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Feasibility of Introducing Particulate Filters on Gasoline Direct Injection Vehicles

A Cost Benefit Analysis

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1 EXECUTIVE SUMMARY

Starting from September 2011 a limit of 6×10^{11} #/km will be introduced for the type approval of diesel passenger cars that will eventually apply to all new registered diesel passenger cars from September 2012. The same limit will also apply to diesel light duty vehicles but with a one year delay (09/2012 for type approvals and 09/2013 for all new registered vehicles). The regulation states that a Particle Number (PN) limit will also be introduced for the certification of Euro 6 technology gasoline-fuelled vehicles but the threshold value was not decided yet. While conventional Port Fuel Injection (PFI) gasoline vehicles can easily comply with the diesel limit, their Direct Injection (G-DI) counterparts are found to emit systematically above this threshold by up to 1 ½ orders of magnitude. It is therefore expected that application of the diesel particle number limit to G-DI vehicles may necessitate the installation of particulate filters.

At the same time, the penetration of G-DI vehicles is expected to rapidly grow in the near future in both the European and US markets. This is due to their improved fuel efficiency compared to the conventional PFIs, that would potentially provide the manufacturers the means to meet the target set in both EU and USA on the fleet-average carbon dioxide (CO_2) emissions of future vehicles. It is foreseen that this vehicle category will dominate the gasoline market eventually replacing the conventional and less efficient PFI vehicles. There exist concerns however, that their elevated particulate emissions may adversely affect the air quality in the future if no measure is taken to efficiently control them.

In this direction the present study examined the feasibility of introducing Gasoline Particulate Filters in G-DI vehicles and investigated the associated implementation cost and environmental benefit.

The possibility to control the particle emissions from G-DI vehicles using Gasoline Particulate Filters (GPF) was already investigated in several studies available in the open literature (some of which from automotive manufacturers), suggesting that some work is already underway. In order to better assess the feasibility of introducing a GPF in G-DI vehicles, a workshop was organized by DG ENTR in which different catalyst and substrate manufacturers presented, in a confidential manner, technical information on prototype GPF systems they have developed as well as their experience from installation of their systems in commercial G-DI vehicles.

The data presented in the workshop suggested that the application of a GPF in G-DI applications is rather straightforward, in good agreement with published data. The preferable solution is that of a relatively small GPF close coupled to the engine. This configuration will enable passive regeneration under most operating conditions due to the relatively high exhaust temperatures of G-DI engines. The main challenge in this case is to minimize any delay in the catalyst light-off imposed by the thermal mass of the GPF. Alternatively, the GPF can also be installed in an underfloor position in which case there might be some need for active regeneration under urban and repeated start-stop operating conditions. In both cases, active regeneration will most probably be achieved through retarded spark timing and split fuel injection for engines running stoichiometric and by post fuel injection for engines running lean, approaches already employed to heat up the catalyst under cold start. The ultimate target is to replace conventional Three Way Catalysts (TWC) with catalyzed GPF systems, using the same precious metal content. A significant progress was already made in this direction and systems exhibiting similar catalytic activity and oxygen storage capacity with three way catalysts were already developed.

A significant progress was also made in optimizing the backpressure for the target filtration efficiency, either by controlling the porosity of the filters or the type and amount of washcoating. Accordingly, no fuel consumption penalty could be identified under urban and rural driving conditions with some studies suggesting a modest increase occurring under high speed/full load conditions (e.g. 180-200 km cruise), that would result in an approximately 1% increase in the fuel consumption under typical motorway conditions. Yet other studies, including experimental data collected at JRC, suggest even fuel consumption benefits associated to increased rates of internal EGR. This minimal effect of the GPF on the fuel consumption could be related to a certain extent to the common observation that there is hardly any soot accumulating in the GPF. This also illustrates that the GPFs can easily regenerate passively and it is therefore expected that there will be little fuel consumption penalty from active regeneration events. It was estimated that the installation of a GPF would result in an additional total 9-18 kg of fuel consumed over the lifetime of a SPC G-DI vehicle. The corresponding figures for MLPCs and LDVs were estimated to be 13-25 kg and 21-59 kg, respectively. Based on the current tax-free price of petrol (0.55 €/lt), the additional cost introduced to the vehicle owner is calculated to be 6-13 € for SPCs, 9-18 € for MLPCs and 16-43 € for LDVs.

The anticipated low soot loading of the GPFs allows the use of smaller GPF volumes and less complex regeneration strategies compared to diesel applications. This effectively results in lower installation costs. In line with these observations, the Federal Environment Energy of Germany has recently submitted a report (UBA HR-6526, 02/02/2011), supporting the installation particulate filters on G-DI vehicles on the ground that the implementation cost is reasonable and there is limited negative effect on the fuel consumption.

Due to confidentiality issues, very limited information on cost figures were provided by the GPF manufacturers, which however suggests that the bare cost should be similar or even less to that of Diesel Particulate Filters (DPFs) of the same volume. Using information on cost figures from earlier studies on the cost effectiveness of DPF systems and the Indirect Cost (IC) multiplier methodology, we derived estimations on the price increase associated with the installation of a GPF in G-DI vehicles of different engine displacement. This was calculated to be 39 to 163 € for small passenger cars, 56 to 264 € for medium to large passenger cars and 70 to 303 € for light duty vehicles, for implementation at a Euro 6 stage. For comparison, system integration costs estimates provided by AECC suggest a price increase of 44 to 168 € depending on engine size, production volume and packaging options. Recently, the International Council on Clean Transportation has also derived installation cost estimates in the range of 62 to 131 € (which according to the IC methodology correspond to 68-169 € price increase). The same study also quoted cost estimates from the Manufacturers of Emissions Controls Association in the order of 35 to 70 € (i.e., 38-91 € price increase).

The IC methodology also allowed for the estimation of the cost reduction resulting from a delayed implementation of the GPF through optimization of the GPF production and vehicle integration, and due to the much larger production volumes. A three years delay was considered to allow sufficient time for the vehicle manufacturer to optimize the installation of GPF systems in their G-DI fleet. Accordingly, the associated price reductions were estimated to be in the range of 2 to 13% (depending on the complexity of the aftertreatment configuration).

The main reason for the consideration of a three years delay though is related to the increasing evidence suggesting potential compliance with the diesel PN limit through improvements in the combustion process. A prototype G-DI engine utilizing an advanced injection system was already shown to comply with the diesel PN limit through careful calibration of the injection strategy. Hybrid PFI/G-DI engines were also introduced in the market by some manufacturers and some others will follow. While the driving force in the development of such engines was the optimization of the fuel consumption over a wider operating range they also offer the potential to drastically reduce particle number emissions. The emission performance of such designs under real world conditions, especially considering the observed strong dependence of particle emissions on fuel properties, as well as durability issues related to deposits formation in the combustion chamber may be of concern and need to be assessed. A potential reduction of size of the emitted particles resulting in significant fractions lying below 23 nm (i.e. the detector cut-off size of the legislative procedure) may also be a concern requiring detailed investigations. In contrast, GPF systems have the advantage of controlling equally (or even more effectively) the more diffusive sub 23 nm particles and at the same time ensuring efficient control under all driving conditions.

In order to collect more information on the potential offered by internal engine measures in controlling the particle emissions from G-DI vehicles, a special workshop was organized by DG ENTR. A number of vehicle manufacturers presented confidential information on research work undertaken in this direction. All manufacturers were in favour of internal engine measures expressing concerns about the feasibility of GPF installation and the associated fuel consumption penalty and engine performance degradation. All seven manufacturers that participated in the workshop stated that will be in a position to achieve significant particle emission reductions through improvements in the combustion process, provided that an additional three years lead time will be granted. Interestingly, three manufacturers were confident that at least some of their G-DI vehicles will comply with the diesel PN limit without the need for a GPF. One of them claimed that will introduce soon in the market vehicles complying with the diesel PN limit. Others had concerns about the lowest achievable particle emission levels and the emission performance degradation due to fuel deposits. Limited information was provided, in a strictly confidential manner, on the implementation cost associated with such internal engine measures. The input suggests though, that the implementation cost can be considerably lower compared to the incorporation of a GPF in the exhaust.

Overall, it was not possible to assess the cost effectiveness of such engine measures in the present study, due to the limited information on the real world emission performance and the associated incremental cost, discounting for the cost related to engine performance improvements. It became evident though that some manufacturers will manage to comply with the diesel PN limit through such less expensive internal engine measures at least for some of their vehicles in a 2017 timeframe. Effectively, this means that the cost efficiency figures presented in this study for a three years delay in the implementation of the diesel PN limit, correspond to a conservative approach and therefore should rather be considered as high estimates.

Some projections of the total emitted solid Particle Number (PN) in EU27 up to 2030 were also performed using information from the TREMOVE and COPERT models. The contribution of G-DI vehicles in the total emitted solid Particle Number (PN) is projected to sharply increase in the future owing to the expected increased penetration of this vehicle category in the market. The projections suggest that despite the steep increase of ambient PN levels from G-DI vehicles, their absolute levels will exceed those of their diesel counterparts at around 2030, due to the presence of some high-emitting non-DPF equipped diesel vehicles. An installation of a GPF will effectively halt the increase of PN emissions from G-DI vehicles, reducing the emitted levels in 2030 by almost one order of magnitude.

Currently there is no available information on the health benefit in monetary terms associated with a given reduction of solid particle number concentrations. Yet, assuming that the biological path through which solid particles trigger the immune system response is related to the available surface area of the deposited particles in the lungs, the particle number seems to be a more relevant metric from a health effect stand point. The translation of the well established, PM-based marginal external costs to calculate the monetary benefit of reducing PN emissions is accompanied by a relatively large uncertainty due to the large variation in the size distributions of G-DI and diesel exhaust aerosol. Experimental data collected at JRC suggest a PN to PM ratio spanning from 0.6 to 5.4×10^{18} #/km, in good agreement with what numerical simulations with reported size distributions and effective density profiles suggest. Accordingly, a $2.2\pm1.5\times10^{18}$ #/km range was employed in the calculations which centers at a value close to what publications also suggest for G-DI vehicles. Combining these figures with latest estimates on the marginal external costs for PM and considering the relative share of vehicle driving in metropolitan, rural and outside built-up areas of each member state, the EU27-average PN benefit resulting from GPF installation was calculated to be 31-162 € for SPCs, 36-190 € for MLPCs and 25-131 € for LDVs, over the service life of the vehicles.

The GPF is estimated to have an overall neutral global warming effect as the reduction of the black carbon was found to counterbalance the CO_2 penalty. Expressed in monetary terms, the benefit from BC reduction over the service life of the vehicles is estimated to be 0.5-40.5 € for SPCs, 0.5-43.8 € for MLPCs and 0.6-47.4 € for LDVs. The corresponding penalty from the elevated CO_2 emissions is estimated to be 0.5-9.8 € for SPCs, 0.8-13.7 € for MLPCs and 1.4-32.9 € for LDVs.

The results of the presented study are summarized in Figure 1 for each vehicle class. Table 1 shows the corresponding EU27 weighted-averaged figures based on the market share of the different vehicle categories over the 2015-2030 timeframe (44% SPC, 50% MLPC and 7% LDV). Overall, there exists a large uncertainty in the calculated figures. Nevertheless, the calculations provided evidence that the societal benefit offered from the effective control of particle emissions is at least of the same order of magnitude with the implementation cost.

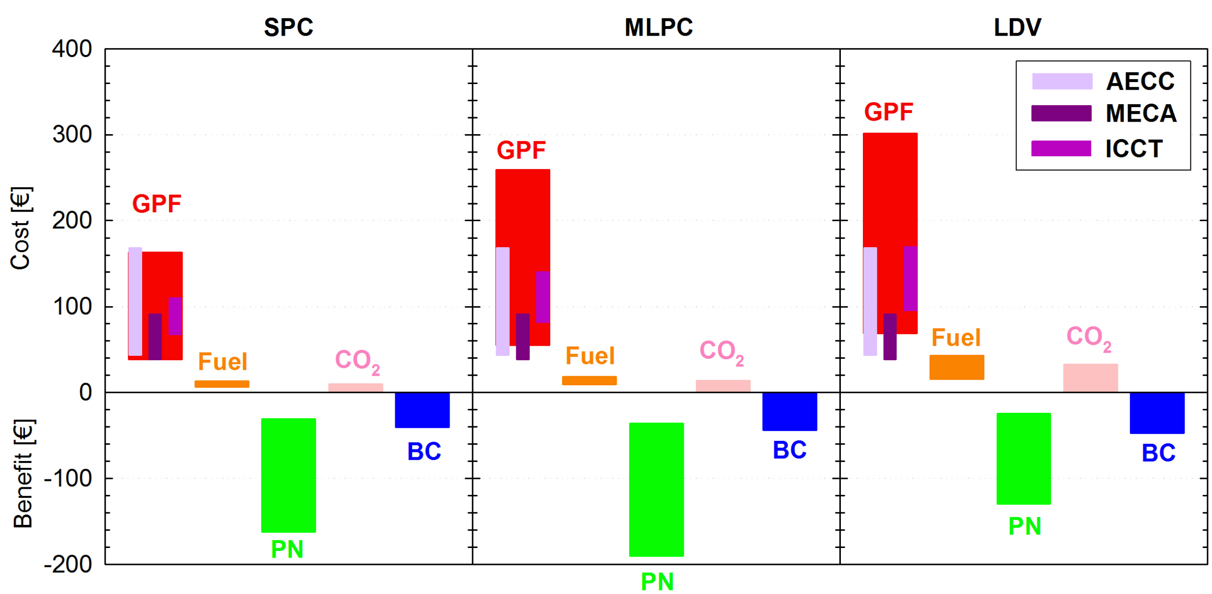

Figure 1: Estimated implementation costs (motorist) and externalities (societal costs) resulting from the introduction of a GPF in G-DI vehicles of different classes. The abbreviations SPC, MLPC and LDV stand for Small Passenger Cars, Medium & Large Passenger Cars and Light Duty Vehicles, respectively.

Table 1: Estimated EU27 sales-averaged implementation costs (motorist) and externalities (societal costs) resulting from the introduction of GPFs in G-DI vehicles.

Vehicle price increase [€]	**48.5 - 222.7**
Fuel penalty [€]	**8.4 - 17.6**
PN benefit [€]	**33.0 - 174.2**
CO_2 penalty [€]	**0.7 - 13.3**
BC benefit [€]	**0.5 - 42.6**

2 INTRODUCTION

2.1 Background

Legislation limiting the pollutant emissions of new registered vehicles is well established in many regions of the world. One pollutant of special concern is Particulate Matter (PM), which is a complex physicochemical mixture of solid and volatile material, ranging in size from a few nanometers to several hundred nanometers. The lack of a robust definition for PM complicates the study of its environmental and health consequences. There is yet no firm opinion as to which components or properties of PM are responsible for detrimental health effects. Nevertheless, several organizations have reviewed the health effects of diesel PM and characterized it as “likely to be carcinogenic” [1, 2, 3] or even “carcinogenic” [4].

Historically, the PM emissions of automotive engines were regulated in terms of mass. Gasoline vehicles were not subjected to regulations due to their relatively low PM mass levels compared to their diesel counterparts. Emission standards limiting the mass of PM from diesel exhaust were first established in 1992 (Euro 1 stage) and gradually became more stringent, with the limit values in 2005 (Euro 4 stage) being set at 80% (passenger cars & light duty vehicles) and 95% (heavy duty engines) of the Euro 1 threshold values. The improvements in diesel PM emissions brought by the progressively tighter emission standards, raised concerns regarding the sensitivity of the traditional gravimetric procedure [5]. At the same time, the Clean Air For Europe (CAFE) study [6] concluded in 2005 that “*significant negative impacts will persist even with effective implementation of current legislation*”. The projected PM levels in Europe were estimated to result in “*a 5.5 months loss in statistical life, or equivalently in 272000 premature deaths*”. In response, the European Commission requested a further tightening of the PM emission standards at a level that would necessitate the mandatory installation of the best technology Diesel Particulate Filters (DPF) to all compression ignition passenger cars [7]. The limit value was provisionally set at 5 mg/km, which corresponded to an 80% reduction over the Euro 4 limit of 25 mg/km, but it was recognized that the gravimetric procedure might not be sensitive enough to discriminate between wall flow Diesel Particulate Filters (DPFs) and flow through particulate filters. Accordingly, it was suggested that the regulated procedure shall be revised and be complemented by a particle number measurement technique drawing from the findings of the Particle Measurement Programme (PMP) that was underway. The shift to a particle number limit also reflects however the growing consensus amongst the health experts that particles in the ultrafine range (smaller than 100 nm), which contribute little to the particulate mass due to their small size, are potentially more toxic and have more adverse health effects on human health [8].

The PMP project was established in 2001 on the initiative of several Member States but it was soon evolved into an international group comprising of Governments (including France, Germany, Greece, Japan, Korea, Sweden, Switzerland and UK), International Institutions (European Commission), industry (associations of car and engine manufacturers, instrument manufacturers), and national vehicle emission laboratories and research institutions. The whole project was conducted under the auspices of the United Nations ECE WP29 GRPE (Working Party on Pollution and Energy) and was managed by a UN-ECE Working Group [9] chaired by the UK Department of Transport.

The mandate given to the PMP Working Group by GRPE was to develop new particle measurement techniques to complement or replace the existing particulate mass measurement, with special consideration to measuring particle emissions at very low levels. PMP was also tasked with accumulating data on the performance of a range of engine/vehicle technologies when tested according to the proposed procedures. The PMP group concluded that a revised filter mass measurement method and a particle number method using a Condensation Particle Counter (CPC) and sample preconditioning to eliminate volatile particles, best met the objectives of the programme.

The proposed PMP methodology was subsequently evaluated in a Light Duty and two Heavy Duty Inter-Laboratory Correlation Exercises (ILCE_LD, ILCE_HD Validation Exercise and ILCE_HD Round Robin). The ILCE_LD [10] and the ILCE_HD Validation Exercise [11] were completed, while the ILCE_HD Round Robin is expected to conclude by the end of 2011. The two concluded studies have proven that the proposed particle number methodology is robust and very sensitive, being capable of quantifying the different emission performance of wall flow Diesel Particulate Filters (DPFs) of different porosities but also the effect of DPF fill state on the filtration efficiency [12]. Following the successful implementation of the ILCE_LD, which verified the superior performance of the particle number methodology compared to the existing and revised gravimetric procedure, the particle number method was introduced in the Light Duty European legislation [13, 14]. The proposed modifications to the particulate mass measurement procedure were also integrated.

Starting from September 2011 (Euro 5b stage), a limit value of 4.5 mg/km and 6×10^{11} #/km, following the PMP procedures, will be introduced for the particulate mass and solid Particle Number (PN) emissions of all diesel passenger car type approvals. Regulation (ECE) No 715/2007 also authorized the Commission to introduce particle number emission limits for gasoline fuelled vehicles. However, at the time of the development of the implementing legislation it was decided that additional information is desirable on the emissions of these vehicles prior to a standard being set. In that respect, the introduction of a particle number limit was postponed at the Euro 6 stage (09/2014) the latest.

JRC has already prepared a report on the emission performance of current technology gasoline vehicles [15], based on information available from the open literature but also dedicated experiments conducted at JRC. The study suggested that while conventional Port Fuel Injection (PFI) vehicles can easily comply with the limit applicable to diesels, this is not the case for Gasoline Direct Injection (G-DI) vehicles. The latter category was found to exceed this threshold by 5 to 25 times over the regulated New European Driving Cycle (NEDC), suggesting that a particulate filter might be required for the G-DI vehicles to comply with the diesel PN limit.

2.2 SCOPE AND OBJECTIVES OF THE STUDY

The main objective of this study is to provide DG-ENTR with technical information for developing the particle number standards for Gasoline Direct Injection (G-DI) vehicles at a Euro 6 level. While generally, both combustion improvements and GPF installations are considered from the manufacturers for compliance with a diesel PN limit, it was not possible in the present study to consider the former option due to limited information on the associated implementation costs and real world emission reduction potential. In that respect, the study focuses on the feasibility and cost effectiveness of introducing a Gasoline Particulate Filter (GPF) in G-DI Vehicles.

More specifically, three policy approaches were examined, namely:

- <u>Policy 1</u>: No introduction of a particle number limit for G-DI vehicles.
- <u>Policy 2</u>: Introduction of the particle number limit applicable to diesel vehicles, i.e. 6×10^{11} #/km, at a Euro 6 stage.

- Policy 3: Introduction of the diesel PN limit three years after the entry into force of the Euro 6 regulation.

The last two policies (2 and 3) were assumed to necessitate the installation of GPF systems to all new registered vehicles. It should be stressed though that a three years delay in the implementation of the PN regulation is expected to allow compliance through less expensive internal engine measures. In that respect the cost-effectiveness of the 3rd policy option should be considered to correspond to a worst case scenario.

2.3 METHODOLOGY

In order to collect the necessary technical information and cost figures associated with the installation of a GPF filter on G-DI vehicles, a workshop was organized by DG ENTR with the active participation of the European Automobile Manufacturer's Association (ACEA), Joint Research Centre (JRC), Member State representatives, the Association for Emissions Control by Catalyst (AECC), and DOW automotive Engineering. Following a presentation given by ACEA on the position of automotive manufacturers, the individual catalyst and substrate manufacturers provided detailed technical information on DG ENTR, JRC and member states requesting confidential treatment for most of it.

Limited and very general information was provided on the cost of the various GPF technologies developed. In that respect, the necessary detailed information on the direct cost of the different elements required for a GPF installation, was derived from similar studies conducted in USA for DPF applications.

To evaluate environmental effects, a special application was developed since the official EU policy assessment model (TREMOVE – [16]) does not discriminate between G-DI and PFI vehicles and furthermore does not include information on particle number emissions. The PN emission factors used in the calculations were derived from the COPERT 4 model [17], reported data in the open literature, and experimental data collected at JRC, applying appropriate assumptions where necessary. Information on the vehicle fleet and activity evolution was drawn from the latest TREMOVE runs (TREMOVE version 3.3.1). Figure 2 shows the general scheme followed, as well as the degree of detail in the characterization of the emission performance and vehicle activity.

