Exhaust Emissions from Road Transport

AIR QUALITY EXPERT GROUP

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Prepared for:

Department for Environment, Food and Rural Affairs;
Scottish Government; Welsh Government;
and Department of Agriculture, Environment and Rural
Affairs in Northern Ireland

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This is a report from the Air Quality Expert Group to the Department for Environment, Food and Rural Affairs; Scottish Government; Welsh Government; and Department of Agriculture, Environment and Rural Affairs in Northern Ireland, on exhaust emissions from road transport. The information contained within this report represents a review of the understanding and evidence available at the time of writing.

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National Centre for Atmospheric Science, University of York and Senior Research Fellow, Department for Environment, Food and Rural Affairs Systems Research Programme

Central Management and Control Unit of the automatic urban and rural networks: **Dr Richard Maggs**, Bureau Veritas

National Atmospheric Emissions Inventory: **Dr Tim Murrells**, Ricardo Energy and Environment

Non-automatic hydrocarbon monitoring networks and metals monitoring network: **Dr Paul Quincey**, National Physical Laboratory (till December 2020), **Dr Nick Martin** (from January 2021).

Quality Assurance and Quality Control of the automatic urban network and the non-automatic monitoring networks: **Dr Paul Willis**, Ricardo Energy and Environment

Assessors and observers

Roger Herbert

Welsh Government

Kate Fitzsimmons
Department of the Environment in Northern Ireland

Andrew Taylor
Scottish Government

Alison Gowers
Public Health England

Secretariat

Shaun Brace
Department for Environment, Food and Rural Affairs
Dr Mohamed Ghalaieny
Department for Environment, Food and Rural Affairs
Michelle Brailey-Balster
Department for Environment, Food and Rural Affairs

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Executive summary

Air pollutants in the exhaust emissions from internal combustion engine (ICE) vehicles result from unburnt fuel, high temperature combustion of fuel in the presence of air, and from exhaust aftertreatment processes. As many of these pollutants are harmful to human health, impact climate as greenhouse gases, or both, emissions from vehicles have been subject to regulation for many years. Europe-wide standards on vehicle exhaust were introduced in the early 1990s and have been progressively made more stringent, extending the range of regulated pollutants included and reducing the permissible levels of emissions in new vehicles introduced to the fleet. A review of the development and evolution of exhaust emissions standards of relevance to UK vehicles is provided in Chapter 2 of this report.

To comply with these standards, technologies for reducing emissions of regulated pollutants have been developed and fitted to vehicle exhaust systems. In combination with advances in engines, these have substantially reduced emissions of many of the regulated pollutants, and a review of the key technologies has been undertaken. Whilst there has been considerable success in substantially reducing exhaust emissions of pollutants such as carbon monoxide (CO), hydrocarbons (HCs) and particulate matter (PM), exhaust emissions from ICE vehicles remain a major source of nitrogen oxides (NO_x) and an important contributor to poor urban air quality.

Ambient atmospheric measurements have previously revealed important disparities between urban and roadside concentrations of nitrogen dioxide (NO_2) and those that would be projected based on the prevailing vehicle emissions standards of NO_x. It is now understood that exhaust gas aftertreatment technologies introduced on diesel vehicles have not always been effectively implemented and not delivered the anticipated reductions in emissions when in real-world use. The on-road effectiveness of exhaust emissions controls has been found to be highly variable even between vehicles that are technically compliant with the same regulatory standards. There is now greater appreciation of the need to evaluate and demonstrate vehicle emissions performance under real-world driving conditions. On-road emissions have been higher than those measured during laboratory-based tests, although in the past five years that gap has closed for some vehicles, and improvements continue. Future European standards will move close to demanding equivalence between on-road and laboratory performance. Real-world driving emissions are reviewed for various vehicle types in Chapter 3.

The dynamic nature of the vehicle fleet means that vehicles with a range of different exhaust control technologies and historic standards are in use on UK roads. Between 1990 and 2015, the UK vehicle fleet transitioned to a greater proportion of diesel cars, compared with gasoline, a change that was motivated by predicted fuel economy savings and lower CO_2 emissions per mile driven. The relative contributions of exhaust emissions from diesel, gasoline, and more recently hybrid electric vehicles continue to evolve as new technologies develop, and the regulatory framework and consumer preferences (in part driven by economic policy such as fuel tax) change. The vehicle fleet will continue to change over the next decade with older more polluting vehicles leaving the fleet and newer cleaner vehicles joining. This report provides, in Chapter 4, modelled estimates of the vehicle fleet composition over time and the resulting exhaust emissions using best available estimates from inventories used for international reporting. Understanding how the future ensemble of vehicles on the roads will contribute to air pollution emissions is significant for policy development since this information can support the evaluation of interventions such as low emissions zones, and the likely compliance with air quality directives. This report was drafted prior to the COVID-19 crisis and projections of future vehicle use are based on knowledge and understanding at that time. Since March 2020 traffic volumes have varied significantly depending on national restrictions and this has influenced roadside concentrations of many vehicle-derived pollutants. AQEG provided an initial analysis of these effects to Defra in June 2020. The post-COVID fall in new vehicle sales and economic conditions which may affect transport in the future are not assessed here.

Ambient observations of air pollution, particularly at the roadside, provide an independent assessment of vehicle exhaust emissions, trends in these emissions over time, and by extension whether emissions controls are working as intended. Methods to assess vehicle exhaust emissions using UK ambient air quality data for a range of different pollutants are presented in Chapter 5, which also includes examples of where ambient observations have detected unanticipated trends in emissions not predicted based on laboratory tests.

Chapter 6 identifies possible future interventions and technologies to reduce exhaust emissions and their impacts, including aftertreatment approaches. The longer-term impacts on vehicle exhaust emissions from increased use of ICE-electric hybridisation propulsion and driver automation technologies are considered, as are potential approaches to retrofitting as a means to improve emissions performance for existing vehicles. Potential regulatory approaches to managing exhaust emissions are also reviewed, including the possible impacts of changed vehicle inspection regimes, new fuels standards and external traffic management controls on vehicle movements and on idling.

Chapter 7 provides the AQEG recommendations.

The key points made in the report are summarised below in a series of answers to questions.

QUESTION AND ANSWER

1. How are exhaust emissions from vehicles regulated and for which pollutants?

New vehicles registered in the UK must currently meet European type-approval emissions standards, referred to as "Euro" standards. The first Euro standards came into force in 1992 with the regulations also defining the test facilities, tests (drive cycles) to be conducted, the measurement methods to be employed and the limits to be complied with. Euro standards have become progressively more stringent and comprehensive over time and currently extend from ‘Euro 1’ to ‘Euro 6’ for light duty vehicles (cars and vans) and Euro I to Euro VI for heavy duty vehicles (HDVs). Euro VI has been mandatory for all new HDVs since December 2013. The Euro 6 standards are being delivered in stages, with implementation dates from 2014 until 2021. Motorcycles have recently transitioned from Euro 4 to Euro 5, with full implementation of Euro 5 in December 2020.

The Euro standards currently limit the tailpipe emissions of: carbon monoxide (CO), oxides of nitrogen (NO_X), total hydrocarbons (THC), particulate matter mass (PM), the number of solid particles larger than 23 nm (PN), and, for heavy duty vehicles, ammonia (NH_3). There are also EU regulations for reductions in fleet average CO_2 emissions for new passenger cars and light commercial vehicles that came into effect from 2015. Euro standards impose exhaust limit values for pollutants that make the application of emissions control technologies unavoidable for vehicle and engine manufacturers, however the Euro regulatory system does not directly mandate the use of particular control technologies or specify how emissions should be reduced. The Euro standards apply to new vehicles. Older vehicles are subject to emissions testing as part of Periodic Technical Inspections (PTIs) (i.e. MOTs in the UK). Petrol and diesel fuel quality has been regulated since the 1990s by Fuel Quality Directives.

2. Are there pollutants which are not currently regulated by the Euro standards, but which are known to be emitted from vehicle exhaust?

Direct emission of nitrogen dioxide (NO_2) is currently regulated via the limits placed on total NO_X emissions (the sum of NO + NO_2) but there is no specific regulation on NO_2 in isolation. Ammonia (NH_3) can be released from vehicle exhaust as a by-product of some of the technologies used to reduce NO_X emissions, but emission of NH_3 is only regulated for HDVs. Other unregulated tailpipe exhaust gases include: nitrous oxide (N_2O), arising from partially reduced NO_x and partially oxidized NH_3; aldehydes and ketones, which result from partial oxidation of hydrocarbons (HC); and isocyanic acid (HNCO), which can be found in exhaust gases from vehicles using selective catalyst reduction (SCR). Solid particles smaller than 23 nm and semi-volatile liquid particles generally smaller than 23 nm are not regulated although they are known to be present in exhaust from both compression (diesel) and spark-ignition (petrol) engines.

3. What are the main engineering approaches used to reduce vehicle exhaust emissions?

The three-way catalyst (TWC) is the main technology used for aftertreatment of engine exhaust in gasoline vehicles. This removes a substantial fraction of the gaseous emissions of CO, HC, and NO_x by converting them to CO_2, water and nitrogen. Gasoline particulate filters (GPF) are used on some gasoline vehicles to control PM and PN.

For diesel engine exhaust, particulate matter is removed using diesel particulate filters (DPF) and CO and HC are controlled though use of diesel oxidation catalysts (DOC). Controlling NO_x is complex because emissions are highly sensitive to combustion conditions in the engine and the condition of the exhaust and aftertreatment system, particularly with regard to temperature and fuel/air ratio. There is generally a narrow window of conditions in which exhaust aftertreatment systems become effective. NO_X is removed from diesel exhaust using direct engine and catalytic approaches. Exhaust gas recirculation (EGR) is used to route engine exhaust gases back into the combustion chamber with the effect of reducing combustion temperature and the formation of NO_X. Exhaust emissions of NO_x are subject to further control using either a lean NO_X trap (LNT) or selective catalyst reduction (SCR). NO_x trapped by an LNT is periodically removed and reduced to predominantly N_2, although NH_3 can be formed. SCR systems inject urea into the exhaust to generate NH_3 which then reduces the NO_X. Unreacted NH_3 can exit the tailpipe, although it can be controlled by an ammonia slip catalyst (ASC) which oxidises it to N_2.

4. How have the introduction of more stringent Euro classes affected exhaust emissions?

The early stages of both light-duty and heavy-duty vehicle legislation in the 1990s principally targeted HC and CO emissions, with limits eventually set to levels that required TWC on gasoline passenger cars, and DOC on diesels. The progressive lowering of emissions limits on diesel NO_X and PM from Euro I to Euro IV, and Euro 3 to 4, led to advances in both fuel injection equipment and exhaust gas recirculation. Euro VI/5 introduced PN limits that could only be attained using DPF, and this technology was universally adopted. DPFs resulted in significant reductions in real-world exhaust emissions of PM. Particle filters are now fitted to gasoline vehicles as well, a result of PN limits imposed by Euro 6. NO_x emission limits for Euro VI led to the widespread introduction of SCR on heavy-duty diesels and had a positive impact on real-world emissions.

New lower limits, and the introduction of a new, more realistic test cycle for light duty vehicles at Euro 6b led to the introduction of “DeNO_x” approaches including advanced EGR, LNT and SCR. The reductions in emissions achieved during test cycles in the laboratory for Euro 6b were not always replicated in real-world use, however. Consequently, three further stages of Euro 6 (6c – from 2018, 6d-temp - 2019, 6d – January 2021) have been added. These require Real Driving Emissions (RDE) measurements from both gasoline and diesel vehicles, using portable emissions measurement systems (PEMS), to comply with not to exceed (NTE) levels that are directly linked to the limits that must be passed in the laboratory drive cycle tests. NO_x control on Euro 6d-temp vehicles, with aftertreatment systems based primarily on SCR, or SCR and NO_x storage approaches, is now proving to be effective across the whole range of RDE conditions.

There is large variability of emissions within classes. Some Euro 6 diesel models emit less NO_x than some Euro 6 gasoline vehicles, but many others emit more NO_x than gasoline cars. There can be large differences in emissions from vehicles within the same class from different manufacturers. This has implications for interventions based only on vehicle Euro standards.

Emissions and limit values of NO_x from motorcycles have been invariant from pre-Euro through to Euro 4 and the emissions are relatively high on a fuel-specific basis.

5. Have ambient concentrations of exhaust pollutants changed in response to the tightening of Euro standards?

Ambient measurements have revealed important disparities between the changes in roadside concentrations and those that would have been projected based on vehicle emissions standards. Improved vehicle emissions standards, if effective, should lead to reductions in roadside concentrations of NO_x, NO_2 and $PM_{2.5}$. Between 2005 and 2010 reductions in concentrations at UK roadside locations were variable and often less than were anticipated based on laboratory vehicle emission performance data. Since ~2010 concentrations of NO_x, NO_2 and $PM_{2.5}$ concentrations have, on average, been decreasing at the roadside, although with considerable spatial heterogeneity. Trends in black carbon, a tracer of diesel exhaust particle, have however been sharply downwards. There is some evidence that concentrations of NO_2 in the UK have been falling more rapidly since around 2015, especially at roadside sites. Concentrations of CO and non-methane volatile organic compounds (NMVOCs), of which HCs are a subset, decreased rapidly from around 2000-2005, but the rate of change has now slowed due to the contribution from other, non-traffic sources. Elevated roadside NH_3 concentrations have also been observed, driven by controls introduced to meet the Euro standards for NO_x.

6. What is the significance of recent changes to the regulatory test cycle?

For more than 25 years new models of cars and vans in Europe were tested for their emissions on a chassis dynamometer before being approved for use on the roads. These tests followed a series of accelerations, decelerations and speeds set out in the New European Drive Cycle (NEDC). The divergence of on-road emissions with those estimated from these dynamometer tests led to the introduction of the Euro 6c standard and laboratory testing with more realistic drive cycles (World Harmonised Light-duty vehicles Test Procedure (WLTP)). The Euro 6d-temp and Euro 6d standards then supplement laboratory tests with limits applied to real on-road driving emissions (RDE) tests. The Euro VI standard for HDV engines also includes new, more representative test cycles and a requirement for in-service, on-road testing.

The specified criteria which define regulatory RDE routes create a very large multi-dimensional space within which valid RDE cycles exist. New vehicles are expected to meet the emission requirements under any possible combination of parameters within that RDE space. The new RDE testing procedures create a far more challenging environment for accurate quantification of emissions than the test laboratory. Individual driver characteristics, behaviour of other road users, environmental conditions (e.g. ambient temperature) and lower accuracy measurement systems all contribute to this.

7. How are emissions affected by engine operating conditions?

Gasoline engines use excess fuel to start the engine (cold start), which results in initially high HC, CO and PN emissions as not all the fuel is consumed during the combustion process. For a TWC to operate efficiently the catalyst must be hot, which means that it is less efficient during, and for a short period after, a cold start. In gasoline direct injection (GDI) engines, PN emissions are higher during cold start due to reduced time for fuel atomisation and the associated mixture heterogeneity and increased fuel impingement. Low ambient temperatures impact the performance of aftertreatment technologies (e.g. catalysts) and demand excess fuelling for engine start and a longer warm up, thus increasing emissions. Increased emissions of NO_x at colder ambient temperatures have also been found from hot diesel engines, separate from any cold-start effect.

8. How do emissions change with different driving conditions?

Traffic congestion leads to vehicle stop-starts and aggressive transient operation (acceleration and deceleration). Acceleration is achieved by delivering more fuel, which results in increased gaseous emissions, and fuel impingement increases resulting in more PN from both gasoline and diesel engines. Uphill driving operating at low engine speed and high load increases all gaseous emission from gasoline vehicles and engine-out NO_x emission from diesel vehicles, although SCR efficiency is improved at higher temperatures, which can be associated with uphill gradients. In gasoline engines, exhaust temperatures can exceed the thermal limit when operating at higher engine speeds and high loads. Therefore, to protect exhaust components, including catalyst materials, additional fuel is injected to cool down the exhaust system. This is called fuel enrichment and leads to high engine out CO and THC emissions.

9. Does the evidence allow conclusions to be drawn on optimum road planning, traffic control, or driving styles that can reduce air pollution?

Modelling suggests that journey-average emissions of NO_x and PM can be effectively reduced by limiting the duration, and degree, of stop-start acceleration events. The relationship between of average speed and exhaust emissions is masked by the greater impact of transient accelerations. Defining the precise road and traffic management interventions that can best achieve this type of smoother, more free-flowing, driving is outside the scope of this report, but the general principle is that reducing congestion will reduce acceleration events and pollution emissions. There is some evidence that 20 mph speed limits in urban areas can help to smooth traffic flows. However, road designs involving vertical deflection (e.g. speed bumps) can cause additional deceleration and then acceleration events, leading to increased emissions from both exhausts and brakes. In practice, different traffic management interventions are likely to be appropriate in different locations.

Driving style is also a significant factor influencing emissions. Differences in CO_2 (up to 7%) and NO_x (up to 55%) emissions have been observed from the same RDE test route arising from drivers switching between ‘normal’ to ‘severe’ driving modes but remaining within ‘legal’ driving styles. The same principles around reducing air pollution from vehicles apply to behavioural aspects including driving style. Since acceleration is a major cause of exhaust emissions, smoother driving reduces fuel consumption and lowers emissions. Avoiding aggressive stop-starts, through limiting sharp braking and rapid acceleration reduces emissions. Moreover, smoother driving reduces non-exhaust emission from brakes.

Although not directly linked to style of driving it is worth restating that a significant fraction of exhaust emissions are associated with vehicle cold starts, and that reducing overall frequency of journeys, particularly short urban drives would be effective at improving air quality.

10. Can advice be provided on the air quality impacts of idling vs engine switch off when stationary, and how this varies with vehicle type, outdoor temperature and time spent stationary?

There is considerable interest in the use of vehicle anti-idling as an urban air pollution management intervention. Switching off an engine immediately ends combustion emissions and even short periods of engine-off have benefits in reducing emissions over a journey of CO_2, the primary greenhouse gas. However, during engine-off periods the exhaust system (for example TWC or DOC) on a vehicle may begin to cool and be less effective once the engine restarts. The effectiveness of stop-start is therefore closely tied to the period the engine is switched off and ambient conditions. The balance between eliminating emissions that would have been released during idling and the possibility of increased emissions on restart will depend on the aftertreatment technology and engine management system of each individual vehicle. It is therefore difficult to provide advice on the effectiveness of anti-idling without further research encompassing the very wide range of ages of vehicles on UK roads that meet different emission standards and use a multitude of different exhaust management systems. Some broad principles however would be that for extended periods of time spent stationary, meaning several minutes and more, engine-off would likely be the right action irrespective of vehicle.

11. How will electrification of the vehicle fleet affect exhaust emissions?

Vehicles running solely on battery electricity have no exhaust emissions, and those operating using fuel cells release only water. Hybrid and plug-in hybrid electric vehicle (PHEV) use a combination of battery storage and ICE to supply power. They offer the potential for lower emissions than conventional ICE vehicles, but they require careful management of the engine-on and engine-off balance for optimal low emissions. Cold engine start events lead to elevated emissions, and time spent in electric mode in hybrids may lead to limited ICE warm-up or cooling of the catalyst systems. Therefore, hybrid vehicles that execute multiple stop-starts in urban environments and that require regular use of their on-board ICE may prove to be significant emitters of some pollutants. Consequently, it is not necessarily the case that a hybrid or plug-in hybrid will generate lower overall exhaust emissions than a modern vehicle with a continuously, or near continuously, operating ICE. There is currently insufficient evidence to gauge the significance of hybrid vehicles in influencing ambient pollutant concentrations. Many hybrid vehicles tend to use gasoline, rather than diesel engines and this may cause changes to fleet-average emissions of certain pollutants. New RDE test regimes as part of Euro 7 standards should limit emissions from hybrids.

12. What are the opportunities to further improve air quality using existing technologies?

It is possible to use current technologies for exhaust emissions controls on older vehicles through retrofitting. The cost of retrofitting is such that the economic case is stronger for vehicles of higher value and those with a long service life. Historically, the main retrofit technologies have targeted city buses and trucks, initially with particle filters and then particles filters with SCR to improve their NO_x control. Recently light duty retrofit systems, designed for Euro 5 passenger cars, have been developed with addition of an SCR system to existing DPF to improve the NO_X control and reduce tailpipe emissions. This kind of retrofit, if implemented, would then allow Euro 5 vehicles to access certain cities which have low emissions zones in place and that ordinarily may restrict access (or charge more).

The future introduction of Euro 7 will require the implementation of both existing and some new technologies to control emissions of currently regulated pollutants plus some newly introduced species. At Euro 7, vehicles will combine ICE with advanced aftertreatment and electrification, enabling operation in electric-only mode or engine operation with minimal emissions, in those areas of air quality concern.

13. What interventions are on the horizon for reducing exhaust emissions?

There are a broad range of possible interventions to reduce the emissions and impacts of exhausts from ICE vehicles. Tests of particle number emissions have been shown to identify the illegal removal of DPFs. The addition of such tests to PTIs would help to prevent the use of vehicles which have had their DPFs deliberately removed.

The introduction of additional exhaust gas aftertreatment, in combination with more realistic testing of vehicle emissions on-road, is likely to lead to further lowered exhaust emissions from new models introduced to the fleet in the future.

Future vehicles are likely to continue the trend for increasing vehicle automation and this has the potential to positively reduce emissions not only through reduction of the amount of aggressive driving, but also through an increased ability to predict and adapt to developing situations, for example avoiding congestion.

Newer vehicles may be better connected to transport infrastructure allowing dynamic selection of energy sources to be made (e.g. battery or ICE), depending on local **c**onditions and air quality requirements.

Whilst new vehicles are likely to have significantly lower exhaust emissions than older vehicles, interventions that manage the entire fleet will still be necessary since fleet turnover is relatively slow and is expected to be slowed down by COVID-19. Substantial numbers of Euro 5 and 6 vehicles will likely be on UK roads for at least the next decade, so clean air zones and new variants such as geofencing and emissions-based charging will play a part in managing urban exhaust emissions.

14. How do inventories and traffic emissions modelling represent the variability in exhaust emissions?

Highly detailed vehicle emission models can simulate instantaneous emissions for an individual vehicle under a specific set of conditions, e.g. on a second-by-second basis during acceleration and deceleration. The simulations are dependent on engine speed and torque, and account for the behaviour of any exhaust aftertreatment system. Conceptually simpler approaches are often used to model national fleet emissions over the course of a longer time period using drive-cycle or speed-average emission factors (EF) and traffic data. On a highly resolved spatial and temporal scale, these models may produce different results.

The UK National Atmospheric Emissions Inventory (NAEI) uses average speed related emission factors in g/km for a range of different vehicle types, engine sizes or vehicle weights, fuel types and Euro classes, along with annual traffic activity data from the Department for Transport (DfT) for these vehicle types for each road link, taking account of road type. Adjustment factors are made to account for degraded and failed emission control systems, road gradient, vehicle load for heavy goods vehicles, and changed fuel quality. Additional emissions due to cold starts are included based on average trip lengths.

Instantaneous emission models are better able to represent the variability in exhaust emissions at street level than average-speed related models but are often not practical to use for predictive air quality modelling. Emissions derived from the average speed approach are considered suitable for modelling and assessments for Local Air Quality Management (LAQM) for reasons of practicalities with the availability of suitable traffic activity data and emission factors and with cost and consistency considerations.

15. How are past changes in exhaust emissions reflected in inventories?

The NAEI represents changes in exhaust emissions at national level using average speed-related emission factors for different vehicle types, fuel types and Euro emission standards in conjunction with trends in UK vehicle activities and fleet composition based on traffic and licensing statistics from DfT as well as Automatic Number Plate Recognition (ANPR) data from DfT's roadside surveys. The emission factors come from sources recommended for national inventory reporting to the EU National Emissions Ceilings Directive (NECD). The trend in emissions shown by the NAEI reflects the changes in traffic and fleet penetration at national level of vehicles complying with tighter Euro emission standards. The trend does not reflect differences in emissions between vehicles conforming to the same Euro standards but using different control technologies, as evidenced by measurements using roadside remote sensing. There is a need for inventories to use more detailed activity data and emission factors that reflect technological differences and the dependencies on environmental factors such as ambient temperature.

The NAEI provides spatially resolved maps of emissions according to DfT traffic count data on individual road links but using national trends in fleet composition, supplemented with local information for London. There are likely to be local differences in the fleet which are not at present fully reflected in the NAEI and the Pollution Climate Mapping (PCM) model used for compliance reporting under the EU Ambient Air Quality Directive (AQD). These differences will be important in understanding current emissions locally and quantifying the effect of local policies restricting vehicle movements according to fuel type or Euro class. Greater access to local fleet data from ANPR sources or vehicle ownership would enable more accurate local inventories to be developed.

16. What are projected future exhaust emissions?

The NAEI projects future emissions at the national level using information from DfT on traffic forecasts, future fleet composition taking account of fleet penetration of emission standards up to Euro 6d/VI as well as battery and hybrid electric vehicles. Detailed traffic forecasts and fleet projections for London taking account of the Ultra Low Emission Zone (ULEZ) are also accounted for, but no other local measures are currently taken into consideration. According to the latest projections from the NAEI, NO_X emissions are predicted to decrease by 65% by 2030 relative to 2017 levels but remain dominated by diesel cars and LGVs (79%). Exhaust emissions of PM are also expected to fall by 80% by 2030 from 2017 levels. These projections do not take account of any short or long-term changes that might result from the COVID-19 crisis (e.g. in terms of vehicle usage or speed of fleet turnover).

17. Does the UK have the right monitoring/measurement programmes in place to assess the effectiveness of future interventions to limit exhaust emissions?

Current UK measurement networks are not optimised to assess the long-term changes in exhaust emissions. This could be improved by creating paired traffic and urban background monitoring sites in the major urban areas of the UK along with localised measurement of traffic flow and composition. An alternative is to make very high time resolution (seconds) pollutant measurements allowing scale separation of very local and background emissions. It is important to retain existing long-term monitoring sites, and routine analysis of rates of change in ambient air pollution concentrations should be undertaken, ideally over periods of five years or more. Comparison of rates of change in ambient concentrations to those projected from emissions inventories would provide improved feedback and verification on the success of exhaust abatement policies.

Analysis of ambient air pollution concentrations shows that the rate of change in exhaust emissions varies from place to place with no clear pattern. More detailed analysis is required to determine the reasons for these differences and the opportunities for optimising policies. Retaining a wide geographical spread of monitoring sites will maximise the potential for identifying disparate changes. Spatially refined monitoring studies may prove useful in identifying any micro-scale variations in emissions associated with existing or new vehicle technologies. If new pollutants are introduced into the emissions standards for vehicles, some revision of ambient monitoring infrastructure (e.g. for NH_3) will be required to evaluate the behaviour of these pollutants. This has proven critical in the past to independently assess emissions.

1. Introduction

Exhaust emissions from internal combustion engines (ICE) result from high temperature burning of hydrocarbons (HC) in the fuel in the presence of oxygen (O_2) and nitrogen (N_2) in the intake air. The vast majority of the fuel is converted via oxidation to carbon dioxide (CO_2) and water (H_2O) which, together with unused O_2 and N_2 from the intake air comprise approximately 99.9% of the exhaust exiting diesel and gasoline engines. Along with CO_2, it is the minor exhaust species, present at parts per million (ppm) and parts per billion (ppb) levels and comprising the remaining ~0.1% of exhaust, that are of environmental concern, either as air pollutants or greenhouse gases (GHG). The fuel itself comprises many individual HC species, for example: gasoline contains ~150 dominant species, and at least a further 1000 minor compounds. As well as the oxidised species, emissions comprise unburned fuel and lubricant, and matter deriving from wear of engine components. Many of the oxidised species have low volatilities and so readily condense forming particles in ambient air. Another important component of exhaust is nitrogen oxides (NO_X), formed from the oxidation of N_2 by O_2, both from the intake air, under the high temperatures of the combustion process.

As many of these exhaust pollutants are harmful to human health or impact climate, or both, emissions from vehicles have been subject to regulation for many years. Euro standards were introduced in the early 1990s and have been progressively made more stringent by extending the range of regulated pollutants and reducing the permissible levels of emissions (Euro 1 to Euro 6 for light duty vehicles (LDV) and Euro I to VI for heavy duty vehicles (HDV)). The pollutants regulated in Europe are carbon monoxide (CO), NO_X, mass of particulate matter (PM) and particle number (PN), along with total HC (THC) and non-methane HC (NMHC) for petrol engines and HC+NO_X for diesel engines. Ammonia (NH_3) is also included in the Euro VI standards for heavy-duty diesel engines. Motorcycles require the control of CO, NMHC, THC, NO_X and from Euro 5, PM. CO_2 is also controlled, but on a fleet-average rather than individual vehicle basis. Regulations requiring higher quality fuels, especially those with lower sulphur content, have been so effective that emissions of sulphur containing pollutants from road vehicle exhausts are negligible.

To comply with Euro standards, new technologies for reducing emissions of the regulated pollutants, such as three-way catalysts (TWC) and diesel particulate filters (DPF), have been developed. The reduction in sulphur content of fuels has been critical in permitting the advancement of these exhaust aftertreatment systems that would otherwise be rendered ineffective by sulphur poisoning of the catalyst systems. In combination with advanced engines, these have reduced emissions of many of the regulated pollutants considerably (i.e. NO_X, CO and HCs from petrol engines and particles from diesel engines), but there are still concerns.

Ambient measurements have previously revealed important disparities between the changes in concentrations and those projected from vehicle emissions standards. Improved vehicle emissions standards should have led to decreases in concentrations of exhaust pollutants from traffic, however analysis of measurements revealed some increases in air pollutants from traffic or downward trends being weaker than suggested by changes to the emissions standards.

Some of these new technology combinations have had unintended consequences or not been as effective as anticipated, particularly for diesel vehicles. For example, the way that air/fuel ratios are adjusted to optimise diesel engine efficiency (reducing fuel consumption and CO_2 emissions) makes NO_x control more difficult. Methods to address this include the use of additional exhaust aftertreatments such as selective catalytic reduction (SCR) systems or lean NO_x traps (LNT), but these can lead to emission of additional pollutants. These include ammonia (NH_3) and nitrous oxide (N_2O) as well as increasing the proportion of NO_X emitted as nitrogen dioxide (NO_2) from diesel vehicles.

In recent years, the vehicle fleet has evolved to contain a greater proportion of diesel cars, stimulated by a desire to reduce CO_2 emissions to protect the climate. In addition, there has been the development of hybrid and semi-automated vehicles and the implications of these for exhaust emissions need to be considered.

Emissions standards require an associated testing procedure, which sets out the drive cycles for which new vehicles are tested and pollutants measured. For many years the New European Drive Cycle (NEDC), which was designed to represent the typical usage of a car in Europe, had been used. However, it was recognised that emissions under real world driving conditions could be much higher than those seen during the laboratory based NEDC tests. This led to the Euro 6c standard which involves laboratory testing with more realistic drive cycles (Worldwide harmonised Light duty vehicles Test Procedure (WLTP)) and to Euro 6d-temp and Euro 6d which are supplemented by real driving emissions (RDE) tests. The emissions limits have not changed from Euro 6b, but the tests have become more demanding and the testing procedures more stringent.

These standards apply when a new vehicle is approved and on first registration, but some exhaust emission control technologies have been shown to become less effective, for example where particle filters become cracked and allow more material to pass out into the air. This is leading to the development of new Periodic Technical Inspection (PTI) procedures.

These changing technologies and regulations have implications for policy. The introduction of new exhaust technologies has implications for the UK in terms of meeting the EU National Emissions Ceilings Directive (NECD) and EU Air Quality Directive (AQD) as well as emissions ceilings under the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP). The UK's National Atmospheric Emissions Inventory (NAEI), which is used to assess compliance with the NECD needs to reflect these changes. Moreover, we need to understand the effectiveness of the new technologies and other interventions, such as low emissions zones, in improving air quality and compliance with the AQD.

This report therefore addresses the following policy-relevant questions:

- How are exhaust emissions from vehicles regulated and for which pollutants?
- Are there pollutants which are not currently regulated by the Euro systems but which are known to be emitted from vehicle exhaust?
- What are the main engineering approaches used to reduce vehicle exhaust emissions?
- How have the introduction of more stringent Euro classes affected exhaust emissions?
- Have ambient concentrations of exhaust pollutants changed in response to the tightening of Euro standards?
- What is the significance of recent changes to the regulatory test cycle?
- How are emissions affected by engine operating conditions?
- How do emissions change with different driving conditions and styles?
- Does the evidence allow conclusions to be drawn on optimum road planning, traffic control, or driving styles that can reduce air pollution?
- Can advice be provided on the air quality impacts of idling vs engine switch off when stationary, and how this varies with vehicle type, outdoor temperature and time spent stationary?
- How will electrification of the vehicle fleet affect exhaust emissions?
- What are the opportunities to further improve air quality using existing technologies?
- What interventions are on the horizon for reducing emissions?
- How do inventories and traffic emissions modelling represent the variability in exhaust emissions?
- How are past changes in exhaust emissions reflected in inventories?
- What are projected future exhaust emissions?
- Does the UK have the right monitoring/measurement programmes in place to assess the effectiveness future interventions to limit exhaust emissions?

These questions define the scope of this report. The report discusses exhaust emissions from all road traffic and focusses on non-CO_2 pollutants, but it is important to note that the impact of road traffic exhaust emissions on both air quality and climate should be considered together in the development of new technologies and policies. This has been highlighted by the impact shifting to diesel vehicles in the interest of CO_2 emissions had on air quality. CO_2 is not the only exhaust gas of concern for climate. For example, N_2O is a greenhouse gas, and NO_x, CO and HCs can lead to the formation of ozone which is also an important greenhouse gas. Moreover, systems thinking is required with consideration of the environmental impact of vehicles throughout their life cycle.

Specific answers to the questions above are given in the Executive Summary, whilst more background information and detail are provided in the following chapters.

Chapter 2 details European regulations, introduces the different exhaust pollutants, describes how the exhaust emission control technologies impact the emissions of pollutants and how these technologies might be affected by operating conditions (e.g. temperature, air/fuel ratio).

Chapter 3 describes the testing procedures, how they have transitioned from NEDC to WLTP and RDE, considers how driving conditions affect emissions and assesses trends in emissions based on measurements of vehicle exhausts.

Chapter 4 describes how exhaust emissions are accounted for in emission models and inventories. It discusses uncertainties, the spatial and temporal variabilities, and trends in emissions in the UK National Atmospheric Emissions Inventory (NAEI), and projected future emissions.

Chapter 5 describes the evidence for past changes in exhaust emissions based on measurements of ambient concentrations and evidence from clean / low emission zones. It discusses whether the UK has the right monitoring programmes in place to assess the effectiveness of future interventions to limit exhaust emissions.

Chapter 6 discusses new vehicle technologies that are being developed and how they might impact future exhaust emissions. It also considers potential new testing procedures for regulating emissions.

Chapter 7 provides recommendations.

2. Exhaust Emissions: regulations, pollutants and technologies

2.1 Emission Regulations

2.1.1 European Standards

When using hydrocarbon-based fuels, the aim is to achieve total combustion of the fuel, leading to the production of carbon dioxide, water and heat and no other compounds. However, due to incomplete combustion, and sometimes during high temperature combustion, other species are formed which need to be controlled. Also, the catalytic components of the exhaust system can create other species which also need to be controlled. Currently, only a limited selection of the species formed by combustion and catalysis are regulated, and others will be regulated in the future.

Tables 2.1 and 2.2 show summaries of the evolution of European emissions legislation for both passenger cars and heavy duty vehicles. Advancing Euro standards have continuously reduced the exhaust emissions limit for on-road transport. For passenger cars there are different emissions limits for gasoline and diesel including grouping of species. For example, nitrogen oxides (NO_x) and total hydrocarbons (THC) are combined for diesels, but separate for gasoline. The heavy duty legislation has to be met over both steady state and transient cycles each with their own emissions limits.

Table 2.3, shows the motorcycle legislation for 4 stroke applications, with Euro 5 tending towards similar limits to gasoline light duty legislation with a delayed timeline.

Table 2.1: European Light Duty (LD) Emissions Legislation

Legislation	Initial Date	CO g/km		THC g/km		THC + NO_x g/km		NO_x g/km		PM g/km		PN #/km	
		Gasoline	Diesel	Gasoline	Diesel	Gasoline	Diesel	Gasoline	Diesel	Gasoline	Diesel	Gasoline	Diesel
Euro 1	1992	2.72	2.72	---	---	0.97	0.97	---	---	---	0.140	---	---
Euro 2	1996	2.20	1.00	---	---	0.50	0.90	---	---	---	0.100	---	---
Euro 3	2000	2.30	0.64	0.2	---	---	0.56	0.15	0.50	---	0.050	---	---
Euro 4	2005	1.00	0.50	0.1	---	---	0.30	0.08	0.25	---	0.025	---	---
Euro 5	2009	1.00	0.50	0.1	---	---	0.23	0.06	0.18	0.005	.005	---	6×10^{11}
Euro 6	2014	1.00	0.50	0.1	---	---	0.17	0.06	0.08	0.005	0.005	6×10^{11}	6×10^{11}

Table 2.2: European Heavy Duty Vehicle (HD) Emissions Legislation

Legislation	Initial date	CO g/kWh		HC g/kWh		CH_4 g/kWh		NO_x g/kWh		PM g/kWh		PN #/kWh		NH_3 ppm	
		Steady state	Transient	Steady state	Transient	Steady state	Transient	Steady state	Transient	Steady state	Transient	Steady state	Transient	Steady state	Transient
Euro I		4.50	---	1.10	---	---	---	8.0	---	0.36	---	---	---	---	---
Euro II		4.00	---	1.10	---	---	---	7.0	---	0.15	---	---	---	---	---
Euro III		2.10	5.45	0.66	0.78	---	1.6	5.0	5.0	0.10	0.16	---	---	---	---
Euro IV		1.50	4.00	0.46	0.55	---	1.1	3.5	3.5	0.02	0.03	---	---	---	---
Euro V		1.50	4.00	0.46	0.55	---	1.1	2.0	2.0	0.02	0.03	---	---	---	---
Euro VI		1.50	4.00	0.13	0.16	---	0.5	0.4	0.46	0.01	0.01	$8x10^{11}$	$6x10^{11}$	10	10

Table 2.3: European Motorcycle Emissions Legislation

Legislation	Initial Date	CO g/km	THC g/km	NO_x g/km	PM g/km
Euro 1	2000	13	3	0.3	---
Euro 2	2003	5.50	1	0.3	---
Euro 3	2014	2.62	0.33	0.22	---
Euro 4	2017	1.140	0.380	0.070	---
Euro 5	2020	0.500	0.100	0.060	.0045

Table 2.4 shows the legislative fleet average CO_2 requirement for passenger cars, over the legislative cycle called the World Light Duty Test Cycle (WLTC). The CO_2 limits are linked to vehicle mass. For 2020 the average mass of a European vehicle is 1380 kg. For a specific manufacturer, its average vehicle mass is used to determine its fleet average limit using the equation below. The specific emissions target for a manufacturer in 2020 is the average of all the specific emissions targets for each registered vehicle.

Specific emissions target (g/km CO_2) = $95 + 0.0333 \times (M - M_0)$, where M_0 is 1380 kg in 2020 and M is the average mass of the specific manufacturers fleet sales.

A heavy-duty CO_2 regulation will be implemented from 2025, requiring a 15% reduction in emissions compared to reference data reported by the manufacturer for the period from 1 July 2019 to 30 June 2020. From 2030 onwards the CO_2 regulation will require a 30% reduction in the reference emissions.

Table 2.4: Passenger Car CO_2 Fleet Average over the WLTC

Passenger Cars		
Date	% Fleet	Fleet Average CO_2 limit g/km
2012	65	130
2013	75	130

Passenger Cars		
2014	80	130
2015 -2019	100	130
2020	95	95
2021	100	95
2025	100	81
2030	100	59

2.1.2 Currently Regulated Emissions for Light Duty Vehicles (LD) and Heavy Duty Vehicles (HD)

- Carbon monoxide (CO) – Formed via partial oxidation of all hydrocarbon-based fuels during all engine operating modes. CO can be formed under high temperature operation when over-fuelling is used to cool the exhaust system, in order to protect exhaust components.
- Carbon dioxide (CO_2) – Formed via total oxidation of all hydrocarbon-based fuels and during all engine operating modes.
- Oxides of nitrogen (NO_x) – Formed via oxidation of nitrogen, in the air drawn in by the engine, at temperatures above 2000 K. Gasoline combustion produces significantly higher engine out NO_X emissions compared to a diesel engine, due to gasoline generally having hotter combustion.
- Total hydrocarbons (THC), which includes methane – This can be either unreacted fuel exiting from the combustion chamber or partial combustion of fuel creating smaller hydrocarbons. There is a lower limit to the volatility of hydrocarbons considered in the THC, which is defined by the measurement technique (See section 3.1.2.1). THC can be formed under high temperature operation when over-fuelling is used to cool the exhaust system for component protection. Hydrocarbons contribute to volatile organic compounds (VOC).
- Methane (CH_4) – This can be in the form of unreacted fuel if the engine operates on natural gas, can be a product of partial combustion of higher hydrocarbon-based fuels or created across certain types of exhaust catalysts.
- Ammonia (NH_3) (regulated for heavy duty vehicles only) – NH_3 is formed over catalysts by the reduction of NO_X emissions, including with selective catalytic reduction (SCR) systems fitted to diesel vehicles and TWC fitted to petrol vehicles.

- Particulate matter mass (PM) – Particulate matter is solid phase materials collected on a filter which can contain black carbon (BC), low volatility hydrocarbons from fuel and oil, nitrates, sulphates and ash. For regulatory purposes, it is defined as the amount of solid residue collected on a filter. PM from exhausts is mainly a product of incomplete combustion of hydrocarbon-based fuel, and a contribution from the engine oil. The type of fuel and combustion approach plays a significant role in the amount of PM formed, for example diesel engines produces more engine-out PM than gasoline.
- Particle number (PN) – This was introduced to enable the mandatory use of particle filters for diesel applications where the mass of emissions was so low as to be difficult, or impossible to accurately measure, despite there being a substantial number of particles emitted. In current automotive regulations, PN is the total number of individual thermally-defined non-volatile single particles and agglomerates produced by the engine, as measured by a particle counter with a lower size limit of 23 nm. The number of volatile particles is not currently subject to legislation in Europe, or elsewhere.

2.1.3 Future Euro 7 Standards and Potential Newly Regulated Exhaust Pollutants

Future exhaust aftertreatment systems (ATS) will become even more complex as they continue to control currently regulated emissions but also address emissions of some species which are not currently regulated. Some of the new species that will be regulated in the Euro 7 emissions regulations are formed over catalyst systems. Hence care has to be taken in order to minimise the formation of secondary emissions via catalyst specification and calibration. Table 2.5 shows the potential new species which may be regulated and how they are formed in combustion engines or over the catalyst system. The new species are either greenhouse gases, such as nitrous oxide (N_2O) and CH_4, or air quality impactors, such as NO_2, ammonia and formaldehyde.

Table 2.5. Potential Future Regulated Emissions in Euro 7

Species	Concerns
Nitrogen Dioxide – NO_2	Harmful to health and biodiversity
Nitrous Oxide – N_2O	Greenhouse gas
Ammonia – NH_3	Odorous, particle formation, harmful to biodiversity
Methane – CH_4	Greenhouse gas
Formaldehyde – CH_2O	Low level ozone, odorous, potentially health relevant
<23nm PN	Carrier of harmful compounds

2.2 Emissions Control Technologies for Light Duty and Heavy Duty Vehicles

Exhaust emissions control technologies can be categorised into two main sections, those that perform oxidation reactions, namely the oxidation of carbon, CO and hydrocarbons, and those that perform reduction reactions, principally the reduction of NO_x. However, a single catalyst, given the correct conditions, can perform both oxidation and reduction reactions. Oxidation catalysts convert CO and hydrocarbons to CO_2, but the relative amounts of CO and hydrocarbons in the exhaust, compared to the fuel burned, mean the contribution of CO_2 formed over the catalyst system is negligible compared to the CO_2 from the engine.

The three-way catalyst (TWC) used in stoichiometric gasoline applications oxidises both CO and HC. It oxidises CO to CO_2 and HC to CO_2 and H_2O. It also reduces NO_x to N_2 where CO is used as the reducing agent for NO_x control. A TWC can be found in vehicles from Euro 1 to Euro 6.

Gasoline particulate filters (GPF) trap particles and periodically oxidise the trapped carbon to give CO_2. The direct oxidation of C with O_2 is the carbon control reaction. GPFs were introduced on direct injection gasoline vehicles at Euro 6c, initially against a less-stringent particle number limit than applies to diesel. At this time GPF with efficiencies >60% were observed. However, earlier introductions will have occurred prior to introduction of PN legislation for gasoline applications and these can show higher filtration efficiencies. Gasoline particle number limits are now harmonised with diesel, and more recent GPF consistently show efficiencies for solid particles from 80%, and in some cases exceeding 90%.

Diesel oxidation catalyst (DOC) applications oxidise CO and HC, with CO oxidising to CO_2 and HC oxidising to CO_2 and H_2O. DOC also oxidises NO to NO_2 which can be used further down the exhaust system, particularly for soot control and SCR systems. The DOC can also provide heat by burning fuel to thermally manage the exhaust system. A DOC can be found in vehicles from Euro 2 to Euro 6

Diesel particulate filters (DPF) trap particles and periodically oxidises the trapped carbon to give CO_2. There are 2 mechanisms for carbon control in diesel, direct oxidation of C with O_2 and oxidation of C with NO_2, which is formed on the upstream DOC or in the DPF itself. In retrofit applications for buses, the DPF is commonly known as a continuously regenerating trap (CRT), where the reaction of NO_2 with C controls the level of PM in the filter by reducing the carbon element. DPF can be found in vehicles from Euro 5 to Euro 6 as well as in retrofit applications. They can be extremely efficient, with efficiencies for solid particles ranging from 95 - >99%.

A lean NO_x trap (LNT) stores NO_2 under lean conditions (i.e. high air/fuel ratio, where air is in excess) and then when full or partially full removes and reduces the NO_x via operating the engine rich (i.e. low air/fuel ratio). The LNT can also act as an oxidation catalyst. LNT is generally only found in certain Euro 6 passenger cars.

Selective catalytic reduction (SCR) uses an added reductant to reduce NO_x to N_2. The added reductant is NH_3 which is formed from the hydrolysis of a urea-water solution which is periodically injected into the exhaust system to control NO_x emissions. The NO_2 that is formed over the DOC assists in improving the NO_x reduction efficiency, in the temperature region of 200-300^{o}C. SCR is generally found in Euro 6 passenger cars and from Euro IV for HD applications. A DPF with SCR activity is known as SCR on-filter (SCRF)

An ammonia slip catalyst (ASC) controls NH_3 emissions from the SCR system. The NH_3 is oxidised to N_2. ASC is generally on found on Euro 6 passenger cars, and Euro VI heavy duty vehicles.

A passive NO_x adsorber (PNA) stores NO at low temperatures and then releases the NO at a higher temperature, ideally when the SCR system is operating efficiently. The PNA can also act as an oxidation catalyst. PNA may be used at Euro 6 for certain passenger cars.

