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Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Improved Emissions Inventories for NOx and Particulate Matter from Transport and Small Scale Combustion Installations in Ireland (ETASCI)

Part 1 of 2*

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Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

www.epa.ie

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DISCLAIMER

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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Executive Summary

The ETASCI project is divided into **two** complementary but distinct work packages (WPs) and final reports. **WP1** focuses on emissions from road transport and is detailed in EPA Research Report 148 – Part 1 of 2. **WP2** examines emissions from non-road transport and from small combustion installations (SCIs) and is detailed in EPA Research Report 149 – Part 2 of 2.

Anthropogenic emissions of NO_x and particulate matter (PM) have been shown to exert serious environmental and health effects. In Ireland, road transport is thought to be the dominant source of these emissions, accounting for an estimated 40%-50% of the national total. The primary objective of this project was to develop improved methods for estimating NO_x and particulate matter (PM) emissions from road vehicles in Ireland. However, accurate determination of NO_x and PM emissions from this sector is challenging, and is burdened with uncertainty.

The difficulty arises from three main sources:

1. The heterogeneity of the vehicle fleet;
2. Accurate estimation of the associated activity data;
3. The **strong, non-linear** dependence of NO_x and PM emission factors on the specific, detailed characteristics of each vehicle, and on exogenous factors, such as drive-cycle kinematics, driver behaviour, ambient temperature, and road topography.

Whilst acceptably accurate records of fleet composition are available, little activity data on road transport is gathered in Ireland; estimates of such activity therefore contain significant embedded uncertainty. Moreover, recent measurements suggest that real-world emission factors may differ significantly from those assumed in transport models, such as Computer Programme to calculate Emissions from Road Transport (COPERT) or Handbook of Emission Factors (HBEFA). This is particularly true of diesel-powered vehicles, in the absence of effective tailpipe treatment systems. Despite ever-tightening legislative emission limits – as measured on a simple, standardised driving cycle – real-world emission factors for these vehicles remain highly uncertain.

Given these constraints, estimates of PM and NO_x emissions from road transport are inherently tentative; the best that can be done is to quantify, and where practical to minimise, the level of their uncertainty.

Attempting to address this problem, the European Commission established the ARTEMIS project (Assessment and Reliability of Transport Emission Models and Inventory Systems) in 2000. By 2007, when the current ETASCI project (Emissions from Transport and Small Combustion Installations) began, ARTEMIS was nearing completion. By combining the ARTEMIS findings with real-world, tailpipe emission measurements from the Irish fleet, ETASCI aimed to establish a significantly more accurate model of NO_x and PM emissions from road transport in Ireland.

Unfortunately, the onset of the Irish financial crisis precluded funding of the associated Portable Emissions Measurement System (PEMS); hence the collection of real-world, Irish, tailpipe emission data was not possible. Moreover, although the ARTEMIS project delivered significant new information and insight, it gradually became clear that the methodology it proposed for determining tailpipe

emissions relies on a wide-ranging, detailed and accurate matrix of inputs. Gathering, organising, and maintaining these input data would require a very substantial investment in new resources and capabilities, which were outside the scope of this project.

Within ETASCI, various approaches to circumventing these problems were attempted, in order to develop a deeper understanding of the underlying issues. The key findings of ETASCI, with respect to road transport, may be summarised as follows:

1. The method currently used to estimate NO_x and PM emissions from Irish road transport contains a high level of embedded uncertainty.
2. A methodological cure for this problem would require very substantial investment in resources and capabilities.
3. Over the short-term (next 5-10 years), therefore, a high level of uncertainty will continue to be associated with such estimates, and should be indicated appropriately.
4. However, the diffusion of new technology into the vehicle fleet over the medium term (10-20 years) will reduce, very substantially, both the absolute level and the variability of these emissions.

ETASCI therefore supports the continued use of COPERT (with latest available emission factors), but recommends that:

1. A programme to monitor real-world driving emissions from road transport, using remote data sensing (RDS), portable emissions measurement systems (PEMS), or both, should be considered.
2. Every effort should be made to reduce the uncertainty of activity data, and fleet composition, by leveraging existing data sets and resources.

The high levels of uncertainty associated with NO_x and PM emission estimates should be appropriately indicated.

1. Introduction

1.2 Overview

Vehicle emissions are a very significant contributor to poor air quality, particularly in urban areas (Reynolds, 2000), where congestion and heavy traffic compound the problem. Specifically, particulate matter (PM) and Nitrogen Oxide (NO_x) emissions from vehicles can have serious health effects (Kraft et al., 2005). The development of an emissions inventory, to quantify the total emissions from a region, is important for understanding the scale of the problem, and ultimately for reducing these emissions. Ireland, as a member of the European Union (EU) and a party to the Convention on Long-Range Transboundary Air Pollution, is required to estimate and report these emissions, and to comply with emission ceilings under the National Emissions Ceilings Directive (EPA, 2011).

From the period 1990 to 2010, the total number of mechanically-propelled vehicles registered in the Irish fleet has more than doubled, increasing from 1,054,259 to 2,416,387 vehicles respectively (DTTAS 2010). This has resulted in a considerable increase in traffic congestion, and associated air emissions, in most urban areas throughout Ireland.

Modern vehicles tend to exhibit an increasing variability in emissions output, as a result of more complex engine and emissions control technology. This leads to generally low emissions levels, with occasional peaks of emissions during periods outside of "normal", or standard operating conditions. Such peaks can occur following a cold start, or when climbing a gradient, for example; emissions levels associated with these events may be orders of magnitude greater than nominal, steady-state values (Smit et al., 2006). Recent studies (Weiss et al., 2012, Franco et al., 2013, Kouridis et al., 2009) further suggest that many "normal" or typical driving cycles may produce emission levels well above those measured on the simple, standardised test cycle used for vehicle certification. This strong, non-linear relationship between emission level, engine operating condition, driving cycle, and ambient temperature, presents a significant challenge to the reliable estimation of national emissions.

1.3 Emissions inventories

The production of an accurate and reliable emissions estimate for all road transport in Ireland (which consists of 2.5 million vehicles, each producing a wide variation in emissions quantity) presents a significant challenge. Several software packages have been developed to assist with the creation of an emissions inventory from a specific area, for a specific time period. The two main programmes used within Europe are COPERT – which assumes that emissions from a given vehicle are essentially a function of its average speed – and ARTEMIS/HBEFA, which bases emissions estimates on a more detailed and sophisticated representation of the vehicle drive cycle.

In addition to these two inventory models – which are focused on road transport only – TREMOVE[1] is used to estimate overall transport demand, modal split, vehicle stock turnover, and vehicular emissions of air pollutants under different policy scenarios. The emission factors

[1] TREMOVE is a policy assessment model, developed on behalf of the European Commission, to study the effects of different transport and environment policies on the emissions of the transport sector.

used in TREMOVE for the road transport sector are closely based on those developed for COPERT.

All emission inventory programmes use the same basic methodology to estimate emissions. That methodology may be summarised as follows:

Total emissions per year =	(emissions per vehicle, per km travelled) x	
	(km travelled per vehicle, per year) x	
	(number of vehicles)	

This can be expressed as:

Total emissions per year =	(vehicle emission factor) x (vehicle activity)	= ***EF*** x ***A***

It is evident that the accurate estimation of fleet emissions requires accurate estimates for both emission factor *EF,* and activity data *A*. Unfortunately, the emission factor for both PM and NO_x is very sensitive to vehicle design, and accurate allocation of *EFs* requires a large number of vehicle classes. However, the existence of a large number of vehicle classes creates a corresponding requirement for highly granular activity data: accuracy in only one of the two parameters will not deliver an accurate estimate of total emissions.

Moreover, it happens that vehicular *EFs* vary with operating mode (activity type), so that it becomes necessary to estimate *EFs* for a range of discrete operating modes, and to weight the *EF* for each mode according to its specific activity level *A*. The approach most commonly used characterises activity type using the single parameter of vehicle average speed, and ignores all other driving kinematics (as described in Section 5). This is the approach adopted by the COPERT model (Computer Programme to calculate Emissions from Road Transport), which is currently used in Ireland and in many other European countries to estimate emissions from the road transport sector.

A second approach, less widely used, acknowledges that vehicle kinematics other than average speed can strongly influence exhaust emission levels, particularly with respect to NO_x and PM (de Haan and Keller, 2000). This more sophisticated approach attempts to incorporate these kinematic variables by defining a wide range of activity types, each associated with a specific “Traffic Situation” (TS). The European Union ARTEMIS programme was established to develop a software model incorporating the TS methodology, with the ultimate objective of implementing the model in all member states, and thereby producing harmonised emission inventories for the transport sector.

Following conclusion of the ARTEMIS project in 2007, a new emissions inventory model was delivered by INFRAS (a Swiss consultancy company and partner in the ARTEMIS consortium), but model stability was poor, supporting documentation was weak, and availability of technical assistance was very

limited. Attempts to implement the ARTEMIS model within the ETASCI project proved frustrating and ultimately fruitless, despite spending time at the site in France where the model was developed. However, INFRAS subsequently secured funding to further develop the software under the HBEFA (Handbook of Emission Factors) project. This resulted in a software name change, to HBEFA (v3.1). HBEFA was developed by INFRAS initially for use by the Swiss, German, and Austrian authorities, and subsequently has been used by several European countries and by the Joint Research Centre of the European Commission. The updated HBEFA (v3.1) model was supplied to ETASCI during the site visit to France, and has subsequently been used in place of ARTEMIS for Irish emissions estimation.

1.4 Report objectives

The primary objective of this project was to develop improved methods for estimating NO_x and particulate matter (PM) emissions from road vehicles in Ireland. The COPERT inventory model is currently used by the Irish Environmental Protection Agency (EPA) for this purpose, although it is clear from the discussion above and from recent research (Barlow and Boulter, 2009) that it suffers from significant shortcomings. In the context of NO_x and PM emissions, these shortcomings are of particular concern, especially given recent advances in engine technology and exhaust emissions after-treatment. These technological developments can result in a vehicle with relatively low emissions under “standard” operating conditions, exhibiting a much higher emission intensity when operating “off cycle”. This produces large emission factor (*EF*) variations that are dependent solely on the vehicle operating condition.

In this study, the authors investigate the impacts of uncertainty in the input data (fleet composition, emission factor, and activity data), and that of the modelling methodology selected (COPERT or HBEFA), on estimates of national emission inventories for NO_x and PM. Inventory estimates for the current fleet were generated using both models, and their sensitivity to inputs assessed and compared. The HBEFA model was then used to project NO_x and PM emissions from road transport forward to 2020 for several scenarios.

Having identified the paucity of activity data as a significant contributor to inventory uncertainty, various methods of logging vehicle activity data have also been examined. A low-cost method of gathering such data was developed, and used to gather real-life activity data from some representative Irish vehicle trips. Various techniques for extracting kinematic parameters from the raw trip data were evaluated, and the effect of parameter choice and extraction method, on the level of emissions predicted, was assessed.

2 NO_x and PM emissions from road transport

2.1 Overview

2.1.1 Environmental and health effects of PM and NO_x emissions

Particulate matter (PM) emissions from road transport are primarily composed of combustion-derived carbonaceous material, onto which organic compounds are adsorbed (Heywood, 1988). Particulates span a wide range of sizes, but are typically classified as either PM_{10}, $PM_{2.5}$, or $PM_{0.1}$, where the subscript refers to the maximum aerodynamic diameter of the particle in micrometres (µm). Abrasion processes, such as tyre wear, brake wear and road surface wear, constitute a secondary source of (mostly larger scale) particulates (Boulter, 2007).

Respiration of PM can have serious health implications (WHO-Europe, 2004). Whilst particles larger than 10 µm are usually filtered or deposited in the nasal passage or throat, PM_{10} can penetrate into the bronchi and lungs, and $PM_{2.5}$ can reach the terminal bronchiole and alveoli (Annesi-Maesano et al., 2007). $PM_{0.1}$, sometimes referred to as ultrafine particles, can eventually spread into the systemic circulation through the alveolar-capillary membrane (Martinelli et al., 2013). The landmark Harvard Six Cities and American Cancer Society studies found that $PM_{2.5}$ was associated with mortality for both cardiovascular and pulmonary causes, but that $PM_{10-2.5}$ was not associated with mortality (Dockery et al., 1993; Pope et al., 1995; Pope et al., 2002).

$PM_{2.5}$, which is associated primarily with combustion of fossil fuels, has therefore become the principal focus of legislative attention.

NO_x has several serious environmental and health implications, which include the following (WHO-Europe, 2004; Kraft at al., 2005):

- It is a greenhouse gas and contributes to global warming.
- It causes smog and acid rain.
- Particles – NO_x can react with ammonia, moisture, and other compounds to form nitric acid and other similar particles.
- Human health concerns include effects on breathing and the respiratory system, damage to lung tissue, and premature death.

2.1.2 Road transport emissions in a national context

As noted under the Ireland National Climate Change Strategy 2007 (DEHLG 2007), the Environmental Protection Agency of Ireland (EPA) is responsible for compiling and reporting annual estimates of pollutant emission (emission inventories), from all main sectors of the Irish economy. These emission inventories are required to comply with Ireland's EU and United Nations (UN) reporting obligations, and also serve to inform national policy initiatives. Moreover, under Article 4.1 of the National Emissions Ceiling (NEC) Directive [2001/81/EC], Ireland is required to limit emissions of NO_x to not more than 65 kilotonnes per year by 2010.

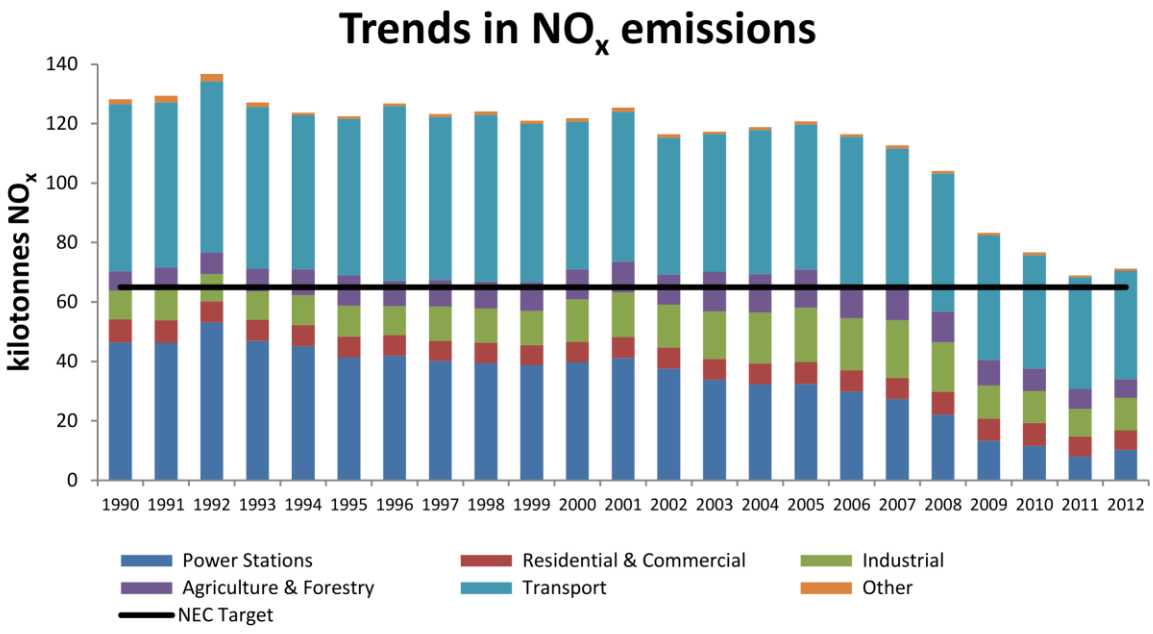

Figure 1 Estimated total NO_x emission for Ireland, by sector, 1990-2011. (EPA 2014)

The Irish EPA currently uses COPERT (Computer Programme to calculate Emissions from Road Transport) software to estimate the contribution of road transport to emissions in Ireland. Based on the COPERT estimates, the Transport sector (road and non-road) is responsible for approximately 50% of Irish NO_x emissions, as can be seen in Figure **1**. Although the absolute level of NO_x emissions from Transport has declined by about one third since 1990, its proportional contribution has steadily increased as emissions from Power Stations have declined.

Figure **2** presents similar data for emissions of $PM_{2.5}$ (particulate matter with an aerodynamic diameter less than or equal to 2.5 µm). The share of this pollutant attributable to transport (again estimated using COPERT 4) is somewhat smaller than for NO_x, but still accounts for approximately 30% of national emissions.

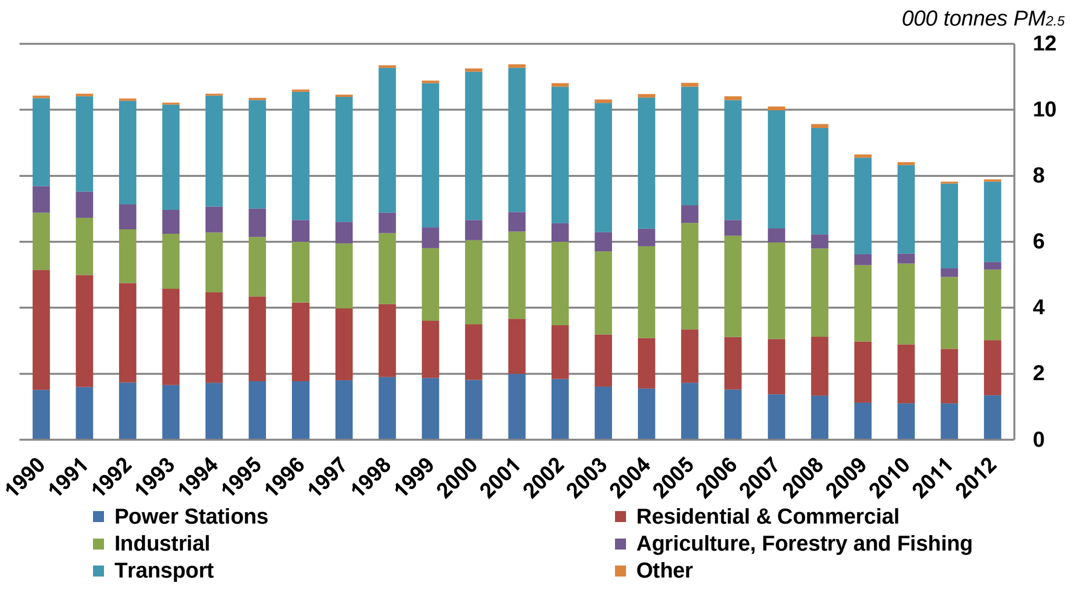

Figure 2 Estimated total $PM_{2.5}$ emissions for Ireland, by sector, 1990-2011. (Central Statistics Office, 2014)

In summary, based on these COPERT estimates, Transport is the dominant source of both $PM_{2.5}$ and NO_x emissions in Ireland.

2.2 Combustion in petrol and diesel engines

The primary source of vehicle emissions is the internal combustion (IC) engine used to propel the vehicle. The basic concept of the IC engine is fairly simple: a mixture of fuel and air is compressed, and then burned, within a closed, cylindrical volume. A movable piston forms a gas-tight “floor” to this cylinder and, when exposed to the high gas pressure resulting from combustion of the fuel-air mixture, is forced downward. The linear motion of the piston is converted to rotary motion by a crankshaft, and its pulsatile nature damped with a flywheel. Rotation of the flywheel is used to propel the vehicle forward, via connection to a gearbox and thence to the vehicle wheels.

Two main variations of the IC engine have been developed: the Spark Ignition (SI) or petrol engine, and the Compression Ignition (CI) or diesel engine, and the two designs differ significantly in combustion principle. In an SI engine, ignition of a relatively homogenous fuel-air mixture is actively initiated by means of an appropriately-timed spark. Combustion occurs as a flame front expands outward from the ignition source in a relatively uniform manner. This contrasts strongly with the CI engine, where fuel is injected into hot, highly-compressed air, and ignition occurs spontaneously as the fuel and air mix. In CI combustion, ignition sites are scattered throughout the combustion chamber, and combustion proceeds in a highly heterogeneous environment encompassing large variations in local temperature and composition.

An important consequence of the above is that petrol and diesel engines differ significantly in the quantity and composition of the emissions they produce, and in the technology required to mitigate those emissions. The relatively homogenous mixture used in SI engines enables, and often requires, operation close to

the ideal, stoichiometric, air:fuel ratio (AFR), at which precisely the required quantity of oxygen is provided for complete combustion of all available fuel. Direct-injection SI engines (DISI), also known as gasoline direct-injection (GDI), can operate with significant quantities of excess air at some part-load conditions, by careful stratification of the fuel-air mixture. This helps increase part-load engine efficiency, by reducing pumping losses. Diesel engines must *always* provide substantially more oxygen (i.e., air) than required by stoichiometry, because the fuel is not fully mixed with air prior to combustion. Therefore, diesel exhaust always contains, and DISI exhaust may contain, a significant amount of oxygen, making the tailpipe reduction of NO_x emissions much more difficult.

2.3 NO_x formation and control in petrol and diesel engines

NO_x refers to all binary compounds of nitrogen and oxygen, *i.e.,* all oxides of nitrogen. Modern vehicle engines produce mostly nitrogen monoxide (NO), and some fraction of this compound is then further oxidised to nitrogen dioxide (NO_2), both in the exhaust system and in the atmosphere.

Formation of NO_x from nitrogen – an inert gas – requires very high temperatures (~ 2000+ K) and a supply of oxygen. Hence, NO_x formation can be curtailed by reducing combustion temperatures, and/or by restricting the availability of oxygen. However, the former generally leads to reductions in engine efficiency and power output, whilst the latter can favour increased production of PM. These disadvantages are avoided if an effective after-treatment technology can be incorporated in the vehicle tailpipe.

For SI engines operating close to stoichiometry, the three-way catalyst (TWC) provides this functionality at relatively low cost, in a compact device with no moving parts. The TWC addresses all three of the principal air quality pollutants from SI engines – NO_x, carbon monoxide (CO), and unburned hydrocarbons (UHC) – by removing oxygen from the NO_x to form Nitrogen (N_2), and adding that oxygen to the CO and UHC to form CO_2 and water (H_2O). Consequently, once the TWC has reached its "light off" temperature, NO_x emissions from stoichiometric SI engines are very low under almost all operating conditions.

Unfortunately this solution is not viable for CI engines, as the presence of oxygen in the exhaust stream makes reduction of the NO_x much more difficult, although a simpler oxidation catalyst is very effective at eliminating CO and UHC. The problem of suppressing NO_x emission from CI engines is compounded by the fact that "engine out" NO_x emissions are higher than for SI engines, a consequence of the CI engine's highly heterogeneous combustion environment, in which localised regions conducive to NO_x formation are far more likely to occur (Heywood, 1988).

Up to and including Euro [2] 5 for light-duty vehicles (LDVs) (vehicles registered before September 2015), and Euro IV for heavy-duty vehicles (HDVs) (vehicles registered before October 2009), mitigation of NO_x emission from CI engines has therefore been based on control of in-cylinder conditions – via ever-increasing injection pressures, use of multiple injection events per cycle, and incorporation of exhaust gas recirculation (EGR). Whilst these measures are effective *under the closely defined*

[2] Euro 1-6, and Euro I-VI, refer to legislative emission limit values for test cycles, for LDVs and HDVs respectively. A detailed discussion of these cycles is presented in Chapter 3; tables of emission limit values and application dates are presented in Appendix 1.

conditions of the legislative test cycle, their ability to control NO_x emissions in real-world operation is much lower. The tighter emission limits and more comprehensive test cycle associated with Euro 6 / VI legislation (see Chapter 3) impose a *de facto* requirement for after-treatment in the tailpipe.

For HDVs, that after-treatment takes the form of Selective Catalytic Reduction (SCR). With SCR, carefully metered quantities of a reducing agent[3], such as ammonia (NH_3) or aqueous urea (AdBlue), are injected into the exhaust stream. The mixture is then passed over an appropriate catalyst where, if conditions are favourable, the NO_x and reducing agent react to form N_2 and H_2O. The vehicle must therefore carry a supply of reductant on board, and the supply must be periodically replenished.

The limited on-board supply, coupled with the fact that reductants employed in SCR may be environmentally objectionable, means that injection of excess reductant is undesirable. Conversely, under-supply will limit the NO_x conversion efficiency that can be obtained. Attaining high conversion efficiency also requires uniform distribution of the reductant in the exhaust stream, and a mixture temperature of 200°C - 430°C when passing over the catalyst. Maintaining these conditions during dynamic engine operation demands a sophisticated, carefully calibrated, and fast-acting control system. SCR is therefore a complex and expensive addition to an emission control system. For that reason, SCR has found its main application in Euro IV, V, and VI HDVs.

For light-duty vehicles (LDVs), such as passenger cars and light vans, lean NO_x traps (LNTs) may offer a simpler and more cost-effective solution. LNTs can remove NO_x from a lean (high oxygen content) exhaust stream by first oxidising all NO_x to NO_2 over a platinum catalyst The NO_2 is then adsorbed onto the catalyst substrate to form a solid nitrate phase. When the substrate's NO_2 adsorption capacity has been exhausted, the control system commands an oxygen-poor exhaust stream to be generated, either by increasing the rate of exhaust-gas recirculation (EGR) or by injection of fuel into the exhaust stream. The presence of an oxygen-poor exhaust stream destabilises the compounds formed by the adsorbed NO_2, causing its release as NO_x or NH_3. The NO_x reacts over a rhodium catalyst with the CO, H_2 and other compounds in the (now) oxygen-poor exhaust, forming N_2, CO_2, and H_2O. An NH_3-absorbing, "dry", SCR catalyst may be positioned downstream of the LNT, so that some of the NH_3 generated during desorption can subsequently be used for NO_x reduction. Once the adsorbed NO_2 has been released from the substrate, enrichment of the exhaust stream is halted, and the LNT starts to adsorb NO_x once again.

Relative to SCR, the LNT has the advantages of reduced complexity, simpler control, lower cost, and elimination of the need for a secondary reductant. On the other hand, there is a fuel consumption (and CO_2) penalty associated with the periodic enrichment of the exhaust required for LNT regeneration, although at least part of this penalty can be offset by tuning the engine for higher efficiency, since engine-out NO_x emissions are not a major concern. LNT may also struggle to match the NO_x conversion efficiency achieved with SCR under steady-state conditions.

[3] A reducing agent (reductant), in this context, is a compound that removes oxygen from the NO_x, converting it to N_2. The reductant, in turn, is oxidised.

Therefore, whilst SCR may continue to represent the solution of choice for HDVs, LNT – with or without "dry" SCR – is likely to capture a significant fraction of the diesel LDV market for Euro 6 and beyond. Currently, LNTs are routinely used to control NO_x in DISI engines, which are achieving significant penetration in the SI market and are likely to dominate within a few years. The DISI technology is not directly transferrable to CI engines, because the exhaust conditions are quite different, but the technical knowledge gained is a significant help in developing LNT for CI engines.

2.4 PM formation and control in petrol and diesel engines

Particulate matter is primarily produced by incomplete combustion of hydrocarbon fuels, although lubricants (such as engine oil) also contribute. Particle nucleation occurs in fuel-rich (oxygen-poor) regions of the combustion zone. Such regions rarely occur in SI engines, but are extremely prevalent in the heterogeneous combustion environment of CI engines. PM emissions are therefore associated almost exclusively with diesel engines.

The formation of PM is a complex multi-stage process. Initially, gas-phase molecules form very small (< 2nm) soot nuclei, which grow by accretion and agglomeration to form bigger particles. Very large, complex chains of particulates are formed during this process. Further chemical and physical changes occur continuously as the nascent particle passes through regions of varying temperature and composition, within the combustion chamber and the exhaust pipe. As the exhaust gases cool, the particles act as condensation nuclei for water and hydrocarbon vapours, which adsorb onto their outer surface. The precise concentration, size distribution, and composition of PM emissions therefore depends on the number of nucleation particles formed in the combustion zone, and on the details of the trajectory followed by those particles in time-temperature-pressure-composition space.

Initially, compliance with PM emission limits was achieved by control of in-cylinder conditions. The enhanced control was achieved using common-rail injection, increased injection pressure, multiple injection events and improved air flow. As legislative limits tightened, the incorporation of diesel oxidation catalysts (DOCs) in the exhaust became necessary. DOCs reduce particle mass mainly by oxidation of hydrocarbons adsorbed onto the particle surface, but have the added benefit of reducing emissions of CO and UHC.

To comply with Euro 5 and later, however, the use of diesel particulate filters (DPFs) became a *de facto* requirement for diesel LDVs. The DPF separates particles from the gas stream by deposition on a collecting surface. It consists of a honeycomb-like ceramic structure (typically cordierite or silicon carbide) with alternate passages blocked. The exhaust gas is thus forced to flow through the porous walls, which act as a filter medium. DPFs have proven to be extremely effective at removing particulate, with filtration efficiencies up to 100% being observed.

The main challenge in DPF applications is efficient and timely removal of accumulated PM, in order to prevent excessive backpressure rise and consequent engine performance degradation. Removal is achieved through periodic oxidation of the PM, in a process called regeneration. Soot oxidation normally requires temperatures of 600^{o}C or more, well above those typical of diesel engine operation, but catalytic coating of the DPF, and/or installation of an upstream DOC, can enable regeneration

to be achieved at somewhat lower temperatures.

If these temperatures are achieved in routine engine operation, so-called passive regeneration of the DPF occurs spontaneously. However, passive regeneration cannot be relied on for all driving conditions, and so an active regeneration system must also be provided. This functions by continuously monitoring the pressure drop across the DPF, and commanding regeneration if necessary. In that case, exhaust temperature is temporarily increased either by delayed fuel injection in the cylinder, or by injection of fuel directly into the exhaust manifold.

Most HDVs have managed to avoid using a DPF whilst meeting legislative PM limits up to Euro V. It is widely expected, however, that Euro VI compliance will require incorporation of a DPF in the tailpipe.

2.5 The influence of drive cycle on NO_x and PM emissions

Consideration of the preceding discussion reveals that, for a given engine design, the level of raw, engine-out, emissions depends (primarily) on engine speed and load. In a broad sense – or in an ideal world, perhaps – engine speed and load both increase with vehicle speed, and so engine-out emissions might be expected to do likewise. In practice, they depend on <u>driver behaviour</u> – *i.e.,* on the commands the driver sends to the Engine Management System (EMS) – and on the <u>EMS interpretation</u> of those commands. As engine technology level increases, “drive-by-wire” has steadily increased the physical disconnect between driver input and engine response. EMS outputs are based not only on throttle pedal position, but also on its rate of change, and on numerous exogenous variables including ambient temperature and pressure, engine coolant and oil temperature, catalyst temperature, recent driving style, EMS mode, etc. It is therefore not possible to predict, *a priori*, the exhaust temperature and composition associated with a particular engine speed and load.

From a legislative and environmental perspective, however, it is the vehicle tailpipe emissions, not the engine-out emissions that are important. Tailpipe emissions depend on a combination of engine-out emissions, tailpipe technology level, and the <u>effectiveness</u> with which that technology functions. That effectiveness depends on three principal factors:

- The details of the installation – location, size, supplier, etc.
- The instantaneous exhaust gas composition and temperature at entry to each device in the tailpipe.
- The *transient* response characteristics of the *combined tailpipe treatment system.*

The first of these is vehicle-specific, and is the reason that vehicle manufacturers devote substantial internal resources to system integration, whilst also working closely with technology suppliers. The fact that a particular technology performs well in one vehicle type, is not a guarantee of similar performance in another.

The temperature and composition of the exhaust gas at entry to a particular tailpipe device depends mainly on the gas properties at exit from the device upstream. Since most tailpipe devices will be effective within a specific range of temperature and composition, and given that both can change significantly across

each device, maintaining appropriate conditions at entry to each device can present a major challenge. The problem is compounded if periodic exhaust enrichment is used for regeneration of an LNT or DPF.

The transient response characteristics of the combined tailpipe treatment system are vehicle-specific, but their *significance* depends on the behaviour of the coupled driver-EMS system. During steady-state operation, the significance tends towards zero, but may be a dominant determinant of emissions during unsteady operation.

In summary, the authors can state that mitigating tailpipe emissions are easiest when:

1. the vehicle is operating in steady state
2. that state is pre-defined
3. the number of individual operating states at which measurements are made is small
4. the operating states cover a small portion of the engine load-speed range
5. external (i.e., ambient) conditions are pre-defined.

If emissions are to be measured during unsteady operation of the vehicle, mitigation will be easiest if changes in engine speed and load occur slowly and are small in amplitude.

With this information in mind, the legislative test cycles used for LDVs and HDVs will now be reviewed.

3 Legislative test cycles and emissions

Legislative test cycles for road vehicles are divided into two categories: those for HDVs – trucks, buses, and coaches; and those for LDVs – passenger cars and light vans. The test procedures used, and emission limit values, vary by geo-political region; only EU test cycles are considered here. The following sections present a high-level discussion of the test procedures and associated emission limit values prescribed for Euro I-VI (HDV) and Euro 1-6 (LDV). Tables detailing the precise emission limits corresponding to a specific Euro certification levels are included in Appendix 1.

For both LDVs and HDVs, emission measurement occurs while the engine or vehicle is stationary in an environmentally-controlled laboratory. The engine (or vehicle) being tested is pre-conditioned prior to the test, and then "driven" at a variety of specified conditions. The engine exhaust produced at these conditions is diluted with ambient air, to simulate atmospheric processes at tailpipe exit and to prevent water condensation. Pollutant concentration is measured using a sample of the diluted exhaust, and the engine-out emissions are back-calculated from these measurements.

3.1 Test cycles: Heavy Duty Vehicles (HDVs)

Testing of HDVs differs from that of LDVs, in that the engine is tested independently of the vehicle. The logic driving this is that a given heavy-duty engine may see service across a wide range of vehicles, with substantial differences in vehicle weight and other characteristics. Therefore, emissions are expressed (and legislative limits defined) in terms of grammes of emission per kiloWatt hour of work done by the engine ($g.kWh^{-1}$), rather than g per km ($g.km^{-1}$) travelled by the vehicle.

3.1.1 Euro I and II

Euro I and Euro II engines were tested using the ECE R49 test cycle, a 13-mode, steady-state test introduced by ECE Regulation No. 49 and then adopted by the European Economic Commission (EEC, 1988) [EEC Directive 88/77]. The R49 test comprises a sequence of 13 speed and load conditions, identical to those of the US 13-mode cycle. Exhaust emissions are measured at each mode, and the final test result calculated from a weighted average of these values. The test conditions and weighting factors of the R49 cycle are shown in Figure 3. The area of each circle in the figure is proportional to the weighting factor for that mode.