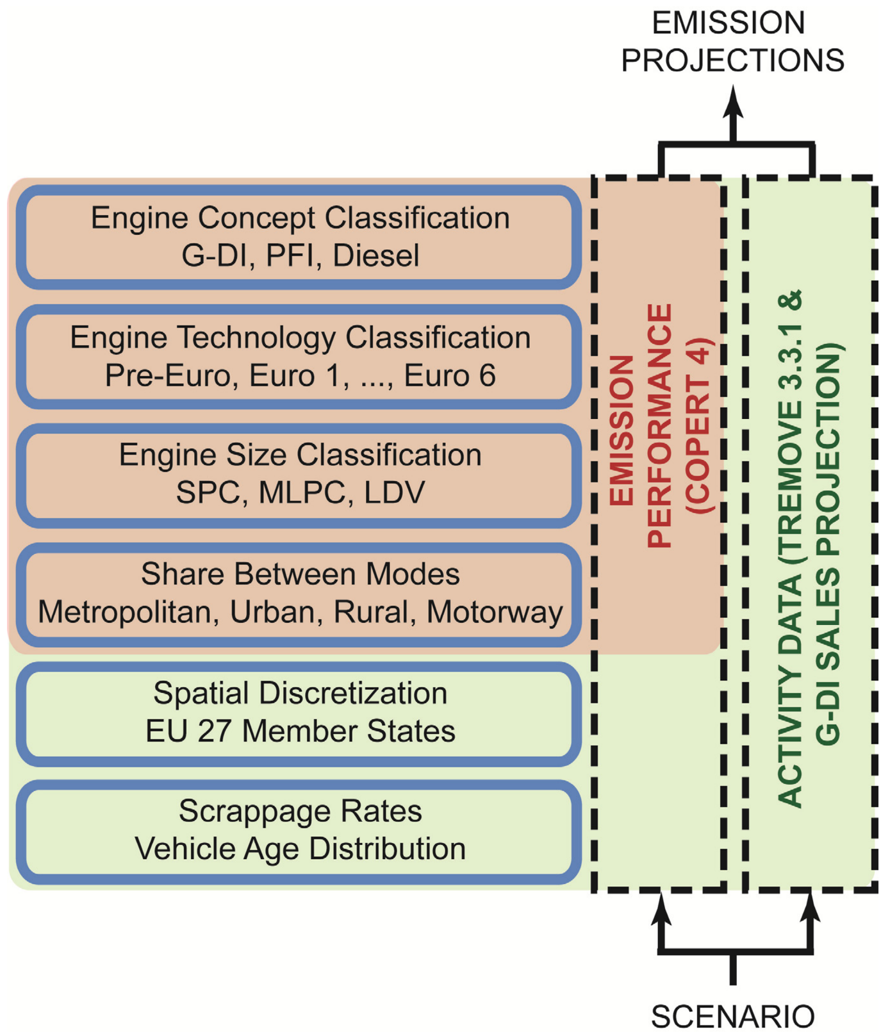

Figure 2: Schematic of the methodology adopted for the calculation of the PN emission projections in EU27. The shaded areas illustrate the degree of detail in the characterization of the emission performance (red area) and activity data (green area).

3 TECHNICAL INFORMATION

3.1 Gasoline Direct Injection Technology

3.1.1 Driving Force - Fuel Consumption

The Gasoline Direct Injection (G-DI) technology emerged from the need to improve the thermodynamic efficiency (and thus reduce fuel consumption) and power output of spark ignited engines. In G-DI engines, the fuel is sprayed directly into the combustion chamber where it evaporates and thus cools the charge. This evaporative cooling, suppresses the knocking tendency (which is caused by excessively high temperatures of the unburned mixture during combustion), thus allowing the use of higher compression ratios (typically by 1 to 2 over baseline [18, 19, 20]), but also increases the volumetric efficiency from the increase in density of the incoming charge [19]. The associated efficiency improvement is somehow counterbalanced by increased parasitic losses due to the higher fuel pressures compared to PFI engines (150-200 bar compared to 3-5 bar) and the absence of "thermal throttling" [19]. In that respect, the overall fuel consumption benefit realized by directly injecting the fuel into the cylinder is in the order of few percent. Fuel consumption reduction estimates range from 1 to 4% [18, 19, 20, 21,22].

G-DI engines can further improve the thermodynamic efficiency when operating in stratified mode (Lean-burn G-DI). Operation in this mode under low-speed, low-load conditions, allows reduced intake air throttling, and thus reduced pumping losses and fuel consumption, as well as reduced heat losses (the excess air reduces combustion temperature, which in turn reduces heat loss to the cooling and exhaust systems) [23, 24]. In the lean-burn mode, fuel is injected near the spark plug during the compression stroke to create a stratified charge. Under certain operating conditions, the overall air-to-fuel ratio can be as high as 20:1 to 40:1 (as compared with 14.7:1 for stoichiometric combustion). The stratified operation allows for an additional up to 8-14% reduction in fuel consumption compared to stoichiometric G-DI concepts [18].

One additional advantage of the G-DI technology is that it allows effective turbocharging and downsizing [25]. In theory the use of a turbocharger to achieve similar engine power with a smaller engine would also improve fuel economy by reducing throttling and friction losses at lighter loads [20, 22]. However, in practice fuel consumption benefit through turbocharging and downsizing was only observed in G-DI applications [20, 26]. The estimated additional reduction in fuel consumption is around 3.5% to 5% [19].

The first G-DI designs utilized either a wall-guided or air-guided fuel injection. In these combustion systems, the injector is placed a long distance from the spark plug, and the fuel spray is directed towards the spark plug by a well defined, in-cylinder air motion or by the interaction of the spray with the piston combustion cavity [24]. Late design G-DI engines utilize spray-guided injection through a close arrangement of the injector and the spark plug, providing close coupling between fuel preparation and ignition. This configuration allows for a wider stratified combustion operation as well as for an improved efficiency of the stratified combustion process [27], thus further improving fuel economy. For example BMW claimed an up to 20% fuel consumption reduction over a baseline port fuel injection vehicle, through the use of a lean-burn spray-guided system [19]. In line with this, engine simulation computations by Alkidas et al. [24, 28] suggested a 15% fuel economy advantage of a G-DI engine over an equivalent PFI engine under the Federal Test Procedure (FTP) cycle. The fuel efficiency was 19% to 23% at near idle and light load conditions were the engine operated in stratified mode and dropped to 5% to 7% at medium load conditions were the combustible mixture was stoichiometric.

Due to their potential for significant fuel efficiency improvements, G-DI gasoline vehicles were identified as a key technology in the direction of reducing greenhouse gases emissions and oil dependency. Regulations controlling the carbon dioxide (CO_2) emissions from passenger cars and light duty vehicles are already in force in Europe [29] and USA [30]. The European regulation sets an average passenger car fleet limit of 130 g/km over NEDC from 2012 with a target to further reduce it to 95 g/km in 2020 and 70 g/km in 2025. The USA regulation limits the fleet average CO_2 emissions of both passenger cars, light duty trucks and medium duty passenger vehicles to 250 g/mi (~155 g/km) by 2016. Work is currently underway to define the target limits in the 2017-2025 timeframe [31].

A number of technologies are considered by the manufacturers to achieve this target, including advanced gasoline and diesel engines and transmissions, electrification, vehicle mass reduction, fuel cells and thermal management technology improvements [31]. Among these options, the downsized, turbocharged G-DI technology appears to be the most mature technology and as such a high penetration of G-DI vehicles is expected in both EU and US markets. It is foreseen that the market share of G-DI vehicles in USA will increase from 3%-5% in 2008 [32] and 5%-6% in 2011 [23] to 58-62% in 2016 [33]. Similarly, the market share of G-DI vehicles in Europe is expected to increase from 7%-9% in 2010 (9%-19% of the gasoline sales) to 13%-18% in 2015 (24%-31% of the gasoline sales) [FEV 2009, CSM 2/2009 – information presented in the workshop]. The global production of G-DI vehicles is expected to increase from 5 millions in 2010, to 21.6 millions in 2015 reaching 35 millions in 2020, constituting 7%, 23% and 35% of the total vehicles produced globally [IHS/CSM March 2011 – information presented in the workshop].

3.1.2 Particulate emission performance

The major shortcoming of the G-DI engines is their elevated particulate emissions. A survey of the literature and experimental data collected at JRC [15], suggested that while G-DI vehicles can easily comply with the PM standard of 4.5 mg/km (already applicable at a Euro 5a stage), their solid particle number emissions exceed the diesel PN standard by 5 to 25 times, over the NEDC.

There is a limited number of studies dealing with particle formation during the combustion process in G-DI engines [27, 34, 35, 36, 37]. All studies though, suggest that the injection strategy has a significant effect on the particulate emissions. By injecting the fuel late in the compressions stroke, a stratified charge develops that allows operation with overall lean mixtures (lean-burn G-DIs) [28]. Early injection in the combustion stroke (Stoichiometric G-DIs) leads to a more homogeneously mixed charge, resembling that of a conventional PFI engine. A shift from homogeneous to stratified charge operation can result in 10-fold to 40-fold increase in particle number emissions [37].

Experimental data on the emission performance of commercial G-DI vehicles however do not show a clear benefit of stoichiometric G-DIs over their lean burn counterparts [15]. PN emissions of stoichiometric G-DIs can exceed those of conventional PFI vehicles by more than one order of magnitude [15, 38]. These observations point towards insufficient time for mixture preparation in G-DI engines compared to PFI engines. In other words, some fuel droplets may survive evaporation resulting in some local regions of rich mixtures were formation of soot is likely to take place. In contrast, in a PFI engine the fuel is injected on the intake ports. Heat from the ports and intake valve facilitate fuel vaporization resulting in a high degree of mixture homogeneity at the moment of the spark event.

Liquid pools forming on the piston walls through the impingement of fuel spray were also identified as a significant source of soot [27, 35, 39]. The Liedenfrost effect can suppress evaporation of such liquid films that will burn belatedly via diffusion flames with highly sooting effects [39]. Late technology spray-guided G-DI engines are thus expected to exhibit improved emission performance. Indeed, Price et al. [40] observed a significant PN reduction potential in spray-guided technology, but still their emissions were an order of magnitude higher than PFI engines. In line with that, solid particle number emissions of two late technology, spray guided G-DI vehicles tested over the NEDC were found to be on average 1.6×10^{12} #/km and 2.2×10^{12} #/km, i.e. 2.5 and 3.5 times above the diesel PN limit [41].

3.2 Potential for compliance with the diesel PN limit through engine improvements

The prospect of a diesel-like particle number limit for G-DI vehicles has recently stimulated considerable research work on the possibility to improve the particle emission performance through internal engine measures. Two principal approaches were investigated, namely a) hybrid PFI/G-DI engines and b) optimization of the combustion/mixture preparation process.

Hybrid PFI/G-DI engines utilize a twin injector system, allowing for fuel injection in either the inlet port (port fuel injection) or inside the engine cylinder (direct injection), or even in both, depending on the operating conditions. This ability to freely switch between the two injection modes provides the means for a significant reduction of particle number emissions alongside a further reduction of fuel consumption [42]. Hybrid PFI/G-DI engines have already been employed since 2007 but mostly in some high performance vehicles (e.g. Lexus LS460 – [43]). Recently, Audi has presented the development of a high performance twin injection engine [42] that according to the manufacturer already conforms with the diesel PN limt. The engine has already entered the production process for installation in the Audi B8 family and the intention is to be introduced in several models of the Volkswagen group within 2012. Yet there is no published data on the particle number emission performance of such vehicles.

There is also an increasing number of publications on research work on calibration of G-DI engines towards low particle number emissions [36, 44, 45]. The studies suggested that that a substantial reduction in PN emissions can be realized by optimizing the number and duration of injections, fuel injection pressure, injection timing, ignition timing and air to fuel ratio and through smooth transition of all these combustion related parameters during accelerations.

By applying such a calibration to a commercial (non-spray guided) stoichiometric G-DI vehicle, a 65% reduction in the PN emissions over the NEDC was observed with no side effects in fuel consumption, driveability or other pollutants [44]. It was not stated though whether the calibrated engine could comply with the diesel PN limit. The potential for a more than an order of magnitude reduction in the particle number emissions of a Euro 5 certified G-DI vehicle were also reported by Dobes et al. [45]. Piock et al. [36] calibrated a prototype engine utilizing a novel outwardly opening nozzle that offers more effective atomization and improved penetration, thus minimizing spray-wall interaction and rich region formations. They managed to achieve PN emissions of 4.4×10^{11} #/km over the NEDC, a figure that is 26% bellow the diesel PN limit. At the same time, the vehicle complied with the Euro 6 standards for all gaseous pollutants. A potential for even further improvements was identified through more precise timing and metering of the injected fuel in all cylinders. Advanced injector systems allowing for up to five split injections were shown to result in two orders of magnitude reduction in particle numbers improving at the same time the stability of the combustion [36].

A recent study [46] has investigated the particle emission reduction potential offered by Exhaust Gas Recirculation (EGR). Up to 65% reductions in the mass of emitted soot were observed, but the effect of EGR on the particle number emissions was not consistent. Under certain operating conditions the use of EGR resulted even in increases of the particle number emissions. This inconsistency was attributed to changes in the size of emitted particles, which was found to decrease with increasing EGR rate.

3.2.1 Real world emission performance and long term emission stability

G-DI engines allow for a very accurate control of the air to fuel ratio especially when starting the vehicle and during warm up where most of the emissions are formed [25]. The rapid increase in computing power and speed allows nearly constant monitoring of the injection and combustion process through the powertrain control software [23]. The ability of the powertrain controllers to monitor and react to the combustion process is critical to G-DI technology, yet could potentially tailor their performance to meet the emission limit over the certification cycle only. There is no information available on the particle emissions of such prototype/calibrated vehicles under unregulated conditions.

Fuel spray preparation was identified as the most crucial design element. However, the optimization of the fuel injection strategy can not be considered independently from the fuel specifications. Tests of a single G-DI vehicle with different commercially available gasoline fuels resulted in up to 70% differences in the emitted number of solid particles [45]. Blending of ethanol with gasoline can also have a significant, yet not consistent, effect on the particle number emissions [45, 47] even at low ethanol content. Generally, the content of oxygenates and the fuel evaporation profile were identified as the most important fuel properties affecting the particle number emission performance of G-DI vehicles [45, 47].

Another concern regarding the emission performance of G-DI engines pertains to the formation of soot deposits in the combustion chamber, the intake ports and most importantly the injector (mainly upon hot engine shut-down [19]). Emission degradation due to fouling of the injector was a well known issue from the very first, unsuccessful implementations of stratified charge spark ignition engines in the 1970s [48, 49]. Similar deterioration of the emissions was observed in the first commercially available G-DI vehicle [50] that entered the European market in 1996 [51]. Increases in the particle number emissions by 60% were also reported for late technology calibrated engines due to extended short term deposits at the injector [45]. Fouling of the injectors may be reduced through Improvements in the injector designs and through optimization of the combustion chamber and intake air system to reduce the temperature and increase the air flow at the injector tip [19, 45]. However, the fuel variability raises some concerns given the strong dependence of the spray formation on the fuel properties [19].

The approaches envisaged to reduce the particle emissions from G-DI engines may also affect the size of the emitted particles. External cooled EGR is expected to be widely employed in future turbocharged and downsized G-DI engines as it has the potential to reduce the propensity of engine knock thus allowing higher boosting level but at the same time also reduce exhaust gas temperature at high loads that would otherwise require fuel enrichment [19, 46, 52]. Brake Specific Fuel Consumption reduction of up to 5% was reported for a prototype turbocharged, downsized G-DI engine [52] through the use of a low pressure cooled EGR system. However, the use of EGR was found to reduce the size of the emitted non-volatile particles with the peak of the distribution shifted by typically 4 to 8 nm [46]. Under a particular operating condition, a large shift of the distribution mode from 42 nm down to 17 nm was observed, with the number concentrations remaining unaffected [46].

This raises concerns regarding the ability of the legislated procedure to quantify the true emission levels of such vehicles, given the specified 23 nm cut-off size.

3.3 Particulate Filters

3.3.1 Diesel particulate filters

Ceramic wall-flow Diesel Particulate Filters (DPF) are widely used in both Europe and USA. The first successful implementation of DPFs dates back in 2000 (by Peugeot [53]) and since then several million commercial passenger cars utilizing DPF have been sold. Following the introduction of a particle number standard at a Euro 5b stage, all diesel vehicles will be equipped with very efficient, wall flow DPF systems.

The operation principle of DPF systems is based on the separation of the airborne particles from the gas stream by deposition on a collecting surface. Several types of filters were developed over the years, including ceramic wall flow, foam and flow-through filters. Only the ceramic wall-flow DPF exhibits sufficiently high filtration efficiencies required by the upcoming PN standards and is therefore widely used. It consists of a honeycomb-like ceramic structure with alternate passages blocked, thus forcing the exhaust gas to flow through the porous walls which act as a filter medium. Available ceramic materials (substrates) include cordierite, silicon carbide, silicon nitride and the recently introduced acicular mullite [54].

The main challenge in DPF applications is to efficiently remove the soot accumulated on the filter in order to prevent excessive backpressure rise (that can lead to significant performance degradation) and even plugging. Removal is achieved through oxidation of the collected soot in a process called regeneration. Soot oxidation through the excess oxygen available in the diesel exhaust, requires temperatures of 600°C or more, which are never reached under normal engine operation. The use of fuel borne catalyst, catalytic coating of the filter and/or an upstream diesel oxidation catalyst can assist the regeneration of the DPF.

The fuel borne catalyst (typically some combination of cerium, strontium and iron) is added to the fuel through a separate tank. During the combustion process, catalytic metallic nanoparticles become intimately mixed with the soot, lowering the combustion temperatures below 400°C [55]. A major shortcoming of this approach is that the fuel borne catalyst contributes to the ash stored in the filter, eventually lowering its soot loading capacity. The first generation of fuel borne catalyst DPFs required periodic service to remove the collected ash (80000 km) but late technology systems are service free (200000 km autonomy). Concerns related to elevated metallic oxide emissions [56] resulted in limited use of fuel borne catalyst systems in the United States of America [25].

In a catalyzed DPF, the ceramic substrate is coated with a catalytic washcoat, typically Platinum, facilitating regeneration, not to the extent that fuel borne catalyst does though [57, 58]. Catalyzed DPF has a relatively higher cost due to the precious metal requirement. It has the advantage of not requiring maintenance though due to the much lower quantities of ash (originating mainly from lube oil) collected.

The use of an oxidation catalyst upstream of the DPF (commercialized as Continuous Regenerating Trap by Johnson Matthey) facilitates regeneration through the production of nitrogen dioxide (NO_2) which is a strong oxidizing agent. The oxidation catalyst converts nitrogen monoxide (NO) in the exhaust to NO_2. NO_2 oxidizes soot at temperatures between about 250 and 450°C, and is itself reduced to NO. This approach has the disadvantage of producing elevated emissions of nitrogen dioxide.

Under certain operating conditions, the exhaust gas temperature can exceed the temperature required for the oxidation of the accumulated soot, that depends on the aftertreatment configuration employed and the soot loading of the DPF, at which point regeneration of the DPF occurs. This type of regeneration is referred to as passive regeneration since it occurs naturally without requiring any kind of interference from the Emissions Control System (ECS). Passive regeneration plays an extremely important role even in active systems because it is associated with little or no fuel-consumption penalty. In practise however, the exhaust gas temperature is too low to sustain regeneration of the DPF under all operating conditions.

In order to ensure that the DPF will regenerate regularly and, most importantly, avoid excessive soot accumulation and stochastic regeneration, the ECS continuously monitors the pressure drop (or some other measure of filter loading) across the DPF. If deemed necessary, the ECS can also trigger regeneration (active regeneration) in order to increase the exhaust gas temperature at levels that would initiate and sustain regeneration. This is achieved by means of generating an exotherm through delayed fuel injection in the cylinder or even a post-injection of fuel in the exhaust manifold. Additional or complementary engine measures to trigger regeneration include Exhaust Gas Recirculation (EGR) shut-off and throttling of the exhaust.

The backpressure introduced by the DPF and the need for periodic active regeneration introduces a fuel consumption penalty. The increase in fuel consumption due to backpressure is quantified to be 1% to 2% or less [25, 57]. Less information is available on the fuel consumption penalty associated with active regeneration of the DPFs. This will strongly depend on driving conditions, with little or no penalty for high driving where regeneration is expected to be mostly passive. Low temperature driving conditions with frequent stops will require more frequent active regenerations leading to fuel consumption penalties of up to several percent [25]. Edwards et al. [58] suggested an average fuel consumption penalty of 2.5%.

DPF systems have been applied to production vehicles since 2000 (with a first unsuccessive application back in 1985 [59]). The DPF technology experienced significant advancements over these years, through regeneration strategy optimization as well as substrate and catalyst improvements [60]. Significant improvement was made in the understanding of the soot oxidation process during active and passive regeneration [61]. At the same time, advanced substrate materials were introduced [62, 63] exhibiting improved filtration efficiency and lower backpressures even at high soot loadings. The dynamics of ash accumulation are better understood [64] and improved designs with higher ash storage capabilities were introduced [65]. Finally, advanced catalyst coatings in DPFs improving contact areas are developed that can enhance regeneration [66, 67].

3.3.2 Gasoline particulate filters

Particulate filters proved to be very efficient in controlling particle number emissions from diesel vehicles and as such they are also considered as an option for G-DI vehicle application. Some information on the potential application of particulate filters to G-DI vehicles is already available in the open literature [68, 69, 70, 71, 72, 73]. During the workshop, several substrate and catalyst manufacturers presented detailed technical information on this subject and requested confidential treatment for most of it. The key conclusions and main challenges were remarkably similar for all manufactures and in good agreement with the published data. In order to respect the request for confidentiality, only some general information will be presented in the report that would assist in the formulation of the possible technology approaches and estimate the associated implementation costs and lead time for series vehicle integration.

3.3.2.1 Differences between diesel and gasoline exhaust

Gasoline engine exhaust temperatures are relatively higher from those of their diesel counterparts. Engine-out temperatures range between 300 and 500°C under urban driving conditions, but can reach 700°C at high speed (motorway) driving [72, 73]. While these temperatures are high enough to sustain passive regeneration of a Gasoline Particulate Filter (GPF), there is generally a shortage of oxygen in the exhaust of G-DIs running stoichiometrically. Accordingly, regeneration is generally observed to occur during vehicle deceleration were the fuel is cut-off and oxygen becomes available in the exhaust [68, 69, 72, 73]. Catalyzed GPF systems with oxygen storage capabilities may also allow short periods of lean operation to initiate passive or active regeneration. Installation in lean-burn engines is more straightforward as there exist generally a surplus of oxygen in the exhaust and the exhaust temperatures are similar to those of their stoichiometric counterparts.

Particle concentrations in G-DI exhaust are much lower, compared to those of diesel engines and therefore much lower soot accumulation is expected [72]. A recent study [74] also showed that soot emitted from G-DI vehicles oxidizes much faster than diesel soot. Measured burnout rates of G-DI soot were found to be 2.5 times higher than those of diesel soot. All these are in good agreement with the common observation of insignificant soot storage in the GPF, even after prolonged operation under real world driving conditions [68, 69]. Consequently, no extreme heat release due to soot oxidation is expected so thermal durability is not considered an issue by substrate manufacturers.