Exhaust Gas Recirculation (EGR) is used to control engine out NO_x emissions. This is where exhaust gases are recirculated to the intake, and hence the combustion chamber, to reduce peak temperatures, limiting NO_x formation by the addition of inert components which dilute the fuel-air mix. EGR is used mainly on diesel applications but may be used on gasoline applications.

For gasoline passenger car applications, historically the main technology used was the three-way catalyst (TWC). Gasoline engines can produce particles hence, in Euro 6 emissions regulations, a gasoline particulate filter (GPF) has been added to the exhaust system to control particulate matter. Current gasoline applications apply both a TWC and a GPF to control regulated pollutants.

Tables 2.6**Tabl** and 2.7 show examples of emissions control technologies used for light duty and heavy duty vehicles from Euro 1/I to Euro 6/VI/5. Earlier emissions standards required only relatively simple aftertreatment, whereas moving to Euro 6/VI the aftertreatment system became significantly more complex, with up to 4 catalysts being required for diesel passenger cars.

The emissions control technology for diesel vehicles is significantly more complex than gasoline. Also, diesel engines are used in more sectors such as passenger cars, trucks and off-highway machinery unlike gasoline which is only used for light duty applications. The nature of the fuel used in diesel combustion, and the type of fuel and air mixing in the engine, leads to high rates of production of PM hence the requirement for an exhaust diesel particulate filter (DPF).

Table 2.6: Example of aftertreatment technologies used for light duty from Euro 1 to Euro 6.

Legislation	Date	Passenger Car Diesel	Passenger Car Gasoline
Euro 6	2014	EGR + DOC + DPF + SCR/ASC EGR + LNT + DPF + SCR/ASC EGR + DOC + SCRF+ ASC EGR + LNT + DPF	TWC + GPF
Euro 5	2009	EGR + DOC + DPF	TWC
Euro 4	2005	EGR + DOC	TWC
Euro 3	2000	EGR + DOC	TWC
Euro 2	1996	EGR + DOC	TWC
Euro 1	1992	EGR + DOC	TWC

Table 2.7: Example of aftertreatment technologies used for Heavy Duty vehicles from Euro I to Euro VI.

Legislation	Date	HD Diesel
Euro VI	2013	DOC + DPF + SCR/ASC
Euro V	2008	DOC + SCR
Euro IV	2005	SCR
Euro III	2000	Engine Control
Euro II	1998	Engine Control
Euro 1	1992	Engine Control

Due to the diesel engine operating with excess oxygen to achieve optimum efficiency, CO and HC are readily controlled using a diesel oxidation catalyst (DOC), but the excess oxygen provides a significant challenge for controlling NO_x. There are two approaches to NO_x control which are used in tandem, exhaust gas recirculation (EGR) and exhaust emissions control. EGR is where exhaust gas is recirculated back into the combustion chamber in order to reduce the peak combustion temperature, and oxygen level, and reduce the combustion-generated NO_X emissions. EGR can be taken from the exhaust prior to, and/or after, the catalysts, known as high pressure and low pressure EGR. NO_x exhaust emissions control can be by lean NO_x traps (LNT) and selective catalyst reduction (SCR) systems. The LNT operates by storing NO_x and periodically removing then reducing the NO_x through operating the engine in an excess fuel condition for a few seconds. The SCR system requires the injection of urea solution into the exhaust which decomposes to form NH_3. The NH_3 is stored on the SCR catalyst and reacts with the NO_x to form N_2. Currently the dominant exhaust technology for controlling NO_x emissions is SCR, with LNT only being used in limited diesel passenger car applications. There are, however, passenger cars that use both LNT and SCR to control NO_x emissions. All Euro VI HD diesel applications use SCR for NO_x control.

All catalyst systems undergo deactivation throughout their lifetime. There are two main deactivation mechanisms, thermal deactivation and chemical poisoning. Thermal deactivation is when the catalyst exceeds its thermal limit and its efficiency is permanently reduced. Chemical poisoning is when certain compounds from the lubricant oil exit the combustion chamber and deposit on the catalyst leading to a physical blockage of the active sites of the catalyst, hence impacting its efficiency. A significant amount of engineering is used to minimise both thermal deactivation and chemical poisoning of the catalyst system to maintain its efficiency.

2.3 Emissions Control Technologies for Motorcycles

Table 2.8 shows examples of emissions control technologies for motorcycles from Euro 1 to Euro 5.

Table 2.8: Example of aftertreatment technologies used for Motorcycles from Euro 1 to Euro 5

Legislation	Date	Motorcycles
Euro 5	2020	TWC
Euro 4	2016	TWC
Euro 3	2014	TWC
Euro 2	2003	Engine Control
Euro 1	2000	Engine Control

Currently motorcycles only require the use of a TWC and not a GPF and hence have no requirement for exhaust control of particulate matter, and a PM limit to be introduced at Euro 5 (2020) is unlikely to result in widespread GPF fitment.

2.4 Impact of Exhaust Emissions Technology on Emissions

The introduction of exhaust emissions control technologies has had the desired impact on reducing emissions of carbon monoxide, hydrocarbons, oxides of nitrogen and particulate matter from internal combustion engines, effects that have been measured by on-road or roadside measurements.

Fundamentally the carbon content of the fuel and the efficiency of the engine determine the CO_2 emissions of the application. Low carbon fuels, such as natural gas, have a lower carbon content compared to gasoline and diesel. However, the diesel engine has a higher efficiency compared to gasoline due to the higher energy content of the fuel, and its ability to operate lean (at high air-fuel ratio) to reduce the energy required to move air into, and exhaust out of, the cylinder ("pumping losses").

In order to continue to control regulated emissions to meet even lower limits, the aftertreatment systems of gasoline and diesel vehicles are becoming increasingly complicated. Gasoline applications historically used only a TWC in many instances. However, now with the requirement for particle number control, gasoline vehicles will use a particle filter, generally post TWC. In addition to more catalysts, exhaust sensors are required to monitor and feedback information to the vehicle's control unit in order to optimise the efficiency of the aftertreatment system.

Diesel applications can readily have 3 or more catalysts in an exhaust system, a typical exhaust having an oxidation catalyst then a particulate filter followed by an SCR system. This level of complexity requires a significant level of control, needing an appropriate number of sensors. A diesel exhaust can have oxygen, NO_x, temperature and pressure sensors to ensure the aftertreatment system is functioning in its optimum efficiency window.

With more catalyst systems in the exhaust care must be taken to ensure limiting secondary emissions which may be formed over the catalyst. For example, NH_3 and N_2O are not formed in the combustion process but are formed over a range of different catalysts under varying exhaust gas compositions. Under low oxygen concentration conditions, NH_3 can be formed, and under higher oxygen concentration conditions, N_2O can be formed. Hence a robust calibration, leading to the catalyst having their optimum conditions, along with optimal catalyst specification can lead to control of secondary emissions. There will be a stronger focus on limiting these secondary emissions for the post Euro 6/VI emissions legislation.

2.5 Impact of Engine Operation on Emissions

Vehicle operating conditions and environmental factors such as pressure, temperature, driving style, road gradient and traffic congestion have a greater influence on the emissions now than prior to the introduction of exhaust emission technology. This is because the exhaust emission technology is often sensitive to the temperature or fuel mix which can be affected by the environmental and driving conditions.

In order for the exhaust emissions control technology to operate most efficiently, the catalyst must be hot (>300^{o}C), which means that during a cold start of the vehicle the catalyst systems require a period of time to warm up and in this period the catalyst is increasing in its efficiency.

For optimum catalyst efficiency the engine needs to operate in stoichiometric air/fuel conditions, which is when the combustion process uses just enough oxygen to burn all the fuel. The engine, including the control strategy, and combined with exhaust sensors, provides the catalyst with the optimum conditions required to maintain efficiency. Diverging from stoichiometry can impact the NO_x conversion efficiency of the TWC and can have the undesired effect of producing nitrous oxide (N_2O). Operating the engine with reduced air (excess fuel) will impact the HC and CO conversion efficiencies resulting in high total hydrocarbon (THC) and CO emissions as not all the fuel is consumed during the combustion process and have the further undesired effect of reducing NO_x to ammonia (NH_3) over the catalyst.

The impact of various operating parameters on emission are illustrated below.

Cold start

- Cold start is the first start of the engine where air, engine coolant and exhaust systems are at same temperature. Currently the legislative cold start temperature is ~25ºC in the laboratory.

- Both gasoline and diesel engines use excess fuel to start the engine to compensate for thermal losses and provide stable start. As described above excess fuel can lead to undesired emissions.

- In gasoline engines, a three-way catalyst is used to reduce the gaseous emissions. As described above the TWC catalyst must be hot (>300°C) to work efficiently, so a cold start results in high tailpipe emissions before the catalyst gets to optimum temperature.

- In gasoline direct injection (GDI) engines, PN emissions are higher during cold start compared to port fuel injection (PFI) engines due to reduced time for fuel atomisation and the associated fuel impingement on the wall of the combustion chamber leading to localised fires and soot/particle production. Most GDI engines generate one to two times more particulate matter (PM) emissions than conventional PFI engines during cold start. Fuel impingement on the wall of the combustion chamber is increased due to poor atomisation during cold engine operation. This leads to higher PN in cold start.

- In diesel engines, cold start also results in an increase in CO, THC and particle emissions due to reduced catalyst efficiency.

Ambient temperature (<25ºC)

- An ambient start occurs when the air temperature is at ambient conditions, but engine coolant and the exhaust system might be warmed from prior operation.

- Both gasoline and diesel engines require excess fuel to start the engine to compensate for thermal losses and provide a stable start. Lower temperatures increase the energy and fuel required.

- Emissions increase with cold ambient conditions due to the catalyst system taking longer to warm up and start controlling emissions.

Driving Style

- Driving style also influences emissions. A gasoline TWC needs to operate at the stoichiometric air/fuel ratio for optimum conversion, with the ratio oscillating to fractionally rich and fractionally lean conditions to allow the catalyst to both oxidise and reduce emissions. Controlling the air/fuel ratio at around stoichiometric conditions is difficult during vehicle transient behaviour, for example during an acceleration. High torque is demanded in a short period during acceleration, and this is achieved by delivering more fuel. The result is that the air/fuel ratio is lower than the required stoichiometric level during acceleration. This results in poor gaseous emissions conversion over the catalyst, and all the gaseous emissions increase compared to steady state operation.

- During acceleration, fuel impingement increases and results in more particle number (PN). This affects both gasoline and diesel engines.

- NO_x emissions from diesel engines peak during load transients, emissions levels depending strongly on engine calibration. Reported data shows examples where 50% of the NO_x emissions occur in only 8% of the journey duration, and, alternatively, where 80% of the NO_x emissions occur in 20% of the journey duration (Mera et al, 2019).

Component Protection

- In gasoline engines, exhaust temperatures can exceed the thermal tolerances of specific components when operating at higher engine speeds and high loads. This can result in melting of exhaust components, including catalyst materials. This is avoided by injecting additional fuel to cool down the exhaust system and to maintain the exhaust temperature below a necessary limit. This excess fuelling is called fuel enrichment and it produces high engine out CO and THC emissions. Catalytic conversion efficiency is also poor during this time, as the engine operates in a very low air/fuel ratio range with no oxygen and tailpipe emissions are high. Emission test cycles like the World Harmonized Light Duty Test Cycle (WLTC) or the US Federal Test Procedure (FTP) do not require measures for component protection, but real drive emission (RDE) can be affected by fuel enrichment.

Engineering Trade-Off

- Engineering approaches to meet limit values trade-off certain pollutants between the engine and the aftertreatment system, by allowing higher engine-out emissions when aftertreatment efficiency is highest, and vice-versa

- Prior to Euro 5, few diesel cars were equipped with SCR or LNT. For these diesels, engine calibration was always focused on the trade-off between NO_x and PM while minimizing CO_2 emissions. Euro 6 diesel applications use DPF plus either (or both of) SCR and LNT, enabling the calibration focus to shift more to fuel economy, since the aftertreatment delivers high efficiency for NO_x and PM/PN.

- Platinum group materials (PGM) are expensive and used for three-way catalyst systems in gasoline engines and various diesel ATS. PGM levels are specified based upon engine out emissions, engineering emission targets and total cost of the ATS.
- The filtration efficiency of a GPF is selected based on engine out PN level, the engineering emissions target and the specified exhaust pressure and temperature limit. Increasing the filtration efficiency will result in lower PN emissions, but this can increase back pressure and temperature, which may affect the full load performance and fuel economy. These engineering trade-offs explain why emission variations exist between vehicles of the same emissions standards, despite vehicles being equipped with similar technologies and aftertreatment systems.

Traffic Congestion

- Traffic congestion forces erratic driving and requires the engine to undertake aggressive transient operation, including harsh accelerations and decelerations, and any number of start and stop manoeuvres. The impact of transient operation and multiple starts increases the emissions as explained in the previous 'driving-style' section. As a result, all the gaseous emissions increase significantly compared to free-flowing traffic. This affects both gasoline and diesel engines.

Road Gradient/Altitude

- Road gradient alters the engine operating speed and load. During uphill driving, the engine operates at low speed and high load, whilst downhill driving operates at high engine speed and low load, although SCR efficiency is improved at higher temperatures, which can be associated with uphill gradients. Uphill driving increases all the gaseous engine-out emissions for gasoline and diesel vehicles. Downhill driving has a limited impact on engine out emissions but may affect catalyst temperature.

The aim of the engine and ATS calibration is to ensure control of emissions over all normal driving conditions, including those mentioned above.

2.6 Alternative Fuels for Passenger Cars and Heavy Duty Vehicles

There are a range of alternative "drop-in fuels" that internal combustion engines can operate on that are direct replacements for gasoline and diesel. The challenges with alternative fuels are production and infrastructure/distribution, plus lowered energy density which can lead to reduced operating ranges for vehicles. The main driver for alternative fuels is to move towards lower CO_2 producing fuels. Fuels with a higher ratio of hydrogen to carbon produce less CO_2.

2.6.1 Spark Ignition (SI) Alternative Fuels

- Ethanol (C_2H_5OH) - Ethanol is currently found at low concentrations (<10%) in European gasoline. Modified SI engines operate on up to 85% ethanol with the remainder being gasoline. However, a modified calibration is required to operate on >20% ethanol. A standard TWC will control emissions from ethanol fuelled vehicles.

- Methane (CH_4) – Dedicated methane internal combustion engines currently operate in Europe in the heavy duty market, and use a spark to ignite the fuel, in a similar manner to gasoline. Storage of methane is a challenge where it is either compressed to 300-500 bar or liquified. When operating on methane an upgraded aftertreatment system is required to control methane emissions which is the most stable hydrocarbon and hardest to oxidise.

- Methanol (CH_3OH) - Currently used in low concentrations in China (<15%), allowed in Europe at 3%, but seldom added. Engine modifications would be required to operate on higher methanol concentrations, such as upgraded seals, as methanol is a strong solvent. The aftertreatment system would need to be upgraded to control the increased methane emissions from the fuel compared to a gasoline engine.

- Liquified Petroleum Gas (LPG) – Can be a mixture of propane and butane. Can be used in conventional gasoline engines. The aftertreatment system would potentially need to be upgraded to control increased light hydrocarbons.

- Hydrogen (H_2) – Can be used as fuel for internal combustion engines. Emissions of PN and NO_x expected which will require emissions control technology to limit tailpipe emissions.

2.6.2 Compression Ignition (CI) Alternative Fuels

- Fatty acid methyl ester (FAME) – Renewable fuel which is currently added at low concentrations (≤7%) in European diesel. There are potential issues operating on FAME blends, though some heavy-duty vehicle manufacturers now warrant their vehicles to 30% FAME. The existing diesel aftertreatment systems control emissions from 7% FAME fuelled vehicles.

- Hydrotreated vegetable oil (HVO) – is processed fatty acids to give a more diesel-like fuel. A current diesel engine can operate on 100% HVO. The existing diesel aftertreatment systems control emissions from HVO fuelled vehicles, and diesel HVO blends.

- Biomass to liquid (BTL) – A diesel replacement fuel, where a current diesel engine can operate on 100% BTL. The existing diesel aftertreatment systems control emissions from FAME fuelled vehicles.

- Dimethyl ether (DME CH_3OCH_3 and larger-molecule derivatives of DME) – DME is a gas at room temperature but derivatives are liquid. It can be made from renewable sources and be used as a direct replacement for diesel, with some fuel system modifications. DME combustion produces low PM/PN but would still require a DPF to meet PN limits. The aftertreatment system would need to be upgraded to control the increased methane emissions.

- Methane (CH_4) – Can be used in diesel engines but needs a combustion initiator. Hence it can be used as a dual fuel with diesel, where diesel ignites first allowing the methane to combust. It can be used at > 90% substitution of diesel fuel leading to low PM/PN compared to diesel alone but would still require a DPF to meet PN limits. The aftertreatment system would need to be upgraded to control the increased methane emissions.

2.7 Unregulated Emissions Formation

Discussions have commenced recently regarding the next stage of European emissions legislation. Nominally “post Euro 6”, the next regulatory stage could be extremely complex. On October 24th 2018, the European Commission hosted a stakeholder event on the future of emission standards for the automotive industry[1], and in her introduction, Szychowska of the European Commission Directorate General for Internal Market, Industry, Entrepreneurship and SMEs (DG GROW), who are responsible for European exhaust emissions legislation, outlined several ambitions for the current and future regulations (Szychowska, 2018):

- There will be increased regulatory oversight of the automotive industry, including greater market surveillance. Compliance will be robustly enforced, with possible market withdrawals of type-approved vehicles and fines of up to €30,000 per vehicle of the offending type sold.

[1] https://ec.europa.eu/growth/content/stakeholder-event-preparing-future-european-emission-standards-light-and-heavy-duty-vehicles_en

- Greater transparency of the regulatory system to all stakeholders is required, including the general public. The RDE performance of vehicles is critical, and it must be demonstrated and proven to be reliable.

- Health studies data should support emissions requirements.

- Fleet CO_2 reductions must continue.

- Emissions control technologies must be good, and proven to be good, for the realistic life of a vehicle (10-15 years). This could mean 300,000 km, by which time current emission aftertreatment performance can deteriorate (e.g. He et al, 2019).

- Connected and automated driving must be used for lower emissions, as internal combustion engines are expected to play a significant role alongside electrification. Connect driving is where the vehicle is in communication with other systems, such as traffic lights, to operate in its most efficient manner.

- New pollutants will be in the spotlight: especially PN and NH_3 for spark ignition (petrol and gas) types.

An overview of several air pollutants and greenhouse gas species, potentially the subject of future European regulations (Martini, 2018), was provided by the Joint Research Centre of the EU. These are:

<23 nm particles (as with >23nm particles currently regulated in Europe, sub 23nm particles will be non-volatile): These particles have been shown to be present in exhaust emissions of both compression and spark-ignition engines, and their presence increases the measured PN compared with the current regulatory size range. By a process known as translocation, nanoparticles can be transported, penetrate and concentrate in living cells of critical regions of the body (Hankin, 2008).

Recent data (Dilara, 2018; Andersson, 2019) including that shown in Figure 2.1, compares >23 nm (PN_{23}) and >10 nm (PN_{10}) particle number emissions from many road transport sources, including gaseous fuelled vehicles (compressed/liquified natural gas, CNG and LNG; LPG) and L-category (L-CAT) vehicles such as motorcycles and mopeds, as well as conventional diesel and gasoline-fuelled applications. In the figure, the value at the foot of each column indicates the number of vehicles of that type studied, and the value in blue at the top of each column indicates the proportion of PN present below 23 nm. The early Euro 6 emissions limit for GDI vehicles of 6×10^{12} particles/km, and later 6×10^{11} particles/km emissions limit that harmonises the GDI limit with diesel, are overlaid on the chart as dotted and solid horizontal lines.

Vehicle categories with highest fractions of <23 nm solid PN (Solid Particle Number - SPN, specifically 10 nm to 23 nm) are gasoline direct injection (GDI) and port-fuel injection (PFI) and gas-fuelled applications. Vehicles such as PFI and some GDI emitting below the Euro 6 particle number limit when measuring particles >23 nm would shift to above the limit if particles between 10 and 23 nm were also included. This potentially drives a regulatory need for particle filters to be more widely adopted.

Particle filter-equipped engines show lower fractions of <23 nm PN than applications without particle filters fitted, especially when the measurement range is extended below 10 nm (Figure 2.2), supporting prior evidence (Andersson 2017) (Figure 2.3) that these particle filters are more efficient for collecting the <23 nm fraction than for the >23 nm fraction.

Particle filters, if fitted to all internal combustion engines will therefore deliver a benefit in the reduction of <23 nm PN, even if that size range remains unregulated. It is unlikely though, that particle filters will be fitted universally without a PN regulation addressing <23 nm particles.

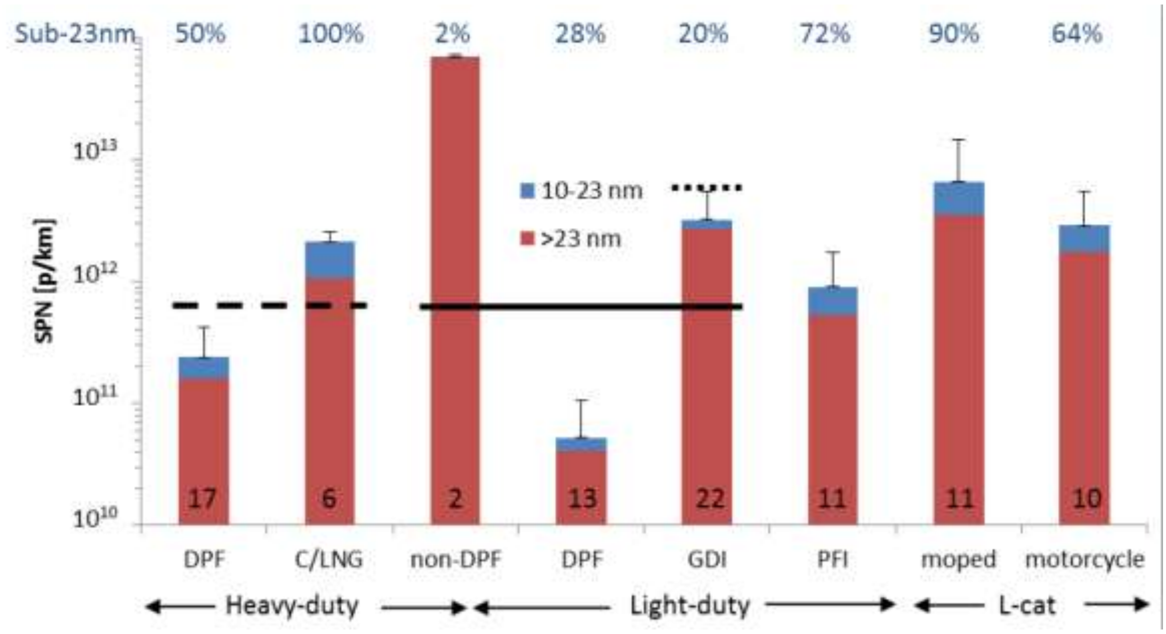

Figure 2.1. PN Emissions (particles/km) from various vehicle types, including 10-23 nm fractions (Dilara, 2018)Error! Bookmark not defined.

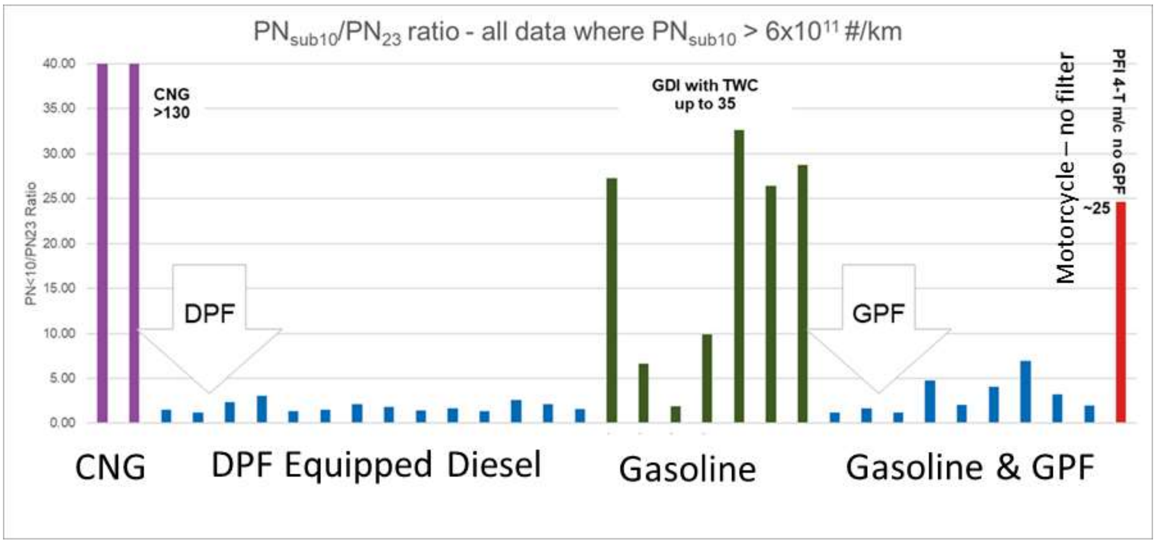

Figure 2.2. PN<10 to PN23 ratio – all data where PN<10 is greater than $6x10^{11}$#/km. Extending the PN measurement range to <10 nm highlights substantial increments in PN emissions from some compressed natural gas (CNG) and gasoline (GDI and PFI) spark-ignition vehicles (Andersson, 2019)

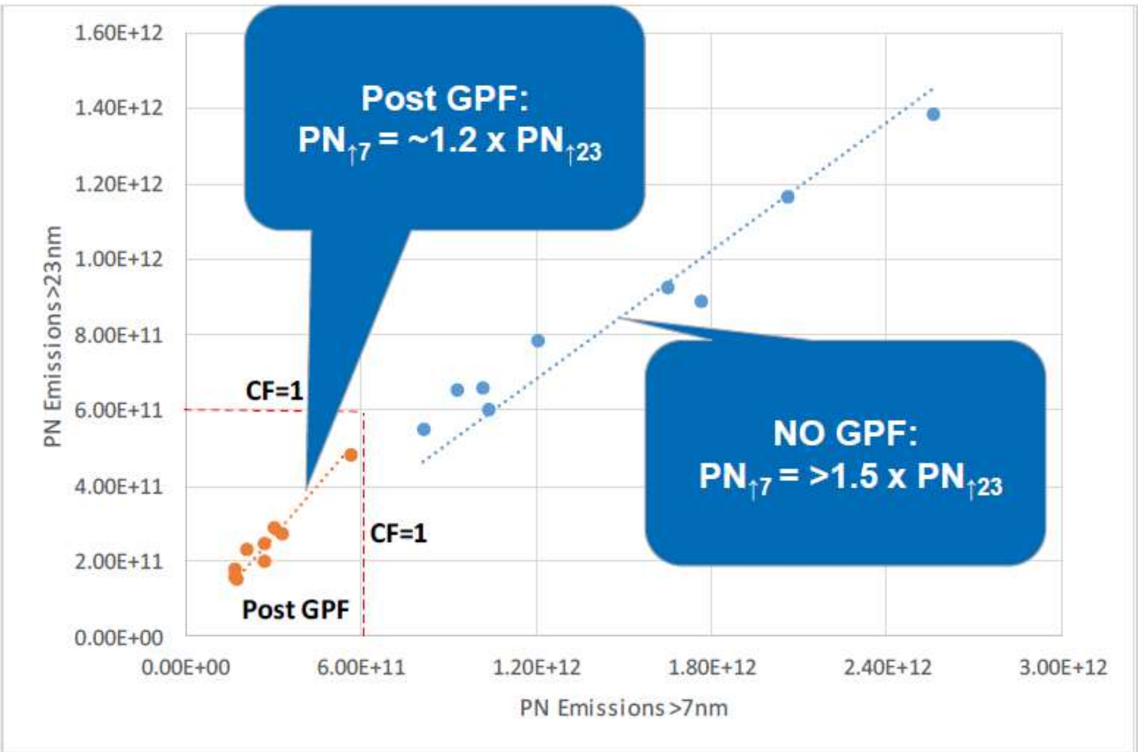

Figure 2.3. The increment in PN (particles/km) when measuring >7 nm rather than >23 nm is ~20% post-gasoline particle filter (GPF), but >50% without a particle filter (Andersson, 2017)

Ammonia (NH_3): ammonia is produced by catalytic emissions control systems designed to reduce NO_x. In spark ignition engines, NO is reduced to ammonia, with the reductant being hydrogen generated across three-way catalysts when combustion occurs with limited oxygen. In diesel engines equipped with lean-NO_x traps (LNT), periodic excess fuel combustion events are required for removal of stored NO_x, and during this process hydrogen is produced and NO is reduced to ammonia. Finally, selective catalytic reduction (SCR) used to control diesel NO_x generates ammonia from a stable source, often aqueous urea, that is injected into, and decomposes within, the exhaust system. The ammonia reacts with and reduces NO_x to nitrogen over a catalyst, but any unreacted ammonia can exit the tailpipe.

Ammonia is a known irritant for the eyes, nose and throat at concentrations around 25-50 ppm (Public Health England, 2015) and emissions from heavy duty engines are limited to 10 ppm[2], in raw exhaust averaged over emissions cycles, though it is not currently controlled on a mass emissions basis. It is also implicated in secondary aerosol production (Behira, 2010), contributing to the negative health effects of PM, and the formation of cloud condensation nuclei (Twomey, 1991) as well as effects on biodiversity. Ammonia is not currently controlled as a mass emission in any global engine emissions legislation.

Studies from light duty vehicles tested in the laboratory on the New European Drive Cycle (NEDC) have shown ammonia emissions in the range 4 mg/km to 70 mg/km (Suarez-Bertoa, 2016). Real-world emissions, measured by FTIR during RDE tests, were lower (Suarez-Bertoa, 2017) than this. Short-term emissions during the cold start period were significantly higher (45 to 134 mg/km) from gasoline vehicles, due to the influence of fuel rich operation for catalyst heating.

[2] European Commission, 2011 http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2011:167:0001:0168:en:PDF COMMISSION REGULATION (EU) No 582/2011

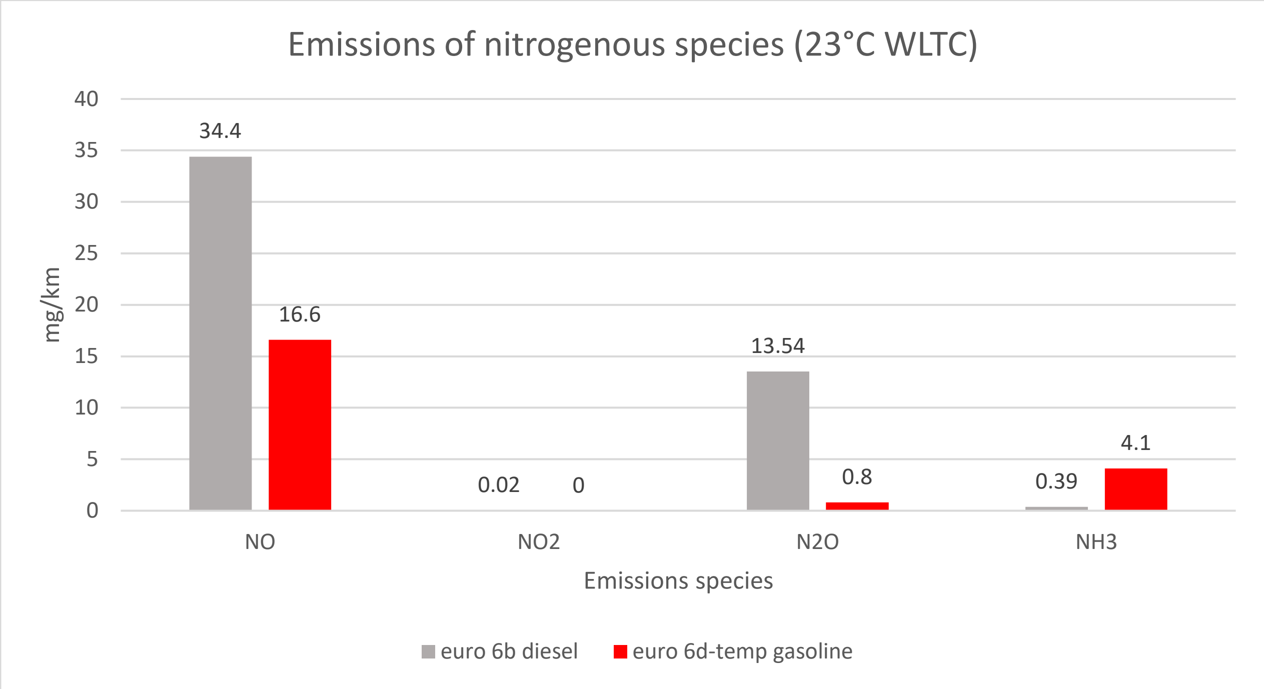

Figure 2.4. Nitrogenous Species' Emissions from a Euro 6b –light-duty diesel and a Euro 6d-temp gasoline (Ricardo Internal Data)

A more recent study on a Euro 6 GDI gasoline vehicle showed emissions, including cold start, which remains the predominant production mechanism, to be in the range 2 to 12 mg/km. Since many cold starts occur in the urban environment, controlling ammonia emissions is potentially an area of concern. Other work, which takes account of ambient measurements made outside of urban areas, has suggested fleet-average ammonia emissions from all petrol cars of 45 mg/km, with those from diesel cars being typically 10 times smaller (Marner et al., 2020).

Nitrous oxide (N_2O): is produced by similar mechanisms to the production of ammonia, though under less aggressively reducing conditions (Nevalainen, 2018). Hydrogen acts as the reductant in spark ignition and LNT-equipped diesel applications, but with SCR-equipped diesels, unreacted ammonia is partially oxidised to nitrous oxide across ammonia slip catalysts (ASC).

Nitrous oxide is not considered to affect air quality, instead its main impact is as a greenhouse gas, where its Global Warming Potential (GWP) on both 20 and 100 year bases is approximately 300 times (IPCC, 2018) that of CO_2. The US (~6.5 mg/km) and China (25 mg/km) already include nitrous oxide emissions limits in their local emissions legislation, though the US allows the N_2O to be traded-off as part of overall GHG targets. Since N_2O is produced by most catalytic aftertreatment systems, but is not an air pollutant, future European regulations may group this with the other exhaust greenhouse gases methane and carbon dioxide, as CO_2 equivalent (CO_2e). There is limited data on production of nitrous oxide from the most modern diesel vehicles, though JRC quotes ~4g/km CO_2e (Suarez-

Bertoa, 2016) supported by recent Ricardo data at ~4.2g/km CO_2e (~14mg/km N_2O, Figure 2.4) from early Euro 6 vehicles. Emissions from a similarly-aged GDI vehicle have been shown to be at ~10% of this level (Rodríguez, 2019).The CO_2e approach would enable catalytic emissions control systems to be employed without the extensive reformulation potentially required to meet aggressive limit values, and N_2O produced in moderation, while the overall GWP is managed by controlling fuel consumption and CO_2.

Formaldehyde (HCHO), acetaldehyde (CH_3CHO) and other aldehydes and ketones: formaldehyde and acetaldehyde are odorous compounds, potentially health relevant, and aldehydes and ketones contribute to low-level ozone formation (Hayman 2002). US regulations set a limit of 4 mg/mile for formaldehyde, but US regulations also control the total HCHO and other selected aldehydes, ketones and alcohols when regulating the composite emission known as non-methane organic gases (NMOG).

Aldehydes are partial oxidation species, so are produced before the engine is fully warmed up, where there are fuel rich regions and before the catalytic aftertreatment is fully effective. Thus, production of these species is minimal beyond cold start periods.

Fuel without alcohol present does lead to aldehyde production, with formaldehyde dominating, produced by oxidation following demethylation of longer chain HC. However, in fuels containing alcohols, methanol (CH_3OH) leads directly to formaldehyde emissions and ethanol (CH_3CH_2OH) to acetaldehyde under cold start conditions. The presence of fuel ethanol in European gasoline at 10% (E10), means acetaldehyde levels can approach those of formaldehyde. Recent data (Ricardo, 2018) from a Euro 6d-temp vehicle fuelled with E10 on the 23°C NEDC indicating around 0.6 mg/km formaldehyde (~14 mg/km cold start) and ~0.2mg/km acetaldehyde (~4 mg/km cold start). Formaldehyde emissions levels from modern light-duty diesels are similar to E10 gasoline, ~1 mg/km from a 23°C WLTC (Suarez-Bertoa, 2018)[i]. Decreasing the WLTC cold start temperature to -7°C led to a 6-fold increase in diesel HCHO emissions, and increases have also been seen with spark ignition engines

Limiting of aldehyde and ketone emissions, specifically under cold start, would help control low-level ozone, exhaust odour and the release of potentially health relevant components in urban areas, although the greatest benefits would be seen in regions of the UK with cooler ambient temperatures.

Isocyanic acid (HNCO): Though it is rarely studied, HNCO, like ammonia and nitrous oxide, is a reduced species potentially to be found in the exhaust of spark-ignition engine (Suarez-Bertoa, 2018) and from diesels equipped with LNT. It is also an intermediate in the decomposition of urea to ammonia, so is a potential emission from diesel engines with SCR.

HNCO is linked to atherosclerosis, cataracts, and rheumatoid arthritis. Emissions levels of HNCO (Suarez-Bertoa, 2016) have been observed as similar in magnitude to those observed for ammonia from spark ignition engines and diesels with NO_X storage catalysts (such as LNT), but lower for urea-SCR diesel engines, indicating the efficient consumption of HNCO during the SCR process. Given the similarity in production mechanism between NH_3 and HNCO for the high emitting engines, it seems likely that control of ammonia would also lead to reductions in HNCO, or vice-versa.

Polycyclic aromatic hydrocarbons (PAH) are highly unlikely to be subject to future legislation as a class of tailpipe pollutants. In order to substantially reduce PM and PN emissions, future regulatory activity is likely to force particle filters on all internal combustion engine applications. These particle filters capture and destroy soot and adsorbed organic molecules, and since benzo[a]pyrene and other PAH are predominantly associated with PM, they will be largely eliminated by these devices.

2.8 Summary

The implementation of successive emissions legislation has required the introduction of complex aftertreatment systems for all transport related internal combustion engines in order to significantly reduce the tailpipe emissions of the vehicle. For gasoline vehicles, a TWC controls emissions of THC, CO and NO_x whereas, a GPF controls particle number and particulate matter. For diesel vehicles, a DOC controls THC and CO, a DPF controls particle number and particulate matter, and SCR/LNT systems control NO_X emissions. In addition to regulated emissions, there are other emissions which are currently unregulated but are emitted at the tailpipe of the vehicles. Pollutants such as NH_3, N_2O, NO_2 and CH_4 are emitted and will be regulated moving towards Euro 7/VII. The current emissions legislation is Euro 6/VI for passenger cars and heavy duty applications and Euro 4 for motorcycles.

CO_2 legislation was introduced in 2012 for passenger cars and reductions in CO_2 emissions from heavy duty applications will be required from 2025. Passenger car CO_2 legislation will demand 95 g/km CO_2 at tailpipe whereas, heavy duty will initially require a 15% reduction compared to 2019/2020 levels. CO_2 legislation is in place to evolve towards 2030.

Moving forwards there are a range of alternative fuels which may become more prevalent due to their low CO_2 potential. However, any shift towards alternative fuels would need to be facilitated by the appropriate infrastructure, in addition to meeting current and future emissions legislation.

Driving style, situation and ambient conditions can significantly impact tailpipe emissions, particularly operations where the catalyst temperature is low, such cold start and low speed driving or low ambient temperatures.

3. Real World Driving Emissions

3.1 Exhaust emissions measurement techniques

Measurement of exhaust emissions have been an important tool to understand the contribution of internal combustion engines to air quality and have enabled interventions which have significantly reduced air quality emissions from transport engine sources in particular. The measurement methods have evolved from steady state engine dynamometer testing to more representative transient engine dynamometer testing and vehicle/chassis dynamometer testing in tightly controlled environments. Although not representative of real-world variability, such environments now offer a means by which repeatable and reliable measurements of emissions can be achieved, essential for technology development and direct comparison of technologies. Given the need for manufacturers to develop products for low emissions during real-world driving, new measurement approaches have been introduced (including remote sensing detection (RSD) and portable emissions measurements systems (PEMS)).

The following subsections introduce current typical test methods and measurement techniques. It is important to recognise that these different methods fulfil different purposes in the process of delivering low emission combustion engines, rather than considering one method to be arbitrarily superior to the others.

3.1.1 Laboratory Sampling

The preferred approach to sampling in emissions regulations is through the use of full-flow dilution (Fig. 3.1) in a constant volume sampling (CVS) system. When a fixed drive cycle is executed on a chassis dynamometer, the whole exhaust of the vehicle is diluted beyond the point of water condensation, with the total flow through the dilution tunnel held at a fixed, pre-determined level. Therefore, as the exhaust flow increases with engine load, the dilution ratio decreases. Throughout the test (or parts of a test) a bag is filled at constant flow with diluted exhaust from the tunnel and, in parallel, another ‘ambient’ bag is filled with dilution air sampled upstream of the point at which the exhaust is introduced into the tunnel. At the end of the test, the gases in the sample and ambient bags are subjected to analysis, and the concentration of pollutant species determined as the difference between the sample and ambient bags.

With the bag sampling at known temperature, it is then a simple step to convert the bag volumetric concentration to mass concentration, and then multiply by the total flow through the dilution tunnel, to get the total mass of pollutant per test, or bag. Subsequently dividing the mass per test result by either the cycle distance (km) for light-duty, or work done (kWh) for heavy-duty engines, gives the quantified level of each specific emission required by automotive regulations.

This full-flow dilution approach is mandatory for light-duty vehicle certification testing. However, due to the sheer size of full-flow dilution systems required for testing large engines with high volumetric exhaust flows, the determination of emissions using direct, raw exhaust sampling and partial flow dilution is permitted. In partial-flow dilution a fixed proportion of the exhaust flow is sampled into a small tunnel supplied with a constant flow of dilution air. This raw emissions process mimics the CVS approach, though has reduced accuracy, as the determination of the exhaust flow and control of the exhaust "split" fraction is complex.

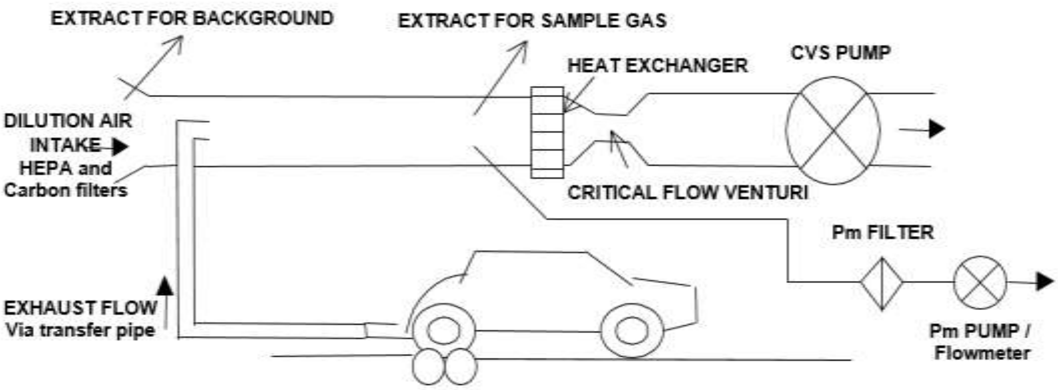

Figure 3.1. Full flow dilution system schematic.

3.1.2 Laboratory Analysis

From either diluted or raw exhaust streams, or from bagged gas, samples are passed to various analysers for emissions quantification. Several standard techniques are used for emissions determination (Nakamura, 2013):

3.1.2.1 Analysis of Hydrocarbons

A flame ionisation detector (FID) is used to detect hydrocarbons by determining their ionization by flame energy. A hydrogen flame produces ionized HC from the sample flow, and the number of ions produced is proportional to the number of carbon atoms in the sample. To reach the FID the HC must pass a heated filter at 192°C, limiting double-counting of HC associated with PM, by setting a volatility threshold.

3.1.2.2 Analysis of Methane and Non-Methane Hydrocarbons

Methane can be determined by gas chromatography, or by the use of a 'methane cutter' (where a FID is used after oxidising the exhaust gases across a catalyst heated at ~350°C, and all HC except methane are converted to CO_2). In the chromatographic approach, the sampled air is passed into a short analytical column and the HC separated. The first component to emerge from the column is methane, and this is then quantified (as described in 3.1.2.1) by FID.

3.1.2.3 Analysis of CO and CO_2

For vehicle emissions measurements, carbon monoxide and carbon dioxide are most frequently measured using a non-dispersive infra-red analyser (NDIR). This measures CO and CO_2 by quantifying the level of infra-red energy absorbed by a sample at specific wavelengths. Different wavelengths are used for CO and CO_2. CO and CO_2 can also be measured by non-dispersive ultra-violet analysis (NDUV).

3.1.2.4 Analysis of NO_x

Oxides of nitrogen (NO_x) are measured by chemiluminescence analysis (CLA). This technique specifically measures nitric oxide (NO), so in order to measure both NO and nitrogen dioxide (NO_2), the instrument contains a NO_x converter which reduces NO_2 in the sample to NO. The measurement principle quantifies NO through its reaction with electrically generated ozone which produces light that is detected by an appropriate sensor. The use of a CLA analyser without a NO_x converter enables the NO fraction of NO_x to be determined independently. Subtracting the NO fraction from total NO_x provides data on NO_2 levels.

NO and NO_2 can also be measured by NDUV.

3.1.2.5 Analysis of PM

For vehicle exhaust emissions, PM is analysed gravimetrically. A sample filter is conditioned in a controlled environment and then weighed. The filter is then placed in a specialised holder designed for uniform deposition, and a stream of diluted exhaust drawn through the filter for the duration of an emissions cycle or phase. The PM mass deposited depends on temperature, residence time, exhaust dilution ratio and flow of gas through the filter, and the sample flow through filter must be a constant proportion of flow through the dilution system. As described in Section 3.1.1, this proportionality of sampling ensures that the mass on the filter, determined following post-test conditioning and reweighing, can be easily scaled to determine the total emission from the vehicle and engine from the test or phase. Conversion of this mass to mg/km or mg/kWh is then straightforward. Due to sampling losses the sampled particles size is typically <2 µm.

3.1.2.6 Analysis of Particle Number

Particle number (PN) measurement is undertaken on a continuous basis, at constant flow, from the dilution system used to determine PM. The measurement system samples diluted exhaust aerosol from the dilution tunnel and then conditions the sample. The sample passes to a condensation nucleus counter (CNC) with a 23 nm lower size limit measurement threshold, a device which uses a super-saturated vapour to grow nanoparticles to diameters where they can be counted optically (McMurry, 2000). The non-volatile particle concentration is determined throughout the emissions test, averaged over the test and converted to particles per km or kWh.

3.1.2.7 Analysis of ammonia

For automotive emissions analysis, ammonia concentration, as average ppm in raw exhaust, is usually determined by infra-red spectroscopy, either broadband infra-red spectroscopy (for example, Fourier Transform Infra-red spectroscopy, FTIR), or by fixed or stepped frequency spectroscopy, such as using a quantum cascade laser (QCL) approach.