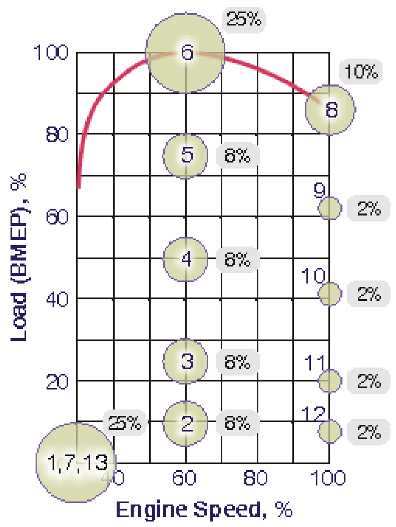

Figure 3 R49 test matrix. (Image source: transportpolicy.net)

3.1.2 Euro III, IV and V

For certification of engines to Euro standard III, IV, and V, three separate tests were employed: a European Stationary Cycle (ESC), a European Transient Cycle (ETC), and an Engine Load Response (ELR), as specified in Directive 1999/96/EC. For compliance, engines must meet the gaseous, PM, and smoke opacity limits specified for each test.

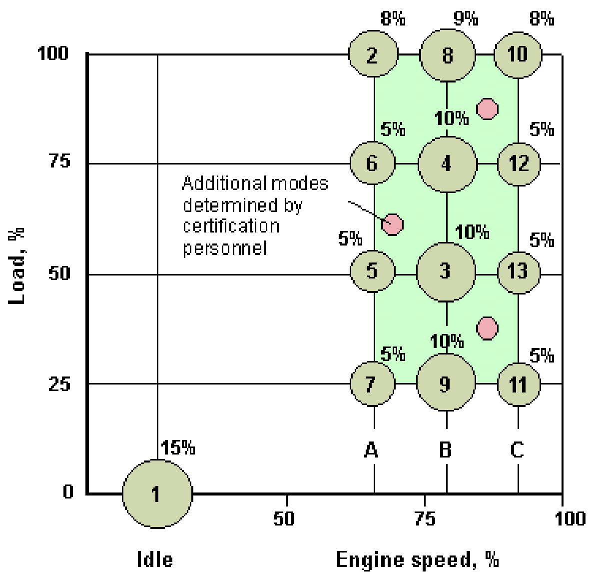

Figure 4 European Stationary Cycle (ESC). (Image source: transportpolicy.net)

The ESC (see Figure 4), also a 13-mode cycle, replaced the R49 test. The weighting applied to idle conditions is reduced, and the modes are more uniformly distributed in the main working range of heavy-duty (HD) engines.

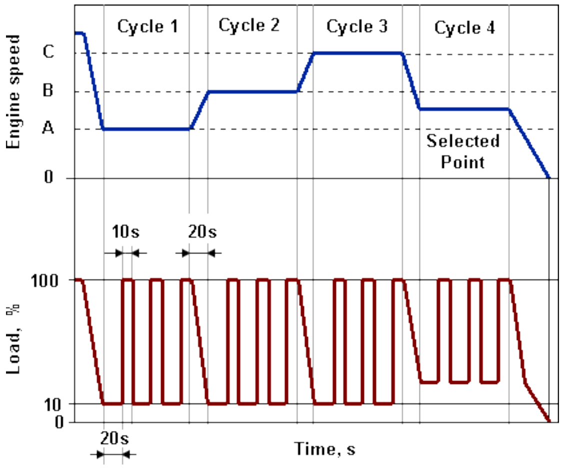

Figure 5 Engine Load Response (ELR) test. (Image source: transportpolicy.net)

The ELR test (see Figure 5) consists of a sequence of three load steps at each of three specified engine speeds, followed by a fourth cycle at a speed and a load selected by the certification personnel. It was used only for measurement of smoke opacity.

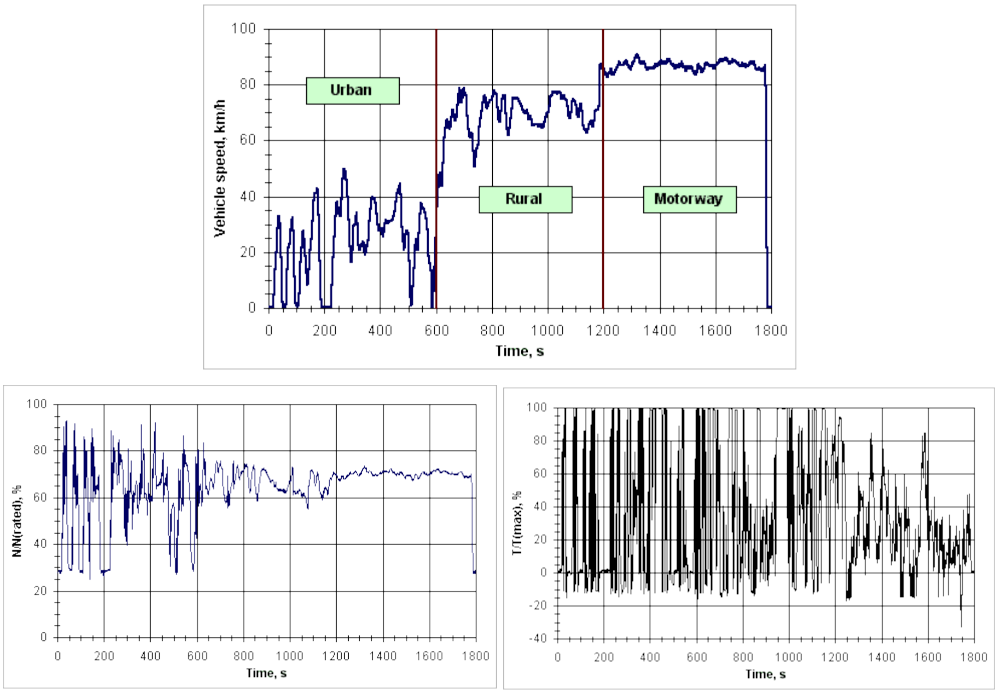

Figure 6 European Transient Cycle (ETC). (Image source: transportpolicy.net)

The ETC (see Figure 6) represented a far more significant change to the test procedure. Based on real road cycle measurements of heavy duty vehicles, the ETC comprises three separate phases of 600 second duration, representing urban, rural, and motorway driving respectively.

Unfortunately, neither the ESC nor the ETC is fully representative of the range of engine

conditions seen during in-use driving. Both of these cycles have relatively high average engine load over the entire test; consequently, average exhaust temperature during the test is relatively high. In addition, the test procedures allow the engine manufacturer to define pre-test engine conditioning. Virtually all manufacturers start the test with the engine fully warmed up and exhaust temperature above 300°C. Under these test conditions, an SCR-equipped engine can meet the ETC test limits of Euro IV and Euro V, even if the system has poor NO_x conversion efficiency at low exhaust temperature.

This approach to emission control, known as “cycle beating”, is widely used by manufacturers who, operating in a highly-competitive and cost-sensitive environment, strive to minimise the cost and complexity of meeting legislated emissions requirements.

3.1.3 Euro VI

HDV engines certified since 2013 must comply with Euro VI legislation, as defined in EU Regulation No. 595 / 2009. As well as reducing emission limit values, Euro VI specifies the use of two new test cycles, the Worldwide Harmonised Steady Cycle (WHSC) (see Figure 7) and its transient counterpart the Worldwide Harmonised Transient Cycle (WHTC) (see Figure 8).

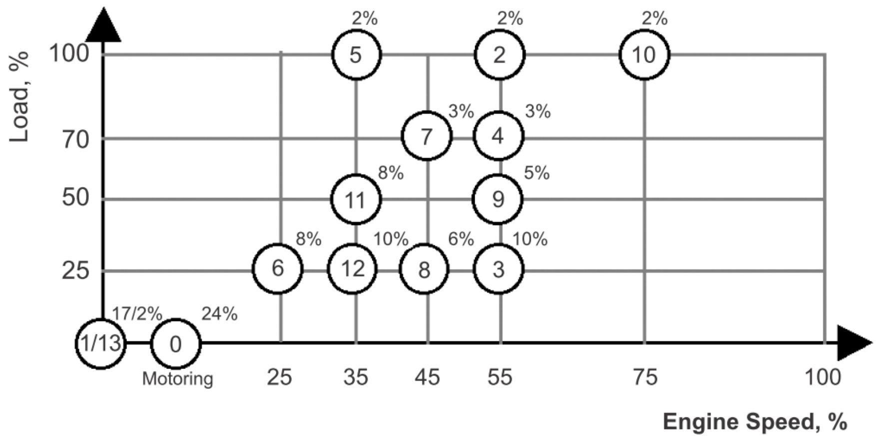

Figure 7 Worldwide Harmonised Steady Cycle (WHSC). (Image source: transportpolicy.net)

These test cycles, developed under the United Nations Economic Commission for Europe (UNECE), are based on research into the world-wide pattern of real, heavy commercial-vehicle use. From the collected data, the WHTC with both cold and hot start requirements, and a hot start WHSC, have been created. These cycles cover driving conditions typical of the EU, the United States, Japan and Australia.

Development and use of a worldwide harmonised test procedure should simplify the certification process for engine manufacturers. At the time of writing (2013), emission measurement procedures and limits vary regionally, but the ultimate objective being pursued by the UNECE Working Party on Pollution and Energy (GRPE) is global harmonisation of all aspects of the certification process.

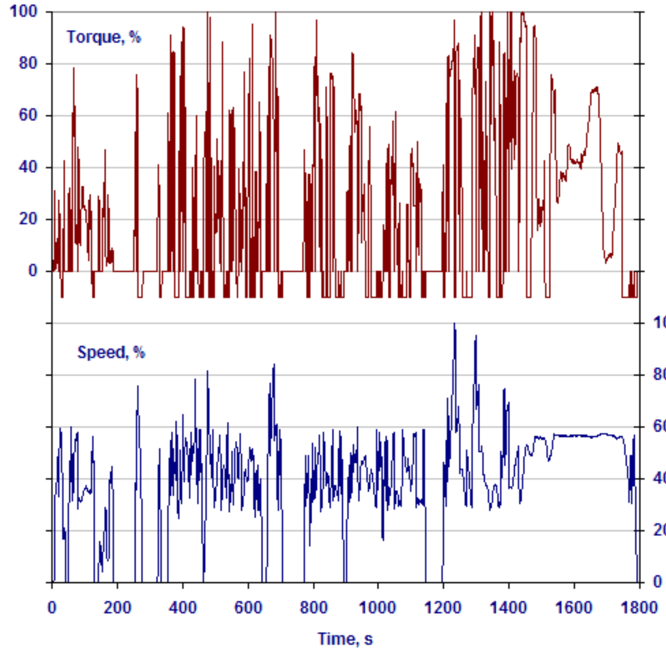

Figure 8 Worldwide Harmonised Transient Cycle (WHTC). (Image source: transportpolicy.net)

Notwithstanding the fact that the WHSC and WHTC have been developed specifically to reflect real-world engine operation, the Euro VI implementing legislation also obliges manufacturers to take steps to limit “off-cycle” emissions. Emissions are required to be “effectively limited, throughout the normal life of the vehicles under normal conditions of use” including “under the range of operating conditions that may be encountered.” To demonstrate compliance, manufacturers must undertake an in-use testing programme, using portable emissions measurement systems (PEMS) for each certified engine family. Testing must be repeated every two years over the normal life of an engine, and average emissions measured must be no more than 1.5 times the WHTC test limit.

3.1.4 Summary of EU HDV test cycle development

It is clear from the discussion above that the EU test cycles for HDVs have progressed from a relatively simple, 13-mode, steady cycle, to a combination of sophisticated steady-state and transient cycles, carefully tuned to reflect real-world driving conditions, and with a requirement to demonstrate continued compliance – under real-world conditions – throughout the life of the engine. The WHTC, in particular, covers a wide range of engine loads and speeds, encompasses rapid transients and engine over-run conditions, and incorporates a cold start requirement. This last is very significant for assessing the efficiency of SCR after-treatment at the low exhaust gas temperatures typical of urban driving.

3.2 Emission limits: Heavy Duty Vehicles (HDVs)

A detailed list of the emission limit values prescribed for each Euro certification level is included in Appendix 1. An overview of the changes in NO_x and PM limits is presented in Figure 9, for certification standards Euro I to Euro VI. Limits are shown only for the steady-cycle test, since transient-cycle testing was not introduced until Euro III.

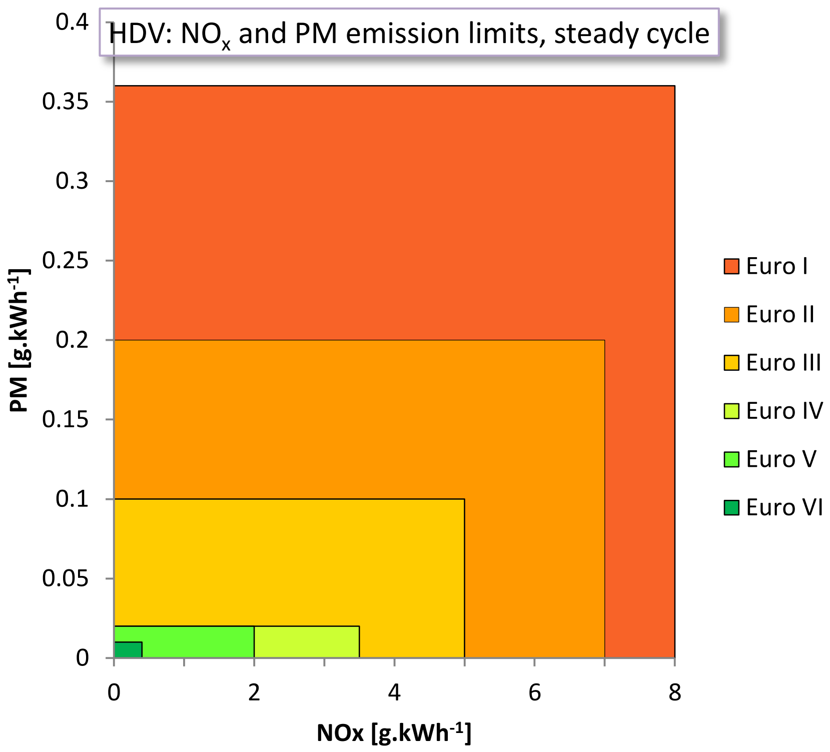

Figure 9 Overview of HDV emission limit reductions, from Euro I (1992) to Euro VI (2013).

Nonetheless, it is clear that the reductions imposed on engine manufacturers have been very significant, with NO_x emission limits reducing by a factor of 20, and PM by a factor of 36, over that period. This is equivalent to an *annual* reduction in emission level of more than 13% and almost 16% respectively. The reductions achieved are even more impressive, given that combustion refinements which reduce PM tend to increase NO_x, and vice versa.

3.3 Test cycles: Light Duty Vehicles (LDVs)

In contrast to HDVs, passenger cars and light commercial vehicles (LDVs) are tested for certification on a chassis dynamometer – *i.e.,* the complete vehicle is tested. Where an engine is used in more than one vehicle configuration, each configuration must be tested and individually certified.

3.3.1 New European Driving Cycle (NEDC)

For certification standards Euro 1 – Euro 6b, all applicable vehicles have been tested using the New European Driving Cycle (NEDC) described below. Prior to the test, the vehicle is allowed to soak for at least six hours at a temperature of 20 – 30°C. The NEDC initially incorporated a forty-second "warm up" period prior to commencement of emissions measurement, but from Euro 3 (2000) onwards this warm-up period was eliminated. The resulting cycle is sometimes referred to as the Modified New European Driving Cycle (MNEDC); however, "NEDC" is typically understood to refer to the current (modified) version of the driving cycle, and that convention is used throughout this report.

The NEDC comprises four "ECE 15" segments and one extra-urban drive cycle (EUDC) segment, performed without interruption. The

ECE 15 segments are intended to represent urban driving, the EUDC to represent extra-urban driving. The cycle, shown in Figure 10, covers a total distance of 11.007 km, at an average speed 33.6 $km.h^{-1}$, and with a maximum speed of 120 $km.h^{-1}$. Each ECE 15 sub-cycle is 1.013 km long, with an average speed is 18.7 $km.h^{-1}$, and a maximum speed of 50 $km.h^{-1}$. The EUDC – intended to simulate extra-urban and motorway driving – is 6.955 km long, with an average speed of 62.6 $km.h^{-1}$, and a maximum of 120 $km.h^{-1}$.

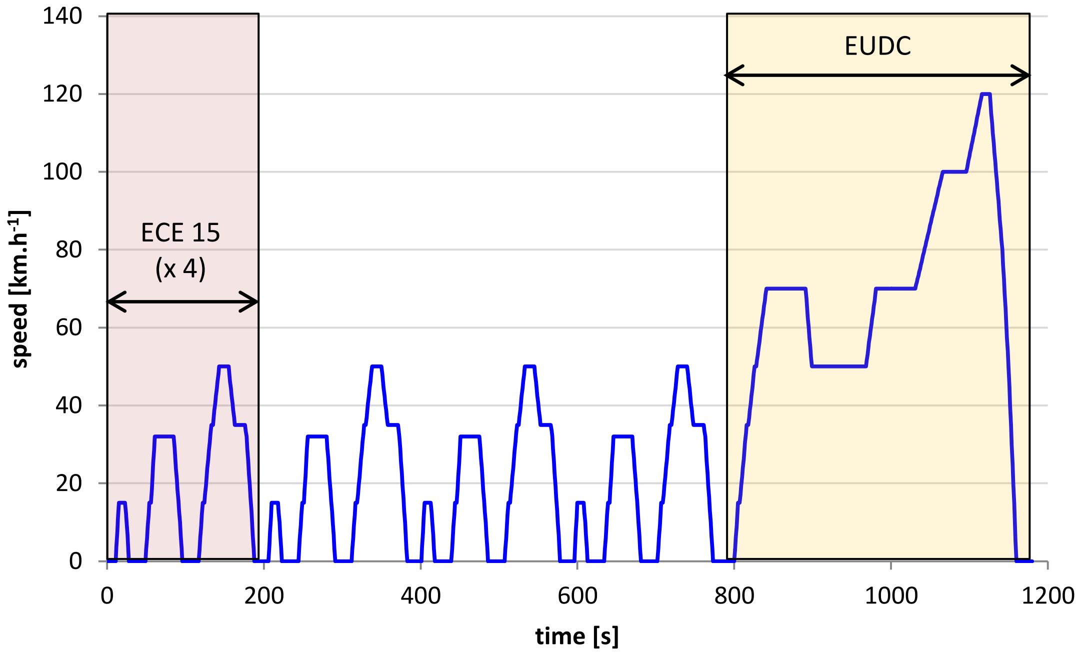

Figure 10 Speed versus time trace for the New European Driving Cycle (NEDC).

3.3.2 Real-world and NEDC drive cycles compared

Although the NEDC incorporates elements of both urban and extra-urban driving, two key weaknesses undermine its usefulness as an approximation to real-world driving. The first relates to the "cold start" temperature employed for the test. As stated above, vehicles are conditioned for at least six hours prior to the test at a temperature of 20-30°C. Since the majority of tailpipe emissions from spark-ignition engines occur prior to catalyst light-off, emissions measured on the NEDC are quite sensitive to the cold-start temperature employed. Whereas 20-30°C might be representative of typical "cold start" conditions for some southern European countries during summer, it is by no means representative for the EU as a whole, and certainly not for Ireland. Real-world tailpipe emissions from Irish vehicles with spark-ignition engines are therefore likely to be significantly higher than those measured on the NEDC, with the divergence being greatest for short, urban trips during winter months.

The second key weakness relates to the kinematic characteristics of the drive cycle itself, and can most clearly be seen when comparing the speed-time trace of the NEDC with its real-world counterpart. As part of the ARTEMIS programme, data on real-world driving cycles was collected across a wide range of European locations and traffic conditions, from which

driving cycles representative of urban, rural, and motorway driving were distilled (Andre, 2004). These are known as the ARTEMIS urban, ARTEMIS rural, and ARTEMIS motorway drive cycles respectively. A "Common ARTEMIS Drive Cycle" (CADC) denotes one urban cycle, followed by a rural cycle, followed by a motorway cycle. Figure 11 and Figure 12 overlay these "real-world" driving cycles on their NEDC counterparts.

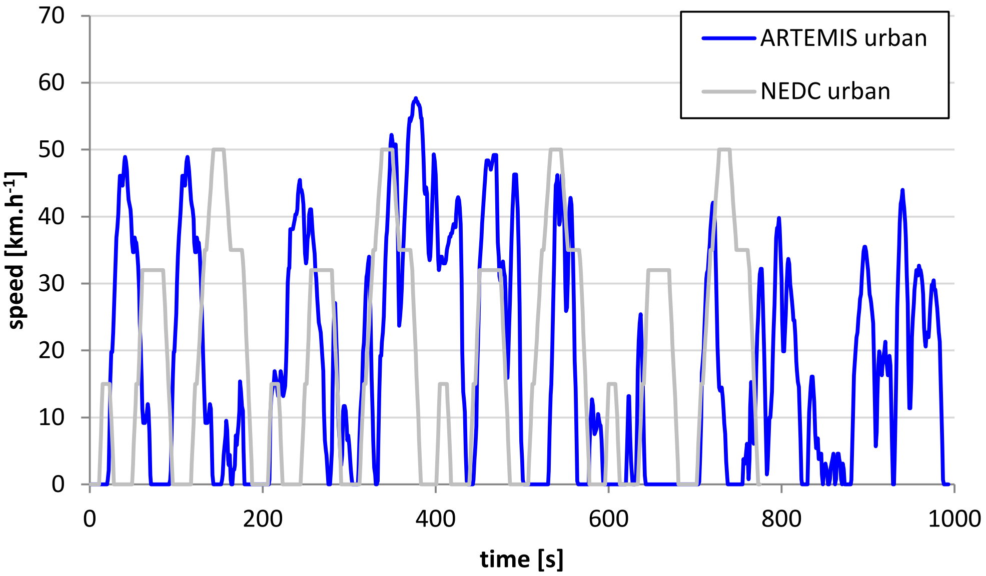

Figure 11 Comparison of "real-world" (ARTEMIS) and NEDC urban driving cycles.

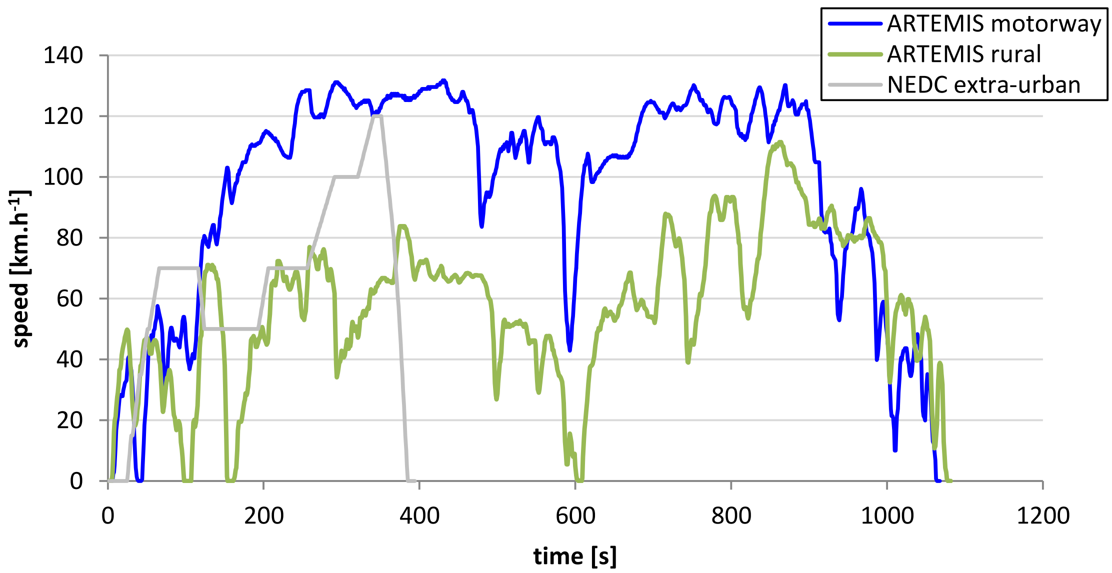

Figure 12 Comparison of "real-world" (ARTEMIS) and NEDC extra-urban driving cycles.

It is evident from these figures that, although the average and peak speeds of the real-world and NEDC sub-cycles are similar, the real-world speed trace is much more erratic than its NEDC counterpart. Less apparent from a visual inspection, but of even more significance, is the

fact that the accelerations associated with real-world cycles are substantially higher than those imposed by the NEDC.

The practical effect of these differences is that real-world driving places significantly heavier demands on the engine than are imposed by the NEDC. During the NEDC, engines operate almost exclusively at light load (small throttle openings); real-world driving explores a significantly larger proportion of the operating map. This discrepancy is highlighted in Figure 13, which plots relative positive acceleration (RPA) vs average speed for the NEDC sub-cycles (red squares), for realistic cycles developed under the ARTEMIS programme (green triangles), and for the Traffic Situations included in HBEFA 3.1 (blue circles).

RPA is discussed in more detail in Section 4.3, but is essentially a measure of the work done in accelerating the vehicle, per metre of distance travelled. Figure 13 suggests that engines work a lot harder during real-world driving than on the NEDC.

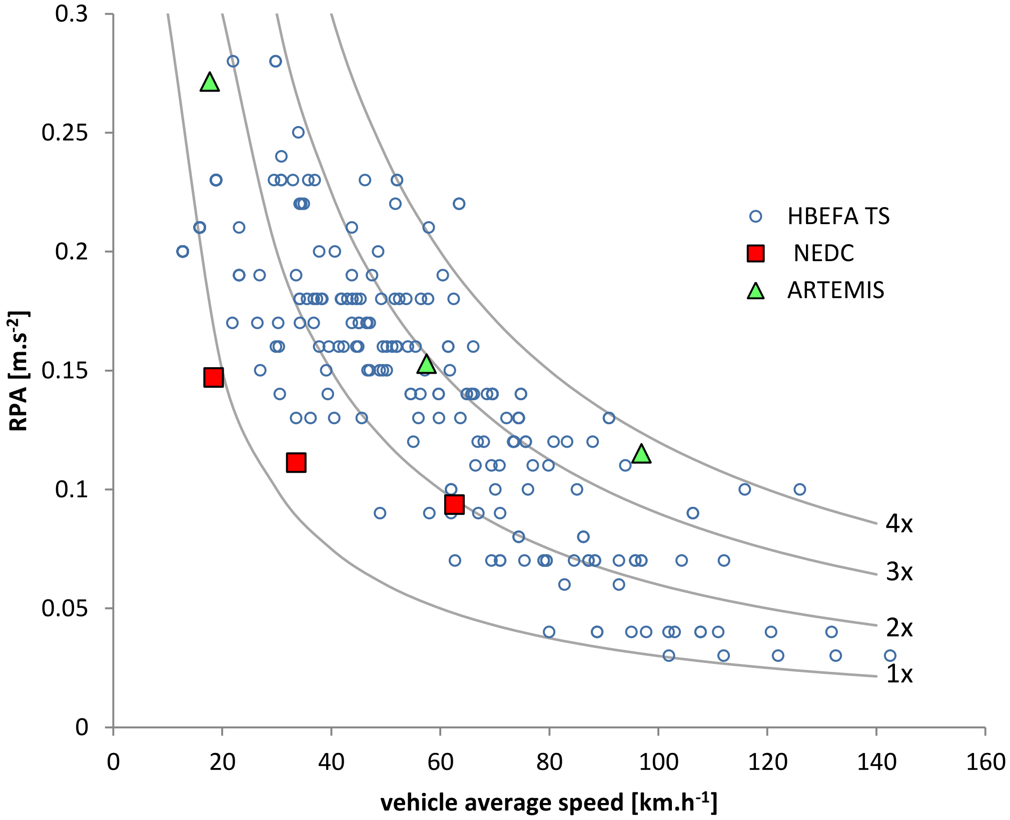

Figure 13 Relative Positive Acceleration (RPA) versus average speed for: HBEFA traffic situations (blue circles); NEDC and sub-cycles (red squares); and ARTEMIS CADC sub-cycles (green triangles). Grey lines show contours of constant engine power.

Operation at high engine load leads to increased in-cylinder temperatures, thereby favouring the formation of NO_x, as discussed in Section 2.3. In the case of diesel-powered vehicles, high engine loads imply reduced air:fuel ratio, which favours the formation of PM. Engine-out emission rate of both pollutants is therefore likely to be higher under real-world

driving conditions, than is measured on the NEDC. Moreover, as discussed in Section 2.5, since large regions of the engine operating map are ignored on the NEDC, tightening of legislative emission limits will not necessarily lead to a reduction of real-world emissions. This issue is discussed in detail in Chapter 4.

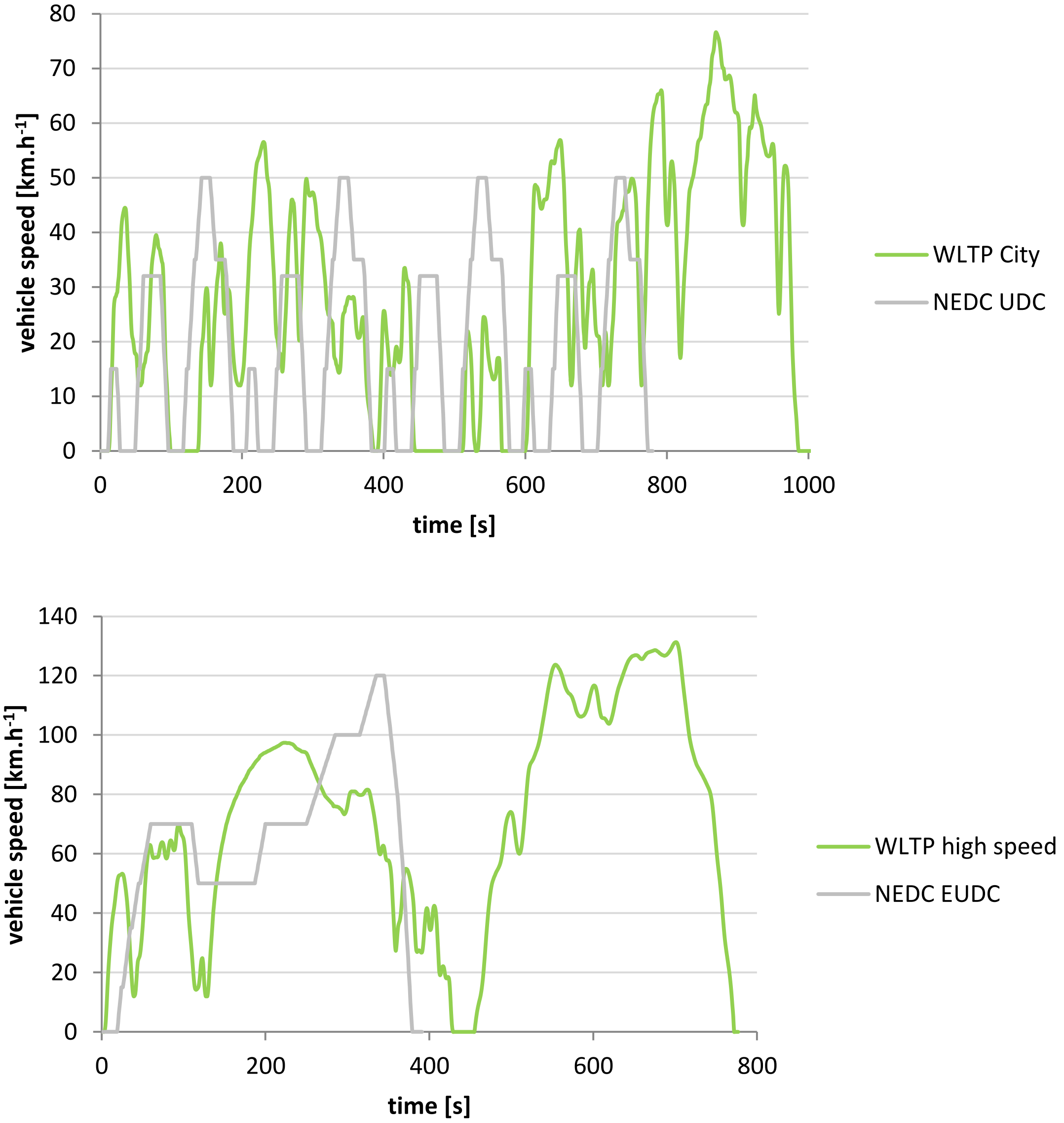

Figure 14 Urban and high-speed segments of NEDC and WLTC compared.

The simplicity and persistence of the NEDC contrasts with the continuously increasing sophistication of test cycles employed for HDV certification, and its retention may be linked to lobbying from powerful industrial groups for stability of test procedures. However, increasing evidence of the divergence between NEDC and real-world driving cycles, coupled with an interest in harmonising certification cycles worldwide, is leading to the development of a Worldwide Harmonised Light Vehicles Test Procedure (WLTP) under the auspices of the UNECE World Forum for Harmonisation of Vehicle Regulations. The WLTP incorporates a revised driving cycle, the Worldwide Light-duty Test Cycle (WLTC), which is compared with the

NEDC in Figure 14. The WLTP will be used for emission certification from Euro 6c (September 2017) onwards [Commission Regulation (EU) No 459/2012].

It is clear from Figure 14 that the WLTC sub-cycles – which are based on a comprehensive analysis of driving cycles in the EU, USA, Japan, Korea and India – include far more frequent, and sharper, fluctuations in vehicle speed than the NEDC, and are in fact similar to those developed under the ARTEMIS programme. The WLTC City sub-cycle also devotes considerably less time to idling than does the NEDC.

3.4 Emission limits: LDVs

Notwithstanding the shortcomings of the NEDC, it is the test procedure by which tailpipe emission levels have been certified for all LDVs to date. Figure 15 illustrates the progressive tightening of PM and NO_x emission limits for diesel LDVs from Euro 1 to Euro 6. The overall reductions mandated are very significant, with NO_x limits reducing by a factor of 12, and PM limits reducing by a factor of 40 over that period. These are equivalent to annual reductions of 10% and 15% respectively.