The low soot loading also allows for the use of more compact and therefore less expensive filters from diesel applications (due to the lower volume and in the case of catalyzed GPFs the lower precious metal content). While in diesel vehicles, the volume of the filter is typically 1.5 to 2.5 times the engine displacement [75], the envisaged GPF systems have much smaller size, equal or smaller to the engine swept volume. Reported GPF volumes in published studies were 0.84 l for a 2 l G-DI engine [68], 2.5 l for a 3.5 l G-DI engine [69] and 0.8 to 1.4 l for 1.4 to 1.6 l G-DI engines [72]. Accordingly, the GPF to engine volume ratios in the studies presented in the workshop ranged from 0.4 to 1.2 with an average value of 0.7. In fact, the optimum size of the GPF will most probably depend on the level of ash emissions, which are expected to be higher in gasoline vehicles due to the relatively higher engine speeds [76]. Some GPF manufacturers have already performed dedicated studies on the assessment of the ash storage capabilities, which did not reveal significant ash accumulation or any performance deterioration for the GPF volumes examined, but more research is required on this issue.

3.3.2.2 Packaging options for GPF

Several approaches are considered for installation of the GPF in the exhaust system, but they can be classified into two main options: a) close coupled to the engine or b) underfloor installation. A close coupled GPF will most probably also incorporate some catalytic coating thus acting as a four way catalyst. Underfloor installations may or may not incorporate catalytic activity.

<u>Underfloor GPF</u>:

An underfloor installation, has the potential shortcoming of potentially requiring some kind of active regeneration due to the relatively lower temperatures (>250°C), particularly under extended operation in urban driving conditions. However, a common observation from all manufacturers was that no accumulation of soot could be observed in the filters. One GPF manufacturer mentioned that in an on-going, on-road durability study, more than 100000 km have been accumulated in a G-DI vehicle retrofitted with an underfloor uncatalyzed GPF, with no need of maintenance or interference of any kind. Nevertheless, excessive soot accumulation was observed [68] under repeated start-stop operation in sub-zero ambient temperatures.

All available information suggest that underfloor GPF systems (both coated and uncoated) will mostly regenerate passively, and there will be minimum requirements for active regeneration. The latter can be achieved through retarded fuel injection and spark timing in the case of stoichiometric G-DIs or post-fuel injection in the case of lean-burn engines, approaches already employed to reduce cold start emissions of gaseous pollutants. The necessary technology is already available, so there should be no need for significant improvements and investigations. This argument is supported by experimental data collected at JRC, presented in Figure 3. The graph shows measured temperatures upstream of the single three-way catalyst of a Euro 5 technology G-DI vehicle tested over NEDC at -7°C and 22°C ambient temperatures. A sharp increase in the exhaust temperature is observed over the first 20 s of the test, exceeding 400° even at -7°C. The temperatures at sub-zero tests reached and even exceeded those at 22°C within less than 100 s.

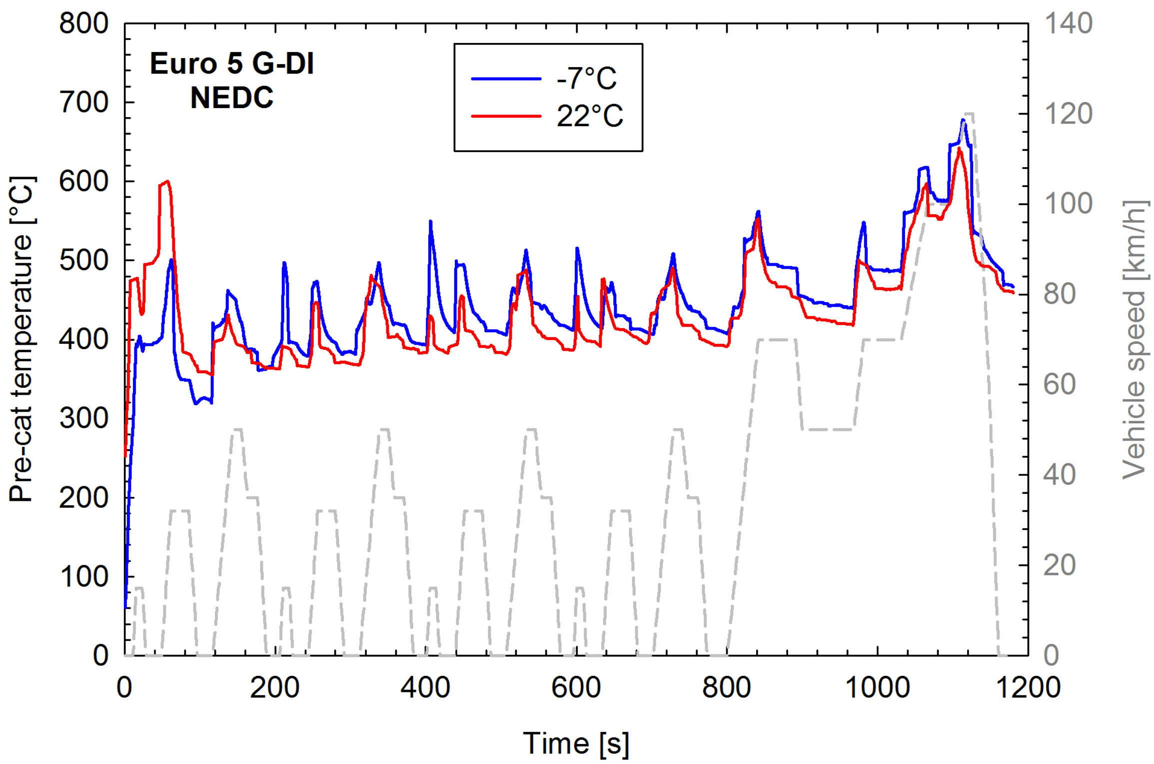

Figure 3: Measured temperatures upstream of the single three-way catalyst of a Euro 5 technology G-DI vehicle tested at JRC over NEDC at -7°C (blue line) and 22°C (red line) ambient temperatures.

Underfloor installation imposes less packaging limitations, thus allowing introduction of relatively large GPF volumes. In that respect, potentially high ash accumulation might necessitate the use of an underfloor relatively large GPF. No evidence of excessive ash accumulation was observed however. A lot of progress was also made in the direction of using catalyzed GPFs in place of underfloor Three Way Catalysts (TWC). Catalyzed GPFs employing the same amount of PMG content, capable of complying with the Euro 6 standards in all regulated pollutants and the diesel PN limit have been presented. The good noise attenuation properties of GPFs allow the use of uncatalyzed systems in place of the muffler to reduce implementation cost.

Close coupled catalyzed GPF:

A close coupled installation of a catalyzed GPF is considered as the most desirable solution. It allows full utilization of the high thermal energy of the exhaust, essentially requiring no active regeneration of the filter. One practical shortcoming is the lack of space that could possibly impose limitations on the GPF volume. The main challenge in these configurations is to achieve similar light-off times with close coupled TWC. The catalyzed GPF (either in a standalone or in a two brick arrangement) increases in general the thermal mass, and consequently the gaseous emissions (carbon monoxide, hydrocarbons and nitrogen oxides) over cold start. All catalyst manufacturers however, presented data showing significant improvements from the first generation concepts investigated, with some late generation systems exhibiting comparable performance with commercial TWC of the same PMG content.

3.3.2.3 Filtration efficiency

Unpublished experimental data on the particle number emissions from 24 in total commercial G-DI vehicles was presented in the workshop. The solid particle number emissions over NEDC ranged from 1.3×10^{12} #/km to 6×10^{12} #/km, averaging at 2.7×10^{12} #/km. Consequently, filtration efficiencies of better than 54% and 90% will be required in order to achieve the diesel PN limit of 6×10^{11} #/km. It is worth noting though that one particular GPF manufacturer presented data from a vehicle emitting as low as 8×10^{11} #/km, verifying that there exists a potential for further PN reductions through engine measures.

All data suggest that under real world operation the GPFs will be clean most of the time and therefore need to have high initial filtration efficiencies [72]. This is in good agreement with the observations of Andersson et al. [71] that current technology open traps did not prove efficient enough to reduce the emissions of G-DI vehicles below the diesel PN standard.

3.3.2.4 Fuel consumption penalty

All experimental data presented in the workshop suggested no fuel consumption penalty over NEDC and generally under urban and rural driving conditions. Increased fuel consumption by a maximum 3% was only observed at high engine speeds and full load, suggesting an approximately 1% increase under typical motorway driving conditions. This is in line with the findings of Saito et al. [72] who observed a 2% fuel consumption penalty only under high speed/high load conditions but no effect over the NEDC. The same group [73] reported in a later study insignificant effect on the fuel consumption over the Common Artemis Driving Cycles [77] and a motorway test cycle developed by ADAC (Allgemeiner Deutscher Automobil-Club).

Likewise, Mikulic et al. [68], did not observe any measurable fuel consumption penalty except at 180 kph and 200 kph cruising where it was quantified to be around 2-3%. Interestingly, the same study found a systematic decrease in the fuel consumption over the NEDC (~2-3%), accompanied by a 30-40% decrease in nitrogen oxides. As the authors suggested, this behaviour is indicative of increased internal Exhaust Gas Recirculation (EGR), caused by the backpressure introduced by the GPF. Increased EGR, reduces the combustion temperature and thus the engine out nitrogen oxide emissions. At the same time it reduces the available oxygen and therefore for the engine to run stoichiometrically, less fuel is injected.

Some tests were also performed at JRC to access the performance of an optimized (uncatalyzed) GPF on the particulate and CO_2 emissions of a commercial Euro 5 technology G-DI vehicle. The results of these experiments will be presented in details elsewhere. Figure 4 compares the measured CO_2 emissions of the vehicle in its OEM configuration and following installation of the GPF downstream of the TWC, over the NEDC, the CADC and selected steady speeds. No fuel consumption penalty could be identified under all test conditions examined including the motorway part of the CADC. Actually the data suggest a small but systematic decrease in the CO_2 emissions in good agreement with what Mikulic et al. [68] have reported.

Overall, the effect of the backpressure introduced by optimized GPF systems on the fuel consumption of the vehicle is expected to be insignificant. The frequency of active regenerations, and therefore the associated fuel consumption penalty, if any, is also expected to be considerably lower compared to diesel applications due to a) the much lower soot emissions and b) the more frequent passive regeneration events.

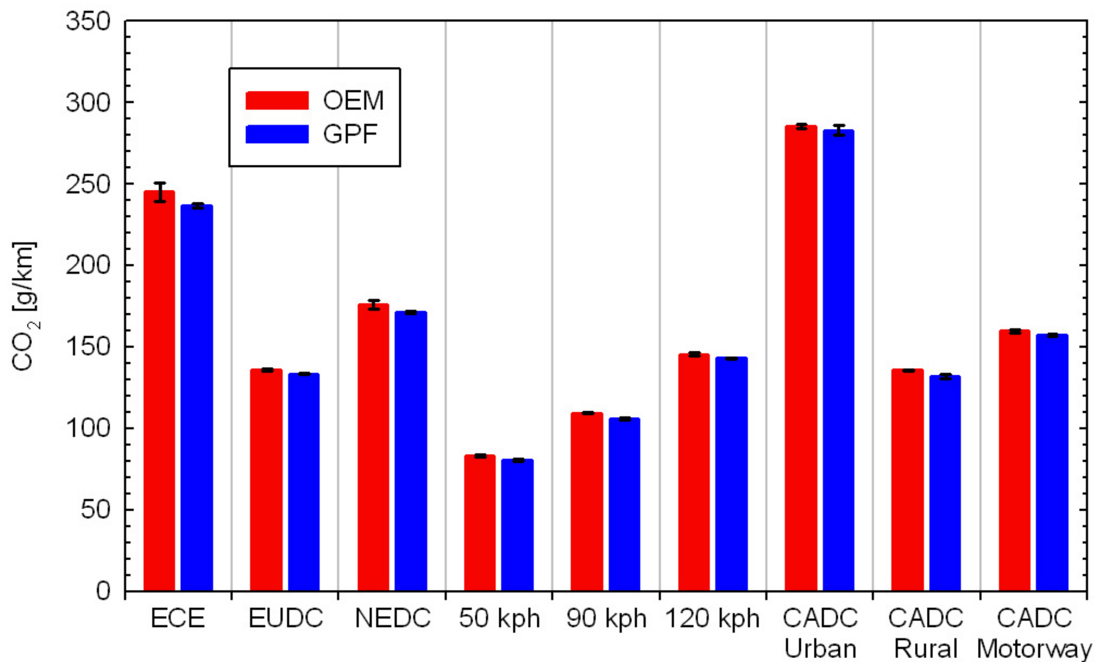

Figure 4: Measured CO_2 emissions from a G-DI vehicle tested at JRC at its OEM configuration (red bars) and retrofitted with an optimized GPF (blue bars). Error bars correspond to ± one standard error calculated from three test repetitions.

4 EMISSION PROJECTIONS

4.1 Fleet evolution

The TREMOVE model [94 – version 3.1.1] was employed in order to project the population and corresponding age distribution of vehicles in Europe (EU 27) up to 2030.

4.1.1 Category classification

Only gasoline and diesel vehicles were considered in the analysis. The vehicles were classified into three main categories according to their size, namely:

- Small Passenger Cars (SPC) assumed to be equipped with an engine smaller than 1.4 dm^3 ("car <1.4 l" categories in the TREMOVE model).

- Medium and Large Passenger Cars (MLPC) assumed to be equipped with an engine larger than 1.4 dm^3 ("car 1.4-2.0 l" and "car >2.0 l" categories in the TREMOVE model).

- Light Duty Vehicles (Light Duty Vehicles) including both vans and light duty trucks ("van - N1 class" and "light duty truck" categories in the TREMOVE model).

The TREMOVE model discriminates between diesel and gasoline vehicles but does not include separate categories for G-DI vehicles. In that respect, some assumptions needed to be made for the share of G-DI vehicles in the gasoline market. Three alternative scenarios were investigated based on input collected during the workshop. These are shown in Figure 5. It is worth noting that the latest projections (35% to 60% in 2017) suggest significantly higher penetrations of G-DI vehicles from what has been assumed in the impact assessment of the Euro 5 emission standards [10% - 78]. This reflects the foreseen direction towards more fuel efficient vehicles to reduce greenhouse gases [29].

Each of the nine in total vehicle classes resulting, were further distinguished into emission technologies according to the emission standards introduced in Europe. The actual implementation dates of the emission standards were employed for this classification. This approach incorporates the inherent assumption that the better performance of vehicles complying with the forthcoming regulations is counterbalanced by the sales of older technology vehicles in the transition period (type approvals versus new registered vehicles). Furthermore, given that the time resolution of the TREMOVE data is one year, the different technologies were assumed to have entered into the market at the start of the year closest to the actual implementation date. The assumed classification of the different vehicle model years to the different emission technologies is given in Table 2.

Table 2: Classification of the different year models to the different emission technologies

Implementation Year	PC diesel	LDV diesel	PC gasoline	LDV gasoline
Euro 1	1993	1995	1993	1995
Euro 2	1996	1998	1996	1998
Euro 3	2000	2001	2000	2001
Euro 4	2005	2006	2005	2006
Euro 5a	2010	2011	2010	2011
Euro 5b	2012	2012	2012	2012
Euro 6	2015	2016	2015	2016

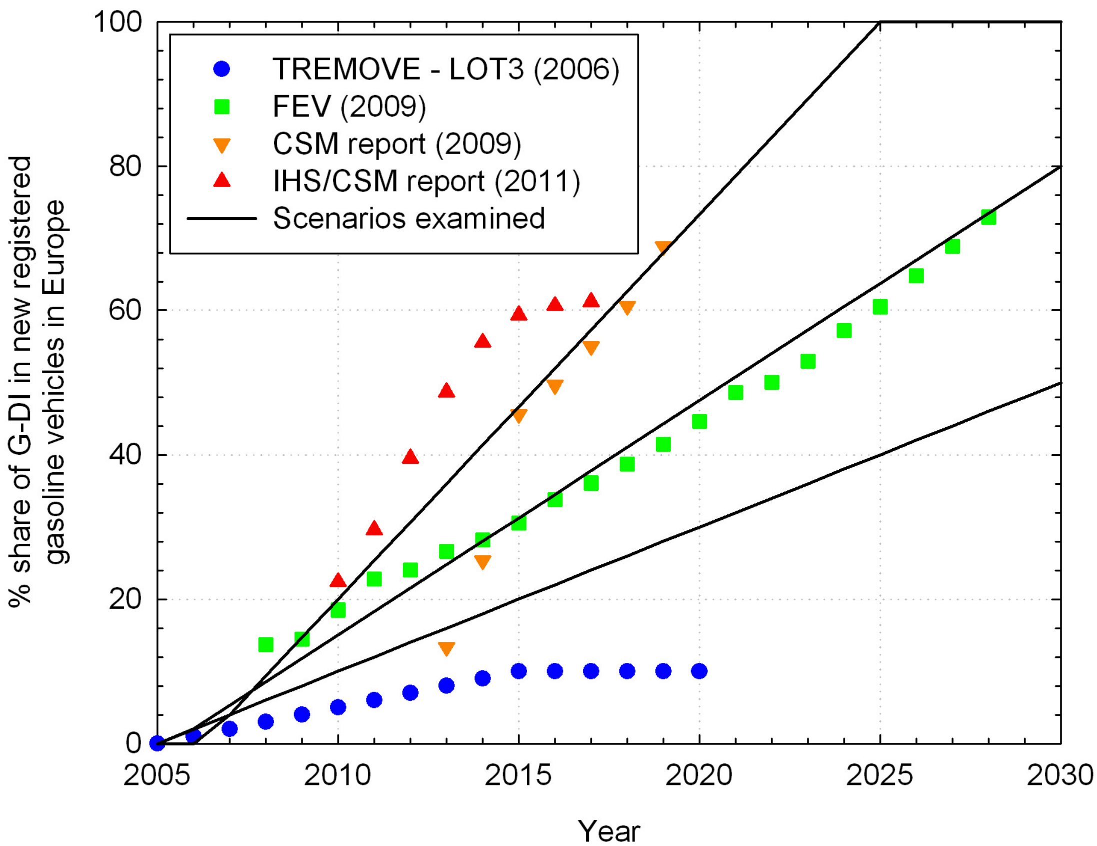

Figure 5: Published projections and formulated scenarios on the evolution of G-DI market share in gasoline vehicles sold in Europe.

4.1.2 Vehicle population and age distribution

The TREMOVE model provides information on the population of the aforementioned vehicle categories as a function of age. As an example, Figure 6 shows the projected evolution of total passenger car and light duty vehicle sales in Europe, categorized according to the engine concept (in particular diesel, PFI or G-DIs). The contribution of G-DI vehicles was externally accounted for, using the three alternative penetration scenarios examined (section 4.1.1). Following a sharp decline of vehicle sales associated with the 2008-2010 financial crisis, the TREMOVE model predicts an increase of vehicle sales that will stabilize at approximately 20 million from 2015 onwards. The share of new registered diesel vehicles is projected to decrease from around 45% in 2005 to ~35% in 2020 where it stabilizes. The TREMOVE model provides also information on the vehicle age distribution of the different category classes. As an example, Figure 7 shows the age distribution of gasoline MLPC vehicles from 1995 to 2030.

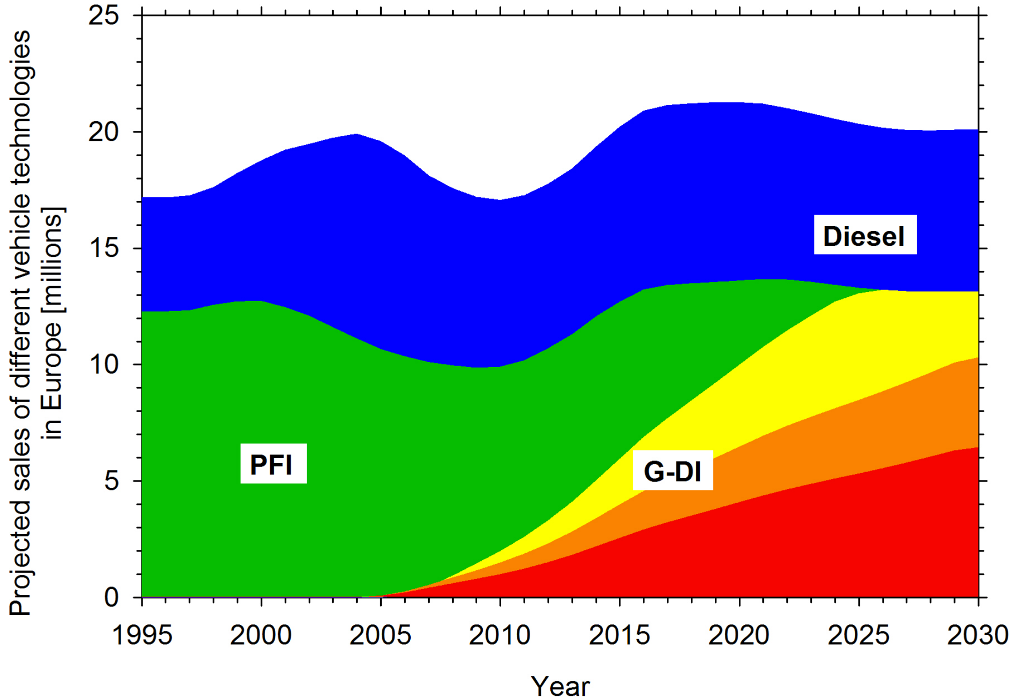

Figure 6: Projected sales of vehicle in Europe categorized according to the engine concept.

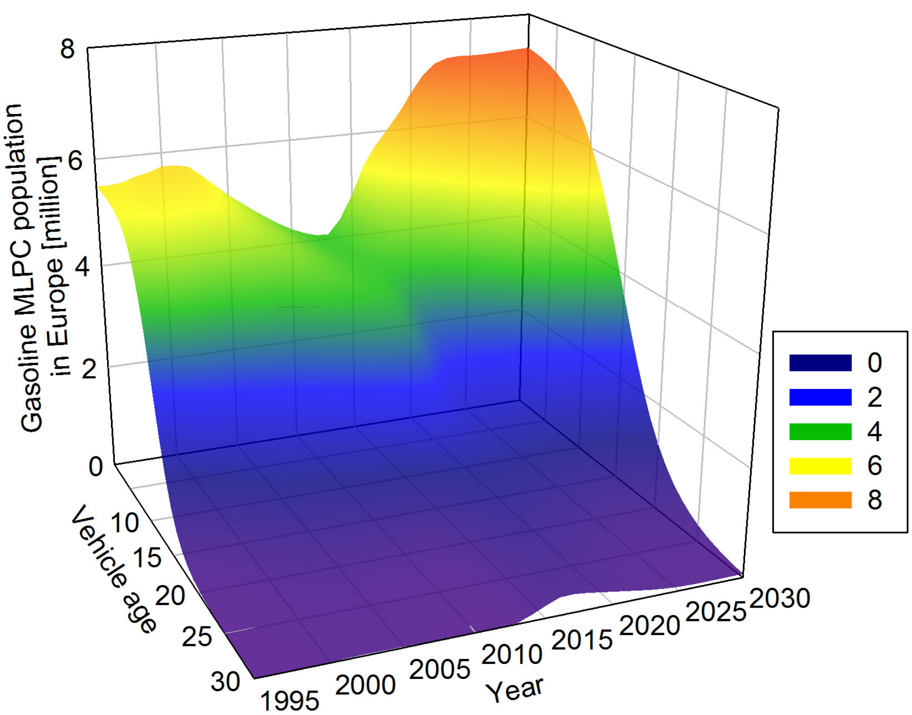

Figure 7: Age distribution of the gasoline MLPC population in Europe from 1995 to 2030.