FTIR provides highly resolved absorption spectra of exhaust gas in the mid infra-red (IR) region of the spectrum on a second-by-second basis. Only compounds with dipoles can be measured, and the absorption strength depends on gas concentration.

QCL is used for the analysis of ammonia and other nitrogenous species (NO, NO_2, N_2O) and also measures absorbance in the mid-IR region of the electromagnetic spectrum. A dedicated laser source of specific wavelength is used for each component of interest, along with solid-state IR detectors targeting a narrow wavenumber range for the species of interest.

3.1.3 Portable Emission Measurement Systems

Portable emission measurement systems (PEMS) are devices that contain all the components required to determine mass emissions for NO_x, THC, CO and CO_2 plus total PN during real world driving, enabling emission measurements on real driving emissions tests (see Section 3.2). Most PEMS can also measure NO_2. Currently, only NO_x and PN are subject to PEMS regulatory control on light duty vehicles, with THC, and methane for compressed natural gas (CNG) applications, also required for heavy duty vehicles. However, both CO and CO_2 must be measured and reported. PEMS include components for sampling and sample transport, dilution (if required), analysis and sample release. PEMS are designed for fitment to mobile applications, including light and heavy-duty on-road vehicles and non-road mobile machinery, as well as to stationary emissions sources, such as power generators (gensets).

Guidance for the use of PEMS during on-road emissions testing of light-duty vehicles has been provided by the European Commission’s Joint Research Centre (Valverde-Morales, 2018), while the RDE regulations defining PEMS instruments, testing and measurements have been introduced in a series of RDE legislative packages. By June 2019, four RDE legislative packages had been released.

The PEMS used for regulatory emissions testing of on-road vehicles feature emissions analysers based upon the same principles as those used in the lab (Section 3.1.2). These are miniaturized systems, with economized functionality, working in less controlled environments, and so are not capable of the same degree of accuracy when quantifying emissions.

Real time concentration measurements must be combined with flow rate measurements to yield emission rates. The inaccuracy, introduced when time-aligning exhaust flow and real-time concentration measurements, is just one reason why on-road measurements using PEMS, which must also sample directly from the exhaust gases, are understood to be less accurate than lab-based measurement results based upon bagged sampling from diluted exhaust. Other reasons include:

- the use of lower accuracy (in comparison with lab-based approaches) pitot flow tubes to measure exhaust volumetric flow
- vibrations experienced by the test vehicle as it drives on the road which can impact analyser stability and function
- temperature changes during an on-road test which can impact analyser stability
- changes in altitude effecting air pressure differences experienced by the analysers and impacting their performance

Together these factors create a far more challenging environment for accurate quantification of emissions than in a test laboratory. The regulatory process for RDE recognises this measurement uncertainty by allowing proportionally higher emissions, when testing on the road, than the limits that must be met during testing in the laboratory.

This is achieved through the introduction of the conformity factor, CF (Equation 1). The target CF defines the not to exceed (NTE) level of emissions a vehicle can release during a compliant RDE cycle. The NTE is defined as the legislated limit value multiplied by the CF.

$$\text{RDE NTE (g/km)} = \text{CF} \times \text{Legislated Limit (g/km)} \qquad (1)$$

At Euro 6d final, the laboratory emissions test limit for a given species on the WLTC, for example: 80 mg/km of NO_X for a light-duty diesel passenger car, must also be achieved on the road. This implies a CF equal to 1. However, an “additive factor” allowance is included within the CF for the measurement uncertainty. This factor increases the allowable emissions threshold *as measured by the PEMS* to account for the uncertainty in the measurement and ensure the on-road target is not more stringent than the laboratory test limits. The additive factor is under regular review as measurement equipment is continually improved by instrument manufacturers.

For example (Table 3.1), the additive factor, *a,* for NO_x was set at 0.5 (RDE package #3)[3], following an introductory phase of CF = 2.1 (which also accounted for statistical uncertainty

[3] Commission Regulation (EU) 2017/1154 of 7 June 2017 amending Regulation (EU) 2017/1151 supplementing Regulation (EC) No 715/2007 of the European Parliament and of the Council on type-approval of motor vehicles with respect to emissions from light passenger and commercial vehicles (Euro 5 and Euro 6) and on access to vehicle repair and maintenance information, amending Directive 2007/46/EC of the European Parliament and of the Council, Commission Regulation (EC) No 692/2008 and Commission

in the procedure and subsequently removed). This indicates CF = 1.5, and that 120 mg/km measured on the road was considered to be equivalent to 80 mg measured in the lab within measurement uncertainty. This additive factor has subsequently been reduced to 0.43 (RDE package #4)[4], and therefore the CF to 1.43. Additive factor, *b,* for PN remains at 0.5.

As a consequence of applying the additive factor and using the Euro 6d final NO_X emissions limit as an example, 114.4 mg/km from an on-road RDE test is now considered to represent RDE NO_X emissions equivalent to 80 mg/km from a chassis dynamometer WLTC test.

Table 3.1: Conformity factors for NO_x and PN at Euro 6d final

CF	NO_x	PN
Euro 6d (final) By 01/01/2022 for all applications	1.0 + *a*	1.0 + *b*
Regulation date: *07/06/2017 (ii)*	1.0 + 0.5 = 1.5	1.0 + 0.5 = 1.5
Regulation date: *05/11/2018 (iii)*	1.0 + 0.43 = 1.43	1.0 + 0.5 = 1.5

3.2 Developments in Engine Emission Testing: Real Driving Emissions (RDE)

Laboratory testing of Euro 6 light vehicles follows the latest requirements defined in the Worldwide harmonised Light duty vehicles Test Procedure (WLTP). These procedures are rigorously defined, with an extensive framework of test automation inherent within facilities and equipment used for certification testing. The highly prescriptive procedures and measurements required, and the carefully controlled laboratory environment, ensure that test-to-test differences (repeatability) and lab-to-lab differences (reproducibility) are

Regulation (EU) No 1230/2012 and repealing Regulation (EC) No 692/2008 and Directive 2007/46/EC of the European Parliament and of the Council as regards real-driving emissions from light passenger and commercial vehicles (Euro 6). http://data.europa.eu/eli/reg/2017/1154/oj

[4] COMMISSION REGULATION (EU) 2018/1832 of 5 November 2018 amending Directive 2007/46/EC of the European Parliament and of the Council, Commission Regulation (EC) No 692/2008 and Commission Regulation (EU) 2017/1151 for the purpose of improving the emission type approval tests and procedures for light passenger and commercial vehicles, including those for in-service conformity and real-driving emissions and introducing devices for monitoring the consumption of fuel and electric energy. http://data.europa.eu/eli/reg/2018/1832/oj

minimized. The resultant emissions data from a regulatory test is therefore a highly reliable indicator of the emissions of that vehicle on the cycle under test.

WLTP was developed to address shortcomings in the previous regulations which were based upon the New European Drive Cycle (NEDC) and widely recognised to be unrepresentative of real-world driving. The test procedures were open to interpretation and lacked legal rigor. This led to significant differences between emissions recorded in the lab and those emitted by vehicles in use (Fontaras, 2016).

Euro 6 legislation based only on the WLTC (Euro 6c and earlier) showed some benefits but failed to reduce NO_x emissions in the real world to expected levels (O'Driscoll et al, 2018). This is likely to be somewhat a consequence of engine and ATS calibrations remaining focused on a singular drive cycle, albeit more representative than the NEDC. Hence the requirement to supplement the chassis dyno cycle with on-road assessment of emissions referred to as Real Driving Emissions (RDE). Emissions requirements that include RDE limits come in two stages of increasing stringency, Euro 6d-temp and Euro 6d, following a Euro 6c phase where on-road emissions were measured and results communicated to the regulators by the manufacturers, but levels were not limited. Indications are that introduction of RDE regulations leads to vehicles emitting air quality emission levels substantially below emission targets (Molden, 2019), however the performance of current vehicles is markedly different between manufacturers and models suggesting that some manufacturers have developed specifically for the 6d-temp stage, while others are opting to immediately adopt the technologies required for Euro 6d.

On-road regulatory RDE tests, performed while meeting all necessary validity criteria and when following the best practice guidance[5] provided by the European Commission's Joint Research Centre (JRC), are subject to a wide range of influences outside of a driver or manufacturer's ability to control. These influences on any given RDE test include individual driver characteristics, the unpredictable behaviour of other road users, and environmental conditions (temperature, pressure, humidity, weather conditions, wind strength and direction). Changing influences mean that repeated on-road drives result in variations of:

- Instantaneous and average speed
- Instantaneous and average acceleration
- Test cycle duration

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[5] COMMISSION NOTICE of 26.1.2017. Guidance on the evaluation of Auxiliary Emission Strategies and the presence of Defeat Devices with regard to the application of Regulation (EC) No 715/2007 on type approval of motor vehicles with respect to emissions from light passenger and commercial vehicles (Euro 5 and Euro 6) http://ec.europa.eu/transparency/regdoc/rep/3/2017/EN/C-2017-352-F1-EN-MAIN-PART-1.PDF

The specified criteria which define regulatory RDE routes create a very large multi-dimensional possibility space within which valid RDE cycles exist. New vehicles are expected to meet the emission requirements under any possible combination. The defining criteria, or boundary conditions, are shown in Table 3.2. Further limiting conditions based upon minimum positive acceleration, and the product of the maximum velocity and positive acceleration thresholds, aim to ensure the driver does not adopt unduly passive, or aggressive, driving styles respectively. Despite this, differences in CO_2 (7%) and NO_x (55%) have been observed from the same RDE route arising from drivers switching between 'normal' and 'severe, but legal' driving styles (Varella et al., 2019).

Table 3.2: RDE Boundary Conditions

Boundary Condition (Definition)	Moderate Conditions	Extended Conditions[#1]
Time of test / Roads used	Test on Working Days, using Paved roads	
Trip Duration	90-120 mins	
Trip Composition: Urban (<60 kph)	Target 34% of total distance, Limits 29-44%, Min 16km	
Trip Composition: Rural (60-90 kph)[#2]	Target 33%of total distance, Limits 23-43%, Min 16km	
Trip Composition: Motorway (>90 kph)[#2]	Target 33%of total distance, Limits 23-43%, Min 16km	
Urban stops (speed <1 kph)	6-30% of urban phase time, 300s max single stop	
Urban Average Speed	15 to 40 km/h	
Motorway Duration >100 kph[#2]	≥5 mins (does not need to be continuous)	
Maximum Speed km/h [Tolerance]	145 kph [>145, ≤160 kph for ≤3% of MW phase time]	
Vehicle Payload	Min: Test equipment, driver, and optional witness Max: ≤90% maximum vehicle payload	
Altitude	≤700m	≤1300m
Ambient Temperature [derogation for 5 years]	≥0°C [3°C], ≤30°C	≥-7°C [-2°C], ≤35°C
Gradient (positive altitude gain)	1200 m/100km	
Test start to end point altitude difference	Max 100m	
Pre-conditioning and soak	Driven at least 30 mins, soak for 6-56 hrs, moderate or extended conditions	
Cold Start (<70°C coolant/First 300s)	Average 15-40 kph, max 60 kph, max stop 90s, initial idle <30s *Emissions analysed as per rest of test. Some tests carried out with hot start*	

[#1] Emissions at "extended" conditions are divided by 1.6 *[#2] Motorway >80 kph and lower speed demands for speed limited LCV's*

The requirements for validity of an on-road RDE test are well-defined and can be verified. However, due to the multiplicity of possible RDE routes, and uncontrollable influences experienced every time the same RDE test is driven, even by the same driver, achieving the same mass/km emission value from two different RDE tests is far more difficult than when driving the same speed, time and gradient in the laboratory. Even in the best case, with the same driver on the same route, repeatability would not be expected to match tests in the lab, and the actual emissions value determined would vary depending on the ambient temperature, driving dynamics and other factors. Discussion over the effect of different factors is in Section 3.3. Such factors lead to significant challenges when testing any vehicle on an RDE test with the hope of determining a single, representative, emissions value, and present a challenge for any emissions factors attempting to account for these emissions.

Works to improve the representativeness of emissions factors is producing improved alternatives to the EU standard emission vehicle calculator (COPERT), including those for NO_X emissions (Bishop et al 2019).

To avoid a manufacturer certifying a vehicle on an “easy” RDE cycle, and not creating a fully robust emissions control solution, the European legal process requires that extra documentation is supplied describing the manner in which a vehicle’s software and hardware technically addresses emissions control under all possible valid RDE tests. This includes a signed commitment that all possible valid RDE tests will be passed by the vehicle in question (European Commission, 2017). The in-service compliance (ISC) requirements of the RDE and WLTP regulations, as defined in RDE package #4 (European Commission, 2018) mean that the certified vehicle is subject to emissions checks by regulatory authorities, independent test organisations and NGOs. These tests can evaluate emissions under all conceivable conditions using on-road measurements, chassis dynamometer tests and also independent approaches such as road-side remote sensing techniques. Failure to meet the emissions limits either within the range of valid RDE tests, or even just outside, will lead to significant scrutiny by the regulators. Potential penalties for deliberate contravention of the emissions control system rules, leading to poor performance in the real-world when compared to the certification tests, include substantial fines and the withdrawal of permission to sell the vehicle in Europe.

The combination of RDE requirements and new in-service compliance testing will ensure robust emissions control on all light-duty vehicles, in order to comply with all possible valid RDE cycles. It should be noted that both the WLTC and RDE cycle were developed from an extensive analysis of a large database of European vehicle trips. With both WLTC and RDE representing all driving up to the 95th percentile of European trips (Tutuianu et al., 2015), the majority of European real-world driving is now representatively covered by either certification testing, in-use compliance testing, or both.

3.3 Remote Sensing of Emissions

Vehicle emission remote sensing provides a way in which to measure the real-world emissions from vehicle fleets. The technique relies on ultraviolet and infrared spectroscopy to measure individual vehicle plumes from passing vehicles. The technique can provide measurements of pollutants such as CO, NO, NO_2 (hence NO_x), NH_3 and hydrocarbons as a ratio to CO_2. Emissions from vehicle emission remote sensing are commonly expressed as fuel-based factors (e.g. grams of NO_x per kg of fuel) from which absolute emissions (in g km^{-1}) can be estimated. Measurements made using remote sensing are complementary to PEMS but have several unique benefits. First, the measurement of a vehicle’s exhaust is made without any contact or interference with the vehicle being measured. Second, the approach can measure 1000s of vehicles in a day from the whole vehicle fleet.

Remote sensing typically only provides a short duration measurement of a vehicle exhaust (around 0.5 seconds) at a sampling location that tends to favour operation at high engine load. Short durations are generally considered insufficient to derive meaningful information, but with tens or hundreds of thousands of measurements over a wide range of conditions, it is possible to derive detailed information on the emissions from vehicles e.g. by Euro class or vehicle make and model. Indeed, one of the benefits of remote sensing is the comparative assessment of vehicle emissions (e.g. by vehicle manufacturer or model) because on average vehicles are typically measured under the same conditions (e.g. traffic and ambient conditions), which is difficult to achieve using PEMS.

3.4 Factors Influencing Emissions

The previous section has highlighted that a number of factors affect emissions, some of which are bounded within the RDE test procedures. Within RDE boundary conditions, and outside, there are a wide range of possible operating conditions experienced and therefore scope for substantial variation in emission rates from a single vehicle as it undergoes similar journeys. This section explores factors which have been identified as important.

3.4.1 Effect of road congestion on the emissions

Congestion on the road leads to increased variations in vehicle speed, often requiring more braking and acceleration events. CO, THC and NO_x emissions typically increase with increasing road congestion (e.g. Lairenlakpam et al., 2018). Tests from Leeds (Khalfan et al, 2015) on Euro IV spark ignition vehicles have shown significant dependence of trip emissions on traffic congestion, with emissions increases similar to the increases in fuel consumption (~50%) and these arising predominantly from increased catalyst warmup duration and a more transient speed during the journey. Increased average velocity of the vehicle from 8 km h^{-1} to 24 km h^{-1} (indicative of less congestion) correlated with reductions in ammonia, and N_2O.

The increased utilisation of, and capability of route guidance systems has the potential to contribute to reductions in exhaust emissions through traffic management. Congestion based route guidance strategies are shown in modelling studies to positively contribute to emission reduction by reducing congestion and reducing the amount of acceleration events on a journey (Cui et al, 2019).

3.4.2 Effect of Stop-Start on Emissions

There are a limited number of published studies quantifying the impact of idling / stop-start systems on air quality emissions. Of those that do exist, most predate the latest technologies. Clear benefits have been recognised with regards fuel consumption and so emissions of CO_2, the primary greenhouse gas. However, tailpipe emissions are dependent on the impact of exhaust gas temperature on the performance of the aftertreatment system. The emission concentrations in the exhaust are dependent on the quality of the combustion, which is often sub-optimal under start or restart conditions. The amount of emissions arising from restart events are therefore highly vehicle and calibration dependent making the optimum stop duration for minimum emissions also highly vehicle dependent.

Increases in emissions due to cooling of aftertreatment systems are rarely reported although some suggest relatively long stop durations, of the order of minutes to tens of minutes are needed (e.g. Taylor, 2003, Gaines et al, 2012) to observe such a detriment. Anti-idling campaigns at four US schools (Ryan et al, 2013) reported significant improvements in air quality at the school with the most traffic during pick-up and drop-off times. The other schools tended to show statistically insignificant changes in air quality which appeared to be primarily dependent on the wider environment.

Further work is needed to quantify the effect of stop period on restart emissions for modern vehicles. The limited data available does not cover well the complex interactions, which vary between different system designs affected by aftertreatment system thermal inertia and idling strategies. The impact of aftertreatment condition resulting from for example recent emissions or ammonia dosing will significantly influence the optimum decision regarding idling or stopping. It is therefore not possible (due to current levels of evidence) to recommend switch-off or idle for all vehicles or all circumstances.

Air quality emissions from ICE hybrid vehicles have been shown to not always correlate well with CO_2 emission reductions (Huang et al, 2019; Yang et al, 2019), often attributed to the increased frequency of stop-start events. The effects are again highly dependent on vehicle design and control strategy employed.

3.4.3 Effect of road design on emissions

NO_X emissions are higher on urban roads compared to rural roads and motorways for Euro 6 diesel passenger car vehicles (Cuelenaere, 2016) with congestion having a major influence. Aftertreatment systems need to be operated at certain temperatures for optimised emissions control. During rural road driving, there is a potential for a drop in temperature of aftertreatment systems such as SCR and it can take a period of time to regain the temperature. Hence, when the exhaust temperature drops, the NO_x emission can increase during this period. Increased spatial density of journey 'start locations' in densely populated areas, when considered with the increased emissions arising from cold starts (Section 2.5) are expected to further contribute to higher emissions on urban roads.

Exhaust emissions from vehicles are closely linked to the magnitude and frequency of acceleration events during a drive. Traffic calming measures such as speed bumps, speed humps and chicanes often affect the number of acceleration and deceleration events as well as the average speed. The impact of traffic calming measures on air quality emissions has been studied using combinations of vehicle emissions models (e.g. the Environmental Protection Agency's MOVES model, or bespoke models) alongside driver models with varying degrees of robustness. There is general consensus that CO, HC and PM_{10} continue to be higher when traffic calming measures are present (e.g. Jazcilevich et al, 2015; Ghafghazi and Hatzopoulou, 2015). NO_x emission predictions are more varied and are primarily dependent on the specific engine and exhaust technologies and calibration (Jazcilevich et al, 2015; Ghafghazi and Hatzopoulou, 2015). The impact on NO_x emissions has been identified as coming from short duration spikes in NO_X emissions, arising during transient operation, contributing up to 82% of the NO_x emissions during the trip (Mera et al, 2019). These have been linked with traffic controls such as speed bumps and traffic lights, the NO_X emission spikes arising from the quality of air-fuel ratio control during the transients (Duckhouse et al, 2019). Such emission spikes are therefore highly vehicle dependent, thereby explaining the diversity in NO_x results in the literature.

3.4.4 Effect of aggressive and soft driving on emissions

Driving style continues to have a major impact on particle number emissions from gasoline engines [for example ref 39]. The cause is somewhat similar to the impact of congestion and traffic calming measures on moving more of the vehicle operation into conditions of higher acceleration and therefore higher fuelling rates. Particularly in small engine diesel vehicles, where catalytic emissions control is sub-optimal, NO_x emissions increase substantially with greater driving dynamicity. In both cases, the amount of variation with driving style is not well correlated with engine size or vehicle power to weight ratio highlighting the dependence on vehicle calibration rather than fundamental process limits. CO and HC emissions continue to be high relative to normal operation in the first few minutes following cold starts when the ability of the TWC to eliminate high engine out emissions is limited.

A study from the German Technical Service, TUV, showed similar trends in light duty diesel engines with a ~6% increase in RDE fuel consumption and a ~26% increase in RDE NO_x between soft and aggressive driving styles on a Euro 6 light-duty diesel vehicle. The same study showed the impact of soft versus aggressive driving severity on fuel economy and NO_X emissions to be substantially greater in the urban RDE section than in the complete RDE cycle where the frequency of acceleration events increases the relative impact of driving style. In this example, urban NO_x emissions increased by ~76% due to driving style in comparison to ~26% in the mixed RDE.

Currently such driving styles are predominantly determined by driver behaviour. Increasing amounts of automation may positively influence such trends not only through reduction of the amount of aggressive driving, but also through an increased ability to predict and adapt to developing situations.

3.4.5 Effect of ambient temperature on emissions

The environment surrounding a vehicle influences the demand on the vehicle's energy system and the performance of sub-systems on the vehicle such as emissions aftertreatment systems. Wind direction, precipitation, humidity and ambient temperature will, for example, influence the force required to propel the vehicle. Ambient temperature and humidity will affect the oxygen available to combustion engines thereby affecting their efficiency and emission factors. Ambient temperatures affect the vehicle's ability to manage powertrain temperatures and the cooling/heating demands of the occupants. The dependence of air quality emissions from transport on environmental conditions will remain difficult to predict and cannot be controlled on RDE tests, and should be acknowledged as a remaining source of variation in RDE testing results. Some of these factors can be relatively easily accounted for, however, the interaction between ambient temperature and emission aftertreatment performance is complex.

Euro 6 gasoline and diesel cars were tested with different ambient initial temperatures, ranging from 5°C to 15°C (Lairenlakpam, 2017) with results presented as emissions per unit mass of CO_2 normalised to the 5ºC total journey result. The results in general show that, low outdoor temperatures influenced the vehicle operation in an RDE test in compression ignition (CI) vehicles (Figure 3.2). In the spark ignition (SI) vehicle studied, outdoor temperature influences the vehicle operation but not as clearly as in CI vehicle.

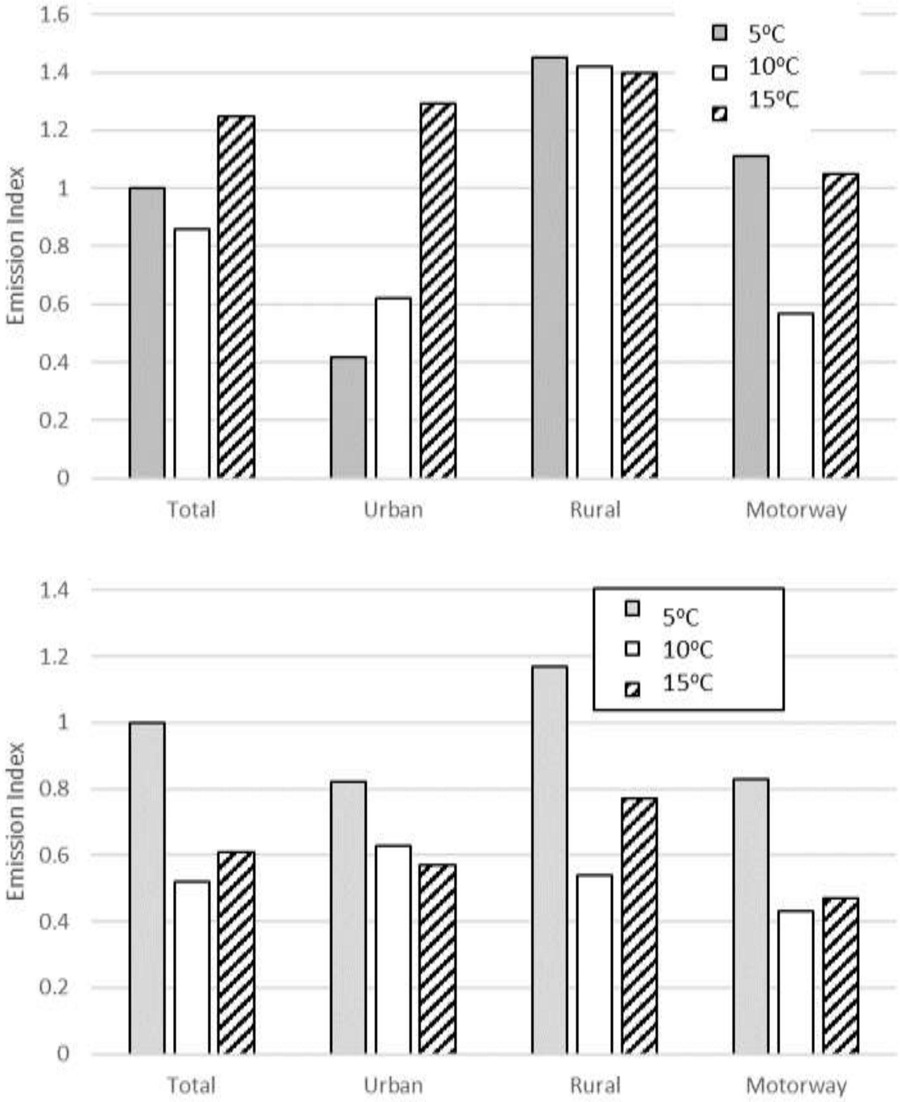

Figure 3.2. Relative NO_X emissions (per unit CO_2) for SI (top) and DI (bottom) Euro 6 vehicles

Ambient temperature is known to affect the emissions of pollutants from vehicles, separate from any cold-start effect i.e. there is an influence of ambient temperature on vehicle emissions from hot engines. These effects have recently been taken into account in emission factor methods such as HBEFA (Handbook Emission Factors for Road Transport) for diesel passenger cars (Keller et al., 2017), at least for NO_x emissions from diesel passenger cars. Some evidence of a temperature dependence of NO_x emissions was also revealed in testing commissioned by the Department for Transport of 19 Euro 5 and 19 Europe 6 diesel passenger cars (Department for Transport, 2016). NO_X emissions for a Euro 6b vehicle have been shown to increase significantly with cold ambient temperatures[iv].

More recently, comprehensive analysis of UK vehicle emission remote sensing data also showed a clear relationship between emissions of NO_x and ambient temperature (Grange et al., 2019). Figure 3.3 shows the relationship between ambient temperature and NO_x emission for light duty diesel vehicles measured between 2017 and 2018 expressed per kg of fuel giving a better indication of emission performance independent of energy demand. These data consist of nearly 300,000 diesel and petrol vehicles measured over a temperature range from 0.5 to 25°C. Figure 3.6 shows that light duty diesel vehicles have a clear and strong relationship with temperature, which is absent from gasoline vehicles.

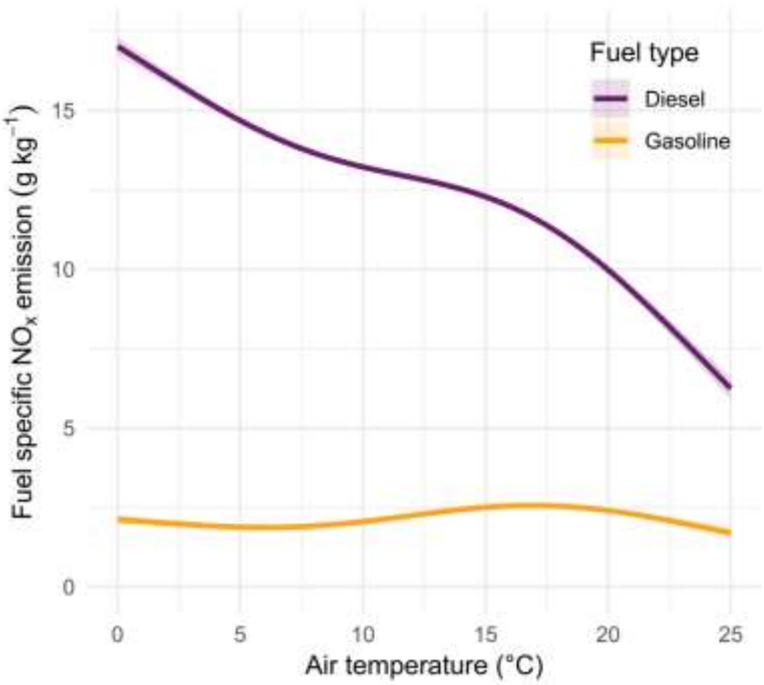

Figure 3.3. Generalized additive models (GAM) of NO_x emissions based on air temperature for light-duty diesel and gasoline-powered vehicles. The shaded zones represent the models' standard error for the prediction Source: Grange et al, 2019.

Emission increases at low ambient temperatures have embedded within them effects of increased warmup duration under cold start and impacts to system performance after warmup. Low ambient temperature demands excess fuelling in both gasoline (SI) and diesel (CI) for the start to reduce the time taken to achieve high aftertreatment performance. Also, it takes more time for the aftertreatment system to warm up completely and operates with excess fuel during this period which increases the amount of fuel rich regions in the cylinder and consequently CO, HC and PM emissions. Aftertreatment system temperature and excess fuel is therefore the main cause of increased emission.

Data from a gasoline engine (Ramadhas, 2017) has shown that CO, THC & NO_x from gasoline engines increase during cold start in colder ambient temperatures compared to normal ambient temperature. Instantaneous CO and HC emissions during the first 60 seconds were observed to increase by a factor of more than 3 and 4 respectively when ambient temperature dropped from 25°C to 10°C. No significant differences were observed at higher temperatures, as the temperature dropped from 45°C to 25°C, showing the highly non-linear relationship between ambient temperature and cold start emissions. Legislated PN emission (>23 nm) were found to be elevated during cold ambient start compared to normal ambient start conditions[v], with the main contribution to PN being carbon particles from incomplete combustion. With an ambient temperature of 10°C, >90% of cold start PN emissions were in the legislated size range, reducing to ~40% and <30% at temperatures of 25°C and 45°C respectively.

Even with these increases, the inclusion of emissions from the cold start event is reported in this study to have a small impact over the entire RDE test (ranging from 1.9% for CO to 0.2% for PN) but over the urban phase the impact was significantly higher (ranging from almost 8% for CO to just over 5% for PN) . The urban phase, which is at the start of the test, contains an increased number of start and stops within a more transient drive cycle. This results in difficulties in controlling the air fuel ratio around stoichiometry reducing three way catalyst effectiveness and increasing CO emissions. PN emissions are higher due to liquid fuel impingement on cold surfaces of the combustion chamber and piston. The magnitude of the effect of these phenomena on emissions are highly dependent on injection and combustion system design which contributes to significant variation in emissions from different vehicle models during cold start.

3.5 Emission Trends from Vehicles

On-road real driving emissions have been used to determine the impact of technology levels and automotive catalysts on tailpipe emissions. Figure 3.4 shows data for an emissions study performed on a specific bus route in Brighton, using a PEMS (Brighton Buses Study). A range of bus technologies were selected to determine the tailpipe emissions for NO_x from Euro IV to Euro VI solutions. The chart shows that for a demanding 18 km urban bus route the total NO_x mass emissions reduce with advancing levels of emissions legislation, with the Euro VI vehicle giving a 98% reduction compared to the Euro IV bus.

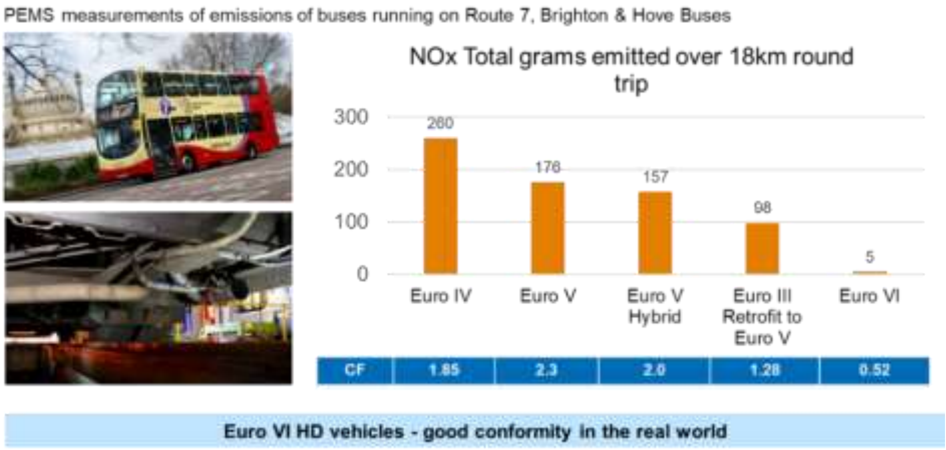

Figure 3.4. PEMS data for bus operation

Studies by O'Driscoll et al. (2018) demonstrated that pre-RDE diesel engines during real world driving were producing emission levels of on average 5.5 times higher than the type approval limit, and up to 31 times higher, with only five percent of the tested vehicles achieving the type approval limit during urban real world driving. Gasoline vehicles of similar generation performed much closer to the type approval limit. This highlights the importance of the RDE test procedures in delivering air quality emission reductions in urban areas, which is starting to be observed in test data.

An overview of recent remote sensing measurements for NO_x is shown in Figure 3.5 for major categories of vehicle. The figure shows that for all classes of diesel vehicles, the move to Euro 6/VI led to a substantial reduction in NO_X. This reduction is greatest for large diesel vehicles and least for smaller vehicles such as passenger cars. It also reveals that older petrol passenger cars are also relatively high emitters of NO_x, which in part will be due to vehicle deterioration effects. Figure 3.9 also reveals that emissions of NO_x from motorcycles have been invariant from pre-Euro through to Euro 3 and that the emissions are relatively high on a fuel-specific basis.

Figure 3.5 Summary overview of emissions of NO_x from different classes of vehicle, split by Euro classification. These data represent summaries of remote sensing data collected between 2017 and 2018 by Ricardo Energy & Environment. The uncertainty intervals relate to the 95% confidence interval in the mean and the numbers at the top of each bar show the number of valid measurements.

While Figure 3.5 showed that NO_x emissions from diesel cars decreased in going from Euro 5 to Euro 6, Euro 6 comprises different stages with potentially very different NO_x performance characteristics. Considering diesel passenger cars specifically, Figure 3.6 shows how emissions of NO_x have changed by the year of manufacture of the vehicle. It is clear from this figure that the introduction of Euro 6 vehicles (from September 2014) led to a considerable reduction in NO_x. However, the plot also reveals that there is markedly different performance between vehicles equipped with LNT and SCR NO_x reduction technologies, with SCR being much more effective under these conditions. This is not always the case though, LNT devices can be more effective than SCR when exhaust temperatures are low, and this effect can be beneficial in demanding urban conditions (Andersson, 2018). Figure 3.7 shows NO_x emissions over the ~9km distance of the low average speed (<15kph), highly dynamic Transport for London Urban Inter-peak cycle for two SCR-equipped vehicles (V1, Euro 6b and V2, Euro 6c) and an LNT-equipped vehicle (V3, Euro 6b). In this cycle the LNT showed lowest emissions for the first 6.4km (>30 minutes) of the test. This suggests that for short city trips, where exhaust warm-up is slow, diesel vehicles with ATS that include LNT might be favourable.

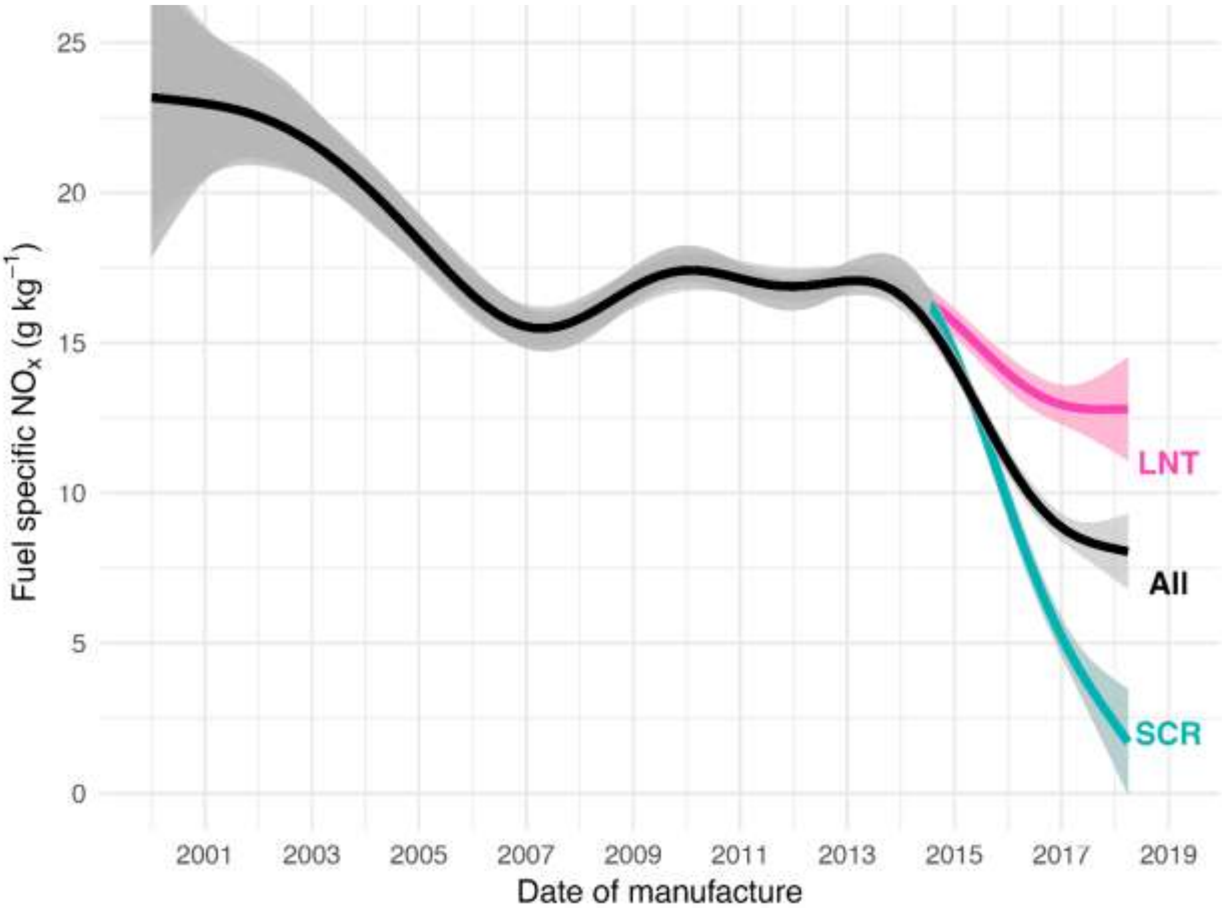

Figure 3.6 Emissions of NO_x (g/kg of fuel) as a function of vehicle manufacture date for diesel passenger cars. Where possible, Euro 6 vehicles have been split by their main emissions control technology (LNT or SCR). The black line shows the emissions for all vehicles regardless of the aftertreatment technology used.

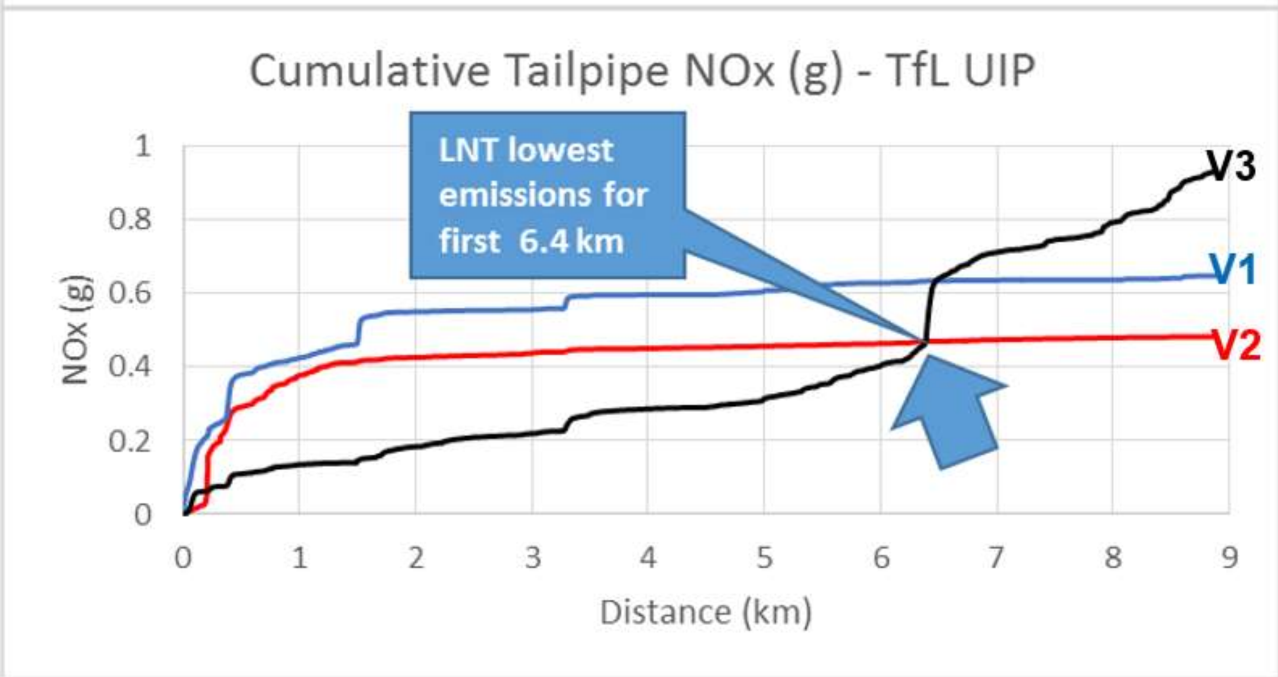

Figure 3.7: Emissions of NO_x from 3 Euro 6 diesel vehicles from the TfL Urban inter-peak cycle

Figure 3.8 shows data from roadside measurements for diesel and gasoline passenger cars demonstrating the reduction of emissions with improving Euro standards. For the diesel applications there is a shift towards lower NO_x from Euro 2 through to Euro 6, predominantly between Euro 2 and 3, and Euro 5 and 6. For gasoline applications, the main shift in NO_x emissions occurred from Euro 2 to Euro 4 - from Euro 4 onwards the average tailpipe NO_x emissions remain low, with a reduction in the number of cars with high emissions (Davison et al., 2021). There is large variability of emissions within classes, with some Euro 6 diesels emitting less NO_x than some Euro 6 gasoline vehicles.

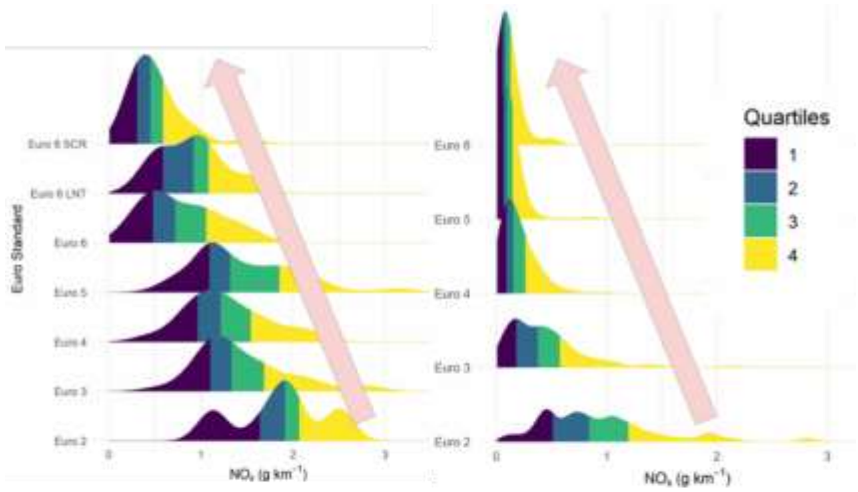

Figure 3.8. Roadside measurement for a range of passenger cars from Euro 2 to Euro 6 (LHS Diesel passenger cars and RHS gasoline passenger cars – based on data used in Davison et al., 2021)

Recent measurements from remote sensing also show the extent to which the total NO_x and amount of NO_2 varies by vehicles of different vehicle models (Carslaw et al., 2019). Figure 3.9 shows that there is at least a factor of 10 difference in fuel-based emissions of NO_x for Euro 6 passenger cars. The figure also reveals the differential performance of NO_2 emissions, with some manufacturers achieving both low total NO_x and low proportion of NO_2.

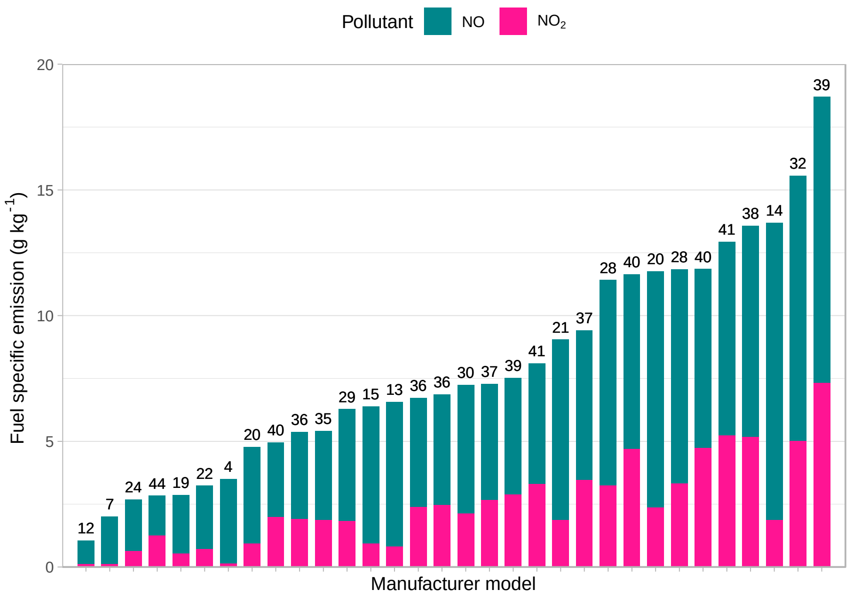

Figure 3.9 Emissions of NO_2 and NO (g per kg fuel) for Euro 6 diesel passenger cars. The results are shown by individual (anonymised) manufacturer family grouping and ranked by their total emission of NO_x. The numbers at the top of each column show

the percentage of within each group NO_x that is NO_2. The NO emissions are calculated as NO_2-equivalent.

Impacts of ageing of exhaust aftertreatment components such as oxidation catalysts has recently been considered by linking individual vehicle emission measurements to vehicle mileage based on the most recent MOT for passenger cars more than three years old. This work shows that for diesel passenger cars at least, the amount of NO_2 emitted by vehicles both in absolute terms and as a ratio to NO_x decreases as the mileage increases (Carslaw et al., 2019).

