]
Euro 6 DPF	1.00	1.10	1.00	1.00	1.00	*0.63*[1]	*1.12*[1]

[1] *Very uncertain value.*

Weighted emission factors for Greater Stockholm

MK1 Trucks	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	0.139	7.548	0.24	1.30	24 035	205	3.3	0.079	0.054
2015	0.076	4.727	0.12	0.65	12 223	181	2.9	0.048	0.077
2020	0.042	2.597	0.06	0.38	5 768	90	1.4	0.033	0.040

EN590 Trucks	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	0.192	8.172	0.24	1.30	116 262	413	6.8	0.119	0.061
2015	0.101	5.123	0.12	0.65	56 483	333	5.6	0.063	0.087
2020	0.054	2.820	0.06	0.38	26 479	165	2.8	0.036	0.046

MK1 Buses	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	0.107	5.755	0.24	1.12	28 404	199	3.2	0.091	0.035
2015	0.056	3.607	0.10	0.54	15 111	181	2.9	0.056	0.068
2020	0.056	3.607	0.10	0.54	15 111	181	2.9	0.056	0.068

EN590 Buses	g/km	g/km	g/km	g/km	ng/km	ng/km	ng/km	mg/km	mg/km
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	0.148	6.225	0.24	1.12	138 899	419	6.8	0.141	0.040
2015	0.074	3.853	0.10	0.54	71 317	342	5.8	0.077	0.076
2020	0.074	3.853	0.10	0.54	71 317	342	5.8	0.077	0.076

	EN590/MK1	Trucks							
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	1.38	1.08	1.00	1.00	4.84	2.02	2.05	1.51	1.13
2015	1.34	1.08	1.00	1.00	4.62	1.84	1.94	1.31	1.13
2020	1.29	1.09	1.00	1.00	4.59	1.82	1.93	1.08	1.14
	EN590/MK1	Buses							
	PM	**Nox**	**VOC**	**CO**	**PAH**	**BaP**	**DB[a,l]P**	**Acrolein**	**Butadiene**
2010	1.38	1.08	1.00	1.00	4.89	2.11	2.11	1.55	1.15
2015	1.33	1.07	1.00	1.00	4.72	1.90	1.98	1.38	1.12
2020	1.33	1.07	1.00	1.00	4.72	1.90	1.98	1.38	1.12

Appendix 2. Road traffic emissions with MK1 and EN590 diesel

Total emissions for Greater Stockholm

2010

	tons	*tons*	*tons*	*kg*	*kg*
	PM	**NOx**	**PAH**	**BaP**	**DB[a,l]P**
MK1	123	6 072	76	1.76	0.115
EN590	156	6 342	124	2.12	0.117
Difference	33	270	48	0.36	0.002
	27%	4.4%	64%	20%	1.7%

2015

	tons	*tons*	*tons*	*kg*	*kg*
	PM	**Nox**	**PAH**	**BaP**	**DB[a,l]P**
MK1	68	4 088	26	1.69	0.0818
EN590	86	4 264	49	1.89	0.0833
Difference	18	176	24	0.20	0.0015
	26%	4.3%	92%	12%	1.8%

2020

	tons	*tons*	*tons*	*kg*	*kg*
	PM	**Nox**	**PAH**	**BaP**	**DB[a,l]P**
MK1	49	2 877	23	1.65	0.0744
EN590	61	2 991	38	1.78	0.0755
Difference	12	114	15	0.13	0.0011
	24%	4.0%	65%	8%	1.5%

Heavy duty vehicle emissions

2010

	tons	*Tons*	*kg*	*kg*	*kg*	*kg*	*kg*
	PM	**Nox**	**PAH**	**BaP**	**DB[a,l]P**	**Akrolein**	**Butadien**
MK1	62	3459	13	0.10	0.0015	41	24
EN590	85	3729	61	0.21	0.0035	62	27
Difference	23	270	48	0.10	0.0020	21	3
	37%	*7.8%*	*385%*	*104%*	*133%*	*51%*	*13%*

2015

	tons	*tons*	*kg*	*kg*	*kg*	*kg*	*kg*
Tunga	**PM**	**Nox**	**PAH**	**BaP**	**DB[a,l]P**	**Akrolein**	**Butadien**
MK1	35	2195	6.5	0.09	0.0015	25	37
EN590	47	2371	30	0.17	0.0030	33	42
Difference	12	176	24	0.08	0.0015	8	5
	33%	*8.0%*	*365%*	*85%*	*100%*	*32%*	*13%*