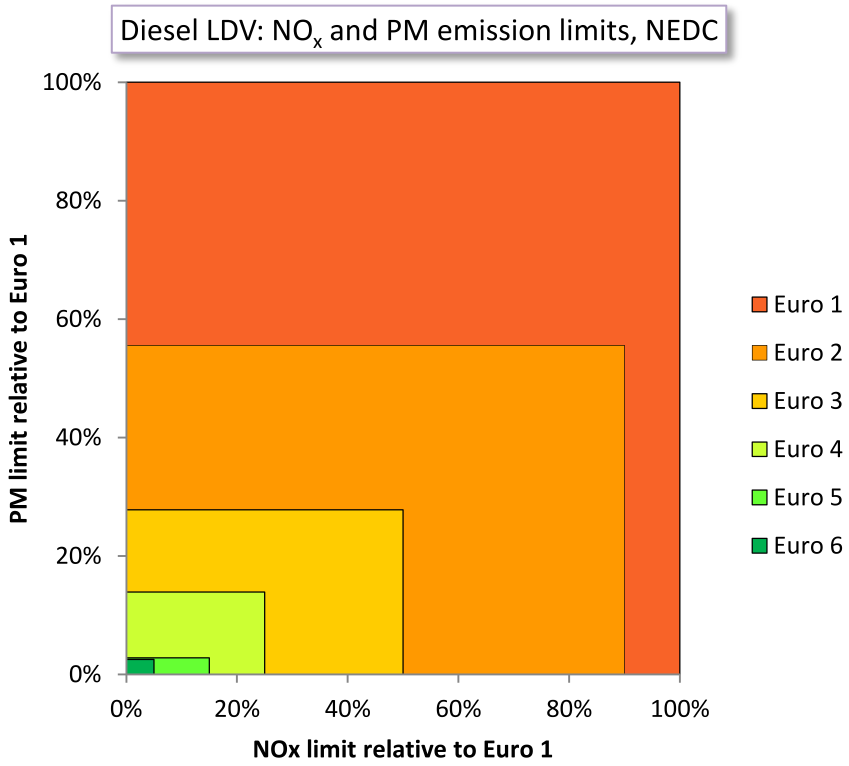

Figure 15 Overview of LDV PM and NO_x emission limits from Euro 1 (1992) to Euro 6b (2015).

Figure 16 compares the rate at which NO_x and PM emission limits were reduced for HDVs (black circles and lines) and for LDVs (light blue squares and lines). Since the emissions from HDVs and LDVs are measured on different bases, both are referenced to their respective initial limit values imposed in 1992. The pale grey diagonal in Figure 16 represents the path for which PM and NO_x emission limits decrease at equal rates.

Interesting observations that can be made with respect to Figure 16 include:

a) For both LDVs and HDVs, the initial emphasis was on decreasing PM emission limits.

b) PM limits decreased more rapidly for HDVs than for LDVs; NO_x limits more rapidly for LDVs.

c) Over the full period from 1992 – 2013, emission limits for PM and NO_x have tightened by roughly equal percentages, for both LDVs and HDVs.

d) Over the full period, the percentage reduction in emissions imposed on HDVs is approximately the same as that imposed on LDVs.

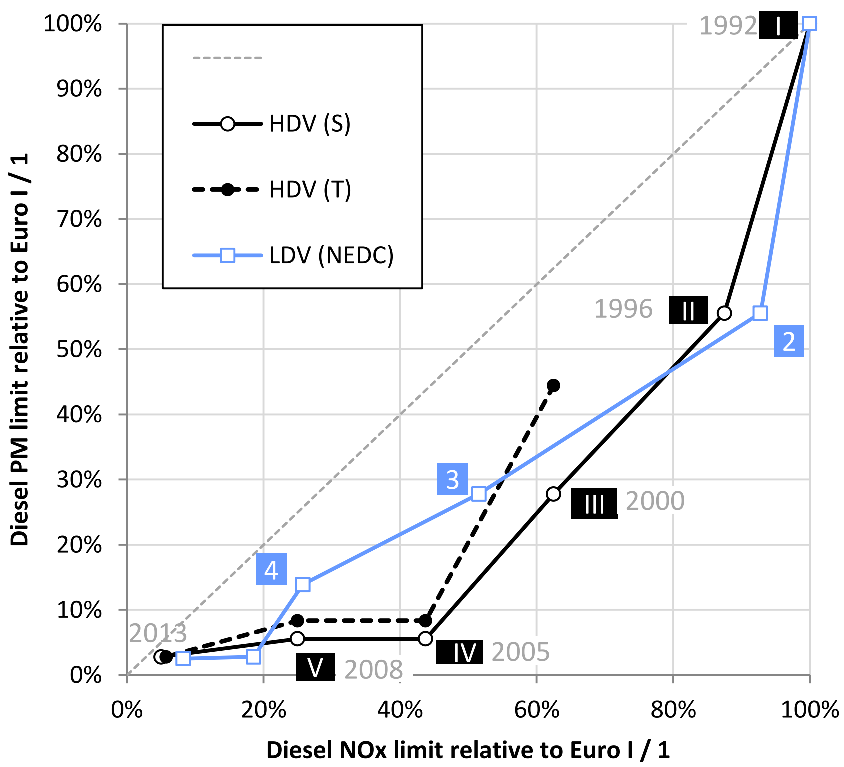

Figure 16 Comparison of relative rates of reduction, of PM and NO_x emission limit values, for HDVs and LDVs (1992 – 2013).

4 Real-world driving cycles and emissions

As discussed in Chapter 1, whereas significant effort has been devoted to establishing representative test cycles for HDVs, certification of LDV emissions is still based on the NEDC. Since the NEDC does not reflect real-world LDV operation very well, a number of alternative approaches to determining real-world emissions from LDVs have been examined. These can be broadly categorised as follows:

a) Computer prediction using measured, vehicle-specific, steady-state emissions maps and a simulated test cycle derived from real-world measurements of LDV operation - Passenger-car and Heavy-duty Emission Model (PHEM).

b) Computer prediction using laboratory measurements of vehicle-specific tailpipe emissions, as measured over a test cycle derived from real-world measurements of LDV operation (ARTEMIS).

c) Direct measurement of tailpipe emissions from a single vehicle during real-world operation (PEMS).

d) Remote sensing of tailpipe emissions from multiple vehicles, during real-world operation, at a single location (RSD).

None of these approaches is perfect. Categories *a – c* suffer from being vehicle-specific; category *d* from being specific to a particular vehicle operating condition. However, estimates obtained using approaches *c* or *d* offer the enormous benefit of being truly real-world: they provide direct measurement of the actual emissions being produced by vehicles during normal, real-world operation. As such, they provide extremely valuable calibration data for emission factors derived using less direct methods.

4.1 Vehicle emission measurement during real-world operation

Obtaining meaningful, accurate and repeatable measurements of vehicle tailpipe emissions during normal operation is very challenging, and until relatively recently required equipment whose mass and bulk precluded their use in passenger cars. Since 2007, however, portable emissions analysers of the requisite quality (such as the Semtech DS shown in Figure 17) have become available.

Figure 17 Semtech mobile vehicle exhaust gas measuring equipment. (Direct-Industry 2011)

These portable emissions measurement systems (PEMS) are capable of determining gaseous emissions at a rate of 1 Hz or more, and have proven extremely valuable for determination of real-world driving emissions (RDEs) from LDVs. It had originally been intended to use such a device within the ETASCI project – technical approval for the funding had been obtained – however, research budget constraints in the wake of the global financial crisis led to the funding being withdrawn. However, teams in other countries have successfully used PEMS to determine RDE, and some of their findings are presented in the following sections.

4.2 Real-world and legislative emission levels compared – HDVs

Wang et al. (2011) report on a field study of on-road emissions from individual diesel HDVs in Beijing, China during November and December of 2009. CO, black carbon (BC) and $PM_{0.5}$ emission factors were determined for 230 individual trucks on four major expressways around the city, and for 57 individual buses within the city, using a chase vehicle equipped with fast-response instrumentation. These emission factors – expressed as g of pollutant per kg of fuel consumed – are presented in Figure 18.

It is evident that the Euro standards have been effective at reducing real-world emission of CO and BC from these HDVs (Figure 18a and Figure 18b). However, no reduction in $PM_{0.5}$ was found. Significantly, both the peak value, and the inter-vehicle variability, of $PM_{0.5}$ is significantly higher for Euro IV HDVs than for Euro II or III (Figure 18c and Figure 18d). It is also noteworthy that the authors found that 5% of diesel trucks sampled were responsible for 50% of total BC emissions.

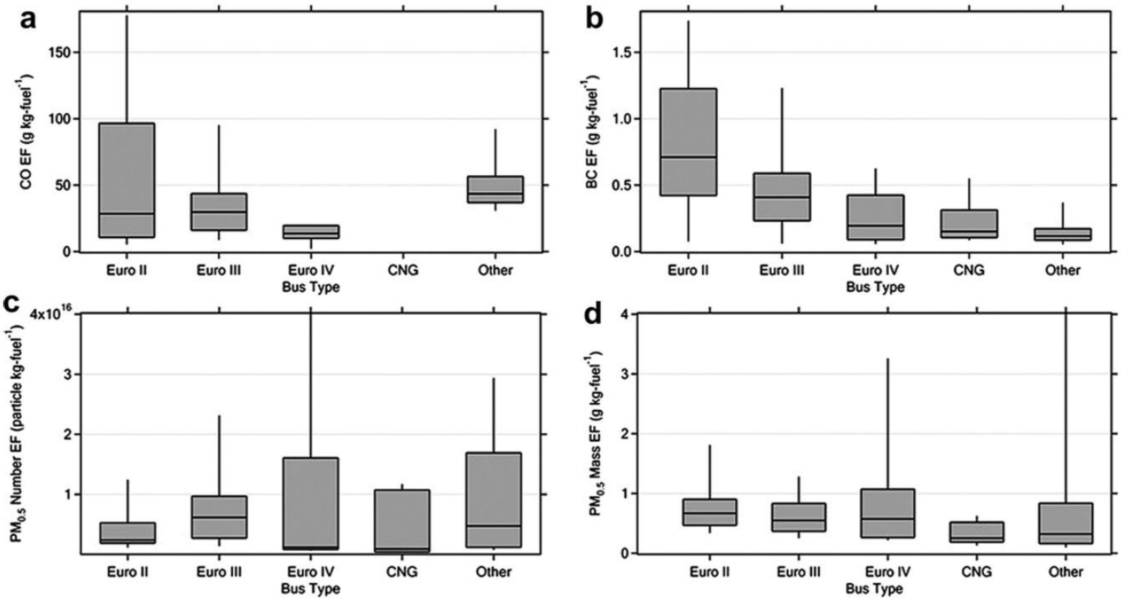

Figure 18 Real-world emission factors for 287 HDVs, as measured in Beijing (a) CO (b) Black Carbon (c) $PM_{0.5}$ Number (d) $PM_{0.5}$ Mass. [Wang et al., 2011]

Liu et al. (2011) equipped four diesel-powered urban buses (2 x Euro II and 2 x Euro IV) with PEMS and ELPI[4]. Measurement of gaseous and particulate emissions were obtained while the vehicles were driven on a standardised test cycle designed by Beijing Public Transportation Group (BPTG), and comprising both low-speed and high-speed elements. Whilst the PM emissions from all buses were found to be within limits, the measured NO_x emissions were up to double the certification limits, and exhibited considerable inter-bus variation, as shown in Figure 19.

Liu et al. (2011) attribute the observed discrepancy between certification and real-world NO_x emissions to differences in the respective driving cycles. Due to high levels of traffic congestion in Beijing, the average engine load on urban buses is low. This characteristic is reflected in the BPTG test cycle, but is not captured in the European Transient Cycle (ETC) used to certify buses to Euro II and IV. The divergence between the BPTG and ETC cycles is graphically demonstrated in Figure 20.

[4] Electrical Low Pressure Impactor: Enables real-time measurement of particle size distribution and concentration.

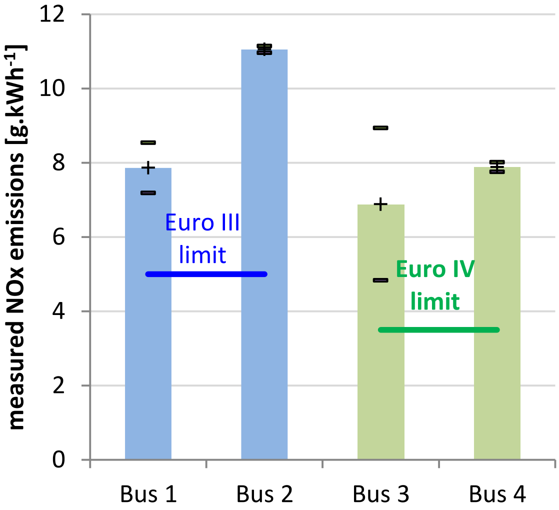

Figure 19 Measured NO_x emissions from four urban buses, compared with certification limits.

The Euro IV buses in this study are particularly sensitive to drive cycle influences, because they are equipped with SCR after-treatment for NO_x reduction. As discussed in Section 2.3, SCR requires exhaust temperatures of 200^0C or more to maximise its effectiveness. However, achieving and maintaining these temperatures at light engine loads, or during stop-start operation, is not straightforward, and SCR-equipped vehicles may struggle to control NO_x emissions during urban operation.

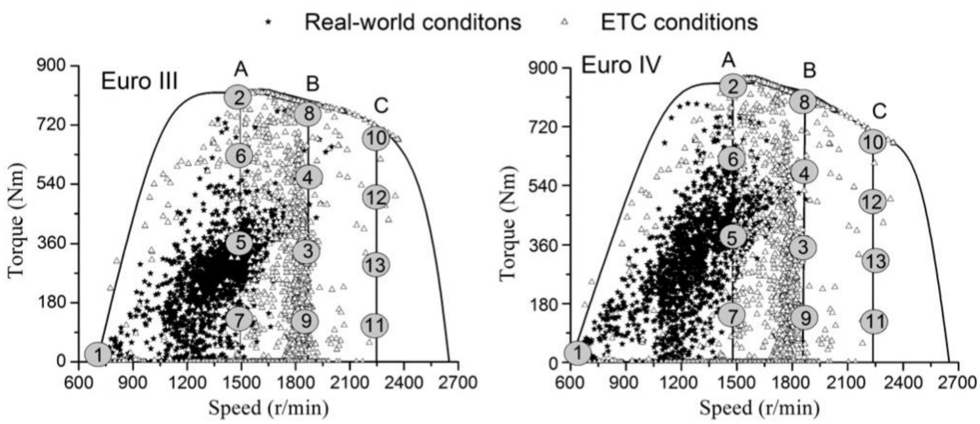

Figure 20 Engine load and speed conditions measured on the BPTG cycle (dark circles), and on the European Transient Cycle (ETC) and European Stationary Cycle (light grey circles). [Liu et al., 2011]

Further evidence of this effect is presented by the International Council on Clean Transportation (ICCT), a subset of which is presented in the following Figures. Figure 21 and Figure 22 show the NO_x emissions, measured using PEMS, for a Euro IV and a Euro V truck respectively. Both trucks are equipped with SCR. The measurements, taken during operation on German roads, show that NO_x emissions from both vehicles are considerably higher during urban driving, and in each case are substantially above the relevant certification limit.

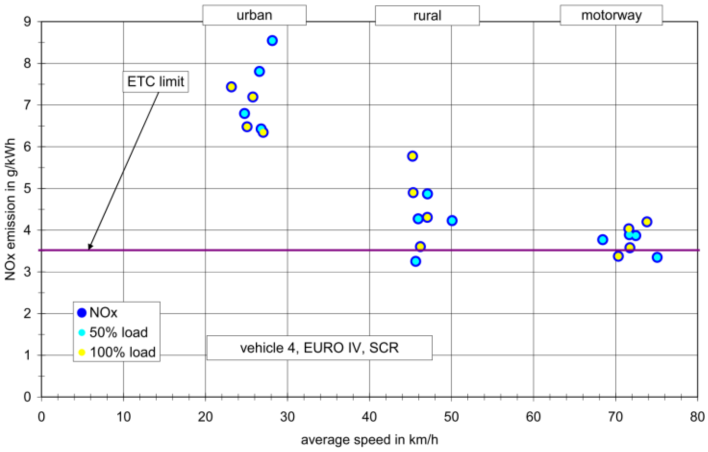

Figure 21 Impact of driving cycle on real-world NO_x emissions from Euro IV truck with SCR (Germany). [Source: White Paper 18, ICCT, 2012]

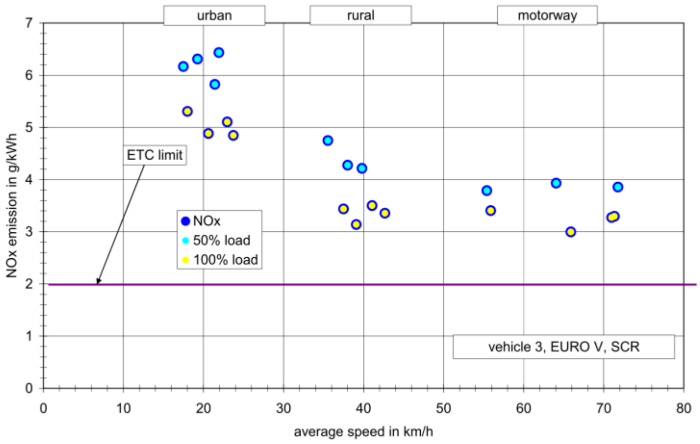

Figure 22 Impact of driving cycle on real-world NO_x emissions from Euro IV truck with SCR (Germany). [Source: White Paper 18, ICCT, 2012]

Figure 23 presents similar PEMS data obtained from 22 China IV (equivalent to Euro IV) buses in Beijing. The buses operate at low average speeds due to congestion in the city. Only three buses in Figure 23 show emissions levels below the limit – although emissions from buses 21 and 22 are very low; most of the buses monitored show emissions levels between two and four times the certification limit.

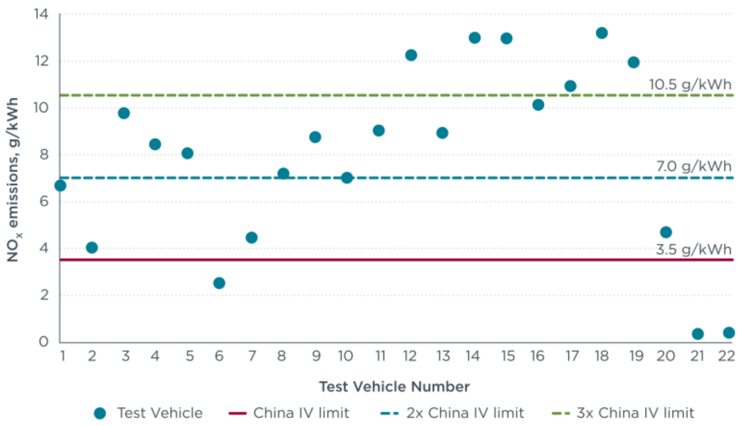

Figure 23 PEMS measurement of real-world NO_x emissions from 22 Euro IV buses in Beijing. {Source: White Paper 18, ICCT, 2012]

Air quality is Beijing is very poor by global standards. Whilst much of the problem relates to concentrations of $PM_{2.5}$, smog – to which NO_x is a significant contributing factor – has been sufficiently bad to force closure of major highways and cause disruption to international flights (Bloomberg News, 2013). In February 2013, the Beijing Municipal Environmental Protection Bureau (EPB) released two new local standards, specifically designed to prevent excess NO_x emissions from HDVs. The two Beijing standards, which are supplements to the existing national China IV and V standards (comparable to Euro IV and V), apply to China IV and V vehicles with Gross Vehicle Weight above 3,500kg and registered in Beijing. The two standards are as follows:

DB11/964-2013, 4 — requires China IV and V engines to be tested over the World Harmonised Transient Cycle (WHTC) in addition to the currently required European Transient Cycle (ETC). In the EU, testing over the WHTC is not required until the Euro VI stage. Both cold-start and hot-start testing are required, with results weighted 14% and 86% respectively.

DB11/965-2013, 5 — establishes in-use, complete vehicle, Portable Emission Measurement System (PEMS) testing requirements for manufacturers, to prove that real-world emissions do not overly exceed certification limit values.

4.3 Real-world and legislative emission levels compared – LDVs

Carslaw et al. (2011) present emissions data compiled from roadside sensing of vehicle exhaust. The data is based on analysis of over 84,000 exhaust plumes, measured in seven urban locations across the UK. Although Remote Sensing Devices (RSDs) provide data only from a single point location, rather than for a complete driving cycle, and suffer from other practical constraints to their application when combined with automatic number-plate

recognition (ANPR), they offer tremendous insight into real-world emissions from a large sample of the vehicle fleet. They are particularly useful as a reality check on emission factors assumed for emission inventory models, as discussed in Chapter 5.

Figure 24 is a summary plot presented by Carslaw et al. (2011). RSD cannot directly measure absolute rates of NO_x emission (in $g.km^{-1}$); instead it measures the ratio of NO_x to CO_2 – effectively, the NO_x emission rate per unit of fuel consumed. This ratio is plotted, as a function of vehicle age, for four classes of vehicle, and two fuel types.

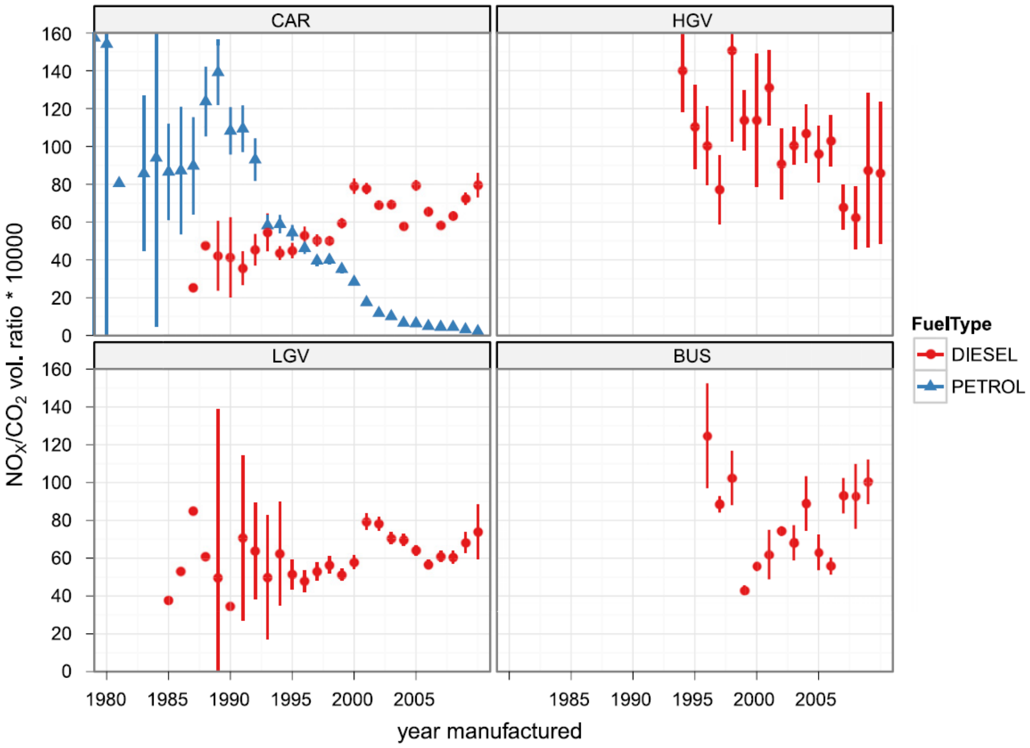

Figure 24 NO_x:CO_2 ratio for major classes of vehicle, from remote-sensing data. Error bars indicate 95% confidence interval in the mean. [Carslaw et al., 2011]

Looking at the "car" data, it is clear that the introduction of three-way catalysts (TWC) on petrol-engined cars has led to dramatic reductions in NO_x emissions from these vehicles. Cars registered since 2003, in particular, exhibit very low emissions of NO_x, although the performance of older, TWC-equipped vehicles suggests that effectiveness may decline with age. NO_x emissions from diesel-powered cars, on the other hand, have been steadily increasing despite the ever-tightening Euro emission limits. NO_x emissions from HGVs appear to be declining with time, whilst emissions from LGVs show little change: if anything, newer LGVs appear to emit more NO_x than their older counterparts.

Lee et al (2013) tested twelve Euro 3–5 light-duty diesel vehicles (LDDVs). The vehicles, all of Korean manufacture, were driven on a chassis dynamometer over the NEDC and over a representative Korean on-road driving cycle (KDC). NO_x emissions were measured over both cycles; average speeds, accelerations and NO_x emissions were also calculated for each 1-km segment on each cycle.

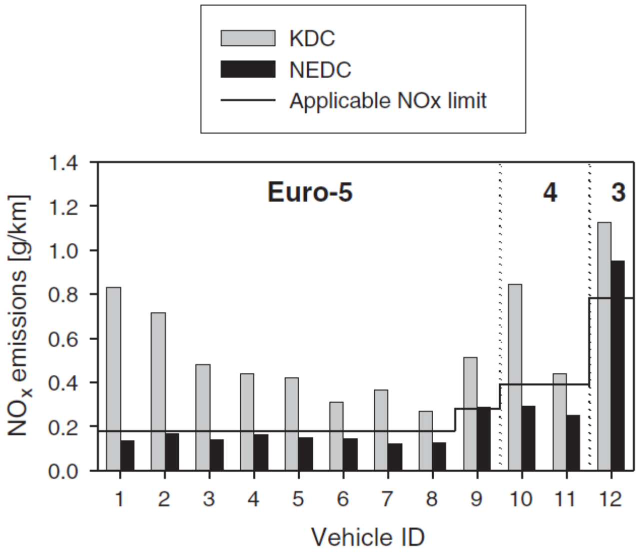

Figure 25 Route-averaged NO_x emissions from 12 selected Euro 3-5 diesel LDVs, over the NEDC and a representative Korean driving cycle (KDC). [Lee et al 2013]

Figure 25 presents the route-averaged NO_x emissions for all twelve vehicles, over both the NEDC and the KDC. It is clear that, whereas all vehicles meet the relevant certification criteria when driven on the NEDC, real-world driving – as captured by the KDC – yields substantially higher levels of NO_x emission. The exceedance ratio (ratio of real-world emissions:certification limits), and vehicle-to-vehicle variability, is highest for the Euro 5 vehicles.

The authors report that that NO_x emissions from the vehicles tested are more susceptible to variations in driving cycle than to those in operating condition, and support this assertion with the plot shown in Figure 26. This presents the NO_x emissions from five Euro 5, diesel LDVs, measured in each 1-km window over the NEDC and KDC. Emissions are plotted as a function of RPA for “low speed” (average speed < 40 km.h^{-1}), “medium speed” (40 – 80 km.h^{-1}) and “high speed” (> 80 km.h^{-1}) windows. It is clear that operation at low vehicle speeds with high RPA produces the highest NO_x emissions, and that NO_x emissions over the KDC are sensitive to RPA. The authors point out that NO_x emissions vary with RPA by up to a factor of 6, 12, and 5 in the “low speed”, “medium speed” and “high-speed” modes of the KDC respectively. This sensitivity is not observed on the NEDC, suggesting that the EGR control strategy for these vehicles has been optimised for the NEDC, rather than for real-world conditions.

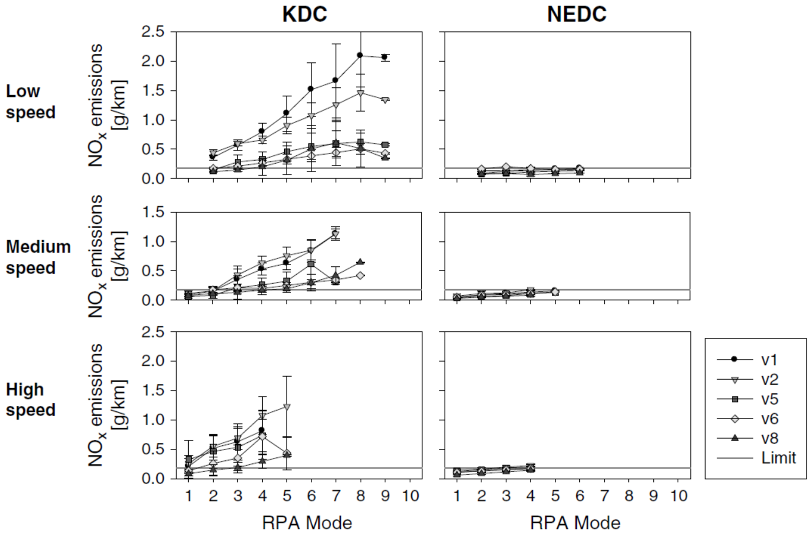

Figure 26 The effect of vehicle average speed and RPA on NO_x emissions over the NEDC and KDC, for five selected Euro 5 diesel LDVs. [Lee et al, 2013]

Weiss et al (2012) measured NO_x emissions from seven diesel passenger cars in a laboratory over the NEDC, and in real-world driving with PEMS over four separate test routes. The test routes are designed to capture a large part of the characteristics of European on-road driving, in terms of altitude profile, speed range, and driving dynamics. Two of the cars were to Euro 4 standard, four to Euro 5, and one to Euro 6. All cars used exhaust gas recirculation (EGR) for NO_x control (see Section 2.3); the Euro 6 vehicle also employed selective catalytic reduction (SCR).

The objective of the testing was to compare the real-world NO_x emissions with those measured over the NEDC, and to evaluate the relative effectiveness of the three certification levels tested. The vehicles were not pre-conditioned for the on-road tests, and were tested in each season of the year within an overall ambient temperature range of 0 – 35^{0}C.

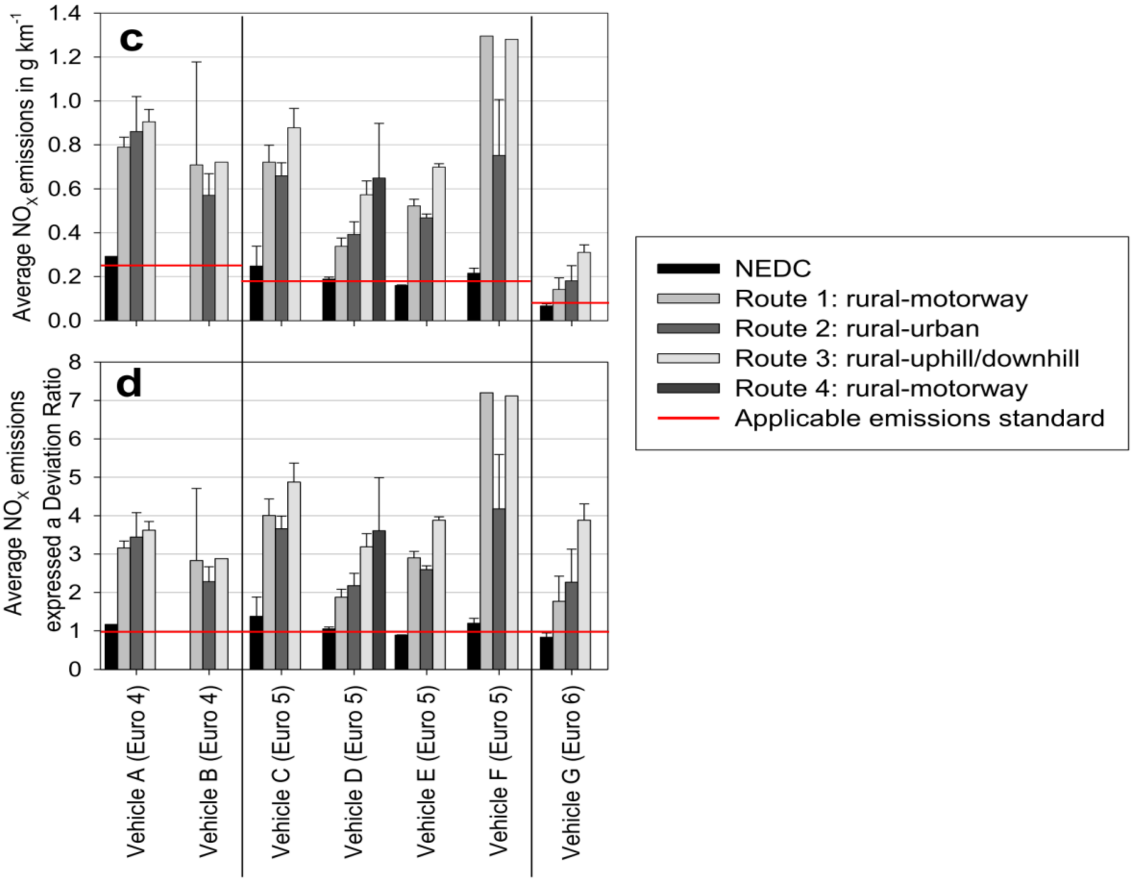

Figure 27 Average NO_x emissions on PEMS test routes, and during NEDC testing in the laboratory, for six diesel-powered passenger cars. (Weiss et al. 2012)

Figure 27 summarises the overall NO_x emissions from each vehicle, averaged over each test route and on the NEDC. The upper graph in Figure 27 (labelled **c**) presents absolute NO_x emissions for each vehicle in g.km^{-1}, along with a horizontal red line denoting the applicable emission level for certification on the NEDC. It is notable that all vehicles meet, or come close to meeting, the relevant certification emissions level when tested over the NEDC. The real-world NO_x emissions, however, are substantially above the relevant certification standard. Moreover, NO_x emissions from the Euro 5 vehicles are equal to – and even exceed – those from Euro 4 vehicles under real-world conditions, even though their NEDC emissions are lower. NO_x emissions from the Euro 6 vehicle, equipped with SCR, are substantially lower than from Euro 4 and 5 cars under both NEDC and real-world conditions.

The lower graph in Figure 27 (labelled **d**) presents the same data, but expressed as a “deviation ratio”: it compares the on-road emissions (in g.km^{-1}) to the appropriate certification level. Expressed in this way, the performance of the Euro 6 car seems somewhat less impressive, although it offers a clear step forward relative to the Euro 5 vehicles tested. It is sobering to note that, under real-world driving conditions, the NO_x emissions from the vehicles tested were between two and seven times higher than the certification values.