Remote sensing data can also be used to quantify NH_3 emissions by fuel type, vehicle type and vehicle age (Farren et al., 2020). The data show that the dominant contribution to NH_3 from road vehicles is petrol passenger cars. The emissions of NH_3 as a function of date of manufacture is shown in Figure 3.10 and shows that emissions of NH_3 are lower for newer vehicles, which is likely to reflect both improvements in TWC design and also deterioration of catalysts as they age. While petrol vehicles dominate the total emission of NH_3, there is also evidence of increased NH_3 emissions from Euro V/VI buses and HGVs fitted with SCR systems.

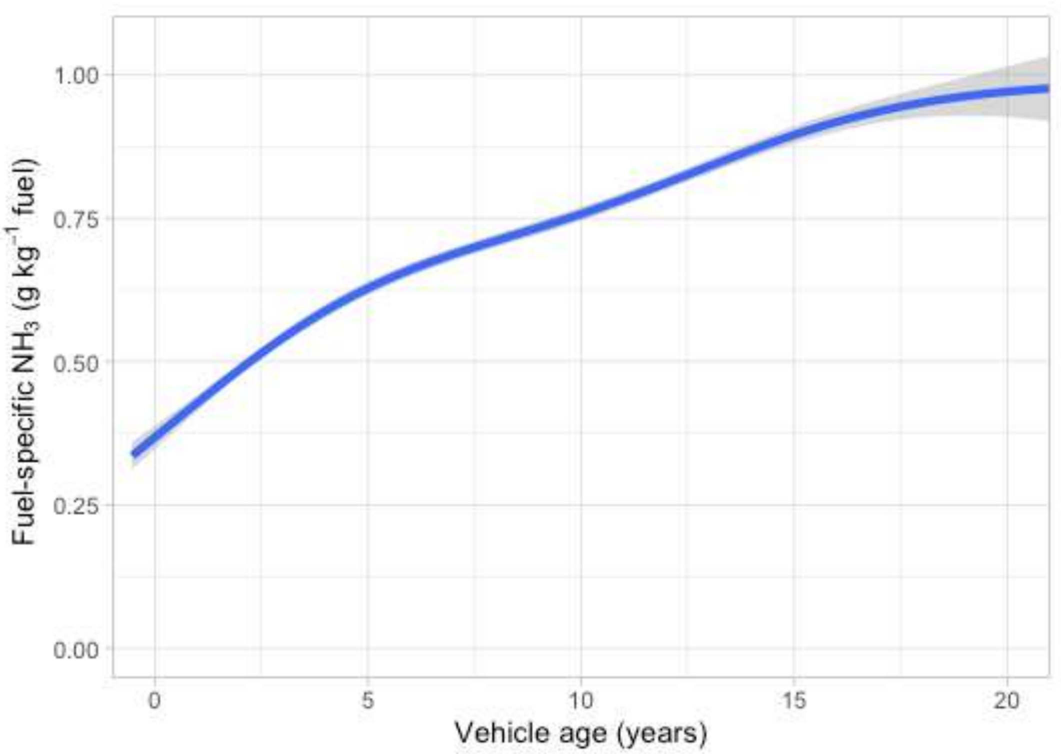

Figure 3.10. Fuel-specific emissions of NH_3 from petrol passenger cars based on 125,000 measurements made from 2017 and 2019.

The trends shown in Figure 3.10 are consistent with similar measurements made in the USA (Bishop and Stedman, 2015), where analysis shows that decreases in NH_3 emissions from vehicles have been less than that for total NO_x.

3.6 Summary

Introduction of more representative test procedures, in particular the Real Driving Emission (RDE) tests is increasing the effectiveness of legislation in driving down real-world emissions.

The intrinsic variability in environmental conditions, traffic conditions and driver behaviour mean that there will remain disparities between any two individual RDE tests. Traffic congestion increases the lower temperature operation of the aftertreatment systems as well as number of accelerations and stop-start events typically resulting in higher emission levels. The effect of longer duration stop-starts are less clear, requiring further evidence for modern vehicles. Reduced ambient temperatures impact on the engine operation and aftertreatment system performance during cold start. Traffic calming measures which result in increasing number and severity of acceleration and deceleration events (e.g. speed bumps) tend to increase emissions.

To achieve NTE limits on RDE tests, manufacturers are designing and calibrating vehicles which produce emission rates substantially lower than legislative limits on most journeys when new to ensure that real driving emissions remain within limits later in vehicle life. The high variability between vehicles is clearly demonstrated through remote sensing data, some of which are already substantially below legislative limits.

Emissions from successive Euro standard vehicles have on the whole been reducing, although limited improvements in diesel engine NO_{x+}

emissions are apparent between Euro 3 and Euro 5 standards. There are differences in performance between LNT and SCR equipped early Euro 6 vehicles. Which technology performs best is dependent on conditions and situation. Early Euro 6 vehicles, on the whole, show a marked reduction in real world NO_X emissions compared to the introduction of Euro 5.

4 MODELLING VEHICLE EXHAUST EMISSIONS

There are different approaches for modelling vehicle exhaust emissions. An approach is generally chosen according to the spatial scale being modelled, e.g. whether the aim is to model emissions on a national scale or on an individual road, and according to the vehicle activity data which are available. Highly detailed vehicle emission models can be used to simulate instantaneous emissions for an individual vehicle under a specific set of conditions, e.g. on a second-by-second basis during acceleration, deceleration and idling. The simulations are in terms of a physical model according to engine speed and torque and accounting for the behaviour of any exhaust aftertreatment systems. Conceptually simpler approaches are often used to model national fleet emissions over the course of a longer time period using drive-cycle or speed-average emission factors (EF) and traffic data. These simpler models are also routinely used to calculate local-scale emissions, but are not intended for this purpose.

There are inherent difficulties and complexities in calculating road traffic emissions to a high level of accuracy because of the complex sensitivities to operating conditions (e.g. temperature in the engine, exhaust and aftertreatment system, driving style etc) and the range of mitigation technologies at different stages of development in different vehicles. Depending on the particular purpose of an emissions model or inventory, taking account in detail of all these complexities is not necessarily appropriate for calculating emissions as uncertainties will always remain

The following sections in this chapter summarise the approaches used for modelling vehicle exhaust emissions. These essentially combine vehicle-, technology- and movement-specific emission factors expressed in grammes emitted per distance or time travelled with an appropriate activity data (e.g. distance travelled at a particular speed or set of driving conditions). A further step is required to derive modelled concentrations of pollutants in ambient air from the exhaust emissions accounting for atmospheric dispersion conditions and chemical processes. This step is briefly described in Section 4.5.

4.1 Approaches Used in National Emission Inventories

4.1.1 Average Speed Approach Provided in the EMEP/EEA Emissions Inventory Guidebook

The EMEP/EEA Emissions Inventory Guidebook (EEA, 2019) provides a methodology for calculating emissions for national inventory reporting. The Guidebook provides three alternative, but conceptually similar, methodologies of varying degrees of complexity. The UK's National Atmospheric Emissions Inventory (NAEI) uses the most detailed (Tier 3) approach in a modelling system which uses average speed related emission factors in g/km for a range of different vehicle types, engine size or vehicle weight class, fuel types and emission regulation (Euro standard). The same approach and emission factors are embedded in the EEA's software "Computer Program to Calculate Emissions from Road Transport" (COPERT 5) which is made available specifically for countries to develop their national inventories, i.e. an annual rate of emissions from each vehicle class consistent with national transport statistics. The NAEI does not use the COPERT software, but the same factors and methodology.

4.1.2 Other Models Used for Emission Inventories

Although the majority of countries in Europe use the Guidebook and COPERT approach, there are other modelling approaches used for the national emission inventories of some other European counties.

Some countries (e.g. Austria, Switzerland) use the **Handbook of Emission Factors (HBEFA**). This model, developed by INFRAS[6] in Switzerland, is essentially a traffic situation model rather than an average speed model. The emission factors are based on the simulation of a huge number of driving situations and vehicle categories. The factors account for traffic situation (e.g. how congested the road is) as well as road type.

VERSIT+ is a model developed by TNO in the Netherlands and is used for the Dutch emissions inventory, as well as more local scale modelling[7]. VERSIT+ is a modal emissions model that relates emissions not just to average speed but also to acceleration and/or vehicle specific power (VSP). Therefore, at any given speed the vehicle can be producing different emissions depending on the acceleration or power requirements on the vehicle. It is essentially an empirical model developing speed-acceleration emission maps for a wide range of vehicle types from measured data and/or vehicle simulations. Regression techniques are used to relate measured emissions data to vehicle driving behaviour such as acceleration, speed and vehicle power. Emission factors are developed for specific vehicle types and drive cycles which are then aggregated up with traffic data to provide the emission results.

TREMOD (Transport Emission Model) was designed in the late 90s on behalf of the German Federal Environmental Agency to build up a suitable tool that covers the state of knowledge for emission calculation in Germany. It is constantly updated and used for Germany's national annual emission inventory reports, the projection of past trends and future scenarios for all transport modes. TREMOD is closely linked to HBEFA, using the same methodology and database of emission factors.

[6] https://www.hbefa.net/e/index.html
[7] https://www.tno.nl/media/2451/lowres_tno_versit.pdf

NEMO (Network Emission Model) is used for the Belgium Emission Inventory and was developed at the Technical University of Graz, Austria as a tool for the simulation of traffic related emissions in road networks. Typical applications range from emission inventories for cities, regions and countries to complex measures like environmental zones or promotion of alternative propulsion systems. The parameterisation of NEMO is based on data from European in-use measurements also used for HBEFA and COPERT. NEMO combines both detailed calculation of the vehicle fleet composition and simulation of emission factors on a vehicle level.

MOVES (Motor Vehicle Emission Simulator) is a modal emissions model developed by the USEPA. It derives emissions estimates based on second-by-second vehicle performance characteristics for various driving modes, calculating emissions rates as a direct function of vehicle specific power (VSP) and speed. The default operating modes and drive cycles are set up to reflect US traffic activity, as do the vehicle categories. It is widely used in the US for area-wide emission inventories and locally specific traffic situations in conjunction with microsimulation traffic models.

4.2 Source of Emissions Test Data Underpinning Emission Models in Europe

What all these models used in Europe (COPERT, HBEFA, TREMOD and VERSIT+) and emission factors in the Guidebook have in common is that they all originate from the same centrally held source of emissions test data, although at any one time each model may be at a different stage of development and upgrade. Vehicle emission measurement programmes are carried out in various countries in Europe. Each will have its own set of priorities and objectives but the European Research for Mobile Emission Sources (ERMES[8]) is a collaborative programme involving a group of institutes and organisation from across Europe which brings the test results together to provide a common set of emissions data from which these models are developed. ERMES is administered by the European Commission Joint Research Centre, Institute for Energy & Transport and its Mission and Objectives are described on the ERMES website. Its Mission includes the "*coordination of research and measurement programmes among European research institutions for the improvement of transport emission inventories and projections in Europe*". The ERMES group meets regularly to share research results and discuss priorities for each year's work-programme.

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[8] https://www.ermes-group.eu/web/

The models overseen by the ERMES group are based on a common set of measurements in order to produce consistent emission factors and emissions estimates for modelling and inventory development across Europe. The test data are provided voluntarily by countries participating in ERMES and include data collected from laboratory studies using chassis and engine dynamometers and more recently from PEMS testing. Some data from remote sensing is also now feeding into ERMES. The test data include that from the vehicle emission testing programme carried out by DfT in 2016 on emissions from Euro 5 and 6 diesel cars (DfT, 2016).

The ERMES database of emission factors covers a range of different vehicles and test conditions and these are processed using the Passenger car and Heavy duty Emission Model (PHEM). PHEM is a vehicle emission simulation model developed by the Technical University of Graz, Austria, originally an output from the EU ARTEMIS project.

The basic approach used by the PHEM model is to simulate a vehicle in terms of a physical model to produce engine speed and torque (a schematic of the PHEM model can be found in the report on version 3.3 of the HBEFA model by TU Graz (2017)). This is then used with an empirically-derived engine emissions map, showing how emissions vary with engine speed and power, to produce engine out emissions. The engine emissions are then corrected to give full vehicle emissions:

- A transient correction – this corrects for the fact that most of the engine maps are steady state engine maps, and so a correction is needed to account for the transient nature of actual driving cycles.
- Catalyst or exhaust aftertreatment module – related to engine temperature.

PHEM simulates vehicle hot and cold start emissions for different driving cycles, gear shift strategies, vehicle loadings, road gradients, vehicle characteristics (mass, size, air resistance, etc.) and is validated by emission measurements from the ERMES database both from light and heavy duty vehicles in laboratories (chassis and engine test bed) and on the road (with PEMS) and under different test conditions.

The vehicle emissions test data from ERMES are run through PHEM to develop emission factors tailored to a particular format for fleet emission models such as COPERT and HBEFA. So, for example, PHEM would be used to develop emission factors at different cycle average speeds for use in COPERT or different traffic situations for use in HBEFA.

A presentation available on the ERMES website[9] shows the COPERT, HBEFA and VERSIT+ model approaches and how these are linked through the common ERMES emissions dataset.

[9] https://ermes-group.eu/web/system/files/filedepot/11/ERMES_presentation_Dec2017.pdf

Taking account of the fact that models are always at a different stage of development at any one time, in principle all these models should produce the same or consistent emission factors under the same drive cycle/speed conditions since they are evolved from the same core database of emission measurements processed through the same vehicle emissions simulation model.

4.3 Approaches Used for Local-scale Emissions Calculations

The preceding sections describe models that are widely used for national scale emission inventories. The same emissions models are also routinely used to calculate local-scale emissions. For example, the Pollution Climate Mapping (PCM) approach for modelling of roadside pollutant concentrations which Defra uses for reporting to the European Commission is based on the NAEI emissions. Similarly, Defra publishes an Emissions Factors Toolkit (EFT) for use by local authorities in local-scale assessments, which is also based on the COPERT-derived emissions factors from the NAEI. Assessments of the impacts of Highways England road schemes and the development of Clean Air Zones have also been based on COPERT-derived emissions factors. This approach is convenient and provides an element of consistency, but the COPERT emissions factors are not intended to inform local-scale assessments and, as shown in Section 4.7.4, need to be used with caution.

4.3.1 Microsimulation Emissions Models

Microsimulation emission models are used to model emissions on a much finer scale, e.g. an individual road link, than models such as COPERT. The PHEM model itself may be used for this purpose, as well as VERSIT+. Microsimulation models do not always improve simulations and are themselves based on detailed results from a limited number of tested vehicles. These models require detailed traffic movement data so are often linked to traffic simulation models such as VISSIM. The additional demands placed on traffic models, when compared with average speed modelling, may add further sources of uncertainty.

4.4 Summary of Relationship between Popular Exhaust Emission Models

The relationship between popular exhaust emissions models is summarised in Figure 4.1 and Table 4.1. This also shows how many of the most popular emission models and tools used in the UK for local air quality management such as Defra's EFT and the emissions screening tool in Highways England's Design Manual for Roads and Bridges (DMRB)[10] also stem from the COPERT average speed modelling approach used in the NAEI and also in the London Atmospheric Emissions Inventory (LAEI)[11].

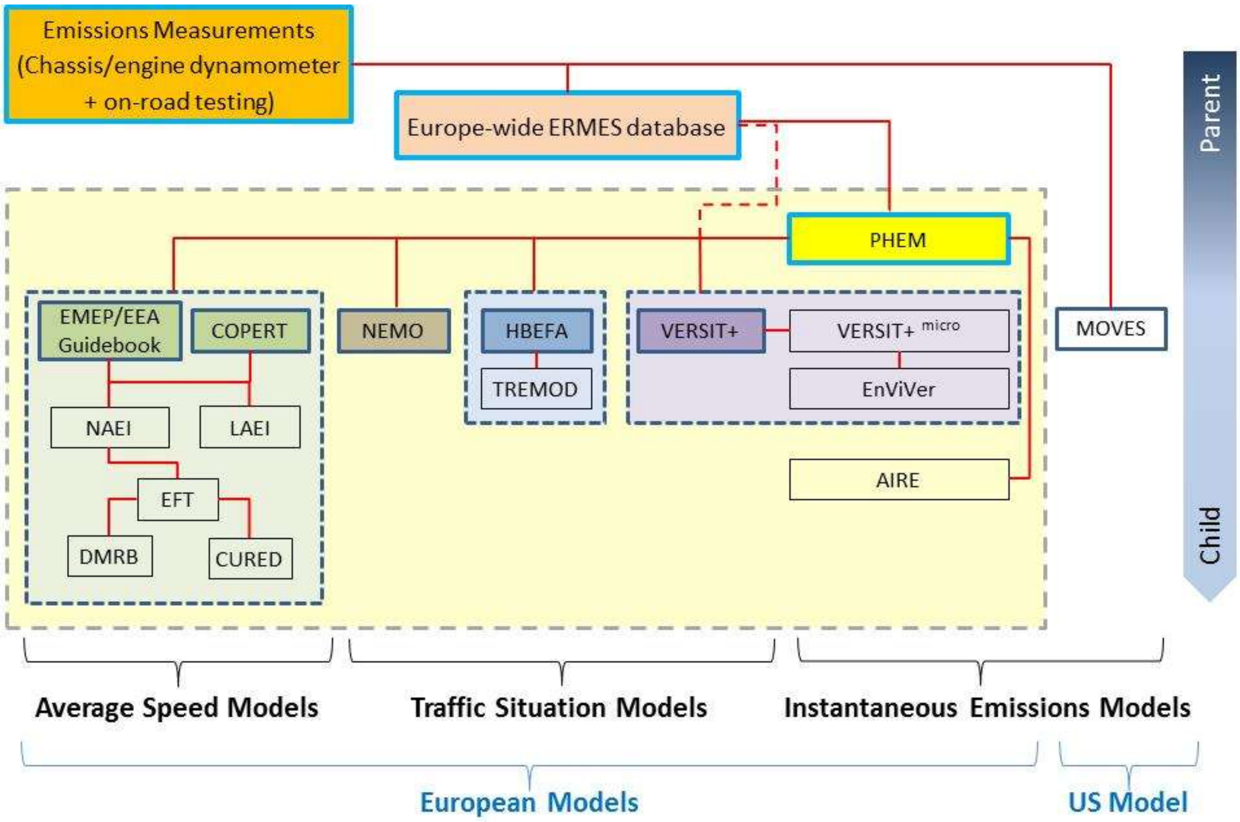

Figure 4.1: Relationship between popular exhaust emissions models. Colours denote closely related models.

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10 https://laqm.defra.gov.uk/documents/DMRB-guidance_V4.pdf

11 https://data.london.gov.uk/dataset/london-atmospheric-emissions-inventory--laei--2016

Table 4.1: Summary description of widely used road transport emission models

Acronym	Description
ERMES	The European Research on Mobile Emissions Sources (ERMES) group unites >50 organisations across 23 countries. It works to harmonise measurement methods and provide a common format for data sharing.
PHEM	Passenger Car and Heavy Duty Emissions Model (PHEM) calculates engine power demand based on driving resistance and transmission loss. Engine power and speed are then used to reference emissions from engine emissions maps.
COPERT	Development of the COmputer Programme to calculate Emissions from Road Transport (COPERT) is coordinated by the European Environment Agency. Exhaust emissions are calculated from speed-emission curves for drive-cycle average speeds.
NEMO	Network Emission Model (NEMO) used for the Belgium Emission Inventory.
HBEFA	Handbook of Emissions Factors for Road Transport (HBEFA) is widely used in Germany, Austria and Switzerland. It provides emissions factors based on traffic situations (e.g. 'urban stop and go', 'urban freeflow' etc.). These are derived using categorised drive cycle segments and PHEM.
VERSIT+	Dutch national emission factors model. The light-duty vehicle module is based on statistical relationships between situation type (driving behaviour) and emissions. The heavy-duty emissions module is based on PHEM.
VERSIT+^{micro}	Simplified version of VERSIT+ to allow linkage to microsimulation traffic models, using average vehicle fleets.
MOVES	The United States Environmental Protection Agency (EPA) Motor Vehicle Emission Simulator (MOVES) combines activity data and emissions measurements, with non-idle emissions referenced to calculated vehicle-specific power.

Acronym	Description
NAEI	UK National Atmospheric Emissions Inventory (NAEI). For exhaust emissions from road traffic, combines average-speed emissions curves from COPERT with UK-specific vehicle fleet inventory data.
LAEI	London Atmospheric Emissions Inventory (LAEI). Emissions calculated using the London Air Quality Toolkit, which is underpinned by the NAEI.
EFT	Defra's Emissions Factors Toolkit (EFT) uses the assumptions from the NAEI to provide average-speed-specific emissions from different vehicle types. This model is recommended by Defra for Local Air Quality Management and local authority modelling carried out to inform the Air Quality Plan for NO_2 in the UK.
DMRB	Highways England's Design Manual for Roads and Bridges (DMRB) air quality model. The published 2007 model uses emissions factors from the 1999 NAEI. The unpublished 2013 model, which is used by Highways England, was developed using EFT V5.2 (itself derived from COPERT 4 v8.1) but can be used with more recent versions.
CURED	Calculator Using Realistic Emissions for Diesels (CURED), intended for sensitivity testing, applies basic adjustments to the assumptions in the EFT.
TREMOD	TRansport Emission MODel (TREMOD) is produced on behalf of the German Federal Environmental Agency and predicts energy consumption and pollutant emissions.
EnViVer	An add-on to the VISSIM micro-simulation traffic model which provides emissions factors from VERSIT+^{micro}.
AIRE	Analysis of Instantaneous Road Emissions (AIRE) was produced for Transport Scotland in 2011. It comprises a series of look-up tables which were derived from the 2005 version of PHEM. AIRE was developed to integrate with the S-Paramics micro-simulation traffic model but can also be used with other sources of activity data.

4.5 Modelling Ambient Concentrations from Exhaust Emissions

The previous sections of this chapter describe methods for estimating vehicle exhaust emissions on a national scale, local area or on a road. A further step is required to model the effect the exhaust emissions have on ambient concentrations at different distances from the roadside. A description of all available models is outside the scope of this report, but a conceptual diagram showing how exhaust emission models and models used for predicting ambient concentrations is shown in Figure 4.2.

Emissions occurring along a length of road need to be expressed in mass emitted per unit time (seconds) and used in an atmospheric dispersion model that accounts for atmospheric conditions and local terrain to calculate the increment caused by the dispersed exhaust emissions to the background pollutant concentrations. Models must also account for chemical processing of the emitted pollutants downstream of the traffic source, such as the conversion of NO_x to NO_2 and formation of $PM_{2.5}$ dependent on location and the prevailing atmospheric conditions.

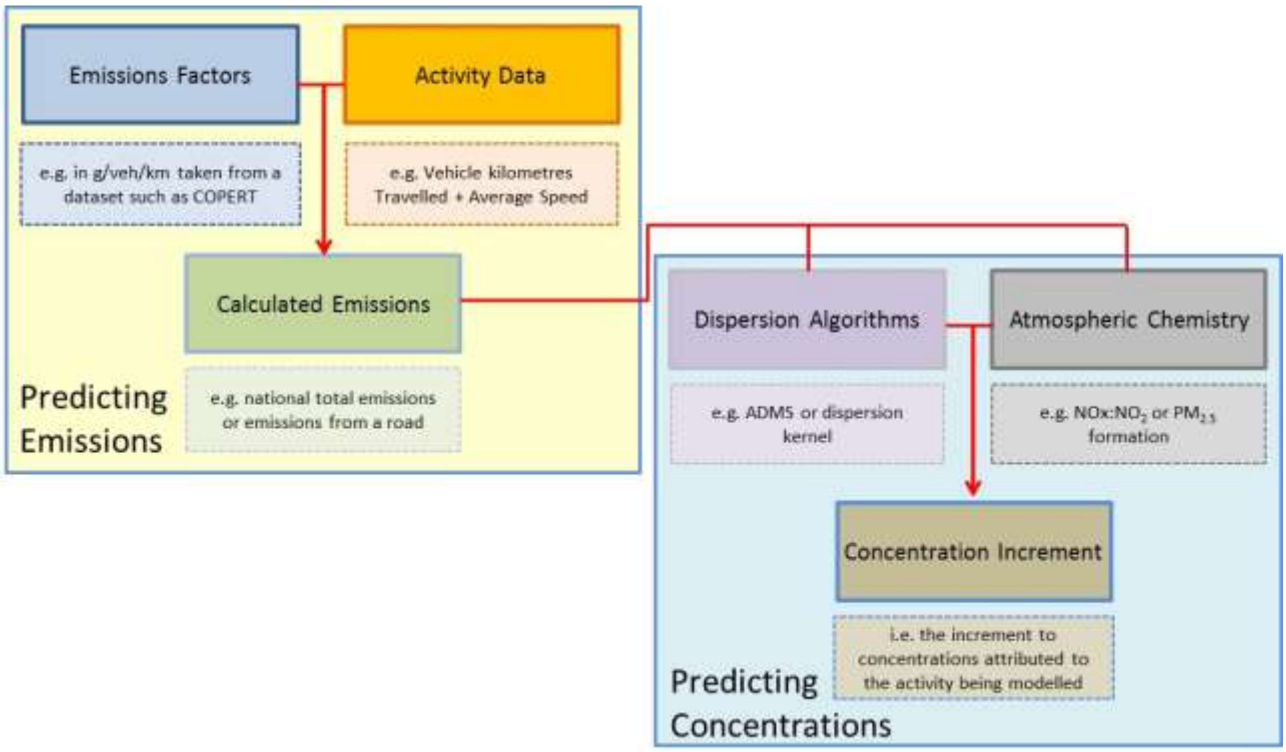

Figure 4.2: Conceptual Diagram of Approaches Used to Predict Emissions and Ambient Concentration Using Models

4.6 The UK's National Emissions Inventory and Projections

The primary aim of the NAEI is to use the average speed-related emission factors in the EMEP/EEA Emissions Inventory Guidebook approach to develop a consistent time-series in UK emissions from the road transport sector for each pollutant from 1990 to the latest inventory year and projected to 2030. This approach (which is often referred to as the COPERT model approach and uses the same emission factors) is chosen because it works well with the traffic activity data available from the Department for Transport (DfT) for both historical years and projections. It is endorsed when the UK's emissions inventory is reviewed by sector experts assessing inventories submitted by Member States under the revised EU Directive 2016/2284/EU on National Emissions Ceilings (NECD)[12] and the United Nations Economic Commission for Europe (UNECE) Convention on Long-Range Transboundary Air Pollution (CLRTAP)[13].

The NAEI calculates exhaust emissions from petrol cars, diesel cars, petrol light goods vehicles (LGVs), diesel light goods vehicles, rigid and articulated heavy goods vehicles (HGVs), buses and coaches and mopeds and motorcycles. Details of the methods, data sources and assumptions used for the road transport emissions inventory are provided in the UK's annual Informative Inventory Report (NAEI, 2019). Hot exhaust emission factors in g/km are available as equations relating emissions to average drive cycle speed for each of these vehicle types, in different ranges of engine capacity, engine weight classes (for LGVs, HGVs and buses) and Euro standard emission category (from pre-Euro 1/I to Euro 6d/VI). In some cases, there are different factors for different vehicle technologies such as conventional hybrid petrol cars, and HGVs using SCR or EGR, but in general the currently available set of emission factors in the Guidebook does not differentiate between the whole range of technology options that are now used in light duty applications, such as lean NO_x traps and SCR.

The NAEI uses the same approach for forecasting emissions from the fleet in future years using sources of information from DfT. The main drivers for future emissions are DfT's latest traffic forecasts from the National Transport Model (NTM) and assumptions on new car sales used in the NAEI's fleet turnover model to predict the future fleet composition.

4.6.1 Vehicle Activity Data Used for Historical Years and Projections

In the NAEI, the COPERT emission factors are combined with distances driven on different road types (urban, rural and motorway roads) according to UK annual vehicle kilometre

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[12] See https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32016L2284&from=EN for Information on the new NEC Directive (2016/2284/EU).

[13] See http://www.ceip.at/ms/ceip_home1/ceip_home/reporting_instructions/reporting_programme/ for reporting requirements of estimating and reporting emissions data under the CLRTAP

figures provided each year by DfT from the UK's traffic census (DfT, 2018). Figure 4.3 shows the trend in total UK vehicle kilometres disaggregated by main vehicle type from 2015 to 2030 according to the NAEI. These come from DfT's traffic statistics for years up to 2017 and DfT's latest Road Traffic Forecasts 2018 for Reference Scenario 1[14]. With regard to ultra-low emission vehicles, this forecast scenario includes implemented and adopted policies only. These do not include future policies or Government ambitions that have not been legislated, but assumes 25% of car and LGV mileage are powered by zero exhaust emission technologies by 2050.

Historically, overall traffic has grown steadily by nearly 30% from 1990 to 2017. According to DfT's traffic forecast the current growth in total vehicle km which has been observed since 2012 is expected to continue to 2030. The dominance of passenger cars is apparent from this figure, but whilst the growth in passenger car km is expected to continue, there is a shift between fuel/power type with a decline in diesel car km, a small growth in petrol car km and a more significant growth (from a current very low base) in activities by electrically-powered cars. In this plot electric cars cover both the mileage driven by battery electric vehicles and the mileage driven in electric mode of plug in hybrid electric vehicles.

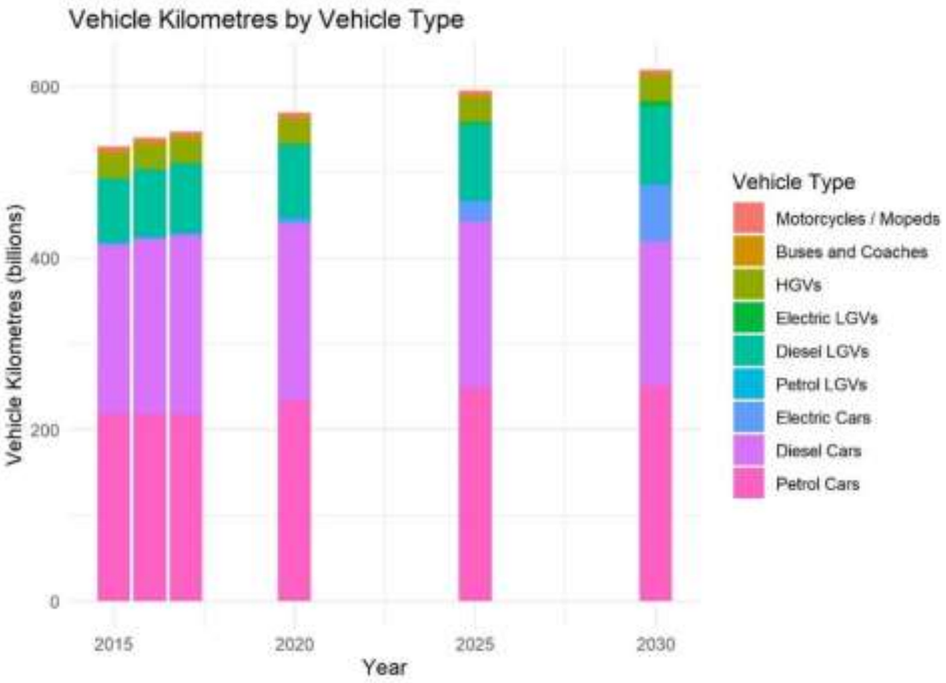


Figure 4.3: Total UK vehicle kilometres by vehicle and fuel type.

[14] See https://www.gov.uk/government/publications/road-traffic-forecasts-2018 for details of assumptions used

4.6.2 Vehicle Fleet Composition Used in the NAEI for Historical Years and Projections

The vehicle kilometres data from DfT are further disaggregated by fuel type, age and Euro classification using the NAEI's fleet turnover model informed by DfT's vehicle licensing statistics, annual mileage data (showing how mileage changes with age of vehicle) and detailed information from a network of Automatic Number Plate Recognition (ANPR) cameras also provided by DfT's roadside survey which shows the fuel split and age mix of vehicles on different types of roads. The ANPR data shows, for example that the fleet of HGVs on the road tends to be newer than in the registered vehicle fleet according to vehicle licensing data, indicating how newer vehicles are used more than older vehicles. This is illustrated in Fig. 4.4 for articulated HGVs. This chart simply shows that there is a greater proportion of Euro VI artic HGVs observed on the road by ANPR cameras in 2017 than is present in the fleet according to vehicle registrations in that same year.

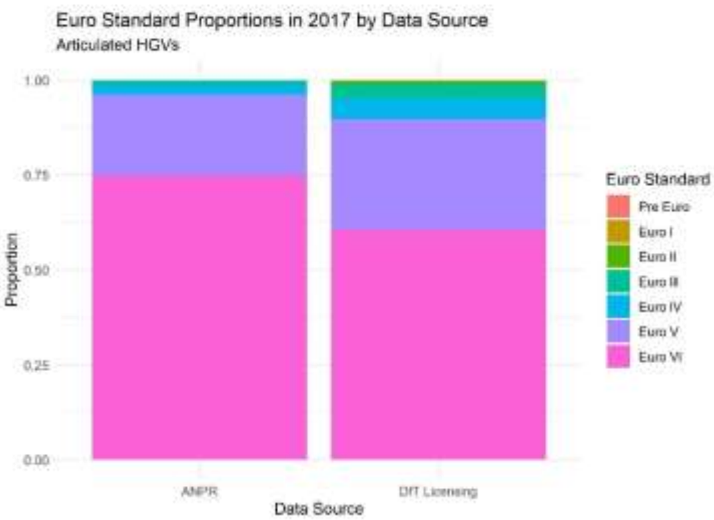


Figure 4.4: Share of articulated HGVs by Euro emission standard classification as observed on the road by ANPR cameras at 250 sites in the UK in 2017 compared with the share according to vehicle licensing data in the same year

Because of significant differences between the emission factors for recent stages of Euro standards of HGVs (Euro IV-VI), the trend in NO_x emissions is sensitive to this rapid turnover in the HGV fleet, as evidenced from the ANPR data.

It is important to recognise, however, that this figure represents the average fleet mix on different types of roads at 256 sites in the UK and masks any regional differences. Insufficient ANPR data from DfT were available to determine whether there are any statistically significant differences in the fleet compositions between regions and cities. For larger articulated HGVs which travel large distances between regions, such differences may be small, but previous analysis by the NAEI of vehicle licensing data according to postcode showed that there were regional differences in the age of the car fleet between cities, with the fleet tending to be older in more rural areas of England for example. However, some caution needs to be exercised when interpreting licensing data because cars may not be used in areas where they are registered, particularly in the case of company-owned cars.

In compiling the national inventory and emission maps, the NAEI has to date not included any regional variations in the fleet mix according to vehicle age and fuel type, apart from in London where detailed fleet information has been provided by TfL. Historically, this may not have had a significant impact on the accuracy of the inventory, but with individual towns and cities considering options for influencing the fleet in Clean Air Zones, taking account of differences in the current and future fleets at individual city level will be much more important, both for the NAEI's national inventory totals and, more specifically, for national and local scale modelling of air quality and development of policies. The NAEI is currently exploring local fleet data as provided by some local authorities based on ANPR measurements to examine how different these are from the national average so that more locally-sensitive modelling can be treated in Defra's Pollution Climate Mapping programme and assessment of current and future policies. The NAEI is also discussing with DfT the use of other data sources such as data from MOT records showing how annual mileage patterns change with vehicle age.

Some fleet compositional data are now available for some cities from publicly available Clean Air Zone reports which show differences from the national trend. This issue on regional fleet differences is further discussed in Section 4.7 considering uncertainties in emission estimates.

The NAEI makes further assumptions on the effect of catalyst failure on petrol cars and SCR systems on diesel cars and vans. Details are given in the Informative Inventory Report (NAEI, 2019). There are no published data on system failure rates and the NAEI assumptions are from advice given by DfT on failure and repair rates largely based on sales of replacement catalysts and the effects of regulation controlling sale and installation of replacement catalytic converters and particle filters for Euro 3 (or above) light duty vehicles since 2009.

A fleet turnover model is used to predict the future fleet composition. This takes into account the fleet penetration of emission standards up to Euro 6d/VI as well as battery and hybrid electric vehicles. However, no account is taken of the penetration of alternative fuels in the fleet such as gas-fuelled HGVs or high strength biofuels. The NAEI projections on biofuel uptake are guided by DfT's Renewable Transport Fuel Obligation (RTFO) which supports the government strategy to reduce carbon emissions from road transport. DfT consulted in 2017 on a range of measures to amend the RTFO and proposed the mix of different biofuel types that DfT expect to achieve its overall uptake target (DfT, 2017). This includes the uptake of 10% bioethanol, E10.

The NAEI uses vehicle activity data projected as vehicle km travelled per year (vkm) at a fairly disaggregated level, including by road and area type, and uses these in conjunction with fleet projections and emission factors from the EMEP/EEA Emissions Inventory Guidebook and the COPERT 5 model (Emisia, 2019) to forecast future emissions. In predicting the future fleet, the NAEI has traditionally relied on a fleet turnover model based on national assumptions informed by DfT, e.g. on projected sales of future petrol, diesel and electrically-powered cars and nationally-averaged assumptions on vehicle lifetimes and usage. Future vehicle activity and fleet composition projections for London are provided directly by TfL taking into account the Ultra Low Emission Zone introduced in 2019. As mentioned earlier, the NAEI is investigating using more localised fleet data in the current inventory. One of the significant challenges that will exist is where locally a fleet is found to be different to the national average situation, then estimating how that fleet will evolve in future when the effect of future national policies at local level is not understood, e.g. will a local fleet always remain older or newer than the national fleet? This will be less of a problem in areas where Clean Air Zones are to be introduced which will, in effect, define the composition of the fleet in future (e.g. where all vehicles are required to be a minimum of Euro 6 standard).

Using recent advice from DfT (April 2019) on future sales and km travelled by petrol, diesel and electric cars, the NAEI fleet turnover model leads to predictions in the breakdown of diesel car km by Euro standard as shown in Fig 4.5. Fleet composition and projections for rigid HGVs are also shown by way of example. These figures reflect both the DfT current trends and forecast in road traffic and the penetration of successive Euro categories in the vehicle fleet. This figure shows how diesel car km have been increasing in recent years and the fleet is dominated by Euro 4 and 5 vehicles. In future years, overall diesel car km are predicted to decrease and the fleet becomes dominated by the lower emitting Euro 6 category of vehicles. In the case of rigid HGVs, there is predicted to be little change in total vehicle km across the time-series, but the fleet becomes dominated by Euro VI vehicles.

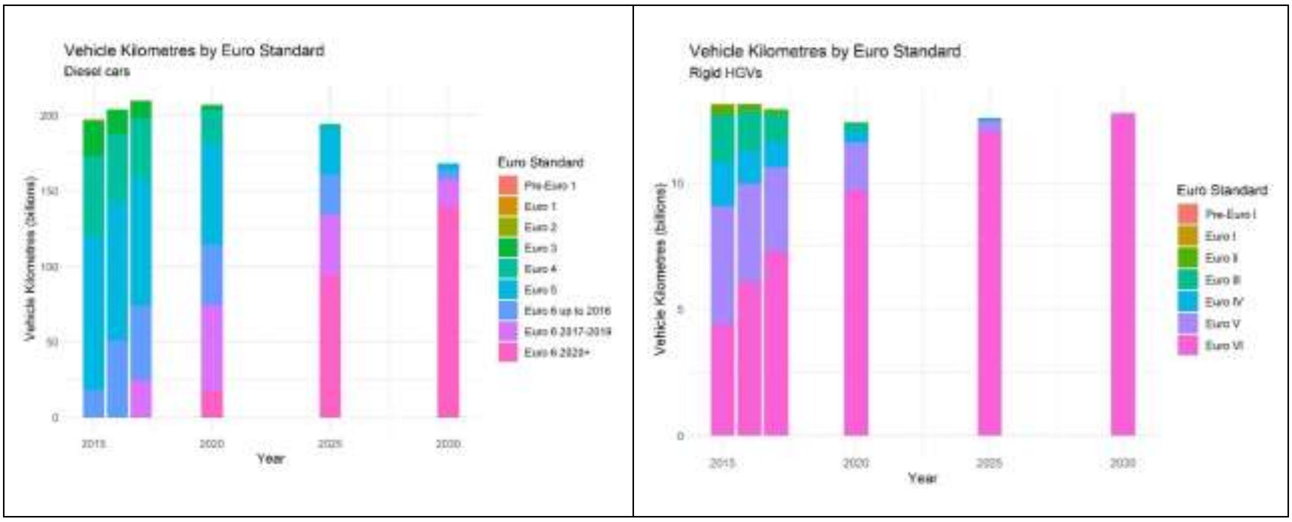


Figure 4.5: Total UK vehicle kilometres by diesel cars and rigid HGVs disaggregated by Euro emission standard

4.6.3 Exhaust Emission Factors Used in the NAEI for Historical Years and Projections

The equations relating exhaust emission factors in g/km to average vehicle speed are provided for each pollutant and vehicle type in an Excel spreadsheet available for download on the website for the EMEP/EEA Emissions Inventory Guidebook. A recent version (as of December 2019) is available at https://www.eea.europa.eu/publications/emep-eea-guidebook-2019/part-b-sectoral-guidance-chapters/1-energy/1-a-combustion/road-transport-appendix-4-emission/view. A slightly earlier version is currently used in the NAEI and has been made available for NO_X and PM for greater ease of use on the NAEI website at https://naei.beis.gov.uk/data/ef-transport. These factors currently underpin the version of the NAEI covering years up to 2017 and projections to 2030 (published in 2019) as well as the latest version of the Defra Emission Factor Toolkit (EFT v9).

As an example, Figure 4.6 shows the trend in NO_x emission factors across the Euro standards from pre-Euro 1/I to Euro 6/VI for petrol cars, diesel cars, diesel LGVs, articulated HGVs (40-50t) and urban buses (15-18t) for a typical urban speed of 36 kph developed from the version of the emission factors used in the version of the NAEI and projections published in 2019. Emission factors for cars and LGVs are given for 3 sub-categories of Euro 6, which assume incrementally lower NO_x emissions for cars and LGVs coming into service over the period 2015-2020. These categories reflect an expectation by the COPERT developers that emissions will reduce over time but are not intended to exactly align with individual steps in the Euro 6 regulation (i.e. Euro 6c, Euro 6d temp, Euro 6d). The NAEI fleet composition data for these Euro 6 sub-categories are aligned with the dates for vehicle first registrations which these COPERT emission factor categorisations refer to. Note that the vertical axes are on different scales for each vehicle type. This figure illustrates the significant reduction in NO_x emissions from petrol cars with the introduction of three-way catalysts to meet Euro 1 standards and the virtual absence of any reduction in emission factors for diesel cars and LGVs up to Euro 5. COPERT is anticipating reduction in factors for Euro 6d with the introduction of Real Driving Emission (RDE) regulation. Emission factors for Euro VI heavy duty vehicles introduced in 2013 show a large reduction in NO_x emissions.

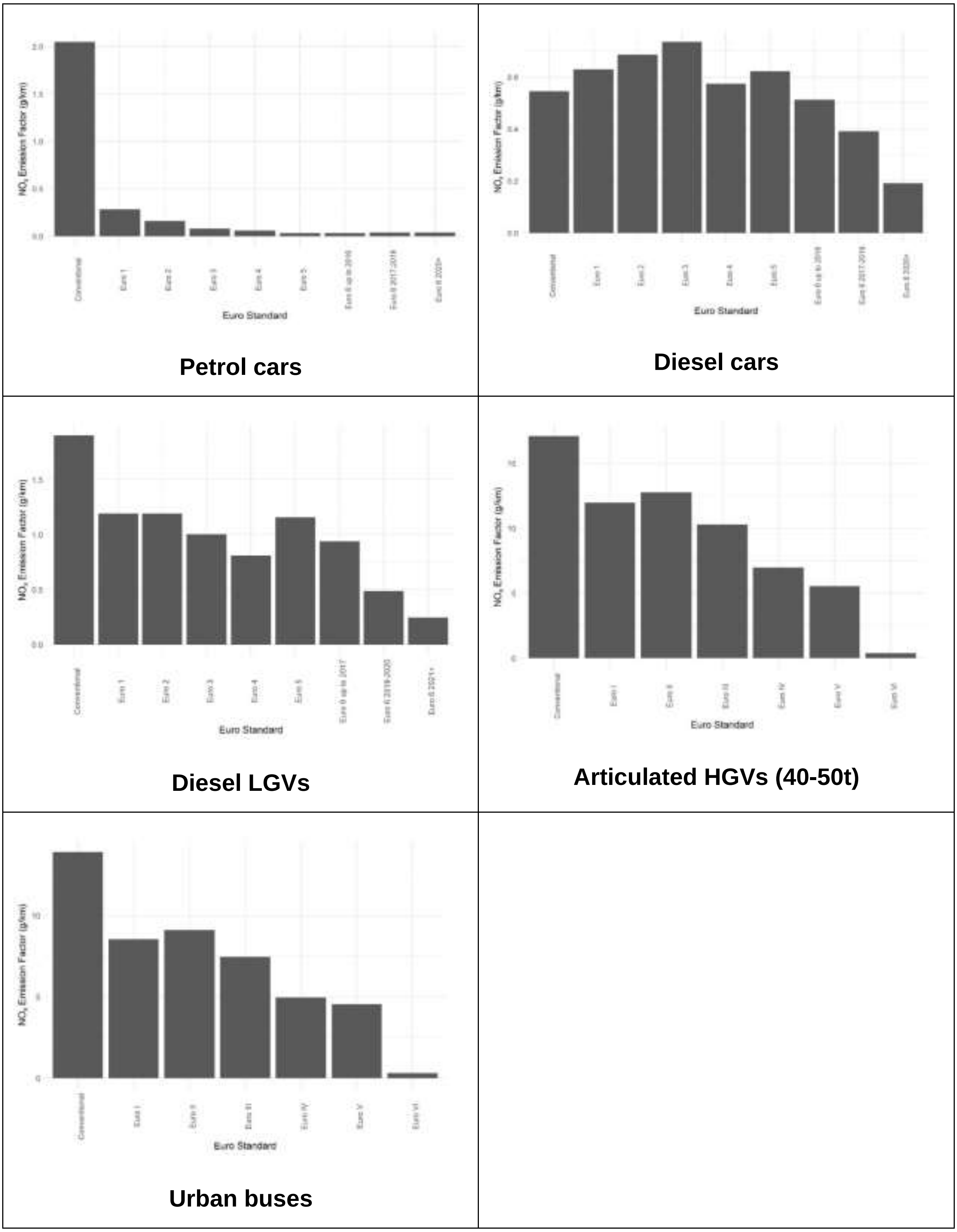


Figure 4.6: NO_x Emission factors for petrol cars, diesel cars, diesel LGVs, articulated HGVs (40-50t) and urban buses calculated from COPERT 5 speed-emission equations at a speed of 36 kph.

4.6.4 Emissions for Alternative Fuels and Technologies

For historical years, emissions are dominated by conventional petrol and diesel fuels and technologies and emission factors in sources such as the Emissions Inventory Guidebook tend to be differentiated according to these primary fuel types and Euro emission standard. The emission regulations themselves have been the main driver for change in emission factors.

