Uncertainty in exhaust emissions from passenger cars; a policy perspective

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Declaration of Originality

I declare that all academic work presented within this thesis is my own, and where work of others is quoted or described it is appropriately referenced.

Copyright Declaration

Abstract

In Europe the regulations that limit vehicle emissions, the Euro Standards, have failed to effectively tackle pollutant emissions in the real world. This thesis contains an appraisal of the real world emissions of modern European vehicles, which were identified as a major cause of uncertainty in UK policy with respect to compliance with air pollution legislation. The thesis includes key background information on air pollution and its control in the UK and a comprehensive review of the existing literature relating to real world emissions of petrol and diesel passenger cars.

The real world emissions performance of modern vehicles was assessed using Portable Emissions Measurement System (PEMS) data, provided by Emissions Analytics, which included 147 Euro 5 and 6 diesel and petrol vehicles. Comparisons were made to the emissions factors of the recommended air quality transport model of the European Union, COPERT, as well as the Euro standard type approval limits. The potential impact of these real world emissions was also assessed using the UK Integrated Assessment Model to perform scenario analysis up to 2030. Scenarios were used to explore the potential effect of different passenger car emissions factors on total UK NO_x (nitrogen oxides) and CO_2 (carbon dioxide) emissions, damage costs and annual mean concentrations of nitrogen dioxide (NO_2). Considering the results of these investigations, wider conclusions were drawn as to how policy makers might effectively reduce passenger car related pollution in European towns and cities. A key conclusion of this thesis is that due to the large variability in the real world emissions of vehicles within a single Euro class, policies could be more effect if real world variability was taken into account, as opposed to relying solely on the Euro standard.

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Chapter 1. Introduction

Air quality in the United Kingdom has been described by many, including the United Nations, as being in a state of crisis. Much of this stems from a report by the *Royal College of Physicians* that estimated in the UK ~40,000 deaths per year are attributable to outdoor air pollution *(RCP, 2016)*. This report, along with a growing body of evidence, links air pollution to a litany of adverse health effects. These include; cancer, COPD, diabetes, stroke, dementia and asthma. Similar reports by the *European Environment Agency (EEA, 2016)* and *World Health Organisation (WHO, 2016)* have revealed the global scale of the air pollution crisis, with an annual estimate of 467,000 premature deaths in Europe attributable to air pollution and 3 million premature deaths worldwide. Whilst these figures are often misquoted and refer to statistical lives rather than cause of death prognosis, the message is clear; action must be taken to tackle air pollution in our towns and cities.

Currently in the UK the most prominent issue relating to air quality is the concentration of nitrogen dioxide (NO_2) at roadside locations. NO_2 is the only statutory limit value the UK consistently fails to meet. Crucially, transport emissions often occur in the urban environment where public exposure is highest. As a result, most policies to address the UK's air quality problem have transport at their heart. Transport emissions, particularly from diesel fuelled vehicles, constitute one third of the UK's total annual nitrogen oxides (NO_x) emissions.

As well as limits on ambient concentrations of NO_2, the European Union (EU) also sets a limit on the maximum allowed annual NO_x emissions of member states in tonnes. These limits are regulated by the National Emissions Ceiling Directive (NECD). In this research, projections of total NO_x emissions and concentrations of ambient NO_2 were modelled by the UK Integrated Assessment Model (UKIAM). UKIAM is an integrated assessment air pollution model, developed by the Integrated Assessment Unit at Imperial College London. UKIAM performs policy appraisals and cost benefit analysis of proposed policies by considering emissions projections, abatement options, atmospheric dispersion and environmental impacts.

To tackle road transport emissions the EU introduced the Euro standards in the late 1990's. The Euro standards use type approval tests to regulate the exhaust emissions of road transport vehicles. Successive Euro standards have set increasingly stringent emissions limits, leading to a reduction in vehicular emissions and improvement in European air quality. However, by the early 2000's it became clear that emissions in the real world were not falling at the same rate as the Euro standard limits.

A key issue, relating to diesel vehicles, is that the emissions recorded when driving in the real world often far exceed the EU type approval limits which are met in the lab. Previously it was thought the difference between real world and type approval limits was due to the type approval test being conducted in a laboratory and the test cycle not being representative of the real world. The Volkswagen Emissions scandal of September 2015 shed light on a more sinister explanation with the discovery of defeat devices.

The US Environmental Protection Agency found Volkswagen had illegally installed software in their diesel vehicles that cheated the type approval tests. Since then cases have been brought against several other motor manufacturers. A study by the German government found evidence of some form of defeat device in 30% of the diesel vehicles they tested. The presence of a defeat device explains some of the deviation between real world emissions and type approval limits, but not all. This leaves policy makers with a serious problem. Diesel vehicles make up ~40% of the UK passenger fleet, but there is huge uncertainty surrounding their emissions performance in the real world. This begs the question; under such uncertainty, how do decision makers design policies that will effectively tackle the health threats posed by diesel emissions?

This thesis aims to address this uncertainty and provide scientific evidence to assist in policy decisions. This is done using a variety of methods including real world emissions measurement data from a Portable Emissions Measurement System (PEMS) and sensitivity analysis using the UKIAM.

1.1 Research Questions

This thesis answers three key research questions:

- What are the key uncertainties relating to emissions from passenger cars?
- How can these uncertainties be minimised?
- How can this be translated into effective air quality policy?

1.2 Research aim and objectives

The main aim of this research is to reduce the uncertainty surrounding real world NO_x and carbon dioxide (CO_2) emissions from passenger cars and provide a more robust evidence base for policy makers. A series of research objectives have been defined in order to thoroughly address the research questions stated above:

1. Develop a framework to assess the possible causes of uncertainty in passenger car emissions and potential risks
2. Use Portable Emissions Measurement System (PEMS) data to explore real world emissions of passenger cars
3. Use modelling to project and estimate the impact and risk associated with real world passenger car emissions and surrounding uncertainty
4. Identify how air quality policies can tackle air pollution from passenger cars given the identified uncertainty

1.3 Scope of research

The analysis presented in this thesis relates specifically to tail pipe emissions from passenger cars; explicitly NO_x, NO_2 and CO_2 from Euro 5 and 6 diesel and petrol passenger cars. The research has a specific UK focus, though many of the findings are transferable to other European countries.

The limitations of this research are therefore the sectors and pollutants not included in the analysis. These include; pre- Euro 5 (2009) passenger cars, heavy goods vehicles, light duty vehicles and buses. Of the pollutants not included, the most important omission is Particulate Matter (PM). Existing literature relating to PM is referred to throughout and factored into the discussion and analysis of results. The reason for the omission of PM was it could not be recorded by the PEMS equipment used in the analysis. Additionally, it is now thought more than half of PM at roadside locations comes from non- exhaust emissions. This analysis also omits well-to-wheel and non-exhaust emissions, though like PM they are considered in the discussion.

1.4 Research methods

In this thesis mixed methods are applied to bridge the science/policy interface. Elements of the research are highly quantitative, such as analysis of the PEMS data and modelling of emissions scenarios, other elements use qualitative methods, such as the Hazards and Operability (HAZOP) technique for risk assessment. The HAZOP technique provides a structural framework for the research, HAZOP assessments first identify an uncertainty, then quantifying it and finally deducing the risk it poses.

The study begins with a case study to demonstrate how HAZOP can be used as an uncertainty framework. A HAZOP assessment of the UKIAM identifies the areas of uncertainty in air quality policy making and the risks attached. A key area of uncertainty is passenger car emissions, with the two biggest concerns being real world NO_x emissions from diesel passenger cars and the emissions factors assumed by air quality models.

In depth analysis is then performed to quantify these risks. First by a PEMS study of Euro 6 diesel cars, followed by a comparison with the emissions factors of the recommended air quality transport model for the EU, COPERT. Scenario analysis is then performed, using the UKIAM, to assess the risk posed by these real world emissions factors and the uncertainty surrounding them.

Following this analysis, the potential risk of increased CO_2 emissions if diesel vehicles were phased out of the UK fleet is assessed. Again, this risk is quantified by a PEMS study which this time includes; Euro 5, petrol and hybrid vehicles. Finally, the risk of increased CO_2 emissions is weighed up against the risk of not reducing NO_x emissions using a cost benefit analysis.

1.5 Structure of thesis

The research presented in this thesis is divided into 7 chapters. Chapters 1 – 3 introduce the research objectives, methodologies and approach taken, provide the necessary background and frame the research. Chapters 4 – 6 are the main results chapters, each contains additional background and literature review specific to the analysis presented in each chapter, as well as a discussion and summary. Finally, Chapter 7 includes discussion, conclusions and a summary of the work presented in the previous chapters.

Chapter 1. Introduction

Chapter 1 provides a brief overview of the thesis and the rationale behind it. The Introduction identifies the key topics the thesis aims to address and outlines the context, aims and structure.

Chapter 2. Background and Literature Review

Following on from the Introduction, Chapter 2 expands on the necessary background information in more detail, much of which is of a technical nature. Chapter 2 also uses a review of relevant academic literature to position the research within the discipline of air pollution (specifically passenger car emissions) and acknowledge the work that has come before.

Chapter 3. The HAZOP approach

Chapter 3 serves a dual purpose. Firstly, it presents Hazards and Operability (HAZOP) as a framework to identify uncertainties in complex systems. Secondly, a case study is outlined using the UKIAM through which an overview and analysis of the UKIAM is presented. Chapter 3 identifies the key areas that will be investigated further in the following chapters, the most important of which is NO_x emission factors from Euro 6 diesel passenger cars.

Chapter 4. NO_x emissions from Euro 6 diesel passenger cars and comparison with COPERT

Chapter 4 is the first core results chapter. It aims to address the most pressing uncertainty identified in the previous chapter (NO_x emissions factors from diesel passenger cars). It includes a study into the real world NO_x and NO_2 emissions from 39 Euro 6 diesel passenger cars measured using PEMS. This chapter highlights the variability of real world emissions, it also reveals emissions much higher than type approval limits and COPERT emissions factors. Real world emissions factors for Euro 6 diesel urban and motorway driving are derived and set in the context of existing literature.

Chapter 5. Scenario analysis of Euro 6 diesel NO_x emissions for 2030

Chapter 5 uses the emissions factors from the previous chapter along with existing literature to develop 5 scenarios for 2030. Each scenario assumes a different evolution of real world diesel Euro 6 emissions factors. Modelling is performed by the UKIAM and the outputs compared include total NO_x emissions in tonnes, damage costs and changes in roadside concentrations of NO_2.

Chapter 6. CO_2 and NO_x emissions from diesel and petrol passenger cars

Chapter 6 extends the scope of the research to include CO_2 and CO as well as petrol and hybrid Euro 5 and 6 passenger cars. This adds important context as diesel vehicles were promoted to reduce CO_2 comparative to petrol. The air quality / climate change trade-off is the focus of this chapter.

Chapter 7. Summary and Discussion

Chapter 7 draws together the research and resets it in the body of existing literature. Results are linked with research aims and the limitations of the work are discussed. This will build on discussion sections included at the end of each of the core results chapters (Chapters 4 – 6). The final chapter pulls out the key conclusions and lessons from the research and identifies areas in which further work is required. Chapter 7 illustrates how the research questions have been answered and aims met.

Chapter 2. Background and literature review

This chapter provides an overview of the general, technical and policy background relevant to this research project as well as a review of existing academic literature. It starts with background relating to air pollution in the EU and the statutes that regulate it. The scope then narrows to more technical background specific to passenger car emissions. The background information presented here draws on existing literature and frames this research within the existing body of academic work. Later chapters also contain additional background sections which include content specific to that section of the research.

2.1 Introduction to air pollution

The World Health Organisation (WHO) defines air pollution as "*contamination of the indoor or outdoor environment by any chemical, physical or biological agent that modifies the natural characteristics of the atmosphere*". Air pollution can be divided into two main categories; pollutants that impact air quality, causing environmental damage on a local scale, and pollutants that impact the global climate. The first group are known commonly as air quality pollutants, the second as greenhouse gases. Though both often stem from the same sources and there is overlap between the two, policy makers have historically tackled air quality and climate change separately. For example, in the UK air quality policy is the remit of the Department for Environment, Food and Rural Affairs (DEFRA) whereas climate change is the remit of the Department for Business, Energy and Industrial Strategy (BEIS, which absorbed the Department of Energy and Climate Change (DECC) in 2016).

Transport is a significant source of both air quality and climate change emissions and there are often trade-offs between the two. The most relevant example of this is diesel and petrol emissions. Diesel vehicles have lower greenhouse gas emissions, but higher emissions of air quality pollutants. The reverse is true for petrol. The following section provides an overview of both air quality and climate change emissions from transport, set in the context of the UK.

2.1.1 Air quality pollutants

The WHO describes air pollution as "*world's largest single environmental health risk*". This is because air pollution is extremely detrimental to human and environmental health. The WHO estimated in their global burden of disease report that indoor and outdoor air pollution carry responsibility for approximately one in every nine deaths (WHO, 2016). In the UK ~40,000 premature deaths per year are linked to air pollution (RCP, 2016) as well as increased hospital admissions due to respiratory and cardiovascular disease (Brunekreef & Holgate, 2002). In Europe road transport is the dominant source of urban air pollution. This is because the vast majority of road transport vehicles still rely on internal combustion of fossil fuels. Many dangerous air quality pollutants are by-products of this combustion process. The pollutants with the greatest impact on European public health are nitrogen dioxide (NO_2), particulate matter (PM) and ozone (O_3) (RCP, 2016). The European Environment Agency estimated in 2013 that these pollutants were responsible for over half a million premature deaths in Europe: 467,000 from PM; 71,000 from NO_2; and 17,000 due to O_3 (EEA, 2016a).

2.1.1.1 Nitrogen oxides (NO_x)

NO_x refers to the combination of two oxides of nitrogen: nitric oxide (NO) and nitrogen dioxide (NO_2). NO_x is the main air quality pollutant considered in this research. Both NO and NO_2 are toxic gases but NO_2 has five times the toxicity of NO. NO_2 irritates lung tissue and causes inflammation of the airways. Long term exposure can reduce lung function and increase the chances of respiratory diseases such as lung cancer and COPD (Hamra *et al.*, 2015; Adam *et al.*, 2015; Gauderman *et al.*, 2002; DeNicola, Rebar & Henderson, 1981).

The NO_2 that is formed during combustion and emitted though a vehicles tailpipe is known as primary NO_2 (sometimes called fNO_2). NO_2 can also be formed by the oxidation of NO in the atmosphere, this is known as secondary NO_2. Given time and well mixed air NO and NO_2 will settle in an equilibrium ratio.

NO_x is of particular concern not only due to the direct health effects associated with its inhalation but also because once in the atmosphere it reacts to form tropospheric ozone (the main component of smog) and ammonium nitrate. NO_x also leads to secondary particulate formation. It is difficult to apportion health effects between NO_2 and $PM_{2.5}$ (see below for definition) exposure because both occur in the same locations, most studies assume an overlap in health effects of ~30% (Walton *et al.*, 2015). An added complication is due to the chemical coupling of O_3 and NO_x, ambient concentrations of NO_2 do not respond linearly to emissions of NO_x (Derwent, 1995). This causes difficulties when modelling NO_2 concentrations, as discussed later in this chapter.

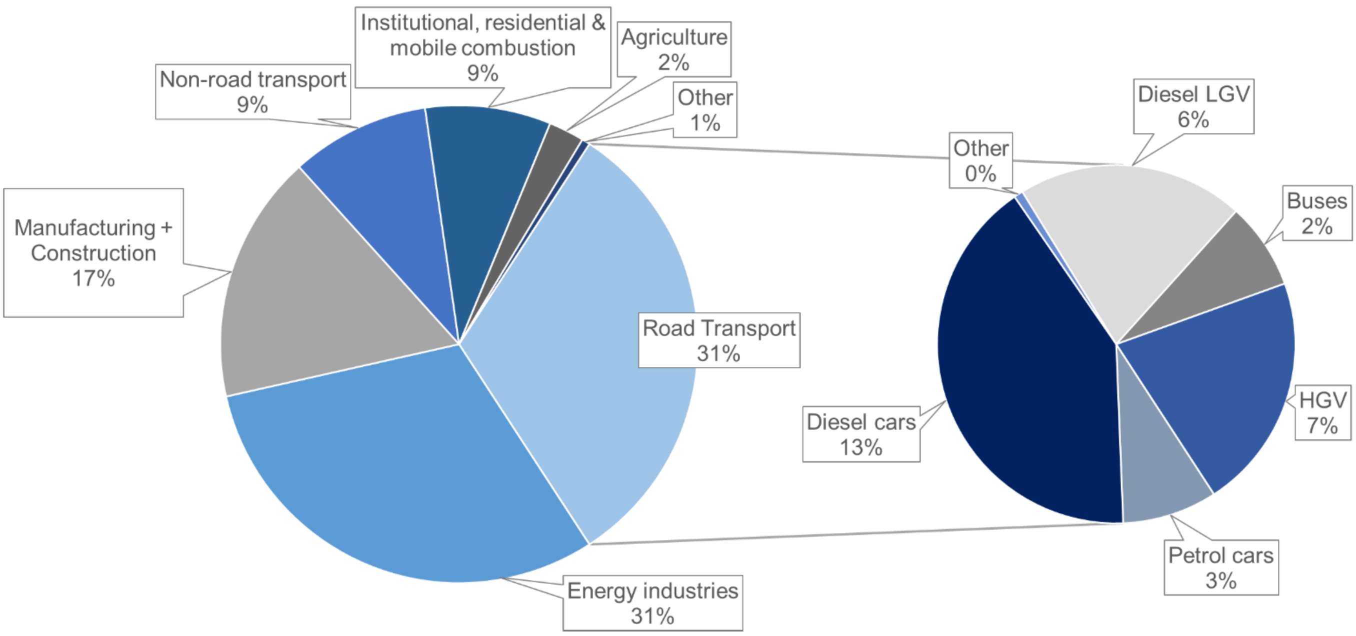

Figure 2-1. UK source apportionment of NO_x (NAEI, 2014a)

Figure 2-1 shows the source apportionment of NO_x by sector in the UK (2014) according to the National Atmospheric Emissions Inventory (NAEI). The total UK NO_x emission was 957 kilotons. Almost one third of all NO_x emissions came from road transport, another third from energy production and the rest from various combustion and industrial sources.

In December 1952 London suffered a severe air pollution episode, since dubbed 'The Great Smog'. At the time the government estimated the smog cost the lives of 4,000 Londoners, but a recent review of the evidence revised this figure up to 12,000 (Bell & Davis, 2001). In reaction to 'The Great Smog' the government passed the first Clean Air Act in 1956 which included measures to relocate power stations outside cites, away from human populations. As a result, today NO_x emissions from energy generation, though similar in quantity to NO_x from road transport, pose much less of a threat to human health. This is because for air quality pollutants the location of the emission determines the magnitude of population exposure. Modern power stations have sufficient stack heights and remote locations to reduce human exposure to the emissions they produce. In contrast transport emissions are often emitted in urban centres where population exposure is highest.

According to the NAEI, of the 31% of UK NO_x emissions attributable to road transport the vast majority came from diesel fuelled vehicles, with 13% from diesel passenger cars alone. Nearly all buses and Heavy Goods Vehicles (HGVs) are also fuelled by diesel. **Figure 2-1** illustrates the magnitude of diesel emissions in the UK, however, it is likely the NAEI underestimated the diesel contribution to NO_x. A recent study found measurements of NO_x in London at sites where traffic was the dominant source were 80% higher than estimated using the NAEI emissions factors (Lee *et al.*, 2015). This was largely attributed to an underestimate of emissions from the diesel vehicles in the fleet. Addressing NO_x emissions from diesel vehicles is a top priority in tackling urban air pollution and this priority is reflected in this research. Petrol and diesel cars in the UK account for approximately the same amount of vehicle kilometres (VKM) annually, yet by the NAEI estimate diesel accounted for 4.3 times as much NO_x emissions. This discrepancy between petrol and diesel car emissions is another focus of this research.

Ambient concentrations of NO_x and NO_2 have decreased in recent years, though at a slower rate than expected, with emissions plateauing around in the early 2000's. **Figure 2-2** comes from a paper by *Carslaw, Murrells, Andersson, et al (2016)*. The study looked at trends in annual average NO_x and NO_2 concentrations from 35 roadside monitoring stations in London from 1996 to 2014.