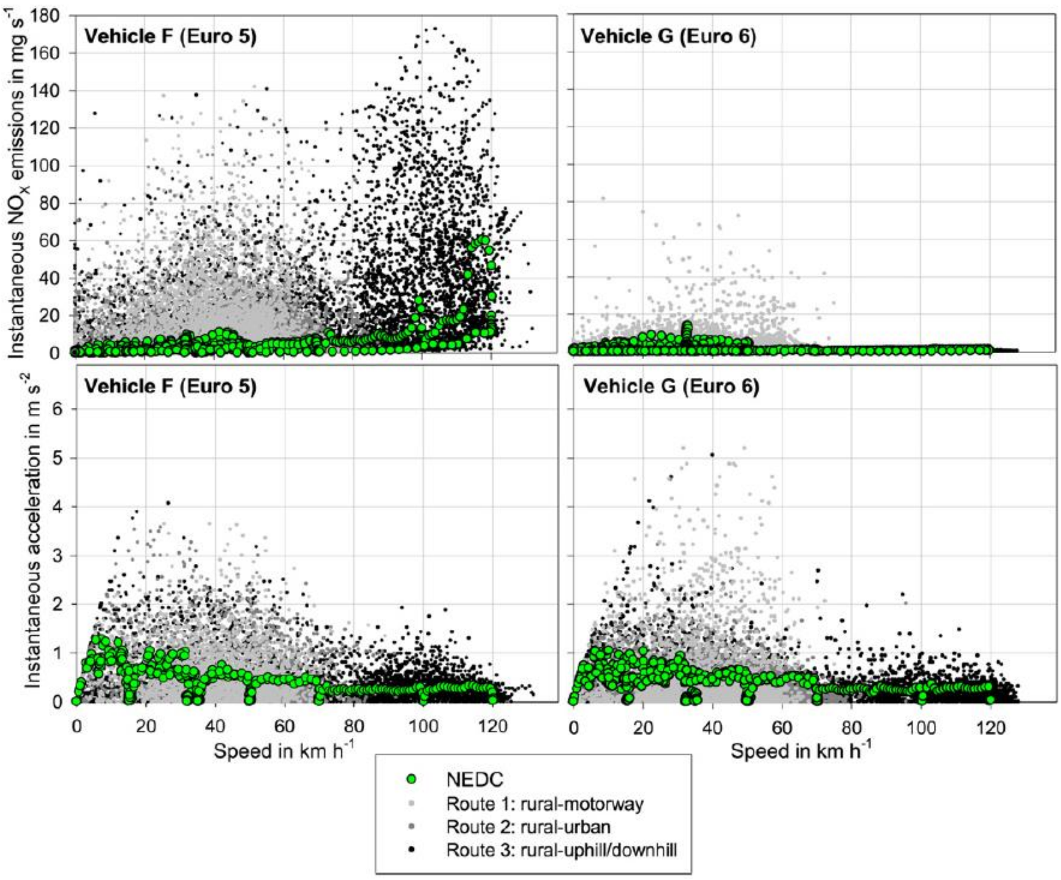

Figure 28 Instantaneous NO_x emissions from a Euro 5 (left) and a Euro 6 vehicle on the NEDC (green circles) and during real-world driving. (Weiss et al. 2012)

Figure 27 clearly highlights the significance of driving cycle for NO_x emissions from diesel LDVs, irrespective of the technology employed. Further evidence of this effect is presented in Figure 28 and Figure 29. Figure 28 compares the instantaneous NO_x emission rate from the Euro 6 car with that of one of the Euro 5 vehicles, as a function of vehicle speed. Emissions measured on the NEDC are plotted as green circles, those measured on the road as circles in various shades of grey. It is clear that both vehicles manage to control NO_x emission rates on the NEDC, although the Euro 5 vehicle struggles somewhat at higher vehicle speeds. Under real-world operating conditions, however, the performance of the Euro 5 vehicle is much worse, at all vehicle speeds. The Euro 6 vehicle (with SCR) manages to contain NO_x emissions very well, except at low vehicle speeds – this behaviour mirrors that discussed earlier in the context of HDVs (see Figure 21 and Figure 22).

Perhaps the clearest evidence of the variability of real-world emissions is presented in Figure 29. Rather than route-average emissions, the data plotted in Figure 29 refers to emissions produced within shorter sub-sections ("averaging windows") contained in each test route. The time duration of averaging windows is chosen so that each window covers exactly the amount of CO_2 the respective test vehicle has emitted during type-approval testing over the NEDC. The distance covered by an averaging window is vehicle-specific, and varies with the instantaneous fuel consumption.

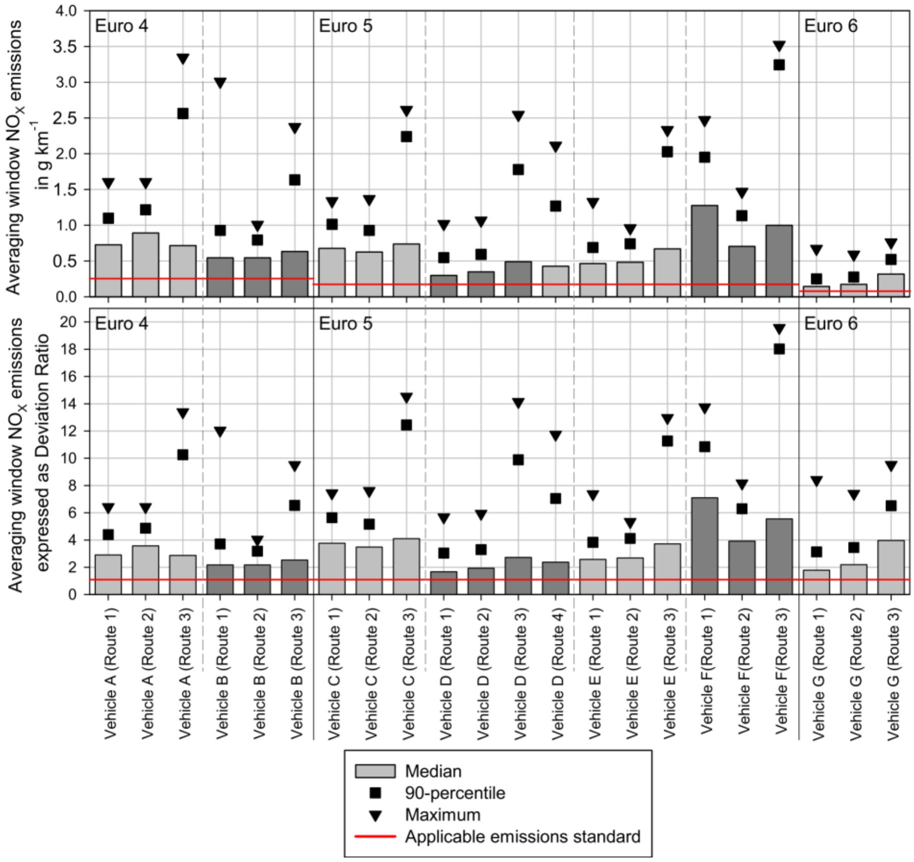

Figure 29 Averaging-window NO_x emissions, from Euro 4, Euro 5, and Euro 6 vehicles, cover a very wide range of values under real-world driving conditions. (Weiss et al. 2012)

Examination of Figure 29 reveals that NO_x emissions during an averaging window could exceed certification levels by a factor of 13 for the Euro 4 cars, by a factor of 20 for the Euro 5 cars, and by a factor of 10 for the Euro 6 car. The authors report that, in fact, all averaging windows for the Euro 4 diesel cars and 97% of the windows for the Euro 5 diesel cars exceed the respective emissions standards. For the Euro 6 diesel car, 90% of the averaging windows also exceed the emissions standard. Nonetheless, this vehicle shows substantially lower emissions than the Euro 4 and Euro 5 cars on all test routes, suggesting that the SCR system is technically capable of reducing NO_x emissions over a wide range of driving conditions.

5 Emissions inventory models

5.1 Overview

The production of an accurate and reliable estimate of national emissions from road transport presents a significant challenge. Several software packages have been developed to assist with the creation of an emissions inventory from a specific area, for a specific time period.

The two main programmes used within Europe are COPERT – which assumes that emissions from a given vehicle are essentially a function of its average speed – and ARTEMIS/HBEFA, which bases emissions estimates on a more detailed and sophisticated representation of the vehicle drive cycle.

All emission inventory programmes use the same basic methodology for estimating emissions:

Total emissions per year =	(emissions per vehicle, per km travelled) x
	(km travelled per vehicle, per year) x
	(number of vehicles)
This can be expressed as:	
Total emissions per year =	(vehicle emission factor (***EF***)) x
	(vehicle activity (***A***))

Usually, the emission factor (*EF*) is expressed in $g.km^{-1}$, and the vehicle activity *A* in vehicle-kilometres (vkm). The *EF* is, in principle, a universally-applicable parameter specific to a particular vehicle type, whereas the activity data *A* is specific to a particular region. It is the responsibility of the model developer to provide accurate *EFs*; it is the responsibility of the user to provide accurate activity data. A realistic emission inventory can be produced only if both sets of data are accurate.

5.1.1 Emission factors (EFs)

The straightforward methodology outlined above is complicated by the fact that *EFs* vary with vehicle type, size, technology level and other factors. Disaggregation of the vehicle fleet is therefore a common feature of most emissions inventory models (Joumard et al. 1994), so that a specific *EF* can be defined for each vehicle type. Fleet decomposition usually takes the following form:

1) Class: *e.g.* passenger car / LGV / bus / HGV / motorcycle
2) Sub-class: based on engine size, vehicle weight, or other vehicle attribute
3) Fuel type: petrol / diesel / LPG / hybrid / etc.
4) Technology level: Euro 4, etc.

The challenge of producing accurate *EFs* is further compounded by the fact that, as many studies clearly show, emissions of PM and NO_x from road vehicles are strongly dependent on a vehicle's *mode of operation* (Weiss et al. 2012, Liu et al. 2011, Hausberger et al. 2009). Such studies demonstrate that the *EFs* of any given vehicle, particularly with respect to NO_x and PM, can vary substantially, depending on driver behaviour (which determines vehicle speed, levels of acceleration, gear selection, etc.) and on external parameters, such as congestion levels, ambient temperature, and so on.

Therefore, programme developers define *EF*s based on particular scenarios, or combinations of vehicle type + activity type. The greater the number of scenarios considered (*i.e.,* the larger the number of vehicle types and activity types included in the model), the more accurate the individual *EFs* are likely to be. On the other hand, the task of determining accurate *EFs* for a large number of scenarios can quickly become overwhelming. As discussed in Section 5.1.2, a large set of scenarios can also impose unrealistic requirements on the granularity, temporal resolution, and accuracy of activity data.

5.1.2 Activity data (A)

The model developer provides *EFs* for a range of scenarios, as described in Section 5.1.1. Regional emission inventories are then compiled by weighting these *EFs* in accordance with local fleet composition and activity levels. Hence, a region-specific Fleet model and Vehicle Activity model are required.

Given this information, the total mass of pollutant *i* emitted by a vehicle fleet disaggregated into *m* types, each undertaking *n* types of activity, can be given by:

$$E_{i,total} = \sum_{j=1}^{m} \sum_{k=1}^{n} \left(EF_{i,j,k} \times A_{j,k} \right)$$

where *j* denotes the vehicle class (as defined by the fleet disaggregation methodology), and *k* denotes the activity type.

Unfortunately, this brings a requirement for highly granular activity data, since accuracy in only one of the two main parameters will not deliver an accurate estimate of total emissions. The greater the number, *m*, of vehicle types, and the number, *n,* of activity types considered, the greater the potential accuracy of the model.

However, the challenge of measuring (and managing) the activity data increases dramatically with increasing *m* and *n*. In practice, activity data with high granularity is not available. Instead, the most common approach is to consider a small number of generic activity types (e.g. rural, urban, and motorway driving), and to estimate, sometimes with very limited empirical data, the level of activity associated with each type.

The approach most commonly used characterises activity type using the single parameter of vehicle average speed (as described in Section 5.2), and ignores all other driving kinematics. This is the approach adopted within COPERT, which is currently used in Ireland and in many other European countries to estimate emissions from the road transport sector.

A second approach, less widely-used, acknowledges that exhaust emission levels, particularly with respect to NO_x and PM are strongly influenced by driving-cycle kinematics (de Haan and Keller, 2000). An attempt is made to capture these kinematic influences by defining *EFs* for a wide range of activity types, each associated with a specific "Traffic Situation" (TS). The TS methodology is

incorporated in HBEFA v3.1 (Handbook of Emission Factors), which is supported by Germany, France, the Joint Research Centre (JRC) of the European Commission, and others, and which was used (alongside COPERT) in the ETASCI project. Sections 5.2 and 0 of this Chapter provide an overview of the COPERT and HBEFA methodologies respectively.

5.2 Emissions inventory based on vehicle average speed: COPERT

In Europe, the most widely-used inventory model based on the average-speed approach is COPERT (Computer Programme to calculate Emissions from Road Traffic)[5]. COPERT is used primarily for compiling national inventories of legislated, atmospheric pollutants from the road transport sector and, less frequently, to project future emissions.

COPERT was originally developed in 1989 by the European Environment Agency (EEA), and has been continuously developed since: the current iteration (as of September 2013) is Version 10.0.

COPERT offers a very high level of fleet disaggregation (see Table 1), implying a requirement for highly granular activity data. The user need not attribute emissions to all vehicle types, but the trade-off for such a decision is reduced confidence in the accuracy of the emission estimates.

[5]COPERT Website:
http://www.emisia.com/copert/

Table 1. **Vehicle types defined in COPERT**

Vehicle class	Sub-classes	Fuel types	Technology levels	Total number of vehicle types
Passenger cars	3	4	12	**56**
Light-duty vehicles	1	2	6	**12**
Heavy-duty trucks	15	1	6	**55**
Buses	5	1	6	**20**
Mopeds	1	1	4	**4**
Motorcycles	4	1	4	**16**
Totals	**29**	**4**	**12**	**163**

On the other hand, only three activity types are considered: urban, rural, or motorway driving. Moreover, each of these activities is characterised only by its average speed – which the user can select to best represent local conditions. The assumption implicit in configuring the model this way – with 163 vehicle types but only three activity types – is that emissions are far more sensitive to vehicle type than to activity type. Unfortunately, as the discussion of real-world driving emissions (RDE) in Chapter 4 reveals, that is not the case.

The decision to characterise vehicle activity using only average speed further undermines confidence in the accuracy of the emission estimates. The *EFs* corresponding to a particular average speed is obtained by fitting a curve to laboratory measurements of *EFs* from several vehicles over a range of driving cycles, with each cycle representing a specific type of driving (i.e., activity). This method works effectively for some pollutants, such as CO_2, for which emissions are closely related to vehicle average speed (TRB 2008). However, as noted in, for instance, (de Haan and Keller, 2000) and (Barlow and Boulter, 2009), for other emissions, such as PM and NO_x, the agreement is not as satisfactory, since trips having very different vehicle operation characteristics, and correspondingly different emission levels, can have the same average speed; this is a particular problem at low-medium average speeds.

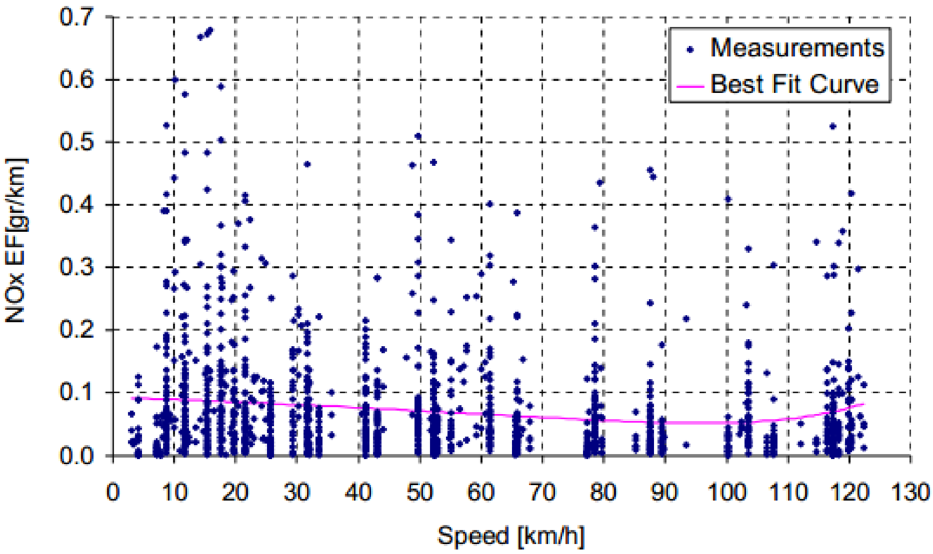

Figure 30 Example of the application of a best fit curve for NO_x emissions for Petrol Euro 3 passenger cars, tested within the ARTEMIS project. Note the large variation of each of the individual tests (blue dots) (Kouridis et al. 2009)

Figure 30 shows the construction of a polynomial curve fit to the NO_x emissions of a sample passenger car, as a function of vehicle speed. As can be seen in the graph, the fit of the polynomial curve (pink) to the data (blue dots) is very poor, and the large variation in NO_x emission factors obtained from individual tests is obvious. Moreover, in modern vehicles emissions frequently occur in short, sharp peaks associated with a particular transient. Therefore, vehicle *average* speed is not a reliable indicator of emission intensity (Barlow and Boulter, 2009).

Notwithstanding these shortcomings, once polynomial curves have been defined for each pollutant, it is possible to calculate an *EF* at any desired average speed. Figure 31 indicates the shape of these COPERT curves for NO_x from petrol and diesel passenger cars up to Euro 4.

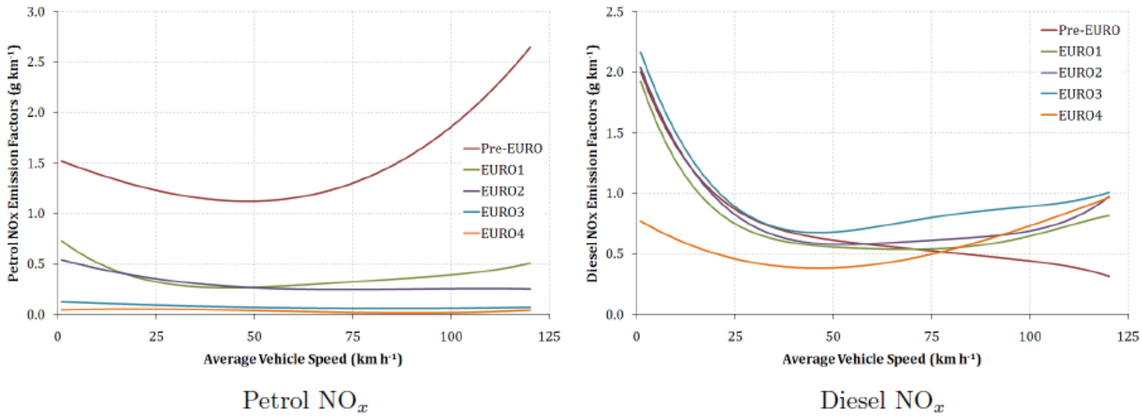

Figure 31 A graphical description of the NO_x emission factors for petrol and diesel private vehicles as a function of average vehicle speed. (Boulter, 2007)

5.3 Emissions inventory based on "traffic situations": (ARTEMIS/HBEFA)

The Handbook of Emission Factors for Road Transport (HBEFA)[6] was originally developed on behalf of the Environmental Protection Agencies of Germany, Switzerland and Austria, to estimate national emissions from road transport for all regulated, and many non-regulated pollutants. Sweden, Norway and France have also commissioned the HBEFA method for national emissions estimation, and it is supported by the Joint Research Centre of the European Commission (JRC). The initial version of the HBEFA software emissions inventory was published in December 1995. Version HBEFA 2.1 was made available in February 2004. The newest version HBEFA 3.1 dates from January 2010, and is very closely linked to the ARTEMIS software inventory model v04d.

Like COPERT, HBEFA offers a high level of fleet disaggregation, with a similar categorisation of vehicle types. However, whereas COPERT offers only three activity types (each characterised solely by its average speed), HBEFA offers no less than 276. Each activity type represents a particular "Traffic Situation" (TS).

5.3.1 Traffic Situations

Traffic Situations (TSs) attempt to capture the effect on emissions of a wider range of driving-cycle-related variables, in place of the average-speed approach adopted by COPERT. A TS is defined by four main characteristics:

1. Type of region (urban or rural)
2. Type of road (motorway, access, etc.)
3. Speed limit on road
4. Traffic flow condition (free-flowing, congested, etc.).

The Traffic Situations developed as part of the ARTEMIS programme are presented in Table 2.

[6] http://www.hbefa.net/e/index.html

Table 2. ARTEMIS Predefined Traffic Situations consider road type, speed limit, and four possible traffic flow conditions (276 individual Traffic Situations).

Area	Road type	Speed limit (km/h)	Traffic flow conditions
Urban	Motorway - National (Through Traffic)	80, 90, 100, 110, 120, 130	4 levels of service
	Motorway - City	60, 70, 80, 90, 100, 110	4 levels of service
	Main Trunk Road - National	70, 80, 90, 100, 110	4 levels of service
	Trunk Road - City	50, 60, 70, 80, 90	4 levels of service
	Distributor - DistrictConnection	50, 60, 70, 80	4 levels of service
	Local Collector	50, 60	4 levels of service
	Access - Residential	30, 40, 50	4 levels of service
Rural	Motorway	80, 90, 100, 110, 120, 130, >130	4 levels of service
	Semi motorway (2+1 lanes, variable)	90, 110	4 levels of service
	Trunk Road	60, 70, 80, 90, 100, 110	4 levels of service
	Distributor-DistrictConnection	50, 60, 70, 80, 90, 100	4 levels of service
	Distributor-DistrictConnection (withCurves)	50, 60, 70, 80, 90, 100	4 levels of service
	LocalCollector	50, 60, 70, 80	4 levels of service
	Local Collector (with Curves)	50, 60, 70, 80	4 levels of service
	Access-Residential	30, 40, 50	4 levels of service

The traffic flow associated with a particular TS is described using one of four “levels of service”. These are, in decreasing order: free-flowing; heavy; quasi-saturated; or stop-and-go. Figure 32 indicates the relationship between level of service, vehicle average speed, and traffic flow rate, for a main road, whilst Figure 33 presents samples of four common road types.

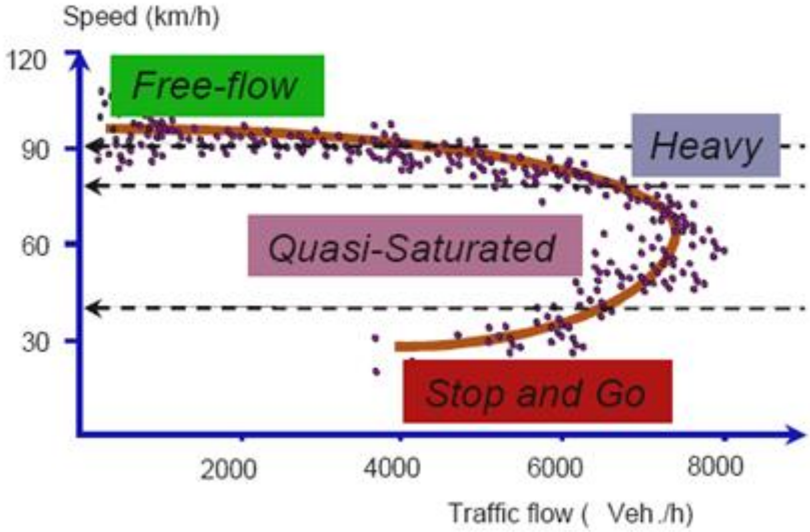

Figure 32 Four levels of service available within the ARTEMIS traffic situation model, which are used to describe the traffic conditions. In general, the level of vehicle emissions will increase as the traffic conditions become congested (Stop and Go).

Figure 33 Sample of road types associated with a particular Traffic Situation (TS). (A) National trunk road, (B) Rural local distributor, (C) Urban district, and (D) Urban local distributor.

In general, TS models are more complex than average-speed models, and require more detailed inputs from the user. Their application at a national level may be challenging, but they offer some hope of providing a representative estimate of road transport emissions at a local level. The accuracy of the estimates produced, of course, will depend on the accuracy of the *EFs* associated with each combination of TS + vehicle-type, and on the accuracy of the relevant fleet composition and activity weighting matrices.

5.3.2 *Relating emission factors (EFs), to traffic situations (TSs)*

As noted in Section 5.3.1, the accuracy of emission estimates depends on the accuracy of the *EFs* associated with each combination of TS + vehicle-type. Given the large number of TSs and of vehicle types available in HBEFA, experimental determination of *EFs* for each pollutant is not practical. In the ARTEMIS programme, a subset of fifteen "Representative Test Patterns" (RTPs) was constructed by combining TSs in various combinations, to yield driving cycles that were statistically representative of typical European driving conditions. *EFs* for these RTPs were then measured, in a laboratory, for a subset of vehicle types; use of this approach for all vehicle types was deemed impractical. However, the *EFs* established in this programme provide useful model calibration data.

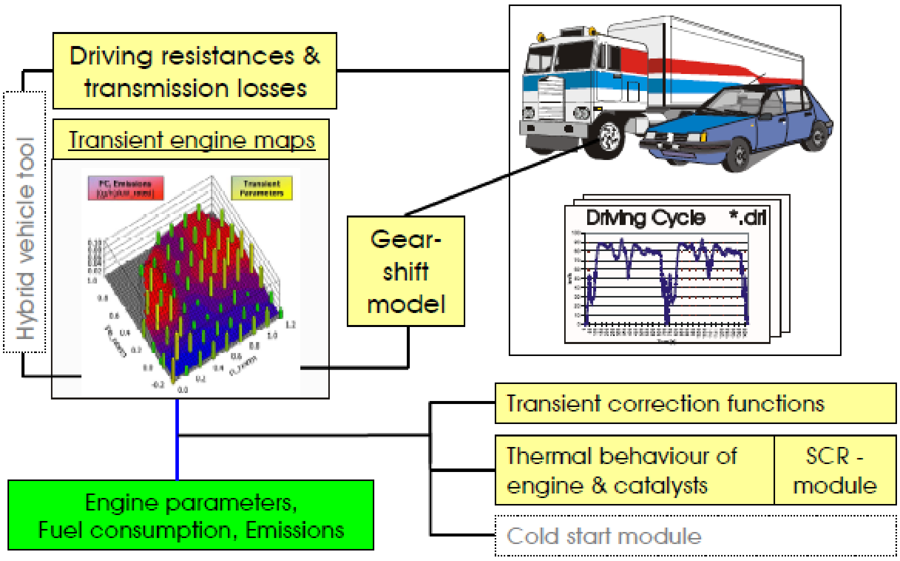

Figure 34 Scheme of the emission model PHEM, used to determine HBEFA 3 emission factors. (Hausberger et al. 2007)

In HBEFA v3.1, the definition of *EFs* for each pairing of (TS + vehicle-type) follows a two-step process. Phase 1 involves direct measurement of emissions from a representative vehicle type on a rolling road, across a range of engine speeds and loads, in order to establish emission “maps”. In Phase 2, these emission maps are fed into a software model of the vehicle, called PHEM (Passenger car and Heavy-duty Emission Model). By “driving” the model of a particular vehicle type through each TS, an *EF* corresponding to each (TS + vehicle-type) pair can be found. Figure 34 presents an overview of the *EF* calculation scheme used in PHEM.

The use of a software model such as PHEM has many advantages: the *EFs* obtained are 100% repeatable; the effects of road gradient, of gear-shift strategy, and of many other factors, can be easily and cheaply investigated; high-emission sectors of the driving cycle can be easily identified; and so on. Figure 35 provides an overview of some of the data obtained in this way: it is evident that emission estimates can be obtained from a large number of (TS + vehicle-type) pairs.

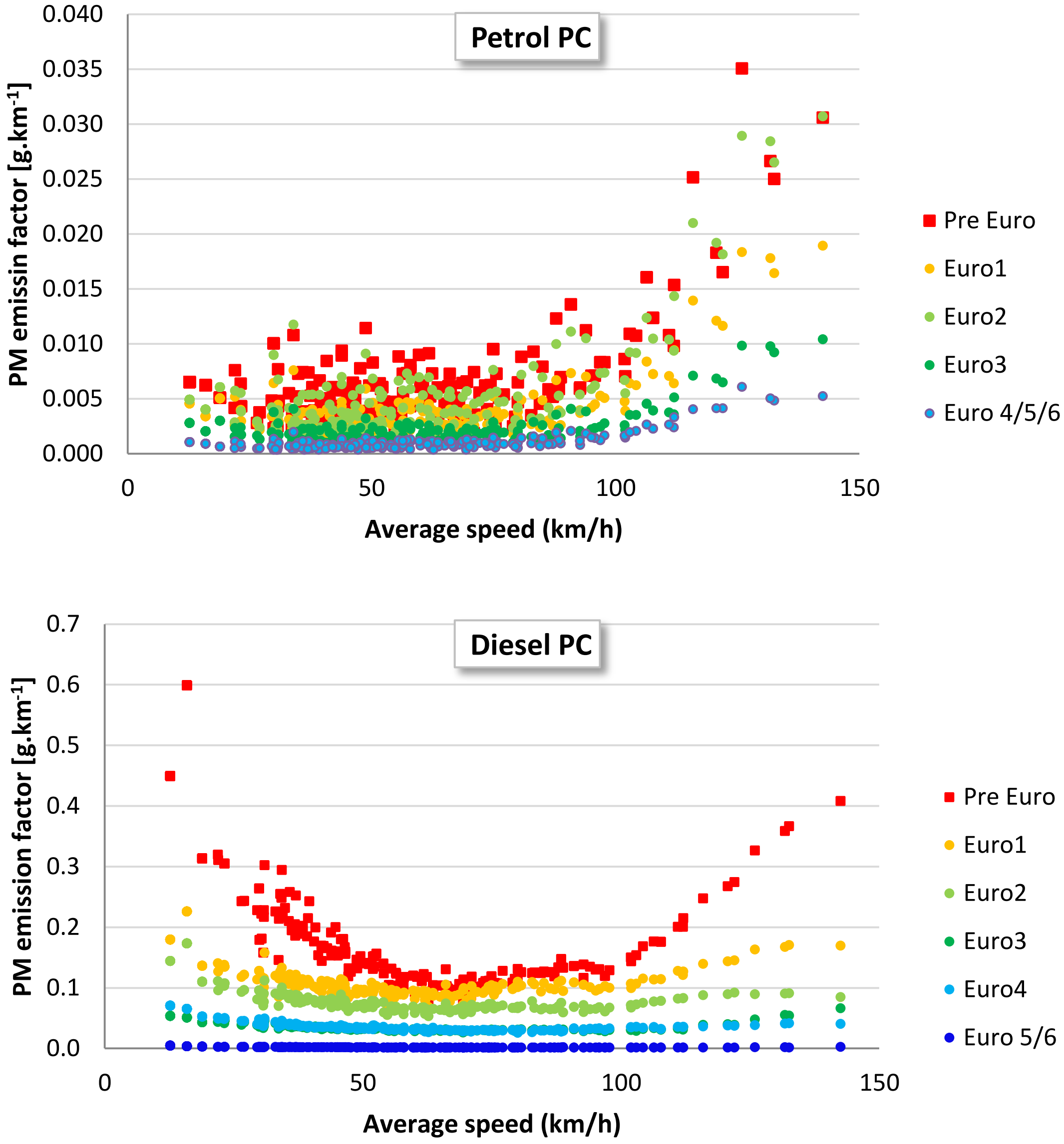

Figure 35 HBEFA v3.1 PM emissions factors for petrol and diesel PCs for all 276 traffic situations.

However, a model is just a model; the accuracy of its output depends on the quality of the input and on the accuracy with which the physics of the system have been modelled. The use of emission maps derived from steady-state testing, to predict emissions during transient operation, raises obvious concerns. As engines, EMS algorithms, and tailpipe treatment systems become increasingly sophisticated and complex, the difficulty of modelling the system's dynamic behaviour increases dramatically. Disconcertingly, verification of the model predictions also becomes more challenging. Despite the sophistication of the PHEM model, and the care taken in establishing emission maps for each vehicle type, the *EFs* incorporated in HBEFA v3.1 must therefore be interpreted with caution.

5.4 Implementing a Traffic Situation (TS) model

As noted in Section 5.2, HBEFA (and ARTEMIS) provide *EFs* for an enormous combination of (TS + vehicle-type) pairs. In practice, it is impossible to accurately map

activity data onto such a comprehensive data set. However, it might be possible to identify a subset of TSs that are representative of traffic movements in a particular region and, using this subset, to produce an emissions inventory that reflects RDEs more accurately than is possible with a simpler, average-speed model. However, in order to implement this approach, a user must:

- Measure local, real-world, driving cycles
- Map the measured driving cycles onto the TS provided.

Each of these tasks presents its own challenges, and both were investigated as part of the ETASCI project.

5.4.1 Measurement of local, real-world, driving cycles

The principal output of a driving cycle measurement event is a time vs speed trace for the vehicle being monitored, as shown for example in Figure 36. Additional data, such as vehicle height above sea level, is also recorded, and can inform the emission estimation process.

As part of the ETASCI project, real-world driving cycle data was collected for a variety of trip and vehicle types. An inexpensive system, based around a mobile-phone, was developed and implemented. Data was collected for urban, rural, and motorway trips, of long and short duration, and across each season of the year. In some situations, the system was extended to monitoring engine operating condition in real-time, using a vehicle's On-Board Diagnostics (OBD) socket. A complete description of the hardware and software employed is presented in Grummell (2013).