For understanding more recent trends in emissions and certainly for forecasting emissions in future years and for running policy scenarios, it becomes increasingly important to consider differences in emissions for a far wider range of vehicle powertrains, alternative fuels and technologies. In this respect, the Emissions Inventory Guidebook is more limited although it does provide emission factors for some alternative fuelled vehicle types and exhaust aftertreatment technologies. For example, in the case of heavy duty vehicles, factors are provided for Euro V vehicles with EGR and SCR technologies and the NAEI uses COPERT default assumptions on the mix of these technologies in the fleet. The Guidebook does not currently provide different emission factors for gasoline direct injection engines (GDI) and port fuel injection engines (PFI) and the NAEI does not have access to data on the mix of GDI and PFI engines in the current and future fleet. The Guidebook also does not differentiate between the whole range of technology options that are now used in diesel light duty applications, such as lean NO_x traps and SCR.

The UK is currently seeing a rapid growth in the number of hybrid and battery electric cars in the fleet, although total numbers in the fleet are still low at the moment. The Guidebook provides factors for full hybrid (i.e. non-plug-in) passenger cars. In 2015, Ricardo-AEA undertook a review for DfT on NO_x, PM and CO_2 emission and energy consumption factors for a range of alternative powertrains including full hybrids and plug-in hybrid cars and vans. This review, which has only recently been published, developed speed-emission factor relationships in typical COPERT format guided by some PEMS measurements and analysis using the PHEM model (Ricardo-AEA, 2015). Factors for plug-in hybrids were developed for vehicles with different electric range using a utility factor approach, the average ratio between electric and hybrid operation mode. These factors have not yet been used in the NAEI.

Few vehicles in the UK run on liquefied petroleum gas (LPG). There are no reliable figures available on the total number of vehicles or types of vehicles running on this fuel. It is believed that many vehicles running on LPG are cars and vans converted by their owners and that these conversions are not necessarily reported to vehicle licensing agencies. Figures from the Digest of UK Energy Statistics (BEIS, 2018) suggest that the consumption of LPG is less than 0.2% of the total amount of petrol and diesel consumed in 2017 and LPG consumption has been declining since 2006. Emission factors for LPG are highly uncertain and whilst factors are available in the Emissions Inventory Guidebook, it is not certain these apply to retrofit conversions.

The NAEI does not estimate emissions from vehicles using compressed natural gas (CNG) as there are no data in the Digest of UK Energy Statistics (DUKES) on the amount of CNG used by road transport, nor are there useable data on the total numbers and types of vehicles equipped to run on natural gas from vehicle licensing sources. Limited information on emission factors for cars and urban buses running on CNG are given in the Emissions Inventory Guidebook (EEA, 2019).

In the UK, gas (CNG or biogas) has been seen as a potentially viable fuel for HGVs. A Low Carbon Vehicle Project study (LowCVP, 2017) on the "Emissions testing of Gas Powered Commercial Vehicles", was completed and published in 2017. The overall view of this study, and consultation with LowCVP's Commercial Vehicle Working Group, recommends that the Government: "should continue to support the development of gas vehicle infrastructure *and gas-powered vehicles, **particularly dedicated gas**, while increasing the supply of low carbon/renewable methane as a sustainable transport fuel in order to realize these benefits.*" This study which was carried out on behalf of DfT included some emissions test data. Prior to the LowCVP report, Ricardo-AEA included estimates of emission factors for gas-fuelled HGVs in its review for DfT on emissions for alternative powertrains and fuels mentioned earlier (Ricardo-AEA, 2015). The review undertaken in 2015 made recommendations on factors for dedicated and dual-fuel gas/diesel HGVs based on information available at the time. For dedicated gas fuelled HGVs, there may be some benefits to NO_x emissions compared with a conventional diesel vehicle comparator, though this has been brought into question in a report by Transport & Environment based on PEMS testing carried out by TNO on three trucks running on liquefied natural gas which suggested NOx emissions over two times higher than from a conventional Euro VI diesel truck on an urban cycle (T&E, 2019). There may also be issues with methane slip emissions occurring particularly for dual fuel gas/diesel vehicles.

4.6.5 Further Methods and Assumptions Used in the NAEI

A further set of adjustment factors taken from the method in the EMEP/EEA Guidebook is used to account for emission factors degrading with accumulated mileage (different rates of degradation are used for different pollutants, vehicle types and Euro emission class) and road gradient and vehicle load factor in the case of HGVs (referring to the freight loading of the vehicles).

The effect of changes in fuel quality on exhaust emission is also accounted for, though this is most relevant to older vehicles on the road in previous years when fuel quality (e.g. sulphur content) was different to current levels. The effect of biofuels on exhaust emissions is accounted for in the NAEI using the scaling factors described in the AQEG (2011) report on the impacts of biofuels on air quality. That report was based on evidence available at the time on the effects of biofuels on exhaust emissions of air pollutants and a more recent review by the NAEI found no further definitive evidence on biofuel effects. The exception to this was that scaling factors for the impact of biodiesel and bioethanol on PM emissions were updated in line with a more recent report published by the European Commission (2015) which suggested smaller emission reductions on PM than implied in the AQEG report, particularly for bioethanol. Most biofuel is consumed in the UK as weak (<10%) blends with fossil fuel petrol and diesel. The NAEI combines the emission effects of biofuels with biofuel uptake rates according to figures published by HMRC (2018).

These factors are used to calculate exhaust emissions from vehicles operating with engines fully warmed up. A separate methodology from the EMEP/EEA Guidebook based on average trip lengths is used to estimate the excess emissions occurring during cold starts. This was a particularly important source of emissions from petrol cars, particularly the early Euro standards because of the time it took for three-way catalyst systems to warm up to their effective operating conditions.

4.6.6 Overall Summary of Methodology Used in the NAEI

The NAEI's road transport emissions and fleet turnover model is set up to run a baseline scenario according to the forecast data available from DfT and as required to support Defra's air quality policy through data supplied to the Pollution Climate Mapping (PCM). The model aims to be sufficiently flexible to run emission scenarios if alterative assumptions can be provided such as different traffic growth rates and different fleet turnover and fleet composition assumptions as defined by Defra and DfT. However, it is not a behavioural model that can predict changes in vehicle activity and new vehicle purchasing and scrappage trends in response to national or local economic and other policy drivers.

Further details on the UK's emission projections are provided in the UK's Inventory Report available on the NAEI website (NAEI, 2019) and projections data submitted to CLRTAP are also available at https://cdr.eionet.europa.eu/gb/un/clrtap/projected/envxio6gq/index_html .

To summarise, a schematic of how the road transport (RT) emissions inventory and projections are calculated in the NAEI is shown in Figure 4.7.

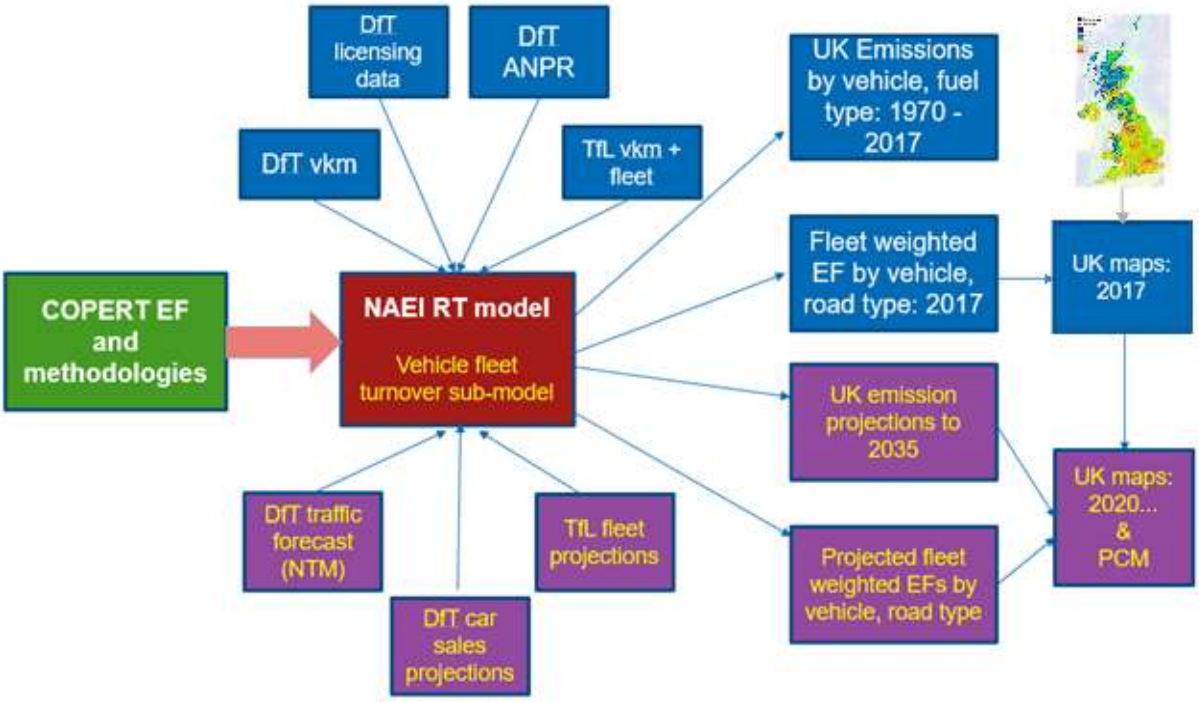


Figure 4.7: Schematic of the methodology and sources of data used by the NAEI in estimating current and future emissions from road transport in the UK. Boxes in blue refer to current and historical years; boxes in purple refer to projecting emissions in future years.

4.6.7 Trends in UK Exhaust Emissions from Road Transport

The NAEI estimates total UK exhaust emissions of each pollutant from 1990 to 2017 and projected to 2030 using the UK vehicle km, COPERT-based emission factors and average speeds for different road types on urban, rural and motorway roads. Figure 4.8 shows the trend in total UK emissions of NO_x from road transport from 2015 to 2030 disaggregated by vehicle type according to the NAEI. Data up to 2017 are based on actual fleet and traffic levels in these years. It should be noted that the UK reports to CLTRAP two types of inventories for road transport: one based on 'fuel used', where emissions are calculated directly from g/km emission factors and vehicle km travelled, and another based on 'fuel sold' where the 'fuel used' estimates are subsequently normalised to fuel sales data as given in DUKES. As DUKES is not able to provide petrol and diesel sales data disaggregated by vehicle type, only as sales totals, the 'fuel sold' inventory time-series can break the link between vehicle km travelled and emissions on a vehicle-by-vehicle type basis and is therefore less useful for policy. The figures shown here are on a 'fuel used' basis. This point will be discussed further in Section 4.7.3 on uncertainties.

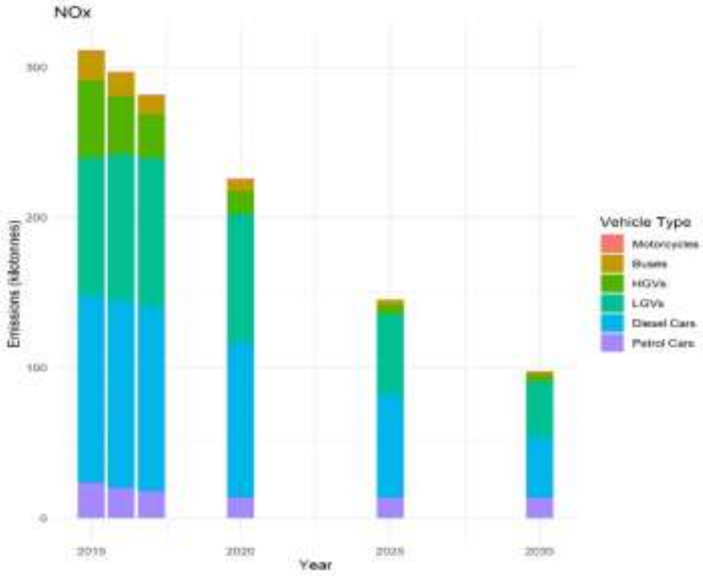


Figure 4.8: UK exhaust emissions of NO_X from road transport by vehicle type and projections to 2030 according to the NAEI (2019).

Emissions of NO_X from road transport in 2017 were 282 ktonnes/year, according to the version of the NAEI published in 2019, which corresponds to 32% of total UK NO_X emissions from all sources. The dominant sources of road transport emissions were diesel cars (44%) and diesel LGVs (35%). Emissions are predicted to decrease by 65% by 2030 relative to 2017 levels, mainly due to the reductions in diesel car and LGV emissions, though these remain dominant sources of road transport emissions in 2030, together contributing 79% of the road transport emissions.

Figure 4.9 shows the trend in UK NO_X emissions from diesel cars by Euro standard where it can be seen how the contribution from Euro 5 and earlier cars gradually vanishes and the remaining emissions by 2030 are due to the Euro 6 vehicles This suggests that with the contribution from other vehicles being so small by this stage (other than diesel LGVs which show a similar trend), further reductions in emission factors than those currently assumed for Euro 6d diesel cars and LGVs would be necessary in order to have a significant impact on UK road transport NO_X emissions in 2030 and beyond.

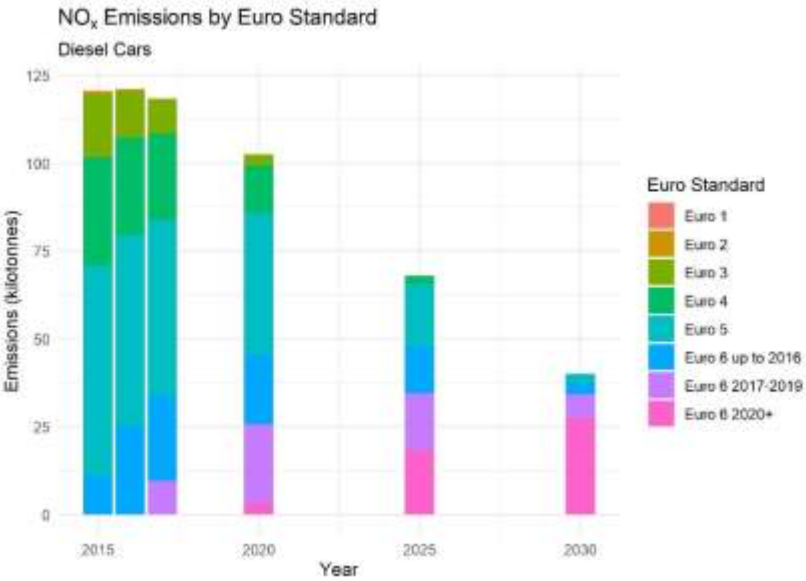


Figure 4.9: UK exhaust emissions of NO_x from diesel cars disaggregated by Euro standard and projections to 2030 according to the NAEI (2019).

Figure 4.10 shows the corresponding trend in total UK exhaust emissions of $PM_{2.5}$ from road transport from 2015 to 2030; non-exhaust sources of emissions from tyre and brake wear and road abrasion are not included here. Again, the figures are on a ‘fuel used’ basis. Exhaust emissions of $PM_{2.5}$ from road transport in 2017 were 4.7 ktonnes in 2017, corresponding to 4.4% of total UK $PM_{2.5}$ emissions from all sources. A much greater share of $PM_{2.5}$ emissions comes from the non-exhaust sources from brake and tyre wear and road abrasion, as discussed in the AQEG report on these sources (AQEG, 2019). The dominant contributions to road transport exhaust emissions were again diesel cars (50%) and diesel LGVs (30%). Total exhaust emissions from road transport are predicted to decrease by 80% by 2030 relative to 2017 levels, mainly due to the reductions in diesel car and LGV emissions. Figure 4.10 shows little change in petrol car exhaust emissions owing to the fact that there are no legislative drivers to further reduce PM mass emissions from these vehicles which are already very low so that by 2030, these vehicles are the largest overall share in PM exhaust emissions (41%) compared with other vehicles. However, this needs to be put in the context that exhaust emissions are expected to be very small relative to the contribution from non-exhaust sources unless further measures are introduced to reduce those emissions.

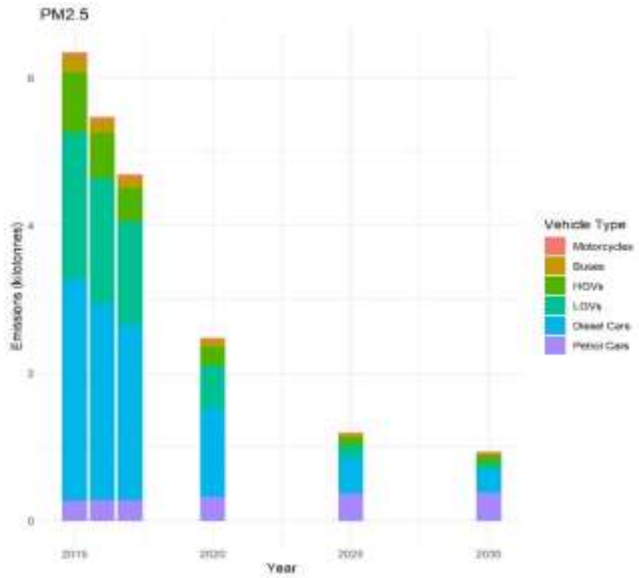


Figure 4.10: UK exhaust emissions of $PM_{2.5}$ from road transport by vehicle type and projections to 2030 according to the NAEI (2019).

Section 3.3.5 discussed the effect of ambient temperature on the emissions of NO_x from diesel cars. Currently, UK emission factors used in the NAEI do not account for ambient temperature. As discussed in Grange et al. (2019) the presence of a temperature dependence on emissions of NO_x from light duty diesel vehicles will likely have several implications. Among the implications is the likely underestimate of wintertime emissions of NO_x in current emission inventories. Another implication of this work is that NO_x and NO_2 concentrations may decrease more quickly than previously thought because newer (Euro 6) vehicles have a smaller temperature dependence than the pre-Euro 6 vehicles they replace.

The NAEI estimates the separate contributions to non-methane volatile organic compound (NMVOCs) emissions from road transport from the tailpipe and from evaporative losses. The detailed methodology in producing these emissions estimates is reported in the UK inventory report (NAEI, 2019). The trends in emissions of NMVOCs from road transport taken from the version of the NAEI published in 2019 are shown in Figure 4.11. The individual vehicle type contributions are shown along with a cumulative figure for total evaporative losses from all road transport as a separate source.

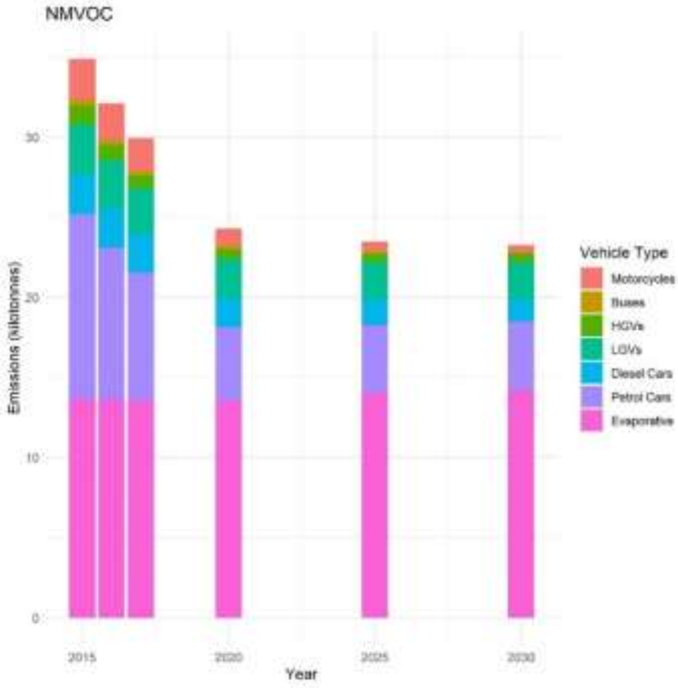


Figure 4.11 UK emissions of NMVOCs from road transport by vehicle type and projections to 2030 according to the NAEI (2019). Total emissions of NMVOCs due to evaporative losses of fuel vapour from petrol vehicles are shown for comparison.

Exhaust emissions of NMVOCs from road transport in 2017 were 16.4 ktonnes in 2017 according to the NAEI (2019), corresponding to 2.0% of total UK NMVOC emissions from all sources. A further 13.5 ktonnes come from evaporative losses from petrol vehicles on the road. Petrol cars are the dominant source of exhaust emissions (8.0 ktonnes in 2017), but these have declined rapidly since the 1990s (>98%) with the fleet penetration of cars with three-way catalyst from the early 1990s and the successive tightening of further Euro standards for new vehicles, supplemented by a gradual switch to lower NMVOC emitting diesel cars in the 2000s. Whilst petrol cars are still the dominant source of NMVOC emissions from road transport, the relative contribution of diesel vehicles has increased to 21%.

Exhaust emissions from road transport are predicted to decrease to 8.9 ktonnes by 2030, but will become dominated by evaporative losses as these are not predicted to change significantly according to the NAEI projections.

Whilst the tailpipe emissions of NMVOCs from diesel vehicles are estimated to be small compared to petrol according to the NAEI, some ambient measurements in London (Dunmore *et al*. 2015) have indicated that this may be something of an underestimate, although it is not clear whether higher emissions of longer chain hydrocarbons (e.g. C_{10} and above) from diesel vehicles arise from the tailpipe, from on-vehicle fuel evaporative losses, or indeed from wider fuel distribution and spillage.

Ammonia is emitted from road transport exhausts mainly as a by-product of catalytic processes in the exhaust aimed at reducing NO_x emissions. Exhaust emissions of NH_3 from road transport in 2017 were 4.5 ktonnes in 2017 according to the NAEI (2019), corresponding to 1.6% of total UK NH_3 emissions from all sources. The trends in emissions of NH_3 from road transport taken from the version of the NAEI published in 2019 are shown in Figure 4.12. Exhaust emissions have been dominated by petrol cars, but were more pronounced for early generation petrol cars with catalysts (Euro 1 and 2). Factors for later petrol vehicle Euro standards are lower according to the Emissions Inventory Guidebook due to improved catalyst systems and this has led to the downward trend in emissions shown in Figure 4.12. Emission factors for diesel vehicles have been much lower than for petrol vehicles, but factors for the later Euro 5/V and 6/VI vehicles are 2-4 times higher than for the earlier Euro counterparts due to emissions from SCR systems using NH_3 as a means to reduce NO_x emissions from the tailpipe. The NAEI estimates that diesel vehicles now contribute 27% of all road transport emissions of NH_3 in 2017. The NAEI currently predicts a small increase in future emissions of NH_3 from road transport mainly due to the increase in diesel vehicle emissions.

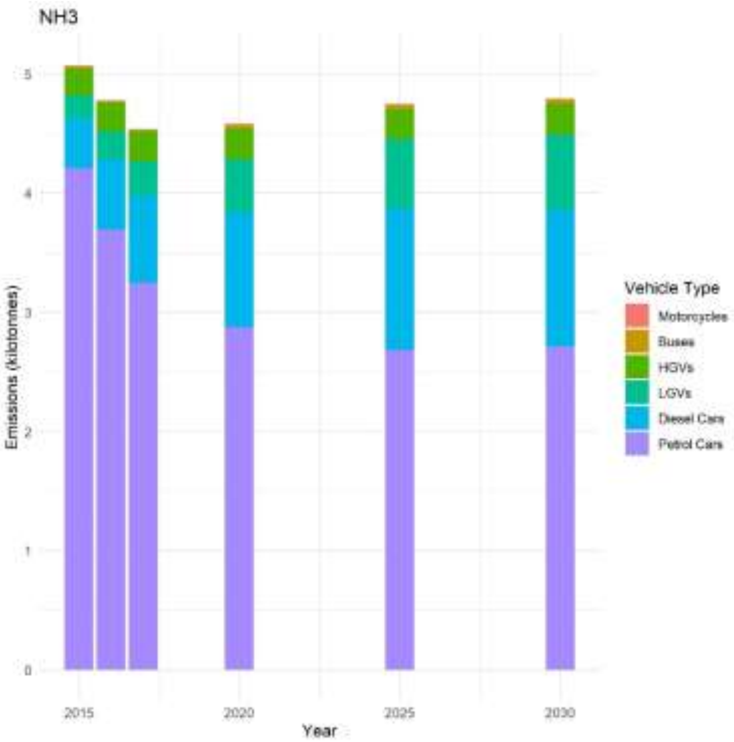


Figure 4.12: UK exhaust emissions of NH_3 from road transport by vehicle type and projections to 2030 according to the NAEI (2019)

4.6.8 The Spatial Distribution of Road Transport Exhaust Emissions

Emissions from road transport are mapped to the UK road network derived from the Ordnance Survey Open Roads using DfT's traffic count point data allocated to a section of the major road network. Traffic flow data are mapped as annual average daily flows for each type of vehicle and combined with fleet-weighted emission factors calculated for the inventory year for each vehicle type according to average speed of the road link. Further details of the method used for spatially mapping emissions from road transport are given in the NAEI mapping reports published each year (Tsagatakis et al, 2019). As stated earlier, whilst the emission maps account for local traffic flows for each main vehicle type according to traffic counts on each road link, the detailed composition of the fleets in terms of age and fuel mix are assumed to be the same everywhere according to the national fleet and no account is taken of any regional differences in the fleet composition apart from in London. The sum of emissions for the latest mapped inventory year are therefore consistent with the UK road transport emission totals. As would be expected, the emissions are highest in major conurbations and trunk roads connecting them.

Emission maps are generated by the NAEI each year after the national totals have been estimated. At the present time, the most recent year for which maps are available on the NAEI website is 2017 which will be consistent with the 2017 version of the NAEI. Interactive emission maps for road transport can be downloaded from the NAEI site at https://naei.beis.gov.uk/emissionsapp/. From here the road traffic statistics from the DfT website used in generating the maps can be accessed.

4.7 Uncertainties in Estimates of UK Exhaust Emissions.

A key question on emission estimations at the national or local level is their level of uncertainty and how confident we are in their 'correctness'. This issue can be considered in several ways:

- *Quantification of uncertainties in national emission estimates* - through a statistical uncertainty analysis and consideration of whether any of the input data (emission factors and activity) used are incorrect

- *Sensitivity tests* – quantifying how sensitive the overall inventory calculations are to each of the key input variables

- *Verification* – using independent datasets or results from other models to compare with the inventory results

- *Understanding the limitations in the approach used* – by consideration of alternative approaches.

4.7.1 Consideration of Inventory Uncertainties

The uncertainties in the NAEI for current/historic years are estimated using an approach described in the EMEP/EEA Emissions Inventory Guidebook which investigates the impact of the assumed uncertainty of individual parameters (such as emission factors and activity statistics) upon the uncertainty in the total emission of each pollutant. Details of the approach are given in the UK Inventory Report. For road transport, the greatest source of uncertainty is in the emission factors. The overall uncertainty in the vehicle kilometres activity data is relatively low according to DfT, although there is greater uncertainty in how these are distributed between the sub-vehicle categories, e.g. in the disaggregation of vehicle kilometres by Euro standard, fuel type and technology. Uncertainty regarding the vehicle fleet mix in the future is high, and the effects of new policies and public attitudes to vehicle choice, such as the rate of uptake of electric vehicles and a move away from diesel cars, may not be well represented.

Assessing uncertainties in emissions inventories for a particular sector is itself an uncertain undertaking, particularly when inventory compilers do not have access to the original data used to derive emission factors. The Guidebook provides a useful discussion around issues that lead to uncertainties in emission inventories for road transport and a set of qualitative precision indicators for individual pollutants and vehicle categories. It also refers to uncertainty estimates made using the COPERT model on inventories for countries with variable quality transport activity data. The uncertainty in the emission factors in a source like COPERT depends on the variability of the individual vehicle measurements for a specific vehicle operation (e.g. speed) which varies with pollutant and vehicle type. The distribution of values around the mean emission factor is considered to follow a log-normal distribution.

The Guidebook itself does not provide quantitative uncertainty estimates in emission factors. The NAEI uses its own expert judgement based on an understanding of the scatter in the data used to define typical speed-emission factor relationships and combines this with consideration of uncertainties in the fleet composition and speeds on UK roads. For NO_x, uncertainties in the emission factors of 35% are estimated for petrol cars and 60% for diesel cars, vans and HGVs. For PM, uncertainties in the emission factors of a factor of 2 are estimated for petrol cars, 40% for diesel cars and 60% for HGVs. For VOCs, uncertainties of 35% in the emission factors are estimated for petrol cars, 40% for diesel cars and vans and 60% for HGVs. These uncertainty estimates are used with uncertainty estimates in activity data for road transport and with corresponding uncertainty estimates for other sectors in the inventory to derive an overall estimate of uncertainty in the emission totals.

Further details on assessing uncertainties in the UK inventory are provided in the NAEI's Informative Inventory Report (NAEI, 2019).

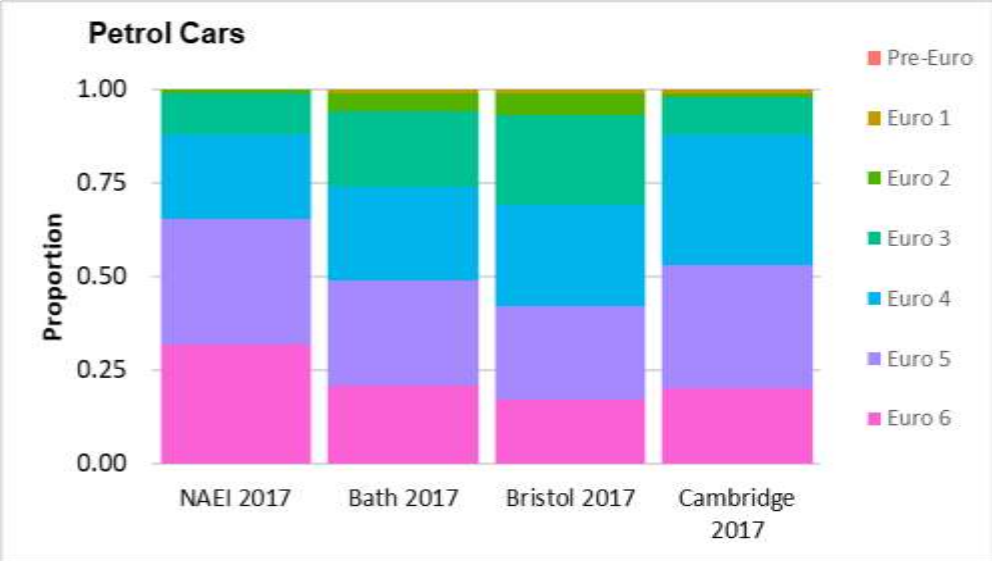
The overall assessment of uncertainties in emissions from the road transport sector at a national level, as calculated by the NAEI, should not be taken to be a measure of uncertainties in emissions at a particular location and moment in time. These are expected to be greater because of the variability in emissions in time and place by individual vehicles according to the specific road and traffic situation the vehicle is in and variability in driving styles as well as variability and uncertainty in the vehicle fleet (e.g. in terms of vehicle model, Euro standard and technology) and their state of operation in any particular time and place. This issue is important to appreciate when comparing inventory data from the NAEI based on nationally and annually averaged information against ambient measurements. The point was discussed in the AQEG report "*Linking Emission Inventories and Ambient Measurements*" (AQEG, 2013).

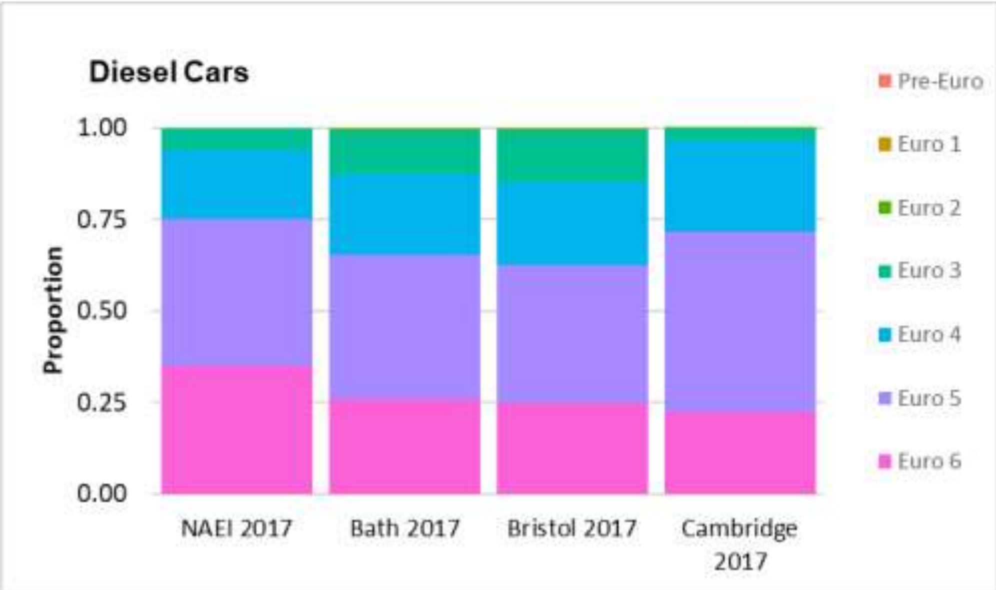
4.7.1.1 How Representative is the Fleet Composition?

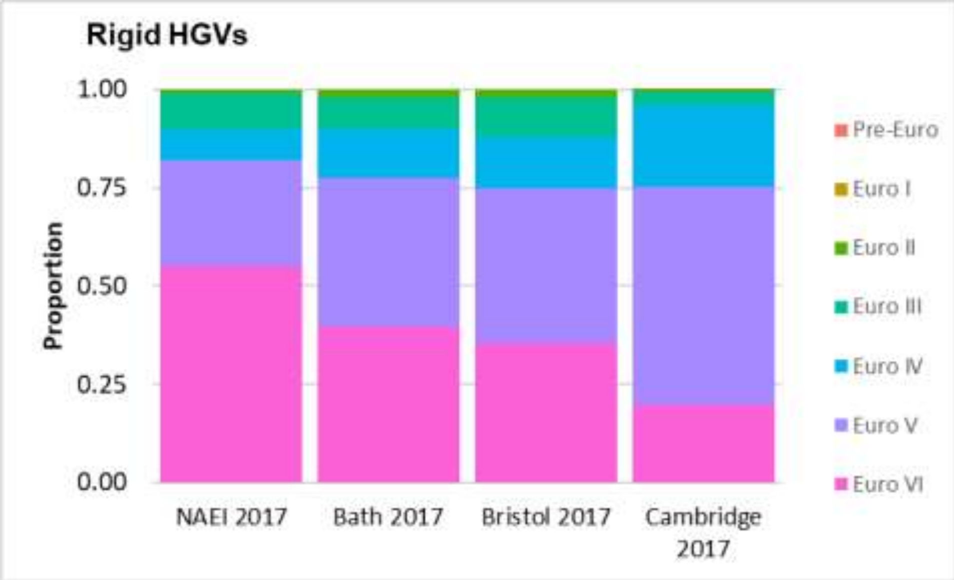
As previously discussed, the NAEI uses national datasets from DfT to define the composition of the vehicle fleet on the roads. The NAEI's national emission estimates and maps do not take into account localised fleet information, except for London. Work is currently exploring the potential for ANPR data provided by some local authorities to help improve air quality modelling done under Defra's Pollution Climate Mapping programme.

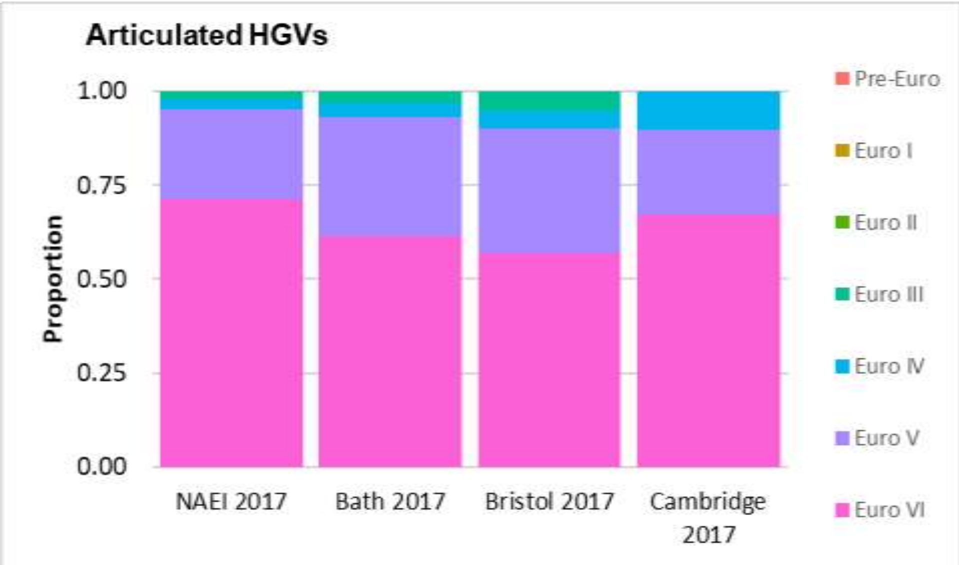
The NAEI currently uses ANPR data from DfT to determine the diesel/petrol car mix on different road types (classed as urban, rural or motorway). Data are provided by DfT grouped by region which does show differences in the fuel mix. These data are not used at this level of detail in the NAEI because of insufficient ANPR data sampled for each road type in each region and instead the data are grouped together and only analysed for differences by road type. Nevertheless, the data do show that the diesel car share on urban roads varied from 41.6% to 47.3% in different parts of England in 2017. The significantly higher diesel share in Northern Ireland is accounted for in the NAEI on the basis of licensing data for this country which shows 56.4% of cars were diesel-powered in 2017.

DfT's ANPR data also show differences in the share of each Euro standard by region. For example, according to the 2017 ANPR data from DfT, the share of Euro 6 classes of diesel cars on urban roads varied from 31% to 45%, but again based on a relatively small sample in each particular region. Plots showing how the Euro category fleet mix varies between some urban cities and from the national average represented by the NAEI can be extracted from publicly available CAZ reports, as can be seen in Figure 4.13 for each main vehicle type for Bath, Bristol and Cambridge. All these cities tend to show a lower share of Euro 6/VI vehicles than the nationally-averaged data from DfT.


Petrol Cars
Proportion
1.00
0.75
0.50
0.25
0.00
NAEI 2017
Bath 2017
Bristol 2017
Cambridge 2017
Pre-Euro
Euro 1
Euro 2
Euro 3
Euro 4
Euro 5
Euro 6


Diesel Cars
Proportion
1.00
0.75
0.50
0.25
0.00
NAEI 2017
Bath 2017
Bristol 2017
Cambridge 2017
Pre-Euro
Euro 1
Euro 2
Euro 3
Euro 4
Euro 5
Euro 6


Rigid HGVs
1.00
0.75
0.50
0.25
0.00
Proportion
NAEI 2017
Bath 2017
Bristol 2017
Cambridge 2017
Pre-Euro
Euro I
Euro II
Euro III
Euro IV
Euro V
Euro VI


Articulated HGVs
1.00
0.75
0.50
0.25
0.00
Proportion
NAEI 2017
Bath 2017
Bristol 2017
Cambridge 2017
Pre-Euro
Euro I
Euro II
Euro III
Euro IV
Euro V
Euro VI

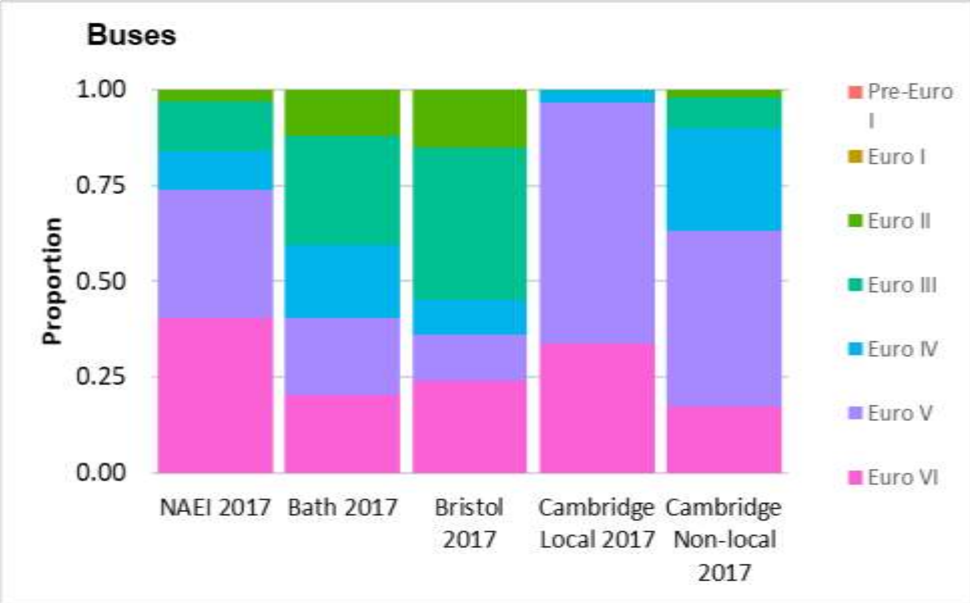


Figure 4.13 Plots showing variation in Euro category fleet mix between Bath, Bristol and Cambridge and from the national average represented by the NAEI for 2017. Data for each city extracted from publicly available CAZ reports.

Over the years, the DfT's ANPR data has consistently shown that the share of diesel cars is higher on motorways than on urban roads and this is accounted for in the NAEI. However, it will be increasingly important to understand local differences when evaluating the effectiveness of policies aimed at restricting vehicles according to fuel type and Euro classification.

Variations in the vehicle fleet composition also relate to understanding the true extent of environmental injustice when it comes to which groups of the population are exposed most to air pollution from traffic and which groups contribute most to it. A study by Barnes et al (2019) considered this issue based on the analysis of exposure to air pollution and vehicle ownership and usage in different socioeconomic and demographic groups. The study examined annual mean ambient background concentrations of NO_2 and PM and vehicle ownership and mileage data from MOT vehicle inspection test records. It found that households with the highest levels of poverty are exposed to the highest levels of NO_2 and PM concentrations, but contribute the lowest amount of emissions due to a) lower car ownership, b) less likely to own a higher polluting diesel car, and c) lower mileage than more affluent population groups. This implies that the higher traffic pollution in poorer areas is more likely to be caused by those living in those more affluent areas. Although the Barnes et al study does not consider *where* the emissions occur, rather which population groups are responsible for them, the variation in vehicle ownership and mileage by different population groups does suggest that a variation in the fleet composition in different urban areas may be expected, although dampened by the fact that vehicles are not used only where they are owned.

4.7.1.2 How Representative are the Emission Factors?

The representativeness of the COPERT emission factors has been brought into question by various studies using PEMS testing, roadside remote sensing and air quality modelling (see for example Carslaw and Rhys-Tyler, 2013, Hood et al 2018). The studies have shown that the values of NO_x emission factors for diesel cars in COPERT are not fully representative of real-world driving and may be underestimated, particularly for the Euro 4, Euro 5 and early Euro 6 categories, which may reflect the low number of vehicles tested and deficiencies in the average speed approach used (see Section 4.7.4). PEMS data for more recent Euro 6 models, conforming to Euro 6-temp standards suggest improved real world NO_x emissions and convergence with COPERT factors. Data from remote sensing suggests a similar trend and although measurements are usually confined to a limited set of driving conditions compared with PEMS, they have the advantage of capturing emissions from a much larger pool of vehicles and should be more representative of the fleet. Remote sensing can also highlight differences in emissions between Euro 6 diesel vehicle models using different technologies to reduce NO_x such as LNT and SCR, when these are manufactured to the same Euro emission standard, a feature that is not captured in COPERT. Remote sensing studies have also shown valuable information on the effects of ambient temperature on emissions.

COPERT and the factors in the EMEP/EEA Emissions Inventory Guidebook are designed for ease of use and comparability in national emissions inventory compilation. There will inevitably be a time lag between when further information on emission factors emerges and when this gets reviewed by the teams responsible for these centralised sources of emission factors, leading to updates in either the emission factors or the manner in which they are parameterised. For helping to understand air quality trends and in developing robust policies to reduce emissions and exposure to air pollution from traffic sources, there is a need to bring this information from PEMS and remote sensing into a common source of emission factors for inventory models more rapidly, and with greater flexibility to allow models to account for the variety of traffic situations and vehicle technologies that have been shown to affect emissions differently. This would need to be aligned with the availability of vehicle activity data at the national and local level.

4.7.2 Sensitivity Tests

Uncertainties in emission projections are best considered as a probable range according to maximum and minimum values of key input data such as traffic forecasts, fleet turnover assumptions including sales of new vehicles and their possible emission factors. It can be fruitful to test the sensitivity of an emission estimate derived from an inventory model to the likely range of each individual input parameter. This will show how important it is to get a particular parameter in the calculation ‘right’.

The NAEI road transport emissions model used to develop the current national inventory and projections was used to model the sensitivity of estimates of urban UK NO_x emissions from road transport when different model input parameters were varied. There are many variables in the model which affect the baseline emission estimates and therefore many possible scenarios that could be run, but the following scenarios were run as illustrative examples to show how sensitive the model results would be for emission estimates in 2017 and for 2030:

- A scenario where the share of diesel cars in the fleet in 2017 and 2030 is 10 percentage points higher than the baseline estimate of 44% in 2017 and 36% in 2030; this might be a case in a city with a particularly high share of diesel car activity.

- A scenario where the diesel car share of the UK fleet falls to 20% by 2030 as a consequence of a reduction in new diesel car sales. Baseline estimate is currently 36% for 2030.

- A scenario where the fleets of all vehicle types are older than the current national fleet estimates by assuming that the proportion of Euro 6/VI vehicles is 5 percentage points lower that the baseline estimates for 2017 and 2030, with the balance occurring in the Euro 4/IV and Euro 5/V fleets. The current fleets currently have the percentages in Table 4.2 assumed for Euro 6/VI vehicles in 2017 and 2030.

  A scenario where the average speed of cars and LGVs in urban areas is reduced from 35 kph to 25 kph; this scenario considers only the effect this has on emission factors according to the average speed emission factor relationships defined in COPERT, as used in the NAEI.

Table 4.3 shows estimates of the percentage difference in urban NO_x emissions for each sensitivity scenario relative to the baseline case. A positive number indicates an increase in the emission estimates.

Table 4.2: Percentage share of Euro 6/VI vehicles in UK fleet in 2017 and 2030 according to the NAEI base case scenario

Baseline Euro 6/VI share in NAEI	2017	2030
Petrol cars	34%	99%
Diesel cars	35%	98%
Diesel LGVs	34%	99%
Rigid HGVs	56%	99%
Artic HGVs	71%	100%
Buses	40%	98%

Table 4.3: Changes in estimates of urban UK NO_x emissions from road transport as modelled by the NAEI for different vehicle fleet and activity scenarios. The changes are shown as percentage differences from the NAEI baseline estimates for 2017 and 2030. A positive value indicates that the scenario would lead to an increase in emissions relative to the base.

Scenario	2017 Difference (%)	2030 Difference (%)
Increase diesel car share in fleet by 10%	+9%	+9%
Diesel car share in 2030 reduced to 20%	-	-14%
5% older fleet	+2%	+10%
Reduced urban speed for cars and LGVs	+10%	+11%

Section 3.4 showed how much variation there appears to be in the NO_x emissions from Euro 6b diesel cars by vehicle manufacture and technology (SCR and LNT) according to measurements from vehicle emission remote sensing (Fig 3.19 and 3.21). A factor of 10 difference is apparent by manufacturer. According to the NAEI, Euro 6 diesel cars contributed around 10% of all NO_x emissions from road transport in 2017 using current COPERT-based emission factors and national fleet data. Whilst the mix of different manufacturer models and diesel car technologies on the road is not known, it is quite evident from the differences in NO_x performances that variations in the mix will lead to variability in overall road transport emissions of NO_x. Moreover, this variability will increase as the share of Euro 6b diesel cars increases in the fleet.