2020

	tons	*tons*	*kg*	*kg*	*kg*	*kg*	*Kg*
Tunga	**PM**	**Nox**	**PAH**	**BaP**	**DB[a,l]P**	**Akrolein**	**Butadien**
MK1	23	1425	4.1	0.06	0.0008	20	24
EN590	30	1539	19	0.11	0.0019	23	27
Difference	6.8	114	15	0.05	0.0011	3.8	3.1
	23%	*8.0%*	*365%*	*86%*	*145%*	*15%*	*13%*

Vehicle share for diesel passenger cars

	2010	**2015**	**2020**
Pre Euro	0.20%	0.08%	0.05%
Euro 1	1.04%	0.16%	0.05%
Euro 2	4.84%	0.69%	0.09%
Euro 2 DPF	0.40%	0.29%	0.07%
Euro 3	8.05%	1.65%	0.29%
Euro 3 DPF	2.52%	1.28%	0.43%
Euro 4	27.7%	7.9%	2.12%
Euro 4 DPF	55.2%	24.7%	8.38%
Euro 5 DPF		60.7%	29.3%
Euro 6 DPF		2.6%	59%

Emission factors diesel passenger cars

MK1			
	g/km	*mg/km*	*ng/km*
	PM	**Butadiene**	**B[a]P**
Pre Euro	0.031	2	250
Euro 1	0.031	2	250
Euro 2	0.031	2	250
Euro 2 DPF	0.031	2	250
Euro 3	0.031	2	250
Euro 3 DPF	0.027	0.9	250
Euro 4	0.010	0.7	75
Euro 4 DPF	0.008	0.4	75
Euro 5 DPF	0.00047	0.14	0.7
Euro 6 DPF	0.00047	0.14	0.7

EN590			
	g/km	*mg/km*	*ng/km*
	PM	**Butadiene**	**B[a]P**
Pre Euro	0.036	2.1	870
Euro 1	0.036	2.1	870
Euro 2	0.036	2.1	870
Euro 2 DPF	0.036	2.1	870
Euro 3	0.036	2.1	870
Euro 3 DPF	0.034	1.0	210
Euro 4	0.011	0.8	261
Euro 4 DPF	0.011	0.5	63
Euro 5 DPF	0.00064	0.22	0.32
Euro 6 DPF	0.00064	0.22	0.32

Ratio EN590/MK1			
	PM	**Butadiene**	**B[a]P**
Pre Euro	1.16	1.05	3.48
Euro 1	1.16	1.05	3.48
Euro 2	1.16	1.05	3.48
Euro 2 DPF	1.16	1.05	3.48
Euro 3	1.16	1.05	3.48
Euro 3 DPF	1.26	1.11	0.84
Euro 4	1.17	1.13	3.46
Euro 4 DPF	1.26	1.25	0.84
Euro 5 DPF	1.35	1.63	0.46
Euro 6 DPF	1.35	1.63	0.46

Weighted emission factors of diesel passenger cars for Greater Stockholm

MK1			
	g/km	*mg/km*	*ng/km*
	PM	**Butadiene**	**B[a]P**
2010	0.0125	0.708	105
2015	0.0044	0.301	35
2020	0.0016	0.181	11

EN590			
	g/km	*mg/km*	*ng/km*
	PM	**Butadiene**	**B[a]P**
2010	0.0151	0.800	239
2015	0.0054	0.389	64
2020	0.0020	0.268	17

Total emissions from LDV diesel vehicles (incl diesel passenger cars) for Greater Stockholm

2010

	tons	*kg*	*kg*
	PM	**Butadiene**	**B[a]P**
MK1	61	724	1.7
EN590	72	818	1.9
Difference	10	94	0.26
	17%	*13%*	*16%*

2015

	tons	*kg*	*kg*
	PM	**Butadiene**	**B[a]P**
MK1	33	307	1.6
EN590	39	397	1.7
Difference	6.1	90	0.13
	18%	*29%*	*8%*

2020

	tons	*kg*	*kg*
	PM	**Butadiene**	**B[a]P**
MK1	27	185	1.6
EN590	32	274	1.7
Difference	5.0	88.5	0.09
	19%	48%	6%

SLB·analys

SLB analys is a department of the Environment and Health Administration. We inform about current and future air quality and make

- environmental & health assessments in urban planning
- air quality and noise measurements
- air quality dispersion calculations for evaluation of source contributions and population exposures

SLB analys is the operator of the Stockholm & Uppsala Air Quality Management Association (www.slb.nu/elvf)

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