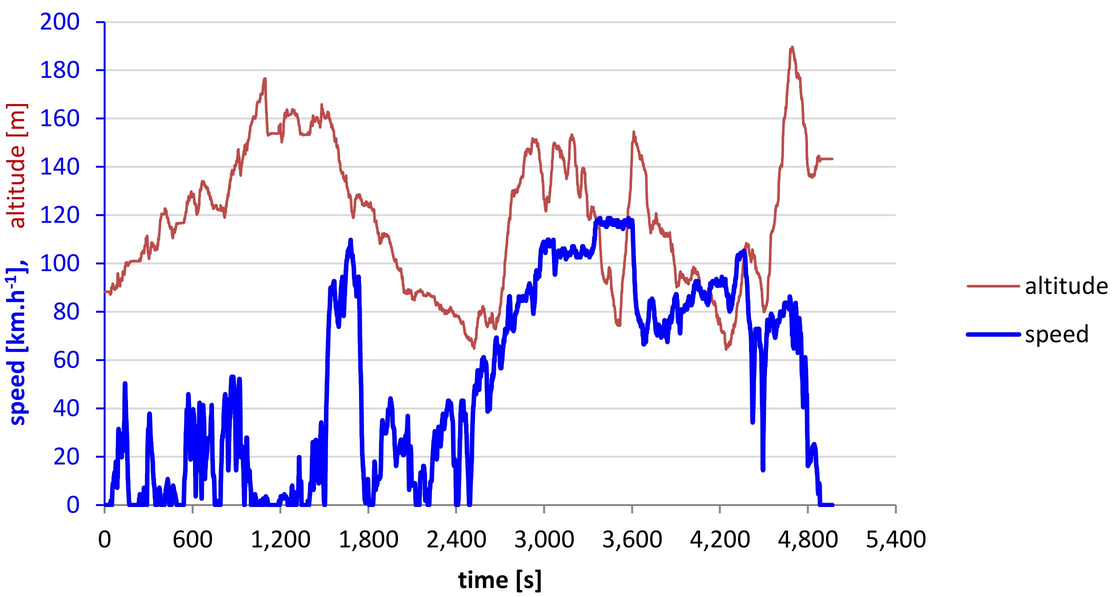

Figure 36 Speed (and altitude) vs time, for a trip monitored in ETASCI.

The major challenges identified during this phase of the project included:

- management of the large quantities of data generated (including storing, indexing, validity checking, integrity checking)
- capturing relevant locational data whilst maintaining source anonymity
- hardware fault detection and correction
- post-processing of the data.

These issues are discussed in Grummell (2013). Section 5.4.2 presents an overview of the approach used to map the measured driving cycles to the TSs provided in HBEFA v3.1, whilst Sections 6.4.1 and 6.4.2 describe a micro-study of that mapping process carried out within ETASCI.

5.4.2 *Kinematic characterisation of driving cycles*

Although kinematic data, such as that presented in Figure 36, is valuable in itself, estimating emissions for such a trip depends on being able to map the measured "speed vs time" (kinematic) trace, onto one or more of the HBEFA traffic situations. That requires a method for characterising an arbitrary kinematic trace using a limited number of scalar parameters. A large variety of parameters is considered in the literature, from which the following were selected during ETASCI:

- Average speed (Vavg)
- Peak vehicle speed (Vmax)
- Relative positive acceleration (RPA)
- Velocity fluctuation index (VFI).

The first two of these are self-explanatory, although there is a subtlety associated with defining the segment over which they are calculated. The **relative positive acceleration (RPA)** is defined as:

$$RPA = \frac{1}{x}\int_{0}^{t} v.a^{+}dt$$

where: x denotes trip segment length [m]

v denotes vehicle speed at time t [$m.s^{-1}$]

a^{+} denotes vehicle positive acceleration at time t [$m.s^{-2}$]

and is a measure of the average engine power required for positive vehicle accelerations.

The **velocity fluctuation index (VFI)** quantifies the variability in vehicle speed, and is defined as:

$$VFI = \frac{\frac{1}{n}\sum|v_i - \overline{v_s}|}{\overline{v_s}}$$

where:.v_i denotes vehicle speed at instant i [$m.s^{-1}$]

v_s denotes average vehicle speed over the kinematic segment being analysed [$m.s^{-1}$]

n denotes the number of samples within the kinematic segment.

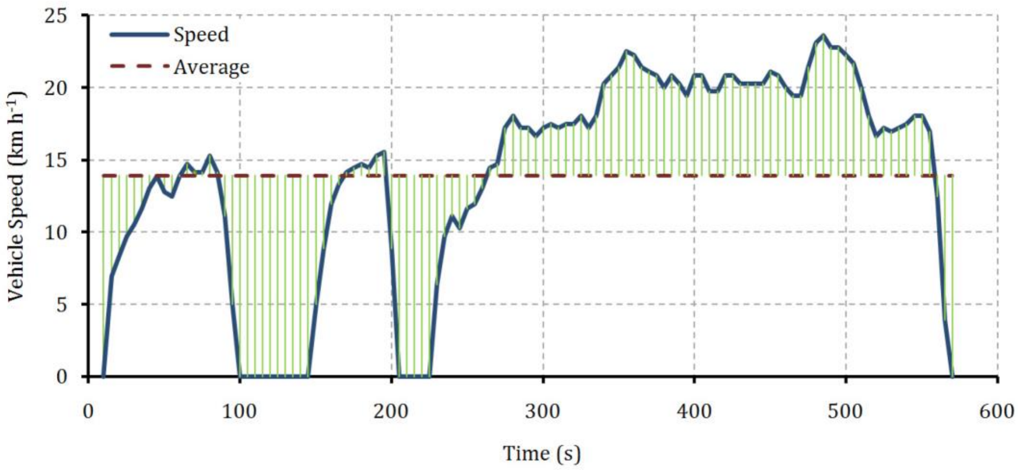

Figure 37 A schematic representation of the velocity fluctuation index (VFI). (Casey, 2011)

VFI was developed as part of the EPA-funded Urban Environment Project[7], and a detailed discussion of VFI can be found in Casey (2011). Figure 37 reveals that VFI is a measure of the average deviation of vehicle speed, from the mean speed over a segment. The green lines in Figure 37 represent the difference between the instantaneous velocity (v_i) and the average velocity ($\bar{v}$) for a segment. The average of these values, normalised against the segment's average velocity, yields the VFI.

As mentioned above, calculation of these parameters is straightforward *once the relevant kinematic segment has been defined*. However, definition of a kinematic segment is itself somewhat challenging. Referring to Figure 38, as an example, it is clear that this particular trip comprises several distinct phases. These might be defined as follows:

- 0 – 1,500 seconds: low-speed (urban), segment A
- 1,500 – 1,800 s: high-speed, segment B
- 1,800 – 2,500 s: low-speed (urban), segment C
- 2,500 – 4,900 s: high-speed (motorway), segment D

However, this segmentation is completely arbitrary; another user might break the trip into more, or fewer, segments, with corresponding changes in each segment's start and finish time. As shown in the micro-study presented in Sections 6.4.1 and 6.4.2, changes in the segmentation scheme applied to a given trip, result in changes to the *EFs* calculated for that trip.

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[7] See www.uep.ie

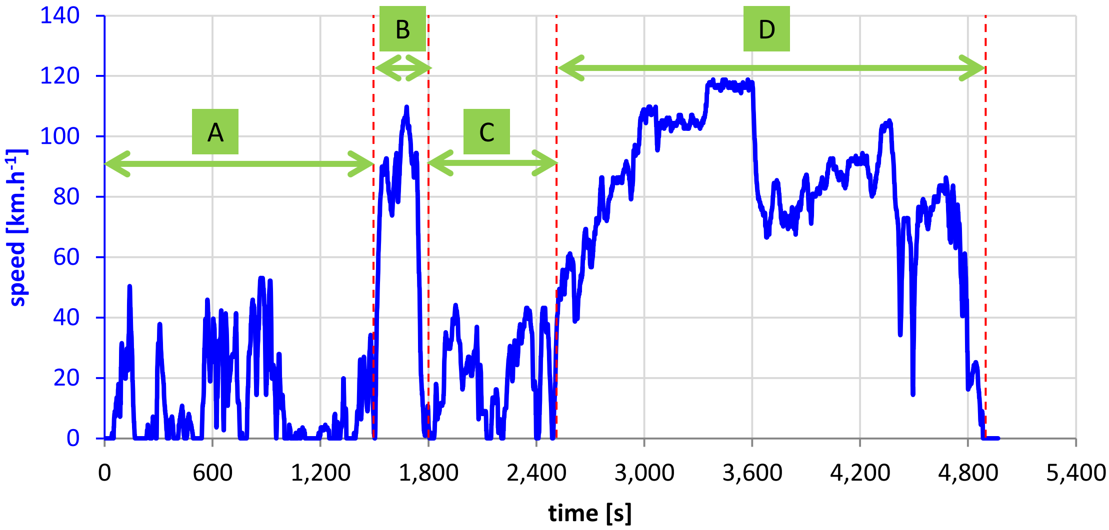

Figure 38 Speed vs time, for a trip monitored in ETASCI.

Once a monitored trip has been segmented, and the kinematics of each segment characterised, it is still necessary to map each segment onto one or more of the TSs for which *EFs* are available. Again, a variety of approaches is suggested in the literature, and the *EFs* attributed to the trip will depend on the scheme adopted and on the number of trip segments employed. Casey (2011) recommends weighting the contribution of a particular TS to the segment *EFs*, based on its proximity to the segment when plotted on a VFI-vs maximum speed plane, as indicated in Figure 39.

In practice, obtaining accurate emission inventories at national level using the TS approach is fraught with challenges and uncertainties, and would require resources massively exceeding those allocated to the ETASCI project. Therefore, the detailed TS approach described above was not used when estimating national emissions with the HBEFA model. Instead, as described in Section 1.1, three TSs were explicitly chosen on the basis of their average speed, so that a direct comparison could be made with emissions estimated by COPERT using the same average speed values for the rural, urban, and motorway activities.

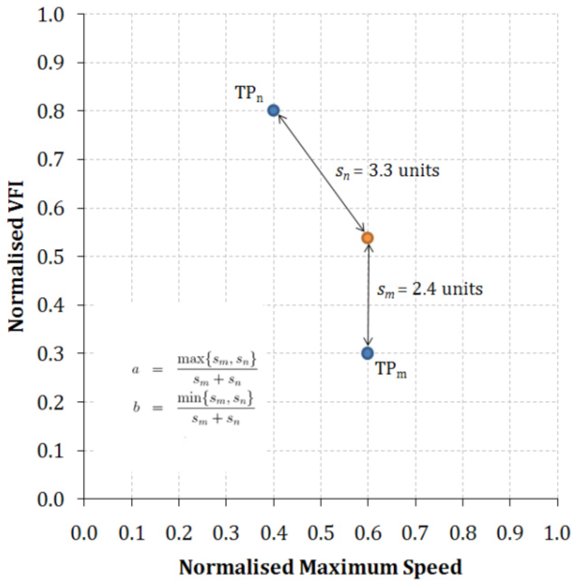

Figure 39 A schematic of the emissions weighting factor scheme proposed by Casey (2011). The yellow dot denotes the coordinates of a measured kinematic segment, the blue dots those of a TS for which the EF is known. The EF calculated for the measured segment is a weighted sum of the EFs for each TS; the weighting is inversely proportional to the distances S_m and S_n: TP_m is given a higher weighting. (Casey 2011)

However, it is worth noting that similar trip-segmentation issues arise even in the context of a simpler, average-speed model, such as COPERT. Because of the "U-shaped" characteristic of most *EFs* vs speed curves (see, for example, Figure 31 or Figure 35), the *EFs* calculated for a two-segment trip, where one segment is at high speed and one at low speed, will be quite different to that calculated for a single composite segment at some intermediate speed. This issue is also raised by Kousoulidou et al. (2013) in Section 5.6.

5.4.3 Application of a TS-based model (ARTEMIS) in Sweden

As part of the ARTEMIS project, an ARTEMIS beta model (version 04d) was used to estimate the total emissions for Swedish road transport, for the period 1990 to 2004. The main aim was to compare the output from the ARTEMIS model to the existing national inventory model. Roadside emissions measurements were also made, using RSD, to validate the ARTEMIS predictions.

Although a comprehensive database of fleet numbers and average vehicle mileage was available, preprocessing of the data was required for many of the datasets for use by the ARTEMIS model. In order to implement the model, a set of TSs relevant to Swedish traffic had also to be identified, and appropriately weighted. Use was made of two existing GIS (Geographical Information System) databases of Swedish roads. The first contains data on all National roads, including data on average daily traffic. The second database contains information on all municipal and private roads, albeit without any information on vehicle flow. For these smaller roads, traffic flows were estimated using existing traffic simulation codes. Urban and rural road types were distinguished by using an additional GIS layer containing an approximate, polygonal representation of built-up areas.

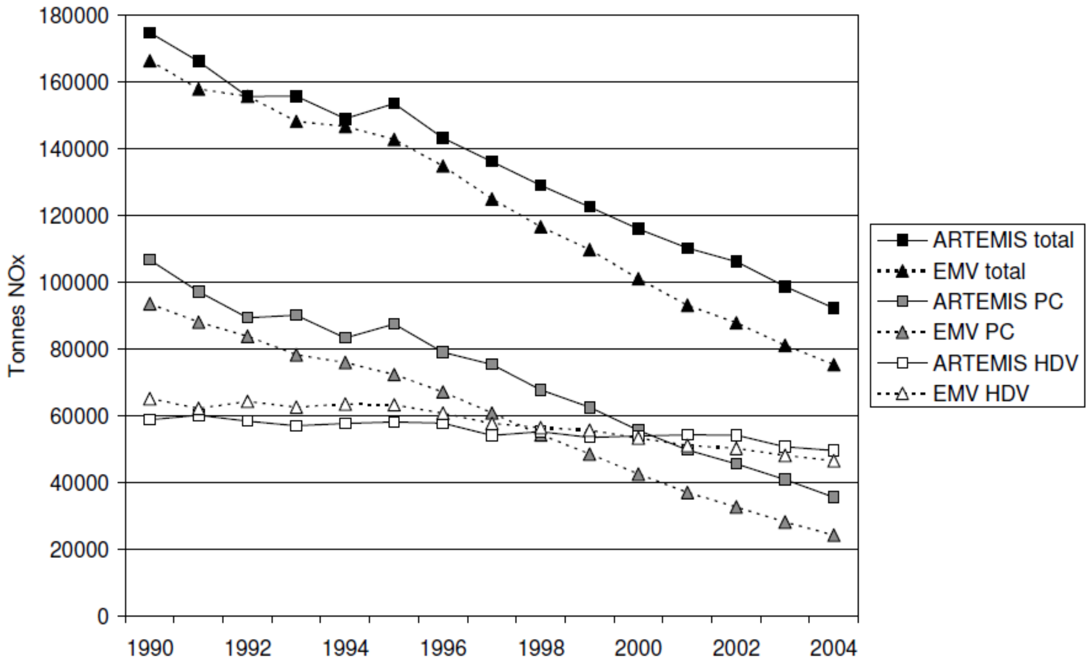

Figure 40 NO_x estimations for Sweden, using both ARTEMIS and the national EMV average speed model.

Following analysis of the traffic flows obtained for the National and Secondary roads, 88 different traffic situations (of a possible 276 available within the ARTEMIS model) were identified as being relevant. The resulting NO_x emission estimates can be seen in Figure 40, where they are also compared to those predicted by the average-speed EMV model. The emission estimates obtained using the TS approach are approximately 10% higher than those obtained with EMV[8], with most of that difference being attributable to differing estimates of emissions from passenger cars (Sjodin et al. 2006).

5.5 Other national inventory models: NAEI and MOVES

HBEFA and COPERT are not the only emission inventory models used in Europe, although they are by far the most widely used. In the UK, an average speed model has been developed as part of the National Atmospheric Emissions Inventory (NAEI). This model is similar in structure to the COPERT model, but has been specifically tailored to the UK vehicle fleet and activity data, and features emissions factors measured from vehicles within the UK. It is notable that the UK devotes considerable resources to monitoring and analysis of activity in the road transport sector, including compilation of an annual National Travel Survey, and development and maintenance of a National Transport Model (NTM).

Motor Vehicle Emission Simulator (MOVES) is the main emissions modelling software in the US, and is the current official model for national inventory estimations. It again is similar to COPERT, and is based on an average speed approach. The vehicle and road structure in the US is considerably different to many other parts of the world and emissions factors specific to

[8] EMV is a PC programme for calculating exhaust emissions from road traffic, used by the Swedish Environmental Protection Agency.

the US have been estimated within this emissions model.

5.6 Real-world and modelled emissions compared

Given the limitations of emission inventory models discussed above, it is of interest to compare the model predictions with emissions observed in practice. There are relatively few reports of such comparisons in the literature, probably due to the difficulty of accurately quantifying real-world driving emissions (RDEs). The advent of (relatively) inexpensive PEMS hardware, allied to increased awareness of discrepancies between certification and real-world emissions, is leading to significant activity in the area, and many more studies are likely to appear over the next few years. It is very unfortunate that the ETASCI project missed the opportunity to be at the forefront of this work due to the funding cuts imposed in 2008.

Kousoulidou et al. (2013) used PEMS to measure on-road (gaseous) emissions from three diesel passenger cars (2 x Euro 5, 1 x Euro 4) and three gasoline cars (Euro 5, Euro 4, and Euro 3). Emissions were measured over two test routes:

- Route 1: Ispra-Milan-Ispra, a trip including a mix of rural and highway driving. Average trip length was approximately 130 km, and average speed about 60 km.h^{-1}.
- Route 2: Ispra-Varese-Ispra, a trip including a mix of urban and rural driving conditions. Average trip length was approximately 63 km, and average speed about 40 km.h^{-1}.

The PEMS *EFs,* measured over “urban”, “rural” and “motorway” sub-cycles identified within each trip, were compared with the COPERT *EFs* for the corresponding average speed. The results, for NO_x only, are shown in Figure 41. It can be seen that the COPERT *EFs* generally lie between the upper and lower bounds of the measured values, but that there is considerable variation between PEMS and COPERT *EFs* for individual vehicles. This difference is particularly strong in the “urban” and “motorway” sub-cycles, and for the Euro 5 diesel vehicles.

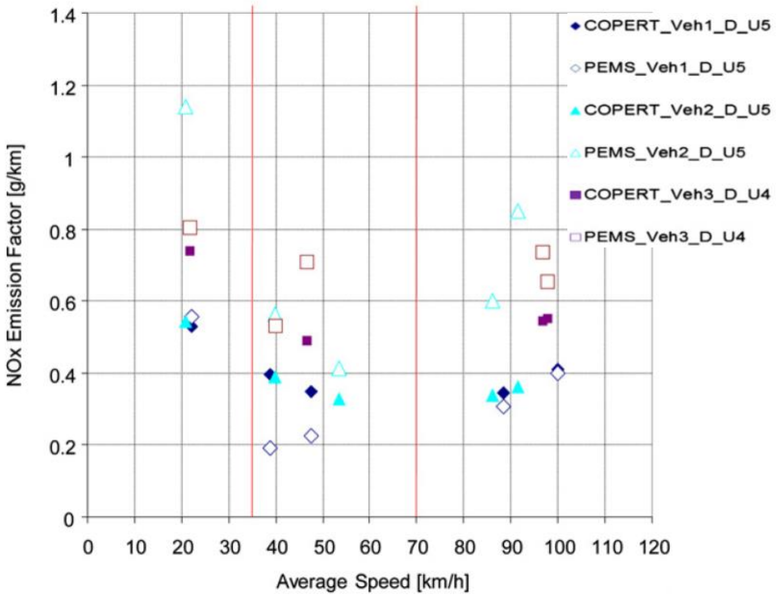

Figure 41 NO_x emission factors: PEMS and COPERT compared. (Kousoulidou et al. 2013)

The issue of trip segmentation is also explored in Kousoulidou et al. (2013), who analysed the PEMS emission data using segment lengths of 1 km, 5 km and 10 km. For each segment, the average speed and the mass of each pollutant emitted was calculated. The results, for one Euro 5 diesel car, are presented in Figure 42.

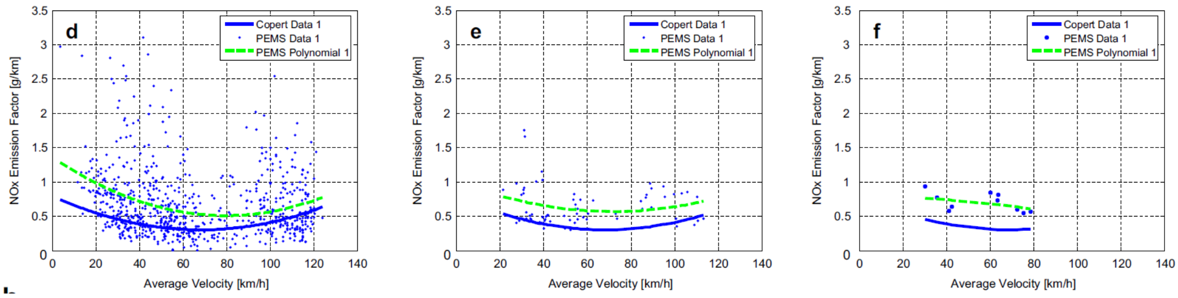

Figure 42 PEMS emission data for (d) 1 km segments, (e) 5 km segments and (f) 10 km segments. (Kousoulidou et al., 2013)

Plot **d** presents the data based on 1 km segments; plot **e** is based on 5 km segments, and plot **f**, on segment lengths of 10 km. The dots denote *EFs* calculated from the PEMS data for each segment; the dashed green line is a polynomial fit to those data points. The solid blue line represents the COPERT *EFs* as a function of average speed. The very large variability of the PEMS *EFs* is evident for short segment lengths, particularly at low and at high speeds. Significant under-estimation by COPERT of NO_x emissions from this vehicle is also apparent. Averaging the PEMS data over longer segments significantly reduces the apparent variability, although even with 5 km segments, the *EF* varies markedly at low speeds.

One of the most interesting studies was carried out by Carslaw et al. (2011), using remote sensing of data (RSD) to measure vehicle emissions at seven urban locations around the UK between 2007 and 2010. A total of 82,469

valid records were analysed. Since RDS measures pollutant concentration relative to CO_2, rather than absolute values, it does not yield *EFs* in a form ($g.km^{-1}$) directly comparable with emission inventory models. However, using CO_2 emission factors for each vehicle type, obtained from the UK NAEI database, it is possible to convert the RDS records from grammes of NO_x per gramme of CO_2, to grammes of NO_x per km. There is obvious potential for error in this process but, as the authors point out, the paper is concerned with how emissions change in a relative way through the different vehicle technologies. These relative changes are not very sensitive to the precise assumptions concerning vehicle speed or other choices affecting the emission estimate.

Once the RDS records had been converted to *EFs*, they were compared with the *EFs* predicted by the UK NAEI emissions inventory model, and with HBEFA v3.1. It was therefore necessary to specify a representative average speed for the NAEI model, and an appropriate TS for HBEFA. The RDS monitoring locations represent urban-type driving conditions, and the mean speed across all campaigns was 31 $km.h^{-1}$. This speed was used directly in the UK emission factor calculations, since the NAEI model (like COPERT) uses vehicle speed as an input. The authors chose 'URB/Trunk-City/50/Satur', where the average speed is 36 $km.h^{-1}$ (and 29 $km.h^{-1}$ for HGVs), as the most appropriate HBEFA TS.

The conclusions of the analysis, for NO_x only, are summarised in Figure 43, where the *EF* derived from RSD records is shown in green, the HBEFA *EF* in blue, and the UK NAEI *EF* in red. Beginning with petrol cars (top-left of Figure 43), it is clear that both modelled and measured *EFs* decrease dramatically with increasing Euro certification. Real-world *EFs* are approximately double those predicted by the models for Euro 0 to Euro 3, but match well for Euro 4 and Euro 5. This may reflect TWC degradation in older vehicles, or the use of "cycle-beating" by manufacturers up to Euro 3.

The comparison of RDE and modelled *EFs* for diesel-powered vehicles is more disconcerting, however. Considering diesel cars and LGVs, the real-world *EFs* appear to have *increased* with increasing Euro level, whereas the modelled *EFs* have decreased strongly from Euro 2 (COPERT) or Euro 3 (HBEFA) onwards. It is interesting to note that the *EF* for Euro 5 cars is the highest observed for any car classification, and exhibits the greatest variability.

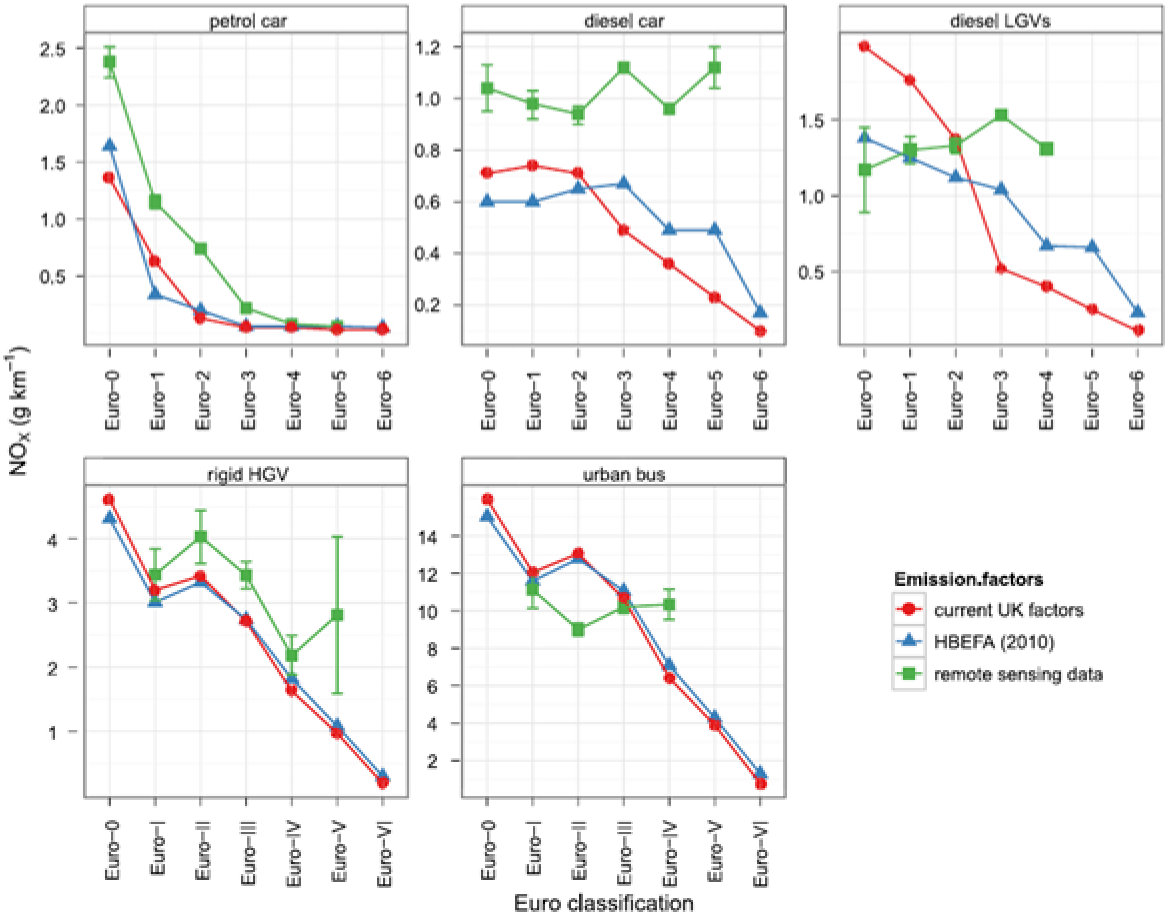

Figure 43 Comparison of different NO_x emissions factors estimates. (Carslaw et al., 2011)

The correlation between observed and modelled *EFs* is rather better when it comes to rigid HGVs. With the exception of Euro V vehicles, the modelled *EF* is very close to that inferred from RDS records. Euro V saw the widespread introduction of SCR on large HGVs for NO_x control which, as discussed in Section 4.2, is often much less effective during urban driving of the type monitored in this paper. However, the smaller, rigid HGVs monitored here generally rely on EGR alone. The authors also note that, due to the urban location used, absolute numbers of rigid HGVs were small, and therefore the inferred *EFs* may not be representative of the type. That may also partly explain the high level of variability in the Euro V *EFs*.

For buses, the agreement between measured and modelled *EFs* is reasonable, although with limited data for Euro IV buses, and none for Euro V or VI, further investigation is merited.

The impact of differences between modelled and observed *EFs,* on the estimate of total emissions, is summarised in Table 3. Total NO_x emissions from road transport in the UK were calculated using three different sets of *EFs*: the standard *EF* used in the UK NAEI model; the *EF* used in HBEFA v3.1; and the *EF* derived from the RDS records. The total emissions estimated in each case were then scaled such that the emissions calculated using the UK NAEI model sum to 100.

Table 3 shows that NO_x emissions based on the RDS records are 78% higher than the estimate obtained using the NAEI model, and 52% higher

than estimated using HBEFA v3.1. Interestingly, although the difference between measured and modelled *EFs* is largest for diesel cars and LGVs, the unexpectedly high proportion of older petrol cars in the observations, coupled with their higher-than-expected *EFs*, makes them responsible for the biggest absolute (and relative) difference between measured and modelled emissions.

Table 3. **Relative NO_x emissions by vehicle type, using three different emission factor estimates. Note that the data have been scaled such that the emission estimates from the UK NAEI model sum to 100. (Carslaw et al 2011)**

Vehicle type	Emission estimate		
	UK	HBEFA	RSD
Diesel car	20.4	26.1	46.4
Petrol car	14.2	15.4	43.4
LGV	20.0	29.8	46.7
HGV + bus	45.4	46.0	41.9
Sum	100.0	117.3	178.3

The work of Carslaw et al. (2011) strongly suggests that the Euro certification process has not been effective in reducing real-world emissions from diesel cars and LGVs. It is also evident that the NO_x *EFs* assumed in current emission inventory models may need significant recalibration, both for diesel LDVs and for older TWC-equipped cars, if they are to accurately reflect real-world emissions. The HBEFA *EFs* are somewhat closer to observations than those employed in the UK NAEI model, but still result in a substantial under-estimation of NO_x emissions from the light-duty, road transport sector.

6 Estimating uncertainty in emission inventories

6.1 Introduction

The primary goal of an emissions inventory programme is to provide an accurate estimate of pollutant emissions from a specific region. However, current emission models contain many sources of error and uncertainty, due to the unpredictable and complex nature of vehicle emissions, as described in Section 1.1.

This chapter begins by outlining uncertainties and errors associated with road transport emissions models in general. It then focuses on the Irish context, explaining how these uncertainties are dealt with in the COPERT and HBEFA models. It concludes with a concrete demonstration of the impact such uncertainties can have on emission estimates, using a sample trip from Dublin to Wicklow, recorded within ETASCI.

6.2 Emissions model error and uncertainty

Accurate quantification of errors associated with an emissions model is inherently challenging – perhaps impossible – because comprehensive validation of the model is not practical. As the authors cannot determine the *actual* level of emissions, errors in model estimates cannot be quantified with certainty.

On the other hand, fundamental physical and chemical considerations constrain the upper and lower bounds of emission for each pollutant and, hence, constrain the *uncertainty* of the emissions estimates. In principle, a more detailed model will reduce that uncertainty, albeit with corresponding increases in model cost and complexity. In practice, the additional requirements imposed on input data by this approach may result in the source of model uncertainty being shifted, rather than eliminated. More detailed models do not necessarily provide more accurate predictions than simple ones (see Figure 44).

This problem is particularly acute when modelling emissions from road transport, because the number of input variables – even for "simple" models – is very large: examples include vehicle numbers, vehicle type, fuel type, average mileage, driving patterns, weather effects, etc. Accurate data pertaining to all of these inputs is simply not available with high granularity, even in highly bureaucratic regimes. It is not surprising, therefore, that a recent study by the Irish EPA found that the uncertainty associated with NO_x emissions from road transport accounts for over 60% of the uncertainty associated with NO_x emissions from all sources in Ireland (EPA, 2011).

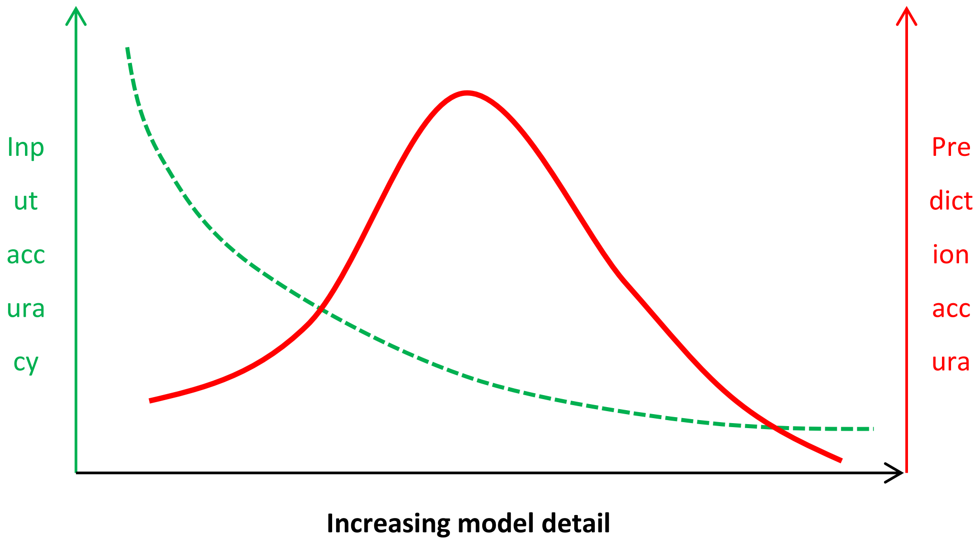

Figure 44 A more detailed model generally requires more detailed input. As input accuracy declines, overall prediction accuracy will also suffer: more detailed models do not necessarily perform better than simple ones.

Given these challenges, model uncertainty is usually estimated using a Monte-Carlo approach. Using this method, the value of each input variable is randomly adjusted within specified limits, and corresponding model outputs are recorded. By repeating this process many times, a range of output values is obtained. The frequency distribution of these output values – e.g. emission quantities – can be interpreted as the probability, or likelihood, of emissions being within a certain range. Although this approach offers useful insight into model sensitivity, it has several limitations when attempting to quantify the uncertainty of emission models for road transport:

- The number of independent inputs to the model is large, so that running a full Monte-Carlo analysis requires the model to be run many thousands – or hundreds of thousands – of times.
- It assumes *a priori* knowledge of the appropriate range, and probability distribution, of values for each input variable.
- It offers no insight into the accuracy or otherwise of the model itself – only its response to changes in the input variables.

Although the first of these can be overcome by allocating sufficient resources to the task, the second and third are of a more fundamental nature, and imply that uncertainty estimates obtained using this approach should be treated with caution.