These sensitivity test results summarised in Table 4.3 are approximate and illustrative and the scenarios themselves are not meant to convey the level of uncertainty in the input variables used in the national inventory, but they do indicate how sensitive emission estimates are to local differences in the vehicle fleet and emission factors. The sensitivity tests that were modelled are also not exhaustive, but they indicate how a range of factors will ultimately influence emissions locally.

4.7.3 Verification of Emission Inventories

There are various forms of inventory verification that can be undertaken. Some of these were discussed in the AQEG report "*Linking Emission Inventories and Ambient Measurements*" (AQEG, 2013) and included examining trends in roadside concentrations of primary pollutants and comparing ratios of pollutant concentrations and emissions where these are dominated by traffic sources. These methods have tended to show the relative changes in emissions of NMVOCs from traffic sources are represented quite well by the inventory, at least for the smaller hydrocarbons, although the possibility remains that inventories, whilst meeting the requirements of official inventory reporting, are not capturing some of the larger hydrocarbons from diesel exhausts (Dunmore et al, 2015). They also highlighted discrepancies with the trends in NO_x emissions which provided the initial evidence that there was something 'wrong' with the inventories in the late-2000s subsequently shown to be due to the underestimation of emission factors for diesel vehicles in earlier versions of COPERT and Guidebook sources, now at least partially addressed.

As explained earlier, despite COPERT being intended for national-level emissions inventories, COPERT-based emissions underpin most of the local-level air quality modelling carried out in the UK (i.e. modelling to predict concentrations at individual roadside locations). Most such studies include a comparison of predicted NO_x or NO_2 concentrations against measurements made using either chemiluminescence samplers or passive diffusion tubes (e.g. Defra, 2018). It is very often the case that such models under-predict measured NO_x and NO_2 concentrations in urban settings. Conversely, the same models often over-predict concentrations beside motorways or other fast-flowing roads. There are many reasons why local-scale models may over- or under-predict concentrations, one of which is the emissions data assumptions.

Concerns regarding the validity of model input assumptions, particularly in the context of historic disparities between projected and measured trends in NO_x and NO_2 concentrations, has led to work being carried out to compare vehicle-specific NO_x emissions from modern diesel vehicles driven in the real world against the assumptions within COPERT.

Marner et al., (2016) collated the results from published dynamometer, PEMS, TNO's Smart Emissions Measurement System (SEMS), and remote sensing data relating to early-model (pre-2016) Euro 6 diesel cars driven in real-world conditions. The authors noted substantial reductions in NO_x, on average, comparing these early Euro 6 vehicles with Euro 4 and Euro 5 variants but they nevertheless suggested that the then-current versions of COPERT (V4.10 and V4.11) over-predicted the improvements delivered by the first tranche of Euro 6 diesel cars. The authors concluded that NO_x emissions from Euro 6 diesel cars predicted using COPERT V4 should be uplifted by 60% to take account of bias in the model.

O'Driscoll *et al*., 2016 reported NO_x emissions measurements made in London using PEMS instruments. These results agreed with those presented by Marner *et al.,* (2016) in that the average under-prediction of NO_x emission in COPERT V4 was approximately 60%. O'Driscoll *et al*., (2016) also showed the significant variability in emissions of both NO_x and NO_2 between vehicles and between emissions in urban and extra-urban settings. This highlighted the limitations of using a single average-speed-based emissions factor to represent all vehicles and all settings.

COPERT V5.0 predicts higher average NO_x emissions from the first tranche of Euro 6 diesel cars than COPERT V4[15] (emissions factors for older vehicles were not substantially altered). Marner and Laxen (2017) showed that the predictions made using COPERT V5.0 were approximately the same as those derived by uplifting COPERT V4 by 60%. Thus, the average NO_x emissions for the first tranche of Euro 6 diesel cars predicted using COPERT V5.0 compare well with the independent real-world emissions tests collated by both Marner *et al.,* (2016) and O'Driscoll *et al.,* (2016).

The non-regulatory PEMS testing of passenger cars described by Marner *et al.,* (2016) and O'Driscoll *et al*., (2016) has continued in recent years and now includes Euro 6d-temp diesel cars. Online data from Allgemeiner Deutscher Automobil-Club (ADAC) in Germany and Emissions Analytics in the UK show large numbers of Euro 6d-temp diesel cars emitting less than 80 mg/km of NO_x as a drive-cycle average. This is much lower than the emissions assumed even from the post-2020 Euro-6 diesel cars in COPERT V5.0, which range from 150-300 mg/km depending on speed. It is thus possible that future emissions from these vehicles may be over-predicted, on average, by COPERT.

[15] COPERT Version 4.11.4 uses the same emissions functions as COPERT V5.0

Marner *et al.*, (2016) also reviewed evidence of real-world NO_x emissions from heavy duty vehicles. It was concluded that the independent measurements supported the assumption in COPERT V4 that NO_x emissions fell by more than 80% between the Euro III and Euro VI standards. Marner and Laxen (2018) went on to conclude that there is no strong evidential basis for deviating from the HDV NO_x emissions assumptions in COPERT V5 on a fleet-average basis.

Whilst not necessarily proving an inventory is correct, it can be fruitful comparing emission estimates and vehicle fleet and activity data used to derive them from different inventory sources. For example, comparing the exhaust emission estimates for London developed for the London Atmospheric Emissions Inventory (LAEI) with the emissions for London extracted from the spatially resolved NAEI. The LAEI is developed from more detailed and highly resolved traffic and fleet data than the NAEI so in principle should be closer to 'ground truth' and therefore provide a good test for the NAEI.

At the national level, the reliability of the NAEI approach and the assumptions and input data used can be judged by comparing total fuel sales figures as given in DUKES with total fuel consumption estimated using the same input data as used to estimate pollutant emissions. Since petrol and diesel sales figures in DUKES exclude biofuels, it is necessary to exclude these from the fuel consumption calculations to enable a like-for-like comparison. For 2017, the 'bottom-up' estimate of fuel consumption underestimates petrol and diesel consumption by 8% and 6%, respectively, compared with fuel sales figures from DUKES, but across the whole time-series from 1990 the maximum deviation from DUKES is 16%. The relative difference between estimates of fuel consumed and fuel sold is used to scale the 'fuel used' version of the air pollutant emissions inventories to derive the 'fuel sold' inventories referred to previously and required for compliance with international inventory reporting guidelines.

There are some difficulties when making such a comparison between fuel sales data and consumption estimates due to fuel purchased overseas and used in the UK and vice versa. This may be significant in the case of fuel consumed by HGVs, though being an island state, this 'fuel tourism' effect is probably less of an issue for the UK than in other countries in mainland Europe. Overall, the agreement between calculated estimates of fuel consumed and official statistics on fuel sold indicates some confidence in the inventory approach at UK level. However, given that fuel consumption factors for individual vehicle types may be known with greater certainty and perhaps show less variability within a specific vehicle category than pollutant emissions, this agreement should not necessarily be taken as an indication of the uncertainty in emissions of air pollutants.

4.7.4 Limitations of an Emissions Modelling Approach

Apart from uncertainties in the key input parameters themselves, the underlying emissions modelling approach ultimately limits the accuracy of an emission estimate. There are inherent difficulties and complexities in calculating road traffic emissions to a high level of accuracy because of the complex sensitivities to operating conditions and vehicle technologies. As previously explained, the particular purpose of an emissions model or inventory needs to be taken into account and the approach used should ideally match the spatial and temporal scale to be modelled. The average speed approach used in the NAEI is considered to be valid for a national inventory and over a network of roads within an urban area larger than approximately a half square kilometre, but less so at specific road links and junctions. For local scale modelling, instantaneous emission modelling approaches may be more valid, at least conceptually. However, microsimulation models themselves have their limitations and, particularly taking account of the increased requirement for predictive traffic data, may not improve simulations over a coarser resolution model. Predictive air quality modelling using emissions derived from the average speed approach is routinely carried out for many purposes, including modelling and assessments for Local Air Quality Management (LAQM) and there are several reasons for this which will be discussed in Section 4.7.5.

Several studies have done a comparison of NO_x concentrations modelled using average speed vs instantaneous emission factors. Marner et al. (2014) used the S-Paramics micro-simulation traffic model, the AIRE instantaneous emissions model, and the ADMS-Roads dispersion model to predict annual mean NO_x and NO_2 concentrations around a single road junction. For each turning movement, the predicted emissions were aggregated by hour, and by 2 m link segment. These 2 m link segments were then used as inputs to the dispersion model in order to predict annual mean NO_x concentrations. The same approach was then repeated using the average-speed based emissions factors in Defra's EFT[16], with average speeds calculated for each hour and for each 2 m link segment. Figure 4.14 compares the predictions derived using each model. There is a much greater range in the concentrations predicted using the instantaneous emissions model. This conforms with expectation, since the EFT can only predict one emission rate for any given average speed, while AIRE can predict a range of emissions at any speed depending on the concurrent acceleration or deceleration. The comparison suggests that EFT-based average-speed model predictions will under-predict concentrations at some locations and over-predict elsewhere, but there is no obvious bias on average between the two models.

The authors went on to investigate some individual differences between the two emissions models. Figure 4.15 shows a line of receptors running approximately along the back of the pavement moving away from the junction. It then compares the predictions made using the AIRE instantaneous emissions model with those using the EFT average speed model. The authors suggested that the road-link-specific differences in Figure 4.15 can be explained by how the same average speed can be achieved by very different driving conditions. For example, the average speeds for each 2 m section of Link 10 (Figure 4.15), which is inbound to the junction and not signal-controlled, are typically achieved through a relatively high degree of deceleration; leading to the EFT over-predicting emissions. Conversely, the

[16] EFT V5.2 which was based on COPERT V4.8.

average speeds along Link 18, which is also inbound to the junction but which approaches traffic light signals, experiences more acceleration than Link 10, leading to the EFT under-predicting emissions.

The authors note that the 'flattened' shape of the scatter plot in Figure 4.14 (i.e. with more variability on the horizontal axis than on the vertical axis) reflects a common observation when comparing average-speed-based model predictions with ambient NO_x and NO_2 measurements. This suggests that average-speed-based modelling might routinely miss the maxima and minima of localised NO_x emissions. Modelling based on average-speed emissions factors is unlikely to fully reflect the spatial heterogeneity in NO_x and NO_2 concentrations on a local scale. However, as stated earlier, there are also uncertainties in the ability of microsimulation traffic and emission models to predict these spatial heterogeneities.

The study by Marner et al., (2014) considered the likely effects of different traffic management options, involving a 20 mph speed limit, changes to lane configuration, and altered traffic light phasing. Each option resulted in localised changes to predicted acceleration profiles at different points on the network and at different times of the day. These changes resulted in some very large (>10 mg/m^3), but highly-localised, predicted changes to annual mean NO_2 concentrations. The authors noted some significant uncertainties regarding both the traffic model and the emissions model[17] but concluded that there was a clear potential to reduce overall emissions from the junction and, perhaps more significantly, to displace peak emissions away from locations with nearby human exposure. It was not possible to validate the findings of the study with respect to changes in concentrations because none of the tested options were implemented.

[17] In particular, treatment of acceleration within microsimulation traffic models is uncertain. Also, and as noted in Table 4.1, the AIRE emissions model uses simplified outputs from a 2005 version of the PHEM model and can thus be considered inexact.

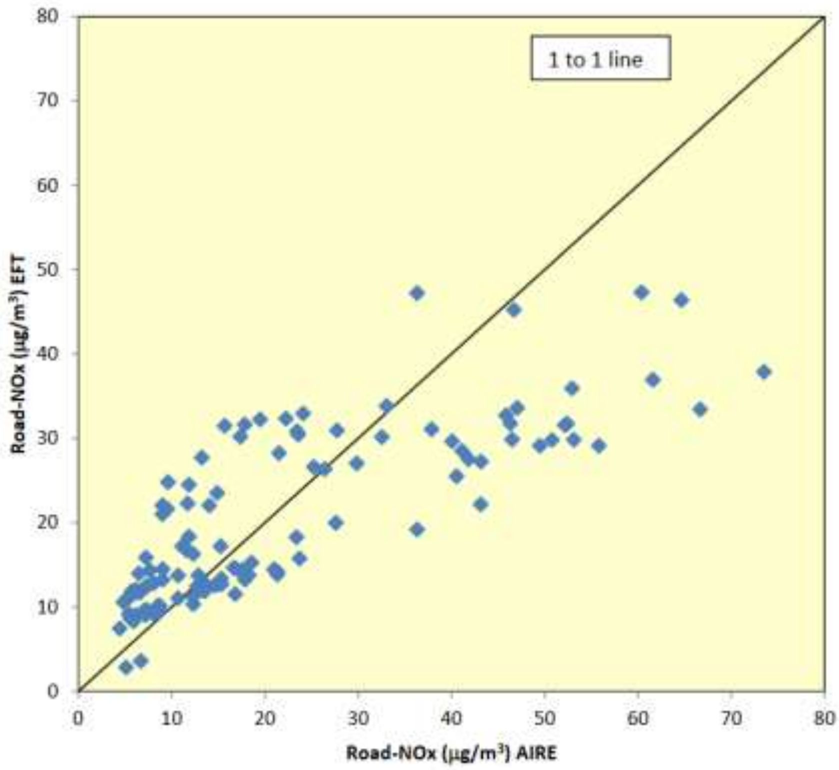


Figure 4.14: Predicted Annual Mean Road-NO_X Based on AIRE Instantaneous Emissions Model vs EFT Average Speed Emissions Model. Taken from Marner et al. (2014)

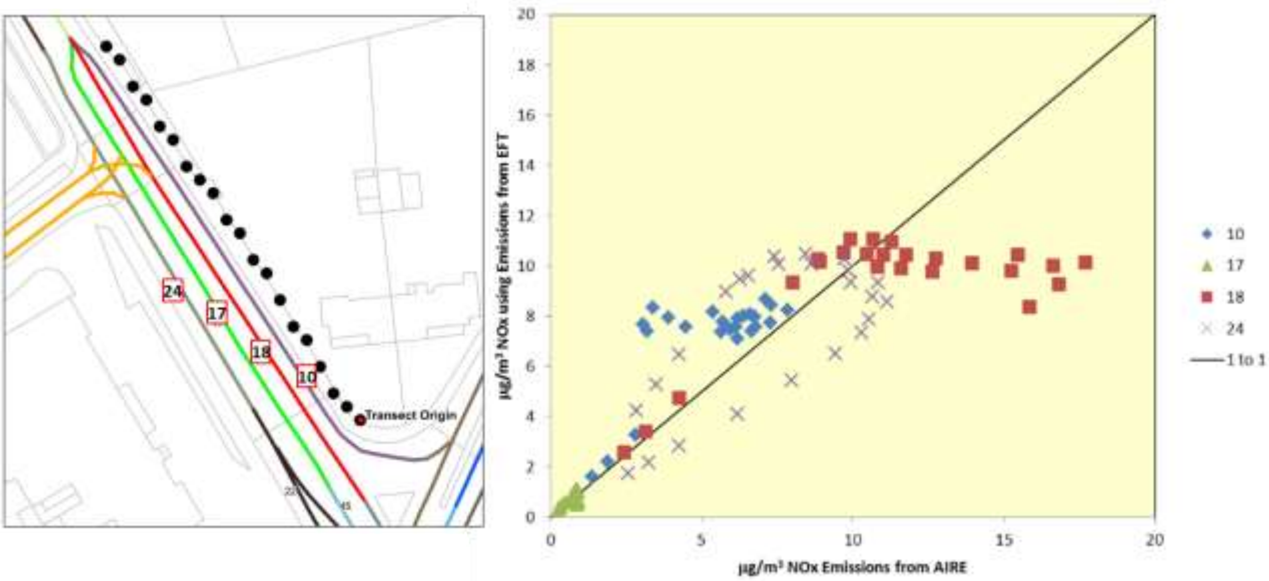


Figure 4.15: Predicted Annual Mean Road-NO_X Concentrations Attributed to Four Traffic Lanes Based on AIRE vs EFT (Both emissions models treated each 2 m road section separately – emissions were not aggregated by lane). Taken from Marner et al. (2014)

Williams and North (2013) also used the AIRE emissions model, but with GPS-derived measured driving profiles. Six routes (3-4 km) in London were driven multiple times to generate instantaneous speed and acceleration data, which were then separated according to the speed limit on each road section (20 mph or 30 mph). The calculated NO_X emissions were validated against PEMS measurements.

On many of the routes surveyed, factors other than speed limits kept driving speeds low. Nevertheless, roads with 20 mph limits tended to experience less overall speed variation and less time spent accelerating and decelerating (>0.9m/s^2). It was concluded that the imposition of a 20 mph speed limit was likely to have mixed effects, increasing NO_X and/or PM_{10} emissions from some vehicles and reducing them from others.

Williams and North (2013) also investigated the effect that specific road features, such as speed bumps and pedestrian crossings, had on speed, acceleration, and predicted emissions. They showed that links with vertical deflection (speed bumps, cushions, and raised junctions) exhibited greater speed-variability than links without these features. Thus, streets where traffic flows were more likely to be interrupted had higher calculated emissions. For example, the authors compared link sections which were similar apart from the method of traffic calming and showed 64% to 98% higher NO_X emissions for the link with speed bumps (i.e. which traverse the whole carriageway) than speed cushions (which cover only part of the carriageway). The magnitude of such effects would not be shown using average-speed-based emissions models.

Both the Marner et al. (2014) and Williams and North (2013) studies suggested that 20 mph speed limits have the potential to reduce roadside NO_2 concentrations compared with 30 mph limits. This is the opposite conclusion that would be reached using average-speed EFT emissions factors which always predict higher emissions at 20 mph than at 30 mph. This demonstrates the limitations of relying on the average-speed emissions factors to demonstrate the effects of changes in driving characteristics.

The modelled vehicle speed and acceleration data used by Marner et al. 2014 are uncertain. Similarly, the GPS-derived speed and acceleration traces used by Williams and North (2013) were from an individual vehicle and will thus be influenced by individual driver behaviour. The increasing adoption of on-board telemetry within vehicles can provide a much larger dataset of vehicle/driver behaviour.

4.7.5 Using Average Speed Approach in Modelling for Local Air Quality Management

Predictive air quality modelling is routinely carried out for purposes including Local Air Quality Management (LAQM), development-control, and informing the design of transport interventions. As mentioned earlier, instantaneous emission modelling approaches would account for the second-by-second variability in emissions from an individual vehicle at a specific location, but almost all predictive air quality modelling for LAQM and assessment of policies and interventions relies on average-speed emissions factors, with instantaneous emissions factors seldom used. There are a number of reasons for this and why the average speed approach is suitable for this purpose:

a) *Availability of suitable traffic data:*

Using instantaneous emissions factors requires a level of detail in traffic data that cannot be taken from typical traffic counts or transport models. While the required parameters can be measured (for example by on-board telemetry), in the context of predicting the effects of future developments or interventions, a microsimulation traffic model is required.

The primary purpose of most traffic models is transport planning and this means that they are often not ideally configured to inform air quality modelling. For example, traffic models will frequently be run for peak-hour periods only, when the metric of primary interest to the air quality assessment is annual mean concentrations. For average-speed-based emission calculations, representative values for the remaining periods of a day can often be estimated reasonably well from diurnal profiles measured elsewhere. The nature of microsimulation traffic models is that the relevant data cannot readily be extrapolated across other periods of the day making it impossible to predict annual mean concentrations. Other issues linked with air quality modelling not being the primary purpose of the traffic model are that the spatial extent of microsimulation traffic models is often smaller than that of air quality models, and that microsimulation traffic models can often be configured to provide outputs on a relatively low resolution (e.g. providing averages between two junctions rather than details at a position close to a junction).

Finally, a key area of uncertainty for many microsimulation traffic models is their ability to predict acceleration profiles. This is also the parameter which links most closely with emissions (i.e. high-acceleration events correlate with high-emission events). There is also a relative paucity of studies where emissions calculated from microsimulation traffic models have been validated against measurements and so air quality modellers may often be sceptical of this type of modelling.

b) *Availability of Emissions Factors*

As explained in Section 4.3.1, instantaneous emissions factors can be taken directly from the PHEM model but PHEM does not contain data for all existing vehicles and cannot contain data for future vehicles. Because microsimulation traffic models can generate large quantities of data and because using PHEM directly can be complex, some emissions models have been configured to link directly to the microsimulation traffic model. However, this introduces additional delays in providing a comprehensive vehicle fleet (for example the AIRE model does not include emissions data for Euro 5/V or later vehicles). By contrast, because the emissions factors in COPERT are more generic, they cover all current and future generations of vehicles.

c) Cost

While microsimulation traffic models can be configured to overcome the issues noted above, doing so is often prohibitively expensive. Using microsimulation traffic data to robustly predict annual mean concentrations then requires expertise and resources which are often not available to many air quality assessments.

d) Guidance and consistency

As well as requiring more time and expertise to use than average-speed emissions factors, instantaneous emissions factors carry a risk of introducing errors when modellers are not familiar with their use. Defra's EFT is produced specifically for LAQM and Defra's guidance to local authorities on LAQM is written from the position that average-speed emissions factors will be used. Similarly, guidance to local authorities developing Clean Air Zones issued by Defra and DfT's Joint Air Quality Unit (JAQU) is that the EFT should be used. Guidance from Highways England related to highway projects favours the use of Highways England's speed band emissions factors but, as explained in Section 4.4 these have been derived from the EFT and so are still, ultimately, average-speed emissions factors.

Air quality modelling carried out for development control purposes typically seeks consistency with other regimes since this tends to simplify the process of gaining permission for new development. The additional costs, both for modelling and also for expected challenges during the planning process, mean that instantaneous emissions factors are seldom used.

As shown in Section 4.7.4, the use of average-speed emissions factors may miss localised maxima and minima but is unlikely to introduce significant bias on average. In situations where the focus of assessment is changes in traffic volumes, or changes in fleet composition, then the degree of error introduced by the average-speed approach is likely to be smaller than if changes to junction design or congestion levels are being considered.

4.8 Summary and Conclusions on Modelling of Vehicle Exhaust Emissions

- There are inherent difficulties in calculating road traffic emissions to a high level of accuracy. Different approaches are available for modelling exhaust emissions and the approach used in a model needs to be consistent with the purpose for which the emission estimates are intended to be used. Many national inventories like the NAEI use simple approaches based on average speed that do not show the variability in emissions evident with different driving conditions. Nevertheless, on average, results on NO_x emissions from Euro 6 diesel cars from PEMS and remote sensing measurements agree reasonably well with the latest average speed factors from sources such as COPERT used in inventories, although emission factors for the more recent Euro-6d-temp vehicles may be overestimated.

- Instantaneous vehicle emission modelling approaches exist that better capture the variations in emissions with speed, acceleration and vehicle specific power. Conceptionally, these are better suited for more local scale modelling and on individual road sections where higher spatial and temporal resolution in emissions are required, but are often not practical to apply, particularly on a national scale covering a longer time-series (e.g. from 1990-2030) and there remain limitations and uncertainties in the microsimulation traffic and emission models themselves. When used with detailed vehicle movement data, instantaneous emissions models tend to show that streets where traffic flows are more likely to be interrupted have higher emissions than would be shown using average-speed-based emission models.

- Overall, predictive air quality modelling using emissions derived from the average speed approach is currently considered suitable for modelling and assessments for Local Air Quality Management (LAQM) for reasons of practicalities with the availability of suitable traffic activity data and emission factors and with cost and consistency considerations.

- In spite of the fairly continued growth in traffic since the 1990s, changes in fleet composition with the penetration of vehicles meeting more stringent Euro standards have led to a fall in pollutant emissions and this downward trend is expected to continue to 2030. According to the version of the NAEI published in 2019, NO_x emissions are predicted to decrease by 65% by 2030 relative to 2015 levels, but remain dominated by diesel cars and LGVs (79%). Exhaust emissions of PM are also expected to fall by 80% by 2030 and overall PM emissions from road transport will be dominated by non-exhaust sources.

- There are likely to be local differences in the fleet which are not at present fully reflected in the NAEI and the PCM model used for compliance reporting under the EU Ambient Air Quality Directive. These differences will be important in understanding current emissions locally and quantifying the effect of local policies restricting vehicle movements according to fuel type or Euro class. Greater access to local fleet data from ANPR sources or vehicle ownership will enable more accurate local inventories to be developed.

- Emission measurements by remote sensing have shown a difference between vehicles complying to the same Euro standards, but using different control technologies, e.g. LNT and SCR. These differences are not reflected in inventories and there is greater need for inventories to use a more granular approach in activity data and emission factors reflecting technological differences and also dependencies on environmental factors such as ambient temperature. There is also an increasing need for emission factors for more advanced and emerging vehicle technologies and powertrains coupled with accurate predictions in the fleet to make better forecasts in future emissions.

- There is often a time lag between when emission measurements are made by the research community and when they get used in inventories. With an increasing amount of research on real-world emissions using PEMS and remote sensing, there is a need to bring this information into peer-reviewed and publicly available sources of emission factors and models more quickly and with greater flexibility to allow models to account for the wide variety of traffic situations and vehicle technologies, aligned with the availability of vehicle activity data at the national and local level. This will allow a more consistent assessment of current emission estimates and development of policy options for reducing emissions.

5 AMBIENT MEASUREMENTS OF EXHAUST POLLUTANTS

5.1 Introduction

Across Europe and many parts of the developed world, legal limits are set for ambient air pollution. In response, regulatory measurement networks are operated by local, city and national authorities to determine compliance. The latest report from the European Environment Agency (EEA,2019) focuses on measurements made in 2017 and shows widespread breaches of EU, limit and target and values for NO_2 and PM across the UK and Europe. Proximity to traffic sources was a major factor in breaches for many pollutants, most especially NO_2.

Recently large-scale data analysis techniques have allowed data to be looked at across city, national or international measurement networks. Rather than focus on measurements from single monitoring sites this chapter tries, wherever possible, to focus on network-wide analysis. By tracking the changes over time, and contrasts between locations with different vehicle mixes, it is possible to detect changes in emissions as fleet technologies change; determining overall trends and also identifying areas where policy interventions are not being effective. These findings can be used to investigate areas or conditions where technology interventions are working optimally and areas or conditions where abatement technology is not performing as expected.

5.2 Using ambient air pollution networks to quantify exhaust emissions

Regulatory measurement sites are typically located in areas with public exposure; in residential areas or close to roads and industry, but networks are not generally optimised to detect air pollution from specific sources or the way in which they change in response to policy (Fuller and Font, 2019).

It can therefore be challenging to use regulatory measurements to assess air pollution from traffic. Measurements made close to roads are often used since traffic is often the dominant source of many pollutants such as NO_x and NO_2 in these locations, but this is not the case for PM_{10} and $PM_{2.5}$ where a large regional or urban component can dominate concentrations, even close to the busiest roads. For this reason, it is often necessary to consider the increment between roadside and background concentrations (Lenschow et al 2001) or alternatively use tracer pollutants from vehicle exhausts to quantify the traffic contribution (Fuller and Green, 2006). For instance, the traffic increment represented on average ~35% and 18% of the total PM_{10} concentration measured at the roadside locations in Paris and London, respectively between 2005 and 2016 (Font et al 2019).

This difficulties of assessing source changes through regulatory networks is made more difficult by changes in network designs. With a focus on determining compliance, measurement networks are often in a state of continuous change. Stations in compliant areas tend to be closed and new ones opened in areas of suspected high concentrations. This preferential sampling leads to biases in any network average or trend (Shadick and Zidek, 2014) and a lack of long-term datasets. Despite the importance of NO_2 measurements for legal compliance, just eight urban areas in the UK had consistent roadside measurements of NO_2 between 2000 and 2017 (Lang et al 2019), severely limiting opportunities for trend analysis and policy feedback. With the notable exception of Marylebone Road, roadside measurement sites do not have long-term co-located data sets of traffic measurements. Although the new NERC supersites will greatly enhance the UK's urban air pollution measurements, these are located in background locations and not designed to specifically focus on traffic and exhaust pollutants.

Measurements from regulatory measurement sites focus on the fleet of vehicles that are already present on our roads and only provide information on new vehicles as they enter service. However, in contrast to emissions modelling or exhaust measurements, data from ambient networks show more clearly how exhaust emission changes affect our outdoor air pollution.

5.3 NO_x and NO_2 trends in the UK 2005 to 2018

Laxen et al. (2019) analysed trends in ambient NO_x and NO_2 from the UK's Defra's Automatic Urban and Rural Network (AURN) and regional networks of chemiluminescence monitors from 2005 to 2018. A total of 112 measurement sites were selected based on data capture; with a focus on data capture during the four years at the start and end of the period. Figure 5.1 shows a relatively steady reduction in NO_x concentrations over this entire period at all site types, with linearity in the average trend. NO_2 concentrations at rural sites also reduced following a linear trend. However, roadside NO_2 shows a more complex pattern, with smaller average reductions before 2010, and steeper falls thereafter. This most likely reflects changes in primary NO_2 emissions over this period; with NO_2 increasing relative to NO_x up to around 2010, and reducing thereafter (see section **Error! Reference source not found.**).

Temporal trends were therefore analysed separately for 2005 to 2018 and the period 2010 to 2018 using the TheilSen method. Focusing on the shorter period also allowed the inclusion of an additional 70 monitoring sites. Table 5.1 shows a significant downward trend in both NO_x and NO_2 at all site types over both periods, an average reduction around 2-3% per year since 2005, with a steeper fall of 3-4% per year since 2010. The reductions in NO_x and NO_2 were quite similar to one another, but there is some evidence in Table 5.1 of steeper falls for NO_2 than for NO_x at roadside sites. The largest reductions, for both NO_x and NO_2, tended to be at rural sites; although there were appreciably fewer such sites in the analysis. This might suggest that reductions in non-traffic sources; notably changes to the UK power generation, have had a greater relative effect than changes to vehicle emissions. It should be stressed that if the trends were expressed in terms of concentrations, rather than percentages, then the changes to rural sites would, by definition, be much smaller.

Figure 5.1 and Table 5.1 comprise averages across all UK sites. Individual sites showed considerable heterogeneity in trend. This is disaggregated, for the period 2010 to 2018 in Figure 5.2. This shows, for example, that there have been no significant reductions in NO_2 concentrations at 15% of the sites but changes greater than 4% per year at 17 % of sites. Geographically there was no clear spatial pattern in the rate of change across the UK.

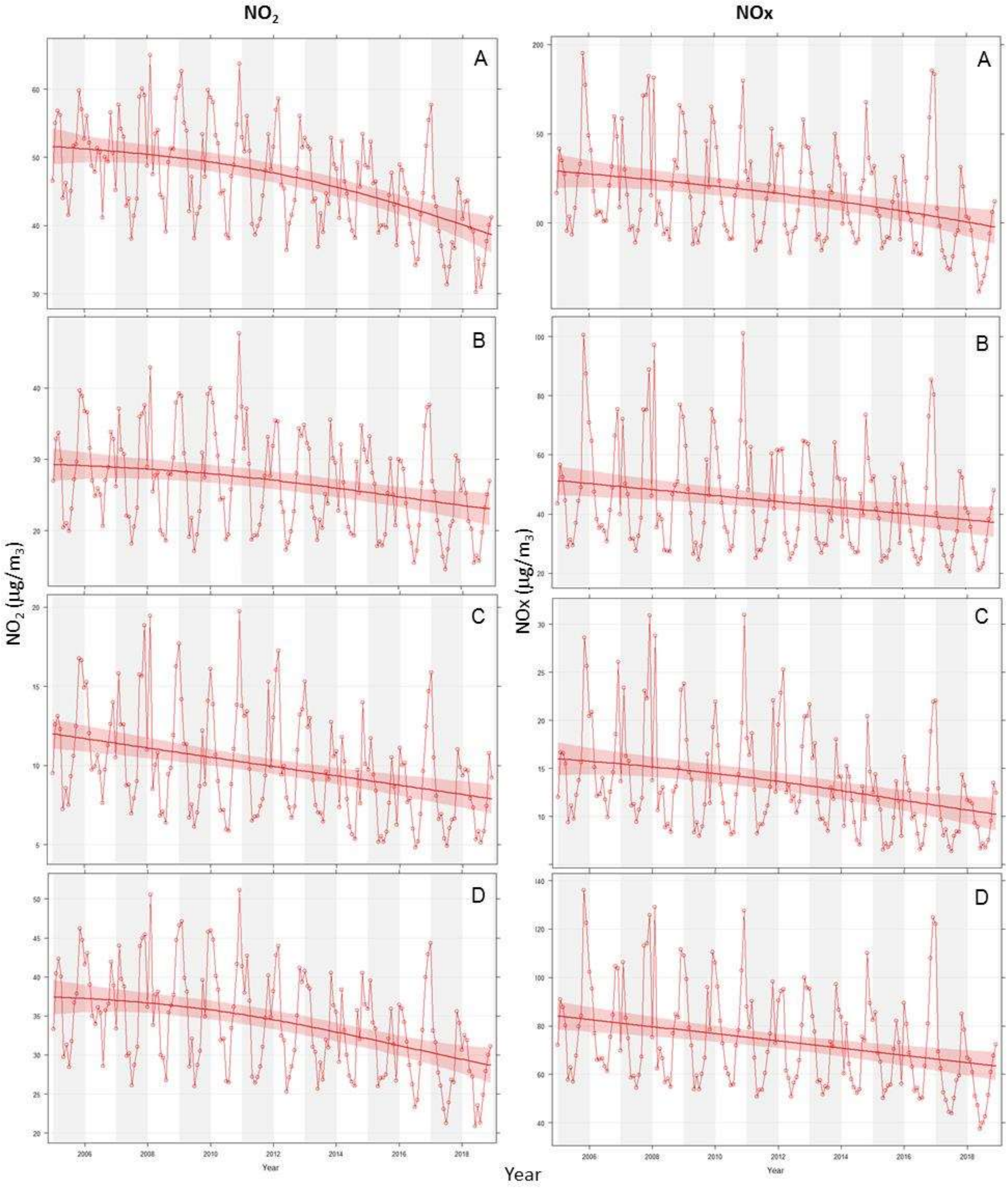


Figure 5.1 Overall NO_2 and NO_x Smooth Trend Fit 2005 – 2018: A) Roadside Sites (n = 52), B) Urban Sites (n = 45), C) Rural Sites (n = 15), and D) All Sites (n = 112). Monthly averages calculated from the hourly concentrations. Generalized Additive Model smoothed trend line

fitted to the monthly data, also showing 95% confidence interval (Laxen et al. (2019). Calculated using Openair software (Carslaw and Ropkins, 2012)

Table 5.1 Results of ThielSen Analysis of Monthly-mean Concentrations at All Relevant UK Sites over 2 Time Periods (Mean Trend is Shown, with the Significance Given as: *** p=0.001; ** p=0.01; * p=0.05). (Data from Laxen et al. (2019))

Site Grouping	Period 2005 - 2018			Period 2010 - 2018		
	Number of Sites	Mean Trend (%/yr)		Number of Sites	Mean Trend (%/yr)	
		NO_2	NOx		NO_2	NOx
All Sites	112	-1.82***	-1.86***	182	-3.13***	-3.07***
Roadside Sites	52	-1.80***	-1.74***	109	-3.10***	-3.02***
Urban Sites	45	-1.65***	-1.99***	57	-3.09**	-3.08*
Rural Sites	15	-2.46***	-2.54***	16	-3.41***	-4.09***

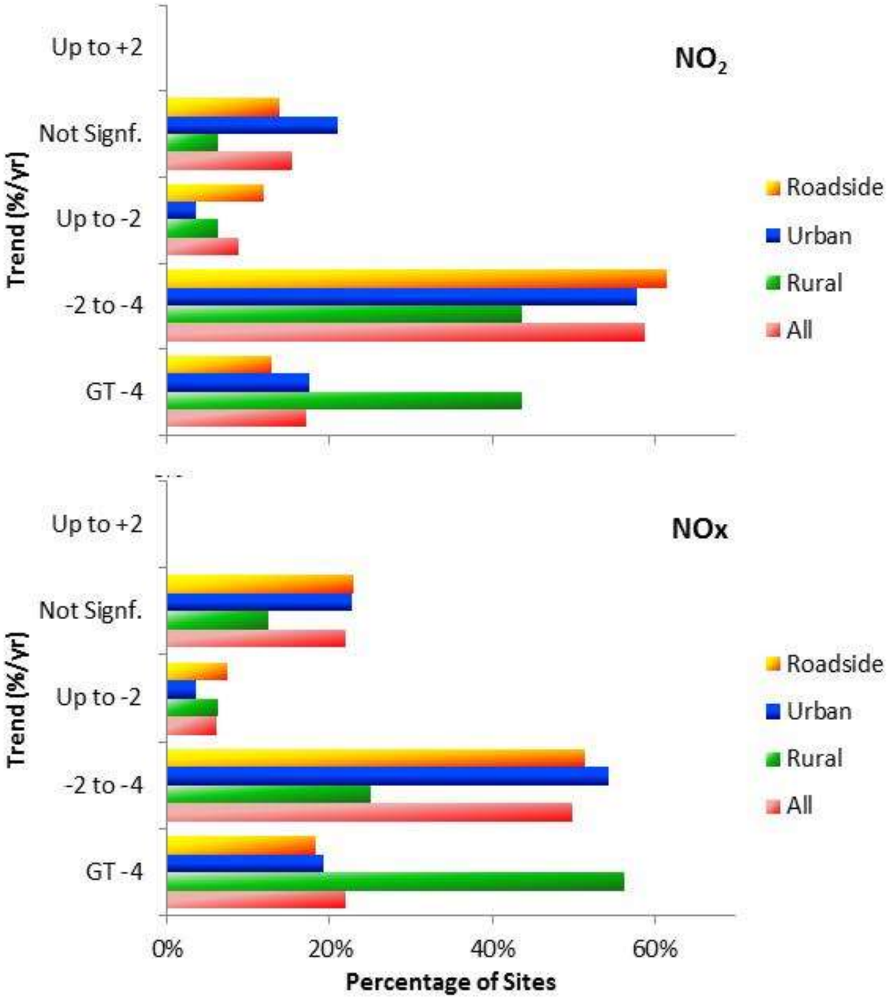


Figure 5.2 Trends in NO_2 and NOx concentrations at UK Sites 2010 – 2018, with sufficient data capture, grouped by site type (Laxen et al (2019).

5.4 European-Wide Trends in Primary NO_2

The initial evidence that primary (directly emitted) NO_2 was increasing was first noted in measurements in London in 1998 and 1999 by Carslaw et al (2001), Clapp and Jenkin (2001) and then in 2003 (Carslaw, 2005). At that time the increase in directly emitted NO_2 was strongly linked to the retrofit of TfL buses in London to use Continuously Regenerating Traps, CRTs – see Chapter 2. Since that time evidence from the UK and around Europe showed that there was a more general increase in NO_2 emissions related to the use of DOC/DPF on diesel vehicles (e.g. Anttila et al., 2011; Casquero-Vera et al., 2019; Font el al., 2019).

The issue of primary NO_2 was considered in depth in an earlier AQEG report (AQEG, 2007). Since that time there have been many developments leading to an improved understanding of the issue.

Notably, Grange et al (2019) analysed measurements of roadside ratios of NO_X and NO_2 measured in 61 urban areas across Europe from 1990 to 2016 as a proxy for primary NO_2 emissions. Aggregations were performed on the mean concentration of each city to ensure that the analysis was not bias to towards cities with more measurement sites such as London and Paris.

Figure 5.3 shows the overall European trend in vehicular NO_2/NO_x ratio based on an updated analysis of Grange et al., (2019). The results show that three main periods where the NO_2/NO_X trend had a distinct characteristic. Period 1 shows relative stability. The large increases in the NO_2/NO_X emission ratio during period 2 correspond to when DOC/DPF were introduced on diesel vehicles and a growth in diesel car share across Europe. However, in period 3, from around 2009/2010, the NO_2/NO_x ratio began to decrease.

The NO_2/NO_x emission ratio shown in Figure 5.3 reflects the European average trend but the trend for individual cities and monitoring sites varies much more, as shown in Figure 5.4. In period 3, the majority of European cities experienced decrease in the NO_2/NO_x ratio coinciding with the introduction of Euro V HDVs, which have been shown to be low emitters of primary NO_2 (Carslaw et al., 2016) and consistent with the analysis of concentration increments by Font et al. (2019) in London and Paris (see section **Error! Reference source not found.**). The downward trend from 2010 was not found in all locations, continued upwards trends were found in some cities including some locations in the UK.

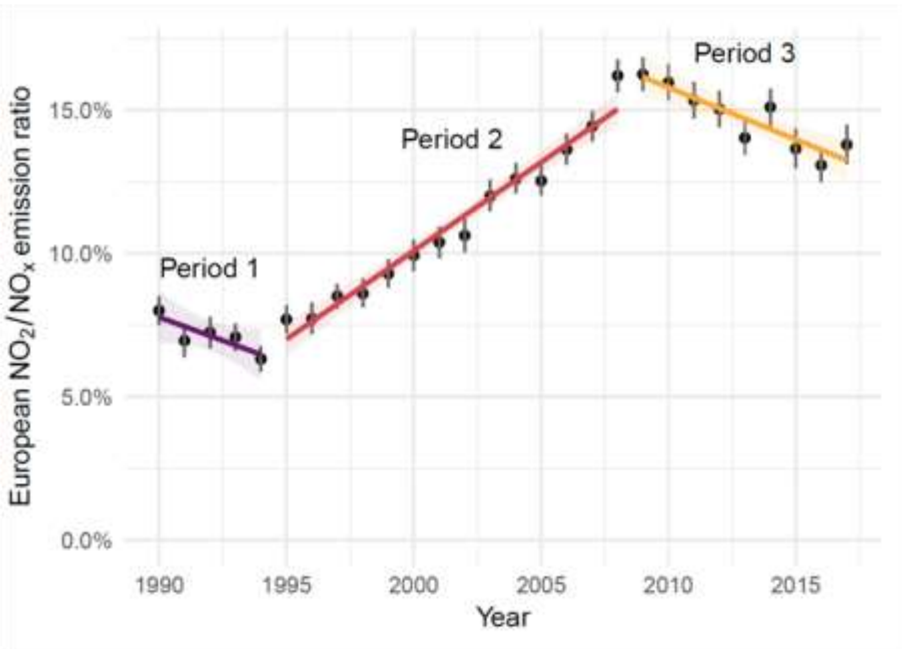


Figure 5.3 Estimated mean vehicular NO_2/NO_x ratio based on the analysis for European roadside ambient monitoring sites based on an updated analysis by Grange et al. (2019).

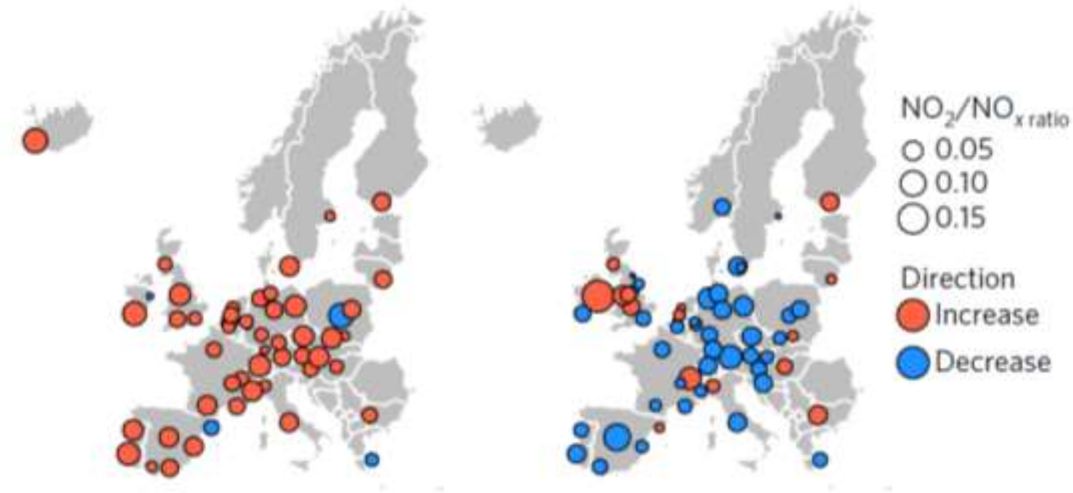


Figure 5.4 Roadside NO_2/NO_x ratio for each urban area for two-time periods: the five years leading up to 2010 (left) and the five years after 2010 (right) and direction of change.

Matthaios et al (2019) considered the primary NO_2/NO_x ratios for 14 urban areas in the UK between 2009 and 2016. The NO_2/NO_x ratio was found to have decreased from 17.5% to 12.5%, consistent with the downward trend found across Europe by Grange et al (2019) during period 3 (Figure 5.3), however this was not uniform across the urban areas, with increasing ratios being found in Chepstow, Sandy (Bedfordshire) and York.

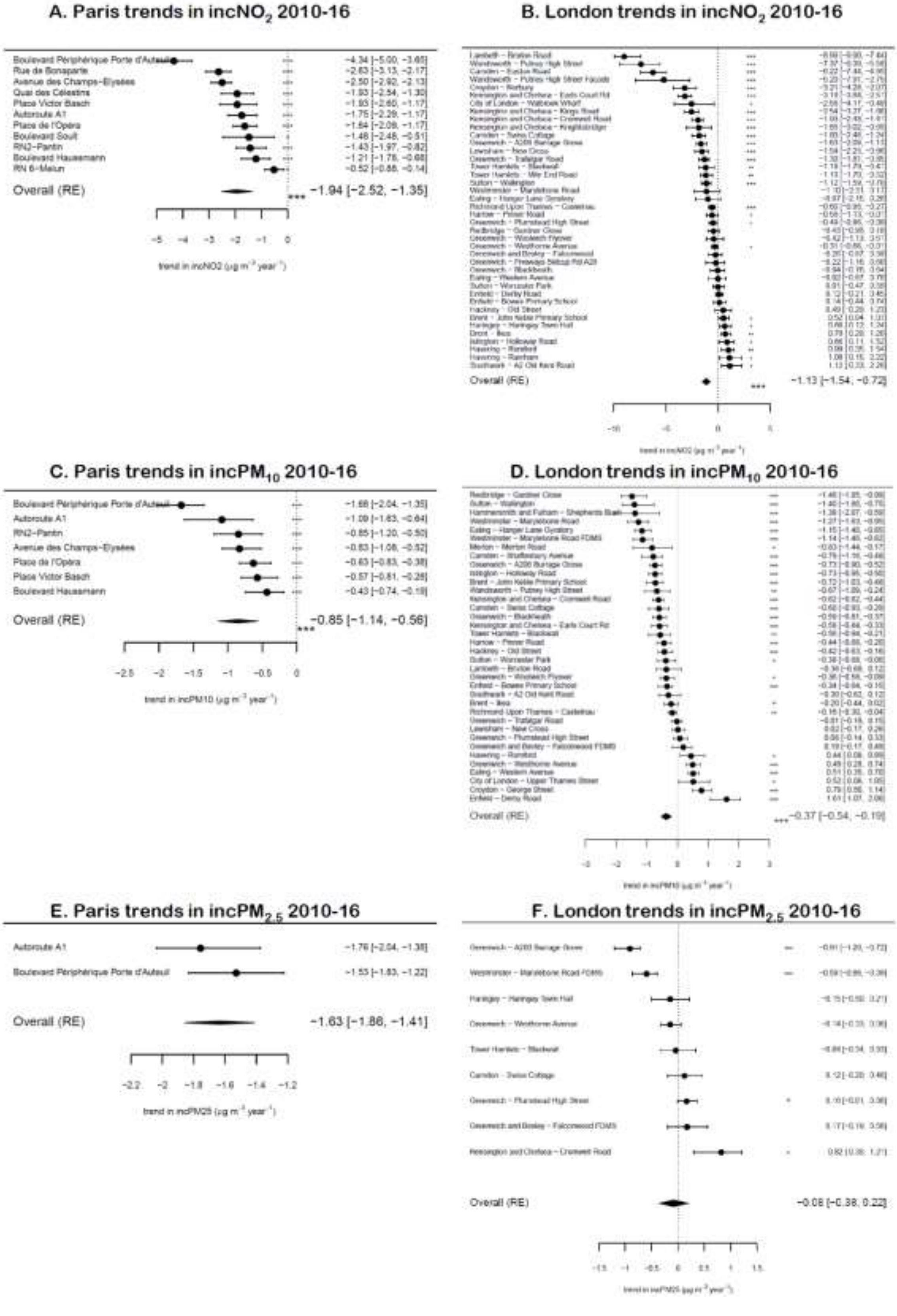
To investigate evidence that exhaust aftertreatment systems are downrated or non-operational in cold temperatures (Chapters 2 and 3), concentrations were also separated by temperature and time of day, specifically rush hour periods when temperatures were less than 5ºC to focus on cold-start emissions. Compared with warmer times, the NO_2/NO_x ratio for these cold-start periods was found to increase by 65% in the morning rush hour and by 75% in the evening rush hour.