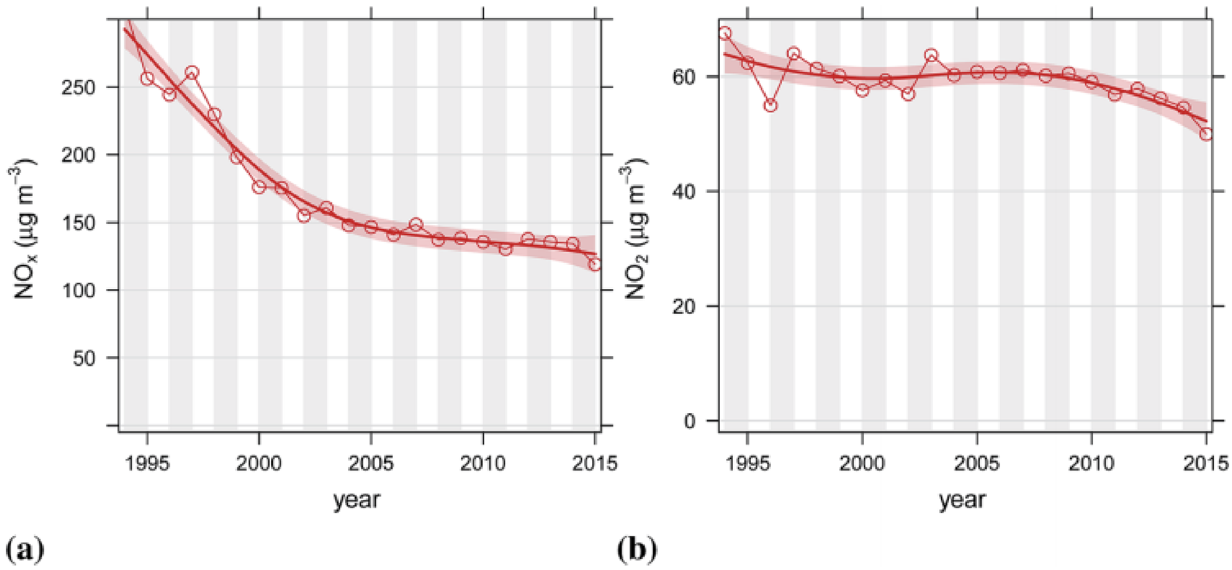

Figure 2-2. (a) Trends in the mean concentration of NO_x across 35 roadside sites in Greater London with at least 10 years of data capture and (b) the same for NO_2 (Carslaw *et al.*, 2016)

The plateauing of emissions has been attributed to the failure of Euro 3 – 5 legislation to reduce real world NO_x emissions from diesel vehicles (see below for further detail). Additionally *Carslaw et. al* found NO_x concentrations had on average reduced by 2.4% per year over the period measured, whereas NO_2 concentrations had only decreased by 0.4% per year. This was attributed to the growing proportion of diesel NO_x emissions being emitted as primary NO_2.

As mentioned earlier, given adequate time and well mixed air atmospheric NO_x concentrations will settle into an equilibrium ratio of NO to NO_2 by reacting with O_3. However, at urban roadside locations where there is little time between source and exposure, and where much of the O_3 is already depleted the proportion of NO_x emitted as primary NO_2 becomes an important factor in ambient concentrations (Grice *et al.*, 2009; Carslaw, 2005; Degraeuwe *et al.*, 2015). This is why research into real world emissions of primary NO_2 from diesel vehicles is important in understanding and developing policies that reduce ambient concentrations of NO_2. As such it is one of the key considerations of this thesis. The technological reasons why fNO_2 emissions from diesel vehicles have been increasing are discussed later in this chapter.

2.1.1.2 Particulate Matter (PM)

Particulate matter refers to a complex mixture of particulates in the air including dust, dirt, smoke and liquid droplets. It and can be anthropogenic or natural. PM is categorised according to its aerodynamic diameter. Particles with a diameter < 10 μm are referred to as PM_{10} and those between 2.5 μm - 10 μm as the “coarse” fraction. These include particles such as dust from construction, pollen and mould which are small enough to be inhaled and accumulate in the respiratory system. Particles with a diameter < 2.5 μm are referred to as $PM_{2.5}$ or “fine” particulates. These include combustion particles, organic compounds and metals. $PM_{2.5}$ pose a greater health risk than the coarse fraction as the particles are smaller and can reach deeper in the lungs.

An important component of $PM_{2.5}$ from diesel vehicles is black carbon or "soot". Black carbon is pure carbon produced by incomplete combustion. Inhalation of black carbon is associated with cardiopulmonary morbidity and mortality (WHO, 2012). Black carbon is also a climate forcer and is thought to be the second most important contributor to global warming after CO_2 (Schmidt & Noack, 2000; Jacobson, 2001).

Particulates less than 0.1 μm are known as ultra-fine particles. These commonly include carbon based and metallic particulates. There is a growing body of evidence linking ultra-fine particles to the most severe health effects of PM exposure (Miller *et al.*, 2017; Oberdorster, Oberdorster & Oberdorster, 2005; Peters *et al.*, 1997). This is because ultra-fine particles are so small they can penetrate the lung tissue and be absorbed into the bloodstream. Once in the body they are not easily removed and have been found to accumulate in the heart and brain.

In Europe urban particulates are mainly attributable to road traffic (Harrison, Smith & Luhana, 1996; Masiol *et al.*, 2012) with ~80% of respirable PM_{10} in cities coming from road traffic sources (Bencs *et al.*, 2010). It used to be the case that PM emissions were much higher from diesel vehicles than from petrol vehicles; however, since 2009 all diesel vehicles have been fitted with a Diesel Particulate Filter (DPF). DPFs have significantly reduced the exhaust particulate emissions of diesel vehicles, including emissions of ultra-fines, reducing the number and mass of primary particles by up to 99% (Bergmann *et al.*, 2009a; Liu *et al.*, 2005). However, where there is a high sulphur content of fuels (e.g. in China and India) the DPF can result in an increase in formation of secondary nucleation mode particles (Kumar *et al.*, 2014).

PM from road traffic can be broadly split into two categories: exhaust and non-exhaust. Non- exhaust emissions include resuspension of particles and road, brake, tyre and clutch wear. In the early 2000's it was estimated the levels of exhaust and non- exhaust particulates in urban environments were approximately equal (Querol *et al.*, 2004; Lenschow, 2001). Recent studies have found that as levels of exhaust particulates have fallen (with the introduction of particulate filters) the proportion of PM from non- exhaust emissions has increased. A 2016 study in the Hatfield tunnel (north of London) found 60% of PM_{10} was attributable to non- exhaust emissions (Lawrence *et al.*, 2013).

The research presented in this thesis relates only to exhaust emissions. The measurement study which provided estimates for real world emissions used a Portable Emissions Measurement System (PEMS) that did not record PM. Academic literature is largely in agreement that DPFs have successfully solved the problem of exhaust PM emissions from diesel cars (May *et al.*, 2014; Mathis, Mohr & Forss, 2005; Bergmann *et al.*, 2009b). This is why the main focus of this research is NO_x emissions, where there is still large uncertainty and discrepancy between diesel and petrol vehicles. However, PM is an extremely important element of urban air pollution and will be kept in consideration as part of the wider context of this work.

2.1.1.3 Ozone (O_3)

Stratospheric ozone is essential for life on earth; it absorbs the majority of harmful ultraviolet rays from the sun. In contrast tropospheric (ground level) ozone is a toxic atmospheric pollutant and enhanced levels are detrimental to human and environmental health (though some O_3 near ground level is needed to produce the hydroxyl radical (OH) and breakdown pollutants). Tropospheric ozone is formed when NO_x and hydrocarbons (HC) from combustion processes such as energy generation and transport react with volatile organic compounds (VOCs) in the presence of sunlight to form ozone. This means control of NO_x is key in reducing ambient levels of ozone (Derwent *et al.*, 2003).

As mentioned previously ozone is a key component of smog. The reaction that forms smog is dependent on sunlight; this is why smog is more common on sunny days. Ozone exposure contributes to poor cardiopulmonary health and mortality and can be a contributing factor in pneumonia, chronic obstructive pulmonary disease and asthma (Ebi & McGregor, 2008). Ozone also effects forests, crops and ecosystems.

2.1.2 Climate change pollutants (Greenhouse gases)

The main greenhouses gases emitted by the UK are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and fluorinated gases. Of these, CO_2 is the most common by far, making up 81% of all 2014 greenhouse gas emissions. Greenhouse gases are collectively measured in megatons of CO_2 equivalent ($MtCO_2e$).

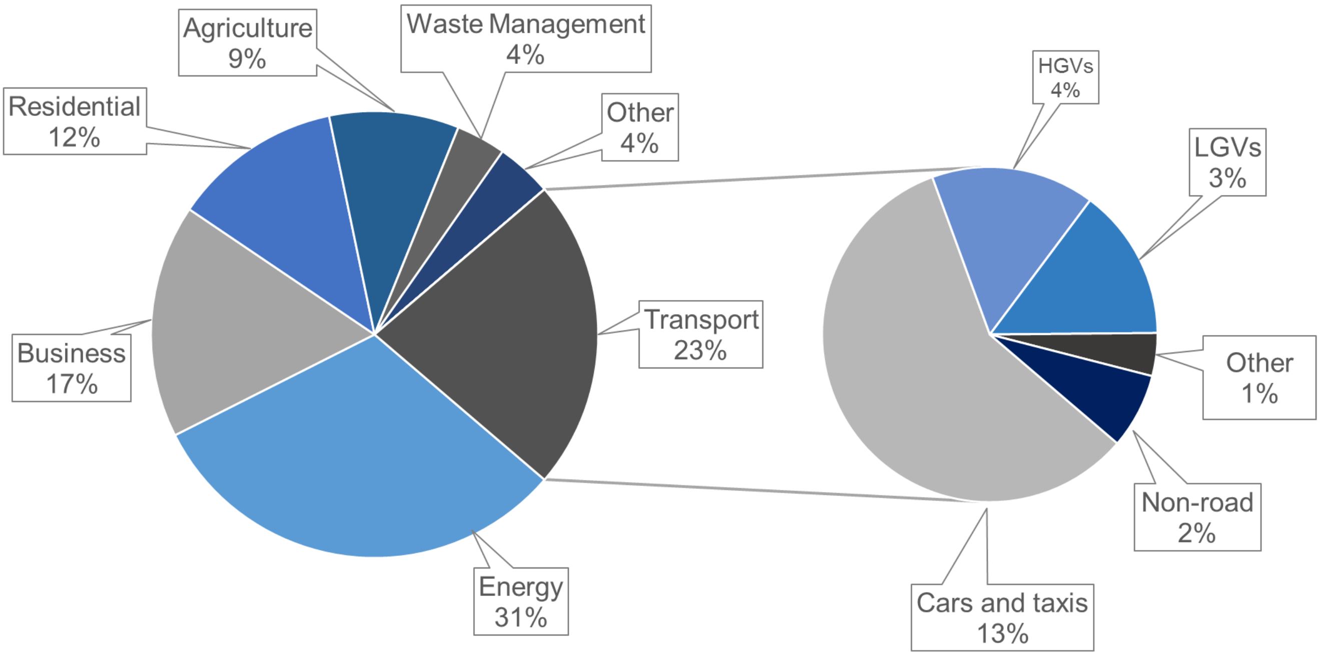

Figure 2-3. UK source apportionment of greenhouse gases for 2014 (DfT, 2015a; DECC, 2016)

Figure 2-3 shows total UK greenhouse gas emissions in 2014 by sector according to DECC (now BEIS). The total emission was 514.4 $MtCO_2e$. Transport (road and non-road) accounted for 23% of all greenhouse gas emissions, compared to 40% of total UK NO_x. Energy generation accounted for a similar proportion of NO_x as greenhouse gas emissions.

As discussed, CO_2 is the main greenhouse gas but NO_2, O_3 and black carbon also contribute to climate change and are emitted at higher rates from diesel vehicles. However, it should also be noted that secondary PM can also have a cooling effect in the atmosphere by reflecting radiation (Fuzzi *et al.*, 2015).

2.1.2.1 Carbon dioxide (CO_2)

CO_2 is a naturally occurring colourless odourless gas that is essential to life on earth. Plants require CO_2 for the process of photosynthesis which produces glucose; this process sustains plant, animal and human life. Since the industrial revolution humans have been burning fossil fuels at an unprecedented rate. This has increased the concentration of CO_2 in the earth's atmosphere. Greenhouse gases are so called because they act like the glass in a greenhouse, trapping heat below the atmosphere. This is known as global warming. The effects of global warming on natural systems are already evident through multiple indicators including changing precipitation patterns, changing migratory patterns and ocean acidification (IPCC, 2014). More difficult to predict are the potentially catastrophic consequences of continued warming. Through initiatives such as the Kyoto protocol and Paris climate accord an international consensus has consolidated around the need for greater mitigation of climate change.

Due to an increase in number of passenger cars on the roads and vehicle kilometres driven by each vehicle, transport is currently the only major sector in the EU for which CO_2 emissions continue to rise (CCC, 2015; Fontaras, Zacharof & Ciuffo, 2017). It is essential that future policies regarding air quality do not have negative impacts on climate change objectives. This thesis therefore includes detailed analysis of CO_2 emissions from both diesel and petrol cars as well as NO_x from diesel cars.

2.2 Air pollution regulation

There are three main bodies of regulation that govern the emissions of air quality pollutants in the European Union (EU): the Euro Standards, the National Emissions Ceiling Directive and the Ambient Air Quality Directive.

2.2.1 Euro Standards

The Euro Standards are the legislation that sets statutory limits for exhaust emissions from road transport vehicles in the EU. Limits are set in terms of grams of pollutant emitted per kilometer driven (g km^{-1}). The first European legislation regarding emissions from passenger cars was passed in 1970 setting legal limits for carbon monoxide and unburned hydrocarbons (EEC, 1970). In 1977 an amendment added the first limit for NO_x. These limit values were successively reduced (Directives 78/665/EEC, 83/351/EEC and 88/76/EEC) and in 1988 a limit value was introduced for particulates from diesel engines (Louka, 2004; Tiwary & Colls, 2010). Since 1988 successively tighter regulations, known as the Euro Standards, have been enacted. These are Euro 1- 6 for passenger cars and light duty vehicles and Euro I-VI for heavy duty vehicles. Successive Euro standards have extended the number of pollutants regulated and reduced limit values. Petrol and diesel vehicles are subject to different limit values for some pollutants. The emission limits and date of implementation for Euro 1-6 are listed in **Table 2-1** below.

The current Euro standards set limits for carbon monoxide (CO), total hydrocarbons (THC), non-methane hydrocarbons (NMHC), hydrocarbons and oxides of nitrogen (THC + NO_x), PM, number of particles (PN) and NO_x. European legislation does not differentiate between NO and NO_2, the type approval only relates to total NO_x.

Table 2-1. Emission Limits for M_1 (passenger cars)

Engine type	Date (new)	Date (all)	CO [g km^{-1}]	THC [g km^{-1}]	NMHC [g km^{-1}]	(THC + NO_x) [g km^{-1}]	PM [g km^{-1}]	PN [# km^{-1}]	NO_x [g km^{-1}]
Euro 1 (Directive 91/441/EEC ((EEC, 1991))									
Petrol	1992	1993	2.72	-	-	0.97	-	-	-
Diesel			2.72	-	-	0.97	0.14	-	-
Euro 2 (Directive 94/12/EC (EEC, 1994))									
Petrol	1996	1997	2.2	-	-	0.5	-	-	-
Diesel			1.0	-	-	0.7	0.08	-	-
Euro 3 (98/69/EC (EC, 1998))									
Petrol	2000	2001	2.3	0.2	-	-	-	-	0.15
Diesel			0.64	-	-	0.56	0.05	-	0.5
Euro 4 (98/69/EC (EC, 1998))									
Petrol	2005	2006	1.0	0.1	-	-	-	-	0.08
Diesel			0.5	-	-	0.3	0.025	-	0.25
Euro 5 (EC 715/2007 (EC, 2007))									
Petrol	2009	2011	1.0	0.1	0.068	-	0.005*	-	0.06
Diesel			0.5	-	-	0.23	0.005	-	0.18
Euro 5b (EC 692/2008 (EC, 2008))									
Diesel	2011	2013	0.5	-	-	0.23	0.0045	6×10^{11}	0.18
Euro 6b (EC 459/2012 (EC, 2012))									
Petrol	2014	2015	1	0.1	0.068	-	0.0045*	6×10^{11}	0.06
Diesel			0.5	-	-	0.17	0.0045	6×10^{11}	0.08

*applies to gasoline direct injection (GDI) only

2.2.1.1 Type approval

Before being sold within the EU single market new vehicle models must pass a type approval test. Standardised tests assure that new models adhere to EU environmental, safety and conformity of production requirements (EC, 2016b). These tests are performed by privately owned technical service providers but the ultimate decision to approve a vehicle lies with national type approval authorities. Currently emissions are tested in a laboratory on a chassis dynamometer (rolling road). The standardised test cycle used to assess vehicle emissions is called the New European Driving Cycle or NEDC. The Urban Driving Cycle (UDC/ECE-15) was introduced in 1970 with the first European emissions legislation. The Extra Urban Driving Cycle (EUDC) which includes more aggressive driving was added in 1990. The NEDC consists of four UDCs and one EUDC as shown in **Figure 2-4**.

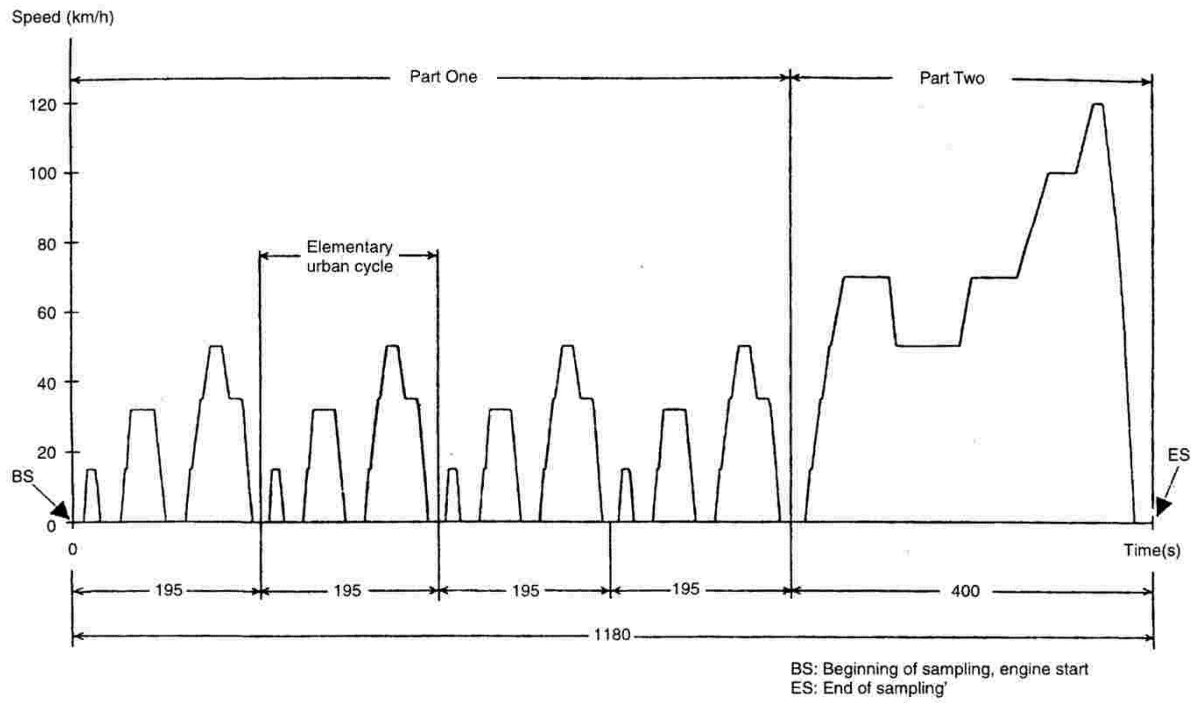

Figure 2-4. Speed profile of NEDC (EC, 1998)

To pass the type approval test new models must have an average emission in g km^{-1} below the limit values stated in **Table 2-1**. Limit values are only legally binding during this type approval test, there is currently no legal consideration of how a vehicle performs outside of this laboratory test. Since the late 90's the NEDC has been criticised for not being representative of real world emissions (Kågeson, 1998; Williams & Carslaw, 2011). These are often referred to as real driving emissions (RDE) or real world driving. Whilst RDE exceeding type approval limits is potentially hazardous for human health it is not illegal under current legislation.

To ensure repeatability strict rules regulate the type approval process, these rules are detailed in UNECE Regulation No. 83 (UNECE, 2015). Some of these controls contribute to the unrealistic emissions from vehicles during type approval. For example, the test must be performed at an ambient temperature between 20 – 30°C, whereas the average ambient temperature in the UK is 9°C. Low ambient temperatures increase NO_x emissions because emission controls are often switched off at lower temperatures to protect the engine (DfT, 2016d; Kwon *et al.*, 2017). Engines are optimised for the NEDC cycle and vehicles are stripped back and streamlined. Though these so called "golden vehicles" produce emissions much lower emissions than in the real world, they operate within the current rules, unlike defeat devices.

2.2.1.2 Introduction of RDE type approval

To address the discrepancy between NEDC assessments and real driving emissions the EU is introducing a real driving component to the passenger car type approval test.

Table 2-2. Future emissions limits include RDE test component

Engine type	Date (new)	Date (all)	CO [g km^{-1}]	THC [g km^{-1}]	NMHC [g km^{-1}]	(THC + NO_x) [g km^{-1}]	PM [g km^{-1}]	PN [# km^{-1}]	NO_x [g km^{-1}]
Euro 6d- TEMP (EC 459/2012)									
Petrol	2017	2019	1	0.1	0.068	-	0.0045*	6×10^{11}	0.06
Diesel			0.5	-	-	0.17	0.0045	6×10^{11}	0.168
Euro 6d									
Petrol	2020	2021	1	0.1	0.068	-	0.0045*	6×10^{11}	0.06
Diesel			0.5	-	-	0.17	0.0045	6×10^{11}	0.120

The test will be carried out using a PEMS and will consist of urban, motorway and rural sections. However, responding to pressure from the motor manufacturing industry the European Commission have allowed a so called "conformity factor" to be applied to the diesel type approval limit. In practice this means up to 2020/21 new diesel cars will be permitted to emit 2.1 times the current type approval limit in the new RDE test. From 2020 the conformity factor will be reduced to 1.5 times the current limit. The potential impact of the new RDE procedure is discussed further in Chapter 5. Detailed characteristics of the RDE testing regime are presented in Chapter 6.

2.2.1.3 Deviation Ratio

The factor by which real-world emissions exceed the relevant limit value is known as the deviation ratio or conformity factor. "Conformity factor" also has an alternative definition relating to the new RDE type approval procedure as stated above. Therefore to avoid confusion this thesis uses the term deviation ratio. The deviation ratio is calculated using **Equation 2-1**.