6.3 Uncertainties related to COPERT

A sensitivity analysis of this type has been carried out on the COPERT 4 emissions model by Emisia SA, on behalf of the European Commission (Kouridis et al. 2009). Emissions estimates for two countries were examined in the Emisia study: Italy, for which a detailed fleet breakdown was available; and Poland, for which

there were significant uncertainties concerning the vehicle fleet.

The primary aim of the study was to assess and compare the uncertainty of model emissions estimates, at national level, for these two countries. A screening technique was used to identify the most influential input parameters. These parameters were then varied using a Monte-Carlo based sampling strategy, and the model outputs recorded: about 6,000 runs of the model were executed for each country. Figure 45 presents the resulting emission estimates for NO_x and PM, in the case of Italy.

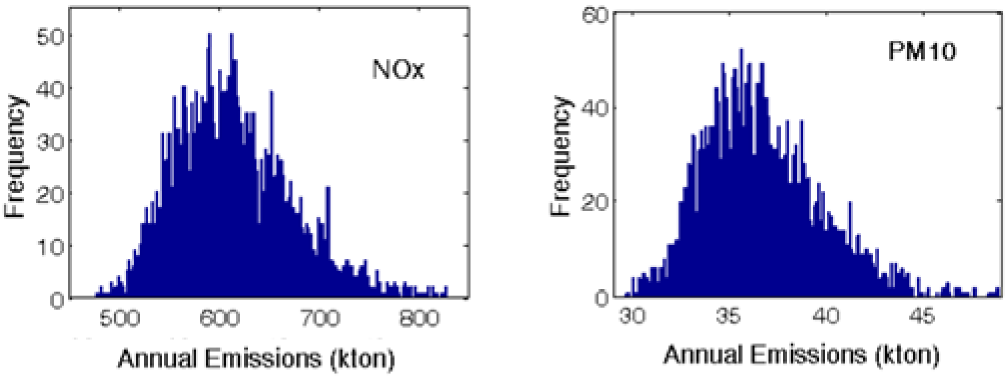

Figure 45 Frequency distribution of NO_x and PM emissions estimates for Italy. (Kouridis et al, 2009)

The overall results from the study are summarised in Table 4. As expected, NO_x and PM emissions display a higher uncertainty (coefficient of variation) than fuel consumption (FC) and CO_2, because the correlation between vehicle average speed and emission rate of PM and NO_x is poor (as was shown in Chapter 5). Limiting the assessment to runs where the calculated fuel consumption was close to the statistical fuel consumption values reported by the countries significantly reduced the calculated uncertainty of the emissions estimates.

Table 4. Summary of coefficients of variation (%) of emissions estimates, for Poland and Italy, using COPERT. Two cases are shown, with and without correction for fuel consumption. (Kouridis et al., 2009)

Case	CO	VOC	CH_4	NO_x	N_2O	$PM_{2.5}$	PM_{10}	PM_{exh}	FC	CO_2	CO_{2e}
Italy w/o FC	30	18	44	15	33	13	13	14	7	7	7
Italy w. FC	19	12	34	10	26	9	8	9	3	4	4
Poland w/o FC	20	18	57	17	28	18	17	19	11	11	12
Poland w. FC	17	15	54	12	24	13	12	14	8	8	8

From the summary statistics (or by observation of Figure 45), we see that 95% of the emission estimates lie within about ± 20% of the mean for NO_x, and within about ± 15% of the mean for PM. In the case of Poland, 95% of emission estimates are within ± 25% of the mean for both NO_x and PM. On this basis, it might be assumed that the uncertainty in NO_x and PM emission estimates produced using COPERT 4 is of the order of ± 25%. That would be incorrect.

The calculations in this study implicitly assume that the emission factors (*EFs*) embedded in COPERT 4 are representative of real-world values. Unfortunately, that is not necessarily the case: as discussed in Chapters 4 and 5, real-world *EFs* for NO_x and PM can be substantially higher than those embedded in COPERT 4.

The Emisia study also contained the following assumptions and omissions:

- Log-normal distribution of parameter values about the observed mean is assumed.
- Uncertainty estimates of EURO 5 and 6 petrol, and EURO 4 diesel PCs are not included.
- Uncertainty of Light Commercial Vehicle (LCV) EFs is not included.
- No uncertainty estimates for climate corrections for NO_x or PM.
- The fact that uncertainty in EFs for LCVs was not included in the Emisia study is particularly significant since, as will be shown in Sections 7.2 and 7.4, LCVs contribute a disproportionate fraction of total fleet emissions.

6.3.1 *Summary of COPERT limitations*

The net effect of these assumptions and simplifications is that the Emisia study under-states the overall uncertainty associated with COPERT 4 emissions estimates. The study itself acknowledges as much: "It should be made clear that the current study does not address neither conceptualisation nor model uncertainty" (Kouridis et al., 2009). The true uncertainty in the COPERT emissions estimates for PM and NO_x is therefore likely to be greater than ± 25%.

The additional uncertainty derives primarily from the fact that COPERT relies on a detailed fleet model but very simplistic activity data. Hence, the *EFs* associated with vehicle activity may not reflect real-world values. It is primarily this shortcoming that the ARTEMIS/HBEFA model seeks to overcome, by defining *EFs* for a far wider range of vehicle activity types.

6.4 Uncertainties related to ARTEMIS / HBEFA

The primary difference between COPERT and ARTEMIS/HBEFA is that the latter allows for a far more detailed and comprehensive description of vehicle activity data. Whereas COPERT relates emissions only to vehicle average speed (V_{avg}), ARTEMIS/HBEFA provides *EFs* for a range of statistically-representative trip segments - Representative Test Patterns (RTPs). Since NO_x and PM emissions, in particular, show poor correlation with V_{avg}, ARTEMIS/HBEFA should provide significantly better real-world estimates of these emissions.

The main difficulty with this approach however – and a major source of uncertainty – is choosing which RTPs most accurately represent drive cycles in a particular region. Since the *EFs* are specific to each RTP, the choice and weighting of segments used will determine the overall emissions estimated by the model.

This is of particular significance in Ireland, since information on Irish drive cycles is essentially non-existent. Although traffic counters are available in some urban areas (*e.g.* the Sydney Coordinated Adaptive Traffic Systems (SCATS) system in Dublin) and on some national roads (via the National Roads Authority (NRA)), they provide only vehicle counts at 15-minute or one-hour intervals, but include no speed or activity data, and very little disaggregation of vehicle type. Collation of a coherent data set from these disparate systems would require substantial resources; using that data to infer representative drive cycles is probably not realistic. A national transport model (NTM) might be useful in this respect, as used for instance in the Swedish study described in Section 5.4.3. At least one feasibility study has been carried out on development of an NTM for Ireland (WSP Group, 2011), but no such model yet exists.

In the absence of drive-cycle data, selection of RTPs could be based on examination of the road types, speed limits, and any relevant traffic data available within that region. The obvious problem with this approach is that selection of the most appropriate RTP is still a matter of personal judgement; empirical support for that judgement is simply unavailable.

Since the main hurdle to implementing the ARTEMIS/HBEFA model seems to be the absence of Irish drive-cycle data, it was decided to collect some of this data and determine its usefulness. A small number of private cars were equipped with a low-cost global positioning system (GPS) and on-board diagnostics (OBD) loggers developed as part of the ETASCI project, and their daily trips recorded. Several parameters such as vehicle speed, location, and trip start/end times were measured, at one-second intervals, during routine vehicle use. The question being examined was this: given accurate and detailed information on a vehicle's drive cycle, would it be possible to extract reliable emission estimates for that cycle from HBEFA?

6.4.1 Kinematic segmentation of recorded trip – micro study

In order to assess the potential of this approach, detailed analysis was performed on a sample trip. The route chosen – along the N11 from Dublin to Wicklow – involves a mixture of rural, motorway, and urban driving, and is representative of many commutes in the Greater Dublin Area. Figure 46 shows (red line) the actual route taken. **Table** 5 summarises the overall trip characteristics, and Figure 47 presents second-by-second vehicle speed data.

In order to obtain emission estimates for the trip, the *measured* drive cycle must be compared to one or more cycles for which HBEFA has stored emission factors (*EFs*). Since no two drive cycles are identical, it is necessary to characterise each cycle using one or more parameters, e.g. average speed, speed range, relative positive acceleration (RPA), etc. Following an extensive analysis of the relevant literature, and building on the EPA-funded work carried out by Casey (2011) in the Urban Environment Project, it was decided to characterise each cycle using average speed (V_{avg}) and RPA.

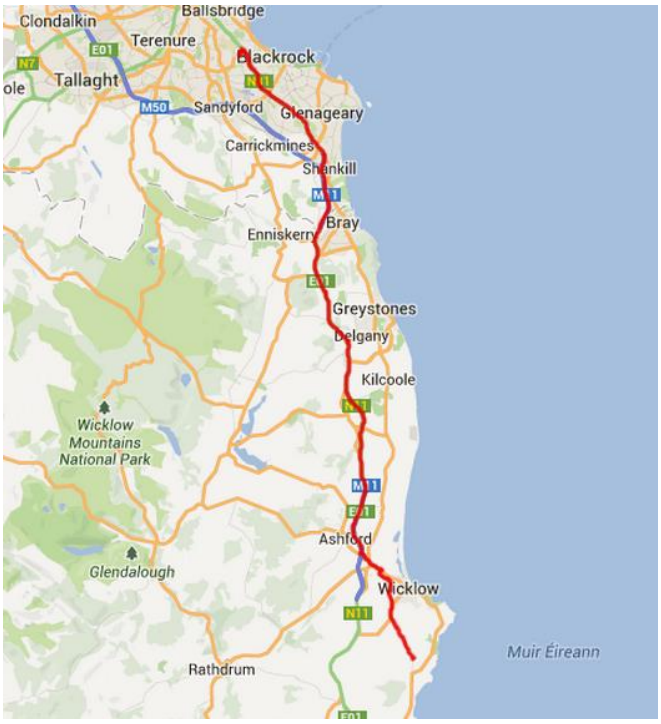

Figure 46 GPS trace of the N11 trip used in the Micro-study. The trip start point is near Wicklow town, with the destination of Stillorgan in south Dublin.

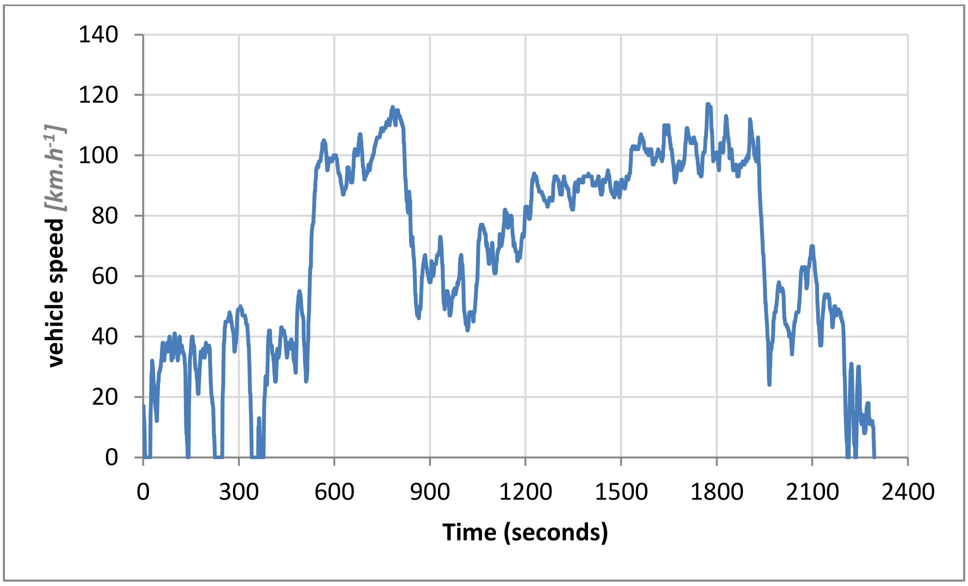

Figure 47 Time speed trace used for the N11 micro-study. The trip begins in an urban environment and, following some motorway driving, concludes in a rural environment.

Table 5 Test route used to study the impact on emissions estimates of trip segmentation strategy.

N11 Trip	
Total Distance	43 km
Average speed	67.5 km.h^{-1}

Although HBEFA contains *kinematic* data for 276 Traffic Situations, comprehensive *emissions* data was available only for the fifteen Representative Test Patterns (RTPs) when the ETASCI analysis was carried out. The characteristic V_{avg} and RPA for these RTPs were therefore calculated. Emissions from the measured drive cycle were then estimated using a weighted sum of the two RTPs with the most similar values of V_{avg} and RPA, as described in Section 5.4.2 and in Grummell (2013).

The final step in the process – and the one of interest here – involves the choice of a trip segmentation strategy. Should the entire trip be characterised using a single average speed and RPA, or – given that the trip contains rural, motorway, and urban elements – should it be broken into shorter segments, and each segment be individually characterised? If so, how many segments should be used, and how should the segment boundaries be selected? In other words, what segmentation strategy should be used, and what impact does the choice of strategy have on the emissions estimated by HBEFA for the trip?

To explore this issue, five different segmentation strategies were examined:

- No segmentation – the entire trip is characterised by a single V_{avg} and a single RPA.
- Kinematic segmentation – segment boundaries are based on a pre-defined change in speed, or some similar event.
- Constant interval segmentation – the trip is segmented every *n* kilometres. Two interval lengths were examined: 5 km, and 1 km.
- Manual segmentation – for this trip, three segments were considered:
    - An urban segment, in the Dublin area.
    - A motorway segment, along the 120 and 100 km.h^{-1} speed limit regions of the M11/N11.
    - A rural segment, after the vehicle exits the M11.

6.4.2 Results of the kinematic micro-study

The results of the N11 micro-study are summarised in Figure 48 and in Table 6. It is clear that segmentation strategy exerts a significant influence on the quantity of NO_x and PM emissions estimated using HBEFA. Importantly, it is also clear that CO_2 emissions are relatively *insensitive* to the choice of segmentation strategy; in other words, each strategy is equally likely to satisfy the requirement to match statistical fuel consumption data.

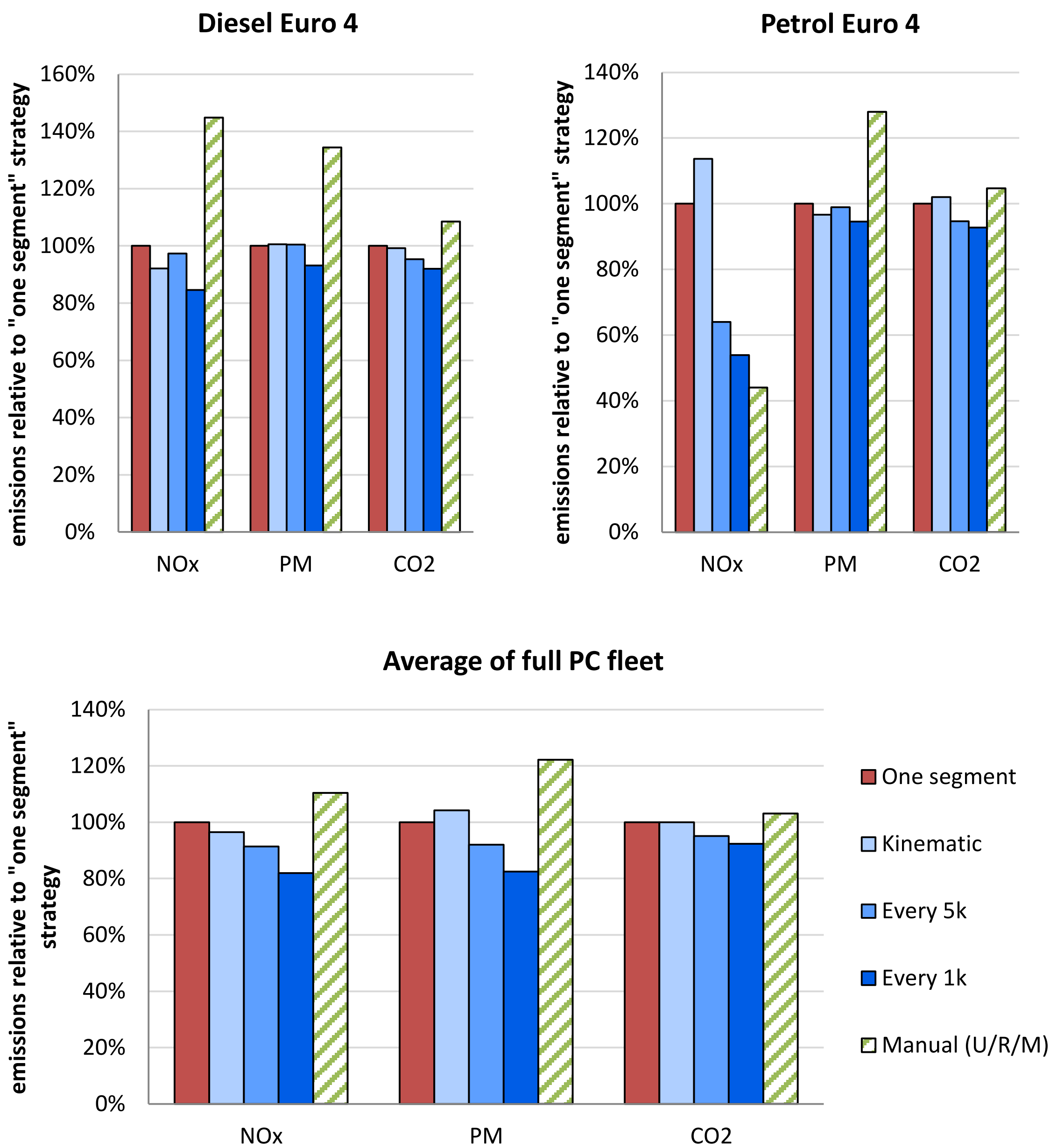

Figure 48 Normalised emissions estimates for the N11 micro-study, as a function of segmentation strategy. Data presented for a Euro 4 diesel, a Euro 4 petrol, and for the full Irish passenger car fleet. Estimates are normalised with respect to the “one segment” strategy (i.e. no segmentation). U/R/M denotes urban / rural / motorway.

Table 6. Sensitivity of HBEFA emissions estimates to choice of segmentation strategy.

	NO_x	PM	CO_2
Euro 4 Diesel	± 30%	± 21%	± 8%
Euro 4 Petrol	± 35%	± 17%	± 6%
Irish PC fleet	± 14%	± 20%	± 5%

6.4.3 Summary of ARTEMIS/HBEFA limitations

The ARTEMIS/HBEFA models set out to circumvent the principal limitation associated with COPERT: its use of vehicle average speed (V_{avg}) as the sole predictor of emissions under normal operating conditions. Instead, ARTEMIS/HBEFA bases its predictions on emissions measured across a range of drive cycles that are statistically representative of real-world conditions. The limitation with this approach – and it's a big one – is that accurate predictions can be attained only if:

a) the characteristics and frequency distribution of real-world drive cycles in the region are known, and

b) these known drive cycles can be accurately, and unambiguously, mapped to drive cycles stored in the model.

Taken individually, these are stringent demands; in combination, they are almost impossible to meet. However, as seen from the micro-study of the N11 trip above, if only one demand can be met, then the uncertainty in emissions prediction remains high.

6.5 Chapter summary

Road transport emission models can never be comprehensively validated, since the *actual* real-world emissions from the fleet cannot be directly measured. However, fundamental physical and chemical considerations constrain their upper and lower bounds, and hence constrain the *uncertainty* of the emissions estimates.

Simple models – such as COPERT – embed very little physics or chemistry, relying instead on statistical correlations of variable quality.

In principle, a more detailed model – such as ARTEMIS or HBEFA – will reduce that uncertainty, albeit with corresponding increases in model cost and complexity. In practice, the additional requirements imposed on input data by this approach may result in the source of model uncertainty being shifted, rather than eliminated.

7 Irish study

The EPA currently uses COPERT 4 to estimate emissions from road transport. This chapter presents and compares results from both COPERT 4 and ARTEMIS/HBEFA v3.5.

The chapter is divided into four main sections. Section 7.1 focuses on the model runs performed using COPERT 4. It begins with a summary of the main inputs used, and presents the corresponding emissions estimates for 2009. This is followed by a look at the sensitivity of those estimates to the uncertainty in some of the inputs: specifically vehicle average speed; the allocation of vkm between motorway, rural, and urban driving; and uncertainties in the fleet composition.

Section 7.2 focuses on model runs performed using HBEFA v3.5. The methodology used to select Traffic Situations (TSs) relevant to Ireland is described, followed by emissions estimates for 2009. Section 7.3 compares the results from the two models for 2009.

Section 7.4 presents projections of future emissions (up to 2020) using HBEFA v3.5. It also examines the potential impact on future NO_x emissions, and on NO_x *EFs* for Euro 5 and Euro 6 vehicles decreasing less rapidly than initially anticipated. This phenomenon has been widely reported in the literature, and has been discussed in Section 5.6.

7.1 COPERT study

7.1.1 Selection of COPERT baseline data

Composition and activity data for the Irish fleet have been published by the Irish EPA (EPA 2011b) for the years 2000 – 2009, and was used for initial COPERT studies within this project. In addition, 2005 data for the urban, rural and motorway percentage share, and average speeds across these three road types for Ireland, have been obtained from the Emisia SA website [9]. However, there is little direct measurement of traffic volume and vehicle average speed in Ireland, so the quality of the Emisia data (summarised in Table 7) is in some doubt.

[9] Website:http://www.emisia.com/tools/FLEETS.html

Table 7. Baseline allocation of vkm and average speed, for each vehicle class (COPERT 4)

	Share of vkm				Average Speed (km.h^{-1})		
	motorway	**rural**	**urban**		**motorway**	**rural**	**urban**
PC	20%	50%	30%		100	60	30
LGV	20%	30%	50%		90	60	25
HGV	60%	30%	10%		90	50	20
Urban Bus			100%		90	50	20
Coach	40%	40%	20%		90	50	20
Motorcycle	15%	55%	60%		90	60	20

7.1.2 Results of COPERT baseline runs

Using this data, the COPERT model was run for 2008, 2009, and 2010.

The resulting emission estimates for 2008, broken down by vehicle category, are shown in Figure 49.

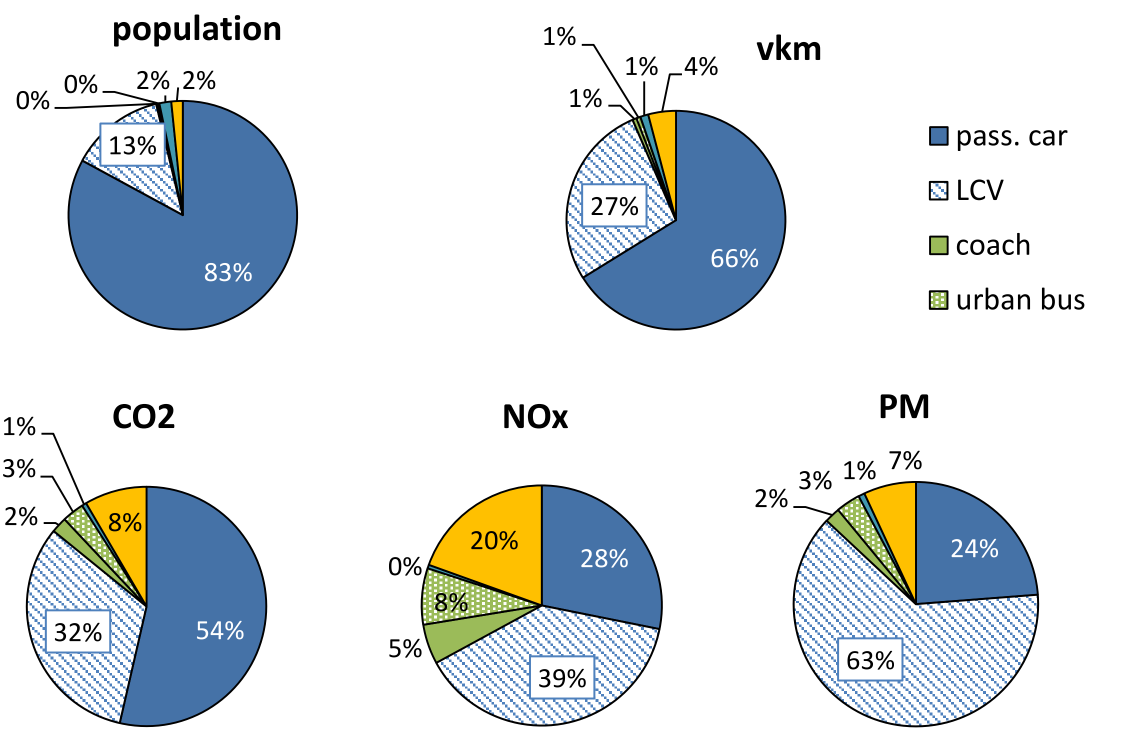

Figure 49 Distribution of the COPERT estimates for 2008 Irish road transport emissions.

The standout result of this preliminary analysis is the large fraction (63%) of the total PM emissions attributed to LGVs, which represent only 13% of the Irish fleet. This is due to their high annual mileage (approximately 70,000 km per year), and the high *EFs* associated with LGVs. A similar, but less pronounced, trend can be observed with the NO_x estimate.

As previously noted, measurements of traffic volume and vehicle average speed in Ireland are sparse, so considerable uncertainty surrounds the values employed in these COPERT runs. Additional COPERT runs were therefore performed, to provide data on model sensitivity to these parameters.

7.1.3 Sensitivity of COPERT emission estimates to assumed average speeds

Three tests on average speed were carried out. First, average speeds for all vehicle types were *decreased* by 20%. Second, average speeds were *increased* by 20%. In the third test, the average speed on urban roads was reduced by 50% (representing increased congestion), and average speeds on motorways were increased by 10% (requiring increased engine power to overcome larger aerodynamic losses). This third test, therefore, might be considered a “worst case” scenario for PM and NO_x emissions.

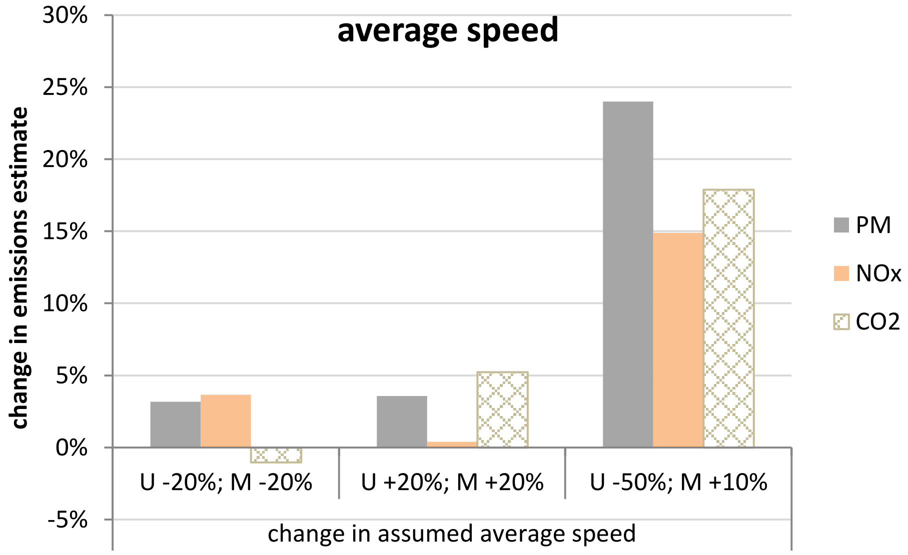

Figure 50 Sensitivity of the total emissions estimate to the average speed assumed for urban (U) and motorway (M) travel.

Figure 50 summarises the results of this study. As can be seen, a blanket increase or decrease of 20% in average speed produces no significant change in COPERT’s emission estimates for the pollutants in question. This is somewhat surprising, since the effect of these changes on journey times would be very noticeable. The third test (representing a “worst case” emissions scenario) produced more significant changes, although still somewhat below what might be expected in a real-world situation.

7.1.4 Sensitivity of COPERT emission estimates to road type

A further three tests were performed to assess COPERT sensitivity to the share of vkm allocated to motorway, rural, and urban driving. Very limited data is available within Ireland on the share of traffic across the three road types; simply disaggregating the Irish road network into urban, rural or motorway road types presents its own challenges. It is therefore reasonable to try and assess the sensitivity of COPERT to this input.

Three test runs were performed. In the first, the Motorway share of vkm for PCs and LGVs was *reduced* from 20% to 10%; the Rural and Urban shares were each increased by five percentage points to compensate (see Table 8). In the second, the Motorway share of vkm for PCs and LGVs was *increased* from 20% to 30%; the Rural and Urban shares were each *reduced* by five percentage points to compensate (see Table 8). The third scenario denotes a very strong shift to Motorway travel: the Motorway share of vkm for PCs and LGVs was increased from 20% to 60%, and the Motorway share of vkm for HGVs was increased from 60% to 80% (see Table 8).

Table 8. Allocation of vkm between Motorway, Rural, and Urban driving respectively, for the COPERT sensitivity study. Red values are changed from the baseline.

	Baseline			M =10%			M =30%			M =60%		
	M'way	Rural	Urban	M	R	U	M	R	U	M	R	U
PC	20%	50%	30%	10%	55%	35%	30%	45%	25%	60%	20%	20%
LDV	20%	30%	50%	10%	35%	55%	30%	25%	45%	60%	20%	20%
HGV	60%	30%	10%	60%	30%	10%	60%	30%	10%	80%	15%	5%

Figure 51 shows the impacts of these changes on the emissions estimated by COPERT. Clearly, the impacts are small: emission estimates, for all three pollutants considered here, change by less than 10% and, in most cases, by less than 5%. These results suggest that COPERT's emission estimates are relatively insensitive to the manner in which vkm are distributed between road types. Whilst this is convenient from the end-user's perspective – reducing the need for accurate input data – it lends support to the assertion that the emissions correlations used by COPERT may be over-simplified.

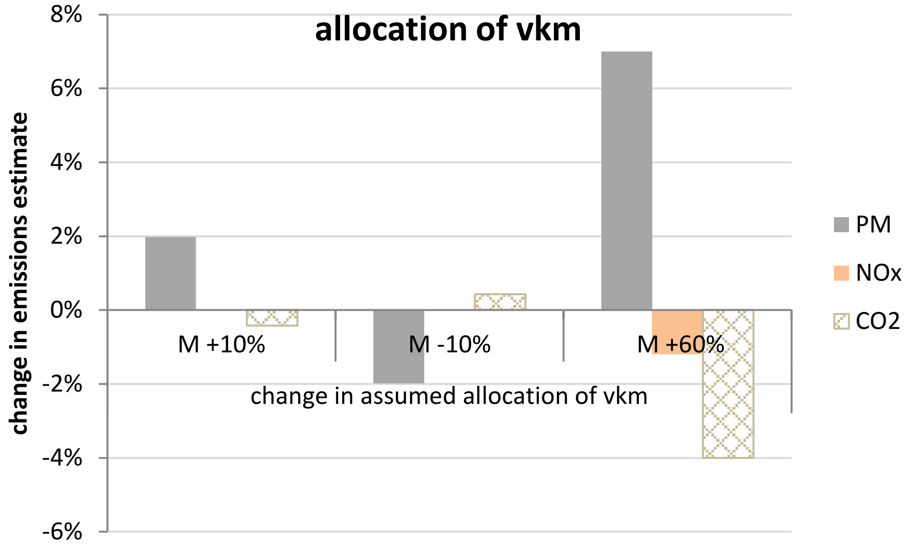

Figure 51 Sensitivity of the COPERT emissions estimate to the assumed allocation of urban (U), rural, and motorway (M) vkm.

7.1.5 Sensitivity of COPERT emission estimates to fleet composition

As noted in Section 7.1.1, the transport fleet presented in EPA (2011) was used to generate emission estimates for the tests discussed above. Unfortunately, whereas COPERT classifies vehicles by Euro standard, HBEFA classifies by age (see 0). Hence, the EPA COPERT fleet could not be used to run the HBEFA simulations.

Instead, data for all registered vehicles from the years 2006 to 2010 was obtained from the Department of Transport, Tourism and Sport (DTTAS). After some minor adjustments, this data was used to provide fleet information for the HBEFA runs. It was noted, however, that the composition of the DTTAS fleet differed in some respects from that used by the EPA, as shown for the 2009 fleets in Figure 52. These discrepancies may be attributable to omission of some minor vehicle classes, such as exempt vehicles, etc.

Although the differences between the two fleets appear minor, it was decided that the COPERT model would be re-run with a fleet that corresponded to the DTTAS data. The results of this test are presented in Section 1.1.

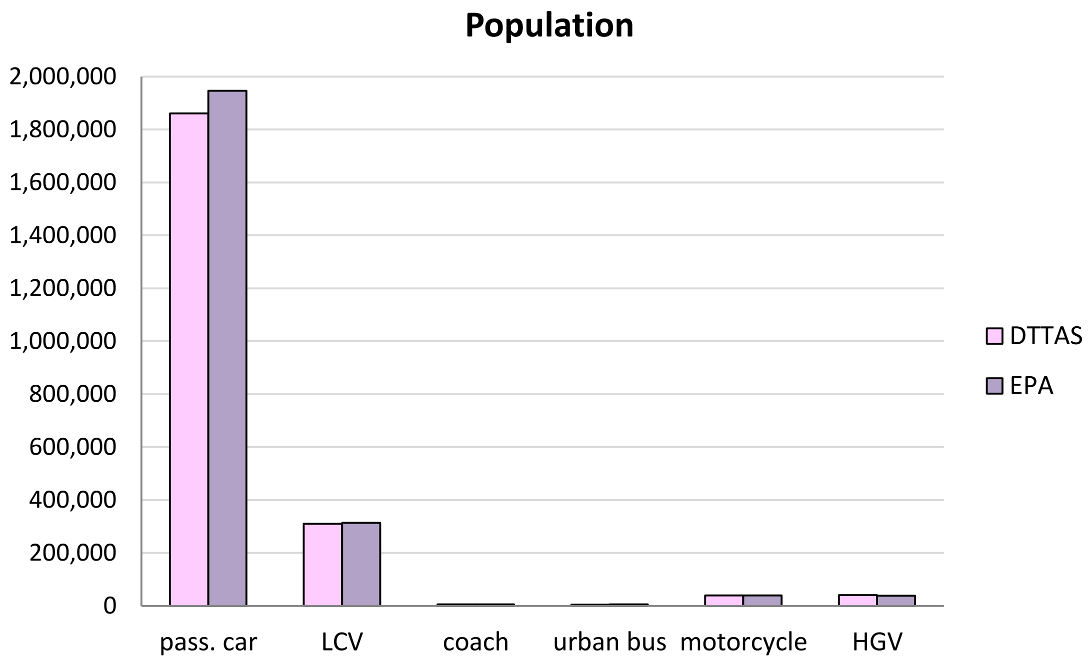

Figure 52 Irish fleet 2009, based on DTTAS and EPA data sets.