5.5 Detailed Analysis of NO_x, NO_2 and PM Trends in London and Paris 2005 to 2016

Font et al. (2019) analysed trends in ambient concentrations in London and Paris. These were selected as Europe's two mega-cities, being separated by less than 400 km. This data set comprised a total of 44 monitoring sites across the Île-de-France region: 30 background locations and 14 roadside sites; and 130 monitoring sites in Greater London with 51 background and 79 roadside sites. The contribution from road traffic was calculated using the increment (termed incNO_2, incPM_{10} etc.) between each road or kerbside site and a single background site in each city. Trends were then calculated using the Theilsen approach, with adjustment for seasonality.

In the five years before the 2010 date for complacence with EU limit value the vast majority of roads showed increases in incNO_2 consistent with the increasing primary NO_2 fraction from 1995 to 2010 found by Grange et al (2019) (period 2 in Figure 5.3). Trends for incPM_{10} were mixed with around half of sites showing increases and half showing decreases.

Figure 5.5 shows the results 2010 to 2016 as Forest plots. Clear differences were seen between this and the preceding five years with all measured roads in Paris and the majority of those in London showing decreasing trends for incNO_2. This was consistent with the decreasing primary NO_2 fraction during period 3 in Figure 5.3 . All measured roads in Paris also showed decreasing trends for incPM_{10} and inc$PM_{2.5}$ but this was not the case for London where increases were measured alongside some roads.



A. Paris trends in incNO2 2010-16
Boulevard Périphérique Porte d'Auteuil
Rue de Bonaparte
Avenue des Champs-Elysées
Quai des Célestins
Place Victor Basch
Autoroute A1
Place de l'Opéra
Boulevard Soult
RN2-Pantin
Boulevard Haussmann
RN 6-Melun
Overall (RE)
−1.94 [−2.52, −1.35]
B. London trends in incNO2 2010-16
Overall (RE)
−1.13 [−1.54, −0.72]
C. Paris trends in incPM10 2010-16
Boulevard Périphérique Porte d'Auteuil
Autoroute A1
RN2-Pantin
Avenue des Champs-Elysées
Place de l'Opéra
Place Victor Basch
Boulevard Haussmann
Overall (RE)
−0.85 [−1.14, −0.56]
D. London trends in incPM10 2010-16
Overall (RE)
−0.37 [−0.54, −0.19]
E. Paris trends in incPM2.5 2010-16
Autoroute A1
Boulevard Périphérique Porte d'Auteuil
Overall (RE)
−1.63 [−1.86, −1.41]
F. London trends in incPM2.5 2010-16
Overall (RE)
−0.08 [−0.38, 0.22]

Figure 5.5 Forest plots for the trends in roadside increments (as µg m^{-3} y^{-1}) in NO_2 (incNO_2), PM10 (incPM10) and PM2.5 (incPM2.5) for Paris and London for 2010-2016. * significant at the 0.001 level; ** significant at the 0.01 level; * significant at the 0.05 level; + significant at 0.1; blank not significant. (Font et al 2019).**

Figure 5.6 shows trends in the roadside increments for NO_2, PM_{10} and $PM_{2.5}$ in 2010-16 against the annual mean concentration in 2010. There were faster downward trends in roadside increments alongside the roads with the highest concentrations in the start of the period. For incNO_2 the greatest downward trend was found alongside locations with high numbers of buses (LB4, WA7, WA8) and especially in Putney High Street (WA7, WA8) where buses had been retrofitted with SCR systems. There was a considerable spread in the trends for measurement sites with annual mean NO_2 less than 80 µg m^{-3}, with many roads showing an increase in concentrations. For incPM_{10} several roads in outer London had increasing trends, and two roads displayed increased inc$PM_{2.5}$.

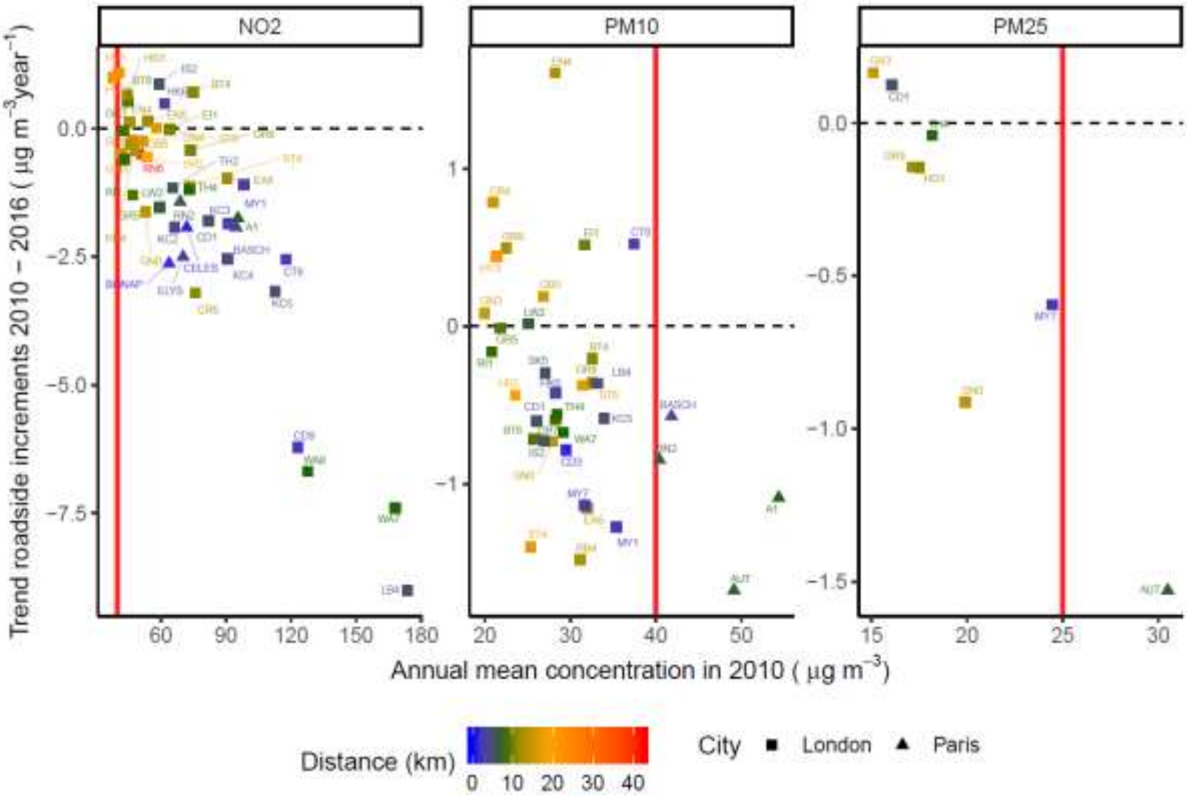


Figure 5.6 Trends in roadside increments in 2010-16 for each pollutant and the annual mean concentration in 2010. Colour scale shows distance to the city centre. Red lines denote the EU annual mean limit value (Font et al 2019).

Linear-mixed-effect models were built to identify the main determinants of changes over time and contrasts between places in annual roadside increments concentrations in Paris and London. Predictors included traffic counts by vehicle type, fuel type (for light duty vehicles) and Euro classes 3, 4 and 5 and III, IV and V reflecting those vehicle types being introduced during the analysis period (which also acted as a proxy for those leaving the fleet which could not be modelled explicitly). A total of eight model formulations were tested for each pollutant and the optimal formulation selected.

Table 5. shows the parameters in the optimal model for each pollutant. The traffic parameters with greatest impact on roadside NO_2 increments were Euro IV heavy vehicles (2.9 µg m^{-3} every 1000 vehicles) > motorcycles (1.8 µg m^{-3} 1000 $vehicles^{-1}$) > Euro III heavy vehicles (1.7 µg m^{-3} 1000 $vehicles^{-1}$) > diesel light diesel vehicles (0.6 µg m^{-3} 1000 $vehicles^{-1}$). Euro V heavy goods vehicles led to a decrease in $incNO_2$ levels at a rate of -4.1 µg m^{-3} 1000 $vehicles^{-1}$.

The optimum linear-mixed-effect model for $incNO_x$ did not separate the different Euro norms for heavy vehicles but did it for light diesels. Also, motorcycles and heavy vehicles were associated with an increase in $incNO_X$ (4.8 and 3.5 µg m^{-3} 1000 $vehicles^{-1}$) and Euro 5 light diesel was associated with a decrease (-2.4 µg m^{-3} 1000 $vehicles^{-1}$).

Roadside increments in PM_{10} were influenced by Euro III and V heavy vehicles; and Euro 2 motorcycles (0.7, 1.2 and 3.5 µg m^{-3} 1000 $vehicles^{-1}$, respectively) whereas Euro 5 light diesels was associated with a decrease in incPM10 (-0.4 µg m^{-3} 1000 $vehicles^{-1}$). Roadside $PM_{2.5}$ increments only showed two traffic parameters with significant coefficients: motorcycles (1.2 µg m^{-3} 1000 $vehicles^{-1}$) and Euro 5 light diesels (-0.3 µg m^{-3} 1000 $vehicles^{-1}$).

Table 5.2. Statistical parameters (coefficients for each vehicle type µg m^{-3} 1000 $vehicles^{-1}$) of the optimum model for each pollutant. Values in brackets denote the 95% confidence interval. Coefficients are expressed per 1000 vehicles. Bold numbers indicate significance at the 95% level; red = positive, green = negative. AADF = Annual Average Daily Flow. Adapted from Font et al 2019.

Traffic parameter	$incNO_X$	$incNO_2$	$incPM_{10}$	$incPM_{2.5}$
	Change in annual mean concentration µg m^{-3} 1000 $vehicles^{-1}$			
AADF light diesels		**0.6 [0.3, 1.0]**		
AADF heavy vehicles	**3.5 [1.1, 5.8]**			0.4 [-0.1, 1.0]
AADF motorcycles	**4.8 [1.5, 8.3]**	**1.8 [0.2, 3.3]**		**1.2 [0.9, 1.6]**

Traffic parameter	$incNO_X$	$incNO_2$	$incPM_{10}$	$incPM_{2.5}$
AADF Euro 4 light diesels	0.3 [-0.2, 0.8]		0.1 [-0.03, 0.3]	0.0 [-0.2, 0.1]
AADF Euro 5 light diesels	**-2.4 [-3.0, -1.9]**		**-0.4 [-0.5, -0.2]**	**-0.3 [-0.5, -0.2]**
AADF Euro III heavy vehicles		**1.7 [0.1, 3.3]**	**0.7 [0.1, 1.2]**	
AADF Euro IV heavy vehicles		**2.9 [1.6, 4.3]**	-0.5 [-1.3, 0.3]	
AADF Euro V heavy vehicles		**-4.1 [-5.6, -2.6]**	**1.2 [0.6, 1.8]**	
AADF Euro 2 motorcycles			**3.5 [1.9, 5.1]**	
AADF Euro 3 motorcycles			0.2 [-0.6, 0.9]	

The linear mixed effect model indicated that introducing Euro V heavy vehicles were responsible for reductions in $incNO_2$. This agrees with real-world emission tests from Euro V HGVs that show a reduction of 22-85% in primary NO_2 emissions compared with Euro II/III standards (Carslaw et al., 2016; Sjödin et al., 2017). By 2015, Euro V dominated the HGV and bus & coach fleets in both cities. Therefore, it is likely that the downward trends in $incNO_2$ concentrations in 2010-16 in the two cities reflected the reduction of NO_2 emissions from these vehicle types. Another factor that might have hastened the decrease in $incNO_2$ in 2010-16 is the introduction of Euro V diesel vehicles that have the negative coefficient in linear mixed effect model for $incNO_X$.

London had a faster HGV fleet turnover than Paris due to the introduction of the LEZ in 2008 as shown separately by Ellison et al. (2013) (see section **Error! Reference source not found.**). Pre-Euro III HGVs were removed from the fleet and replaced by Euro IV (up to 2010) and then by Euro V. The tightening of the LEZ in 2012 led to the remaining Euro III HGVs being retro-fitted to meet Euro IV PM standards (e.g. fitting a diesel particle filter -

DPF). However, the LEZ did not produce faster reductions in incNO_2 concentrations when compared with Paris. The LEZ in London was designed to reduce PM emissions but was also predicted to reduce NO_x concentrations in London by 18% (Cloke et al., 2000) based on assumed real-world emissions declining in line with Euro standards. The linear-mixed effect model suggests that reductions in incNO_2 would be optimised by tightening the LEZ access standard for heavy vehicles to at least Euro V rather than Euro IV as it is currently.

Paris and London observed downward trends in incPM_{10} in 2010-16. According to the linear-mixed effect model (Table 5.), the introduction of Euro 5 light diesels significantly reduced roadside PM_{10} concentrations. This was consistent with exhaust diesel particle filters (DPFs) (Fiebig et al., 2014) being effectively compulsory in Euro 5 cars and LGVs (2011).

DPFs in heavy vehicles were first introduced in Euro IV (2009) but the model for incPM_{10} gave non-significant factors for Euro IV heavy vehicles and positive factors for Euro V. A possible explanation is that the reduction of PM_{10} emissions from tailpipes (in the fine fraction) might have been counteracted by increased non-exhaust particle emissions (resuspension, brake-wear, tyre-wear) (dominant in the coarse fraction) from an increase in the traffic flow, especially that of heavy vehicles. Some suburban roads had upwards trends suggesting that the control on vehicular PM_{10} emissions did not have the same response everywhere.

Decreasing trends in inc$PM_{2.5}$ concentrations on the two Parisian roads with $PM_{2.5}$ data were greater than those observed in background locations indicating the success of traffic-related policies however, it was the opposite in London where trends in roadside inc$PM_{2.5}$ in 2010-16 showed a non-significant downward trend. Trends in inc$PM_{2.5}$ were not monotonic in 2010-16 and the consistent downward trend observed in 2010-14 was broken by an increase in the roadside $PM_{2.5}$ concentration in 2015-16 (see Font et al 2019).

The linear-mixed effect model for inc$PM_{2.5}$ identified Euro 5 light diesels as the vehicle category associated with reducing roadside $PM_{2.5}$; and motorcycles with increasing factors (Table 5.). This is an important issue since Euro standards for motorcycles do not regulate PM emissions (except for quads). An increase in the number of motorcycles in London might therefore have led to an increase in fine particle emissions, offsetting the expected benefits of DPFs on other vehicle classes.

5.6 Particle composition and traffic pollutants at “supersites” in London.

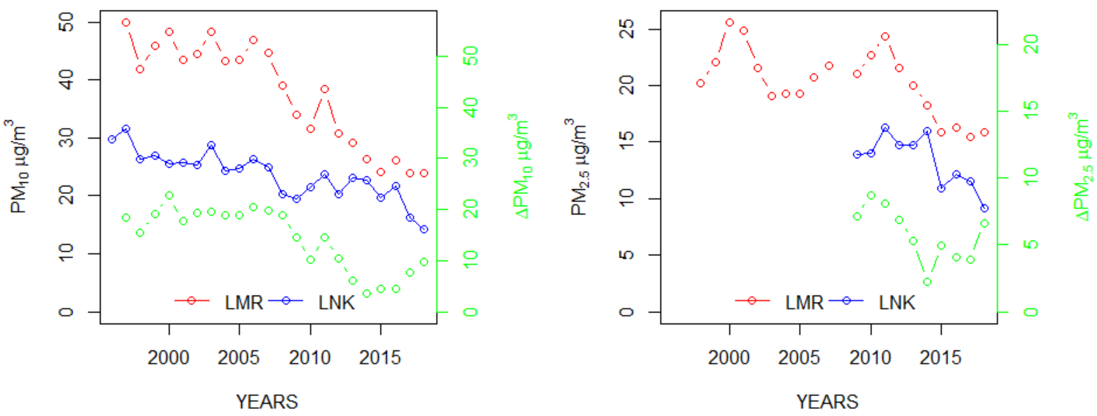
Measurements of particle composition are far sparser, compared with those of PM_{10} and $PM_{2.5}$, constraining the opportunities to be able to explain changes in concentrations from different sources. Harrison and Beddows (2017) examined the changes in annual mean concentrations of PM_{10}, $PM_{2.5}$, elemental carbon (EC) / black carbon (BC), organic carbon

(OC) and particle number at Marylebone Road and at the central London background site in North Kensington. These are the only UK roadside and background paired sites in a UK urban area that have extensive composition measurements.

Figure 5.7 shows decreases in all particle metrics since 2010 and decreases in the roadside increment. This suggests that, some of the change in PM_{10} and $PM_{2.5}$ concentrations are being caused by a decrease in the roadside increment of EC / BC and OC from traffic. Earlier analysis by Font and Fuller (2016) found a close to 1:1 agreement for trends in the roadside increment of black carbon and $PM_{2.5}$ at Marylebone Road between 2010 and the end of 2014 suggesting that a decrease in black carbon from traffic could explain the decrease in $PM_{2.5}$ confirming the effectiveness of diesel particle filters. It is however unclear if these changes in exhaust particles match those expected from the roll out of diesel particle filters as the London fleet changes. Singh et al (2018) also found declining concentrations of black carbon across urban and rural areas of the UK. Font et al (2019) noted that the widespread downwards trends in $PM_{2.5}$ from traffic found between 2010 and the end of 2014 were not continuing at all locations from around 2015 onwards.

Figure 5.7 also shows decreases benzene and CO concentrations reflecting the introduction catalytic exhaust controls. Decreases were seen at both sites with a decrease in the roadside increment. These species exhibited the greatest changes of any of the commonly measured traffic pollutants with dramatic improvements in benzene concentrations in the final years of the last century and rapid improvements in CO too. The WHO guideline for CO has not been breached in London since 2002.

As shown in Figure 5.8, particle number concentration at Marylebone Road declined by around 60% in late 2007, with a ban on the sale of diesel road fuel with a sulphur content of more than 10 ppm in the UK (Jones et al 2012). The steady decrease in the particle number concentration at Marylebone Road from 2010, would again point to the effectiveness of diesel particle filters.



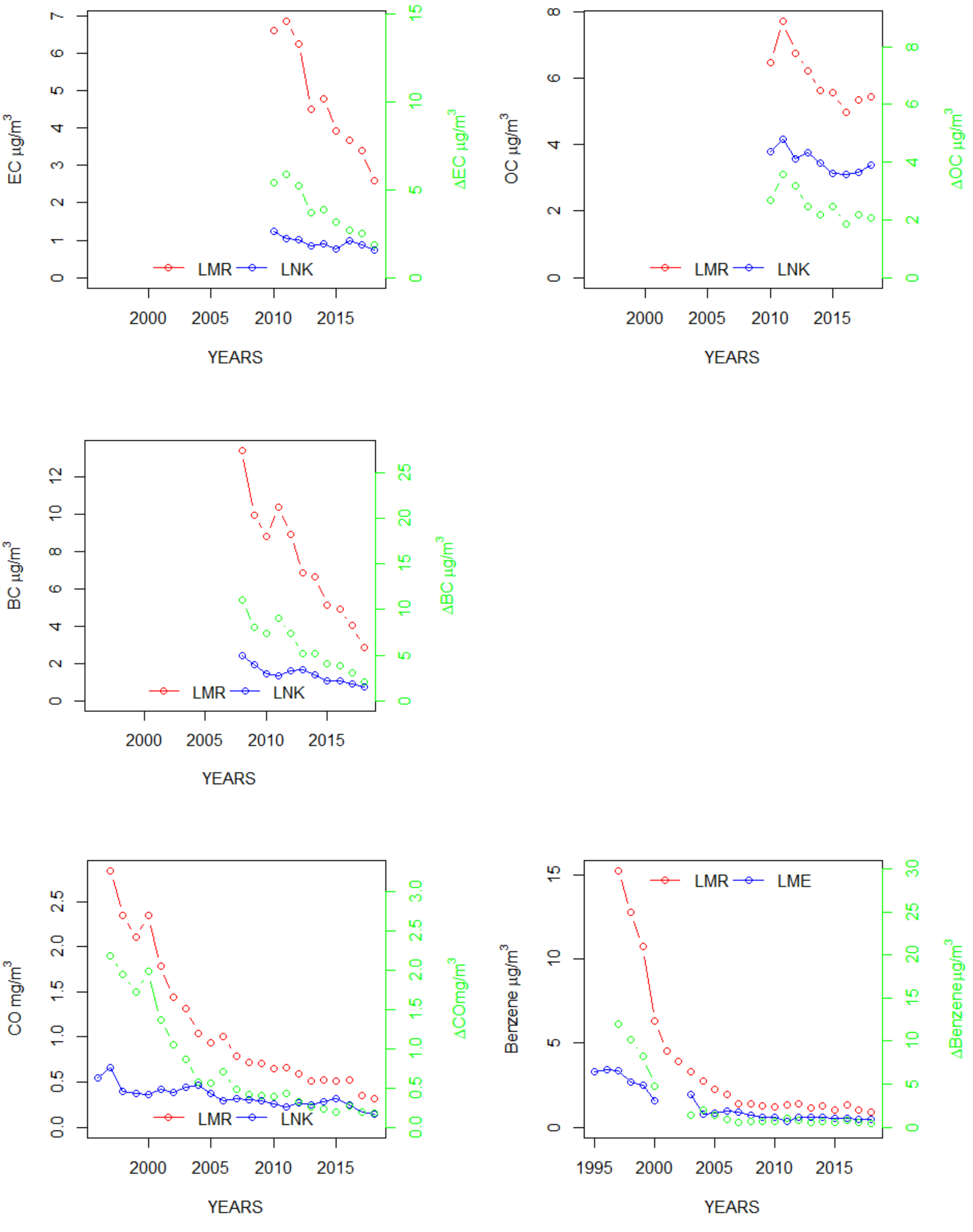


Figure 5.7 Annual mean pollutant concentration trends at Marylebone Road (LMR, red), North Kensington (LNK, blue), Eltham (LME) and the road traffic increment (Δ) (green). (PM_{10} and $PM_{2.5}$ pre-2005 are not EU reference equivalent).

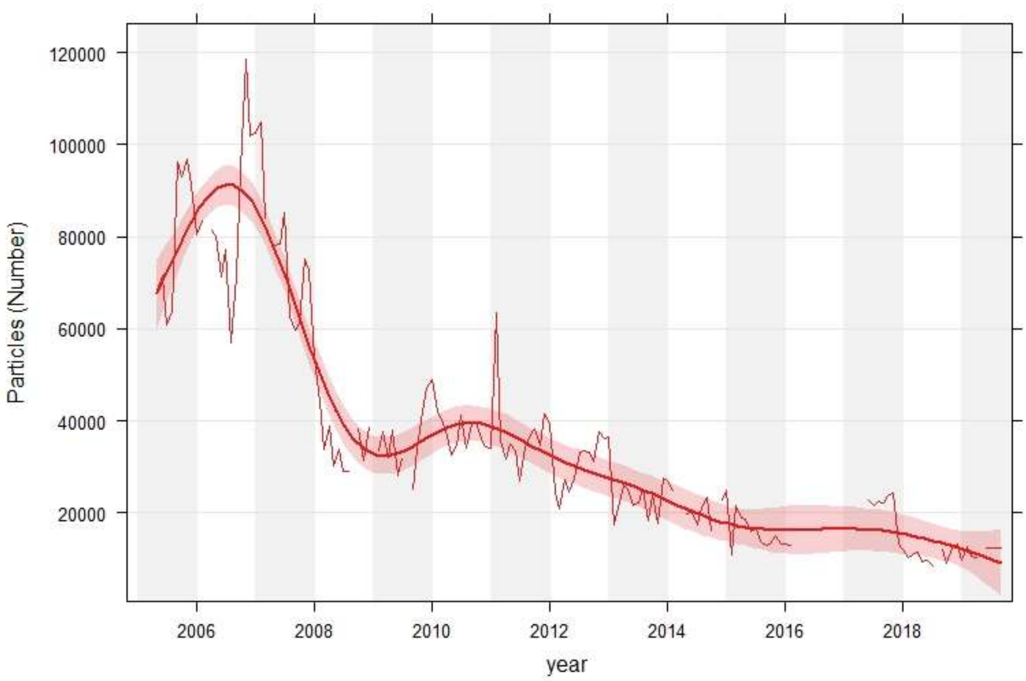


Figure 5.8 Monthly mean particle number concentrations at Marylebone Road with Loess smoothed trend line.

5.7 Hydrocarbons from traffic

Roads transport is a long-standing source of emissions of non-methane hydrocarbons (NMHC) and certain other small volatile organic compounds such as ethanol and acetaldehyde (collectively non-methane volatile organic compounds NMVOCs). Road vehicles contribute emissions to air through two major pathways, *via* the engine exhaust system and through evaporative losses of petrol and diesel from the fuel tank, pipework and engine, the latter being particularly significant in older carburettor intake petrol engines. Tailpipe emissions of NMVOCs are a mixture of unburnt fuel, for example compounds such as *iso* and *n* pentane, toluene, and octane, and products of partial combustion, such as 1,3 butadiene, butenes and pentenes. The introduction of the three-way catalytic convertor as an emissions control technology for gasoline vehicles has been very effective in reducing overall tail-pipe emissions of NMVOCs, and other measures such a fuel tank vapour recovery have reduced evaporative losses.

The atmospheric impacts on the reduction of harmful exhaust emissions such as 1,3 butadiene have been particularly significant, with roadside concentrations well below the target value of 2.25 µg m^{-3} right across the UK, and even in the most heavily trafficked locations such as Marylebone Road (Figure 5.9. **Monthly mean concentrations of 1, 3 butadiene and benzene at Marylebone Road. The solid line indicates the change predicted from emissions modelling. Note the log scale.**9) (AQEG, Volatile Organic

Compounds In The UK, 2020). Similar large decreases have been measured in benzene concentrations as shown in (Figure 5.9).

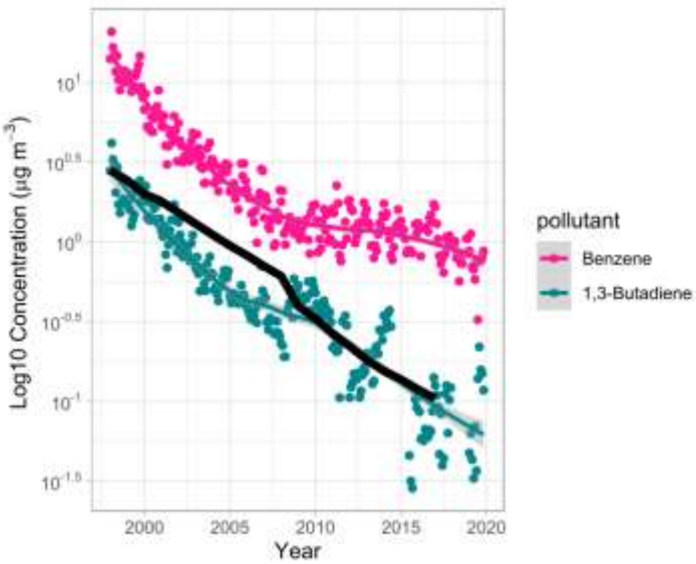


Figure 5.9. Monthly mean concentrations of 1, 3 butadiene and benzene at Marylebone Road. The solid line indicates the change predicted from emissions modelling. Note the log scale.

In addition to primary aerosols, exhaust emissions are known to contribute to secondary aerosols through the emission of NO_x and VOCs, which, after emission, can be subsequently oxidised in the atmosphere to form particulate matter in the form of nitrate and secondary organic aerosols (SOA). They also generally perturb the oxidation chemistry of the atmosphere that may influence the production of other secondary aerosol types, such as SOA from natural VOCs (Hoyle et al., 2011). While aromatic VOCs such as toluene and xylene have long been recognised as SOA precursors, more recently larger molecules have recently been recognised as important (Genter et al., 2012). Measurements undertaken in London by Dunmore et al. (2015) highlighted larger alkanes, such as those emitted from diesel engines and the so-called semi- and intermediate-volatility organic compounds (SVOCs and IVOCs), which are sufficiently large (containing more than approximately 13 carbons) that they exist in equilibrium in both the particle and gas phases (Robinson et al., 2007). These are largely unspecified in nature and challenging to study, and while they are emitted in lower quantities than lighter VOCs, they are recognised as producing much more SOA per molecule emitted, so are potentially highly significant.

The exact contribution exhaust emissions make to the secondary $PM_{2.5}$ budget in polluted environments is difficult to explicitly determine, partly because of the challenges of disentangling it from the SOA arising from natural VOCs. While there is evidence that it can be the dominant source of SOA in certain environments (Liu et al. 2012) their overall importance is currently the subject of much scientific debate (Akherati et al., 2019; Jathar et al., 2014), so this should be regarded as uncertain at this stage. Exhaust SVOC/IVOC emissions are mitigated through the use of particle filtration and oxidation catalysts (along with conventional VOCs), so it is reasonable to expect that whatever impact this class of emission is currently having on air quality will diminish in the future as these technologies become more commonplace.

5.8 Ammonia

More than 85% of the ammonia emission in the UK originates from agricultural sources such as livestock wastes and fertiliser applications. Traffic exhaust is thought to be one of the dominant non-agricultural sources of ammonia in urban areas of the UK (Sutton et al., 2000) and thus the trends in urban NH_3 concentrations provide some indirect insights into the trends of exhaust emissions of NH_3. The relationship is not straight-forward, however, because NH_3 combines with HNO_3, which originates from the oxidation of NO_x, to form aerosol ammonium nitrate (NH_4NO_3). Gas-phase NH_3 and HNO_3 then continue to co-exist in an equilibrium with the NH_4NO_3, which depends on temperature and relative humidity as well as the detailed overall aerosol composition. Thus, urban NH_3 concentrations are not only dictated by local emissions, but also by concentrations of HNO_3 and meteorological conditions, as well as the regional background. Thus, trends in concentrations could partly reflect temporal changes in any of these other factors.

The trends of ammonia measured monthly with a denuder technique (DELTA; Sutton et al., 2001) in the two cities with sites of the UK National Ammonia Monitoring Network (NAMN) are shown in Figure 5.10. In London, measurements were initially made at Bridge Place from the 3rd floor of a building, before sampling moved to the grounds of London's Natural7 History Museum about 15 m from Cromwell Road. In Edinburgh, measurements were started in a quiet road (Castle Terrace) and then moved to The Pleasance, where sampling took place initially from a window of the Medical School and later at the St Leonard's monitoring station, about 20 m from the road. Measurements in Edinburgh stopped at the end of 2016. Changes across site changes are meaningless. However, both sites show a declining trend in urban NH_3 concentrations up to about 2010, with little apparent change since then. The observed trend at these urban sites is more pronounced than those at rural / agricultural sites, where the NAMN has detected a significant decline only in areas dominated by pig and poultry emissions (Tang et al., 2018).

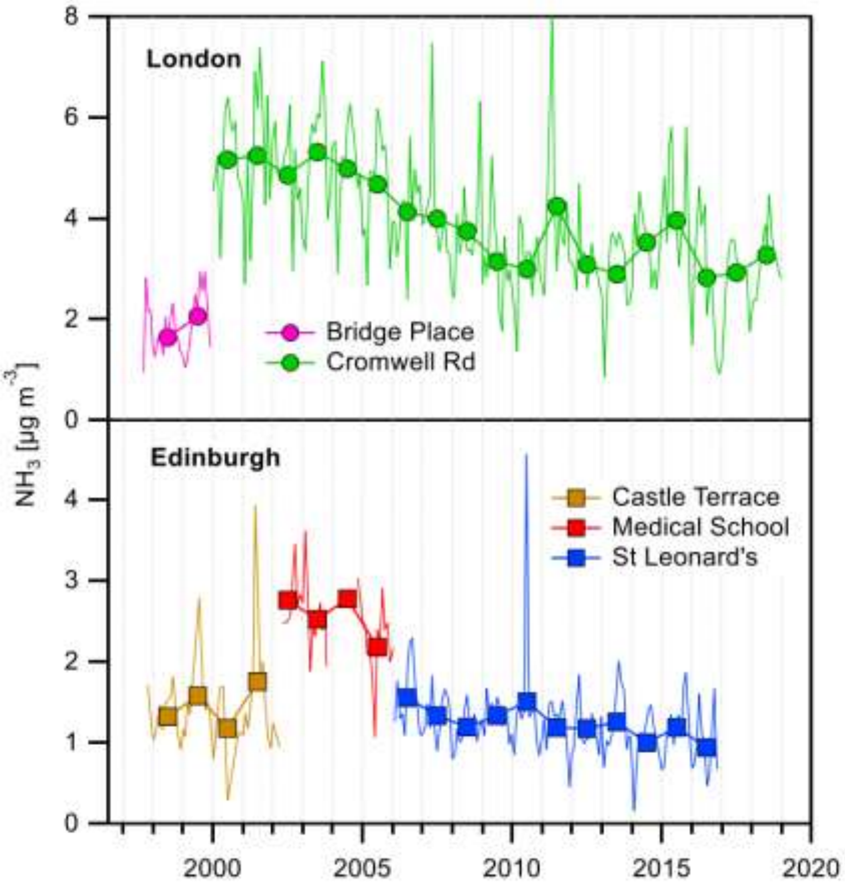


Figure 5.10. Trend in ammonia concentrations in two UK cities.

Both urban and rural concentrations exhibit a seasonal cycle with the lowest concentration in winter and the highest concentration in summer. This is consistent with the temperature effect on gas-particle partitioning which favours higher gas-phase concentrations under warmer conditions, but it is also consistent with the temperature response of evaporative NH_3 sources which is under similar control.

Additional ambient measurement evidence on vehicle emissions of NH_3 stems from shorter-term studies of roadside concentrations. Marner et al. (2018) used a network of ALPHA (Tang et al., 2001) and DELTA samplers (Sutton et al., 2001), NO_2 diffusion tubes and a chemiluminescence reference sampler to measure concentrations of a range of nitrogenous pollutants at 29 sites in southern England over a period of two years (summer 2014 to summer 2016). The network was spread over an area of 27 km^2 and included sites adjacent to both major and minor roads, background sites, and roadside transects. NH_3 and NO_2 concentrations measured well away from roads were subtracted from the roadside measurements to estimate the traffic-related increment. Total NO_X concentrations were estimated from the NO_2 diffusion tube measurements using the finite difference model described by Abbott and Stedman (2005) and published by Defra (Abbott et al., 2017).

All of the roadside monitoring sites at which appreciable roadside NO_X increments were recorded also showed marked roadside increments of NH_3. Furthermore, concentrations of NH_3 and NO_x exhibited similar spatial distributions close to roads (e.g. Figure 5.11 A). A comparison of the roadside increments of NO_x to those of NH_3 showed that, on average, the molar ratio of NH_3 to NO_X was approximately 6% (e.g. Figure 5.11 B).

Vehicle emission remote sensing provides detailed measurements of NH_3 based on the direct measurement of individual vehicle exhaust composition (see chapter 3). Over 300,000 vehicle measurements made by Ricardo and the University of York (Grange et al., 2019; Carslaw et al., 2019), have been used to consider the emission of NH_3. These measurements will tend to represent fleet-weighted emissions because almost all vehicle types are measured, and the measurements will be dominated by the most popular vehicle types in use. The remote sensing data suggests a mean ratio of 7.5% i.e. similar to the value of 6% calculated by Marner et al. (2018).

The roadside NH_3 measurements of Marner et al. (2018) may also be compared with those of Cape et al. (2004), which were collected 12 years earlier (2002 to 2003). Cape et al. (2004) used ALPHA samplers to measure NH_3 concentrations, as well as diffusion tubes to measure NO_2 concentrations, alongside roads and at representative background locations; subtracting the background values to indicate the vehicle-derived increment. Figure 5.16 summarises some of the results, alongside those of Marner et al. (2018). The roads surveyed by Cape et al. carried between 3,500 and 58,000 vehicles per day (vpd) and comprised between 8% and 15% HDV (HGV + buses). Those recorded by Marner et al. ranged from 1,400 to 16,000 vpd and 2% to 7% HDV. There are a number of further differences in the methodology of these studies, hindering a direct comparison. Nevertheless, there is no evidence in Figure 5. that NH_3 emissions per average-vehicle have changed appreciably over the 12 years separating the studies, in contrast to the urban trends in NH_3 concentrations described above. The same observation is also apparent for concentrations of NO_2 in Figure 5.12, again despite the more comprehensive analyses described in above showing evidence of a decline in traffic-related NO_x and NO_2 concentrations over this period. The simplistic comparison given in Figure 5.12 may thus be misleading.

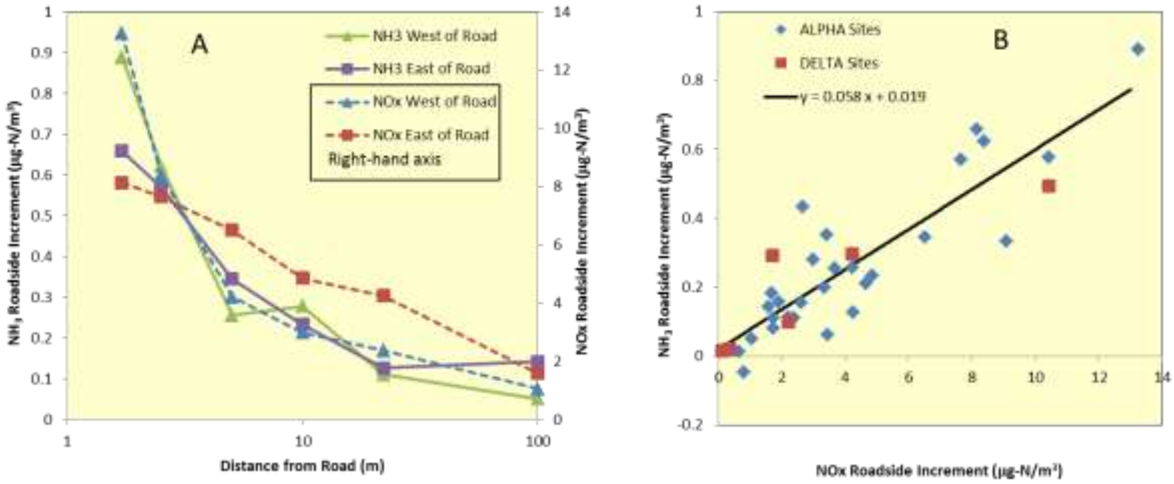


Figure 5.11 A: Roadside Increments (i.e. total concentration minus local background) of two-year (2014-2016) average NH_3 and NO_x concentrations along two transects running perpendicular to the A22 in East Sussex. B: Roadside increment of NH_3 vs roadside increment of NO_x at all sites with co-located measurements, showing bivariate least squares regression line.

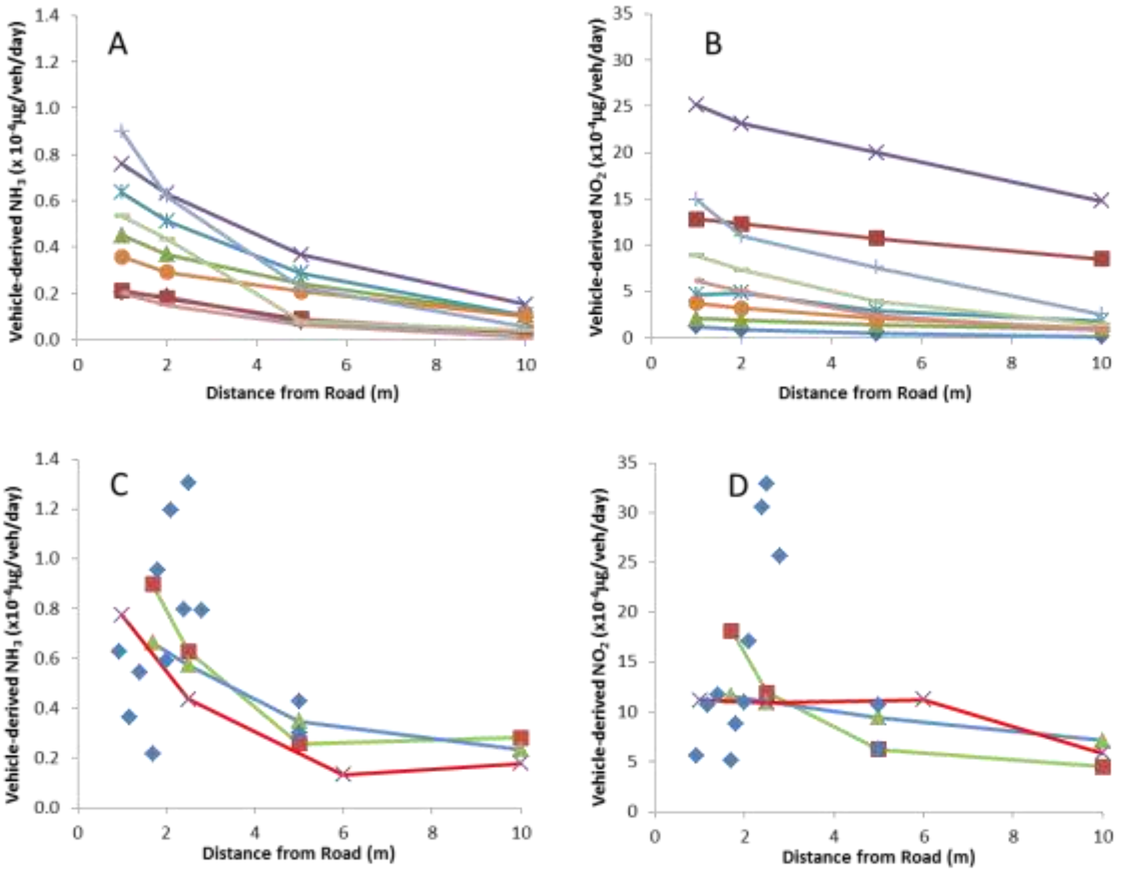


Figure 5.12 Roadside Increments (i.e. total concentration minus local background) of NH_3 and NO_2 per vehicle over nine transects in 2002-2003 (Panels A & B) and three transects and 12 roadside monitors in 2014-2016 (Panels C & D). 2002-2003 data from Cape et al. (2004). 2014-2016 data from Marner et al. (2018).

5.9 Evidence from Clean Air / Low Emissions Zones

Clean air or low emissions zones aim to improve air pollution in an area by restricting the entry of certain types of vehicles. Zones are usually based on the exclusion of older vehicles based on the assumption that they were constructed with less effective exhaust abatement compared with newer vehicles. Some zones differentiate between petrol and diesel fuelled vehicles because of their differential emissions standards and real-world emissions.

Holman et al (2015) found over 200 low emission zones operational in 12 European countries. Low emission zones vary in design making them hard to compare. For instance, some apply to heavy vehicles only while others also encompass passenger cars. Also, the success of a low emission zone relies on the differential in real-world emissions between vehicle types that are excluded and the vehicles that are allowed in the zone. This differential in real-world emissions has not always been found in the fleet for some pollutants, hampering the success of these zones (Fuller, 2018). Nevertheless, in German cities with low emission zones, reductions in annual mean PM_{10} and NO_2 concentrations up to 7% and 4% respectively have been reported with greater changes found in specific exhaust pollutants, such as black carbon concentrations (Holman et al 2015).

Before considering changes in air pollution the first metric of success for a low emission zone should be to consider it impact the vehicle types in circulation. The effectiveness of the London low emission zone in changing London's vehicle fleet was shown by Font et al (2019) who contrasted it with Paris, where no zone was present.

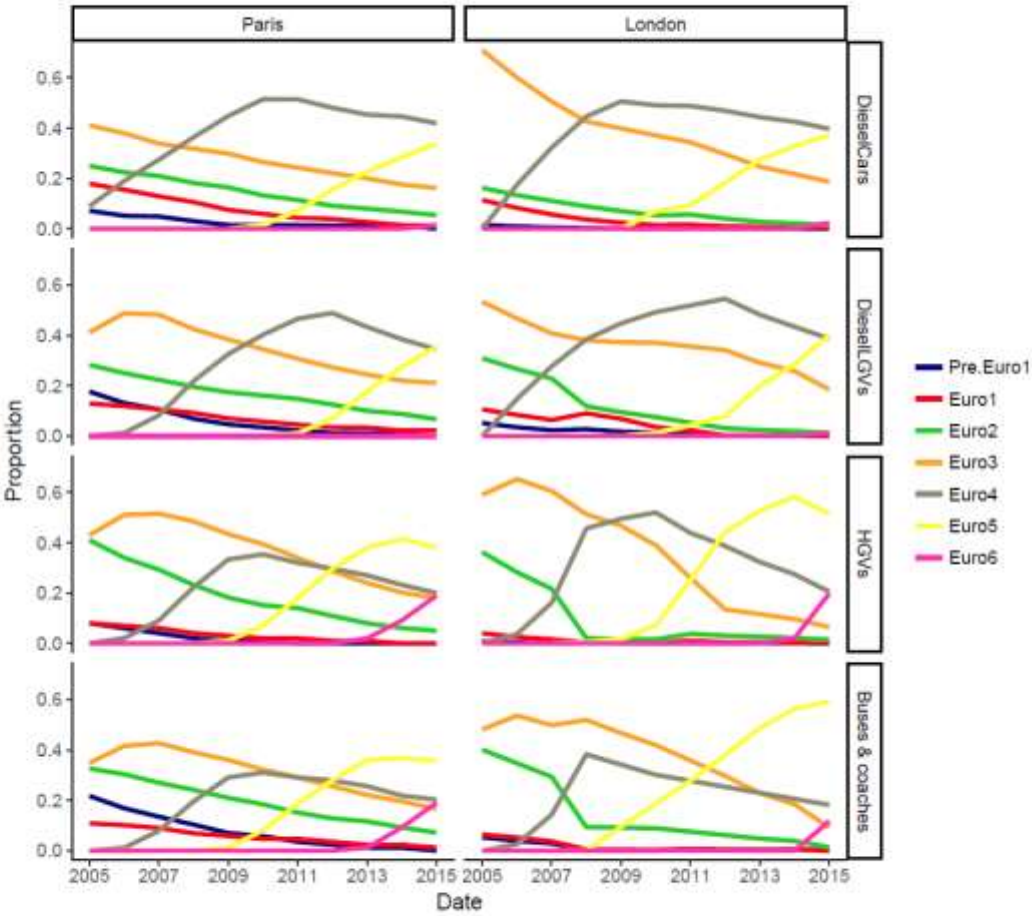


Figure 5.13 distribution of Euro classes for diesel vehicles (cars, LGVs, HGVs and buses & coaches) for Paris and London (Font et al 2019).