4.1.3 Activity data

The exhaust vehicle emissions generally depend on the driving conditions. The TREMOVE model provided also information on the annual mileage driven by the different vehicle categories under metropolitan, urban, rural and motorway conditions. The relative share of metropolitan, urban, rural and motorway driving is given as a function of the main vehicle classes (passenger cars or light duty vehicles) with the data suggesting small fluctuations over the years. Table 3 gives the relative share of each driving condition for each EU27 country averaged over the timeframe examined (1995-2030 - TREMOVE).

The TREMOVE model also provides information on the usage of vehicles from 1995 to 2030 as a function of their age, accounting for the fact that older vehicles are generally used less frequently. As an example, Figure 8 shows the projected annual mileage driven by SPC in Europe from 1995 to 2030 as a function of their age.

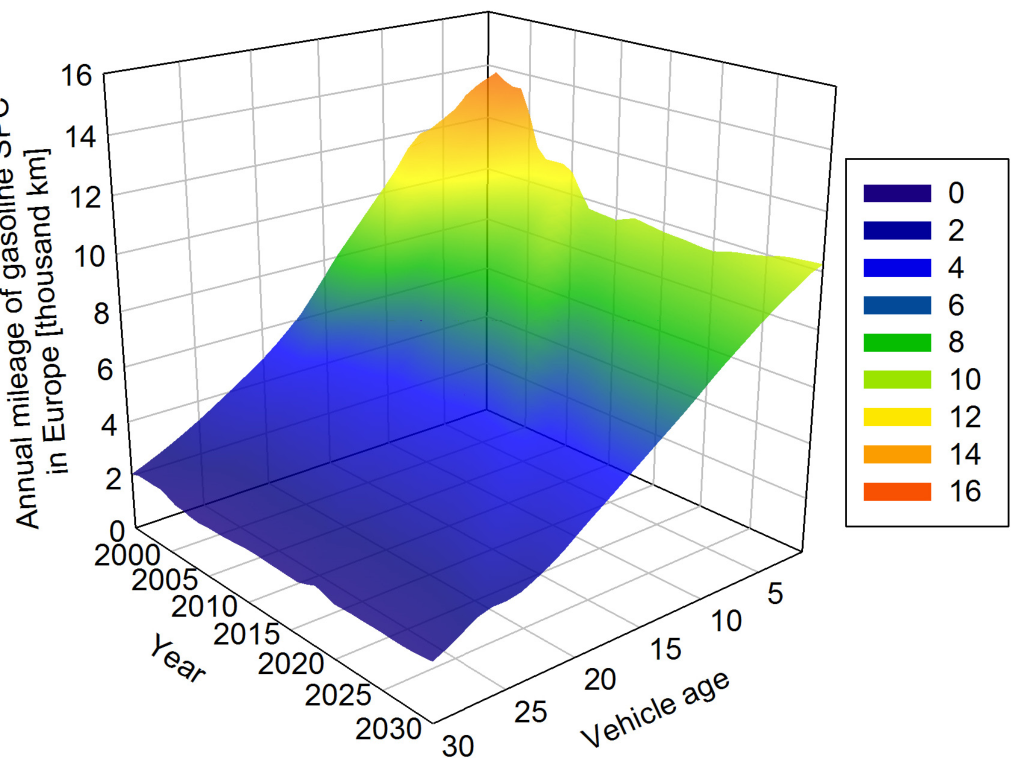

Figure 8: Projected annual mileage driven by gasoline SPC in Europe from 1995 to 2030 as a function of vehicle age.

Table 3: Share of metropolitan, urban, rural and motorway driving for passenger cars and light duty vehicles.

Country	Passenger cars				Light duty vehicles			
	Metrop.	**Urban**	**Rural**	**Motor.**	**Metrop.**	**Urban**	**Rural**	**Motor.**
AT	2%	19%	66%	13%	1%	6%	42%	52%
BE	3%	18%	55%	24%	1%	17%	33%	48%
CZ	6%	23%	65%	6%	2%	11%	72%	15%
DE	2%	25%	54%	20%	1%	6%	28%	65%
DK	10%	12%	65%	13%	10%	12%	66%	12%
ES	6%	22%	61%	11%	3%	8%	65%	25%
FI	12%	15%	65%	8%	8%	6%	68%	18%
FR	2%	26%	57%	15%	1%	3%	48%	48%
GR	9%	24%	57%	9%	8%	14%	60%	19%
HU	4%	6%	76%	14%	5%	5%	74%	16%
IE	14%	10%	72%	5%	19%	4%	74%	2%
IT	3%	16%	63%	18%	0%	5%	39%	55%
LU	1%	54%	36%	9%	0%	66%	30%	4%
NL	5%	21%	48%	25%	1%	4%	36%	59%
PL	8%	13%	75%	4%	0%	1%	93%	6%
PT	11%	12%	55%	21%	10%	2%	59%	29%
SE	5%	26%	61%	8%	3%	6%	60%	31%
SI	5%	27%	49%	19%	0%	0%	88%	12%
UK	5%	38%	47%	10%	7%	23%	50%	20%
EE	15%	24%	61%	0%	0%	1%	93%	6%
CY	2%	8%	86%	4%	0%	1%	93%	6%
LV	12%	19%	69%	0%	0%	1%	93%	6%
LT	14%	13%	64%	9%	0%	1%	93%	6%
MT	2%	11%	73%	14%	17%	9%	31%	43%
SK	5%	15%	72%	9%	0%	0%	95%	5%
BG	3%	20%	72%	4%	5%	5%	75%	15%
RO	4%	6%	76%	14%	5%	5%	73%	17%

4.2 Emission factors

Solid particle number[1] emission factors for the different vehicle categories were derived from the latest version of the COPERT model [17], making some assumptions where necessary. Table 4 summarizes the categories for which PM and PN emission factors are available in the COPERT 4 methodology. PM emission factors for diesel vehicles are provided as a function of average vehicle speed. For the remaining available emission factors, three fixed values representative of urban, rural and motorway driving conditions are given.

Table 4: Vehicle categories for which the COPERT 4 methodology includes PM and PN emission factors.

Emission Standard	Diesel		PFI		G-DI	
	PC	LDV	PC	LDV	PC	LDV
Pre Euro	PM(V)	PM(V)				
Euro 1	PM(V), PN	PM(V)	PM, PN			
Euro 2	PM(V), PN	PM(V)	PM			
Euro 3	PM(V), PN	PM(V)	PM, PN		PM, PN	
Euro 4	PM(V)	PM(V)	PM			
Diesel + DPF	PM, PN	PM, PN				

In order to estimate the emission factors for the remaining vehicle categories the following assumptions were made:

- Average vehicle speeds of 20 km/h, 60 km/h and 100 km/h were assumed as representative of urban, rural and motorway driving conditions, respectively. These figures are very close to the average vehicle speeds over the Urban, Rural and Motorway parts of the Common Artemis Driving Cycles [77].

- Emissions in metropolitan areas were assumed to be the same to those under urban driving conditions.

- Solid particle number emissions were assumed to scale up with Elemental Carbon (EC). The COPERT 4 methodology includes information on the EC content of the PM (Table 6) emitted from diesel and PFI vehicles of all technology levels (not discriminating between LDV and PC). This provided the means to calculate PN emissions of:

    - pre-EURO diesel PC: $PN_{pre-EURO} = PN_{EURO1} \dfrac{PM_{pre-EURO}\left(\dfrac{EC}{PM}\right)_{pre-EURO}}{PM_{EURO1}\left(\dfrac{EC}{PM}\right)_{EURO1}}$

[1] Solid particle number emission factors were derived from data collected during the EU Particulates project (Ntziachristos et al. – SAE Technical Paper 2004-01-1985). In the PARTICULATES project, the solid particle number emissions were measured with an Electrical Low Pressure Impactor sampling downstream a thermodenuder. This allowed for a classification of particles into three main size bins: 30-50 nm, 50-100 nm and 100-1000 nm (aerodynamic diameters). In our calculations, the sum of these three size bins was employed, which corresponds to the number concentration of thermally treated particles larger than 30 nm.

- Euro 4 diesel PC: $PN_{EURO4} = PN_{EURO3} \dfrac{PM_{EURO4}\left(\dfrac{EC}{PM}\right)_{EURO4}}{PM_{EURO3}\left(\dfrac{EC}{PM}\right)_{EURO3}}$

- Diesel LDVs: $PN_{LDV} = PN_{PC} \dfrac{PM_{LDV}\left(\dfrac{EC}{PM}\right)_{LDV}}{PM_{PC}\left(\dfrac{EC}{PM}\right)_{PC}} = PN_{PC} \dfrac{PM_{LDV}}{PM_{PC}}$, since the COPERT methodology does not discriminate between LDVs and PC in terms of their EC content.

- Euro 2 PFI PC: $PN_{EURO2} = PN_{EURO1} \dfrac{PM_{EURO2}\left(\dfrac{EC}{PM}\right)_{EURO2}}{PM_{EURO1}\left(\dfrac{EC}{PM}\right)_{EURO1}}$

- Euro 4 PFI PC: $PN_{EURO4} = PN_{EURO3} \dfrac{PM_{EURO4}\left(\dfrac{EC}{PM}\right)_{EURO4}}{PM_{EURO3}\left(\dfrac{EC}{PM}\right)_{EURO3}}$

- Particulate emissions of Pre Euro PFI vehicles were assumed to be the same to those of Euro 1 technology PFI. The contribution of this vehicle category in the projections is insignificant due to the low population and mileage of these very old vehicles.

- Particulate emissions of gasoline LDVs were assumed to be the same to those of the smaller PC of the same technology.

- All Euro 5 and Euro 6 diesel vehicles were assumed to be equipped with a DPF.

- Particulate emissions of Euro 5 and Euro 6 PFI vehicles were assumed to be the same to those of Euro 4 PFIs.

- PN emissions of Euro 4 and Euro 5 technology G-DI vehicles were assumed to be the same to those of Euro 3 G-DIs. Particulate emissions from non-GPF equipped, Euro 6 technology G-DIs were assumed to be 30% lower. This reduction reflects the improvements in the combustion process (e.g. spray-guided G-DIs) mainly addressing fuel efficiency and to a lesser extend the PM limit that was introduced at a Euro 5b stage.

- The GPF is assumed to have an average 90% filtration efficiency in reducing solid particle number emissions, which corresponds to the target set by the GPF manufacturers (section 3.3.2.3). The same filtration efficiency was assumed for all the driving conditions, since it is anticipated that the GPF will be empty most of the time. This figure is lower that the >99.6% efficiency of DPF systems assumed in the COPERT methodology.

- The effect of cold start on the particulate emissions was not considered since such information was only available for PM and not for PN in the current COPERT methodology.

The resulting solid particle number emission factors are given in Table 5.

Table 5: PN emission factors for the different vehicle categories examined.

Technology	PN [#/km] × 10^{12}					
	Passenger Cars			Light Duty Vehicles		
	Urban	Rural	Motorway	Urban	Rural	Motorway
Diesel						
Pre Euro	718.3	407.9	230.9	671.2	852.4	482.7
Euro 1	232.0	202.0	185.0	344.5	225.8	267.5
Euro 2	215.0	189.0	173.0	390.8	296.7	528.9
Euro 3	211.0	171.0	162.0	431.9	254.7	375.6
Euro 4	207.0	157.8	96.1	230.9	136.2	200.8
Euro 5	0.094	0.068	0.352	0.094	0.068	0.352
Euro 6	0.094	0.068	0.352	0.094	0.068	0.352
PFI						
Pre Euro	5.120	3.770	1.320	5.120	3.770	1.320
Euro 1	5.120	3.770	1.320	5.120	3.770	1.320
Euro 2	5.120	3.770	1.320	5.120	3.770	1.320
Euro 3	0.166	0.198	0.134	0.166	0.198	0.134
Euro 4	0.166	0.198	0.134	0.166	0.198	0.134
Euro 5	0.166	0.198	0.134	0.166	0.198	0.134
Euro 6	0.166	0.198	0.134	0.166	0.198	0.134
G-DI						
Euro 3	18.70	11.80	6.200	18.70	11.80	6.200
Euro 4	18.70	11.80	6.200	18.70	11.80	6.200
Euro 5	18.70	11.80	6.200	18.70	11.80	6.200
Euro 6	13.09	8.260	4.340	13.09	8.260	4.340
Euro 6-GPF	1.309	0.826	0.430	1.309	0.826	0.430

Table 6: Elemental carbon content of PM emitted from port fuel injection gasoline and conventional (non-DPF equipped) diesel vehicles, assumed in the COPERT model.

Technology	Non-DPF Diesel	PFI
Pre Euro	55%	30%
Euro 1	70%	25%
Euro 2	80%	25%
Euro 3	85%	15%
Euro 4	87%	15%

4.2.1 Fuel consumption emission factors

In order to estimate the external (society) and internal (motorist) costs associated with the potential CO_2 penalty resulting from a GPF installation, it is necessary to estimate the fuel consumption of future G-DI vehicles. This is a rather challenging task considering the direction taken towards decarbonisation of the road transport and the target set for a fleet average CO_2 emission of 95 g/km in 2020 [29]. While downsized direct injection vehicles will play an important role, it is anticipated that this ambitious target will necessitate significant electrification, hybridization and potentially introduction of fuel-cells [31].

The COPERT 4 model incorporates fuel consumption emission factors for petrol PCs up to Euro 4 technology and petrol LDVs up to Euro 1 stage, which practically are representative of PFI technologies. For the purpose of the present study, the expected fuel consumption benefits of G-DIs and vehicle downsizing were not considered in the analysis. That is, it was assumed that Euro 6 G-DI vehicles will emit the same amount of CO_2 with Euro 4 petrol PCs and Euro 1 gasoline LDVs. This conservative assumption is expected to result in an overestimation of the true CO_2 emissions.

The COPERT 4 model provides emission factors as a function of the average vehicle speed. Accordingly, average vehicle speeds of 20 km/h, 60 km/h and 100 km/h were assumed as representative of urban, rural and motorway driving conditions, respectively. The COPERT 4 model also includes a methodology to estimate the cold start effect on the fuel consumption as a function of the ambient temperature and average trip length. Application of the methodology using the annual average daily temperature [79] of the EU27 capitals suggest a 10% (±2%) increase of fuel consumption under urban conditions. Accordingly, a fixed 10% increase was employed in the calculations performed in the present study. The resulting base case emission factors are summarized in Table 7. Table 7 also shows the associated CO_2 emissions assuming that the carbon content of the fuel is fully oxidized into CO_2 based on the carbon balance for 5% ethanol/gasoline blends [13, 80]:

$$CO_2 = FC \frac{44.011}{12.011 + 1.008 \cdot 1.89 + 16.000 \cdot 0.016}$$

The backpressure introduced by optimized GPF systems was assumed to result in a 0.5-1.5% increase of fuel consumption over motorway driving. Active regeneration is expected to be required only in cases of extensive driving under urban conditions. In order to calculate the associated fuel consumption penalty some rough calculations were performed drawing from DPF applications. A DPF-equipped diesel is assumed to regenerate every 500 km with the regeneration event lasting 20 min, resulting in a 30% increase of CO_2 emissions (experimental data at JRC suggest a 20-25% increase). G-DI vehicles emit much lower soot concentrations, and considering that the regulated solid particles are mainly soot, the actual soot emissions over urban driving are 18.7/207 times lower from conventional diesels (Table 5). The GPF volumes is also expected to be approximately half that of their DPF counterparts and therefore should have half their storage capacity. Accordingly, a GPF would require active regeneration every 500×207/18.7/2=2750 km. However, since the G-DI engines would be generally throttled under urban driving, it is anticipated that the manufacturers would avoid having the GPF heavily loaded. In that respect, it is assumed that active regeneration would occur every 1500 km. It needs to be stressed though that the whole analysis assumes that during this long driving distance the vehicle will not be driven over rural or motorway conditions where the high exhaust temperatures would initiate passive regeneration. Therefore, this should correspond to a worst case scenario. Finally assuming a 10 min regeneration event (half that of the diesel due to the lower volume and soot loading) at 50 km/h, the average distance driven during active regeneration events is approximately

8.5 km. Accordingly a 30% increase of the fuel consumption over active regenerations would result in a (1.3×8.5+1500)/1508.8=0.16% fuel consumption increase under urban driving over the service life of the vehicle. The resulting emission factors are also summarized in Table 7.

Table 7: Fuel consumption and ultimate CO_2 emission factors for the different classes of Euro 6 G-DI vehicles, with and without GPF.

Category	Fuel Consumption [g/km]			Utlimate CO_2 [g/km]		
	Urban	Rural	Motorway	Urban	Rural	Motorway
Base case Euro 6 G-DI						
SPC	78.8	44.2	48.1	244.8	137.2	149.5
MLPC*	103.8	59.3	58.6	322.3	184.2	182.0
LDV	148.3	73.7	74.9	460.6	228.7	232.4
Euro 6 G-DI with GPF						
SPC	78.9	44.2	48.3-48.8	245.2	137.2	150.2-151.7
MPC*	104.0	59.3	58.9-59.5	322.8	184.2	182.9-184.7
LDV	148.5	73.7	75.3-76.0	461.3	228.7	233.6-235.9

*a weighted average figure was employed according to the relative EU27 population of MPC and LPC, averaged over the 2015-2030 timeframe.

4.2.2 Black Carbon emission factors

There is a growing consensus in the scientific community that Black Carbon (BC) is a strong climate forcer. Late studies suggest a 100 to 2000 times higher global warming potential of BC compared to CO_2 [81]. A literature survey conducted by JRC suggested that most of the PM emitted by G-DI vehicles is Elemental Carbon (EC/PM=70-90%). Although there exist some uncertainties in translating EC to Black Carbon (BC), the former is expected to be at most equal to BC (BC/EC estimates range from 1 to 3 - [82]). The COPERT 4 methodology has only information on the PM emissions from Euro 3 technology G-DI vehicles. In order to derive BC emission factors for Euro 6 technology G-DI vehicles, it was assumed that 70% of the emitted PM is EC, which additionally was considered to be BC. Furthermore, a 30% reduction in the base case Euro 6 BC emissions was assumed over the Euro 5 vehicles, in line with the reduction employed in the solid PN emissions, while GPF systems were assumed to be 90% efficient in reducing BC. The resulting emission factors are given in Table 8.

Table 8: Black Carbon emission factors for Euro 6 G-DI vehicles with and without GPF.

BC [mg/km]	Urban	Rural	Motorway
Euro 6 G-DI	3.7	1.7	3.7
Euro 6 G-DI with GPF	0.37	0.17	0.37

4.3 Scenario formulation

In order to assess the environmental benefit that would result from the installation of GPFs in G-DI vehicles, three scenarios were formulated:

- Scenario 1 (baseline): This baseline scenario reflects the evolution of particulate emissions based on the measures decided thus far. It therefore does not include any particle number limit for G-DI vehicles that would necessitate further control of their particulate emissions. The effectiveness of different legislative approaches tackling particle number emissions of this vehicle category could then assessed by means of quantifying the emission reduction they would bring over this basecase scenario.

- Scenario 2 (diesel PN at a Euro 6 stage): This scenario assumes that the diesel PN limit will also apply to G-DI vehicles at a Euro 6 stage (09/2014 for PC and 09/2015 for LDV), requiring the installation of GPF systems to all G-DIs.

- Scenario 3 (diesel PN 3 years after the Euro 6 stage): This scenario assumes that the diesel PN limit will also apply for G-DI vehicles but will be introduced three years after the Euro 6 standards enter into force. Again, it is assumed that such a limit will necessitate the use of a GPF in all G-DI vehicles. It is recognized that in such a long time-frame, most probably some manufacturers will manage to achieve the diesel PN limit through internal engine measures. However, there is no information yet on the emission performance of such vehicles under real world conditions and over the lifetime of the vehicle, so it is impossible to assess their emission performance at this point.

4.4 Results

4.4.1 Baseline scenario

Figure 9 shows the projected evolution of the total number of non-volatile particles emitted annually in EU27 by the three main vehicle categories (diesel, PFI and G-DI), under the baseline scenario.

The simulations suggest that diesel vehicles will remain the main contributor to ambient PN emitted from passenger cars over the time frame examined (1995 to 2030). Interestingly, the total PN emitted from this vehicle category remained relatively stable from 1995 to 2009, despite the almost tenfold reduction in the PM limits over this period. This was a direct consequence of the increasing penetration of diesels in the European market. From 2010 onwards, however, the widespread use of DPFs driven by the PN regulations is expected to effectively reduce their emissions by more than one order of magnitude in 2030.

The total emitted PN from PFI vehicles was also reduced by approximately 80% from 1995 (6.6×10^{24} #/year) to 2010 (1.4-1.5×10^{24} #/year) due to both the decreased penetration of gasoline vehicles in Europe (resulting in a 30-38% reduction in the total annual mileage driven by this vehicle category) and the improved PM performance brought by the better control of the air/fuel ratio. Any further reductions after 2010 will be mostly related to the replacement of this category by the more fuel efficient G-DI vehicles.

A distinctly different trend is observed for G-DI vehicles, whose contribution to ambient PN is found to sharply increase from approximately 10^{24} #/year in 2007 to 8-16×10^{24} #/year in

2030, approaching or even exceeding the emissions from diesel vehicles. This is a direct consequence of their increased penetration in the market.