Equation 2-1. Deviation ratio

$$\mathbf{DR_i} = \frac{\mathbf{m_i}/\mathbf{s_i}}{\mathbf{ES}}$$

DR_i = deviation ratio of trip for pollutant *i*

m_i = mass of pollutant *i* emitted over trip in *g*

s_i = distance of trip

ES = emission standard in g km^{-1}

For diesel vehicles the deviation ratio has been steadily increasing as type approval limits have become more stringent. This is illustrated in **Figure 2-5.**

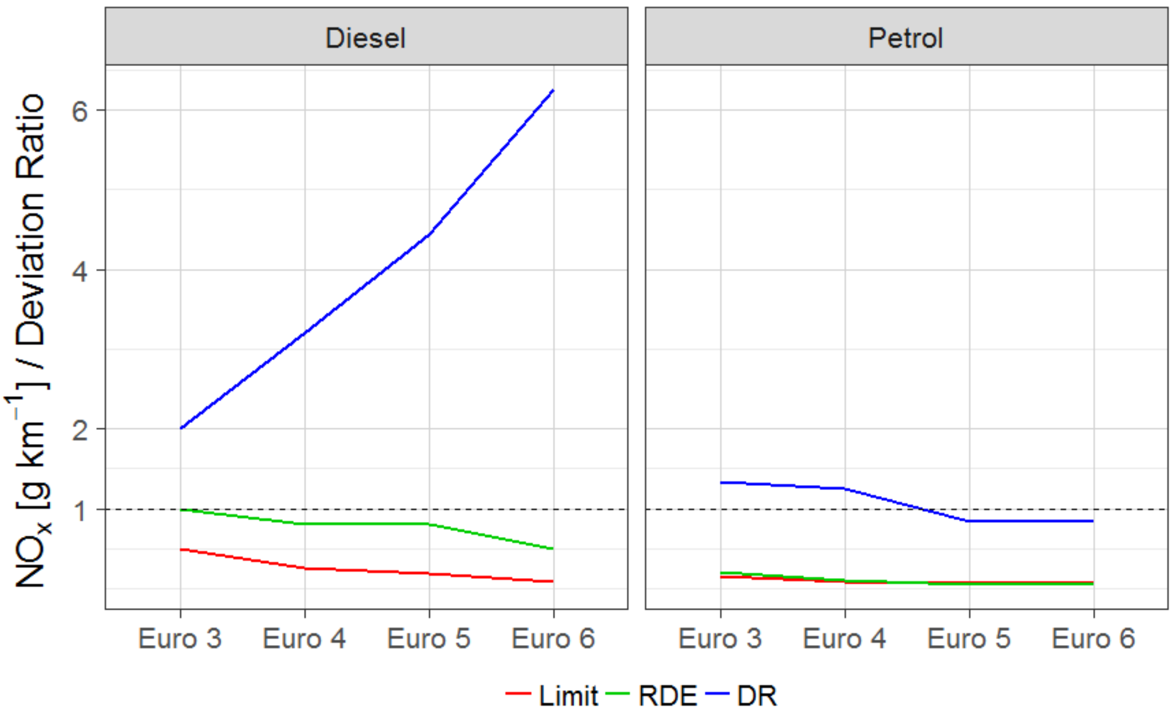

Figure 2-5. Type approval limit, RDE estimate and deviation ratio (ICCT, 2016a, 2012)

Figure 2-5 shows the type approval limits (red), estimate of real world emissions (green) and estimated deviation ratio (blue) for Euro 3- 6 diesel and petrol passenger cars. The dashed horizontal line represents the point at which the deviation ratio is equal to type approval limit. A deviation ratio < 1 means real world emissions is below the relevant type approval limit, deviation ratio > 1 means RDE is above the type approval limit. Since Euro 5 petrol cars have had real driving emissions lower than their type approval limit. In contrast whilst diesel real world emissions has fallen, it has not fallen by as much as the type approval limit and the deviation ratio has steadily increased to approximately 6 for Euro 6 diesel. The International Council on Clean Transport (ICCT) calculated these real world estimates from PEMS measurements and remote sensing data. They estimated NO_x emissions from Euro 6 diesel cars were 10 times higher than Euro 6 petrol cars.

Until recently it was assumed that the difference between real world emissions of diesel passenger cars and NEDC emissions was entirely due to the unrepresentative nature of the test. The 2015 Volkswagen emissions scandal called this into question. A recent study found that the limited driving conditions of the NEDC could account for no more than 20% of the discrepancy between real world and type approval emissions (Degraeuwe & Weiss, 2017). Real world emissions of diesel vehicles are often many times higher than type approval limits. There is no such increase for petrol cars and no satisfactory explanation as to why this should be the case.

2.2.1.4 Defeat devices ("Dieselgate" Scandal)

Whilst the behaviour of manufacturers described above is undesirable, it is not illegal. Defeat devices, like those installed in 11 million Volkswagen vehicles between 2008 and 2015, are illegal. The energy consumed running emissions controls reduces fuel economy, increasing the cost of running a vehicle. Defeat devices are used to cheat the type approval test in order to deliver real world fuel economy benefits that are appealing to consumers. A defeat device is defined in Article 3 (10) of Regulation (EC) No 715/2007 as;

"any element of design which senses temperature, vehicle speed, engine speed (RPM), transmission gear, manifold vacuum or any other parameter for the purpose of activating, modulating, delaying or deactivating the operation of any part of the emission control system, that reduces the effectiveness of the emission control system under conditions which may reasonably be expected to be encountered in normal vehicle operation and use"

In 2015 the US EPA discovered that Volkswagen had written a "switch" code into their diesel vehicles electronic control module (ECM). The "switch" identified type approval conditions by vehicle behaviour (e.g. position of steering wheel, atmospheric pressure). When type approval conditions were identified the ECM fully implemented the NO_x emissions controls. However, at all other times (i.e. when the vehicles were being used in the real world) emissions controls were only partially implemented, resulting in higher NO_x emissions. This is an example of a "cycle detection" defeat device (T & E, 2016). A recent study found 1,200 early deaths in Europe, each losing as much as a decade of life, were attributable to the presence of the defeat device in Volkswagen cars sold in Germany alone (Chossiere *et al.*, 2017).

Though Volkswagen was the first company to admit to the use of a defeat device, evidence points to widespread use. The German Federal Motor Transport Authority (KBA) has accused Fiat of installing certain models with a defeat device that turns off emissions controls after the first 22 minutes of a journey (the NEDC is 20 minutes long). Other types of defeat device are thought to be in use, the most common being the “thermal window”. This exploits the rule that allows emissions controls to be disengaged at certain temperatures to protect the engine. It is thought many manufacturers disengage emission controls at much higher temperatures than necessary. For example, Opel (Vauxhall) and Renault- Nissan until recently reduced emission controls below 17°C, far above the UK average temperature of 9°C (T & E, 2016).

Another defeat device, known as the “hot restart” is thought to only fully engage emission controls after a cold start. Cold start is when the engine has cooled to ambient temperature before it is switched on. The type approval procedure stipulates engines must be “soaked” (rested) overnight to ensure the test is performed after a cold start. Independent testing in Europe has found many vehicles produce higher emissions after a hot restart than a cold start. A UK government report found 32 out of 38 diesel vehicles tested had higher emissions after hot restart then a cold start (DfT, 2016d). Similarly the German government found the same phenomenon for 48 out of the 53 vehicles they tested (BMVI, 2016).

These results indicate a defeat device is present that activates when an engine is started from a cold start. During a cold start the engine temperature is too low for NO_x reduction technologies to be effective, therefore NO_x emissions should be higher. Indeed, in the USA where the type approval procedure includes both a cold start and a hot restart in a sample of 30 diesel vehicles not a single one had hot restart emissions higher than cold start emissions (ICCT, 2016b). In contrast over 80% of vehicles tested in the UK and Germany measured higher emissions after a hot restart

(DfT, 2016d; BMVI, 2016). Manufacturers have provided no explanation for lower emissions after a cold start that makes engineering or physical sense. This has led to accusations of widespread use of defeat devices.

2.2.1.5 CO_2 fleet average target

The NEDC also regulates fuel consumption and CO_2 emissions for new vehicles. Unlike air quality pollutants the legal CO_2 limit applies to a manufacturer's fleet, not an individual model. There is a fixed "target" that the fleet average of each manufacturer must fall below. Manufacturers are permitted to achieve this by having lighter vehicles with lower CO_2 and heavier vehicles with higher CO_2. Individual vehicles are supposed to fall within a limit curve that is proportional to the vehicles weight (Error! Reference source not found.).

Equation 2-2. CO_2 specific emissions target (EC, 2009a)

$$CO_2 = Target + a(M - M_0)$$

Target = target fleet average
a = gradient of line
M = mass of vehicle
M_0 = average mass of new passenger cars in the previous 3 years

Table 2-3. Parameters for the 2015 and 2020 CO_2 specific emissions limit curve

	2015	**2020**
Target	130	95
a	0.0457	0.0333
M_0	1372 kg (2012-15)	1392.4 kg (2016)

The fleet average target introduced in 2015 was 130 g CO_2 km^{-1}. This will be reduced to 95 g CO_2 km^{-1} in 2020. The parameters used to calculate the limit curve are listed in Error! Reference source not found. and the curves are plotted in **Figure 2-6.**

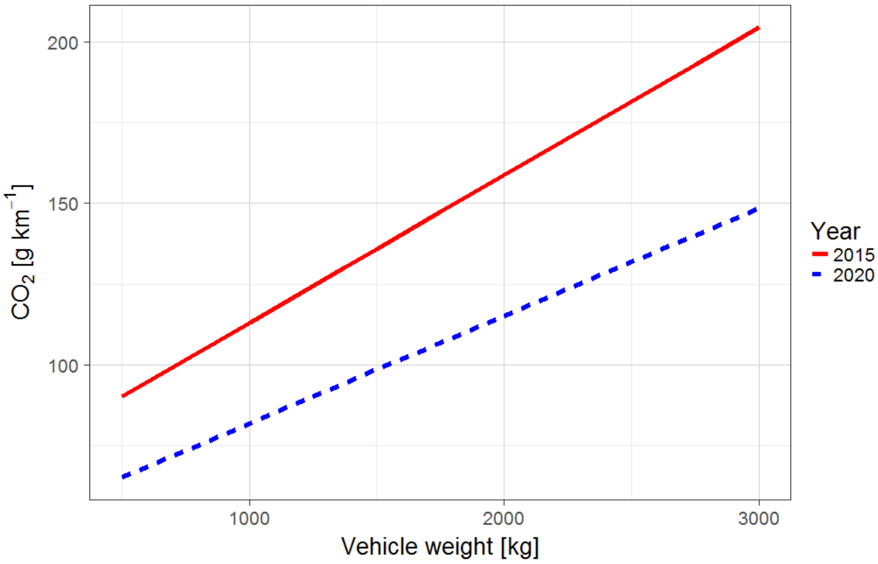

Figure 2-6. 2015 and 2020 CO_2 specific emissions target curve by weight

As discussed, there is substantial evidence of growing discrepancies between real world NO_x emissions and type approval limits (Franco *et al.*, 2014; Weiss *et al.*, 2011a; Carslaw *et al.*, 2011b; O'Driscoll *et al.*, 2016; Kågeson, 1998). The same phenomenon has also been observed for CO_2, with the gap between type approval and real world emissions growing from 8% in 2001 to 31% in 2012, and increasing to 40% in 2014 (Fontaras & Samaras, 2010; Fontaras *et al.*, 2014; T & E, 2015). This will be discussed further in Chapter 6.

In addition to the Euro Standards the two main pieces of legislation that regulate emissions of air quality pollutants in the EU are the National Emissions Ceiling Directive and the Ambient Air Quality Directive.

2.2.2 National Emissions Ceiling Directive

The National Emissions Ceiling Directive (NECD) sets legal limits (ceilings) on the emission of pollutants in kilo-tonnes that member states are permitted to emit annually. The amount decreases for successive target years and is measured relative to the baseline year of 2005, and is specified as a percentage of 2005 emissions. There have been two tiers of legislation, Directive 2001/81/EC which legislated from the period 2010 – 2020 and Directive 2016/2284/EU which legislates from 2020 – 2030. The NECD sets limits for NO_x, $PM_{2.5}$, non-methane volatile organic compounds (NMVOCs), sulphur dioxide (SO_2) and ammonia (NH_3) and has been legally binding since 2010. Compliance is monitored through annual reporting by member states national emission inventories to the European Environment Agency.

The UK met its first emissions ceilings for every pollutant (EEA, 2016b). **Table 2-4** lists the UK's ceilings for NO_x from 2005 to 2030.

Table 2-4. UK National Emission Reduction Commitments (EEB, 2017)

	2005	2010	2013	2020	2030	2030 CLE	2030 MTFR
NO_x reduction relative to 2005 [%]	-	-29%	-35%	-55%	-73%	-72%	-80%
NO_x ceiling [kilo-tonnes]	**1592**	1130.3	1034.8	716.4	429.8	445.8	318.4

CLE = The Current Legislation scenario is a national emissions projection that assumes full implementation of existing EU policies without any additional measures

MTFR = The Maximum Technically Feasible Reductions is a national emissions projection of the maximum emission reduction that could be achieved if all readily available technical measures were implemented.

The UK's latest CLE predicts a 3.7% exceedance of the 2030 national emission reduction commitment. However, it should be noted that the CLE is subject to change and differs depending on the baseline emissions figure used for 2005. It is likely that the value stated in the EEB report used COPERT 4v11 speed dependent emissions factors which would lead to an underestimate in projected UK 2030 NO_x emissions.

2.2.3 Ambient Air Quality Directive (2008/50/EC)

In contrast to the NECD which sets a limit to the total emission of pollutants, the Ambient Air Quality Directive sets Air Quality Limit Values limiting pollutant concentrations in ambient air. Whereas the UK has so far succeeded in meeting the NECD commitments the same cannot be said for the Ambient Air Quality Directive.

Limit values for air quality pollutants were first legislated in the EU Ambient Air Quality Framework Directive (96/62/EC) and fourth Daughter Directive (2004/107/EC). Most recently the Ambient Air Quality Directive (2008/50/EC) set legally binding limits for concentrations of the pollutants SO_2, NO_2, PM_{10}, $PM_{2.5}$, Lead, Benzene and CO. Unlike the NECD that regulates only total NO_x emissions the Ambient Air Quality Directive regulates also for concentrations of NO_2, the more harmful component of NO_x.

For NO_2 there are two Air Quality Limit values, one for hourly mean concentration and one for annual mean concentration. These are listed in **Table 2-5**.

Table 2-5. NO_2 limit values Ambient Air Quality Directive (2008/50/EC)

Averaging period	Limit value	Date by which limit is to be met
One hour	200 μg m^{-3} not to be exceeded > 18 times per calendar year	1 January 2010
One year	40 μg m^{-3}	1 January 2010

Directive (2008/50/EC) allowed that if member states had particular difficulty in achieving compliance of the NO_2 limit value by January 2010 the member state could apply to the Commission and postpone the date of compliance to January 2015 (at the latest) on a zone by zone basis. In 2009 the UK applied for a time extension in 24 zones and was successful (granted the extension) in 9. However, the period of extension has now ceased and even at the time of writing (2017) the UK is still in exceedance in 37 of the 43 reporting zones (DEFRA, 2017a). The UK is also failing to comply with the hourly mean limit. In 2017, Brixton Road in London breached the yearly exceedance allowance (18 exceedances) of the hourly mean limit in just five days (Guardian, 2017).

2.2.4 Climate change legislation

In 2008 the UK passed the Climate Change Act which enacted the commitments of the Kyoto Protocol into UK law (Parliament of the United Kingdom, 2008; UNFCCC, 1998). Explicitly the commitment to reduce greenhouse gases in 2050 by 80% compared with a baseline year of 1990. The Act also created the independent

Committee on Climate Change (CCC). To achieve the goal of an 80% reduction the CCC has set 5 “carbon budgets” which each set an incremental reduction target up to 2050 to keep the UK on track to meet its Kyoto commitment. The UK is currently on track to outperform its 2nd carbon budget which commits to a 31% reduction in GHGs relative to 1990 from (2013 – 2017).

2.3 Recent developments in UK air quality

The rejection in 2009 of the UK government’s time extension bid for 15 of the zones it applied for an extension in was due to lobbying of the European Commission by the environmental law NGO Client Earth.

In April 2015 Client Earth took DEFRA to the Supreme Court challenging DEFRA’s 2011 plan to clean up air quality in the UK, “*Air Quality Plans for the achievement of EU air quality limit values for nitrogen*” (DEFRA, 2011). The plan did not bring the UK into compliance with Directive (2008/50/EC) until 2030, 20 years after the original deadline. Client Earth argued this was unacceptable and ministers should devise a new air quality plan. The Supreme Court unanimously ruled against DEFRA and stipulated a new air quality plan should be produced before December 2015 to bring the UK into compliance as soon as feasibly possible.

In December 2015 DEFRA released a second air quality action plan “*Improving air quality in the UK- Tackling nitrogen dioxide in our towns and cities*” (DEFRA, 2015b). The key policy instrument of the new action plan was Clean Air Zones (CAZ’s) to be introduced in five UK cities. Again the plan was deemed unsatisfactory by Client Earth and the air quality community as a whole. This was partly due to the limited scale of additional measures but also the use of COPERT 4v11 emission factors, which greatly underestimated NO_x emissions from Euro 6 diesel passenger cars (O'Driscoll *et al.*, 2016). This was acknowledged by DEFRA and in the accompanying technical report they stated the COPERT underestimate "*could result in up to 22 additional zones being in exceedance of the NO_2 limit value in 2020*".

Client Earth brought DEFRA back to the High Court in November 2016 and again won the case. The judge ruled the government was not taking sufficient action to bring the UK into compliance "*as soon as possible*". The government was ordered to produce another air quality action plan by May 2017. The additional measures in this latest plan included additional CAZ's to be implemented by local authorities, speed limits on motorways near areas of exceedance and introduction of Euro 6d. At the time of writing Client Earth have described the latest action plan as "*weak*" and indicated they intend to return to the courts.

2.4 Passenger cars in the UK

There were 31 million licensed passenger cars in the UK in 2016. Petrol cars accounted for 59.7%, diesel cars 39.1% with the remainder made up of hybrids (1.0%), gas (0.1%) and electric (0.1%) (DfT, 2016c). The average annual mileage of the diesel vehicles (17,220 km) is 65% higher than that of a petrol vehicle (10,460 km) (DfT, 2015b). As a result, the split in total passenger car vehicle kilometres (VKM) driven is roughly equal. In Great Britain the total VKM in 2015 was 510 billion km, 80% of which was driven by cars and taxis (DfT, 2016b). Additionally petrol and diesel passenger cars are not evenly distributed throughout the road network.

Table 2-6. Diesel and petrol split of GB VKM (NAEI, 2014b; DfT, 2016b)

	Urban	**Motorway**	**Rural**	**Total**
Total passenger car VKM (Billion km)	151.2	80.3	171.9	403.4
Total diesel VKM (Billion km)	70.0	48.7	85.6	204.3
Total petrol VKM (Billion km)	81.2	31.8	85.9	198.9
% of total diesel VKM	34%	24%	42%	
% of total petrol VKM	41%	16%	43%	

The figures in **Table 2-6** are derived from NAEI fleet composition data and Department for Transport (DfT) statistics for 2014. The data shows different behavioural patterns for drivers of petrol and diesel vehicles. Both spend similar proportions of total VKM on rural roads (~43%), but petrol vehicles spend the majority of the remaining VKM (41%) on urban roads with only 16% on motorway. In contrast the urban / motorway split for diesel (34% / 24%) is less substantial.

2.4.1 The difference between petrol and diesel engines

Petrol and diesel vehicles both use petroleum fuel in an Internal Combustion Engine (ICE). However, there are key differences in the chemical makeup of the fuels and engineering of the engines. This section presents a basic overview of these differences and how they result in very different exhaust gas compositions.

Crude oil is subject to a process called fractional distillation by which it is separated into its many “fractions” using differences in boiling points of component parts. Diesel oil and gasoline (petrol) are two such components. Some of the differences in diesel and petrol exhausts, mainly relating to CO_2 emissions, can be attributed to the differences in chemical composition of the fuel. Fundamentally diesel is a denser fuel than petrol. As a result less diesel fuel is required to produce the same amount of energy. This is one reason CO_2 emissions are lower from diesel vehicles. The second reason is diesel has a more efficient combustion process. The elemental trade-off between NO_x and CO_2 occurs because fuel- efficient combustion requires a higher temperature, and at higher temperatures more NO_x is formed.

2.4.2 Combustion in diesel engines

Diesel engines are also known as compression ignition (CI) engines. In a diesel engine the fuel is injected into the combustion chamber under high pressure (>2000 bar) causing the fuel to rise to a temperature at which it ignites. This mechanism means the mixture of air and fuel occurs during combustion. Diesel is dense and therefore does not mix easily with air as it is injected into the combustion chamber. When air and fuel are not well mixed irregular combustion occurs, allowing pockets of incomplete combustion which result in the formation of particulates (Überall *et al.*, 2015).

Diesel engines operate under a wide range of air / fuel ratios, though nearly always higher (i.e. more air) than the “stoichiometric” ratio. The stoichiometric ratio is the ideal ratio for combustion, at this ratio there is the exact amount of fuel and air for complete combustion. Diesel engines run “lean”, meaning there is a higher ratio of air. This means less fuel is required for the same power output, resulting in lower CO_2 emissions. However, the lean air/ fuel ratio is also the main reason NO_x emissions from diesel vehicles are much higher than from petrol. Lean combustion creates a lot of heat which contributes to the efficiency of diesel engines but is key in the formation of NO_x. At temperatures >1500 °C nitrogen in the air reacts with oxygen and NO_x is formed.