7.2 HBEFA Ireland study

7.2.1 Selection of Irish Traffic Situations for use with HBEFA

COPERT and HBEFA differ mainly in their descriptions of traffic activity. Whereas COPERT relies on estimates of vehicle average speed, HBEFA relates vehicle emissions to so-called Traffic Situations (TSs). Whilst the HBEFA approach has the *potential* to be much more accurate, that potential can be realised only if the TSs used by the model match real-world traffic activity.

Unfortunately, such activity information is difficult to collect, and not available for Ireland. The approach adopted in the ETASCI project was to identify a set of TSs, for each vehicle category, that aligns with the activity types employed in COPERT. Where more than one TS aligned with a particular COPERT activity, the TS whose average speed was closest to that specified in COPERT was used. These TSs are listed in Table 9.

Using these TSs, and a transport fleet derived from the DTTAS data, emissions estimates were generated for the each of the years 2008 – 2010. These results for 2009 are summarised in Section 1.1.1

Table 9 Traffic Situations (TSs) selected for the Irish HBEFA study. Average speed of each TS is similar to that used for the corresponding activity in COPERT.

	Motorway	**Rural**	**Urban**
PC	Rural-MW-100-Freeflow	Rural-Trunk-70-Freeflow	Urban-Local-50-Saturated
LGV	Rural-MW-90-Freeflow	Rural-Local-60-Freeflow	Urban-Local-50-Saturated
HGV	Rural-Trunk-90-Freeflow	Rural-Local-50-Freeflow	Urban-District-30-Freeflow
Urban Bus	Rural-Trunk-60-Freeflow	Rural-Local-50-Freeflow	Urban-Access-30-Freeflow
Coach	Rural-Trunk-100-Freeflow	Rural-Local-60-Freeflow	Urban-Access-30-Freeflow
Motorcycle	Rural-MW-100-Freeflow	Rural-Local-60-Freeflow	Urban-Local-50-Saturated

7.2.2 Results from the HBEFA model

Figure 53 summarises the HBEFA emission estimates for Ireland for 2008. The distribution of emissions between the vehicle categories is broadly similar to that predicted using COPERT (see Figure 49), with LCVs again contributing a disproportionate fraction of NO_x and PM emissions in particular. The substantial contribution of HGVs, coaches, and urban buses to calculated NO_x emissions is also apparent.

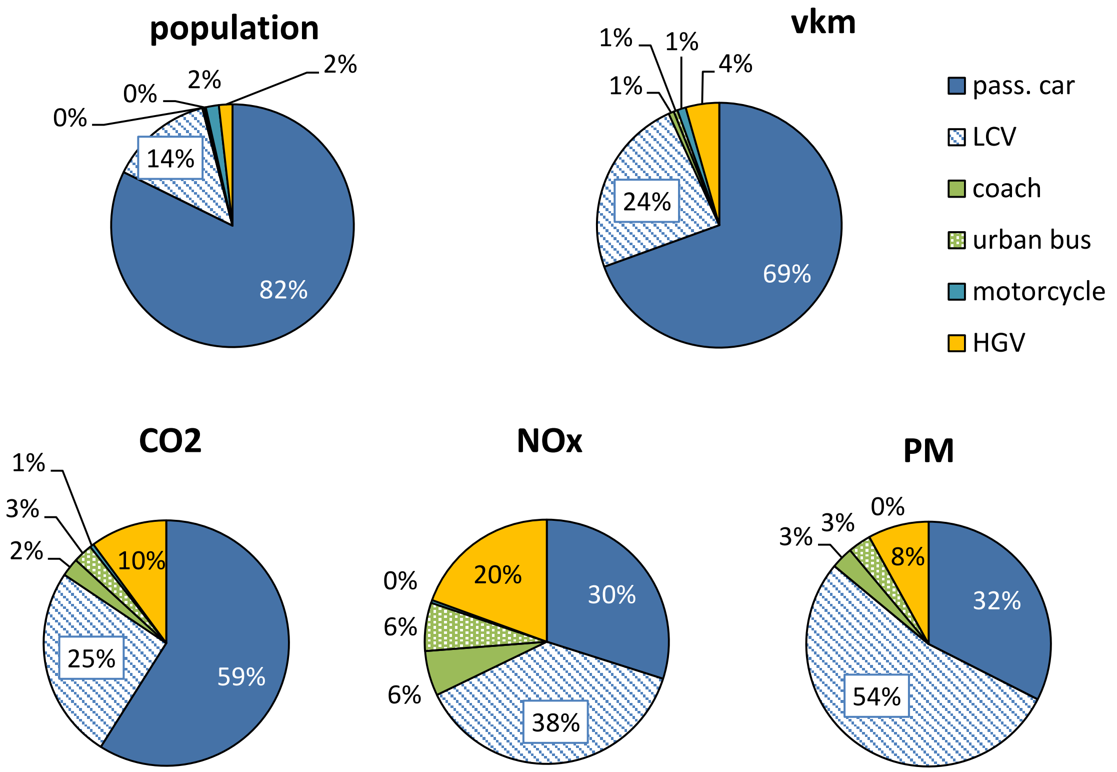

Figure 53 Distribution of the HBEFA estimates for 2008 Irish road transport emissions.

7.3 Comparison of emissions estimates from HBEFA and COPERT

Road transport emission estimates from both COPERT 4 and HBEFA v3.5 have been presented in this chapter. The COPERT estimates were produced using the EPA fleet model; the HBEFA estimates using the DTTAS fleet data. To allow direct comparison of emission estimates from the two models, a third simulation was performed using the DTTAS fleet with COPERT.

Emission estimates for 2008 (NO_x, PM and CO_2), from each of the three model runs, are presented in Figure 54 and in Table 10. The following conclusions can be drawn from this study:

- Emission estimates are more sensitive to changes in fleet composition than to the choice of model used. Since the differences between the DTTAS and EPA fleets were relatively small, this suggests that maintaining an accurate fleet model is very important.
- Overall NO_x emission estimates from both models are very similar when using the DTTAS fleet, although the distribution of emissions amongst vehicle categories shows small differences.
- More significant difference between models are apparent for PM (about 7%) and, surprisingly, for CO_2 (about 5%).

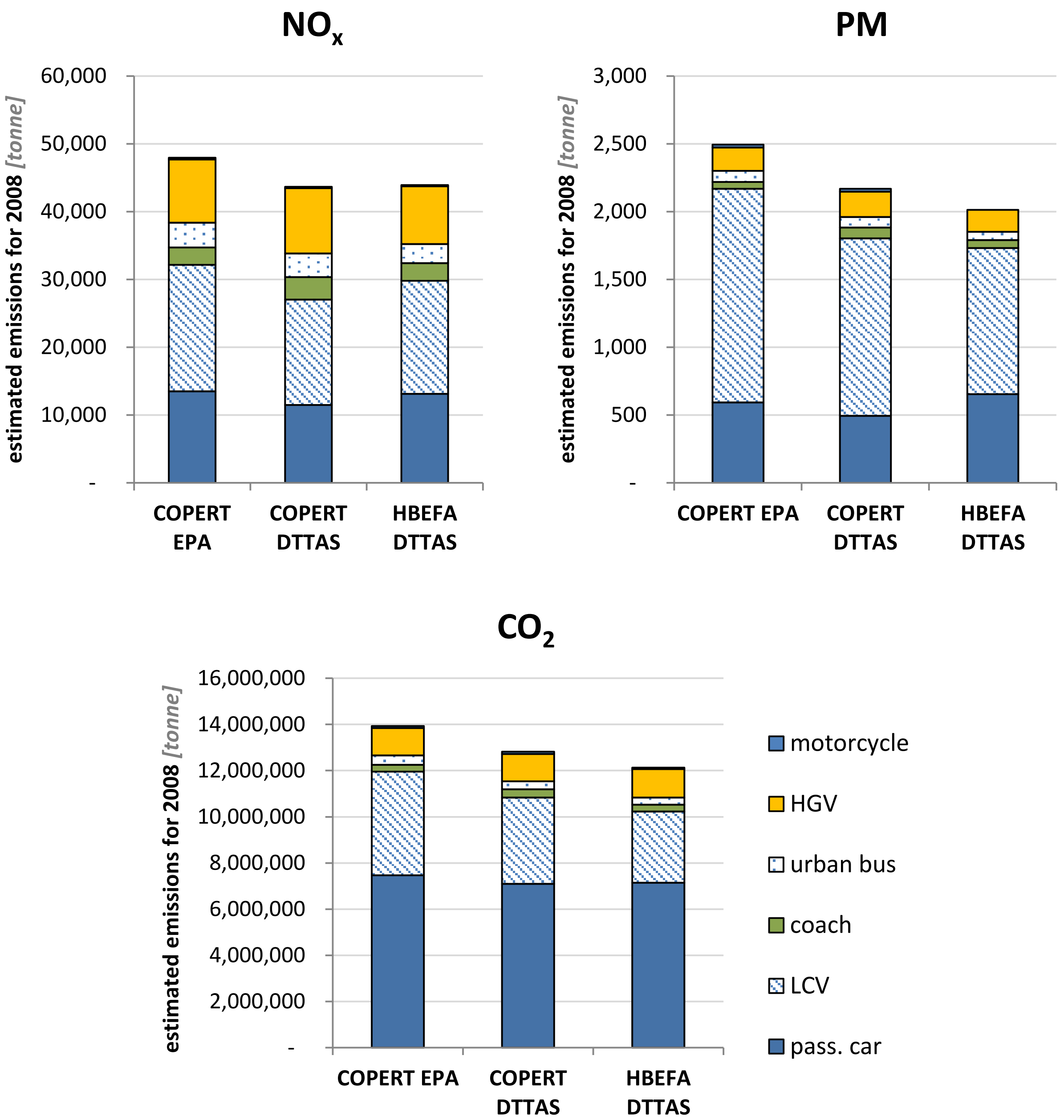

Figure 54 Emission estimates for 2008, using HBEFA and COPERT. HBEFA estimates are based on the fleet composition provided by DTTAS. Two COPERT estimates are presented: one based on the DTTAS fleet, and one based on the fleet composition used by EPA.

Table 10. 2008 emission estimates compared.

	PC	LCV	Coach	Urban Bus	Motor-cycle	HGV	Total	% of COPERT EPA
NO_x [tonnes]:								
COPERT EPA	13,499	18,661	2,568	3,636	232	9,350	47,946	100%
COPERT DTTAS	11,469	15,574	3,321	3,475	201	9,626	43,666	91%
HBEFA DTTAS	13,132	16,654	2,615	2,805	166	8,542	43,914	92%
PM [tonnes]:								
COPERT EPA	593	1,576	51	80	21	173	2,494	100%
COPERT DTTAS	495	1,309	78	80	22	186	2,169	87%
HBEFA DTTAS	653	1,078	58	61	-	162	2,013	81%
CO_2 [million tonnes]:								
COPERT EPA	7.46	4.49	0.30	0.41	0.09	1.17	13.93	100%
COPERT DTTAS	7.10	3.74	0.35	0.35	0.09	1.19	12.82	92%
HBEFA DTTAS	7.15	3.08	0.29	0.31	0.06	1.23	12.13	87%

7.4 Projections of emissions in 2020

Following successful simulation of emissions estimates for 2008-2010, a further study was undertaken, to project the evolution of road transport emissions out to 2020.

The primary objective of the study was to examine the impact of tightening emissions standards on future emissions from road transport. The size, composition and activity of the transport fleet were therefore “frozen” at 2010 values. Replacement vehicles are introduced at whatever rate is necessary to

maintain a constant age profile in the fleet. In practice, this implies an average PC replacement rate of 135,000 per year between 2010 and 2020 – significantly above the average of 83,000 actually recorded between 2010 and 2013. It also results in the number of diesel-powered PCs remaining constant at 480,000. Hence, <u>the projections presented here are not forecasts.</u>

The evolution of the PC fleet composition in this scenario is shown in Figure 55. New PCs are assumed to be Euro 5 compliant if introduced between 2010 and 2014, and Euro 6 compliant after that. On this basis, Euro 5 and Euro 6 cars will constitute 80% of the PC fleet in 2020.

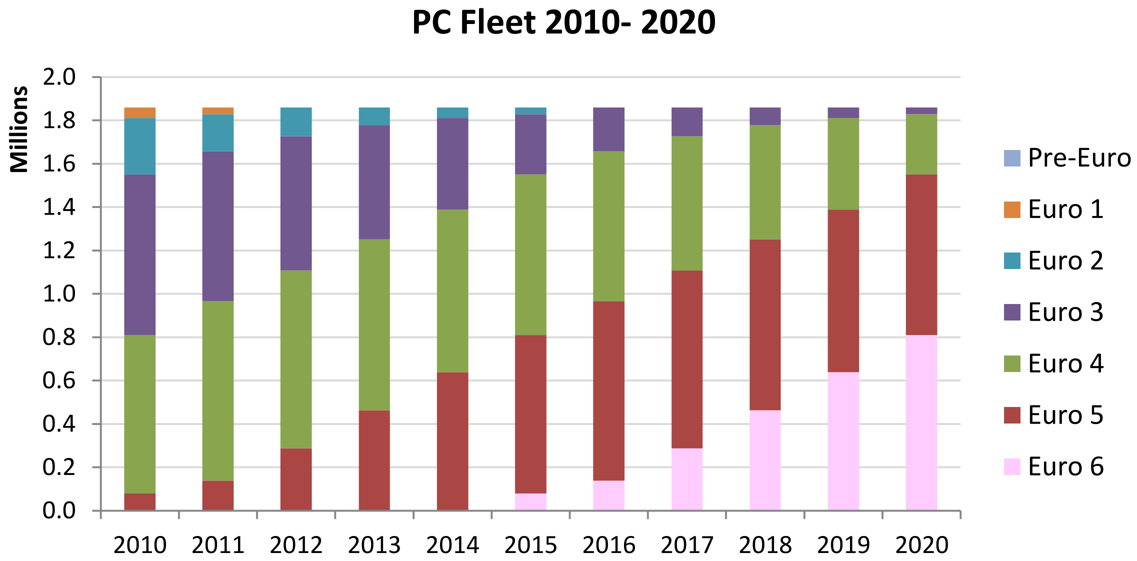

Figure 55 Projection of Irish PC population evolution from 2010 to 2020.

7.4.1 HBEFA projection results

The projected emissions estimated with HBEFA in this scenario are presented in Figure 56. As can be seen, the projected reductions in overall NO_x and PM emissions are substantial. By 2020, NO_x emissions in this scenario are projected to decrease by 57% of their 2010 level. Although national NO_x emissions increased in 2012, NO_x emissions from transport decreased from 38 kt in 2010, to 36 kt in 2012. The increase in national NO_x emissions is associated with increased use of Moneypoint for generation, and an increase in NO_x emissions from the industrial sector. Projected reductions in PM emissions are even more dramatic, decreasing by 84% during the same period, driven by the effectiveness of diesel particulate filters (DPFs) fitted to Euro 5 and Euro 6 light-duty vehicles. The projected reduction in CO_2 emissions, on the other hand, is a relatively modest 13%.

The projected CO_2 reductions are associated mainly with PCs and, to a lesser extent, LCVs. However, the CO_2 *EFs* used by HBEFA may under-estimate the strong and rapid decrease in CO_2-intensity of new PCs, driven by the 95 $g.km^{-1}$ fleet-average limit to be imposed on manufacturers by 2021. It is quite likely, in the authors' opinion, that CO_2 *EFs* for Euro 5 and Euro 6 PC will continue to evolve. The enabling technology will also be shared with small LCVs, leading to further reductions in CO_2 emissions from that source.

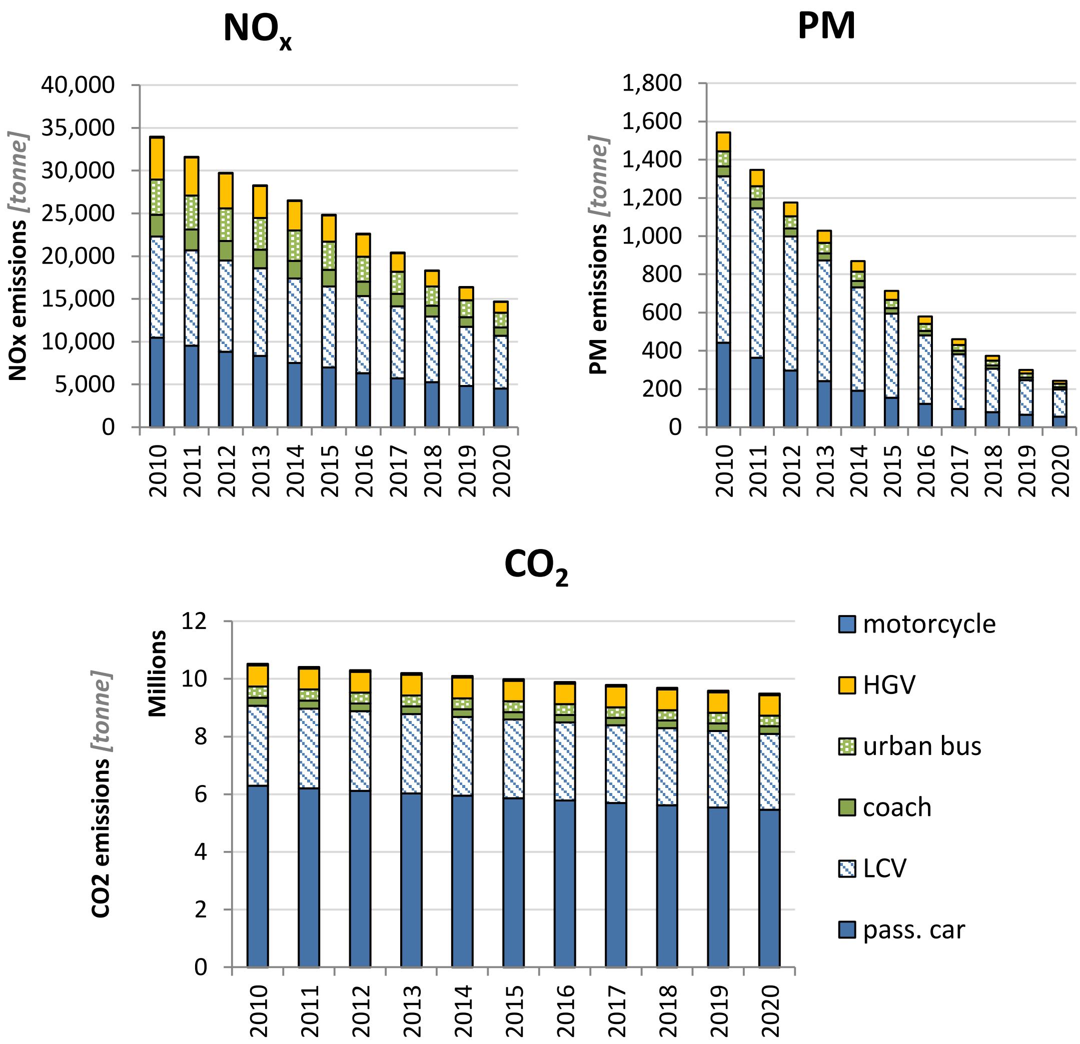

Figure 56 NO_x, PM and CO_2 emission projections for road transport, from 2010 to 2020, using HBEFA. A significant reduction in both NO_x and PM is evident, with less of a reduction in CO_2 emissions.

Projected reductions in NO_x emissions are also dominated by PCs and LCVs. It should be noted however, that the NO_x *EFs* for Euro 5 and Euro 6 vehicles contain a high level of embedded uncertainty: measured NO_x emissions from Euro 5 vehicles show a high degree of variation, whilst the pool of Euro 6 vehicles that can be sampled is still very small. Hence, the NO_x *EFs* for these PCs and LDVs are likely to evolve, as more definitive and reliable data becomes available.

7.4.2 Impact of TS selection on NO_x emission estimates

As discussed in Section 7.2.1, selection of the specific TSs used to model traffic activity for the HBEFA model runs was based on matching their average speed to that associated with the corresponding [road + vehicle] combination in COPERT. In most cases, several TSs had average speeds close to the desired value, and selection of the most appropriate TS came down to “expert judgment”. It was therefore decided to re-run the HBEFA model with a range of different TSs, whilst maintaining the “average

speed" constraint of ± 5%. Only the TSs associated with PCs and LCVs were changed.

The study was carried out for both the 2010 and 2020 emission estimates, and the results are presented in Figure 57. The estimates produced by COPERT, using the same DTTAS fleet model, are also shown for comparison.

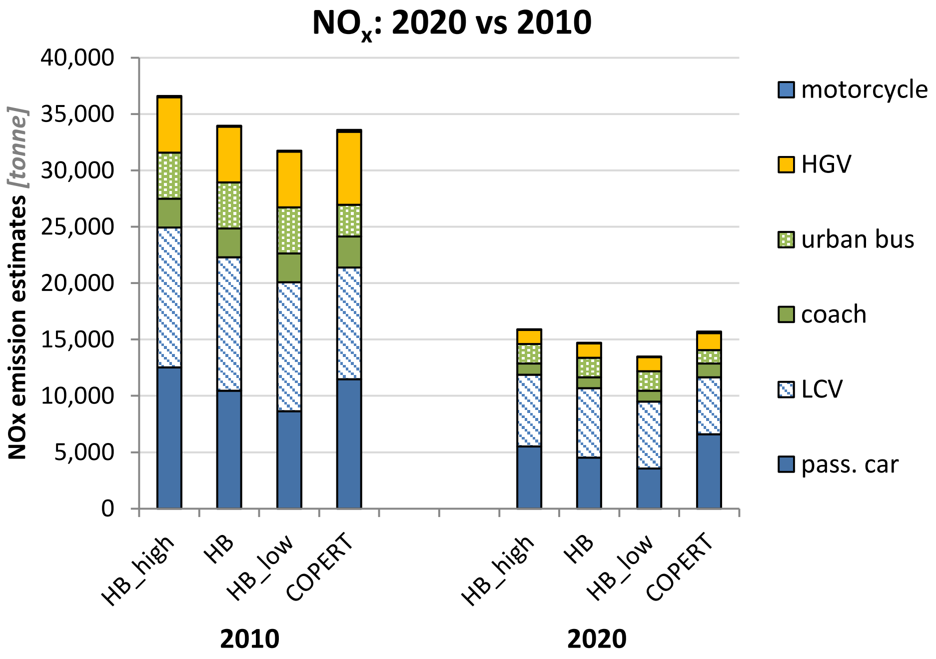

Figure 57 NO emission estimates for 2010 and for 2020. Projected decrease is about 50% for all four model runs. HB denotes HBEFA model; HB_high and HB_low denote highest and lowest emission estimates obtained by varying TSs as described above. All estimates use the DTTAS fleet profile.

For 2010, the NO_x emission estimate varies by about ± 7% when alternative TSs are used; for 2020, the variation is about ± 8%. However, the absolute variation is much smaller in 2020 as a result of the large reduction in total projected NO_x emissions. Given that the changes to the model inputs were small, these are fairly significant changes in the output.

It is interesting to note that HBEFA estimates appear to be much more sensitive to changes in TSs – even when the TSs have almost the same average speed – than COPERT estimates are to changes in assumed values for vehicle average speed (see Section 7.1.3). This suggests that HBEFA does a better job of capturing changes in NO_x emissions resulting from changed driving conditions. However, it also highlights the major challenge associated with HBEFA – identifying which TSs most accurately reflect real-world driving conditions for the region and period in question.

7.4.3 Impact of uncertainty in Euro 5/6 NO_x emission factors on emission projections

As discussed in Chapters 4 and 5, there are substantive grounds for querying whether NO_x *EFs* for Euro 5/6 and Euro V/VI vehicles will decrease as rapidly as initially expected. The substantial reductions projected for NO_x emissions in Figure 56, however, are predicated on that assumption being valid. A further series of HBEFA runs was therefore performed to test the significance of this assumption, in which it was assumed that the real-world *EFs* of Euro 5 and Euro 6 vehicles remained at Euro 4 levels. The results can be seen in Figure 58.

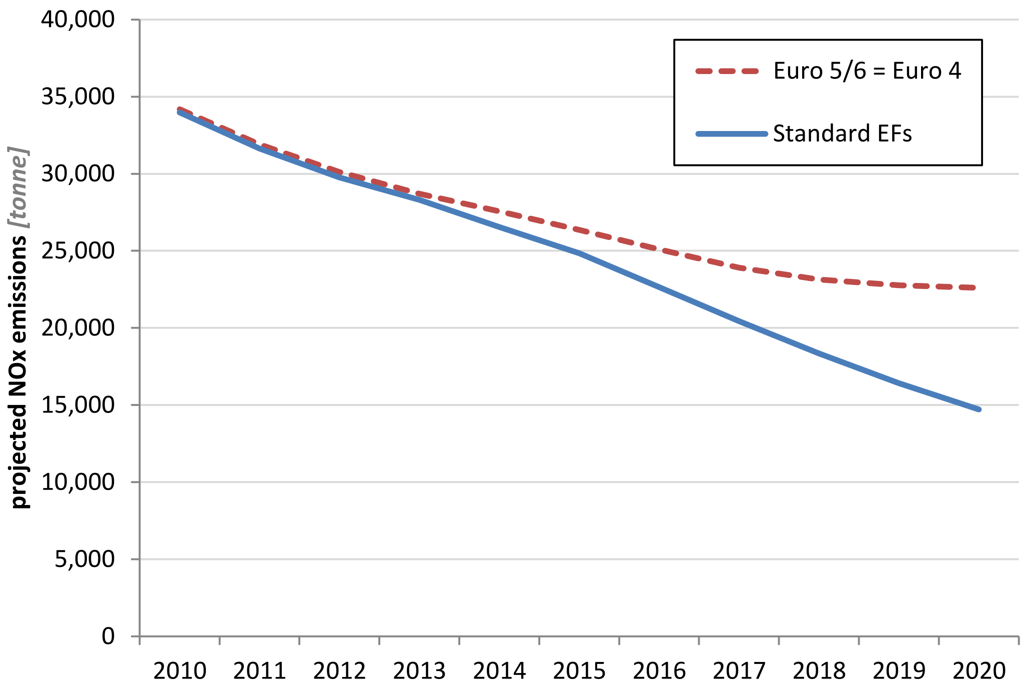

Figure 58 Projected NO_x emissions using HBEFA. Solid line: using standard EFs for Euro 5 and Euro 6 vehicles; dashed line: using Euro 4 EFs for Euro 5 and Euro 6 vehicles.

The impact of such an assumption becomes significant as the proportion of Euro 5/6 vehicles in the fleet increases. On a positive note, even in the scenario where there is zero improvement in real-world NO_x EFs for new LDVs, the overall NO_x emissions continue to decrease, albeit significantly more slowly than in the "standard" scenario. It is probably fair to say that these two *EF* scenarios are about equally likely, and therefore interesting to note that the NO_x emissions projected for 2020 in the "Euro 4" scenario are about 50% higher than those projected under the "standard" scenario.

In other words, even if the fleet description were perfect, *and* the TSs used in the model accurately captured real-world driving conditions, *and* the *EFs* for all other vehicle classes were exactly correct, the uncertainty in the emissions projection for 2020 would be of the order of 25%. Given the difficulty of complying with these constraints, the true uncertainty in the NO_x emission estimate is probably significantly higher.

The developers of COPERT have also highlighted the difficulty of measuring accurate NO_x EFs for Euro 5 and 6 diesel LDVs, in the

documentation for COPERT 4 V10.0 (Katsis et al. 2012). They recommend that further testing of diesel Euro 5 and 6 LDVs is carried out, and suggest that, as an interim solution, a "reduction factor" is used with LDV diesel vehicles. This factor is applied to Euro 4 NOx *EFs*, for diesel LDVs only. The reduction factor suggested for Euro 5 vehicles is -0.21 – i.e., the NO_x emissions from a Euro 5 diesel LDV are estimated to be 21% *higher* than from a corresponding Euro 4 vehicle – and for Euro 6 vehicles a reduction factor of 0.57 is proposed.

These two reduction factors will not be implemented in COPERT until the next release, but it is possible to apply these factors manually to the emissions values exported from the model. A comparison of the COPERT future projections with and without these two reduction factors is shown in Figure 59. A minimal change in overall emissions is evident, with a small increase in total NO_x being projected for 2011-2019 when the revised *EFs* are used.

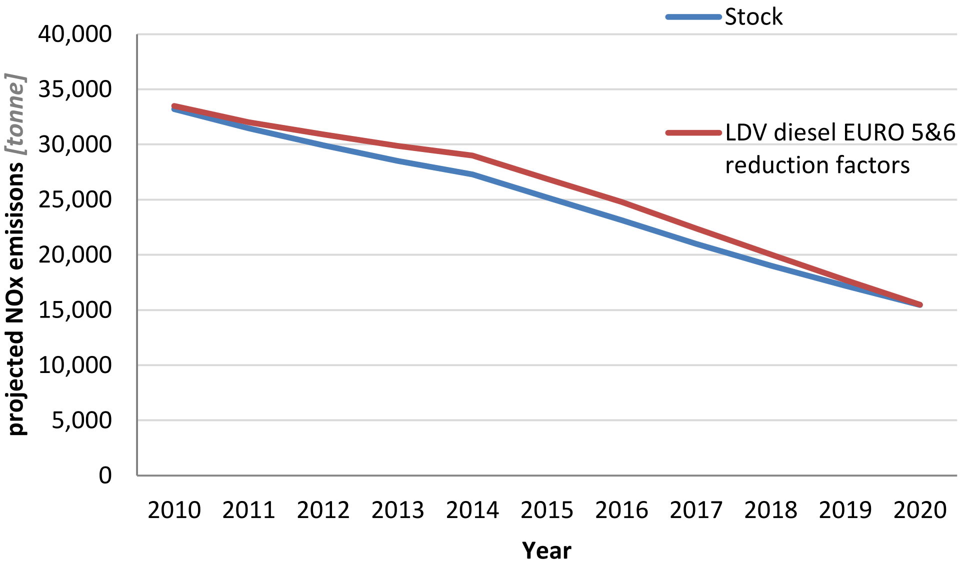

Figure 59 Comparison of COPERT NO_x projections, with and without Euro 5 and 6 LDV reduction factors.

8 Conclusions

The primary objective of the ETASCI project was to develop improved methods for estimating NO_x and particulate matter (PM) emissions from road vehicles in Ireland. In pursuit of that goal, the following tasks have been completed:

- The physical and chemical mechanisms governing formation of these pollutants have been explored (Chapter 1);
- Current and imminent technology for mitigation and control of pollutant emissions has been assessed (Chapter 1);
- Existing and scheduled emissions legislation and test procedures have been reviewed (Chapter 1);
- Direct-measurement techniques, for determining emissions during real-world driving, have been examined (Chapter 4 and Chapter 5);
- The COPERT and ARTEMIS/HBEFA inventory models have been analysed, and their relative strengths and weaknesses identified (Chapter 1);
- Emissions estimates for the current transport fleet have been generated using the ARTEMIS/HBEFA and COPERT models, and the findings compared (Chapter 1);
- The potential of the ARTEMIS/HBEFA and COPERT models to model future emissions has been investigated (Chapter 1).

Based on this analysis, the following conclusions have been reached with respect to estimating NO_x and PM emissions from road transport in Ireland:

1. <u>Estimates of NO_x and PM emissions incorporate a high degree of uncertainty: *at least* ± 20%, and almost certainly much higher.</u> This is an unavoidable consequence of the strong, non-linear dependence of NO_x and PM emissions on the details of vehicle design and operation, and the impossibility – at a practical level – of capturing that level of detail in an inventory model.

2. <u>Use of a more accurate emissions inventory model does not ensure a more accurate emission inventory.</u> The more sophisticated the model, the more detailed the input requirements become. In practice these input data are not available with the accuracy and resolution required to justify use of the more sophisticated model.

3. <u>The only reliable method for determining real-world emission factors is by direct measurement during vehicle operation</u>. This can be approached in two ways: a Portable Emissions Measurement System (PEMS) can extract detailed emissions data from a single vehicle across a wide range of drive cycles; Remote Data Sensing (RDS or RSD) can extract emissions data from a large number of vehicles at a single operating condition.

4. <u>Choice of inventory model is less important than having an accurate representation of the vehicle fleet.</u> Given the same fleet data, COPERT and HBEFA produced very similar emission estimates. However, minor changes to the fleet description produced relatively

significant changes in COPERT's emissions output.

5. With COPERT, accurate emission factors are more important than detailed activity data. The very simplistic activity model in COPERT renders it relatively insensitive to changes in activity data other than vkm. National model enhancement efforts should prioritise measurement of real-world emission factors ahead of the production of detailed activity data.

6. LCVs are responsible for a disproportionate share of NO_x and PM emissions, due to their high annual mileage and relatively high emission factors. The concentration of LCV operations in urban areas aggravates this issue.

7. PM emissions from road transport will decrease strongly over the next decade, as the use of diesel particulate filters (DPF) spreads throughout the fleet. Consequently, the significance of this sector for national emissions will decline, as will the absolute uncertainty concerning PM emission levels from this source.

8. NO_x emissions will decrease relatively slowly in the short-term, but more rapidly from 2020 on. Vehicles currently for sale comply with stringent NO_x emission standards on the official test cycle, but do not maintain that performance in real-world operation. The introduction of a much more challenging test regime with Euro 6c in 2017 should eliminate that discrepancy, with penetration of these vehicles into the fleet becoming significant from 2020 on.

9 Recommendations

Based on the conclusions drawn in Chapter 8, the authors offer the following recommendations:

Recommendation 1:

COPERT should be retained as the principal tool for estimating national NO_x and PM emissions from road transport.

However, the quality of the estimates produced can be significantly improved by specific actions in three key areas: emission factor estimation, traffic activity measurement, and fleet specification. Recommendations and actions related to each of these areas are presented below.

Recommendation 2:

Improve emissions factors for NO_x and PM.