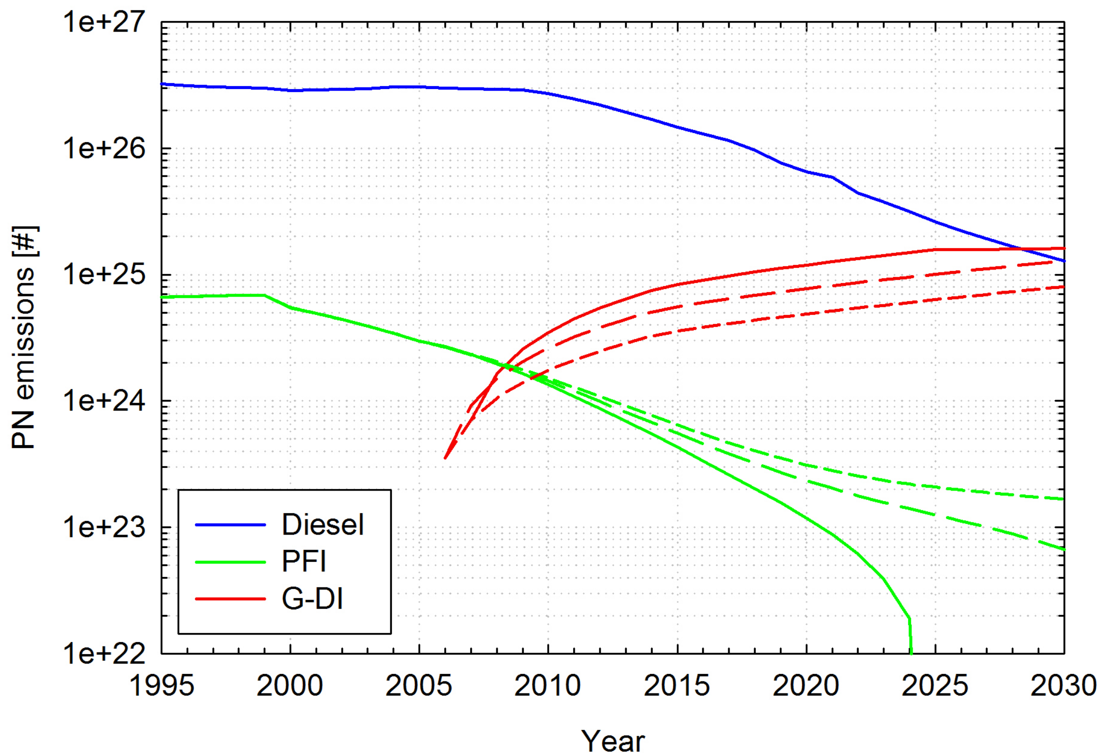

Figure 9: Evolution of solid PN emissions from PC and LDV in Europe according to the baseline scenario. The three different line types correspond to the different projections of G-DI market share.

Figure 10 shows the evolution of the fleet average PN emission rates of diesels, PFIs and G-DIs. These figures were simply calculated by dividing the total PN emitted yearly to the total annual mileage of the corresponding vehicle category. Fleet-average PN emission rates from PFIs progressively decrease as more efficient vehicles gradually replace older technology cars. The effect of this transition on the fleet average emissions takes more than two decades, due to the relatively long vehicle service life. This is more evident in the case of diesels, where a non-DPF equipped vehicle present in 2030 would emit at least 3 orders of magnitude more solid particles than a DPF equipped vehicle. Stated differently, the emissions of single non-DPF equipped vehicle would be equivalent to those of more than 1000 DPF equipped diesels. Accordingly, the fleet-average emissions from diesel vehicles in 2030 are projected to be 7.3×10^{12} #/km, i.e. 98.6% higher the average emission factor of DPF equipped diesels (9.9×10^{12} #/km).

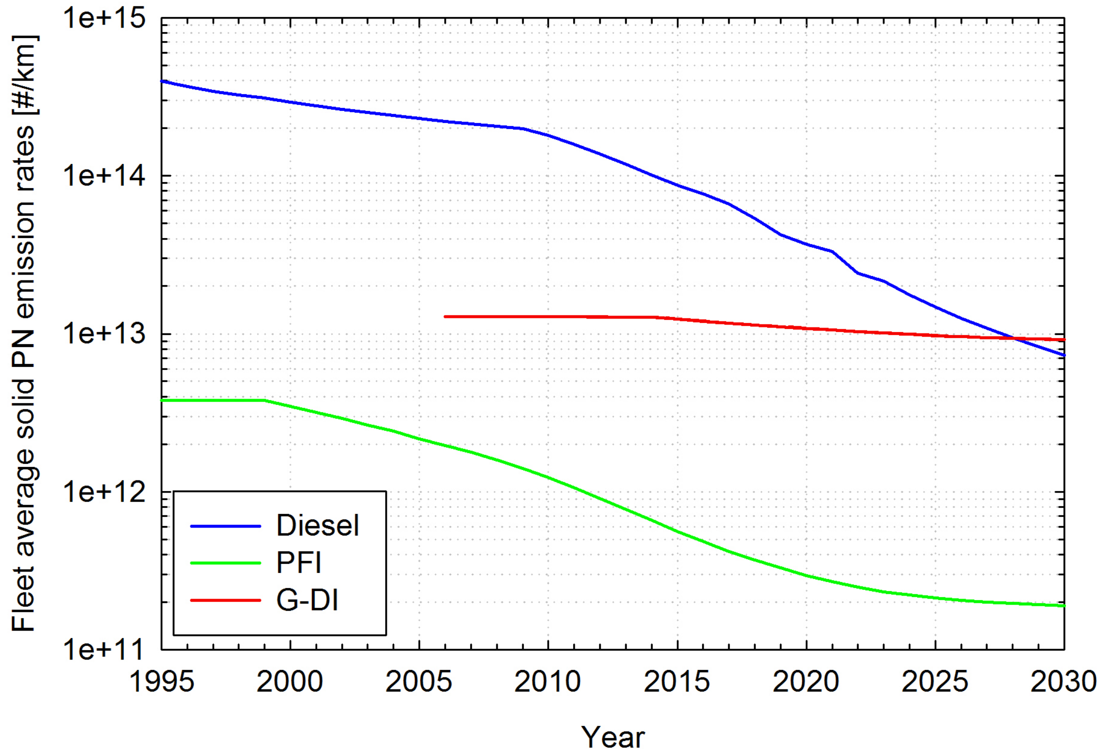

Figure 10: Evolution of fleet average solid PN emission rates for the three main vehicle categories under the baseline scenario.

4.4.2 Environmental benefit resulting from the installation of GPF

Figure 11 compares the evolution in the total non-volatile PN emitted annually in EU27 by G-DI vehicles under the baseline case (no GPF installation) and the two scenarios investigated for GPF installation, namely mandatory use at a Euro 6 stage (2015 for PCs, 2016 for LDVs) or three years after the Euro 6 stage enters into force (2018 for PCs, 2019 for LDVs). Results are shown separately for the three market penetration scenarios examined.

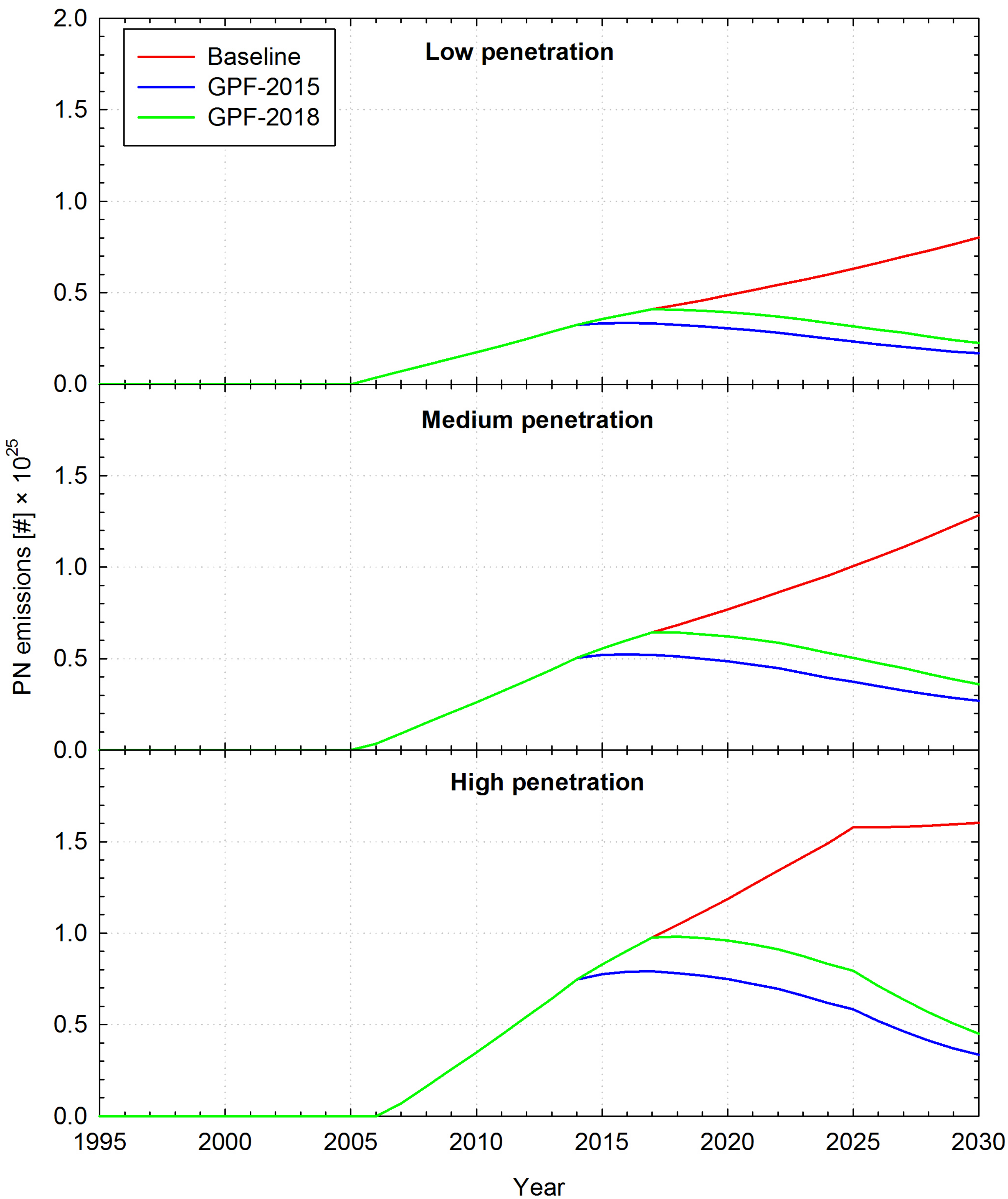

Figure 11: Effect of GPF installation on the evolution of the total PM emitted from G-DI vehicles for three different market penetration scenarios (different panels) and two possible implementation dates (different line colors).

The projections suggest that the introduction of a GPF will immediately halt the rise of the total non-volatile particle emissions. Following the introduction of GPFs, the PN emissions will be gradually reduced as old non-GF equipped G-DIs will be progressively scrapped. The installation of a GPF at a Euro 6 stage will effectively reduce the PN levels in 2030 by 79%. A three years later introduction of GPF will result in a 72% reduction of the PN levels in 2030.

The annual reduction in the total non-volatile PN emitted by G-DI vehicles for the different scenarios investigated are summarized in Table 9 and Table 10. The tables also present the PN reductions as a fraction of the total non-volatile particles emitted from passenger cars and light duty vehicles.

Table 9: Projected total PN reductions from the installation of GPFs at a Euro 6 stage for the three penetration scenarios examined.

Year	Low Penetration		Medium Penetration		High Penetration	
	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]
2015	2.25E+23	0.15%	3.51E+23	0.23%	5.24E+23	0.34%
2016	4.83E+23	0.36%	7.57E+23	0.56%	1.14E+24	0.83%
2017	7.74E+23	0.66%	1.22E+24	1.01%	1.85E+24	1.51%
2018	1.09E+24	1.09%	1.72E+24	1.69%	2.63E+24	2.52%
2019	1.43E+24	1.79%	2.26E+24	2.76%	3.47E+24	4.11%
2020	1.79E+24	2.64%	2.84E+24	4.08%	4.39E+24	6.08%
2021	2.20E+24	3.55%	3.48E+24	5.49%	5.40E+24	8.19%
2022	2.61E+24	5.54%	4.15E+24	8.52%	6.45E+24	12.64%
2023	3.05E+24	7.54%	4.86E+24	11.56%	7.58E+24	17.12%
2024	3.50E+24	10.27%	5.57E+24	15.71%	8.72E+24	23.20%
2025	3.97E+24	13.89%	6.33E+24	21.17%	9.93E+24	31.12%
2026	4.44E+24	18.13%	7.09E+24	27.55%	1.06E+25	38.70%
2027	4.92E+24	22.92%	7.86E+24	34.76%	1.12E+25	46.83%
2028	5.40E+24	28.93%	8.63E+24	43.77%	1.17E+25	56.66%
2029	5.87E+24	35.61%	9.39E+24	53.76%	1.22E+25	67.02%
2030	6.34E+24	43.10%	1.01E+25	64.95%	1.27E+25	78.15%

Table 10: Projected total PN reductions from the installation of GPFs three years after the Euro 6 stage enters into force for the three penetration scenarios examined.

Year	Low Penetration		Medium Penetration		High Penetration	
	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]	PN reduction [#]	PN reduction [% of total PN emissions from PCs & LDVs]
2018	2.63E+23	0.26%	4.15E+23	0.40%	6.35E+23	0.60%
2019	5.79E+23	0.72%	9.15E+23	1.10%	1.41E+24	1.63%
2020	9.29E+23	1.35%	1.47E+24	2.07%	2.27E+24	3.06%
2021	1.32E+24	2.10%	2.09E+24	3.22%	3.24E+24	4.76%
2022	1.73E+24	3.61%	2.75E+24	5.49%	4.28E+24	8.03%
2023	2.18E+24	5.26%	3.47E+24	7.98%	5.41E+24	11.65%
2024	2.64E+24	7.56%	4.20E+24	11.41%	6.57E+24	16.55%
2025	3.14E+24	10.66%	5.00E+24	16.01%	7.85E+24	23.09%
2026	3.64E+24	14.39%	5.81E+24	21.51%	8.67E+24	29.65%
2027	4.16E+24	18.70%	6.64E+24	27.87%	9.45E+24	36.90%
2028	4.69E+24	24.21%	7.49E+24	35.95%	1.02E+25	45.80%
2029	5.23E+24	30.52%	8.36E+24	45.22%	1.09E+25	55.61%
2030	5.76E+24	37.73%	9.22E+24	55.79%	1.15E+25	66.38%

5 IMPLEMENTATION COST

5.1 Methodology

5.1.1 Background

Generally, the implementation cost of a new technology consists of some direct costs associated with the construction of the product (like raw material, labour, energy costs, etc.) but also includes several indirect costs due to changes in other parts of a company's processes. Such indirect costs can be as diverse as research and development, hiring new staff, retooling and training of salespeople among others, and perhaps more importantly depend on the complexity of the end product/technology as well as the time frame involved [83].

A straightforward approach would be to teardown a technology into the fundamental materials, labour and capital required to manufacture it, and then to estimate the cost of every single component and every step in the manufacturing process [19, 84, 85]. However this method is costly and very difficult to apply as it requires a thorough knowledge of and experience with automotive manufacturing processes [19, 86]. Furthermore, such teardown studies are difficult to apply to new technologies that have not yet been implemented in a mass-production, whose designs are not yet finalized and whose impact on changing related parts is not yet known [19].

An alternative, simpler approach to calculate indirect costs is the use of Retail Price Equivalent (RPE) mutlipliers, widely employed in the automotive industry [83]. The RPE is defined as the ratio of the total revenue of a company to the direct manufacturing cost [83] and as such, it takes into account, manufacturing costs, production overheads (research and development, warranty, depreciation and amortization, maintenance, repair and operation costs), corporate overhead (general and administrative, retirement, health care), selling (transportation and marketing), dealer and profit [83]. The RPE multipliers are intended to represent long-run, high-volume, industry-average production costs [19].

Reported RPE values [83, 87, 88, 89, 90, 91, 92] are in the range of 1.26 and 2.0. This relatively large uncertainty in the RPE estimations reflects inconsistencies in the methodologies and assumptions employed, but also differences in the complexity of the technologies investigated. For example, Vyas et al. [87] derived an RPE multiplier of 1.5 for outsourced components and an RPE value of 2.0 for internally developed components.

The RPE methodology has three major disadvantages:

- It does not account for the complexity of the technology since it assumes that every single increase in the production cost will have the same percentage increase in the retail price.
- It also assumes that every technological adjustment affects all sources of indirect cost, which might not be valid.
- Finally, it includes all forms of indirect costs for a manufacturer, regardless of whether all of these costs change in response to the regulatory action.

Rogozhin et al. [83] amended the RPE methodology in an attempt to tackle the aforementioned shortcomings. The resulting Indirect Cost (IC) multiplier methodology, employed in the present study, is described briefly in the following section.

5.1.2 Indirect cost multiplier methodology

Rogozhin et al., [83] described a methodology to ascribe the RPE multiplier to the different components of the indirect costs, discriminating also between different levels of complexity (low, medium and high) as well as between short and long time frames.

The various levels of complexity are categorized as follows [83]:

- Low complexity technologies are defined as innovations that do not affect links between core components and systems of an automobile, but just reinforce existing core components.

- Medium complexity technologies are defined as either a) changes of a core concept of a system or a component that does not change how components interact with each other or b) changes in the way in which the product's components are linked together without changing the design of the core concepts.

- High complexity technologies are defined as innovations establishing a new dominant design and, hence, a new set of core design concepts embodied in components that are linked together in a new way.

The resulting indirect cost multipliers are summarized in Table 11. Note that Rogozhin et al. [83] did not include the profit in their derivation of indirect multipliers, even though they calculated their contribution to be 0.06. The reasoning was that an increased production cost will not be fully reflected in the vehicle price due to elasticities of supply and demand. However this was criticized in later studies (e.g. [19]) on the ground that the global automotive industry approximates a monopolistically competitive market (that is a market in which there is a product differentiation but a high degree of competition among many firms), where the long-run cost of production will eventually be passed on to customer. In response, a later study by US EPA [31] has accordingly amended the IC multiplier figures to include the manufacturer profit. This is also the approach employed in the present study.

The formulation of the IC multipliers was based on the assumptions of a) full scale economies and b) sufficient large implementation times. Full-scale economies and full learning are generally considered to be reached at between 100000 and 500000 units per year [19, 93]. Projected production of G-DI vehicles from twelve European manufacturers [HIS/CSM March 2011 – information presented in the workshop] in 2014, ranged from 120000 to 2230000. These figures are large enough to ensure that the indirect implementation costs will be efficiently distributed over sufficiently large number of vehicles.

Although the IC multipliers take into account large volume production and full learning, the economic literature suggests that manufacture learning cost reduction occurs indefinitely. Accordingly, a late study by USA Environmental Protection Agency (EPA) [31] employed some additional volume-based and time-based reductions in the cost figures. However, as the same study suggested, there is a lot of uncertainty in the corrections employed and further research is required in order to better understand industry learning curves. For this reason, no additional manufacture learning cost reductions were employed here.

The same study [31] also provided some uncertainty figures for the IC multipliers. The coefficients of variance (defined as the standard deviation divided by the mean) were estimated to be 12%-13%. This uncertainty was taken into account in the cost figures derived in the present study.

Table 11: Indirect cost multipliers as a function of technology level and implementation time frame.

IC multiplier contributors	Short run effect			Long run effect		
	Low complexity	Medium complexity	High complexity	Low complexity	Medium complexity	High complexity
Manufacturing						
Cost of sales	1.00	1.00	1.00	1.00	1.00	1.00
Warranty	0.04	0.05	0.06	0.02	0.02	0.03
Research & Development	0.01	0.06	0.10	0.00	0.00	0.02
Depreciation & amortization	0.00	0.00	0.02	0.00	0.00	0.02
Maintenance, repair & operations	0.00	0.00	0.03	0.00	0.00	0.03
Corporate overhead						
General & administrative	0.00	0.00	0.03	0.00	0.00	0.03
Retirement	0.00	0.00	<0.01	0.00	0.00	<0.01
Health care	0.00	0.00	<0.01	0.00	0.00	<0.01
Selling						
Transportation	0.00	0.00	0.01	0.00	0.00	0.01
Marketing	0.00	0.04	0.05	0.00	0.00	0.00
Dealer						
Dealer contributors	0.01	0.06	0.08	0.00	0.02	0.06
Manufacturer profit						
Net income	0.06	0.06	0.06	0.06	0.06	0.06
Indirect cost multipliers						
Total contribution to cost of sales, including profit	**1.11**	**1.26**	**1.51**	**1.08**	**1.11**	**1.32**

5.2 Cost estimations

5.2.1 Formulation of the expected vehicle technologies

Limited information was provided in the workshop on the cost of the GPF technologies. All manufacturers though stated that the construction cost should be similar to that for diesel applications. It was also clear, that the automotive manufacturers do not look at the GPF as an additional exhaust component but rather as a replacement to a TWC or at least the muffler. Furthermore, the introduction of the GPF should not result in an increase of the total Platinum Group Metals (PMG) content used. This means that if the GPF will incorporate catalytic activity it should replace a TWC or reduce accordingly its size.

The cost of the GPF will depend on its volume which should be proportional to the engine displacement. Three vehicle categories were assumed, in line with the classification employed in the TREMOVE model [94] that was employed for the emission projections (section 4). These included:

- Small Passenger Cars (SPC), assumed to be equipped with a 1.4 l engine.

- Medium/Large Passenger Cars (MLPC), assumed to be equipped with a 2.0 l engine.

- Light Duty Vehicles (LDV), assumed to be equipped with a 2.5 l engine.

Since it is not clear yet what would be the required size of the GPF, separate calculations were performed for catalyst volume of 1 and 0.8 times the engine displacement. It was further assumed that the use of a catalyzed GPF would result in a reduction of the TWC volume by the volume of the GPF. In the case of MLPC and LDV, separate calculations were performed for the case of a single and dual exhaust system.

It was further assumed that active regeneration of the GPF, if needed should be achieved through retarder (split) fuel injection and spark timing in stoichiometric concepts and through post fuel injection in lean burn engines, approaches already employed in G-DIs to warm up the catalyst during cold start operation. It is not certain however whether there will be a need for active regeneration, especially in the case of catalyzed GPFs. In that respect, the cost of the equipment required to monitor the status of the GPF was included in the uncertainty range. The necessary equipment was assumed to be a differential pressure sensor monitoring the pressure drop in the GPF and a thermocouple monitoring the temperature upstream of the GPF (twice as many in the case of two exhaust lines), as in diesel applications [95].

Accordingly, two levels of complexity could be identified (section 5.1.2):

- <u>Low complexity technology</u>, corresponding to a straightforward application in which the GPF will be installed without any kind of monitoring of its status and no need for active regeneration. This was only considered for the case of catalyzed GPFs.

- <u>Medium complexity technology</u>, which will require some kind of interaction with the ECU (through pressure and temperature sensors) to initiate active regenerations by means of retarded (split) fuel injection and spark timing or post fuel injection.

5.2.2 Time frame

The introduction of a new vehicle technology in mass production requires generally 2 to 3 years [19, 23], to allow for the technology to pass through the normal product cycles (redesign and product turnover schedules). Accelerated rates of implementation can increase costs by decreasing amortization periods and by demanding more engineering and design resources than those available. This period of 2-3 years is rather considered as the quickest possible timeframe, requiring significant carry-over technology and engineering from other models. Given the accumulated experience from diesel applications, it is expected that the automotive manufacturers will be in a position to introduce GPF at a Euro 6 stage, if a number limit mandating the use of GPFs will be established by the end of 2011.

Longer implementation times of 4 to 8 years are required for more substantial changes, especially those requiring engine development [19]. While compliance with the diesel PN limit was demonstrated with prototype engines utilizing advanced injection systems [36], and G-DI engines utilizing advanced piezo-injectors are already available in the market [96, 97, 98], it is expected that further advancements (e.g. variable area or spray angle and narrower cone angles [57]) will be required to effectively reduce PN emissions through engine measures. The development of such technologies is anticipated to require at least six years.