2.4.3 Combustion in petrol engines

Petrol engines are commonly known as spark or positive ignition (PI) engines. There are now two different types of petrol engine; conventional port fuel injection (PFI) and the relatively new gasoline direct injection (GDI).

2.4.3.1 Port Fuel Injection (PFI)

In a Port Fuel Injection (PFI) petrol engine the fuel is injected through an intake track at a low pressure and ignited by a localised high temperature supplied by an external energy source (i.e. a spark). This is possible as petrol fuel is much lighter than diesel and readily evaporates to mix efficiently with the air in the combustion chamber. As a result a small spark can produce smooth combustion throughout the well- mixed combustion chamber. Petrol engines operate at much lower air / fuel ratios than diesel engines, oscillating around the stoichiometric ratio. This means less particulates are formed because all the fuel is completely burned.

2.4.3.2 Gasoline Direct Injection (GDI)

40% of the petrol vehicles tested for this research project were Gasoline Direct Injection (GDI). GDI engines in theory have higher fuel efficiency (therefore lower CO_2 emissions) than conventional PFI petrol engines. In recent years the GDI market share has rapidly increased, now making up ~50% of new petrol vehicles (Saliba *et al.*, 2017; Wolfram *et al.*, 2016).

In GDI engines the petrol fuel is injected at higher pressure (up to 200 bars) straight into the combustion chamber and then ignited. GDIs create a lean air / fuel mixture meaning, like diesel combustion, there is a higher ratio of air. As a result less fuel is consumed and less CO_2 produced. Chan, Meloche, Kubsh, *et al.*, (2012) found GDI engines deliver a fuel consumption saving of between 3-6% compared with PFI.

However, higher emissions of pollutants characteristic of diesel engines also effect GDI engines. GDI engines have much higher emissions of particulates than PFIs, higher than a diesel vehicle with a Diesel Particulate Filter (DPF) (Peckham *et al.*, 2011; Liang *et al.*, 2012; Wang *et al.*, 2014). GDIs also emit a higher number of smaller, ultra-fine particles < 100 nm diameter which, as discussed previously, are associated with many adverse health effects.

DPFs successfully reduce diesel particulate emissions (Mathis, Mohr & Forss, 2005). It is thought the introduction of Gasoline Particulate Filters (GPF) for GDIs will have a similar reduction effect (Chan *et al.*, 2012).

2.4.4 Abatement technologies

This section describes the most common exhaust after treatments used to reduce harmful emissions from petrol and diesel cars.

2.4.4.1 Three way catalyst (TWC)

All modern petrol vehicles are fitted with a catalytic converter known as a three way catalyst (TWC). TWCs use the chemical processes of oxidation and reduction to turn harmful pollutants into harmless by-products. These reactions are facilitated by the precious metals Platinum, Rhodium and Palladium spread over a 3D ceramic honey comb structure to maximise surface area. TWCs reduce ~95% of CO, NO_x and HC emissions (Santos & Costa, 2008).

A TWC essentially completes the combustion process via three reactions listed below;

1. The reduction of NO_x to nitrogen and oxygen ($2NO_x \rightarrow O_2 + N_2$)
2. The oxidation of CO to CO_2 ($2CO + O_2 \rightarrow 2CO_2$)
3. The oxidation of unburnt HC to CO_2 and H_2O (water)

The optimum efficiency for these three reactions is facilitated by the exhaust emissions of an engine running close to its stoichiometric ratio. As discussed the air / fuel ratio in a petrol engine oscillates around the stoichiometric ratio, creating ideal conditions for all three of the reactions above to take place. The TWC will only work for stoichiometric engines (like petrol engines) where the O_2 concentrations are <1%.

2.4.4.2 Diesel oxidation catalyst (DOC)

Another key reason NO_x emissions are higher from diesel engines is they cannot use a TWC. Diesel engines run lean, operating far above the stoichiometric ratio. As a result the exhaust gases from diesel vehicles have much higher levels of oxygen. This means oxidising reactions (2 and 3) are favoured at the detriment of the reduction reaction (1). For this reason diesel vehicles cannot use a TWC and instead use a diesel oxidation catalyst (DOC). The DOC works in a similar way to the TWC above to effectively reduce CO and HC but is unable to remove NO_x. The oxidising effect of the lean exhaust gases is the reason diesel vehicles have lower emissions of CO and HC than petrol vehicles. It also oxidises some of the NO to NO_2, which is partly why diesels emit a higher proportion of primary NO_2 (Carslaw *et al.*, 2016). As the DOC is unable to reduce NO_x from the exhaust, diesel vehicles deploy a number of additional NO_x abatement technologies.

2.4.4.3 Exhaust gas recirculation (EGR)

As discussed above the key reason for heightened NO_x emissions from diesel vehicles is the high temperatures that facilitate the oxidisation of nitrogen. EGR reduces NO_x formation by lowering the temperature of combustion. It does this by taking a proportion of the exhaust gas and returning it into the combustion chamber. The exhaust gas is inert, meaning it is virtually void of oxygen and will not support combustion. Some of the heat energy generated during combustion is absorbed by exhaust gas, reducing the peak combustion temperature and formation of NO_x.

All diesel passenger cars Euro 5 and later use EGR to reduce NO_x formation. However EGR alone is no longer sufficient to meet type approval limits and most Euro 6 diesels also deploy additional technologies to remove NO_x from the exhaust gases once it has formed.

2.4.4.4 Selective catalytic reduction (SCR)

SCR was first used in stationary sources such as large municipal waste boilers. It then began to be used by large diesel engines on ships and trains, eventually being installed on buses and HGVs and finally in recent years it has been used in diesel passenger cars (Malpartida *et al.*, 2012). In a passenger car the SCR is installed after the DOC and DPF.

First an injection of Diesel Exhaust Fluid (DEF) is mixed with the exhaust gases. The most common DEF, AdBlue, is 30% high purity urea dissolved in deionised water. When mixed with exhaust gases the DEF is rapidly hydrolysed to form ammonia (NH_3) and CO_2, this is the first reaction in the SCR process. The NH_3 and NO_x then pass into the SCR catalyst where a reaction is facilitated by a honeycomb structure of precious metals, turning NO_x into nitrogen (N_2) and water (Ofoli, 2014). Finally the gases pass through an oxidation catalyst that turns any remaining ammonia into N_2 and water. SCR is called "selective reduction" because the ammonia catalyst reduces only the NO_x in an oxidising environment.

One of the criticisms of SCR is the process is very sensitive to the quantity of DEF injected. If there is too much DEF NH_3 can be released into the atmosphere, this is known as "ammonia slip". However, if there is too little there is insufficient NO_x conversion. Another problem is the DEF tank must be refilled sporadically by the vehicle owner. It is also the case the SCR is more effective at removing NO_x when there is a higher ratio of primary NO_2 (Malpartida *et al.*, 2012). As a result systems using SCR encourage a higher fNO_2 within the SCR process, which results in a higher fNO_2 tailpipe emission. This will be explored further in Chapter 4.

2.4.4.5 Lean NO_x traps (LNT)

As the name indicates the lean NO_x trap (LNT) is a device that reduces or "traps" NO_x emissions from a lean burn engine. The LNT is the latest diesel NO_x reduction technology to be introduced. Exhaust gases are filtered over a molecular "sponge" of alkali or alkaline- earth metal oxides. This removes the NO_x and stores it in the form of nitrites and nitrates (Larson *et al.*, 2008). The stored NO_x is periodically released and reduced. This is done by briefly creating reducing conditions (i.e. lower levels of O_2) by a short period of rich engine operation which generates reductants such as CO, H_2 and HC. These reductants stimulate the release of the stored NO_x and catalytically reduce it to N_2 and O_2. However there is also potential for the formation of harmful by-products such as N_2O and NH_3.

Another issue with LNT is the period of rich burn required for regeneration incurs a fuel/ CO_2 penalty. This is the main element of emission control that was manipulated by the Volkswagen defeat device. Volkswagen reduced the number of LNT regenerations when the vehicle was in normal operation (not on a chassis dynamometer) in order to save fuel and deliver efficiency savings. When the LNT is not regenerated at regular intervals the "sponge" becomes saturated with NO_x unable to absorb any more. As a result tail pipe NO_x emissions increase.

2.4.4.6 Particulate filters (DPF and GPF)

All diesel vehicles Euro 5 and after (since 2009) have been fitted with a Diesel Particulate Filter (DPF) to reduce emissions of PM. As discussed previously a particulate filter was not necessary for PFI vehicles as only low levels of particulates are formed during stoichiometric combustion. However the new GDI petrol vehicles require Gasoline Particulate Filters (GPF) to pass the Euro 6 PN limit. DPFs have been found to reduce particulate emissions by up to 99% (Liu, Skemp & Lincoln, 2003; Mayer *et al.*, 2002).

Exhaust gases pass through the particulate filter after the DOC but before the SCR. DPFs are essentially filtration devices that successfully filter the majority of soot particles (Mathis, Mohr & Forss, 2005). DPFs have a honeycomb structure made of microscopic channels which the exhaust gases flow through. These channels trap the soot particles removing them from the exhaust. The soot particles accumulate on the walls of the microscopic channels and must be burned off regularly in regeneration events.

The main concern relating to particulate filters are the high levels of particulate emissions, especially ultrafine particulates, during regeneration (Giechaskiel *et al.*, 2007; Hawker *et al.*, 1998). Regeneration occurs spontaneously when the DPF reaches a temperature of > 600 °C, which usually only occurs during motorway driving, where public exposure is low. DPFs also slightly increase CO_2 emissions by 2-5% because of an increase in back pressure requiring additional mechanical work to be overcome (Liu, Skemp & Lincoln, 2003; Mayer *et al.*, 2002).

2.4.5 Cold start emissions

A cold start refers to an engine starting once it has cooled fully to ambient temperatures (< 30°C) (Heimrich, 1990). In the first minutes of engine operation low temperatures result in incomplete combustion, producing higher emissions than normal operation (Cao, 2007; Weilenmann *et al.*, 2005). This coincides with the catalytic converter being below its optimum operating temperature (~ 400°C), preventing the removal of HC, CO, PM and NO_x (Chang *et al.*, 2014; Mathis, Mohr & Forss, 2005). As a result pollutant emissions in the first few minutes of operation can be many times those from normal operation.

Cold start emissions are extremely sensitive to ambient temperature (with the exception of NO_x from gasoline vehicles (Reiter & Kockelman, 2016; Weilenmann, Favez & Alvarez, 2009)) and are a significantly lower proportion of total emissions for diesel vehicles than for petrol. For petrol vehicles the vast majority of HC, CO and NO_x emissions are from the cold start period (Weilenmann *et al.*, 2005).

Previous studies estimated the cold start period covers the first 1 – 5 km of urban driving (Favez, Weilenmann & Stilli, 2009; André & Joumard, 2005). In terms of duration Chen, Chiang, Chen, *et al.*, (2011) reported emissions stabilised after the first 120s of a journey. In the US and the UK almost half of all car journeys are less than 5km (de Nazelle *et al.*, 2010; DfT, 2016a). For short journeys cold start emissions are the dominant source of total pollutant emission and can last the duration of the journey. Most journeys start in urban conurbations close to people's homes and workplaces, meaning the cold start often occurs in the areas of highest public exposure. Miller & Franco (2016) estimate cold starts make up 8% of vehicle kilometres driven.

2.5 COPERT

COPERT stands for Computer Program to calculate Emissions from Road Transport. It is an air quality transport model developed by the European Environment Agency. The European Monitoring and Evaluation Program (EMEP) recommends COPERT as the preferred tool in the calculation of vehicle emissions (EEA, 2013). COPERT is also widely used in modelling studies. The COPERT model provides speed dependent emissions factors for regulated pollutants such as CO, NO_x, VOC and PM along with unregulated pollutants such as N_2O, NH_3, SO_2 and NMVOC.

COPERT is currently used for road transport inventories and emissions projections by 22 of the 28 EU member states (Kioutsioukis *et al.*, 2010). It is used in the UK for NO_x emissions projections, road transport emissions modelling and provides speed dependent emissions factors for the Emissions Factor Toolkit and the Pollution Climate Mapping (PCM) model (DEFRA, 2014; Kousoulidou *et al.*, 2013). The PCM model is used by DEFRA in scenario assessment and population exposure calculations to inform policy development. It is therefore extremely important that

COPERT emissions factors are representative of real world driving. The COPERT emissions factors for Euro 6 diesel vehicles are evaluated in Chapter 4.

The version of COPERT used in this analysis is 4v11. This was the recommended version at the time the work was carried out and published (*see* O'Driscoll, ApSimon, Oxley, *et al.*, 2016). COPERT has since been updated to Version 5, this will be discussed briefly at the end of Chapter 4.

2.6 Real driving emissions (RDE) measurement

Much of the research presented in this thesis relates to real driving emissions of passenger cars. As discussed RDE refers to exhaust emissions during 'normal operation', which are often higher than type approval limits. The discrepancy between type approval limits and RDE has been apparent since the early Euro standards (Kågeson, 1998). The development of Portable Emissions Measurement Systems (PEMS) for passenger cars has undoubtedly played an important role in exposing real driving emissions.

2.6.1 Portable Emissions Measurement Systems (PEMS)

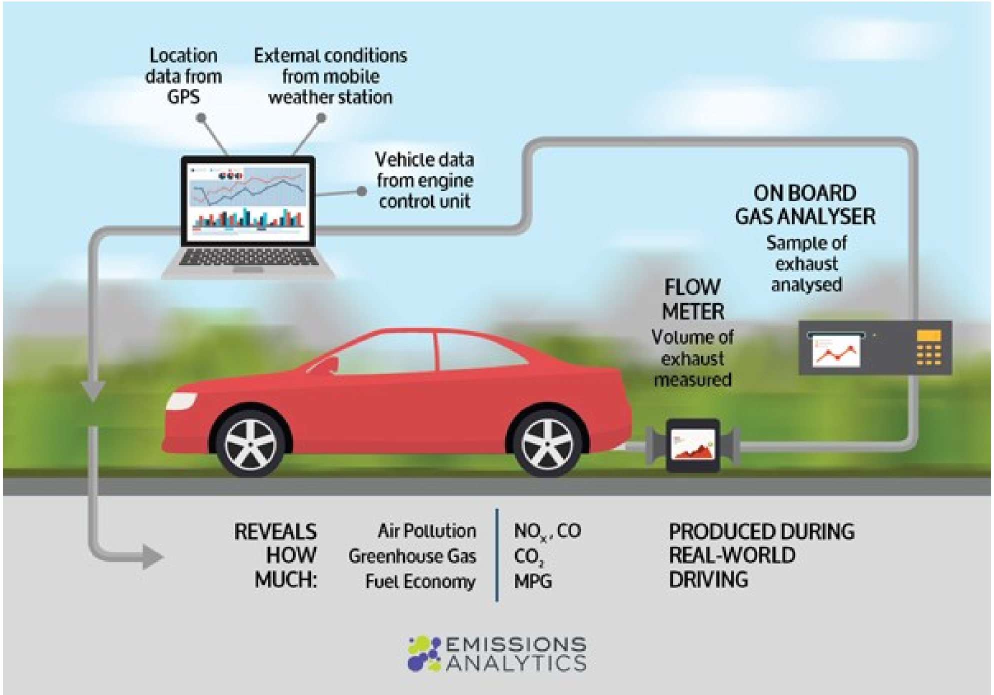

Figure 2-7. Diagram of PEMS (Source: Emissions Analytics)

PEMS are a mobile laboratory that record a constant real time measurement of exhaust emissions of CO, CO_2, NO_x, NO,NO_2 and THC (and experimentally CH_4, PM and total PN, though these were not measured by the PEMS used in this study). They can be fitted to the tail pipe of practically any passenger car without the need for vehicle modification. Vehicle emissions can then be measured during normal operation on public roads. **Figure 2-7** is a diagram of a PEMS system in operation. Typically the on board gas analyser will be placed in the boot or back seat of the vehicle.

The RDE component of the light duty type approval process being introduced in 2017 will be enforced using PEMS. PEMS were developed in the late 90's, were approved for use in the heavy duty type approval process in 2009 and became a mandatory component in 2011 (EC, 2009b, 2011). Along with remote sensing PEMS studies have exposed the discrepancies between certification and on road emissions (Rubino *et al.*, 2007; Weiss *et al.*, 2011b; Carslaw, 2005). The introduction of a PEMS component to the light duty test procedure is expected to tackle the issue of NO_2 exceedance in urban areas (Degraeuwe *et al.*, 2015; Weiss *et al.*, 2012).

Previous studies have found vehicle emissions are strongly correlated with a variety of operating and environmental conditions including congestion, driving style, wind and ambient temperature (Kousoulidou *et al.*, 2013; Gallus *et al.*, 2017; De Vlieger, 1997; Weiss *et al.*, 2011b). This introduces a level of variability in PEMS tests because external factors cannot be controlled and regulated the way they are in laboratory based chassis dynamometer tests (Weiss *et al.*, 2011b). Reduced repeatability is the main criticism of PEMS but it is also what makes PEMS measurements more representative of the real world. TNO state "*as a rule of thumb a 15 to 20% variation for similar trips appears natural*" (Gerrit *et al.*, 2016).

PEMS already play a substantial role in compiling emissions inventories, developing emissions factors for projections and use in emissions models (Collins *et al.*, 2007; Frey *et al.*, 2003). These include the air quality model COPERT. The PEMS testing presented in this thesis was performed by Emissions Analytics using a SEMTECH-DS. Further details relating to the measurement procedure can be found in Chapter 4.

The strength of PEMS data is that it is recorded at 1 Hz resolution. This allows analysis of emissions in real time, facilitating analysis of the relationship between driving characteristics (e.g. speed) and exhaust emissions. The main limitation of PEMS is that the equipment is very expensive, bulky and energy intensive. As a result sample sizes can be small (often less than 10 vehicles) and tests run no longer than a few hours. To address this TNO have developed the Smart Emission Measurement Systems (SEMS) that measures CO_2 and NO_x. It is smaller than the PEMS and lower-cost but also less accurate. The other benefit of the SEMS is it can conduct measurements over a longer timescale (weeks as opposed to hours).

2.6.2 Remote sensing

Remote sensing (in contrast to PEMS) provides a snapshot of exhaust emissions for a very large sample of vehicles (>70,000). A key advantage of remote sensing is a large number of vehicles can be sampled relatively quickly (Williams & Carslaw, 2011). Remote sensing involves a non- mobile measurement station being installed close to the traffic stream. The Remote Sensing Detector (RSD) consists of an emitter and a detector positioned facing each other with the traffic flowing between them. The emitter emits light at various frequencies which the detector detects. Spectroscopy is performed on the exhaust gases of vehicles as they pass between the emitter and detector breaking the beam. Essentially the RSD records to what extent the exhaust plume absorbs energy in different frequency bands (Bishop *et al.*, 2010).

If the RSD is paired with an Automatic Number Plate Recognition (ANPR) camera individual vehicles can be matched to emissions measurements. This creates the potential for remote sensing to be used for in service conformity tests. It also allows profiles to be built up for specific technology types (e.g. Euro 5 diesel) and conclusions drawn as to the average performance across the fleet. A limitation of remote sensing is that emissions are recorded as a ratio of CO_2, and CO_2 is not recorded by the RSD. It is therefore not possible to make a direct comparison with type approval limits which are measured in g/km without using an estimate for CO_2.

Another limitation of RSD is that (as it provides a snapshot) it is not possible to know whether the emission recorded is a peak or spike in emissions occurring due to acceleration or DPF regeneration or if the vehicle in question has consistently high emissions. PEMS data reveals that pollutants are emitted in peaks and troughs, with RSD it is impossible to say which part of the emission you are capturing. RSD is good at producing large statistical overviews but is not as good as PEMS for evaluating individual vehicles.

2.6.3 Other measurement techniques

Other measurement techniques include chase measurements where the vehicle being measured is followed by a vehicle containing a mobile emissions laboratory. Though these studies have been found to be within the accuracy of laboratory based testing the requirement that the vehicles are within 10m of each other means in practice these studies cannot be conducted on open roads (Bergmann *et al.*, 2009a).

Tunnel studies use measurements of pollutant concentrations at the entrance and exit to a tunnel along with a measurement of airflow to estimate the total amount of pollutant emitted in the tunnel. These studies are useful for estimating aggregate real world emissions data, but difficult to focus on specific vehicle technologies.

2.7 The UK Integrated Assessment Model (UKIAM)

The UKIAM is an integrated assessment model developed by the Integrated Assessment Unit at Imperial College London designed to evaluate emissions control strategies in the UK (Oxley, ApSimon & Valiantis, 2011). It fulfils the requirement of the DEFRA Support for National Air Pollution Strategies contract and explores the impacts of a potential policy developments and cost-effective strategies for improving future air quality. The UKIAM has some similarities to the Pollution Climate Mapping (PCM) model, the official DEFRA air quality model used to fulfil the requirements of Ambient Air Quality Directive (2008/50/EC). Both UKIAM and PCM model background locations at a 1 km^2 resolution and both consider roadside contributions as a separate increment. However UKIAM is more deterministic and PCM is semi-empirical.

It brings together data from the other DEFRA contractors to create a multifaceted projection and appraisal of proposed strategies. The pollutants modelled by the UKIAM are SO_2, NO_x, PM_{10}, $PM_{2.5}$ and NH_3 (ammonia, mainly from agriculture, contributes to nitrogen deposition and health risks). The initial scoping exercise of this research project was to identify possible uncertainties in the UKIAM and infer any action required to mitigate or allow for these uncertainties. This is presented in the next chapter. The UKIAM has also been used to generate the 2030 emissions projections reported in Chapter 5.