Figure 5.13 shows the distribution of Euro classes for diesel vehicle in Paris and London. This shows both historic differences in the car fleets and also the impact of London's low emission zone on the HGV and bus & coach fleet.

At the start of the 2005-09 period both Paris and London had similar HGV Euro distribution with Euro II and III dominating the fleet. But at the end of the period the distribution was very different in the two cities. Euro II HGVs were quickly replaced in London's fleet by Euro IV and had almost disappeared by 2008, when the LEZ was introduced. By contrast, in 2009 the presence of Euro II HGVs was still notable in Paris at 20%. Euro IV was introduced into the fleet of the two cities but reached a higher share in London (49% by 2009) than Paris (30%). The introduction of phase 3 and 4 of the LEZ in London in 2012 induced a faster decrease in Euro III compared to Paris. Most remaining Euro III HGVs after 2012 in London were adapted to meet Euro IV standards for PM emissions (63-93%) and therefore permitted in the LEZ. Euro V was introduced in 2010 and by 2015 it had become a large part of the fleet; 40% in Paris and 51% in London. The Euro class distribution of buses & coaches in Paris was similar to that of HGVs for the whole-time period.

Phases 1 to 4 of the London Low Emission Zone were evaluated by Ellison et al (2013) who found a 3 % decrease on PM_{10} inside the zone compared with a 1 % decrease elsewhere. No statistical different change in NO_x concentrations was found between the zone and outside.

London's Ultra Low Emission Zone (ULEZ) was introduced in April 2019. Nested within the original low emission zone this new scheme applied to the central area only. It followed a period of upgrade for bus fleets and an additional "toxicity" or T change for pre-Euro IV petrol and diesel cars on top of London's congestion charge.

Initial analysis (GLA, 2019) of emerging data from London's ULEZ for the period up to the end of September 2019 showed that the roadside increment of NO_2 within the zone decreased by 36% averaged across all sites. This was a mean change of 29% (24 µg m^{-3}) when compared with a no-ULEZ scenario based on concentration changes in outer London, well away from the zone. This change was relative to a baseline of February 2017 when the central London T charge, a precursor to the ULEZ, was confirmed. The change in concentrations was brought about by changes in the vehicle fleet and also a decrease in traffic volume of around 3 to 9%. The change in ambient concentrations is similar to the predicted decrease in NO_x emissions modelled from observations of traffic flow, vehicle type and Euro class and composition. Emissions modelling also predicted a change of 11-13% for $PM_{2.5}$ but an absence of appropriate measurements in the zone means that this is not verifiable.

However, despite evidence of change in concentrations from low emissions zones, the schemes are often complex and have not generally been of sufficient magnitude to provide insights into the contribution of different vehicle types and technologies to pollutant emissions and concentrations in urban air.

Some insights into real-world exhaust emissions can be gained from more simple schemes that focus on a single vehicle type. Low emissions zones focused on buses have met with some success. These include the zone in Brighton (Brighton & Hove, 2019), which met with early success and the 12 low emission bus zones in London that mandated Euro VI buses only. Annual mean NO_2 decreases between 2016 and 2019 varied along the different London zones. These ranged from 7 to 58 µg m^{-3} (14 to 49%) (GLA, 2019). This confirms that there is a large improvement in emissions from Euro VI buses compared with the older ones that they replaced. It also confirms the large contribution that buses make to total road transport emissions in these specific locations.

5.10 Summary

Air pollution measurements across Europe and the UK are focused on determining compliance with legal limits and are not optimised to quantify air pollution from specific sources, including traffic. Despite the importance of NO_2 measurements for legal compliance, just eight urban areas in the UK had consistent roadside measurements of NO_2 between 2000 and 2017, severely limiting opportunities for trend analysis and policy feedback. Other weaknesses in current network design include on-going changes in measurement location, and sparse measurements of particle composition and traffic. Pairing roadside measurement sites with nearby background and rural measurements would improve the opportunities for source apportionment.

There were widespread breaches of EU limit and target values for NO_2 and PM across Europe in 2017. Proximity to traffic sources was a major factor in breaches for many pollutants, most especially NO_2.

Analysis of UK, London, Paris and European-wide ambient air pollution datasets has revealed important disparities between the decreases in concentrations that would be inferred from vehicle emissions standards and those measured in ambient air. Whilst concentrations of some pollutants have decreased others have not fallen as fast as expected. There was also large heterogeneity in trend in different locations suggesting that air pollution control technologies are not working effectively in all road conditions. The reasons for this heterogeneity are not explained by this analysis.

Rather than decreases, there were overall increases in NO_x and NO_2 from traffic between 2005 and 2010 as we approached the dates for compliance with EU limit values. This was followed by decreases between 2010 and the end of 2016. Analysis of concentrations measured at 112 locations across the UK between 2005 and 2018 showed overall decreases in NOx and NO_2 concentrations of 1.9 and 1.8 % per year, respectively. However, 15 % of roads in London had increasing concentrations of NO_2 from traffic between 2010 and 2016. Analysis data from London and Paris showed that motorcycles and heavy vehicles were associated with an increase in NO_X while Euro 5 light duty diesels were associated with a decrease. Primary emissions of NO_2 from traffic were important in determining localised NO_2 close to roads. These emissions increased Europe-wide from the late 1990s, coinciding with the introduction of oxidation catalysts and particle filters on diesel cars and a growth in the proportion of diesel vehicles in the car fleet. A decreasing trend in primary NO_2 began around 2009/2010, coinciding with the introduction of Euro V heavy duty vehicles however this downward trend has not been observed in all locations in the UK and in Europe.

PM_{10} measurements from traffic in London showed a statistically significant decrease in London between 2010 and 2016, but, as with NO_X and NO_2 there was considerable heterogeneity in trend on different roads and not all roads showed a decrease. There was no statistically significant change in traffic $PM_{2.5}$. Analysis data from London and Paris showed that increased PM_{10} was associated with Euro 3 and Euro 5 heavy duty vehicles and Euro 2 motorcycles. Motorcycles were also associated increased $PM_{2.5}$. The introduction of Euro 5 light duty vehicles was associated with a decrease in both PM_{10} and $PM_{2.5}$, due to particle filters on these vehicle types. Particle number concentrations showed an overall downward trend since 2005 with a rapid step change coincident introduction of diesel with a sulphur content of less than 10 ppm in the UK in late 2007.

VOC and carbon monoxide exhibited rapid deceases in the first five years of this century, but the rate of change has slowed since this time. Ammonia concentrations are elevated close to roads and it is currently unclear from the available ambient measurements if the road component of concentrations has been falling over time.

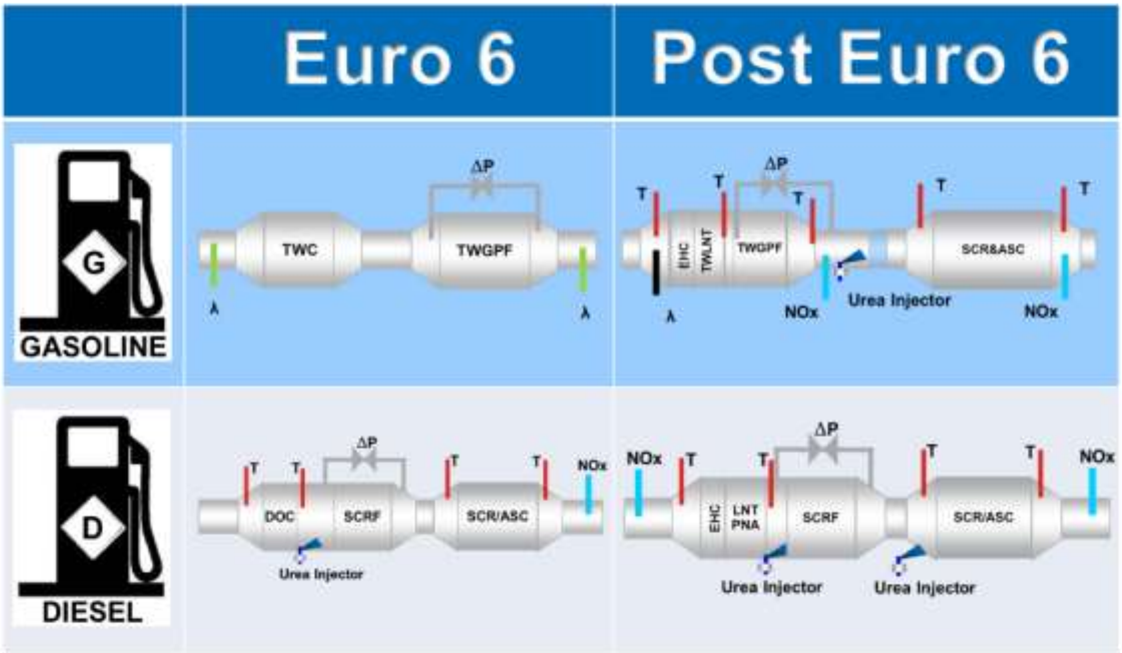
Clean air or low emissions zones can be an effective tool to accelerate the decrease of air pollution from traffic, but their impact depends on the scheme design. Zones focusing on busy bus corridors have led to deceases in NO_2. Initial analysis of emerging data from London's ULEZ for the period up to the end of September 2019 showed that the roadside increment of NO_2 within the zone decreased by 36% averaged across all sites This was a mean change of 29% (24 µg m^{-3}) when compared with a no-ULEZ scenario.

6 FUTURE INTERVENTIONS

6.1 Future engine and aftertreatment combinations

There is a potential that future gasoline engines will be operated lean (in conditions of excess oxygen) to achieve improved fuel economy, which will require significant changes to exhaust emissions aftertreatment systems. Operating a gasoline engine lean for part load fuel economy benefits means that conventional three-way catalysis will not operate to control NO_x emissions at these conditions. Reducing NO_x emissions therefore requires more complex aftertreatment systems than currently employed under stoichiometric conditions. Figure 6.1 shows an example of such a post Euro 6 gasoline emissions control system. The system contains an electrically heated catalyst for rapid warm up, followed by a combined three-way catalyst and lean NO_x trap. Next is the gasoline particulate filter followed by an SCR system designed specifically for gasoline application.

Additional engine technologies such as external EGR and water injection will be used in gasoline powertrains which can have secondary emissions benefits but are often not the primary role of the technologies. EGR will generally be applied for fuel economy benefit but can also be applied in order to maintain stoichiometry and thereby reduce the requirements of aftertreatment components. Water injection can be applied for a performance benefit, to achieve stoichiometry, or to get a fuel economy benefit via increasing the effective compression ratio of the engine. Both technologies under stoichiometric operation reduce engine out NO_X emissions, through reduction of peak temperatures during combustion. Reducing peak temperatures in the combustion chamber reduces NO_x formation which can reduce the demands on the aftertreatment system.



Source: Ricardo analysis

Figure 6.1. Typical current and potential future exhaust emissions aftertreatment systems
(EHC – Electrically heated catalyst, PNA – Passive NOx Adsorber, LNT – Lean NOx Trap, TWC – Three Way Catalyst, TWGPF – Three Way Gasoline Particulate Filter, SCR – Selective Catalytic Reduction, ASC, Ammonia Slip Catalyst & SCRF – Selective Catalytic Catalyst on Filter).

Current diesel exhaust emissions control systems typically consist of an oxidation catalyst followed by a combined SCR on filter technology with an underfloor SCR catalyst and finally an ammonia slip catalyst. Changes expected to further optimise efficiency will include independent control of each NO_x catalyst. This could be achieved, for example, by the use of two urea injectors servicing two SCR catalysts. In addition, an electrically heated catalyst may be required to rapidly heat the first catalyst in the exhaust system providing efficient emissions control sooner after engine start. Unconventional ammonia creation routes such as those developed by Wilson and Hargrave (2018) offer lower temperature SCR operation through the use of on-line generated intermediary solutions. This could result in significant reductions of NO_X emissions under cold operation and cold start.

It is clear from these examples that technologies are available that can further reduce air pollutant emissions and fuel consumption. The compromise between cost, emissions reduction performance and vehicle performance systems should therefore result in further reduction of future vehicle emissions provided there is appropriate commercial incentive.

6.2 Impact of increased hybridisation

With increasing hybridisation there is opportunity for future internal combustion engines and range extender engines to be designed and optimised to operate over a much more limited range of operating conditions compared to current engines. This means that the calibration complexity, cost and performance compromises made in conventional engines could be somewhat mitigated resulting in improved emission performance, reduced cost or reduced complexity.

The level of reduction observed is however highly dependent on the operating strategy and relative importance of different product attributes during development. Huang et al (2019) compared CO_2, HC and CO emissions for two vehicle types, both with conventional and hybrid powertrains and observed significant and consistent reductions in CO_2 emissions. Equivalent reductions were not observed for CO and HC emissions, with CO emissions for hybrid vehicles being consistently higher than conventional vehicles. This was attributed to the increased number of engine stops and restarts which are likely to arise from less controlled combustion and aftertreatment systems during the restart events. Similarly, Yang et al (2019) showed that both PFI and GDI gasoline fuelled hybrid vehicles emitted higher particle numbers than their conventional powertrain counterparts as a consequence of increases in particle emissions during restart events and therefore urban driving. The gasoline hybrid vehicles tested using PEMS by O'Driscoll et al (2018) did show NO_x emissions ~20 times lower than non-hybrid equivalents.

6.3 Impact of increased automation

Availability of driver assist and autonomous vehicle technologies is increasing. With this comes the potential to design into the wider transport system behaviours which affect the operation of vehicles on the road. Wave like behaviour in traffic systems resulting in part from the human factor in vehicle control is known to result in fluctuating speeds and typically therefore increased emissions. Limited testing by Stern et al (2019) suggest that in congested traffic conditions the influence of autonomous vehicles extends significantly beyond that of the vehicle itself to dampen out such waves. If considered in the control system design of the autonomous vehicle, they showed that a fleet of 21-22 vehicles with only one of them being autonomous could achieve speed load characteristics corresponding to a 73% reduction in nitrogen oxides from the entire fleet.

6.4 Abatement technologies for existing fleet - Retrofit options

Retrofit emissions control technologies are relatively mature for heavy duty applications with many buses in major cities being retrofitted to control both NO_x and PM emissions. Figure 3.2 shows the retrofit results for a Brighton bus (Ricardo Internal data, 2016). The retrofit was on a Euro III bus with the aim of achieving Euro V emissions. The retrofit Euro III bus gave lower tailpipe NO_x emissions than the market Euro V bus (98 g compared to 176 g) over the same 18 km bus route (Ricardo Internal data, 2016).

However, the retrofit for diesel passenger cars is not so mature but is becoming more important especially in certain cities where bans on diesel cars will be implemented for Euro 5 or lower legislated vehicles (Reuters.com, 2018). The retrofit system mainly consists of an SCR system, with or without reforming of urea, for NH_3, off-line for use under low temperature exhaust conditions. Initial studies have shown good NO_x control can be achieved by passenger car retrofit and there are a range of solutions now on the market. Figure 6.2 shows the cumulative NO_x emissions for a Euro 5 vehicle with a DOC and DPF which has been retrofitted with an SCR system over the WLTC and an urban RDE (Real Driving Emissions) cycle. When the SCR system is activated, the tailpipe NO_x emissions reduce significantly (Giechaskiel et al, 2018).

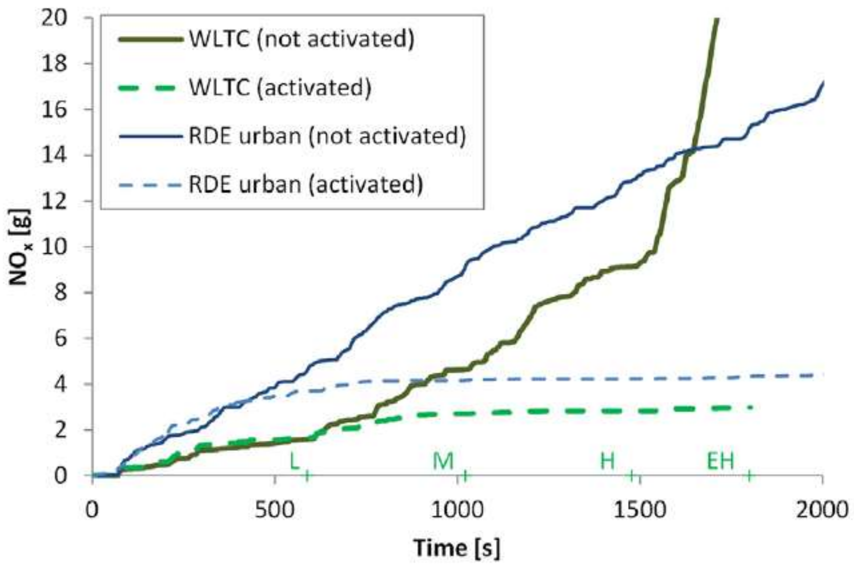


Figure 6.2. Euro 5 vehicle with a DOC + DPF vehicle retrofitted with an SCR system (Giechaskiel et al, 2018).

Particulate filters for gasoline retrofit remains possible and show potential provided passive regeneration can be achieved and packaging constraints met. For example, Chan et al (2012) showed filtration efficiencies of ~80% for a non-catalysed GPF retrofitted to a US Tier 2 standard vehicle.

6.5 Improvements to Periodic Technical Inspection (PTI) of Existing Vehicles

In early DPF-equipped passenger car diesel vehicles, city-only driving can lead to DPF blocking, a situation where the DPF can only be regenerated at the vehicle manufacturer's dealership. Such issues can lead to elevated user costs and inconvenience. Consequently, there has been interest in, and a proliferation of, services offering the removal of DPF hardware and the associated modification of the engine management system and on-board diagnostic systems to maximise fuel economy, while ensuring vehicle performance and driveability are retained, as exemplified by the website mecs.io[18]. This issue has generated considerable media coverage[vi] (The Guardian, 2016) in the past few years, with the European Commission suggesting up to 10% of Euro 5 diesels, or later, in Switzerland, the Netherlands and Belgium have damaged, modified or removed DPFs (Suarez-Bertoa et al, 2018).

Since February 2014, the periodic technical inspection (PTI) test, or MOT test in the UK, has required a visual check for the presence of a DPF[19] relevant vehicles. This check has proven largely ineffective, as in many cases the exhaust can containing the DPF is inaccessible, or the can is still present but replaced after the particle filter has been removed. Additionally, the current MOT method for visible smoke (EU Directive 2014/45/EU, 2014)[20] emissions of diesel vehicles, is insufficiently sensitive to identify either if a DPF has been damaged (intentionally or otherwise) or if a DPF has been removed. Hence such vehicles can exhibit substantially increased particle emissions compared to their design, but still pass the MOT test/PTI.

For Euro 5b, in September 2011, a non-volatile particle number-based (PN) methodology was introduced to European emissions type-approval legislation for light-duty diesel vehicles. The exercise conducted to validate the PN methodology (Andersson et al, 2007) demonstrated substantially increased sensitivity compared with gravimetry, and a separation factor of around 100x between non-DPF and DPF-equipped diesel particle number emissions levels (Figure 6.3). During this study, PN emissions were measured in a chassis dynamometer facility, from a dynamic driving cycle, and so any particle number-based PTI application, if applied to a static vehicle, would require a modified measurement approach for measuring with the engine at idle.

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[18] https://www.mecs.io/dpf-removal-uk.html, accessed 13th January 2020.

[19] https://www.gov.uk/government/publications/diesel-particulate-filters-guidance-note.

[20] DIRECTIVE 2014/45/EU OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL (of 3 April 2014) on periodic roadworthiness tests for motor vehicles and their trailers and repealing Directive 2009/40/EC.

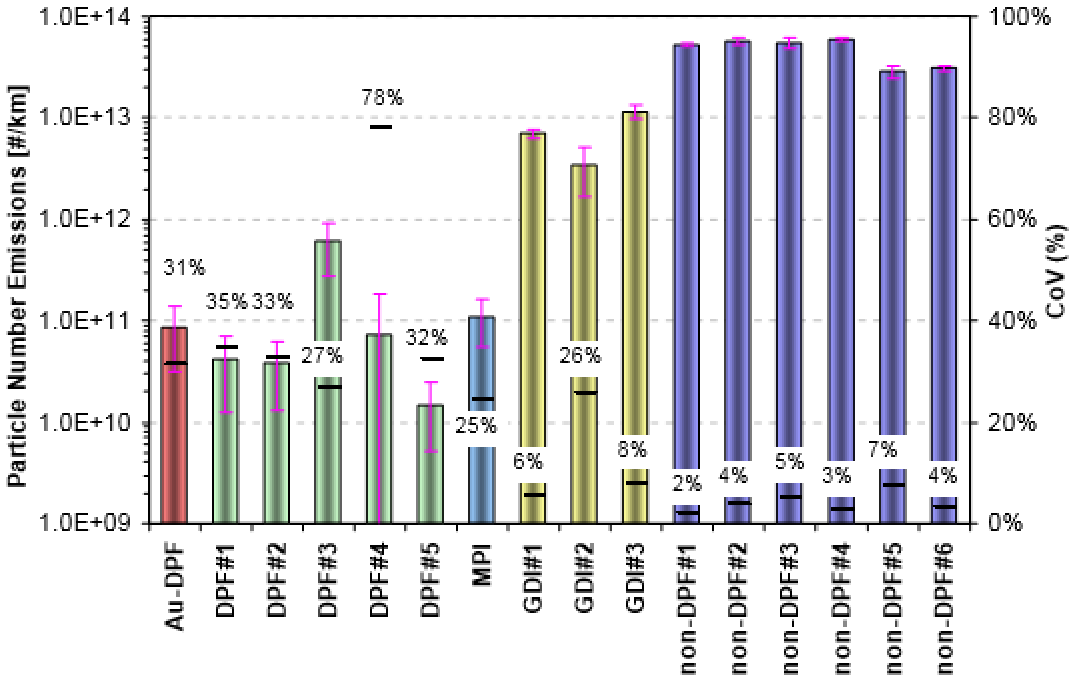

Figure 6.3. Non-DPF diesel emissions are at least 100x higher than with DPF.

In a report for the Department for Transport, Andersson and Keenan (2015) concluded that it appears viable to use a particle number measurement approach to discriminate a vehicle equipped with a DPF from the same vehicle with the DPF removed. The approach considered no change to the current UK MOT procedure, except requiring the measurement of non-volatile particle number concentration rather than the currently required smoke opacity during the free-acceleration smoke test[21].

In order to achieve this, the exhaust sample would need to be pre-conditioned so that only non-volatile particles were measured, and the size range addressed by the particle counter would ensure that any judgement is based upon filtration of soot particles that are most abundant above 20 nm. Particle number-based methods are highly sensitive, so measuring directly from raw exhaust under the positive exhaust flow produced with engine load can ensure virtually no background contribution to the particle count, and consequently very high discrimination power of the method. Based upon chassis dynamometer data, non-DPF particle concentrations under free acceleration were estimated to be between 10^6 and 10^7 particles/cm^3, with post-DPF levels being ≤ 10^4/cm^3. These data indicate that a pass/fail threshold around 10^5/cm^3 (~10% of the engine-out levels and 10x the DPF levels) would be suitable to identify the absence of a DPF.

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[21] https://www.nidirect.gov.uk/articles/vehicle-test-procedures, accessed 13th January 2020

Other non-volatile particle number-based approaches for PTI are being applied, or are under consideration, in Europe. A white paper led by the VERT Association (VERT, 2018), a Swiss organisation dedicated to the promotion of Best Available Technology for emission control, concluded "In order to secure the long-term emission performance of catalysts and filters, regular maintenance and periodic technical inspections are needed. The current PTI smoke emission test for diesel vehicles is not suitable to judge the emission performance of DPFs." (VERT, 2018).

The main differences between the approach of Andersson and Keenan and the European approaches, are the application of cheaper methods for particle measurement, including the use of particle charging and charge measurement, potentially avoiding the expense of components within the measurement system for the elimination of volatiles. The European approaches also address measuring at idle, or raised idle, rather than the free-acceleration method used for current smoke measurement.

The Dutch PTI DPF Programme that is running between 2013 and 2019 has been described by Kadjik (2018). The objectives are:

- Definition of a relevant emission test
- Definition of a feasible PN limit value
- Definition and specification of a low cost PN-tester

All these aspects must be treated holistically to determine a realistic procedure.

In a screening exercise of 220 public-owned DPF-equipped vehicles, 161 vehicles (76%) showed PN emissions of <5000 #/cm^3, 52 vehicles (24%) had PN emissions of 5000 to 250,000 #/cm^3. 10% of the vehicles showed PN emissions of > 250.000 #/cm^3.

Particle number concentration at (warmed up) low idle speed was shown to correlate with NEDC PN emissions (at least for vehicles with no, or failed/leaky, DPF; Figure 6.4). On this basis a pass/fail threshold based upon particle number concentration measured at idle appears viable, and measured concentrations can be linked to certification cycle emissions in #/km.

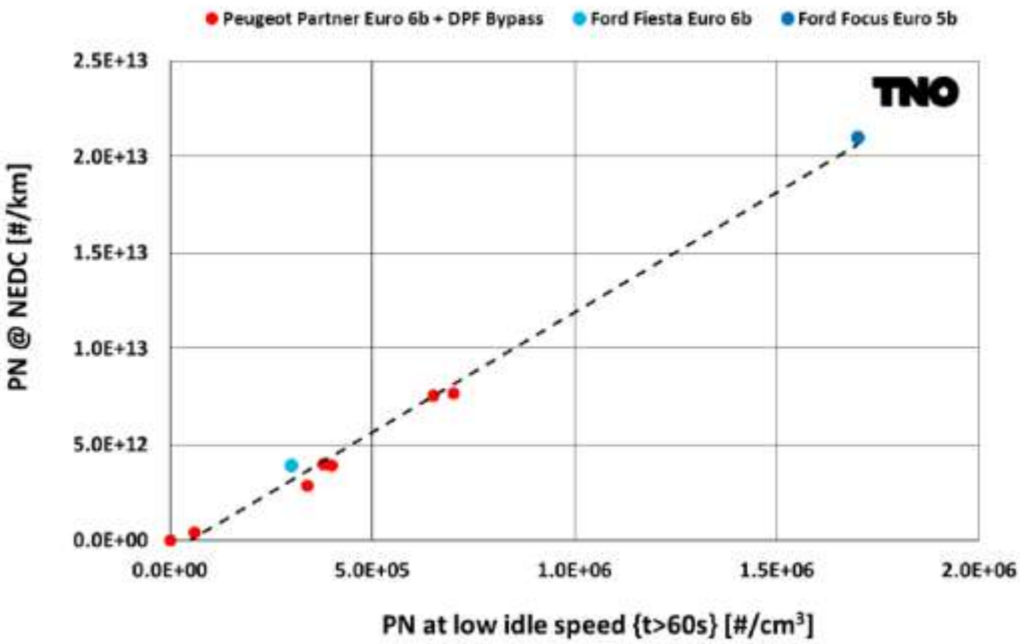

Figure 6.4. #/cm^3 (hot engine, low idle) correlates with #/km (cold NEDC; Kadjik (2018)

A test procedure was therefore proposed: continuous particle number concentration measurement is conducted starting with ambient air (20 s) and then switching to a tailpipe sample taken from a vehicle with the engine at hot idle (60 s sample), before returning to ambient.

A possible limit value might be between 2.5×10^5 #/cm^3 and 1.5×10^6 #/cm^3, but the measured concentration will depend on the specific measurement device used. A proposal for the equipment specification is that it measures:

- Solid (non-volatile) particles
- A size distribution with a mode at 70nm

An initial proposal for PTI is given in Table 6.1 below:

Table 6.1: Proposed Particle Concentrations for Limit Values at PTI (Kadjik (2018))

Euro class	Type Approval & In Service Conformity (Manufacturer)		PTI (Vehicle Owner)
	PM [mg/km]	PN [#/km]	PN [#/cm^3]
3, 4, 5a	5.0	-	1.000.000
5b, 6	4.5	$6 * 10^{11}$	100.000 to 500.000

Fierz and Rüggeberg proposed the use of the low-cost charge-based particle sensor technology for diesel DPF PTI. Particles entering the instrument enter a charging zone, collect charges in a predictable manner and then carry those particles to a detector. The Partector is tuned for high detection efficiency of soot particles >20 nm. Since diesel engines run fuel lean (lambda ~10) they produce exhaust with low water content, even at steady state, low engine speed and load (low idle). Low idle is desirable to avoid any need at PTI for driver intervention (for example to depress the accelerator pedal and increase engine speed). In exhaust with low water content, such as diesel low idle, the need to eliminate volatiles, which will also collect charges and be detected as particles, is reduced and the detector can therefore run at ~40°C, simplifying the hardware, reducing weight, warm-up time and reducing power demand. A light-weight, simple hand-held device is therefore possible, suitable for use in MOT stations. The instrument can measure ~10^3#/cm^3 to ~$5x10^6$#/cm^3, ideal for a pass/fail threshold in the range 10^5 to 10^6#/cm^3. The instrument will not be accurate in comparison with the regulatory PN approach, but suitable for pass/fail PTI. This instrument would not be suitable for gasoline fuelled engines, where more fuel is burned per mass of air and this generates high levels of water that would condense and flood the instrument during sampling.

Suarez-Bertoa et al, 2018 stated that the current MOT opacity method is not considered to be suitable to determine even complete DPF removal. To explore PTI options, several PN-based methods were used measuring directly from raw exhaust at low idle, and at raised idle (2000 rpm). Some instruments sampled using just dilution and a detector, while others used dilution, volatile removal and detection. Both diffusion charger and particle counter methods of enumeration were tested. At both low idle and high idle all instrument responses were linear (R^2 ranging from 0.95 to 0.99 for 6 instruments) when compared to a 23 nm regulatory PN system measuring from diesel exhaust. It was concluded that both condensation nucleus counter and diffusion charger-based measurement systems were viable candidates, but that non-volatile PN should be measured, especially if the method must also address gasoline particle filters in the future. Low idle was a suitable engine mode at which to assess presence of removal of a DPF. More work is required to determine a threshold emissions value for a failed/removed DPF, though the data presented suggests this would be between $5x10^4$#/cm^3 and 10^6#/cm^3.

In 2017 Switzerland implemented a PTI check for non-road mobile machinery (NRMM) based upon the application of a non-volatile particle number measurement system (D'Urbano and Bonsack, 2018). One permitted instrument[22] is essentially identical in function to the in-lab regulatory PN measurement system, including a condensation nucleus counter, but is portable. The Swiss administration, through regulation SR 941.242, mandates compliance testing for off-road vehicles and the procedures include specifications

[22] https://www.tsi.com/npet/, accessed 13th January 2020

for the measurement equipment[23], test procedures and reference value[24]. The regulation defines the particles to be measured as "Solid, carbonic components of the hot exhaust gas in the exhaust pipe of combustion engines … (with) a mobility diameter in the range from 20 nm to 300 nm. The volatile portions are not considered as nanoparticles." Particle measurement equipment is subject to a rigorous calibration and maintenance regime, similar to that required for certification equipment.

Measurements in the Swiss PTI test are made from raw exhaust at high idle, with a pass/fail threshold "reference value" generated by equating raw exhaust particle concentrations to the certification Non-road Transient Cycle limit value of $1x10^{12}$ particles/kWh. The resultant threshold value is $2.5x10^5$ particles/cm^3. Use of the method has shown clear discrimination between DPF equipped and non-DPF equipped engines, and the method also identified a 'failed DPF' (Figure 6.5).

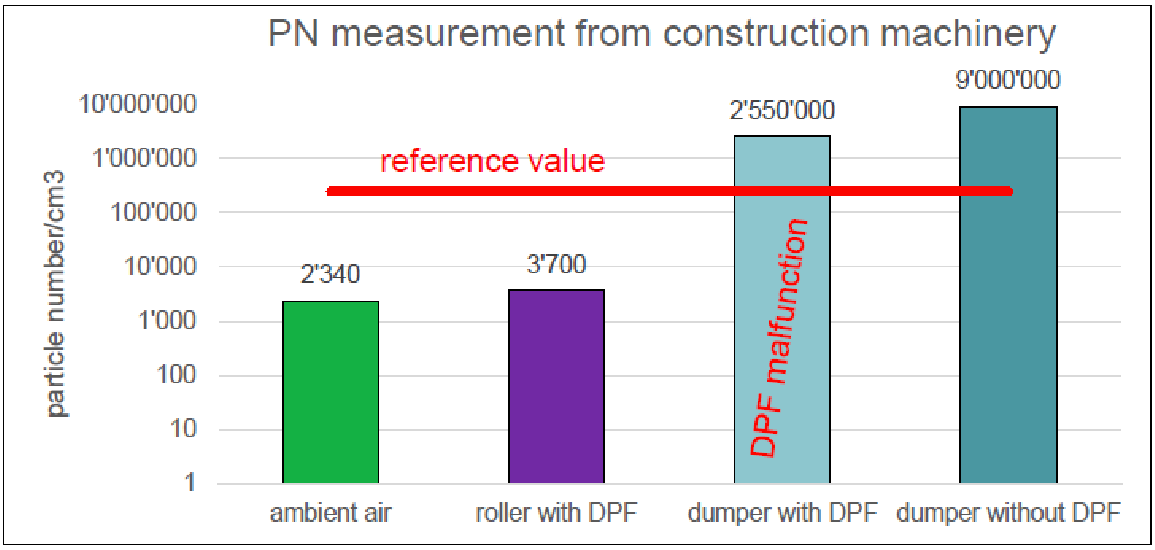

Figure 6.5: Results from Swiss NRMM PN PTI Approach indicating DPF, Non-DPF and Failed DPF measurements (D'Urbano and Bonsack, 2018)

In conclusion, the Swiss non-road measurement experience, and research studying on-road passenger car diesel emissions, clearly indicates that an approach employing particle counting (or similar metric) could be used to identify the removal of DPFs during a static, MOT emissions test. One vehicle with illegally removed DPF would emit particulate emissions equivalent to 10-1000 DPF-equipped vehicles. The predominant view in Europe is that the measurement should address non-volatile particles, but this will require measurement devices of high cost. The alternative approach exemplified by Fierz and Rüggeberg may also be a viable alternative for diesel applications.

[23] https://www.admin.ch/opc/de/classified-compilation/20051389/index.html, accessed 13th January 2020
[24] https://www.bafu.admin.ch/bafu/de/home/themen/luft/publikationen-studien/publikationen/luftreinhaltung-auf-baustellen.html, accessed 13th January 2020

Further work would be required to establish the "reference value", as used in the Swiss NRMM regulation, for passenger car diesels, as this will be instrument dependent, but data suggests this would lie in the range $5x10^4$ to $1.5x10^6$ particles/cm^3.

6.6 Impacts of connectivity and geofencing

Increased vehicle connectivity offers opportunities for more complex management of urban transport. The presence of multiple energy stores on hybrid vehicles allows decisions to be made, currently within the vehicle, as to which energy source to use out of typically the fuel tank and combustion engine, or battery and electric drive. Factors such as state of charge, fuel reserves, driver led decisions (e.g. driving modes), ancillary demands (e.g. air conditioning) can inform the vehicle's decision as to which energy source to use. Interconnected cities and transport systems can allow external information such as current local air quality to influence the mode of operation of the vehicle and energy source used.

Applications of geofencing have been proposed to include to manage clean air zones (Foss et al, 2019) and to create emission-based charging zones (Wu and Sperling, 2018). Limited data is yet available about the relative success of information providing services versus charging based systems. Success will likely depend on development of integrated smart transport systems which remain disaggregated. Successful implementation of connective systems to influence vehicle operation should be beneficial to displacing emissions from areas of lowest air quality through increasing the number of vehicles running in low emission modes in those regions.

6.7 Fuel composition and origin

Figure 6.6 shows a fuels roadmap (Ricardo Internal Data, 2019) on how the evolution of internal combustion engine future fuels may progress towards 2040. The roadmap shows, increasing levels of renewable ethanol for gasoline fuels as a drop in component moving towards niche (renewable) power or waste to fuels production. Diesel shows a drop in renewable content of up to 7% with a similar niche as gasoline towards (renewable) power or waste to fuels production. Increasing the bio content of gasoline and diesel can reduce well to wheels CO_2 for the existing parc and require no base engine changes.

Natural gas naturally is low CO_2 forming compared to gasoline and diesel due to its favourable carbon to hydrogen ratio and will be used for certain applications. Renewable methane can further reduce the well to wheels CO_2. The final step for internal combustion engines is to use renewable H_2 as the fuel, to significantly decarbonise engines.

Figure 6.6: Ricardo Future Fuels Roadmap (Ricardo Internal Data, 2019)

6.8 Summary

Future emissions legislation will require more complex aftertreatment systems, in terms of both hardware and control. This will, however, facilitate reduced tailpipe emissions, for both gaseous and solid emissions, and will lead to a reduction on the impact of air quality especially in urban areas.

Retrofit technology mainly for Diesel applications will be available to reduce tailpipe NO_X emissions of Euro 5 vehicles tending towards those of Euro 6 diesel applications. This is being driven by local air quality requirements in certain European cities.

Connected vehicles of the future will be flexible enough to deliver zero emissions in geofenced areas, such as city centres and areas of poor air quality and operate on an internal combustion engine where required.

The evolution of fuels is tending towards more renewable hydrocarbon-based content and a reduction in carbon content. Moving to fully synthetic fuels produced from renewable sources will limit the increase in carbon emissions.

7 RECOMMENDATIONS

- The management of air pollution emissions from vehicle exhaust is only one part of a complex set of decisions needed to optimise and minimise the wider environmental impacts of road transport. It is important that exhaust emissions of air pollutants are placed in a wider systems context that also considers vehicle non-exhaust emissions of particles and the overall energy efficiency of the vehicle, from original energy source through to final propulsion. The air pollution and greenhouse gas consequences of vehicle manufacture, full lifecycle operation and final disposal should also be considered in decisions on future vehicle emissions standards.

- Engineering improvements in catalyst formulation and engine calibration optimization should continue to be pursued since these may lead to reduced emissions during transient driving events such as acceleration, and improved control over the species released from exhaust systems.

- There is currently only limited information available on the air pollution emission behaviour of more advanced and emerging vehicle technologies and powertrains, for example associated with hybrid vehicles. As these grow in number in the UK fleet, the ability to fully represent their emissions will be an important component of estimating impacts and predicting air pollution, particularly in urban locations.

- The UK ambient air monitoring networks are designed primarily for the purposes of demonstrating compliance with air quality standards and are not ideally structured or located for the more specific purposes of evaluating vehicle exhaust emissions. A consequence is that trends in vehicle emissions over time, or the impact of interventions, including clean / low emission zones can be difficult to quantify and that only pollutants which are of direct interest for LAQM are routinely measured at many sites. A useful approach for examining the impact of interventions in exhaust emissions is to pair monitoring sites (e.g. urban background and roadside) and combining this with measurement of traffic data at the roadside sites. The routine co-location of remote sensing observations, vehicle counts and ambient air quality monitors would allow the relationships with between these to be routinely evaluated. An alternative is to make very high time resolution (seconds) pollutant measurements allowing scale separation of very local and background emissions.

- The next set of European vehicle emissions legislation will require a new set of exhaust tailpipe species to be considered for control as well as lowering emission limits of existing regulated pollutants. If new pollutants are introduced into the emissions standards for vehicles, some revision of ambient monitoring infrastructure (e.g. for NH_3) will be required to evaluate the behaviour of these pollutants. This has proven critical in the past to independently verify emissions performance.

- Whilst more data is emerging on real-world emissions in general, more measurements are needed, in particular to assess whether increased vehicle hybridisation is impacting on emissions and air quality. The increasing trend for vehicle automation and vehicle connectivity is in its early phases, but an active programme of evaluation of the impacts of these technologies would provide valuable foresight of their potential effectiveness in delivering air quality benefits.

- With an increasing amount of research on real-world emissions using PEMS and remote sensing, there is a need to bring this information into inventories more quickly to allow models to account for the wide variety of traffic situations and vehicle technologies. The measurement data are somewhat fragmented and often held on proprietary databases, and the limitations in PEMS data are not necessarily fully described in all datasets or for the technique as a whole. There is an urgent need for open access, and peer-reviewed datasets on real-world emission factors expressed with associated uncertainties.

- Predictive air quality modelling using emissions derived from the average speed approach is currently used for modelling and Local Air Quality Management (LAQM) assessments. This is for reasons of practicalities with the availability of suitable traffic activity data and emission factors and with cost and consistency considerations. However, this approach limits the ability to evaluate interventions that impact transient driving conditions. Moreover, to understand emissions locally and quantify the effect of local policies, for example restricting vehicle movements according to fuel type or Euro class, data on local differences in the fleet composition will be required.

- Prediction of air pollution would be improved if emissions inventories used more granular information, specifically related to vehicle activity data and emission factors that reflect technological differences between vehicles and dependencies on environmental factors such as ambient temperature. Greater access to local fleet and vehicle activity data from sources such as automatic number plate recognition (ANPR), or satellite navigation companies, or vehicle ownership would enable more accurate local emission inventories and models to be developed, potentially improving decision-making around city schemes such as clean air zones.

- Any approach used for modelling exhaust emissions needs to be consistent with the purpose for which the emission estimates are intended to be used. An improved understanding of instantaneous emissions and how they are affected by driving and operating conditions is better suited for more local scale modelling and on individual road sections where higher spatial and temporal resolution in emissions are required.

- Traffic management interventions and urban planning and development that can reduce traffic congestion, and therefore minimise vehicle acceleration / deceleration events would be beneficial for air quality. Similarly, air quality impacts need to be assessed when developing new traffic management schemes to avoid inadvertent detrimental effects. Improved modelling of the impacts of vehicles' changing speeds would allow these air quality benefits to be quantified more directly and with greater confidence.

- Use of existing retrofit options which show the ability to bring older generation vehicles up to or beyond emission levels of newer vehicles should be explored, noting the need to also consider emissions from the aftertreatment systems themselves. Retrofitting of modern diesel vehicles (Euro 5 and later) with NO_X control is technically possible. It is easier to exclude these from urban areas than it is to economically retrofit them. However, given the typical lifetimes for diesel vehicles are between 10 and 15 years, widespread retrofitting of Euro 5 and possibly early Euro 6 diesels, incentivised by reduced congestion charging, should result in lower NO_X emissions outside cities too. This is likely to require research to establish a cost-benefit analysis.

- Improvements to Periodic Technical Inspection (PTI)/MOT methodologies would allow detection of modified/removed particulate filter systems which are undetectable during the current inspections.

List of Acronyms

AADF	annual average daily flows
ANPR	Automatic Number Plate Recognition
ARTEMIS	Assessment and Reliability of Transport Emission Models and Inventory Systems
ASC	Ammonia Slip Catalyst?
ATS	Aftertreatment Systems
AURN	Automatic Urban and Rural Network
BC	Black Carbon
BEIS	Department for Business, Energy and Industrial Strategy
CAZ	Clean Air Zone
CERC	Cambridge Environmental Research Consultants
CI	Compression Ignition
CLRTAP	Convention on Long-Range Transboundary Air Pollution
CNG	Compressed Natural Gas
CO	Carbon monoxide
CO_2	Carbon dioxide
COPERT	Computer Program to Calculate Emissions from Road Transport
CRT	Continuously Regenerating Traps
Defra	Department for Environment, Food & Rural Affairs
DfT	Department for Transport
DMRB	Design Manual for Roads and Bridges
DOC	Diesel Oxidation Catalyst
DPF	Diesel Particulate Filter
DUKES	Digest of UK Energy Statistics
EC	Elemental Carbon
EEA	European Environment Agency
EF	Emission Factor
EFT	Emissions Factor Toolkit
EGR	Exhaust Gas Recirculation
EHC	Electrically heated catalyst
ERMES	European Research for Mobile Emission Sources
EMEP	European Monitoring and Evaluation Programme
FID	Flame Ionization Detector
FTIR	Fourier Transform Infra-Red
GDI	Gasoline Direct Injection
GPF	Gasoline Particle Filters
HC	Hydrocarbon
HDV	Heavy Duty Vehicles
HGVs	Heavy Goods Vehicles >3.5 tonnes gross vehicle weight
HBEFA	Handbook of Emission Factors
HMRC	Her Majesty's Revenue and Customs

H_2O	Water
ISC	In-Service Compliance
JAQU	Joint Air Quality Unit
LAEI	London Atmospheric Emissions Inventory
LAQM	Local Air Quality Management
LAQN	London Air Quality Network
LDV	Light Duty Vehicles
LGVs	Light Goods Vehicles < 3.5 tonnes gross vehicle weight
LNT	Lean NO_X Trap
LowCVP	Low Carbon Vehicle Partnership
LPG	Liquefied petroleum gas
MOVES	Motor Vehicle Emission Simulator
NAEI	National Atmospheric Emissions Inventory
NDIR	Non-Dispersive Infra-Red
NDUV	Non-Dispersive Ultra-Violet
NECD	National Emissions Ceilings Directive
NEDC	New European Drive Cycle
NEMO	Network Emission Model
NGO	Non-Governmental Organisation
NH_3	Ammonia
NMVOCs	Non-methane volatile organic compounds
NO	Nitric oxide
NO_x	Oxides of nitrogen
NO_2	Nitrogen dioxide
NTE	Not to exceed
NTM	National Transport Model
PCM	Pollution Climate Mapping
PEMS	Portable Emissions Measurement Systems
PFI	Port Fuel Injection
PHEM	Passenger car and Heavy duty Emission Model
PN	Particle Number
PNA	Passive NOx Absorber?
PTI	Periodic Technical Inspection
QCL	Quantum Cascade Laser
RDE	Real Driving Emissions
RT	Road Transport
RTFO	Renewable Transport Fuel Obligation
SCR	Selective Catalytic Reduction
SCRF	SCRF + soot filter?
SEMS	Smart Emissions Measurement System
SI	Spark Ignition
TfL	Transport for London
THC	Total Hydrocarbons

TREMOD	Transport Emission Model
TWC	Three Way Catalyst
TWGPF	Three-way gasoline particle filter?
TWLNT	Three-way lean NOx trap?
UNECE	United Nations Economic Commission for Europe
vkm	Vehicle kilometre
VSP	Vehicle Specific Power
Vpd	Vehicles per day
WLTC	Worldwide harmonised Light duty vehicles Test Cycle
WLTP	Worldwide harmonised Light duty vehicles Test Procedure

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