Conclusively, it was assumed that introduction of a GPF system at a Euro 6 stage will correspond to the minimum anticipated time frame, having a short-run effect on the vehicle price. A possible implementation date of late 2017 should allow sufficient time for the vehicle manufacturers to incorporate a GPF in the most efficient way, having a long-run effect on the vehicle price (see section 5.1.2). Actually, a 2017 implementation date, will almost certainly allow compliance of some vehicles through less expensive internal engine measures, even if the quantification of the cost-effectiveness improvement could not be quantified in the present study.

5.2.3 Cost elements

Limited information on the GPF technology cost was provided in the workshop, with most manufacturers suggesting that the cost should be similar to or even lower than that of DPFs. In that respect, estimations were performed using cost information available from studies on DPF systems. Unfortunately, most of these studies provide price increase figures associated with the introduction of a DPF system without discriminating between the contribution of the different components.

To our knowledge, the only studies providing detailed information of the volume-dependent cost of particulate filters are the US EPA regulatory impact analysis on 2007 model year heavy duty diesel engines [99] and the U.S. EPA regulatory impact analysis on non road diesel engines [100]. According to these studies, the substrate cost per litre for a DPF is 62 $\$/dm^3$ (2002 prices) for silicon carbide and 31 $\$/dm^3$ (2002 prices) for cordierite. The substrate cost for a catalytic converter is 5.4 $\$/dm^3$ (2002 prices) while the total cost of washcoating and canning is 23 $\$/dm^3$ (2002 prices) for both catalyzed particulate filters and catalytic converters. According to the catalyst manufacturers however, the fabrication of the washcoat is more expensive for wall flow particulate filters than for flow through TWCs. Accordingly a 20% lower washcoating cost was assumed for TWCs (18.4 $\$/dm^3$).

The aforementioned studies also provided cost figures for mufflers installed in diesel engines of 6, 8 and 13 dm^3 total swept volume, averaging at 6 $ per engine volume. The same study also included some cost figures on the can housing, which was assumed to be constant at 7 $ (2002) for catalyzed DPFs smaller than 7 l. Using an average USA$ to € exchange rate of 1.2 $/€ (1999-2010) and a harmonized index of consumer prices for transport in EU of 2.88% (1999-2011) [101], 1 $ in 2002 corresponds to 1.06 € in 2011.

Cost estimates for the differential pressure sensor and thermocouples were taken from a recent study by the National Research Council (NRC) [19]. The particular study suggested a 13 $ (2007 prices) cost for a thermocouple and a 25 $ (2007 prices) for the differential pressure sensor. Again, using the above average statistical data from the European Central Bank [101], 1 $ in 2007 corresponds to 0.93 € in 2011.

The above figures, suggest a cordierite DPF brick (substrate, washcoat & canning but no PMG) cost of 114 € (2011) and 200 € (2011) for a 2 l and 3.5 l volume. The corresponding costs of silicon carbide DPFs are 180 € (2011) and 315 € (2011), respectively. The calculated figures are in good agreement with the study of NRC [19] suggesting a cost of 124 $ (2007)

and 270 $ (2007), that is 115 € (2011) and 250 € (2011), respectively, for advanced cordierite DPFs of the same volume.

The cost figures employed in the present work are summarized in Table 12. Note that from this point onwards all cost and price figures are expressed in € in 2011 monetary terms, unless otherwise stated.

Table 12: Cost elements employed in the study. Cost figures are in € (2011 monetary terms)

Element	Cost
Substrate cost per GPF volume (€/dm^3)	32.8-65.3
Washcoating and canning cost per GPF volume (€/dm^3)	24.2
Washcoating and canning cost per TWC volume (€/dm^3)	19.5
Can Housing cost per GPF or TWC brick (€/GPF or €/TWC)	7.4
Muffler cost per GPF volume (V) (€/dm^3)	6.36×V
Differential pressure sensor cost (€)	23.1
Thermocouple cost (€)	12.0

5.2.4 Calculated vehicle price increase

The results of the cost calculations following the approach and assumptions described in the previous section are summarized in Table 13 to Table 15. The calculated price increase associated with the introduction of a GPF at a Euro 6 stage ranged from 39-163 € for a small passenger car to 70-303 € for a light commercial vehicle. The corresponding figures for a three years delay in the implementation ranged from 38-142 € to 68-263 €, respectively.

For comparison, the system integration costs suggested by AECC [102] ranged from 40 to 130 € depending on engine size, production volume and packaging options, including engines with a dual exhaust system. Similarly, a recent study by the International Council on Clean Transportation [103]. suggested GPF production costs in the range of 62 to 131 € for the engine sizes considered in the present study. The same study quoted cost estimated from the Manufacturers of Emissions Controls Association in the order of 35 to 70 €. Translation of these figures to price increase, through the indirect cost multipliers, yield an estimate of 38 to 169 € for a 2014 implementation. Therefore, published cost estimates lie in the low to mid range of the calculations in the present study, suggesting that the GPF installation can be twice as much cost efficient from what our calculations suggests.

Table 13: Calculated price increase associated to the introduction of a GPF system in G-DI SPC.

SPC	Uncoated GPF		Coated GPF	
	Min	Max	Min	Max
Exhaust lines / Number of GPFs	1	1	1	1
Engine displacement [dm3]	1.40	1.40	1.40	1.40
GPF volume [dm3]	1.12	1.40	1.12	1.40
Incremental production costs				
GPF substrate cost [€]	36.80	92.01	36.80	92.01
GPF canning [€]	7.42	7.42	-	-
GPF washcoat [€]	-	-	19.89	19.89
Pressure sensor [€]	23.25	23.25	-	23.25
Thermocouple [€]	12.09	12.09	-	12.09
Decremental production costs				
Muffler cost reduction [€]	7.12	8.90	-	-
TWC substrate cost reduction [€]	-	-	6.41	8.01
TWC washcoat cost reduction [€]	-	-	14.42	19.89
Total production cost				
Total implementation cost [€]	72.44	125.86	35.85	126.16
Implementation at a Euro 6 stage				
Vehicle price increase [€]	**88.83**	**162.84**	**39.28**	**163.23**
Implementation 3 years after the Euro 6 stage				
Vehicle price increase [€]	**79.37**	**141.51**	**38.35**	**141.84**

Table 14: Calculated price increase associated to the introduction of a GPF system in G-DI MLPC.

MLPC	Uncoated GPF		Coated GPF	
	Min	Max	Min	Max
Exhaust lines / Number of GPFs	1	2	1	2
Engine displacement [dm3]	2.00	2.00	2.00	2.00
GPF volume [dm3]	1.60	1.00	1.60	1.00
Incremental production costs				
GPF substrate cost [€]	52.58	131.44	52.58	131.44
GPF canning [€]	7.42	14.84	-	-
GPF washcoat [€]	-	-	31.59	33.92
Pressure sensor [€]	23.25	46.50	-	46.50
Thermocouple [€]	12.09	24.18	-	24.18
Decremental production costs				
Muffler cost reduction [€]	10.18	12.72	-	-
TWC substrate cost reduction [€]	-	-	9.16	11.45
TWC washcoat cost reduction [€]	-	-	23.79	24.17
Total production cost				
Total implementation cost [€]	85.16	204.24	51.22	200.42
Implementation at a Euro 6 stage				
Vehicle price increase [€]	**104.42**	**264.25**	**56.12**	**259.31**
Implementation 3 years after the Euro 6 stage				
Vehicle price increase [€]	**93.31**	**229.63**	**54.78**	**225.34**

Table 15: Calculated price increase associated to the introduction of a GPF system in G-DI LDV.

LDV	Uncoated GPF		Coated GPF	
	Min	Max	Min	Max
Exhaust lines / Number of GPFs	1	2	1	2
Engine displacement [dm3]	2.50	2.50	2.50	2.50
GPF volume [dm3]	2.00	1.25	2.00	1.25
Incremental production costs				
GPF substrate cost [€]	65.72	164.30	65.72	164.30
GPF canning [€]	7.42	14.84	-	-
GPF washcoat [€]	-	-	41.34	46.11
Pressure sensor [€]	23.25	46.50	-	46.50
Thermocouple [€]	12.09	24.18	-	24.18
Decremental production costs				
Muffler cost reduction [€]	12.72	15.90	-	-
TWC substrate cost reduction [€]	-	-	11.45	14.31
TWC washcoat cost reduction [€]	-	-	31.59	33.92
Total production cost				
Total implementation cost [€]	95.76	233.92	64.02	232.86
Implementation at a Euro 6 stage				
Vehicle price increase [€]	**117.42**	**302.65**	**70.15**	**301.27**
Implementation 3 years after the Euro 6 stage				
Vehicle price increase [€]	**104.92**	**263.00**	**68.48**	**261.80**

6 UTILIZATION COST

A potential fuel consumption penalty will also result in an additional internal cost for the vehicle owner. The price of petrol in Europe is subject to high taxation, either in the form of excise taxes on oil products or in the form of value added tax applicable to all consumer goods. Since this taxation serves for the common welfare, and therefore is redistributed across each Member State's citizens, it does not constitute a net cost to the society. In that respect, only the pure industrial cost of the additional petrol fuel consumed, is considered in the analysis.

The pre-tax petrol price closely follows the cost of crude oil and shows little variation across the EU Member States since both the crude oil and its derivatives are traded on the global market. Latest cost estimates for tax-free petrol across EU27 are in the order of 0.55 $€_{2011}$/lt [104]. It is impossible to project the evolution of crude oil price in the future, especially considering that significant changes may affect the vehicle usage and even the market sales. Such investigations however were outside the scope of the present study, in which a fixed price of 0.55 $€_{2011}$/lt was employed.

In order to calculate the total additional fuel cost resulting from the introduction of a GPF, it is necessary to estimate the total mileage driven under urban, rural and motorway conditions over the service life of the vehicle. The starting point was to calculate the total service life of the different vehicle categories, by means of convoluting the age-dependent population (Figure 7) and driven mileage (Figure 8) of gasoline vehicles. This calculation suggested an average useful life of 130000 km for SPC, 135000 km for MLPC and 150000 km for LDVs. These figures were subsequently apportioned to urban, rural and motorway driving, by means of weighting the figures specified in Table 3 by the vehicle population in each country. The fuel consumption penalty can then be calculated as the difference between the total fuel consumed from a non-GPF vehicle to that from a GPF equipped vehicle. The results of these calculations are summarized in Table 16.

The calculations suggest an additional cost of 6.3-13 € for SPCs, 9.3-18.3 € for MLPCs and 15.7-43.1 € for LDVs. The higher figures for LDVs are due to both the higher fuel consumption and the higher (more than double) motorway mileage covered by this vehicle category.

Table 16: Calculated fuel cost increase over the lifetime of the G-DI vehicles resulting from the installation of a GPF.

	SPC			MLPC			LDV		
	Urban	Rural	Motor.	Urban	Rural	Motor.	Urban	Rural	Motor.
Total useful life [km]	130000			135000			150000		
Share [%]	25.0%	60.5%	14.5%	29.2%	55.3%	15.5%	7.5%	59.3%	33.3%
Useful life [km]	32500	78650	18850	39420	74655	20925	11250	88950	49950
Fuel consumption – Non-GPF									
FC [g/km]	78.8	44.2	48.1	103.8	59.3	58.6	148.3	73.7	74.9
Fuel consumption – GPF									
Min FC [g/km]	79.0	44.2	48.4	104.0	59.3	58.9	148.6	73.7	75.2
Max FC [gkm]	79.0	44.2	48.9	104.0	59.3	59.5	148.6	73.7	76.0
Total additional fuel consumed over the useful life									
Min total fuel [kg]	8.6			12.7			21.4		
Max total fuel [kg]	17.7			24.9			58.8		
Fuel cost increase over the useful life									
Min cost increase [€]*	**6.3**			**9.3**			**15.7**		
Max cost increase [€]*	**13.0**			**18.3**			**43.1**		

*A fuel density of 749.5 kg/m^3 was employed in the calculations, which corresponds to the average density of the reference fuels in the European regulation [14].

7 ENVIRONMENTAL BENEFIT FROM PARTICLE NUMBER REDUCTIONS

7.1 Background

Road transport activities give rise to health and environmental impacts, the costs of which are generally not borne by the transport users alone, and as such are referred to as externalities in economic science. Internalisation of these effects [105], that is measures taken to translocate these externalities to the transport users, requires a monetary valuation of these external costs.

However, the estimation of external costs is a challenging task. Road transport is responsible for a number of externalities related to air pollution, noise, congestion, accidents, global warming and energy dependency. Furthermore, external costs of transport activities depend strongly on parameters like location, time of the day as well as vehicle characteristics.

In the case of traffic related air pollution, there is a general consensus that the best methodological approach is that of the Impact Pathway Analysis employed in the ExternE [106] and the CAFE CBA [107] methodologies. Both methodologies follow a bottom-up approach, simulating the chain from the emission of the pollutants, their dispersion and finally the response of receptors (humans, flora, materials and ecosystems). This approach has the advantage of allowing for a separate, detailed consideration of different emission sources (road transport, stack, etc.) and pollutants (primary PM, Nitrogen Oxides, etc.). It is computationally intensive though, especially for the development of representative national average figures. In such cases, simplified top-down approaches have also been developed as an alternative, especially in Switzerland [108, 109], using national data. The main difficulty in these approaches lies in the differentiation between specific traffic situations (metropolitan, urban and rural sites) and emission sources (road and rail transport, industry, households, etc.).

Once the exposure levels are determined, the detrimental effects of the different pollutants tackled are quantified, for those types of impacts for which dose-response relationships have been established. A wide range of impact categories are covered, including health effects, crop losses, damages in buildings, materials and on ecosystems (e.g. eutrophication). The damages are then translated to monetary units according to the Willingness To Pay principle, that is the maximum amount a person is willing to pay to avoid detrimental effects. While the various methodologies employed may differ in the impacts covered, there is a general consensus that the dominant effect is that on human health.

Of all pollutants, particulate matter is one of the most difficult to assess, yet it is recognized to cause the most detrimental health effects, even premature deaths. One main difficulty is the large spatial variation which is a direct consequence of the variation of exposure levels (depending on the population density and the meteorological conditions) but also reflects differences in the Gross Domestic Product (controlling the amount of money the individuals are willing to pay) across EU member states. For example, the CAFE CBA study [110] has produced country average figures for EU25, with the external cost of road transport PM varying from 4.2 to 12 €/kg for Estonia to 63 to 180 €/kg for Netherlands. These figures derived in 2005 were found to be 3 to 9 times higher from those estimated in a 2002 study conducted by the same group (BeTa) [111]. This was attributed to a certain extent, to the use of a finer mesh grid in the latter calculations. The updated figures, however, still represented something of an average of damages between rural and urban emissions.

The BeTa study provided a simplified methodology to quantify the marginal external costs of road transport PM emissions in urban areas of different populations. Application of this methodology, suggests that the health effects of particulate matter are considerably more significant in densely populated cities, as evident in Figure 12. The excess damage in metropolitan areas (cities with population larger than 500000) can be as much as 13 to 36 times higher compared to that in rural background.

Also shown in Figure 12, are the maximum and minimum estimates for EU25 derived with the CAFE CAB methodology. The uncertainty in the estimates is about a factor of 3 (87 €/kg compared to 30 €/kg). These large differences reflect the uncertainties in the monetary valuation of health impacts of PM, especially regarding mortality. The latter is usually expressed through either the Value of Statistical Life (VSL) or the Value Of a Life Year (VOLY).

A similar European study, which was based on the ExternE methodology, was conducted in 2006 (HEATCO) [112]. Recognizing the importance of population density, the study derived external costs for road transport PM for EU25 countries, discriminating between “urban” and “outside built-up” areas. The cost figures for urban environments (80 to 590 $€_{2002}$) were 2.5 to 12 times higher of that for outside-built up areas, in good agreement with what the BeTa study suggested (Figure 12).

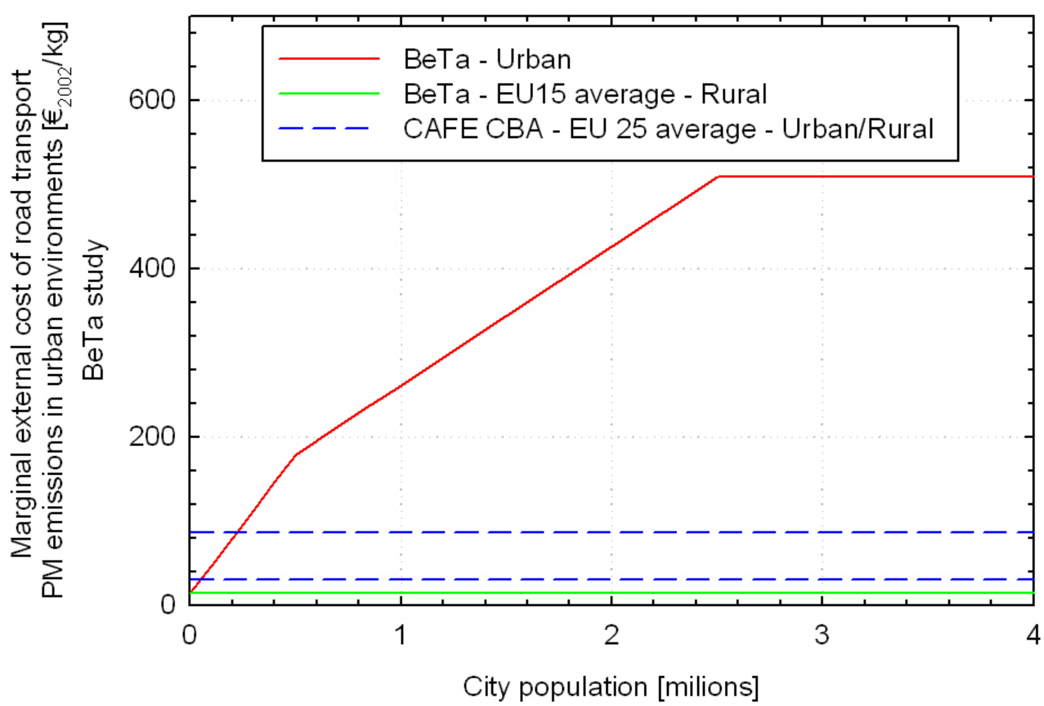

Figure 12: Dependence of the marginal external costs of road transport PM on the population of the city (red line), according to the BeTa study [111]. The EU average figures derived in the same study (green line) and the subsequent CAFE CBA study (blue lines) are also shown for reference.

Another source of uncertainty in the calculation of marginal external costs for PM, lies in the relative contribution of different PM constituents on the health effects. While both the CAFE CBA and the HEATCO studies discriminate between primary and secondary particles, attributing the health effects of secondary aerosol to the primary pollutant from which they were formed (sulphur dioxide for sulphate particles, nitorgen oxides and ammonia for nitrate aerosol), a different approach is employed for the evaluation of their relative risk rates. The need to assign the relative risk of different PM constituents comes from the fact that health effect figures are available for exposure to the total ambient PM collectively. The CAFE CBA study treats all particles as equally aggressive in lack of quantitative information on the different risk rates of potentially more hazardous compounds like metals, organic matter and endotoxins. The HEATCO study employed a latter version of the ExternE methodology [113], which based on accumulating evidence on the higher toxicity of internal combustion engines aerosol, weights differently the toxicity of the various components of ambient PM. In particular it is assumed that the toxicity of sulphate and nitrate particles is only 40% and 20%, respectively, of that of automotive exhaust particles. The toxicity of particles emitted from power plants is assumed to 40% of that of road transport PM.

Table 17: Marginal external costs for road transport PM in €$_{2011}$/kg. IMPACT values were adjusted to 2011 values following the guidelines of the HEATCO study [112 – Annex B].

Area	Metropolitan	Urban	Outside built-up areas
Austria	507.8	164.3	85.2
Belgium	528.2	170.4	114.0
Bulgaria	74.1	23.8	19.0
Cyprus	309.7	100.0	26.2
Czech Republic	308.0	99.2	76.4
Denmark	475.9	153.4	56.0
Estonia	195.7	63.7	33.0
Finland	398.6	128.4	33.2
France	477.7	153.8	95.5
Germany	446.0	143.8	87.0
Greece	327.9	105.6	46.1
Hungary	337.7	108.7	86.7
Ireland	567.0	183.0	59.3
Italy	464.3	150.1	84.5
Latvia	182.0	58.5	33.8
Lithuania	185.2	60.2	37.0
Luxembourg	864.3	278.3	123.2
Malta	307.8	98.7	25.6
Netherlands	522.1	168.6	102.1
Poland	228.5	73.3	68.6
Portugal	324.8	104.6	48.2
Romania	79.0	25.4	20.3
Slovakia	278.6	89.1	75.2
Slovenia	387.1	124.4	80.3
Spain	392.3	126.2	53.9
Sweden	419.9	135.1	40.8
United Kingdom	475.3	153.0	74.1

A recent European project (IMPACT) consolidated the results from all relevant European studies on marginal external costs in the transport sector, and derived a recommended set of methods and default values for estimating external costs [114]. Table 17 summarizes the recommended external cost figures for road transport PM, which were also employed in the present study. The proposed figures were based on the studies of HEATCO [112] and UBA [115].

The cost figures for outside built-up areas were on average 2.5 times higher (1.3 to 5.4) compared to those derived in the CAFE CBA study [110]. These higher costs, reflect to a certain extend the elevated risk rates for automotive exhaust aerosol employed in the HEATCO study.

It is interesting to compare these figures to marginal external costs suggested in different national studies. The UK Department for Environment, Food and Rural Affairs has recently proposed a UK average damage costs for road transport PM of 10-125 $£_{2010}$/kg [116], which increases to 46-590 $£_{2010}$/kg for central London. Despite the ten-fold uncertainty in the proposed figures, the mean values are in good agreement with those derived in the IMPACT study (74 $€_{2011}$/kg for outside built-up areas, and 475 $€_{2011}$/kg for metropolitan areas).

The Swiss study on the cost benefit analysis for retrofitting construction machinery with DPF systems [108, 109, 117], employed a benefit of 460 CHF_{2002} per kg of PM_{10} reduced. The figures derived in the HEATCO for Switzerland (not included in Table 17) are 76 $€_{2002}$ and 460 $€_{2002}$ for outside built-up areas and metropolitan areas, respectively. The relatively high value employed in the Swiss study, might reflect extensive use of such construction machineries in densely populated areas. However it could also point towards uncertainties in the valuation of PM health effects, especially considering that a different (top-down), approach was employed in the particular study.

7.2 Marginal external cost for particle number

There is currently no available information on the marginal external costs for Particle Number (PN). Yet there exists increasing evidence suggesting that particle number concentrations (especially those of automotive exhaust lying mostly in the ultrafine size range), are more relevant than PM from a health effects standpoint [e.g. 118, 119].