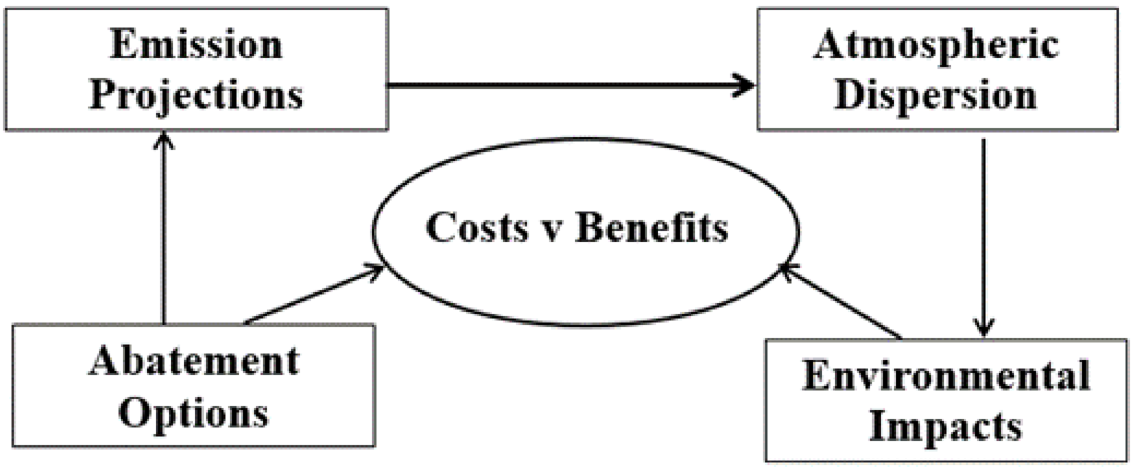

Figure 2-8. Basic schematic of the UKIAM

As shown in **Figure 2-8** the UKIAM can be subdivided in to five main topic areas; emission projections, costs and benefits, atmospheric dispersion, abatement options and environmental impacts. A brief overview of each of these main topic areas is given below.

- **Emissions projections** are calculated for future scenarios up to 2030. These are used in atmospheric modelling and defining potential abatement measures. Emission projections depend on activity data and emission factors as emission activity data which comes from a number of different sources such as the NAEI, LAEI (London Atmospheric Emissions Inventory) and DfT.
- **Atmospheric Dispersion Models** use emissions data to predict concentrations and deposition of pollutants across the country. The UKIAM can interchange the Source Receptor framework from various models including FRAME and EMEP.
- **Environmental Impacts** are split into two main categories; ecosystems and health impacts. For ecosystems the UKIAM puts emphasis on Sites of Special Scientific Interest (SSSIs) and risk classes. The UKIAM accesses improvements in protection for different potential abatement strategies. Deposition of ammonia (eutrophication) is an important factor in ecosystem protection and it is the pollutant with the biggest uncertainty attached to it. Health Impacts are calculated using change in Population Weighted Mean Concentration (PWMC) of pollutants as an indicator of exposure of the UK population.

- **Abatement Options** can be categorized as technical measures and behavioural changes. Technical measures are based on the Multi Pollutant Measures Database (MPMD compiled by Amec Foster Wheeler) and work by Rothamstead on emissions of agricultural NH_3.
- **Cost Benefit Analysis** in the UKIAM consists of applying marginal damage costs related to changes in PWMC to estimate monetised health benefits and compare with abatement costs.

The outputs of the UKIAM include total emissions (in tonnes) of SO_2, NO_x, PM_{10}, $PM_{2.5}$ and NH_3, 5 km resolution deposition maps for NH_x, SO_2 and NO_x, 5 km resolution concentration maps for NO_3, SO_4 and NH_4, 1 km resolution maps for PM_{10}, $PM_{2.5}$, NO_x and NO_2, 1 km resolution of PWMC for all pollutants and source apportionment.

The UKIAM is a series of nested models that are called in succession. Components of the UKIAM include the Abatement Strategies Assessment Model (ASAM) which models the European imported contribution and Background, Road and Urban Transport modelling of Air quality Limit values (BRUTAL) which models road transport emissions. UK non-transport sources are modelled by the UKIAM itself. This is illustrated by **Figure 2-9**. The part of the UKIAM used most in this research was the BRUTAL model.

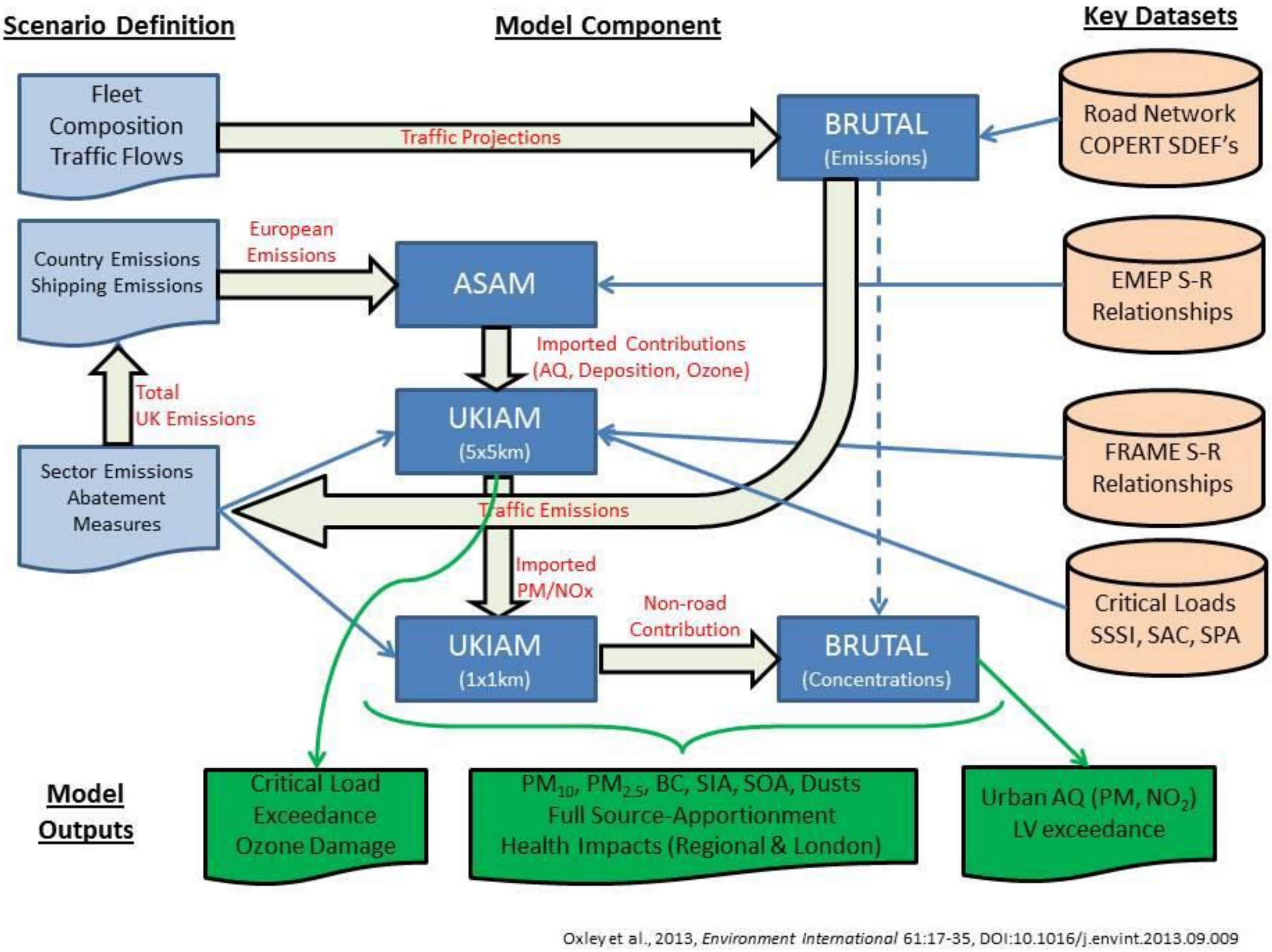

Figure 2-9. Schematic representation of the integrated structure of the multi-scale UK Integrated Assessment Model (UKIAM)

A final point relating to the UKIAM is that it is not designed to model aerial sources (i.e. planes) and has limited vertical resolution. This leads to an over estimate of pollutant concentrations in grid squares containing airports. The UKIAM should therefore not be used for modelling in the areas surrounding airports (particularly Heathrow). For this reason results presented in later chapters will include the caveat (excluding Heathrow).

2.7.1 The BRUTAL model

The BRUTAL model is the high resolution (1 km) urban scale transport sub-model of the UKIAM. It adopts a bottom up approach to calculate the total UK transport emissions (in tonnes), the background and roadside pollutant concentrations (in $\mu g\ m^{-3}$) and the number of grid-squares containing roads at risk of exceeding the Air Quality Limit Value for annual mean NO_2.

The BRUTAL model is described fully by Oxley et. al 2009 (Oxley *et al.*, 2009). This section presents an overview of the model relevant to this analysis. The version the model used for this analysis was BRUTAL v4.3 with the baseline year 2014.

Within the BRUTAL model vector based GIS maps of the UK road network (including all motorways, major roads and most minor roads) are mapped onto a 1 x 1 km grid. Traffic flows are assigned to the road lengths in each grid-square using monitored traffic flow data from the NAEI where available and aggregated regional traffic flow data from the DfT. London traffic flow data is taken from the London Atmospheric Emissions Inventory (LAEI). The vehicle technology mix varies by road type and region and is taken from NAEI projections. Grid-squares also include data relating to population density.

2.7.1.1 Modelling of NO_x/ NO_2

As discussed there is a non-linear relationship between NO_x emissions and NO_2 concentrations. As a result modelling NO_2 concentrations is not straightforward. BRUTAL uses a quadratic relationship to calculate background NO_x concentrations. This approach compares well with the empirical total oxidant method used by Jenkin and Clapp (Clapp & Jenkin, 2001). This method deduces that due to the strong chemical coupling between NO_x and O_3 it is beneficial to regard NO, NO_2 and O_3 as a set species rather than NO and NO_2 alone.

Before the BRUTAL model is called by the UKIAM another sub-model produces 1 x 1 km background NO_x and PM_{10} concentration maps from the contribution of all non-traffic sources including transboundary European emissions (fNO_2 from non-traffic sauces assumed 10%). Ambient concentrations of urban NO_2 are affected by changes in NO_x emissions, the proportion of NO_x emitted as primary NO_2 and background concentrations of NO_x and O_3.

Each grid-square is assigned a location type (eg. urban, suburban, rural) dependent on the traffic flow, population density and NO_2 concentration. A different value of β (ratio of NO to O_3) is applied for the different types of location. A background concentration for NO_x, O_3 and NO_2 relating to the emissions contribution from non-traffic sources is calculated for each grid-square by the UKIAM.

To add the component of background concentration which relates to traffic sources BRUTAL adds together fNO_2 and NO_x emissions from all traffic sources in the grid-square and calculates the total oxidant. The total NO_x from all traffic sources is then numerically dispersed and added to the non-traffic background. This combined value (traffic + non- traffic) is the predicted background concentration for each grid-square. Finally, to calculate the predicted roadside concentrations a roadside enhancement/ increment (related to the trafic flow, fleet mix, speed and emission factor of the road) is added and multiplied by a street canyon factor (if the grid-square is classed as an urban location). The street canyon factor is derived using ADMS-Urban, is modified for different road types and is related to the population density in the grid-square (Oxley *et al.*, 2009; Vardoulakis *et al.*, 2007).

2.7.1.2 Identifying exceedance of annual mean NO_2 limit

For each 1 x 1 km grid-square the BRUTAL model predicts the roadside concentration of the busiest road within that square. For example, **Figure 2-10** is a 1 x 1 km grid-square in central London. Though there are many roads within this grid-square the BRUTAL output relates only the highest annual roadside reading. For the grid-square depicted this will always be Marylebone Road (highlighted by red dashed line), one of the main arterial roads into London with Annual Average Daily Flow (AADF) ~100,000 vehicles.

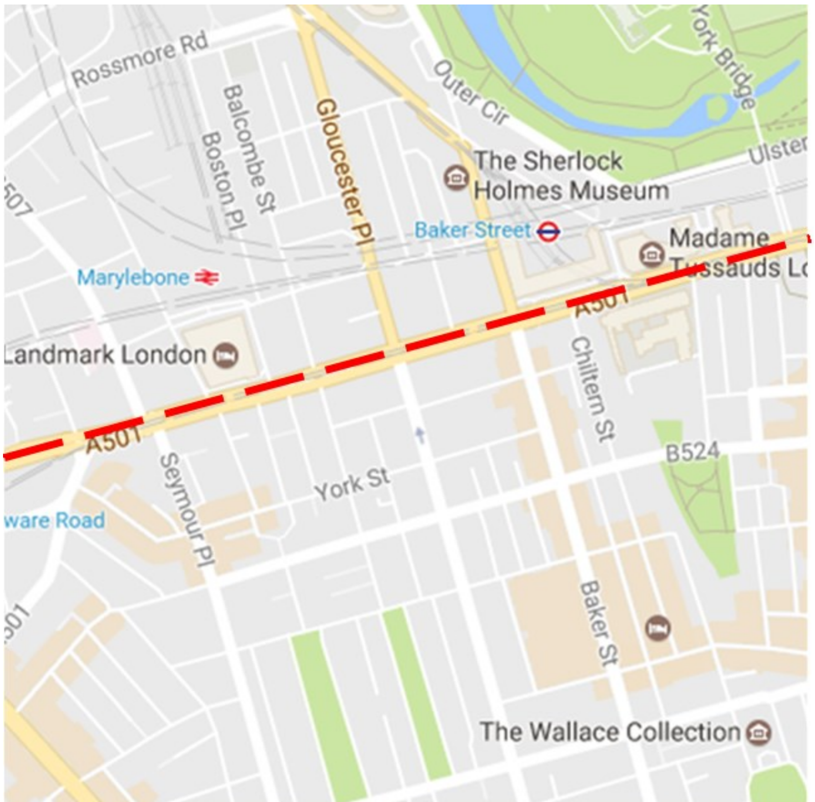

Figure 2-10. Map of 1 x 1 km square in London, Marylebone Road highlighted
"Map data ©2017 Google"

The UKIAM identifies grid-squares where the busiest road is at risk of exceeding the limit value based on the road-side increment superimposed on the background. These are grid-squares where more detailed modelling is required, for example using the ADMS model.

2.7.1.3 BRUTAL model validation

BRUTAL v4.3 model has been validated against 2014 measurement data for sites across the UK. For model validation the official (at the time) COPERT 4v11 speed dependent emissions factors were used. The locations of the 150 sites used for validation are plotted in **Figure 2-11**. The validation sites were located across the UK, with a higher concentration in London. The measurement data for the London sites came from the London Air Quality Network (LAQN), the measurement data for the national sites came from the Automatic Urban and Rural Network (AURN).

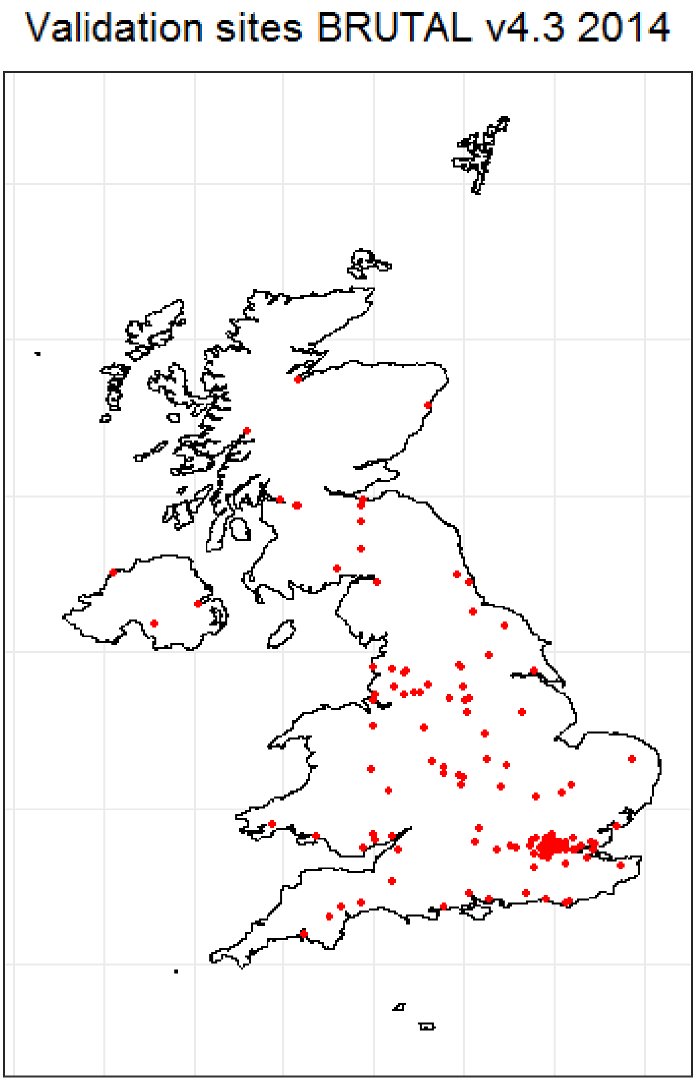

Figure 2-11. Validation sites across the UK

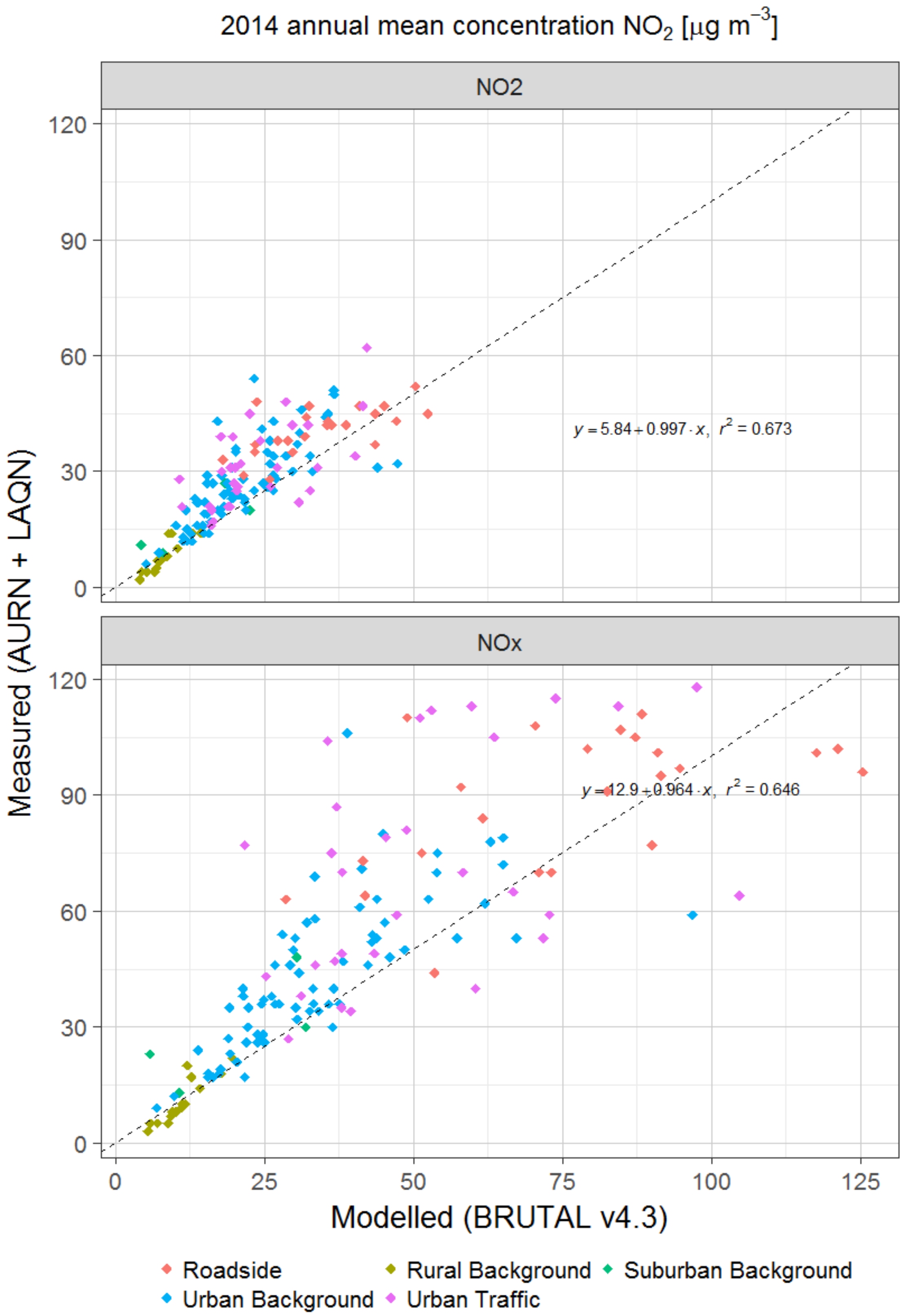

Figure 2-12. Measured vs. modelled NO_2 and NO_x for 150 sites across the UK

Figure 2-12 is a scatter plot of BRUTAL predictions against measured values for NO_2 and NO_x at 150 background and roadside locations. There was good correlation between the measured and modelled values. However, **Figure 2-12** shows the model had a negative bias, meaning it underestimated when compared with measurements. This bias is explored in **Table 2-7** using the "modStats" function in the R package "openair". The statistics presented are defined in the package as follows:

FAC2	*fraction of predictions within a factor of two*
MB	*the mean bias*
MGE	*the mean gross error*
NMB	*the normalised mean bias*
NMGE	*the normalised mean gross error*
RMSE	*the root mean squared error*
r	*the Pearson correlation coefficient*

Table 2-7 shows that the MB (the negative bias) is largest for traffic related sites (Roadside and Urban Traffic). A potential explanation for this is that for this comparison BRUTAL used the COPERT 4v11 emissions factors, which (as shown in the Chapter 4) underestimate Euro 6 diesel emissions factors and potentially other vehicle categories also. *Oxley, ApSimon & O'Driscoll* (2016) investigated the effect on model validation of increasing the COPERT 4v11 emissions factors used by BRUTAL to match real world emissions. They found the model was better able to replicate the measured roadside concentrations of NO_2 when the emissions factors were increased, indicating an underestimate in emissions factors contributed to the negative bias of

the BRUTAL model. This is similar to the findings of Lee *et al.* (2015) (discussed previously) who found in London measurements of NO_x at traffic related sites were far higher than predicted by the NAEI (which uses COPERT emissions factors).