COPERT's use of vehicle average speed as the main determinant of emission factor is a fundamental weakness, and embeds substantial uncertainty in the estimates. Use of a more sophisticated model (e.g. HBEFA) will shift, not reduce, that uncertainty. The only reliable method for determining real-world emission factors is by direct measurement during vehicle operation.

Action 1: Establish a programme to monitor real-world driving emissions. Perform a pilot study with one or more Third Level Institutions to assess options and build capacity, prior to establishing a permanent, national capability.

Action 2: Leverage the emission factor data by relating it to ambient air quality monitoring stations.

Action 3: Closely monitor COPERT emission factor development, in the context of Euro 6c (Sep 2017) and the 2021 CO_2 target of 95 g. km^{-1}. The NO_x and CO_2 intensity of passenger cars and related commercial vehicles will evolve rapidly as manufacturers adapt to these targets.

Recommendation 3:

Improve vehicle activity estimates.

These actions should focus on capturing the quantity – not quality – of traffic activity. COPERT is insensitive to drive cycle kinematics, but does capture changes in the quantity and distribution of total activity. Significant value can be extracted from existing data sets, although a National Transport Model (NTM), if developed, could offer additional insight.

Action 4: Establish a Traffic Activity Office (TAO) to act as a centralised "clearing house" for all data related to traffic activity. The Office will collect data from all available sources, format it for use with COPERT, and upload it to an online repository on a monthly basis.

Action 5: Establish a robust system for collecting, cleaning and compiling mileage data that is routinely gathered during passenger car NCT and commercial vehicle CVRT tests, and for transferring that data to the Traffic Activity Office on a weekly or monthly basis.

Action 6: Establish a robust system for providing the Traffic Activity Office with traffic volume data, gathered by the NRA and some municipal authorities, on a daily or weekly basis.

Action 7: Develop a National Transport Model. The data sets compiled by the TAO could provide both input and validation data for such a model.

Recommendation 4:

Improve vehicle fleet description.

COPERT emission estimates are sensitive to the details of the fleet composition used. Hence, maintaining an accurate and up-to-date description of the fleet is fundamental to obtaining useful emission estimates. The latest DTTAS data should be used to define fleet composition.

Action 8: Upload DTTAS fleet data to an online repository on a monthly basis.

Action 9: Develop a robust method for automatically transforming DTTAS data to a format compatible with COPERT's fleet description.

Recommendation 5:

COPERT can be improved.

In addition to the recommendations listed above – which can be implemented at national level – there are additional actions that could enhance the COPERT model itself, but are best pursued at Task Force on Emissions Inventories and Projections (TFEIP) level.

Action 10: Encourage a shift to vehicle classification by age rather than by Euro standard within COPERT. This approach is more likely to be compatible with national vehicle statistics, retains useful information within the model, and reduces the likelihood of vehicle classification errors.

Action 11: Suggest a review of vehicle categorisation by engine size. The rapid, widespread move to engine downsizing and turbocharging renders disaggregation using the current engine size bands somewhat meaningless.

Action 12: Advocate a stronger focus on determining emission factors for LCVs, which are responsible for a disproportionate share of NO_x and PM emissions. These emissions are aggravated by the fact that LCV activity is concentrated in urban areas.

Action 13: Propose that all emissions estimates produced by COPERT should explicitly state the associated confidence interval. This would build confidence in the estimates produced, and highlight the estimates most urgently requiring further attention.

References

Andre, M. (2004). "The ARTEMIS European driving cycles for measuring car pollutant emissions." Science of the Total Environment **334**: 73-84.

Annesi-Maesano, I., Forastiere, F. et al. (2007). "Particulate matter, science and EU policy." European Respiratory Journal **29**(3): 428-431.

Barlow, T. and Boulter, P. (2009). "Emissions factors 2009: Report 2 - A Review of the Average-speed Approach for Estimating Hot Exhaust Emissions". Published Project Report PPR309, Transport Research Laboratory, UK.

Bloomberg News. (2013). "Beijing smog closes highways as travellers return after holiday." http://www.bloomberg.com/news/2013-10-07/beijing-smog-closes-highways-as-travelers-return-after-holiday.html

Boulter, P.M.I. (2007). "Assessment and reliability of transport emission models and inventory systems - Final report." Unpublished Report IE/044/07, TRL Limited, Wokingham, UK.

Carslaw, D.C., Beevers, S.D. et al. (2011). "Recent evidence concerning higher NO_x emissions from passenger cars and light duty vehicles." Atmospheric Environment **45**(39): 7053-7063.

Casey, E. (2011). Estimating vehicular emissions and informing air quality assessments for Dublin, using descriptions of vehicle activity. Ph.D. Thesis, School of Electrical, Electronic & Mechanical Engineering, University College Dublin, Ireland.

Central Statistics Office. (2014). Environmental Indicators Ireland 2014. Available at http://www.cso.ie/en/media/csoie/releasespublications/documents/environment/2014/eii2014.pdf

de Haan, P. and Keller, M. (2000). "Emission factors for passenger cars: application of instantaneous emission modeling." Atmospheric Environment **34**(27): 4629-4638.

DEHLG. (2007). "Ireland National Climate Change Strategy 2007 - 2012". Department of the Environment, Heritage, and Local Government, Stationery Office, Dublin, Ireland. Also available online at www.environ.ie

Direct-Industry. (2011). "On Board Emissions Analyzers: SEMTECH-DS." Retrieved 26-04-2011, from http://news.directindustry.com/press/sensors-europe/on-board-emissions-analyzers-semtech-ds-60795-342005.html.

Dockery, D.W., Pope, C.A. et al. (1993). "An Association between Air Pollution and Mortality in Six U.S. Cities". N Engl J Med **329**:1753-1759 DOI: 10.1056/NEJM199312093292401

DTTAS. (2010). Irish Bulletin of Vehicle and Driver Statistics 2010. Department of Transport, Tourism and Sport, Dublin, Ireland. Available at http://www.transport.ie

EEC. (1988). "Council Directive 88/77/EEC of 3 December 1987 on the approximation of the laws of the Member States relating to the measures to be taken against the emission of gaseous pollutants from diesel engines for use in vehicles." Official Journal of the European Communities **L 36:** 33-61.

EPA. (2011). "Ireland's Transboundary Gas Emissions in 2009." Irish Evironmental Protection Agency, Dublin, Ireland. Also available online at: http://www.epa.ie/downloads/pubs/air/airemissions/.

EPA. (2011b). "National Inventory Report 2011". Irish Environmental Protection Agency, Dublin, Ireland. Available online at http://coe.epa.ie/ghg/nirs/NIR_2011_IE_v2.1.pdf.

EPA. (2014). "Ireland's Transboundary Gas Emissions in 2012". Irish Environmental Protection Agency, Dublin, Ireland. Available online at

http://www.epa.ie/pubs/reports/air/airemissions/NECD_Summary_Rpt_%202014.pdf

Franco, V., Kousoulidou, M. et al. (2013). "Road vehicle emission factors development: A review." Atmospheric Environment **70**(0): 84-97.

Grummell, T. (2013). "Uncertainty of Emissions Inventories for NO_x and Particulate Matter from Road Vehicle Transport in Ireland." M.Eng.Sc. Thesis, University College Dublin, Ireland.

Hausberger, S., Rexeis, M. et al. (2009). "Emission Factors from the Model PHEM for the HBEFA Version 3." Graz University of Technology, Report Nr. I-20/2009 Haus-Em 33/08/679. Available online at (http://www.hbefa.net/e/index.html (accessed 20/10/11)).

Heywood, J.B. (1988). Internal Combustion Engine Fundamentals. New York, McGraw-Hill International.

Joumard, R., Jost, P. et al. (1994). "Hot passenger car emissions modelling as a function of instantaneous speed and acceleration." The Science of the Total Environment **169 (199.5)** 167-174.

Katsis, P., Ntziachristos, L. et al. (2012). "Description of new elements in COPERT 4 v10.0." Emisia SA Report No: 12.RE.012.V1, Thessaloniki, Greece. Available online at http://www.emisia.com/files/COPERT4_v10_0.pdf

Kouridis, C., Gkatzoflias, D. et al. (2009). "Uncertainty estimates and guidance for road transport emission calculations." EUR 24296 EN, Publications Office of the European Union, Luxembourg. Available online at http://publications.jrc.ec.europa.eu/

Kousoulidou M. et al. (2013). "Use of portable emissions measurement system (PEMS) for the development and validation of passenger car emission factors". Atmospheric Environment **64:** 329-338.

Kraft, M.E.T., Kappos, A., et al. (2005). "The German view: effects of nitrogen dioxide on human health--derivation of health-related short-term and long-term values." International Journal of Hygiene and Environmental Health **208**: 305-318.

Lee, T. et al. (2013). "Variability in operation-based NO_x emission factors with different test routes, and its effects on the real-driving emissions of light diesel vehicles". Science of the Total Environment **461-462:** 377-385

Liu, Z. et al. (2011). "Real-world operation conditions and on-road emissions of Beijing diesel buses measured by using portable emission measurement system and electric low-pressure impactor". Science of the Total Environment **409**: 1476-1480

Martinelli, N., Olivieri O., and Girelli, D. (2013). "Air particulate matter and cardiovascular disease: a narrative review". Eur J Intern Med. **24**(4):295-302. doi: 10.1016/j.ejim.2013.04.001.

Pope, C.A., Thun, M.J., Namboodiri, M.M., Dockery, D.W., Evans, J.S., Speizer, F.E. and Heath, C.W. (1995). "Particulate Air Pollution as a Predictor of Mortality in a Prospective Study of U.S. Adults", American Journal of Respiratory and Critical Care Medicine, Vol. **151**, No. 3_pt_1, pp. 669-674. doi: 10.1164/ajrccm/151.3_Pt_1.669

Pope, C.A., Burnett, R.T. et al. (2002). "Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution". Journal of the American Medical Asssociation **287**(9):1132-1141. doi:10.1001/jama.287.9.1132.

Reynolds, A.W.B.M.B. (2000). "Development of an emissions inventory model for mobile sources." Transportation Research Part D: Transport and Environment. **vol. 5**: p 77-101.

Sjodin, Å., Ekstrom, M. et al. (2006). Implementation and evaluation of the road model for Sweden's International reporting obligations on air emissions. 2nd Conference Environment & Transport. Reims, France.

Smit, R., Smokers, R. et al. (2006). "A New Modelling Approch for Road Traffic Emissions: VERSIT+ LD - Background and Methodology". TNO Report 06.OR.PT.016.1/RS, TNO Science and Industry, Delft, The Netherlands. Available online at:
http://www.emissieregistratie.nl/ERPUBLIEK/documenten/Lucht%20(Air)/Verkeer%20en%20Vervoer%20(Transport)/Smit%20et%20al.%20(2006)%20A%20new%20modelling%20approach%20for%20road%20traffic%20emissions;%20VERSIT+%20LD%20-%20background%20and%20methodoly.pdf

TRB. (2008). "Real-World Carbon Dioxide Impacts of Traffic Congestion." Transportation Research Record: Journal of the Transportation Research Board **2058**: 163-171.

Wang, X. et al. (2011). "On-road emission factor distributions of individual diesel vehicles in and around Beijing, China." Atmospheric Environment **45** (2): 503-513.

Weiss, M., Bonnel, P. et al. (2012). "Will Euro 6 reduce the NOx emissions of new diesel cars? – Insights from on-road tests with Portable Emissions Measurement Systems (PEMS)." Atmospheric Environment **62**(0): 657-665.

WHO-Europe. (2004). Health Aspects of Air Pollution. World Health Organisation. Copenhagen, Denmark.

WSP Group. (2011). Irish National Transport Model Feasibility Study. Dublin, Ireland.

Appendix 1 Euro-standard emission limit value, test procedures, and application dates

Table 11. **HDV steady-cycle emission limit values, application date, and test cycles (Euro I – VI)**

HDV Steady Cycle							
Tier	**Date (effective from)**	**NO_x**	**PM**	**Test procedure**	**CO**	**HC**	**Smoke**
		$[g.kWh^{-1}]$			$[g.kWh^{-1}]$		
Euro I	1992 (< 85 kW)	**8.0**	**0.612**	R-49	4.5	1.1	
	1992 (> 85 kW)		**0.36**		4.5	1.1	
Euro II	October 1996	**7.0**	**0.25**		4.0	1.1	
	October 1998		**0.15**		4.0	1.1	
Euro III	*October 1999 (EEVs only)*	*2.0*	*0.02*	*ESC & ELR*	*1.5*	*0.25*	*0.15*
	October 2000	**5.0**	**0.10** **0.13**[a]	ESC & ELR	2.1	0.66	0.8
Euro IV	October 2005	**3.5**	**0.02**		1.5	0.46	0.5
Euro V	October 2008	**2.0**	**0.02**		1.5	0.46	0.5
Euro VI	January 2013	**0.4**	**0.01**	WHSC	1.5	0.13	

Notes:
[a] for engines of less than 0.75 dm^3 swept volume per cylinder, and a rated-power speed of more than 3000 min^{-1}

EEV - enhanced environmentally-friendly vehicles

Table 12. HDV transient-cycle emission limit values, application date, and test cycles (Euro I – VI)

HDV Transient Cycle						
Tier	**Date (effective from)**	**NO_x** [g.kWh^{-1}]	**PM** [g.kWh^{-1}]	**Test procedure**	**CO** [g.kWh^{-1}]	**NMHC** [g.kWh^{-1}]
Euro III	*October 1999 (EEVs only)*	*2.0*	*0.02*	ETC	*3.0*	*0.40*
	October 2000	**5.0**	**0.16** **0.21**[a]		5.45	0.78
Euro IV	October 2005	**3.5**	**0.03**		4.0	0.55
Euro V	October 2008	**2.0**	**0.03**		4.0	0.55
Euro VI	January 2013	**0.46**	**0.01**	WHTC	4.0	0.16[b]

Notes:

[a] for engines of less than 0.75 dm^3 swept volume per cylinder, and a rated-power speed of more than 3000 min^{-1}

[b] THC for diesel engines

EEV - enhanced environmentally-friendly vehicles

Table 13. **Euro LDV passenger car emission limit values, and application dates (Euro 1 – 6)**

	LDV – passenger cars							
Tier	**Date (vehicle registration)**	**Engine type**	**NO_x**	**PM**	**CO**	**HC**	**HC + NO_x**	**PN**
			[$mg.km^{-1}$]		[$g.km^{-1}$]			[$\#.km^{-1}$]
Euro 1	Jan 1993	Diesel	**970**	**140.0**	2.72			
		Gasoline						
Euro 2	Jan 1997	Diesel	**900**	**100.0**	1.0			
		Gasoline	**500**		2.20			
Euro 3	Jan 2001	Diesel	**500**	**50.0**	0.64		0.56	
		Gasoline	**150**		2.30	0.20		
Euro 4	Jan 2006	Diesel	**250**	**25.0**	0.50		0.30	
		Gasoline	**80**		1.0	0.10		
Euro 5a	Jan 2011	Diesel	**180**	**5.0**	0.50		0.23	
		Gasoline	**60**		1.0	0.10		
Euro 5b	Jan 2013	Diesel	**180**	**5.0**	0.50		0.23	6.0×10^{11}
		Gasoline	**60**		1.0	0.10		
Euro 6b	Sep 2015	Diesel	**80**	**4.5**	0.50		0.17	6.0×10^{11}
		Gasoline	**60**		1.0	0.10		6.0×10^{12}
Euro 6c	Sep 2018	Diesel	**80**	**4.5**	0.50		0.17	6.0×10^{11}
		Gasoline	**60**		1.0	0.10		6.0×10^{11}

Table 14. LCV emission limit values, and application dates (Euro 1 – 6)

	LCV – N_1 Class I and Class II (diesel)							
Tier	**Date (type approval)**	**Engine type**	**NO_x**	**PM**	**CO**	**HC**	**HC + NO_x**	**PN**
			[$mg.km^{-1}$]		[$g.km^{-1}$]			[$\#.km^{-1}$]
Euro 1	Oct 1994	Class I		**0.140**	2.72		**0.970**	
		Class II		**0.190**	5.17		**1.400**	
Euro 2	Jan 1998	Class I		**0.100**	1.0		**0.900**	
		Class II		**0.140**	1.25		**1.300**	
Euro 3	Jan 2001	Class I	**0.500**	**0.050**	0.64		0.560	
		Class II	**0.650**	**0.070**	0.80		0.720	
Euro 4	Jan 2006	Class I	**0.250**	**0.025**	0.50		0.300	
		Class II	**0.330**	**0.040**	0.63		0.390	
Euro 5a	Sep 2010	Class I	**0.180**	**0.005**	0.50		0.230	
		Class II	**0.235**		0.63		0.295	
Euro 5b	Sep 2011	Class I	**0.180**	**0.005**	0.50		0.230	6.0×10^{11}
		Class II	**0.105**		0.63		0.295	
Euro 6	Sep 2015	Class I	**0.080**	**0.005**	0.50		0.170	6.0×10^{11}
		Class II	**0.105**		0.63		0.195	

Table 15. **LCV emission limit values, and application dates (Euro 1 – 6)**

	LCV – N_1 Class III and N_2 (diesel)							
Tier	**Date (type approval)**	**Engine type**	**NO_x**	**PM**	**CO**	**HC**	**HC + NO_x**	**PN**
			[mg.km^{-1}]		[g.km^{-1}]			[#.km^{-1}]
Euro 1	Oct 1994	Class III		**0.250**	6.90		**1.70**	
		N_2						
Euro 2	Jan 1998	Class III		**0.200**	1.50		**1.60**	
		N_2						
Euro 3	Jan 2001	Class III	**0.780**	**0.100**	0.95		0.860	
		N_2						
Euro 4	Jan 2006	Class III	**0.390**	**0.060**	0.74		0.460	
		N_2						
Euro 5a	Sep 2010	Class III	**0.280**	**0.005**	0.74		0.350	
		N_2						
Euro 5b	Jan 2013	Class III	**0.280**	**0.005**	0.74		0.350	6.0×10^{11}
		N_2						
Euro 6	Sep 2016	Class III	**0.125**	**0.005**	0.74		0.170	6.0×10^{11}
		N_2						

Appendix 2 EPA and DTTAS fleet models for COPERT and HBEFA

Updated Ireland fleet numbers (for HBEFA model)

COPERT classifies vehicles by emissions compliance standard (Euro standard), not age. HBEFA classifies vehicles by age, and uses an internal lookup table to determine the corresponding Euro standard. HBEFA therefore allows the user to input the age profile of each vehicle class into the programme – data which is generally readily available from national databases.

This makes it difficult to directly compare the COPERT fleet to the HBEFA fleet, as only the emissions-level breakdown of the fleet is provided in the EPA COPERT data, and not the actual age profiles of the vehicles. However a fleet for use with the HBEFA programme has been compiled from the Department of Transport, Tourism and Sport (DTTAS) data, and is presented and explained in the following section.

Ireland national fleet database overview

In August 2011, data for all of the registered vehicles from the years 2006 to 2010 within the Republic of Ireland was obtained from the Department of Transport, Tourism and Sport (DTTAS). A chart of the relevant fleet breakdown for Ireland for use as inputs to the HBEFA model is shown in Figure 62. As has been stated before, the HBEFA model does not need the Level 4 (emissions limits) breakdown, as it can calculate this from the inbuilt library within the software. A breakdown in the data from the DTTAS was requested in the format similar as described in Levels 1, 2 and 3 of Figure 62, and further to this, a breakdown of the age profile of each of these vehicles subclasses. The age profiles were of the format of the number of vehicles registered in each of the previous 16 years, followed by the sum of all the vehicles that were registered more than 16 years previously.

Figure 60 shows the age profiles of Irish passenger cars in 2010. This data can be directly imported to the HBEFA programme. Data for commercial vehicles for Ireland obtained from the DTTAS was classified in term of vehicle weight, with no disaggregation of vehicle type (LCV/HGV). A transformation table for diesel vehicles is shown in Figure 61; a similar table exists for petrol LCVs. Note that the HBEFA HGV fleet contains only “rigid” trucks, because the DTTAS fleet does not distinguish between rigid and articulated trucks.

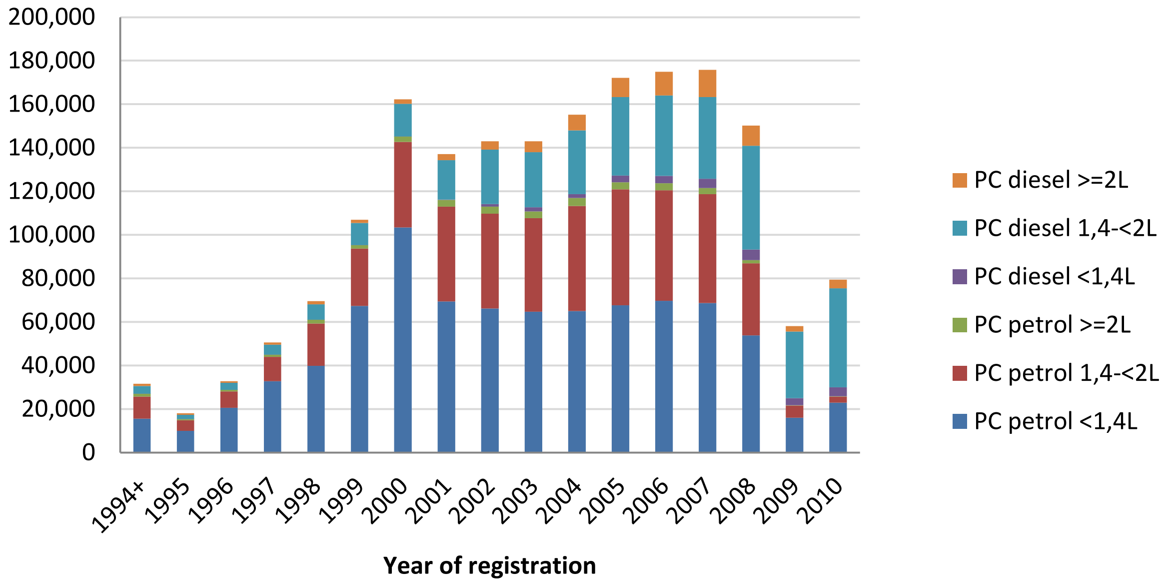

Figure 60 DTTAS age profile of all passenger vehicles in Ireland for 2010. A spike in sales of petrol <1.4L vehicles can be attributed to the 2000 scrappage scheme, and the low volume of new car registrations in 2009 is due to the economic downturn.

DTTAS Database of LCV and HGV (by weight in kg)							HBEFA Catogory	Weight combination (kg)
Not Exceeding 610	611 - 813	814 - 1016	1017 - 1270				LCV M1+N1-I	<1270
1271 - 1524	1525 - 1778						LCV N1-II	1271-1778
1779 - 2032	2033 - 2286	2287 - 2540	2541 - 2794	2795 - 3048			LCV N1-III	1779-3048
3049 - 3302	3303 - 3556	3557 - 3810	3811 - 4064	4065 - 5080	5081 - 6096	6097 - 7112	Rigidtruck<7,5T	3049-7112
7113 - 8128	8129 - 9144	9145 - 10160	10161 - 11176	11177 - 12192			Rigidtruck 7,5-12T	7113-12192
12193 - 13208	13209 - 14224						Rigidtruck 12-14T	12193-14224
14225 - 15240	Exceeding 15241						Rigidtruck 14-20T	> 14225

Figure 61 Transformation table for diesel LCVs and HGVs Irish database, as obtained from DTTAS. 28 individual weight categories are rationalised to seven different vehicle classes, (3 LCVs and 4 HGVs), which can be exported to the HBEFA model.

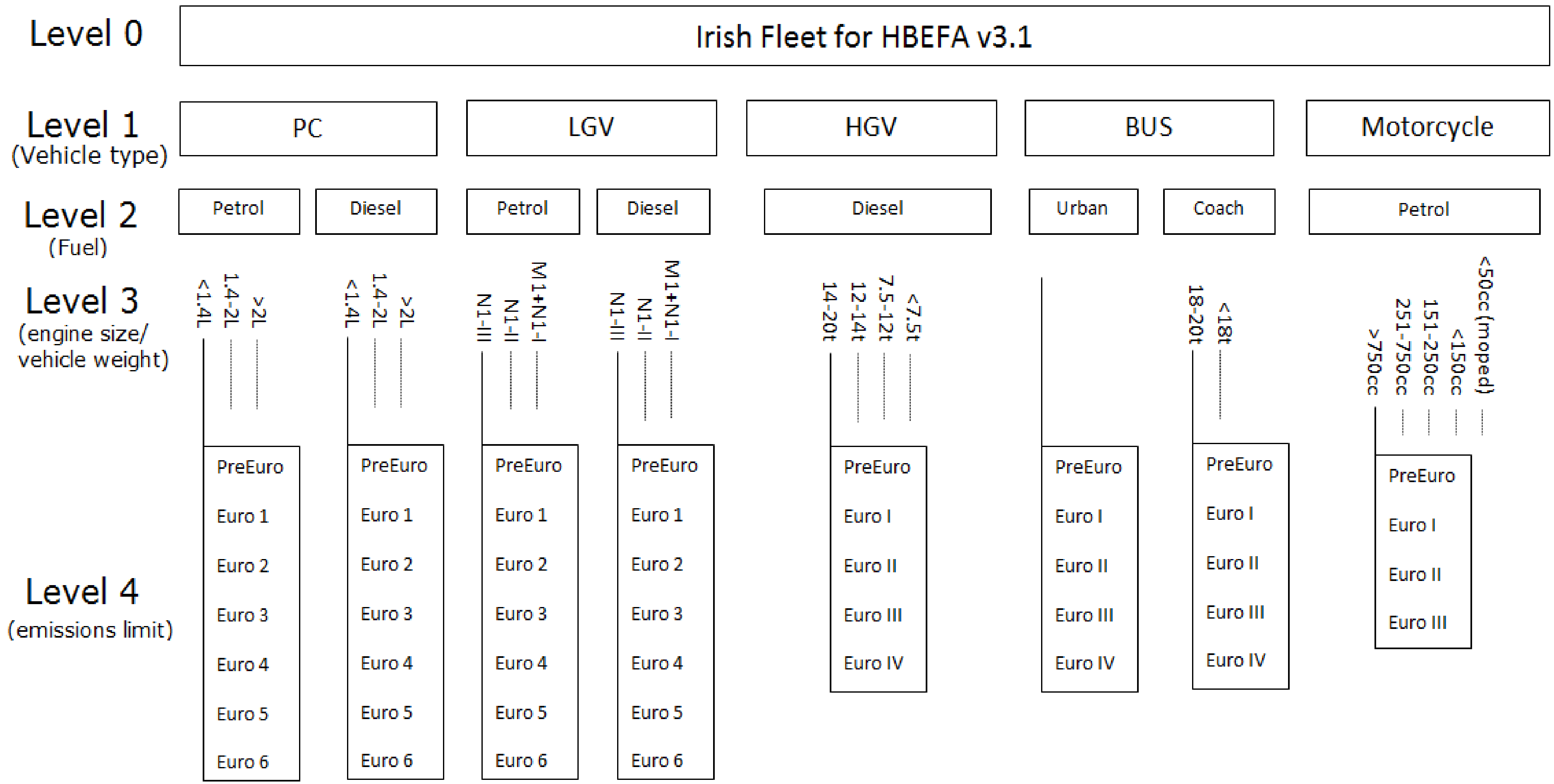

Figure 62 HBEFA Irish fleet description (NB: Vehicles classes with very low vehicle count may not be fully accounted for. These include petrol engined HGVs, LPG passenger cars, etc.).

List of Acronyms

AdBlue.................... Aqueous Urea

ADC........................ ARTEMIS Drive Cycle

AFR........................ Air:Fuel Ratio

ANPR..................... Automatic Number-Plate Recognition

ARTEMIS............... Assessment and Reliability of Transport Emission Models and Inventory Systems

BC.......................... Black Carbon

BPTG..................... Beijing Public Transportation Group

CADC..................... Common ARTEMIS Drive Cycle

CI............................ Compression Ignition

CO.......................... Carbon Monoxide

CO_2......................... Carbon Dioxide

COPERT................ Computer Programme to calculate Emissions from Road Transport

CVRT..................... Commercial Vehicle Road Test

DECLG................... Department of the Environment, Community, and Local Government

DEHLG................... Department of the Environment, Heritage, and Local Government

DISI........................ Direct-Injection, Spark-Ignition

DOC....................... Diesel Oxidation Catalyst

DPF........................ Diesel Particulate Filter

DTTAS................... Department of Transport, Tourism and Sport

ECE........................ Economic Commission for Europe

EEA........................ European Environment Agency

EEC........................ European Economic Commission

EEV........................ Enhanced Environmentally-friendly Vehicles

EF........................... Emission Factor

EGR....................... Exhaust-Gas Recirculation

ELPI....................... Electrical Low Pressure Impactor

ELR........................ Engine Load Response

EMS....................... Engine Management System

EPA........................ Environmental Protection Agency

EPB........................ Environmental Protection Bureau (Municipal Beijing)

ESC........................European Stationary Cycle

ETASCI ....Emissions from Transport and Small Combustion Installations

ETC ....European Transient Cycle

EU ....European Union

EUDC ....Extra-Urban Drive Cycle

FC ....Fuel Consumption

GDI ....Gasoline Direct-Injection

GIS ....Geographical Information System

GPS........................Global Positioning System

GRPE ....Working Party on Pollution and Energy (UNECE)

H_20 ....Water

HBEFA ....Handbook of Emission Factors

HD ....Heavy-duty

HDV........................Heavy-duty Vehicle

HGV ....Heavy Goods Vehicle

IC............................Internal Combustion

ICCT ....International Council on Clean Transportation

ICE ....Internal Combustion Engine

INRETS LTE ....Institut National de Recherche sur les Transports et leur Sécurité, Laboratoire Transports et Environnement

JRC ....Joint Research Centre

KDC........................Korean Driving Cycle

LCV ....Light Commercial Vehicles

LDDV......................Light-duty Diesel Vehicles

LDV ....Light-duty Vehicles

LGV ....Light Goods Vehicles

LNT ....Lean NO_x Trap

MNEDC ....Modified New European Driving Cycle

MOVES ....Motor Vehicle Emissions Simulator

N_2 ....Nitrogen

NAEI ....National Atmospheric Emissions Inventory

NCT........................National Car Test

NEC........................National Emissions Ceiling

NEDC.....................New European Driving Cycle

NH_3.........................Ammonia

NO..........................Nitrogen Monoxide

NO_2.........................Nitrogen Dioxide

NO_x.........................Nitrogen Oxide

NRA........................National Roads Authority

NTM ....National Transport Model

OBD ....On-Board Diagnostics

PC ....Passenger Car

PEMS.....................Portable Emissions Measurement System

PHEM.....................Passenger-car and Heavy-duty Emission Model

PM..........................Particulate Matter

POWCAR...............Place of Work Census of Anonymised Records

R49.........................13-mode, steady-state, diesel engine test cycle introduced by ECE Regulation No. 49

RDE........................Real-world Driving Emissions

RDS........................Remote Data Sensing

RPA........................Relative Positive Acceleration

RSD........................Remote Sensing Device / Remote Sensing of Data

RTP........................Representative Test Pattern

SCATS...................Sydney Coordinated Adaptive Traffic System

SCR........................Selective Catalytic Reduction

SI............................Spark Ignition

TAO........................Traffic Activity Office

TCD........................Trinity College Dublin

TFEIP.....................Task Force on Emissions Inventories and Projections

TRB........................Transportation Research Board

TS...........................Traffic Situation

TWC.......................Three-Way Catalyst

UHC ....Unburned Hydrocarbons

UK .... United Kingdom

UN .... United Nations

UNECE................... United Nations Economic Commission for Europe

US .... United States

Vavg .... Average Speed

VFI.......................... Velocity Fluctuation Index

Vmax .... Peak Vehicle Speed

WHO .... World Health Organisation

WHSC .... Worldwide Harmonised Steady Cycle

WHTC .... Worldwide Harmonised Transient Cycle

WLTC .... Worldwide Light duty Test Cycle

WLTP .... Worldwide Harmonised Light Vehicles Test Procedure

EPA Research Report 148

Improved Emissions Inventories for NOx and Particulate Matter from Transport and Small Scale Combustion Installations in Ireland (ETASCI) Part 1 of 2*

epa Research
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Authors: Dr. William J. Smith, Timothy Grummell B.E., Dr. David J Timoney

Combustion-related emissions of NOx and particulate matter (PM) have been shown to exert a strongly negative impact on the environment and on human health. As part of its contribution to international mitigation efforts, Ireland is committed to quantifying and reporting national emissions of these pollutants on an annual basis. This project provides improved estimates of Irish emissions from small, distributed, combustion sources, specifically from the Road Transport sector (Work Package 1), and from Non-road Transport, and stationary combustion installations such as domestic boilers (Work Package 2).

Identifying Pressures

Small combustion installations, whether stationary or mobile, emit pollutants near ground level, and in close proximity to humans. It is therefore vital that the levels of emission from these sources are accurately quantified: a difficult challenge. The emissions are estimated by combining activity data – which details fuel use by type and sector – with emission factors (EFs), which quantify the mass of each pollutant emitted per unit of fuel consumed. Whereas accurate activity data is, in general, available, emission factors for many of the fuel and appliance combinations commonly used in Ireland are highly uncertain.

Informing Policy

This project can inform policy in relation to emissions from both stationary, and mobile, combustion sources. With regard to small, stationary sources, emission factors specific to Irish conditions have been developed within the project for some combinations of fuel and appliance. These emission factors should be implemented in future emission inventory estimates, and corresponding emission factors for solid-fuel appliances should be developed as a matter of urgency. With respect to road transport, the project has demonstrated that an inherently high level of uncertainty is embedded in the NOx emission factors associated with current vehicle technologies. Policymakers should therefore be wary of ascribing too much weight to these data when formulating policy.

Developing Solutions

The Ireland-specific emission factors determined in this project are an intrinsically valuable output; the facilities and expertise developed within the project, however, will be of even greater benefit. That new capacity is currently being used to develop emission factors for small appliances burning a range of solid fuels. This important, but very challenging, task will also identify fuel-appliance combinations that are particularly injurious to the environment or to human health, and if necessary could inform the development of policies that might limit or proscribe their use.

** The ETASCI project is divided into two complementary but distinct work packages (WPs) and final reports. WP1 focuses on emissions from road transport and is detailed in EPA Research Report 148. WP2 examines emissions from non-road transport and from small combustion installations (SCIs) and is detailed in EPA Research Report 149.*

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