A theoretically straightforward approach would be to establish a correlation between PM and PN. Figure 13 compares PN and PM emissions from five G-DI vehicles tested at JRC over NEDC and CADC cycles. Two of them were Euro 4 certified (G-DI Lean #1 and G-DI Stoich. #1), while the remaining three were late technology Euro 5 vehicles. It is difficult to establish a clear trend from the data, with four vehicles yielding on average a PN to PM ratio of 1.5(±0.1) to $2.5(\pm0.8)\times10^{18}$ #/kg_{PM}, yet another one gave an average figure of $4.6(\pm0.6)\times10^{18}$ #/kg_{PM}.

Several reasons can be identified for this lack of a clear correlation between PM and PN. Firstly, the gravimetrically determined PM consists of both solid (mostly soot) and volatile materials, while the PN emissions measured in accordance to the regulations only target the solid core of the emitted PM. The higher the volatile fraction, the lower the PN to PM ratio gets. Furthermore, accurate quantification of the volatile fraction at such low PM emission levels becomes challenging. Several studies [120, 121, 122] have identified artefacts associated with adsorption of gaseous compounds onto the filter. The artefact strongly depends on the filter media employed [120, 122], with the TX40[2] filters collecting as high as ~4 mg/km adsorbed material, depending on the hydrocarbon emissions [120]. Teflo filters[3] are less prone to gaseous adsorption, collecting less than ~1 mg/km artefact [120]. In the results presented in Figure 13, TX40 filters were employed, and the PM emissions determined ranged between 0.5 to 4 mg/km, with the exception of two tests conducted at -7°C test cell temperature (6-7 mg/km). At such low levels, the adsorption artefact can have a strong effect on the quantified PM emissions, and correspondingly on the calculated PN to PM ratios.

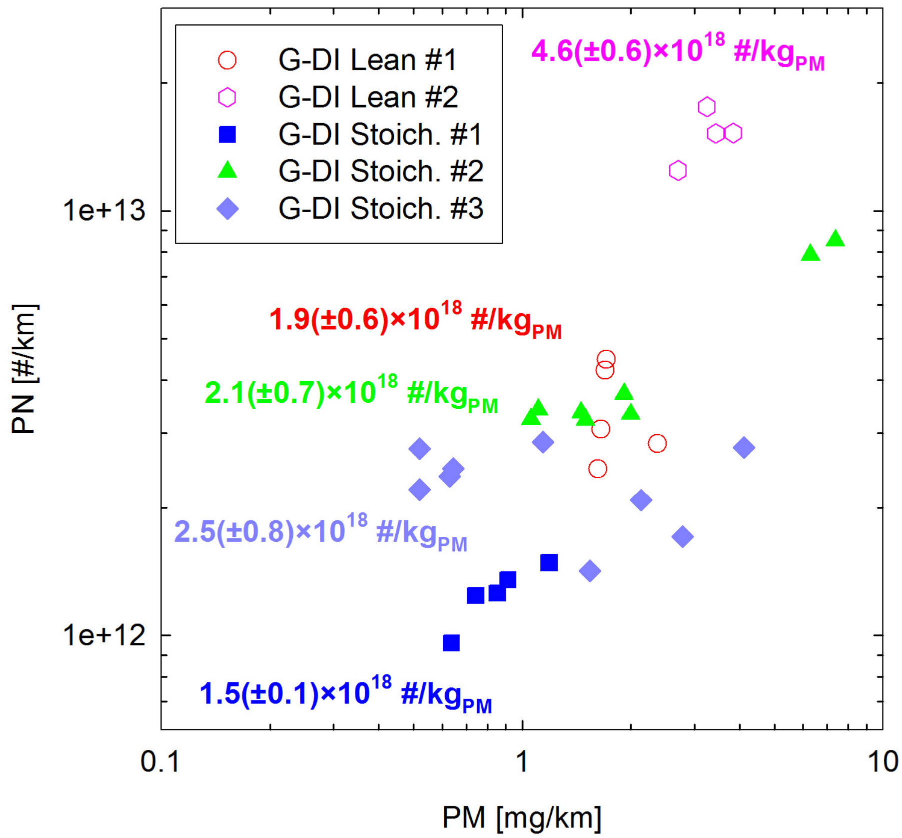

Figure 13: Comparison of solid particle number (as defined in the European regulations) and PM emissions from five G-DI vehicles tested at JRC. The numbers shown correspond to the average ± one standard deviation ratio of PN to PM for each vehicle tested.

Three recent studies [47, 121, 123] have also investigated the possibility to establish a correlation between PM and PN for late technology G-DI vehicles. All of them employed Teflon filters for the quantification of the PM mass. The studies yielded more consistent results suggesting a PN to PM ratio of 2.3×10^{18} #/kg_{PM} [47], 2.2×10^{18} #/kg_{PM} [123] and 2.0×10^{18} #/kg_{PM} [121]. It should be stressed that the system employed in two of the aforementioned studies [47, 121] for the quantification of the solid particle numbers was not compliant with the European regulations, the most important difference being the use of detectors having lower mobility cut-off size. However, the measured size distribution spectra

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[2] Teflon-coated fibreglass filter with an initial mat layer bonded to a woven layer.

[3] Expanded Teflon material stretched across a hard plastic ring.

also presented in these works, suggested limited contribution of the non-regulated sub-23 nm particles.

Yet, it is difficult to speculate that the large scatter observed in Figure 13 is a consequence of the large adsorption artefacts in the TX40 filters employed, as this could not explain the large PN to PM ratios for second lean burn G-DI vehicle. In this case, a correction for the volatile artefact would result in an even larger figure. This rather suggests that there does not exist a unique correlation between PN and PM. In line with this, the proposal by the California Air Research Board (CARB) to introduce a solid particle number limit as an alternative to the gravimetric PM procedure, assuming a correlation of 10^{18} #/kg_{PM} [124], has received strong criticism, especially with respect to the equivalence of the two limit values (i.e. the assumed correlation factor). Eventually, it was decided that the introduction of a particle number limit in USA will be assessed at a later stage (2020).

Some studies suggested that the number of solid particles correlates better with the mass of elemental carbon [47, 121], with the agreement holding down to DPF-out levels [123]. However this correlation strongly depends on the size and effective density distribution of the emitted particles, which may differ from vehicle to vehicle or may even depend on the driving conditions. The size distributions of particles emitted from G-DI vehicles are found to be approximately lognormal in shape, exhibiting a geometric standard deviation (σ_g) of 1.8-2.2 and a geometric mean diameter (d_g) of 40 to 90 nm [15, 37, 46, 47, 121, 125, 126, 127, 128, 129, 130, 131, 132]. Their effective density is found to follow a power-law dependence on the particle size, characteristic of fractal-like aggregates with a fractal dimension (DF) of 2.35±0.1 [133]. For comparison, diesel exhaust aerosol distributions are narrower (σ_g=1.6-1.9) and peak to larger sizes (55 to 100 nm). The effective density of diesel exhaust is also found to follow a power-law dependence suggesting a fractal dimension of 2.25±0.1. By combining these figures it is possible to calculate the mass of airborne particles as a function of the underlying size distribution and the effective density profile [134]. The calculated mass of airborne particles was shown to correlate very well with the mass of soot for both diesel [123, 135, 136] and G-DI [121] exhaust aerosols.

Figure 14 summarizes the results of these calculations for typical size distributions and effective density profiles reported for diesel and G-DI exhaust aerosols. The calculations suggest a 1.0-7.8×10^{18} #/kg for diesel exhaust aerosol and 0.7-11.0×10^{18} #/kg for G-DI exhaust aerosol, with the upper range of values corresponding to smaller geometric diameters and narrower distributions. The PN to PM ratios derived from the JRC data (0.6-5.4×10^{18} #/kg - Figure 13) fall well within the calculated range for G-DI vehicles. Therefore the scatter in the experimental data could just originate from differences in the size and/or structure of the emitted particles. This highlights the difficulties associated with the translation of PM emissions to PN, and correspondingly the derivation of marginal external cost figures for PN from established data on PM.

Most importantly, this seemingly straightforward application results in an intuitively misleading conclusion that smaller particles are less hazardous that larger ones, since more of them are required to accumulate the same amount of mass. There is no consensus yet as to which property/properties of PM are responsible for adverse health effects. Some studies suggest that the main toxicity arises from organic compounds attached onto the particles [137] while others identify the soot core of PM that stimulates the most adverse reaction [138]. When it comes to the non-volatile fraction that is the target of the present study, there exist numerous studies suggesting that the accessible deposited surface area is a more appropriate quantitative measure of the biological effects of solid particles [139, 140, 141, 142, 143]. For fractal aggregates (like G-DI soot), this “active” surface area available for interaction with the carrier gas (or the lung fluid) is best described by the so called Fuchs surface area [143, 144]. The Fuchs surface area shows a power law dependence on the mobility diameter, with the exponent changing from 2 (i.e. geometric surface area) in the free molecular regime (< ~5 nm particles) to 1 (i.e. particle length) in the continuum regime (supermicron particles) [145].

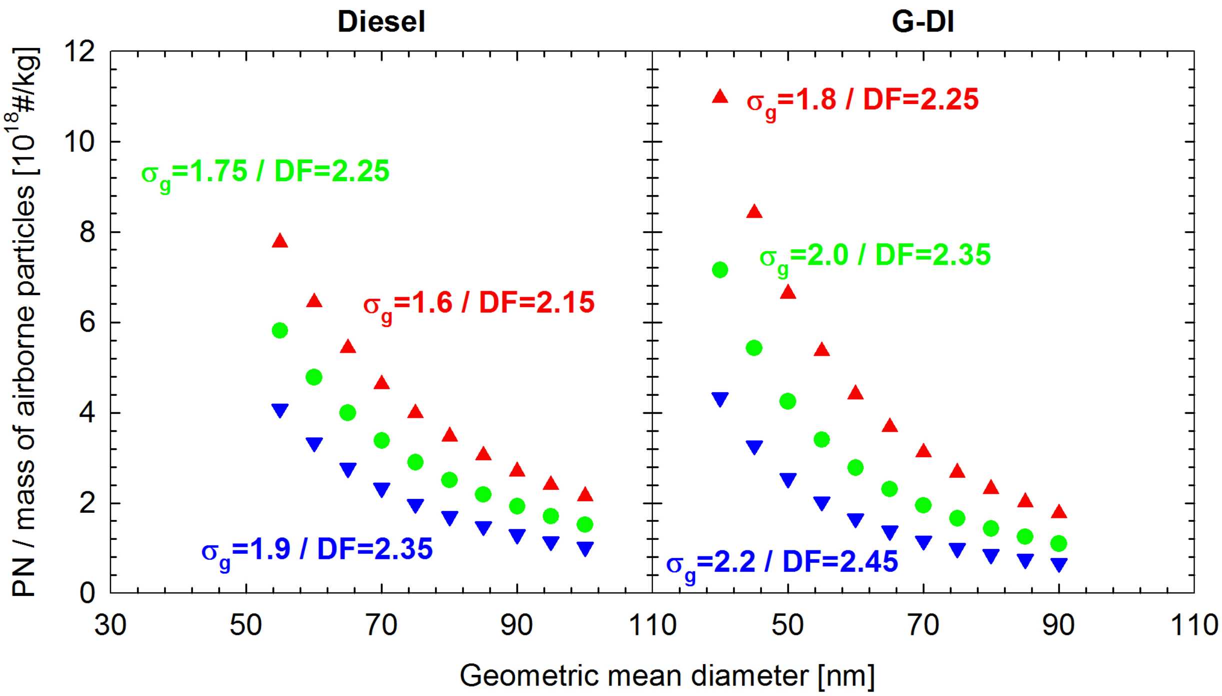

Figure 14: Ratio of particle number to airborne particle mass for diesel (left-hand panel) and G-DI (right hand panel) vehicle exhaust aerosol, plotted as a function of the size distribution properties (geometric mean diameter and geometric standard deviation) and the fractal dimension (DF).

It is generally accepted that the health effects of diesel exhaust particles are associated with the biological response of the human body to a deposited particle in the respiratory tract [140, 146]. In that respect, to assess the relative health risk of different size distributions, it is also necessary to take into account the size-dependent deposition of particles into the respiratory tract [147]. In the size range where automotive exhaust aerosol lies, the alveolar deposition shows a peak in the 10 to 20 nm size range (depending on the activity level) and therefore smaller particles are expected to penetrate deeper into the lungs. As an example, Figure 15 shows size dependent alveolar deposition rates for two extreme cases, namely, heavy exercise and sleeping, based on numerical computations performed by International Commission on Radiological Protection [148].

Convolution of the number weighted size distributions typical for diesel and G-DI exhaust aerosol with the Fuchs surface area and the alveolar deposition rates, allows for the calculation of the PM mass required for different distributions to result in the same accessible deposited surface area of particles in the alveoli. The results of these calculations are summarized in Figure 16. Not surprisingly, the results suggest that the larger the mean size of the distribution, the larger the amount of mass required to accumulate the same surface area in the lungs. The results are also broadly similar for G-DI and diesel vehicles for a given mean diameter. For example, the incremental mass required for a distribution peaking at 90 nm to accumulate the same particle active surface area in the lungs with that of a distribution peaking at 55 nm is 112-146% for diesels and 123-141% for G-DIs.

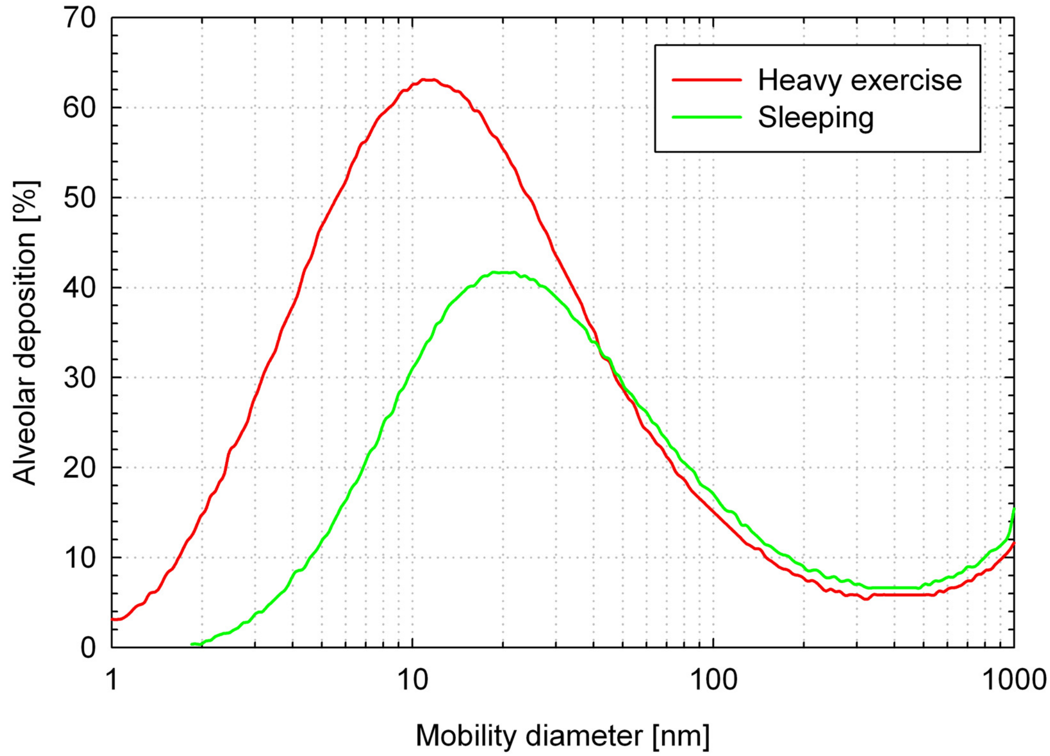

Figure 15: Percentage fraction of different sized particles depositing in the alveoli [148].

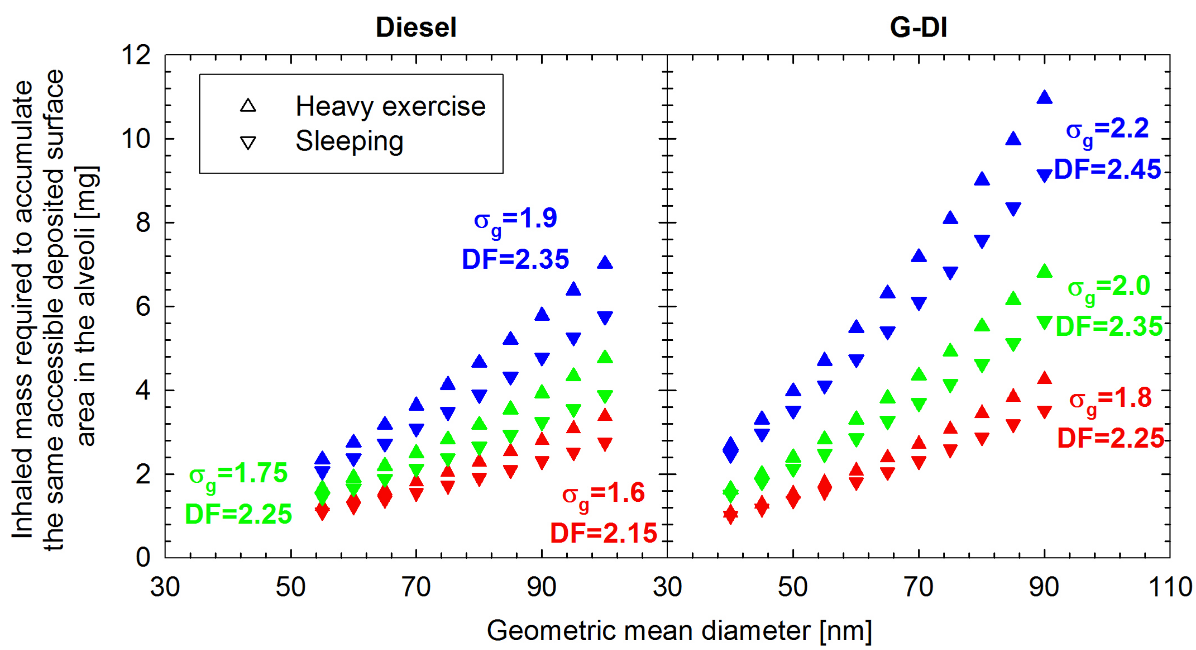

Figure 16: Required mass of inhaled particles to deposit the same (arbitrarily selected) particle surface area in the alveoli for typical size distributions of diesel (left-hand panel) and G-DI (right-hand panel) exhaust aerosol.

Combination of the number to airborne mass ratios (Figure 14) with the mass per deposited surface area in the lungs ratios (Figure 16) allows for a calculation of the number concentration required to deposit the same particle surface area in the lungs for the different distributions examined. The calculated ratio of inhaled number concentrations to deposited surface area in the lungs is plotted in Figure 17. The results suggest practically no dependence on the width of the distribution (σ_g) and as such are broadly similar for G-DI and diesel exhaust. It should be noted that the correlation between number and surface does not depend on the density of the particles, since deposition in the alveoli at this size range is dominated by diffusion. The ratio of inhaled number to deposited surface area also shows a relatively low dependence on particle size, varying in the case of diesel exhaust between 5.2 and 6.9×10^{14} #/m^2 for heavy exercise and between 4.2 and 6.1×10^{14} #/m^2 when sleeping. The corresponding figures for G-DI exhaust are 5.3 to 8.5×10^{14} #/m^2 and 4.4 to 7.9×10^{14} #/m^2, respectively, the difference stemming from the wider size range covered by this vehicle technology.

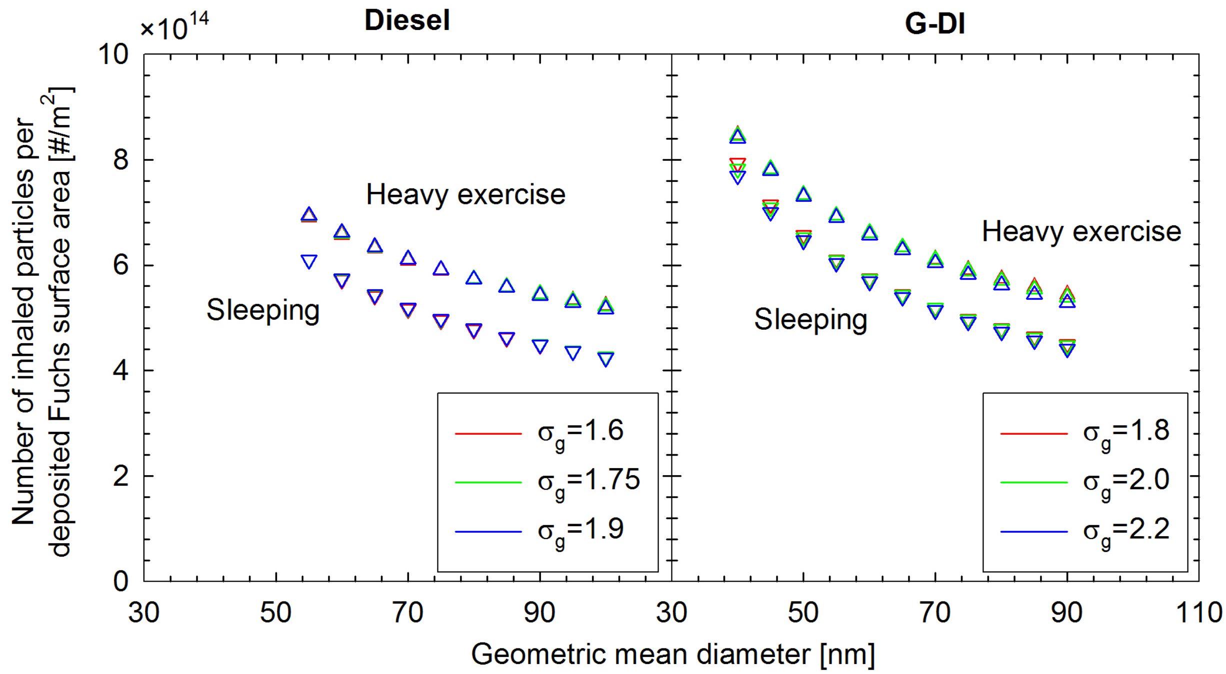

Figure 17: Number of inhaled particles per surface area of deposited particles for diesel (left-hand panel) and G-DI (right-hand panel) exhaust.

Overall, the low size dependence of the number of inhaled particles per deposited particle surface area in the lungs (Figure 17), suggests that a marginal external cost expressed in € per particle number reduction would better capture the health effects of exhaust non-volatile particles, given the large variability in the size distributions of exhaust aerosols. Yet, the derivation of an absolute value for such a marginal external cost figure can only be based on established figures in total mass units. This unavoidably introduces a large uncertainty due to the strong dependence of PM on particle size (Figure 14).