Table 2-7. Summary of NO_2 model evaluation statistics (against measurements)

Type	n	FAC2	MB	MGE	NMB	NMGE	RMSE	r
Roadside	27	0.96	-7.49	8.84	-0.18	0.21	10.56	0.63
Urban Traffic	30	0.80	-19.78	26.86	-0.28	0.38	33.18	0.44
Rural Background	15	0.93	-0.01	1.43	0.00	0.17	2.09	0.88
Suburban Background	4	0.75	-3.42	4.70	-0.20	0.28	5.59	0.82
Urban Background	74	0.97	-5.75	6.79	-0.21	0.25	9.10	0.74

Table 2-7 and **Table 2-8** quantify the correlation between modelled annual NO_x and NO_2 emissions from BRUTAL v4.3 and measurement data. The best overall predictions by site type were for rural background locations. As discussed other sites (which may be more affected by traffic emissions) displayed a negative bias. This analysis used an updated version of BRUTAL to that used in DEFRA's *Model Inter-comparison Exercise (MIE)* (Carslaw, 2011) where BRUTAL was compared to ADMS-Urban, ERG-Toolkit, CMAQ-Urban and PCM. The MIE focused solely on London so a direct comparison is not possible. However, in general the analysis presented here indicates improvements to the BRUTAL model have been a success. The magnitude

of the MB has significantly reduced compared to the MIE findings, and the R values have significantly increased.

Table 2-8. Summary of NO_x model evaluation statistics (against measurements)

Type	n	FAC2	MB	MGE	NMB	NMGE	RMSE	r
Roadside	27	0.93	-15.15	21.99	-0.17	0.24	26.44	0.56
Urban Traffic	30	0.80	-19.78	26.86	-0.28	0.38	33.18	0.44
Rural Background	15	1.00	0.37	2.35	0.03	0.22	2.98	0.89
Suburban Background	4	0.75	-8.78	9.73	-0.31	0.34	12.39	0.75
Urban Background	74	0.97	-9.14	11.13	-0.21	0.26	15.87	0.74

2.8 Summary

This chapter introduced the relevant background required to frame the research presented in the following chapters and an introduction to the models used in this analysis. It also provided the rationale behind the investigation into exhaust emissions from (particularly diesel) passenger cars and their effect on air quality. A key theme was the difference between diesel and petrol vehicles and the air quality / climate change trade off. This chapter also provided an overview of the UKIAM, the following chapter explains how the UKIAM was used to structure and guide this research project.

Chapter 3. The HAZOP approach

This chapter describes how the Hazards and Operability (HAZOP) technique for risk assessment was used to underpin this research and identify areas of interest. A preliminary HAZOP assessment formed the basis of the research project and acted as a scoping exercise. HAZOP has been a vital tool in determining the direction and content of the subsequent research. Chapters 4 – 6 contain separate methodologies that refer specifically to the analysis presented in those chapters, the methods described in this chapter relate to the structure and direction of the research project as a whole.

3.1 HAZOP

The HAZOP technique for risk assessment can be used as a heuristic method to identify possible causes of uncertainty within environmental models and provide a framework for research (ApSimon, Warren et al. 2002). The technique was endorsed by the Chemical Industries Association (CIA, 1989) for chemical engineering plants but can be applied to environmental issues. HAZOP assessments contain four main steps:

1. Identify and consider each component of the process/ model being assessed
2. Define the function of each identified component
3. Consider deviations from this function and how such deviations might occur. Deviations from the function may be described by words such as; “NOT”, “LESS”, “MORE”, “AS WELL AS” and “REVERSE”
4. Consider the consequences of these deviations, identify hazards (uncertainties) and define them. Consider the possible deviations from current procedure and then consider the consequences of these deviations

The same four step process used in chemical plants can be used to assess environmental models. First the model is broken down into its component parts, then the functionality of each part individually assessed. The initial results are concise, allowing for further detail to be added where required. More in depth analysis can then be performed for the components of the model perceived to pose the biggest risk/ uncertainty. The 4th step of a HAZOP assessment of an environmental model will usually include a form of sensitivity analysis, an already well-established uncertainty technique.

The HAZOP technique can be performed multiple times and continuously updated throughout the lifetime of an assessment or project. It can also be used to monitor the progress of a project as it breaks down complex processes into measurable stages and components. Similarly it can be used as a checklist at the end of a project to evaluate the delivery of objectives.

3.2 HAZOP assessment of the UKIAM

A full HAZOP assessment of the UKIAM can be found in the Appendix. The results from Stage 1 (identify and consider each component of the process) are depicted in **Figure 3-1**. The process started by first identifying the 5 main topic areas; costs and benefits, abatement options, emission projections, atmospheric dispersion and environmental impacts. The main topic areas were then divided into subgroups. The HAZOP assessment did not include costs and benefits as this topic area has been assessed separately by DEFRA contractor Mike Holland using the Treatment of Uncertainty in a Benefit Assessment (TUBA) approach.

The component of the UKIAM thought to contain the largest uncertainty was “Emissions factors” (highlighted in red in **Figure 3-1**), specifically emissions factors from passenger cars. Rationale for this is provided in the next section. This topic was subsequently subject to the in-depth analysis presented in this thesis and is presented as a case study for application of the HAZOP technique.

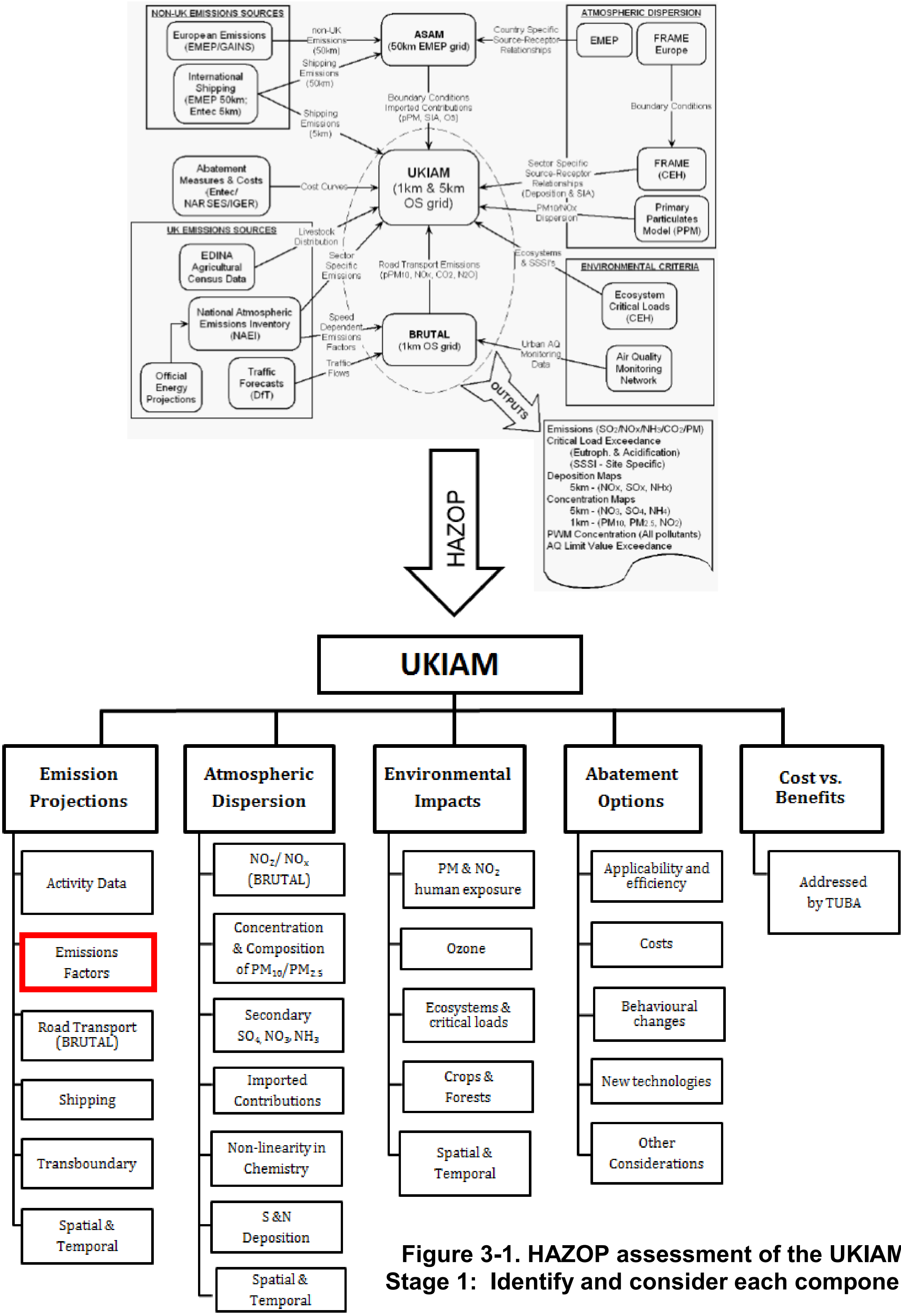

Figure 3-1. HAZOP assessment of the UKIAM
Stage 1: Identify and consider each component

3.3 Case study: NO_x emissions from Euro 6 diesel passenger cars

The component of the UKIAM chosen as a case study was Euro 6 NO_x emission factors from diesel passenger cars. This section describes why this particular component was chosen for closer examination and how HAZOP provided the framework for the subsequent research. An overview of the importance of diesel emissions to air quality in the UK has already been presented in the previous chapter.

3.3.1 Rationale for focusing on Euro 6 diesel NO_x emission factors

The first reason to focus on diesel NO_x emissions was the consistent failure of the UK to meet the Air Quality Limit Value for NO_2. UK exceedances of the limit value occur almost exclusively at roadside locations and Euro 4 and 5 diesel cars did not deliver the real world reduction in emissions promised (Carslaw *et al.*, 2011b). As discussed in the previous chapter the failure of successive Euro standards to reduce real world NO_2 emissions is thought to contribute to current NO_2 roadside exceedances.

As shown in **Figure 3-2** the NAEI estimated in 2014 diesel fuels contributed 28% of the UK's total NO_x emissions. After energy generation and manufacturing, diesel passenger cars were the single biggest source of NO_x. Energy and manufacturing are stationary sources and as such their emissions are much easier to monitor and regulate. They also tend to be located outside urban conurbations meaning public exposure is low. In contrast diesel NO_x emissions are emitted throughout our towns and cities where exposure is high.

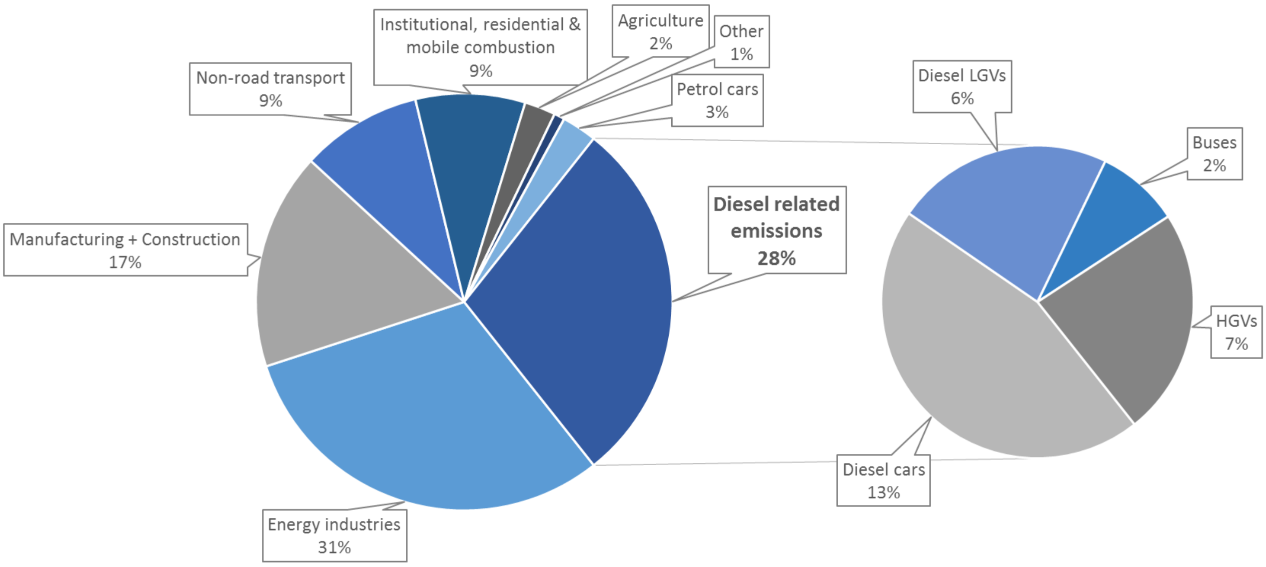

Figure 3-2. Proportion of UK NO_x emissions that come from diesel vehicles (NAEI, 2014a)

At the time of the HAZOP assessment (2014) much faith was being put in the latest Euro 6 diesel cars launched in September 2014. COPERT 4v11 was also published in September 2014 and predicted a significant reduction in NO_x between Euro 5 and 6 diesel cars. Given similar hopes had been ascribed to the introduction of Euro 4 and 5 it seemed prudent to approach these optimistic predictions with a degree of scepticism. Improvements in NO_x emissions from Euro 6 diesel technology went on to form the cornerstone of the DEFRA 2015 air quality plan as well as Clean Air Zones and London Ultra-Low Emission Zones.

Lastly the UNECE GAINS model predictions for UK total NO_x emissions were much lower than NAEI projections. As the NECD ceilings are dictated by the GAINS figures there is a risk that the ceilings will be set lower than the UK could achieve. It was therefore important to have more certainty in the emission projections.

3.3.2 Stage 1: Identify the components of the process

The component of the UKAIM being considered in greater detail for this case study is the emissions factors used in the BRUTAL model. Specifically the Euro 6 NO_x emissions factors for diesel passenger cars. Like the NAEI the BRUTAL model uses COPERT speed dependent emissions factors. At the time of this research the recommended version of COPERT in use was 4v11 (this has since been updated to version 5).

3.3.3 Stage 2: Define the function of the component

The BRUTAL model takes the COPERT emissions factors and multiplies them by activity data to calculate the total emission in tonnes.

$$\text{Emissions [grams/tonnes]} = \text{Emission Factor [g km}^{-1}\text{]} \times \text{Activity Data [km]}$$

BRUTAL then disperses this total emission and superimposes it on the background to calculate the roadside concentrations of NO_2. The diesel car contribution to total national NO_x emissions is also calculated.

3.3.4 Stage 3: Consider deviations from this function

Deviations from this function were considered using the guide words “MORE/ LESS”, “NOT” and “AS WELL AS”.

3.3.4.1 “MORE/ LESS”

The possible outcomes of this investigation were the COPERT emission factors would be “MORE” or “LESS” than the real world Euro 6 diesel emissions. The evidence available indicated they would be “LESS”, i.e. an underestimate. The most likely cause for the COPERT 4v11 emissions factors being an over/ under estimate was the limited sample size (only six vehicles) of Euro 6 diesels used to inform the model (Rexeis *et al.*, 2013; Pastramas *et al.*, 2014).

3.3.4.2 "NOT"

Emissions are effected by a wide range of input factors, it is likely estimates that depend only on speed will "NOT" be representative. They fail to take into account parameters such as ambient temperature, road gradient and wind speed. Also there are many other driving parameters (such as acceleration and vehicle specific power) that are "NOT" considered by the COPERT speed dependent emissions factors, this will be explored further in Chapter 4.

3.3.4.3 "AS WELL AS"

Diesel cars were promoted in order to reduce CO_2 emissions, it is therefore important when considering NO_x emissions from diesel passenger cars to also consider CO_2 "AS WELL". In the recent past an increase NO_x emissions relative to petrol cars was the acceptable trade-off for reduced CO_2 emissions. For this reason (and for context) it is important to consider petrol passenger cars "AS WELL AS" diesel. This is done in Chapter 6.

It is also important, for reasons discussed in the previous chapter, to consider primary NO_2 "AS WELL AS" total NO_x emissions.

3.3.5 Stage 4: Consequences of this deviation

The potential consequences of COPERT underestimating NO_x emission factors could be an underestimate in projections of total UK NO_x emissions in tonnes, possibly effecting the likelihood of meeting the NECD ceilings. Underestimating NO_x emissions from diesel passenger cars may also lead to an underestimate in roadside concentrations of NO_2 and over confidence in meeting the Air Quality Limit Value.

To assess the potential risks it was first important to narrow down the possible range of the error in the COPERT emission factors. This was done using PEMS testing presented in Chapter 4, which indicated the magnitude of the deviation.

Next the potential consequences of this deviation were explored using modelling and sensitivity analysis, presented in Chapter 5.

3.4 Summary

The HAZOP technique for risk assessment was applied to the UKIAM to identify key uncertainties requiring more in depth analysis. The area of emissions factors, specifically NO_x emissions from diesel passenger cars was identified and provided a case study of the HAZOP approach. The four step process was followed, resulting in the research presented in the following chapters.

Chapter 4. NO_x emissions from Euro 6 diesel passenger cars and comparison with COPERT

This chapter presents the real world NO_x and NO_2 emissions from 39 Euro 6 diesel passenger cars measured using a Portable Emissions Measurements System (PEMS). These measurements are then compared to the EU type approval limit and COPERT version 4v11 speed dependent emissions factors. The instantaneous PEMS measurements are also analysed for relationships between driving mode and NO_x emissions with a focus on urban driving.

4.1 Methods

Instantaneous NO_x emissions from 39 Euro 6 diesel passenger cars were measured by Emissions Analytics using a PEMS in the Greater London area. The test route comprised of ~80 km urban and motorway driving. By dividing the accumulated NO_x emission in grams by the distance travelled, the average NO_x emission over the entire test route was calculated (hereafter referred to as "trip" emission). Given the importance of urban air quality, real world emissions factors were also calculated for the composite urban and motorway sections. The COPERT model emissions estimates for the trips were calculated from the PEMS speed profile and compared with real world measurements.

4.1.1 Test fleet

All vehicles in the test fleet were category M1, defined as; "*Vehicles used for the carriage of passengers and comprising not more than eight seats in addition to the driver's seat*" (ECOSOC, 2011). The vehicle characteristics are listed in **Table 4-1**. The test fleet has been anonymised due to the commercial sensitivity of the data. Each vehicle was assigned a Vehicle ID according to the NO_x abatement technology used and the engine displacement size. The manufacturers of the vehicles sampled made up 70% of new vehicle registrations in the UK in 2015 and included 13 of the 20 most popular manufacturers in the Europe (SMMT, 2016; ICCT, 2015).

Table 4-1. Characteristics of the test fleet

Vehicle ID	Year of manufacture	Engine displacement [ℓ]	Mileage at start [km]	NO_x after-treatment
E1.5	2015	1.5	1675	EGR
E1.6	2014	1.6	2363	EGR
E2.2a	2012	2.2	6013	EGR
E2.2b	2012	2.2	225	EGR
E2.2c	2013	2.2	1164	EGR
E2.2d	2015	2.2	590	EGR
E2.2e	2015	2.2	531	EGR
L1.4a	2014	1.4	2245	EGR + LNT
L1.4b	2014	1.4	1463	EGR + LNT
L1.5	2015	1.5	1263	EGR + LNT
L2.0a	2015	2.0	1059	EGR + LNT
L2.0b	2014	2.0	2568	EGR + LNT
L2.0c	2014	2.0	745	EGR + LNT
L2.0d	2015	2.0	451	EGR + LNT
L2.0e	2015	2.0	1312	EGR + LNT
L2.0f	2013	2.0	2019	EGR + LNT
L2.0g	2014	2.0	640	EGR + LNT
L2.0h	2014	2.0	2563	EGR + LNT
L2.0i	2015	2.0	2910	EGR + LNT
L2.0j	2014	2.0	1000	EGR + LNT
L2.0k	2014	2.0	1492	EGR + LNT
L2.0l	-	2.0	742	EGR + LNT
L2.0m	2014	2.0	4356	EGR + LNT
L2.0n	2015	2.0	4276	EGR + LNT
L2.0o	2014	2.0	1696	EGR + LNT
L2.0p	2014	2.0	4192	EGR + LNT
S1.6a	2014	1.6	2406	EGR + SCR
S1.6b	2014	1.6	544	EGR + SCR
S1.6c	2013	1.6	2178	EGR + SCR
S1.6d	2014	1.6	2028	EGR + SCR
S2.0a	2015	2.0	2502	EGR + SCR
S2.0b	2015	2.0	2093	EGR + SCR
S2.0c	2014	2.0	2567	EGR + SCR
S2.0d	2014	2.0	5270	EGR + SCR
S2.0e	2013	2.0	4061	EGR + SCR
S2.0f	2014	2.0	3842	EGR + SCR
S2.0g	2015	2.0	1184	EGR + SCR
S3.0h	-	3.0	1861	EGR + SCR
S3.0i	-	3.0	1393	EGR + SCR

4.1.1.1 NO_x abatement technology

The vehicles in the test fleet were all equipped with a Diesel Oxidation Catalyst and one of the three main diesel NO_x abatements; Exhaust Gas Recirculation (EGR), Lean NO_x Traps (LNT) and Selective Catalytic Reduction (SCR). Further explanation of each technology can be found in Chapter 2. All vehicles were fitted with EGR, as are all diesel vehicles Euro 5 and above. The majority were fitted with EGR in combination with either LNT or SCR. Vehicles labelled as EGR used only EGR. The mixture of abatement technologies in the test fleet (7 EGR, 19 EGR + LNT, 13 EGR + SCR) was representative of the distribution of these technologies in the 2014 EU diesel passenger car sales mix (ICCT, 2015). All vehicles were also fitted with a Diesel Particulate Filter (DPF) which has been standard for diesel vehicles since Euro 5.

4.1.1.2 Engine displacement

As discussed in Chapter 2 diesel engines tend to be larger than petrol engines. The average engine displacement of the Euro 6 test fleet was 2ℓ, and the engines ranged between 1.4 ℓ - 3 ℓ. This is representative of the distribution of engine sizes in both the UK (**Table 4-2**) and EU as a whole.

Table 4-2. Distribution of engine displacements in the test fleet compared to UK 2015 sales (DfT, 2015c)

	≤1 ℓ	**1 to ≤ 1.55 ℓ**	**>1.55 to ≤ 2 ℓ**	**>2 ℓ**
UK 2015 sales diesel cars (%)	0.1%	12%	65%	23%
Test fleet share (%)	0	10%	72%	18%

4.1.1.3 Mileage of test fleet

The majority of vehicles in the test fleet had a low mileage, the average mileage at the start of the trip was 4105 (sd. 3000) km with most vehicles having an initial mileage of between 2000 - 5000 km. Historically manufacturers recommended an engine “break-in” period for new vehicles of 100 miles. In modern vehicles the engine “break-in” is part of the production process, meaning all engines in the test fleet were properly broken-in and settled into normal operation. As the vehicles in the test fleet had a relatively low mileage, emissions degradation (usually observed > 50,000 km (Borken-Kleefeld & Chen, 2015)) was not a consideration.

4.1.2 Test route

The test route was comprised of urban and motorway driving in the Greater London area. Each test followed a similar route with slight variations due to unavoidable circumstances such as road works or traffic. The average trip length was 84.3 (sd. 16.6) km and the average duration was 112 (sd. 22) minutes, of which roughly three quarters was urban driving and one quarter motorway.

4.1.2.1 Urban and motorway section selection

To assess the differences between urban and motorway emissions the relevant sections of the route were identified and analysed separately. Sections were identified using GPS co-ordinates and purpose built software in the statistical package R. As with the manufacturer of the vehicles, the exact location of the test route is commercially sensitive. **Figure 4-1** is a schematic of the test routes general characteristics. The urban section comprised A, B and C roads (UK) in residential areas with a speed limit of 50 km h^{-1} (30 mph). The motorway section was an M road with a speed limit of 110 km h^{-1} (70 mph).

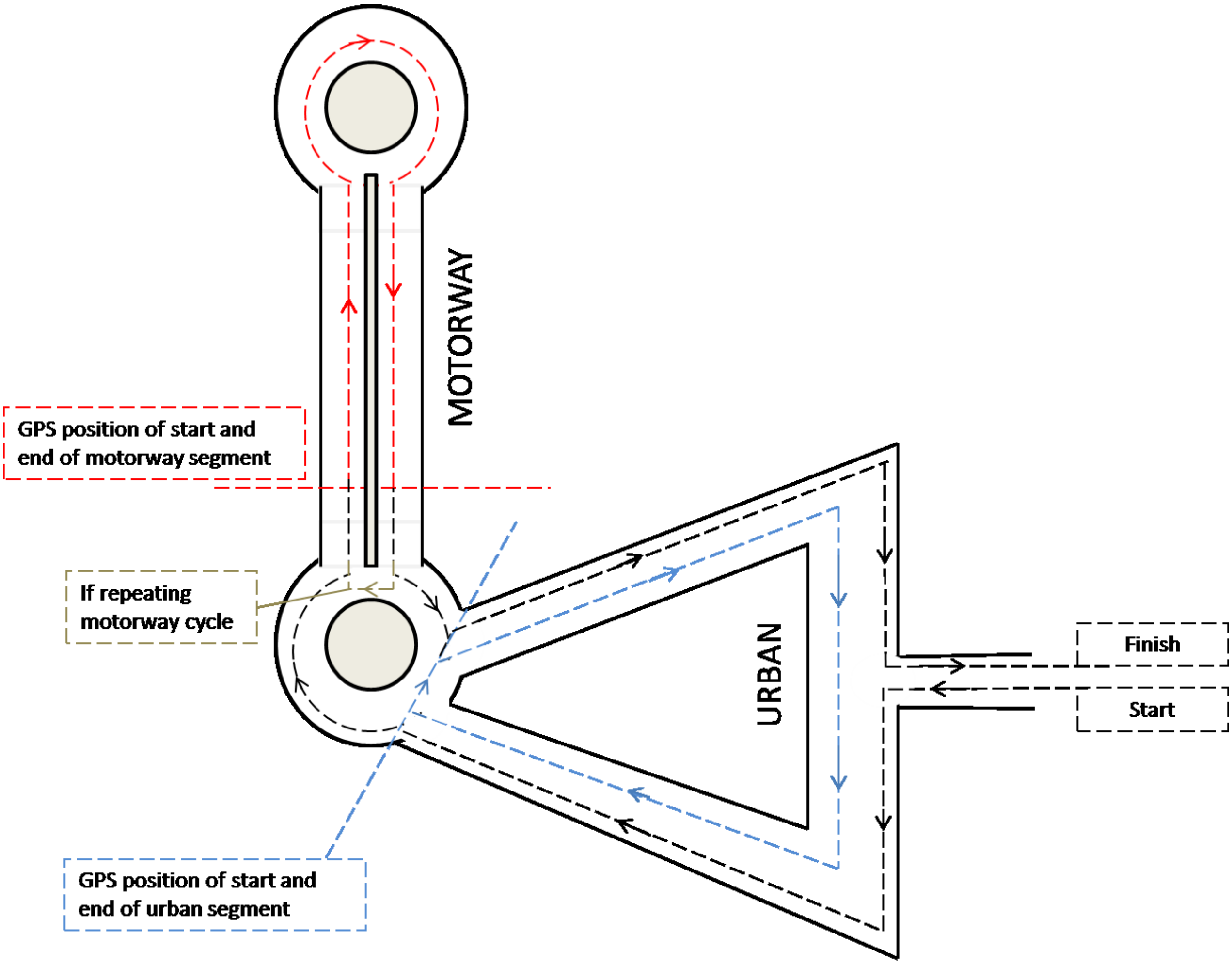

Figure 4-1. Illustration of test route and section selection

The test routes started at the test centre in an urban area, vehicles first completed an urban loop. This was followed by a period down and back a stretch of motorway, usually repeated twice. The vehicles then repeated the urban loop and ended back at the test centre. The purpose built software identified when the vehicle passed onto the motorway using GPS, as illustrated in **Figure 4-1**. Thus each trip was broken down into its motorway and urban constituent parts. This does not follow the RDE test procedure detailed in Regulation (EU) 2016/646. The aim in this chapter is to explore real world (particularly urban) emissions exactly as they occur. For that reason the data processing and dynamic conditions specified in (EU) 2016/646 were not used to clean the data. However, the RDE test procedure is followed in Chapter 6 and comparisons are drawn between the two methods.

4.1.2.2 Driving characteristics

Table 4-3 lists the average characteristics of the test trips and the constituent urban and motorway sections. For comparison the characteristics of the NEDC (New European Driving Cycle) are also listed. As the test route was relatively flat (< 60 m elevation gain over 85 km) the effect of road gradient is not considered here.

Table 4-3. Characteristics of test route and sections

	Trip	Urban	Motorway	NEDC
Route distance [km]	84.3 (sd. 16.6)	34.8 (sd. 6.0)	37.7 (sd. 5.3)	11.02
Avg. duration [minutes]	112 (sd. 22)	80 (sd. 17)	22 (sd. 3)	13
Avg. vehicle speed [km h^{-1}]	45.6 (sd. 4.9)	26.5 (sd. 2.9)	103.8 (sd. 5.6)	34 (sd. 31)
Avg. RPA [m s^{-2}]	0.25 (sd. 0.12)	0.26 (sd.0.12)	0.15 (sd. 0.16)	0.15 (sd. 0.03)
Avg. VSP [kW t^{-1}]	3.9 (sd. 8.7)	1.5 (sd. 6.0)	11.8 (sd. 12.3)	
Max elevation [m]	54.7 (sd. 22.2)	28.6 (sd. 7.6)	51.9 (sd. 22.4)	-
Min elevation [m]	-4.7 (sd. 6.6)	-4.5 (sd. 6.6)	4.7 (sd. 4.0)	-
Idle [%] (time) ($v \leq 2$km h^{-1})	10.2 (sd. 4.8) %	13.7 (sd. 6.7)%	1.5 (sd. 0.8)	22.9
Low [%] (time) ($2 < v \leq 50$km h^{-1})	62.5 (sd. 7.4) %	84.0 (sd. 6.7)%	5.1 (sd. 5.1)	55.3
Medium [%] (time) ($50 < v \leq 90$km h^{-1})	8.7 (sd. 3.0) %	2.4 (sd. 2.5)%	13.7 (sd. 4.8)	14.6
High [%] (time) ($v > 90$km h^{-1})	18.6 (sd. 4.5) %	0 (sd. 0)%	83.7 (sd. 7.6)	7.2

4.1.2.3 Speed distribution

PEMS tests capture a large range of driving characteristics that are not well represented in the laboratory based NEDC. **Figure 4-2a** shows the cumulative frequency speed distributions for the PEMS trips (grey) and compares these to the NEDC (red) and the new WLTC (Worldwide harmonized Light vehicles Test Cycle) (blue). The WLTC and PEMS speed distributions were much smoother than the NEDC, though the distribution of speeds over the PEMS trips were similar to the NEDC and WLTC. However, as discussed in Chapter 2, it is now known differences in driving dynamics account for only a small part of the difference between RDE and NEDC NO_x emissions (Degraeuwe & Weiss, 2017).

Figure 4-2b shows the cumulative frequency speed distributions for the urban and motorway sections. The urban sections (green) fell within the range 0 – 50 km h^{-1} whereas the vast majority of the motorway sections (orange) fell within the range 70 – 110 km h^{-1}. As stated previously emissions vary significantly with driving characteristics including speed. It was therefore important to have consistency in the speed distribution between different trips to ensure comparisons were fair. Plotting the cumulative frequency speed distribution (**Figure 4-2**) is a useful way to compare large numbers of trips and ensure continuity and comparability. With the exception of one motorway section all speed distribution lines formed distinct groups. This indicated a good level of correlation in the speed distributions of trips from different vehicles. The one motorway section in **Figure 4-2b** that stood out from the group had an average speed of 80 (sd. 36) km h^{-1}. This was below the motorway section average of 103.8 (sd. 5.6) km h^{-1}. This test coincided with road works on the motorway. As the aim of this analysis is to represent accurately real world emissions (including congestion which occasionally slows motorway traffic) this vehicle was not excluded from the study. Emissions from this vehicle were not found to be anomalous.

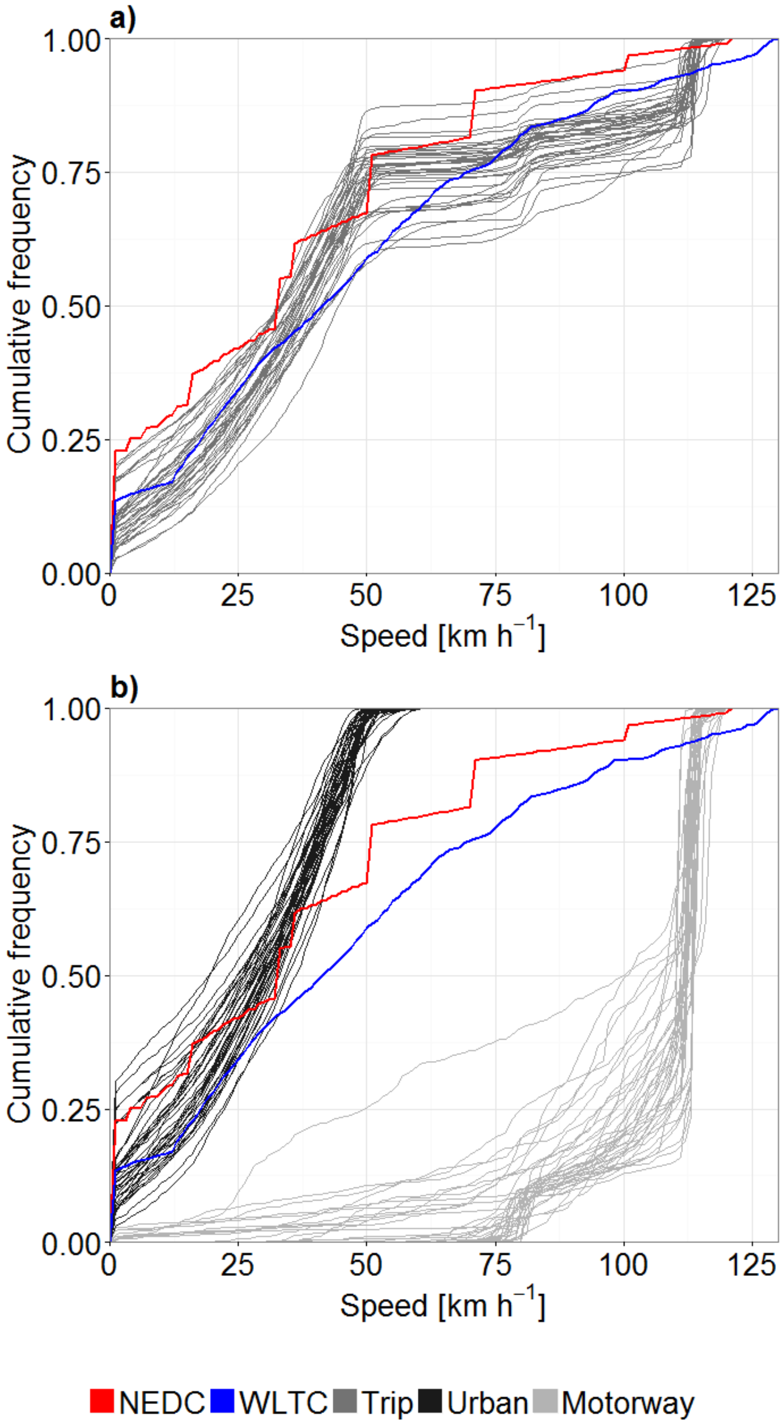

Figure 4-2. Cumulative frequency speed distribution for a) whole trip and b) urban and motorway sections (test cycle data from Tutuianu et al (Tutuianu *et al.*, 2013))

4.1.2.4 Relative Positive Acceleration (RPA)

Relative Positive Acceleration (RPA) is a parameter used as a proxy for driving style and to assess the comparability of different measurement trips. RPA is the integral of the product of instantaneous speed and positive acceleration. This chapter adopts the definition of RPA used by Weiss *et al.*, (2011). This definition is slightly different to the definition in (EU) 2016/646 (the new RDE type approval legislation). Weiss et al. divide the whole trip into numerous "sub-trips" and calculate the RPA of each whereas (EU) 2016/646 calculates the RPA of the trip as a whole. The (EU) 2016/646 definition of RPA will be used in Chapter 6. **Equation 4-1** was used to calculate the RPA plotted in **Figure 4-3**.

$$\mathrm{RPA} = \frac{\int_0^{t_j} (v_i \times a_i)\,dt}{x_j}$$

Equation 4-1. Relative Positive Acceleration (Weiss *et al.*, 2011a)

t_j = time

x_j = distance of sub-trip *j*

v_i = speed during each increment *i*

a_i = Instantaneous positive acceleration during each increment *i* contained in the sub-trip *j*

A "sub-trip" is defined as "*any part of the test route, in which the vehicle speed is at least 2 km h^{-1} for a period of at least 5 seconds"*. Meaning when the vehicle speed falls below 2 km h^{-1} one "sub-trip" ends and when a vehicle accelerates again above 2 km h^{-1} for a duration of 5 seconds a new "sub-trip" begins.

Figure 4-3 shows the RPA of urban and motorway sections and compares these to the NEDC (red) and WLTC (blue).

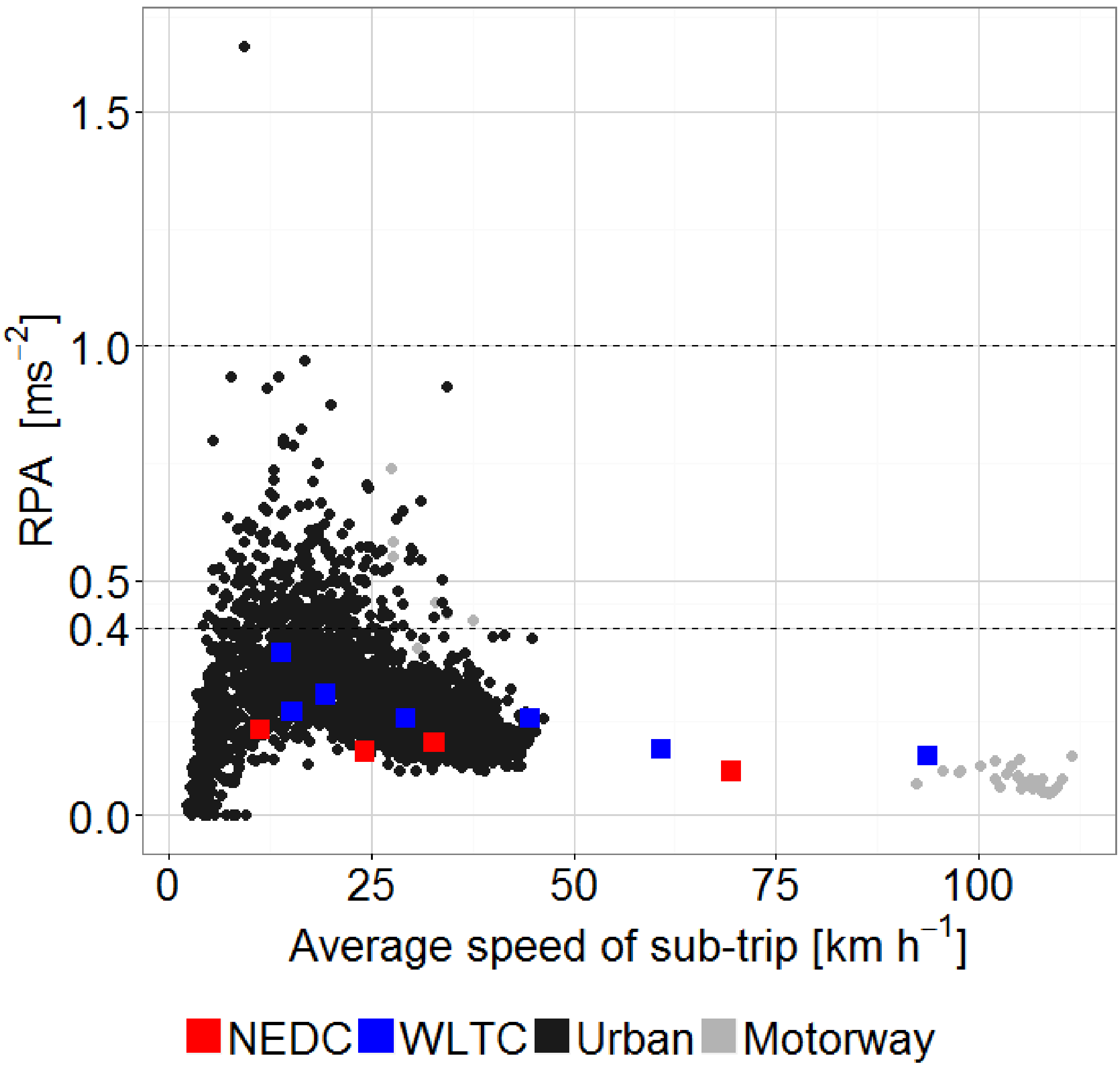

Figure 4-3. Relative Positive Acceleration (RPA) of urban and motorway sections (test cycle data from Tutuianu et al (Tutuianu *et al.*, 2013))

RPA is a measure of acceleration as well as speed. As a result there was a much wider range of RPA for real world driving than for laboratory based test cycles where acceleration/ deceleration is controlled. **Figure 4-3** shows the RPA range of the WLTC is much more representative of real driving than the NEDC. The majority of the PEMS sub trips fell into the “low” RPA bracket (lowest horizontal dashed line). “Low” is classified as RPA within the range 0.1 - 0.4 m s^{-2} and velocity under 50 km h^{-1}. “Extreme” is classified as RPA above 1 m s^{-2} at low velocity or a low RPA at a velocity above 120 km h^{-1} (Weiss *et al.*, 2011a). Only one PEMS sub-trip was classed as being “extreme”. The WLTC defines “normal European driving” as having an average RPA of 0.2 m s^{-2} for urban driving and 0.1 m s^{-2} for motorway driving (Tutuianu *et al.*, 2015)). The PEMS trips average RPA was 0.25 (sd. 0.12) m s^{-2} for urban and 0.15 (sd. 0.16) m s^{-2} for motorway sections. These values fit the definition of “normal European driving”, meaning the tests in this study were representative of normal European driving.