The situation becomes more complicated, considering the aerosol dynamics in the atmosphere which can reduce the number concentrations through coagulation. Since the coagulation rate depends on the initial concentration, the size of aged G-DI aggregates may be lower from those emitted by conventional diesels for the same suspension time. While aged aerosols might be more relevant for exposure and health effect studies, the quantification of the size evolution is a rather challenging task and out of the scope of the present work.

The aforementioned approach, only considers the solid fraction of the emitted particles, which is also what is mostly controlled by a particulate filter. Little is known on the chemistry of the particles emitted from G-DI vehicles. However, a survey of published data on the emission performance of G-DI vehicles [15], suggested that like in diesels, most of the emitted PM is elemental carbon (70-90%). Given the large uncertainty associated with the translation of marginal external costs from PM to PN, any correction for the particle-bound volatiles is deemed to be insignificant.

The above analysis illustrates the difficulties associated with the quantification of health effects of particles on a number basis. In lack of other information, a figure of $2.2 \pm 1.5 \times 10^{18}$ #/kg was employed in the present study. The center value of 2.2×10^{18} #/kg is in agreement with all recent experimental data (both published information [47, 121, 123] and JRC data -) while the uncertainty range ($0.7\text{-}3.5 \times 10^{18}$ #/kg) brackets most of the individual points of experimental data from the Euro 5 vehicles tested at JRC ($0.6\text{-}5.4 \times 10^{18}$ #/kg - Figure 13). Interestingly, this range is in good agreement with the experimental data collected in the PARTICULATES project [130], which was employed for the derivation of the emission factors in the COPERT model. In particular, the individual measurements of five in total Euro 3 technology G-DI vehicles, suggested a solid PN over PM ratio of $2.6 \pm 1.9 \times 10^{18}$ #/kg.

7.3 Calculation results

The starting point of the calculations was to derive the national average mass-based marginal external costs for each EU-27 Member State based on the figures proposed in the IMPACT study (Table 17). The marginal external costs at the three main areas considered in the IMPACT study, i.e. metropolitan, urban and outside built-up areas (considered to consist of rural and motorway sites), were weighted according to the total mileage driven by Euro 6 technology G-DI vehicles (Table 3) and the corresponding PN emission factors (Table 5) for each driving condition. The results of these calculations are summarized in Table 18 and Figure 18. The national average marginal external cost was found to range from as low as 23 to 25 €/kg_{PM} for Bulgaria and Romania, to as high as 144 to 190 €/kg_{PM} for Belgium, Netherlands, UK and Ireland.

The EU27-average marginal external cost was then calculated by means of weighting the national average figures by the projected population of G-DI vehicles. This resulted in a total-fleet, EU27 average marginal external cost figure of 115 $€_{2011}/kg_{PM}$, which furthermore was found to show little dependence on the different G-DI share projections. The marginal external cost differed somehow between vehicle classes due to the different driving behaviour (i.e. share of metropolitan, urban, rural and motorway) and the different market shares of the different vehicle classes in each member state. Accordingly, the EU27-average marginal external costs for SPC, MLPC and LDVs were calculated to be 109 $€_{2011}/kg_{PM}$, 121 $€_{2011}/kg_{PM}$ and 92 $€_{2011}/kg_{PM}$, respectively. Using the range of ratios of PM to PN emissions established in the previous section (7.2), these figure translate to number-based marginal external costs of 29 to 156 $€_{2011}/10^{18}$# for SPC, 33 to 173 $€_{2011}/10^{18}$# for MLPCs and 25 to 131 $€_{2011}/10^{18}$# for LDVs.

Table 18: National average marginal external cost of road transport PM for each EU27 member state.

Country	Marginal External Cost ($€_{2011}/kg_{PM,road\ transport}$)
Austria	122.8
Belgium	149.4
Bulgaria	22.6
Cyprus	44.9
Czech Republic	105.9
Denmark	140.0
Estonia	70.4
Finland	114.0
France	130.6
Germany	116.9
Greece	102.7
Hungary	110.1
Ireland	190.5
Italy	116.5
Latvia	63.4
Lithuania	68.8
Luxembourg	240.3
Malta	48.8
Netherlands	157.9
Poland	88.5
Portugal	109.1
Romania	25.4
Slovakia	92.6
Slovenia	120.8
Spain	106.7
Sweden	101.3
United Kingdom	143.5

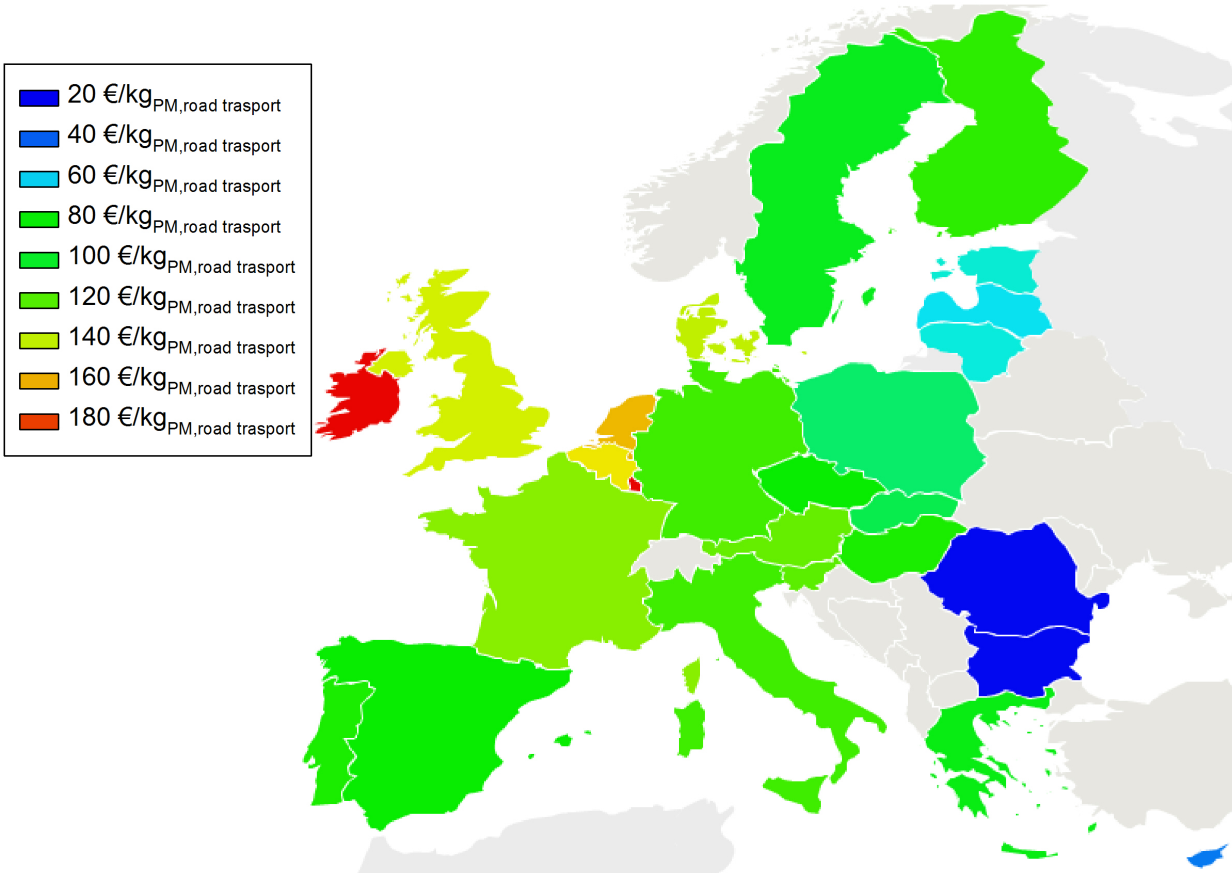

Figure 18: National average marginal external cost of road transport PM for each EU27 member state.

The benefit in monetary terms resulting from the installation of a GPF can then be derived by calculating the total number of solid particles saved over the useful life of the vehicle. The calculations are summarized in Table 19. The estimated reduction of solid particle number emissions is calculated to be around 10^{18} #/vehicle life. Accordingly, the installation of a GPF is expected to result in a societal benefit in the range of 31 to 162 € per SPC vehicle, 36 to 191 € per MLPC vehicle and 25 to 130 € per LDV.

Table 19: Calculated reduction in the external cost of emitted PN through the use of GPF.

	SPC			MLPC			LDV		
	Urban	Rural	Motor.	Urban	Rural	Motor.	Urban	Rural	Motor.
Total useful life [km]	130000			135000			150000		
Share [%]	25.0%	60.5%	14.5%	29.2%	55.3%	15.5%	7.5%	59.3%	33.3%
Useful life [km]	32500	78650	18850	39420	74655	20925	11250	88950	49950
Emission factors - Non-GPF									
PN [10^{12}#/km]	13.0	8.3	4.3	13.0	8.3	4.3	13.0	8.3	4.3
Emission factors - GPF									
PN [10^{12}#/km]	1.3	0.83	0.43	1.3	0.83	0.43	1.3	0.83	0.43
Emission reduction over the useful life									
PN [10^{18}#]	1.04			1.10			0.99		
Marginal external cost									
Min estimate [€]	29.5			32.8			24.9		
Max estimate [€]	155.9			173.2			131.5		
Externalities reduction									
Min estimate [€]	**30.7**			**36.0**			**24.6**		
Max estimate [€]	**162.2**			**190.5**			**130.1**		

8 GLOBAL WARMING EFFECT

A potential drawback of installing a GPF in G-DI vehicles is related to a possible increase of the emitted CO_2 emissions and the associated contribution to global warming. GPF systems however, have also the benefit of effectively reducing the emitted black carbon which is recognized to be a much stronger climate forcer than CO_2. Latest estimates on the global warming potential of 1 g of BC over a period of 100 years range between 100 to 2000 times that of 1 g of CO_2 [81]. Furthermore, BC has a significantly shorter lifetime compared to CO_2 and therefore proven emission control strategies like particulate filters were recognized to have immediate benefits in achieving near-term targets [81].

Economic valuation of the climate change effect is a rather challenging task, due to the long term effects in a global scale and the general lack of knowledge about the physical impacts caused by global warming [112]. Two main methodologies are employed. The first one follows the impact pathway approach and uses detailed modelling to assess the physical impact of climate changes which are then combined with estimations of the societal cost in monetary terms resulting from these physical impacts [149]. Other studies propose the use of an avoidance cost instead, that is the least-cost option to achieve a required level of greenhouse gas emission reduction [113]. It is generally considered that the avoidance cost approach is better suited for short-term effects while the impact pathway methodology is conceptually more relevant for long-term effects. The IMPACT study [112] reviewed all relevant studies and proposed different external cost figures depending on the time horizon considered. The external cost associated with a 1 tonne CO_2 equivalent ranges from 17 to 70 € in 2020, to 22-100 € in 2030 and 20-180 € in 2050.

Due to the different lifetimes of the two pollutants (30-95 years for CO_2 [150] compared to several weeks for BC), the global warming penalty resulting from increased CO_2 emissions will be persistent long after the service life of the vehicle and will furthermore have a global impact. On the contrary, BC emissions will have an effect an impact on climate only over the useful life of the vehicle (order of 10 years). Accordingly, short term external costs (2020 timeframe) were employed in the study for the emitted BC while CO_2 externalities were calculated assuming a long-term effect (2050 timeframe).

The results of the calculations are summarized in Table 20. The total increase in the amount of CO_2 emitted over the service life of the vehicle was estimated to range 26 to 54 kg for SPC, 39 to 76 kg for MLPCs and 68 to 183 kg for LDVs. The CO_2 equivalent reduction resulting from the control of BC was calculated to be 29-573 kg for SPCs, 31-621 kg for MLPCs and 34-667 kg for LDVs. The associated externalities resulting from CO_2 increase were calculated to be 0.5 to 9.8 € for SPCs, 0.8 to 13.7 € for MLPCs and 1.4 to 32.9 € for LDVs. The corresponding figures for the BC benefit were 0.5 to 40.5 € for SPCs, 0.5-43.8 € for MLPCs and 0.6 to 47.4 € for LDVs. The global warming benefit from BC is therefore estimated to be of the same level with the global warming penalty from the CO_2 increase.

Table 20: Estimated global warming effect and associated externalities resulting from the BC reduction and CO2 increases through the installation of GPF.

	SPC			MLPC			LDV		
	Urban	Rural	Motor.	Urban	Rural	Motor.	Urban	Rural	Motor.
Total useful life [km]	130000			135000			150000		
Share [%]	25.0%	60.5%	14.5%	29.2%	55.3%	15.5%	7.5%	59.3%	33.3%
Useful life [km]	32500	78650	18850	39420	74655	20925	11250	88950	49950
CO_2 emission factors – No GPF									
CO_2 [g/km]	244.8	137.2	149.5	322.3	184.2	182.0	460.6	228.7	232.4
CO_2 emission factors – No GPF									
Min CO_2 [g/km]	245.2	137.2	150.2	322.8	184.2	182.9	461.3	228.7	233.6
Max CO_2 [g/km]	245.2	137.2	151.7	322.8	184.2	184.7	461.3	228.7	235.9
Increased CO_2 emissions over the useful life									
Min CO_2 [tn]	0.026			0.039			0.068		
Max CO_2 [tn]	0.054			0.076			0.183		
CO_2 externalities - Long term (2050: 20-180 €/tnCO_2)									
Min [€]	**0.5**			**0.8**			**1.4**		
Max [€]	**9.8**			**13.7**			**32.9**		
BC emission factors - No-GPF									
BC [mg/km]	3.7	1.7	3.7	3.7	1.7	3.7	3.7	1.7	3.7
BC emission factors - GPF									
BC [mg/km]	0.37	0.17	0.37	0.37	0.17	0.37	0.37	0.17	0.37
Decreased BC emissions over the useful life									
BC [kg]	0.29			0.31			0.34		
CO_2 equivalent reduction from BC									
Min [tn] (100:1)	0.029			0.031			0.034		
Max [tn] (2000:1)	0.573			0.626			0.667		
BC externalities - Short term (2020: 17-70 €/tnCO_2)									
Min [€]	**0.5**			**0.5**			**0.6**		
Max [€]	**40.5**			**43.8**			**47.4**		

9 CONCLUSIONS

The study assessed the feasibility of introducing a GPF system in G-DI vehicles as well as the associated cost and environmental benefit. The key findings can be summarized as follows:

- The penetration of G-DI vehicles in the market is expected to rapidly grow in the following decade, eventually replacing their PFI counterparts. This is due to their improved fuel efficiency and the potential they offer for efficient downsizing and turbocharging that would significantly improve fuel consumption.

- The particle number emissions of commercial G-DI vehicles over the NEDC are consistently above the diesel PN limit of 6×10^{11} #/km, with some vehicles exceeding it by 1 ½ orders of magnitude.

- The installation of Gasoline Particulate Filters to G-DI vehicles seems rather straightforward due to a) the lower soot emissions b) the much more frequent passive regenerations requiring little engine interference and c) the more than a decade experience collected from diesel applications. Accordingly, the associated fuel consumption penalty introduced by the installation of a GPF is expected to be low.

- While the total annually emitted solid PN emissions from diesel and PFI vehicles are projected to decrease over time, the contribution of G-DI vehicles to ambient PN will gradually increase. If no measure is taken for the control of their particle number emissions it is expected that G-DIs will become the major contributor of PN by 2030. The situation will become even worse in subsequent years as the remaining non-DPF equipped vehicles will be eventually scrapped.

- The use of a GPF has the potential to drastically reduce emissions, resulting in an almost one order of magnitude reduction (79% over the basecase) by 2030. A three years delay in the implementation of a GPF has a relatively small long-term effect in ambient PN levels (72% reduction over basecase by 2030), due to the currently limited penetration of G-DIs in the market.

- The price increase associated with the introduction of a GPF at a Euro 6 stage was estimated to be in the range of 39 to 163 € for small passenger cars, 56 to 264 € for medium to large passenger cars and 70 to 303 € for light duty vehicles, depending on the level of complexity. The lowest cost estimates correspond to lower GPF volumes, constructed by less expensive substrate materials, incorporating catalytic activity thus reducing (or even replacing) the volume of the TWC, and requiring no monitoring of the GPF status. Highest cost estimates correspond to uncoated GPFs of larger size that would require active regeneration and therefore monitoring of the temperature and the pressure drop across the GPF, and especially in the case of MLPCs and LDVs to systems incorporating two exhaust lines. Published data on implementation costs by AECC range between 40 and 130 €, while MECA suggest an implementation cost of 35 to 70 €, respectively, depending on engine size, production volume and packaging. These low figures reflect to a certain extent the confidence of GPF manufacturers that much lower GPF volumes will be required with no need for monitoring of the GPF status but also verify the availability of less expensive substrate materials. In line with this, a recent study by the International Council on Clean Transportation derived a GPF implementation cost (including pressure monitoring) of 62-85 € for SPCs, 75-108 € for MLPCs and 87-131 € for LDVs.

- A three years delay in the implementation of a diesel PN limit for G-DI applications will not reduce significantly the GPF installation cost (2 to 13% reduction). It is expected however to allow the vehicle manufacturers sufficient time to comply through less expensive internal engine measures, at least for some of their models. Some recent studies suggested that there exists a significant potential for particle emission reduction through optimized injection strategies and improved injector designs. Dual injection systems allowing both port- and direct-fuel injection, were also introduced in some expensive models which according to the manufacturers already comply with the diesel PN limit. Confidential information presented by several vehicle manufacturers in a special workshop organized by DG-ENTR, verified that all manufacturers are working in this direction and some of them were confident that will be in a position to comply with a diesel PN limit at a 2017-2018 time frame. Some of the challenges faced include the strong effect of fuel quality and deposit formation on the particle emissions, raising concerns about the real world emission performance.

- The installation of a GPF is estimated to result in a maximum 9-18 kg increase in the total fuel consumed over the useful life of SPCs. The corresponding figures for MLPC and LDVs were estimated to be 13-25 kg and 21-59 kg, respectively. Based on the current industrial price of petrol (tax-free 0.55 €/lt), the additional cost introduced to the vehicle owner is calculated to be 6-13 €/useful life for SPCs, 9-18 €/useful life for MLPCs and 16-43 €/useful life for LDVs.

- External costs for particulate emissions are only available for PM and not for solid particle number which is the focus of the present study. A calculation of external cost for PN from the established PM figures is not straightforward. Such a translation strongly depends on the underlying size distribution which is known vary considerably from vehicle to vehicle and also depend on the operating conditions. Numerical calculations suggested that inhaled number concentrations correlate well with the active surface area of the particles deposited in the lungs. Accordingly it was assumed that particles of different sizes are equally toxic, even if smaller particles have less mass.

- Latest estimates on the marginal external cost of road transport PM suggest a EU27 average figure of 109 €/kg_{PM} for SPCs, 121 €/kg_{PM} for MLPCs and 92 €/kg_{PM} for LDVs (the differences reflecting the frequency of use in densely populated areas were health effects are more important). Experimental data from late technology G-DI vehicles suggest a solid PN to PM ratio at around 2.2×10^{18} #/kg_{PM}, but individual data fall within the 0.6 to 4.5×10^{18} #/kg_{PM} range, to a large extent due to difference in the size distributions (as verified by numerical calculations). Using a $2.2\pm1.5\times10^{18}$ #/kg_{PM} figure suggested a PN health benefit from GPF installation of 31-162 € for SPCs, 36-191 € for MLPCs and 25-130 € for LDVs, over the useful life of the vehicles.

- The global warming penalty resulting from increased CO_2 emissions is expected to be counterbalanced from the reductions in black carbon. The external cost resulting from the elevated CO_2 emissions is estimated to lie in the range of 0.5-10 € for SPCs, 0.8-14 € for MLPCs and 1.4-33 € for LDVs, over the service life of the vehicles. The associated benefit resulting from the total BC reduced over the useful life of the vehicles is estimated to be 0.5-40.5 € for SPCs, 0.5-43.8 € for MLPCs and 0.6-47.4 € for LDVs.

- Overall, the results of the study suggest that the societal benefit offered from the installation of GPF is at least of the same order of magnitude with the associated implementation cost.

10 LIST OF SPECIAL TERMS AND ABBREVIATIONS

ACEA	European Automobile Manufacturer's Association
AECC	Association for Emissions Control by Catalyst
BC	Black Carbon
CAFE	Clean Air For Europe
CO_2	Carbon Dioxide
CPC	Condensation Particle Counter
DG ENTR	Directorate General for Enterprise
DPF	Diesel Particulate Filter
ECS	Emissions Control System
EGR	Exhaust Gas Recirculation
EPA	Environmental Protection Agency
EC	Elemental Carbon
ECE	Economic Commission for Europe
EU	European Union
FTP	Federal Test Procedure
G-DI	Gasoline Direct Injection
GRPE	Working Party on Pollution and Energy
GPF	Gasoline Particulate Filters
HD	Heavy Duty
IC	Indirect Cost
ILCE	Inter-Laboratory Correlation Exercise
JRC	Joint Research Centre
LD	Light Duty
LDV	Light Duty Vehicle
MLPC	Medium to Large Passenger Car
NEDC	New European Driving Cycle
NO	Nitrogen Monoxide
NO_2	Nitrogen Dioxide
NRC	National Research Council
PC	Passenger Car
PM	Particulate Matter
PMP	Particle Measurement Programme
PMG	Platinum Group Metals
PN	Solid Particle Number
PFI	Port Fuel Injection
RPE	Retail Price Equivalent

SPC	Small Passenger Car
TWC	Three Way Catalyst
UN-ECE	United Nations Economic Commission for Europe
UK	United Kingdom
USA	United States of America

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Abstract

Starting from September 2011 a limit of 6×1011 #/km will be introduced for the type approval of diesel passenger cars that will eventually apply to all new registered diesel passenger cars from September 2012. The same limit will also apply to diesel light duty vehicles but with a one year delay (09/2014 for type approvals and 09/2015 for all new registered vehicles). The regulation states that a Particle Number (PN) limit will also be introduced for the certification of Euro 6 technology gasoline-fuelled vehicles but the threshold value was not decided yet. While conventional Port Fuel Injection (PFI) gasoline vehicles can easily comply with the diesel limit, their Direct Injection (G-DI) counterparts are found to emit systematically above this threshold by up to 1 ½ orders of magnitude. It is therefore expected that application of the diesel particle number limit to G-DI vehicles may necessitate the installation of a particulate filter.
At the same time, the penetration of G-DI vehicles is expected to rapidly grow in the near future in both the European and USA markets. This is due to their improved fuel efficiency compared to the conventional PFIs, that would potentially enable the target set in both EU and USA on the fleet-average carbon dioxide (CO_2) emissions of future vehicles. It is foreseen that this vehicle category will dominate the gasoline market eventually replacing the conventional and less efficient PFI vehicles. There are concerns however, that their elevated particulate emissions may adversely affect the air quality in the future if no measure is taken to efficiently control them.
In this direction the present study examined the feasibility of introducing Gasoline Particulate Filters in G-DI vehicles and investigated the associated implementation cost and environmental benefit.

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