4.1.2.5 Vehicle Specific Power (VSP)

Vehicles specific power (VSP) is another metric used to characterise driving behaviour. VSP is an instantaneous measure of a power per unit mass of a vehicle. It is a function of vehicles speed, acceleration/deceleration and road gradient (**Equation 4-2**). Emissions show strong correlation with VSP (Zhai, Frey & Rouphail, 2008; Carslaw *et al.*, 2013).

$$VSP = v[1.1a + 9.81(\sin(\arctan(r))) + 0.132] + 0.000302v^3$$

Equation 4-2. Vehicle specific power (Jiménez-Palacios, 1999)

VSP = vehicle specific power in [kW t^{-1}]

v = velocity [m s^{-1}]

a = acceleration [m s^{-2}]

r = road gradient [slope]

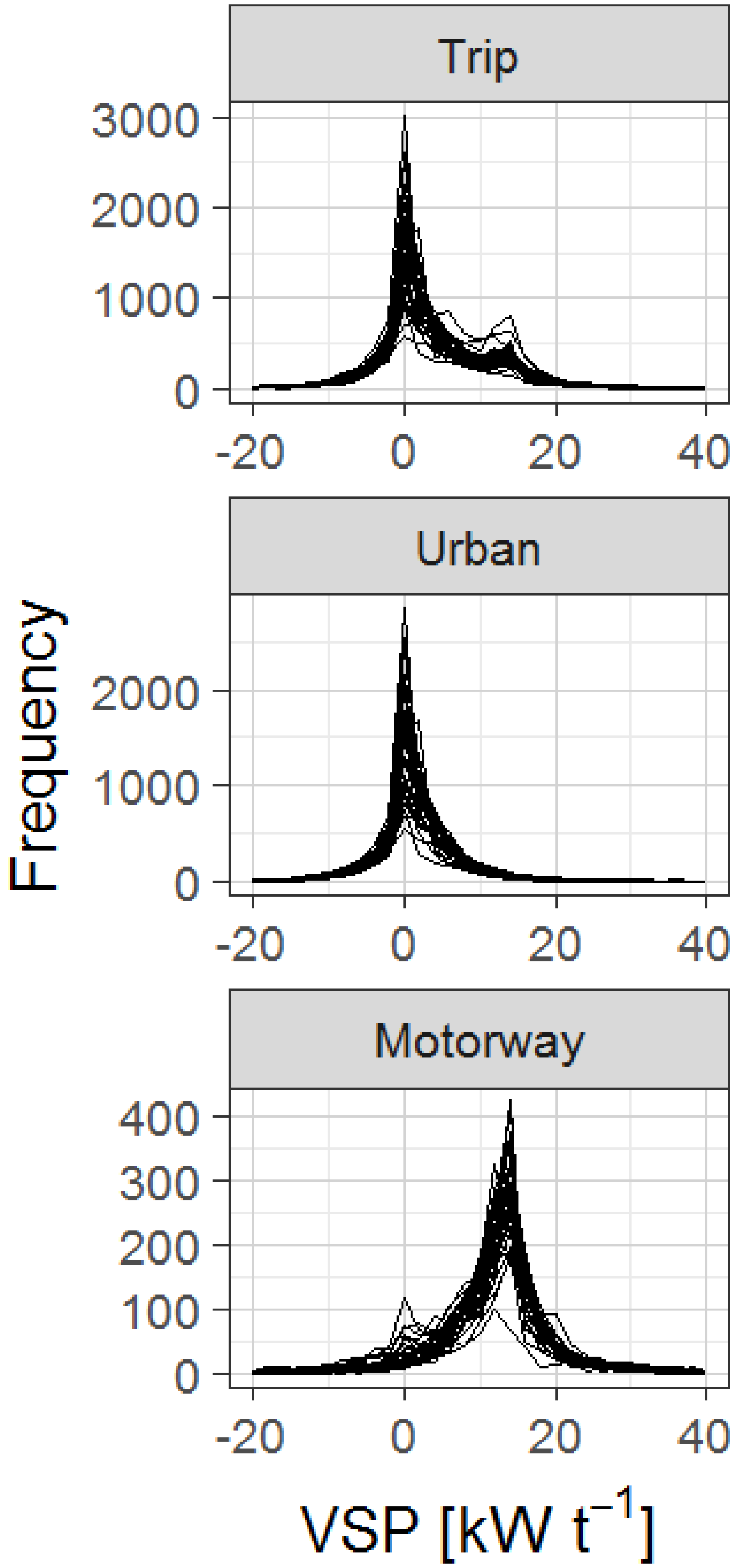

Figure 4-4. Vehicle Specific Power for whole trip, urban and motorway sections

The VSP frequency distribution for each trip/section is represented by a black line in **Figure 4-4**. The distributions formed distinct groups indicating consistency throughout the tests. The motorway sections VSP profiles also highlight the trip for which there was congestion on the motorway. This is an example of how looking at the VSP frequency distribution of different RDE trips can be a quick way to identify similarities and differences.

The VSP frequencies for the whole trip had a bimodal pattern with a central peak corresponding to urban driving and a secondary positive skewed peak corresponding to motorway. The urban peak was higher as the test route had a much greater frequency of urban driving (~75%). The urban sections had much lower VSP with a bigger proportion negative due to lower speeds and a greater prevalence of sharp deceleration during urban driving.

4.1.3 Ambient temperature

The tests were performed over a range of ambient temperatures (3 - 29˚C) within the normal range for Europe (EC, 1998). Previous studies found correlation between NO_x emissions and ambient temperature, with NO_x emissions increasing as temperature decreased (DfT, 2016d; Kwon *et al.*, 2017). TNO in the Netherlands concluded this is due to the “thermal window” phenomena whereby manufacturers disable NO_x controls at a certain temperature theoretically to protect the engine (Kadijk *et al.*, 2016; DfT, 2016d). Some manufacturers have reportedly been exploiting the “thermal window” regulation and disabling NO_x controls at temperatures as high as 17 ˚C (T & E, 2016), the UK average ambient temperature is 9 ˚C (DfT, 2016d). Results were not corrected for ambient temperature as the aim was to accurately present real world European driving emissions which must cover a range of temperatures to be representative.

4.1.4 Portable Emissions Measurement System (PEMS) testing

PEMS testing was conducted by Emissions Analytics using a SEMTECH-DS developed by Sensors Inc (Sensors Inc, 2010). The SEMTECH-DS consists of a flow meter that measures the volume of exhaust emissions connected to multiple gas analysers. The SEMTECH-DS contains a GPS receiver which records latitude, longitude, altitude and vehicle speed. There is also an interface that connects to the vehicles on-board engine diagnostics (OBD) port. NO and NO_2 are measured simultaneously and separately using Non-Dispersive Ultraviolet Light (NDUV). NO is reported as NO_2 and NO_x is calculated as the sum of both (Sensors Inc, 2014).

The SEMTECH-DS was installed and operated following manufacturers recommendations. A leak test along with zero and span (known gas concentration) calibrations were performed before and after each test-run. Results were deemed invalid if the zero or span test at the end of the trip had an error greater than 3%. The SEMTECH-DS fulfils both EU and US testing requirements and previous studies have found SEMTECH-DS to be accurate within the range of lab based testing methods (EPA, 2008b, 2008a; EC, 2011; Weiss *et al.*, 2012).

The PEMS unit was powered by external batteries meaning engine operation was not affected, apart from the additional weight. The PEMS itself weighs approximately 95 kg. With the addition of drivers the total load weight was 220 kg. This was uniform for all tests and supplemented by additional weights if required. Studies have found that this additional weight affects the power mass ratio of a vehicle and can potentially increase CO_2 emissions by up to 3%; it is reasonable to assume a similar margin for NO_x (Fontaras & Samaras, 2010; Weiss *et al.*, 2012). This is less than the 10 – 20 % variability associated with any PEMS measurement (Kadijk *et al.*, 2016). The 220 kg is also roughly equivalent to 2 – 3 passengers, and therefore not outside a vehicles normal operating weight.

As mentioned previously, driving style can have a large impact on emissions. The tests were performed by drivers trained in "normal" non-aggressive driving as evidenced in **Figure 4-3**. The same drivers performed all the tests to ensure driving style was consistent.

The SEMTECH-DS samples at a frequency of 1Hz (i.e. 1 second time resolution). This allows for real-time scrutiny of instantaneous emissions which is not possible using other emissions measurement techniques. **Figure 4-5** is an extract from the urban section of vehicle L2.0p showing instantaneous NO_x (blue) on the left-hand y axis and speed (red) on right. **Figure 4-5** shows NO_x emissions were delivered in peaks that correlated with acceleration events.

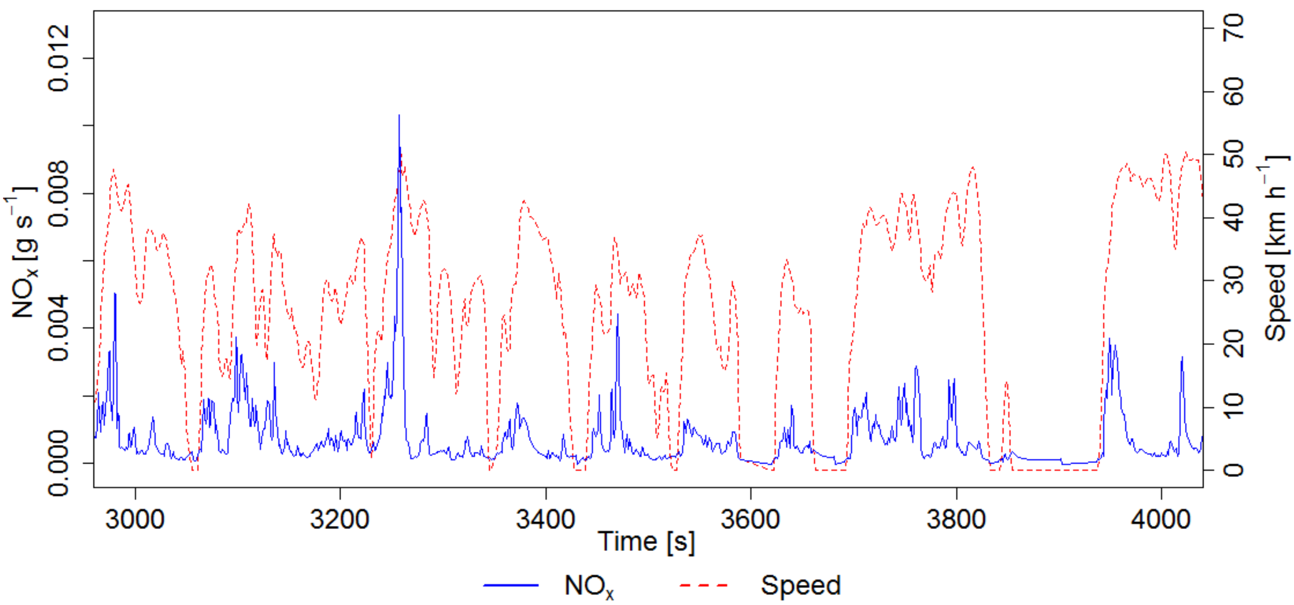

Figure 4-5. Extract from vehicle L2.0p urban section showing instantaneous NO_x and speed

4.1.5 COPERT emissions factors

The speed dependent emissions factors used in this research were the Euro 6 COPERT 4v11 NO_x emissions factors introduced in September 2014[1]. COPERT derives emissions factors from the Handbook on Emission Factors of Road Transport (HBEFA). HBEFA has been developed from chassis dynamometer tests using the ERMES drive cycle. The PHEM model (Passenger car and Heavy duty vehicle Emission Model) is then used to expand the chassis dynamometer emissions factors to cover all driving conditions. For COPERT 4v11 the Euro 6 emission factors were inferred from measurements of 20 vehicles of which only 6 were diesel (Rexeis *et al.*, 2013; Pastramas *et al.*, 2014).

The aim in this chapter is to compare the COPERT estimates for NO_x and NO_2 to the real world emissions PEMS measurements using the speed profile of the trips. COPERT emissions factors are speed dependent but are not designed to be used with instantaneous speed. When modelling with COPERT the road network is broken down into a series of links for which the average speed is known. The speed dependent emission factor corresponding to this average speed is then applied uniformly to the entire link.

To ensure the comparison between COPERT and PEMS was accurate and fair the COPERT estimates were calculated using the speed profile from each individual PEMS trip. For this comparison the approach of the INCERT (Interface for the Comparison of Emissions from Road Transport) model (Kousoulidou *et al.*, 2013) was adopted using purpose built software in the statistical package R. The INCERT model

[1] COPERT released version 5 in October 2016 after this research had been published

splits the PEMS real world speed profile into links of equal length. The average speed of each link is then calculated and applied to that whole link, emulating application of the COPERT model.

The reliability of COPERT increases with link length up from a minimum of 400m (Samaras *et al.*, 2014). This study used a link length of 1 km. This technique is illustrated in **Figure 4-6**. The blue line is the PEMS speed profile for a 30 minute extract from vehicle L2.0p. The red line is the mean speed of each 1 km link. This extract includes part of the motorway section where each 1 km link has a much shorter duration and there is less variability in the speed. The extract also includes part of the urban section where the PEMS profile has much more variability and there is far more acceleration and deceleration. The 1 km links in the urban section had much longer durations. COPERT has limited modelling capability at low speeds (< 10 km h^{-1}). Using the 1 km links avoids modelling in this range.

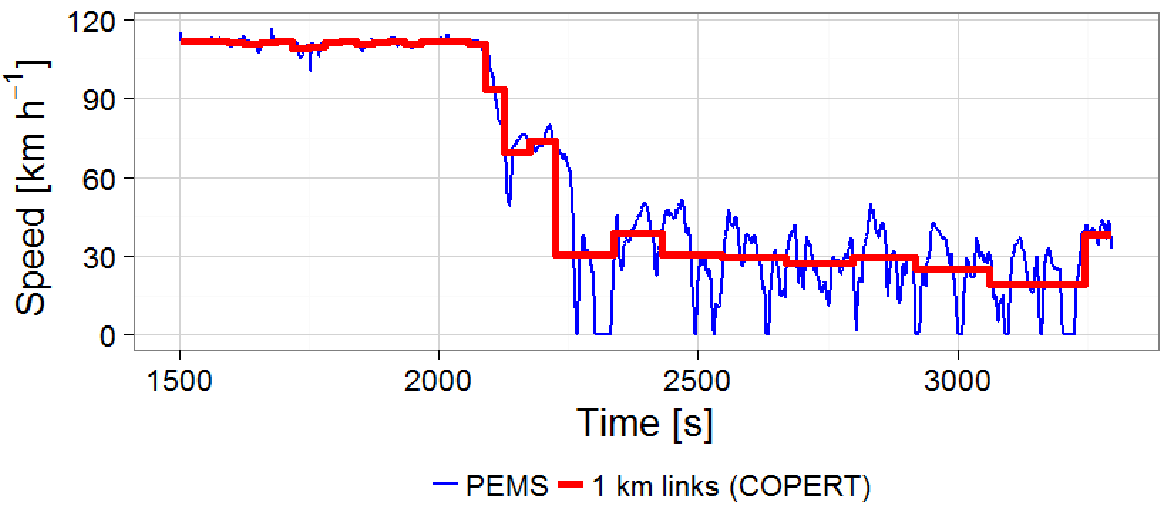

Figure 4-6. PEMS and 1 km link speed profile generated to calculate COPERT emissions estimates

Once the required speed profile was generated the appropriate COPERT emissions factor was assigned to each link. This was done using the iMove model (Valiantis, Oxley & ApSimon, 2007). iMove is a purpose built software that applies COPERT emissions factors to custom speed profiles. iMove is embedded in the BRUTAL model (Oxley *et al.*, 2012), the road transport sub-model of the UKIAM (Oxley *et al.*, 2013)).

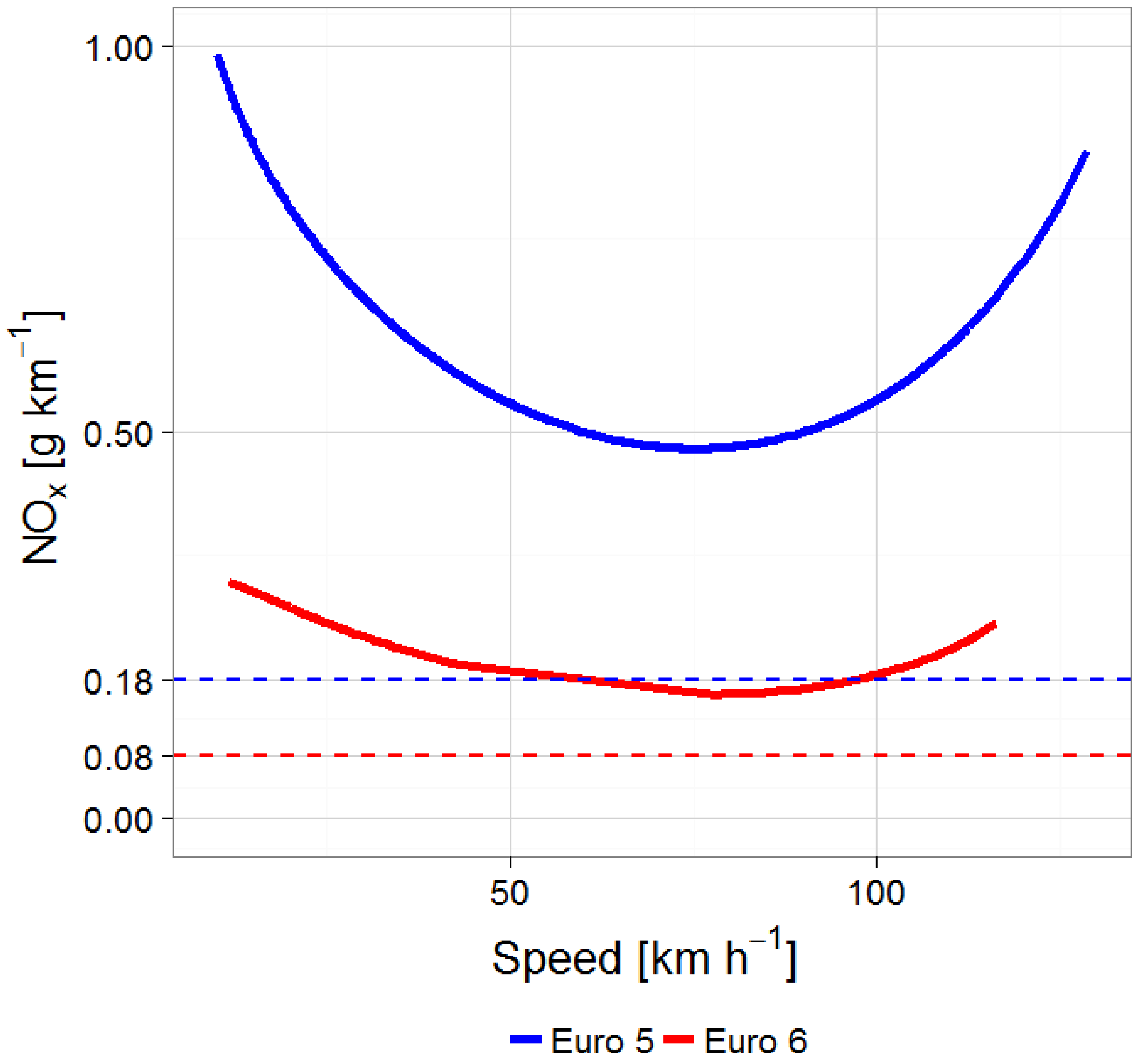

Figure 4-7. COPERT 4v11 speed dependent emissions curves for NO_x, dashed lines = type approval limits

Figure 4-7 shows the Euro 5 (blue) and Euro 6 (red) COPERT 4v11 speed dependent emissions factors for NO_x along with the relevant type approval limits. COPERT 4v11 estimated an approximate -64% reduction in NO_x between Euro 5 and Euro 6 and the relationship with speed became less pronounced (the curve became flatter). The Euro 6 emission factors were between 2 and 4 times the emissions limit of 0.08 g km^{-1}. COPERT does not have a specific function for NO_2, it assumes a constant proportion of NO_x. For Euro 6 diesel COPERT estimates a flat rate of 30% primary NO_2 (Pang, 2015).

4.1.6 Data analysis

4.1.6.1 Emissions factor calculations

Average NO_x emissions in this chapter are stated in g km^{-1}. PEMS record emissions in g s^{-1}. The g km^{-1} emissions were calculated by summing the total trip/ section emission in g s^{-1} to get a total in grams and then dividing by distance travelled. As the PEMS did not record the distance travelled this was calculated using **Equation 4-3**. This distance was also verified using the "geosphere" package in R which calculates the total distance travelled on a route using GPS co-ordinates.

$$S = \sum_{i=1}^{n} v_i \times t_i$$

Equation 4-3. Calculating distance of each section

S = total distance of section [m]

v_i = velocity [m s^{-1}] at time *i*

t_i = time [s] (1 second)

4.1.6.2 Acceleration

Following EU Commission Regulation (EU) 2016/646 acceleration was calculated using **Equation 4-4**:

$$a_i = \frac{v_{i+1} - v_{i-1}}{2 * 3.6} \qquad i = 1 \; to \; N_t$$

Equation 4-4. Acceleration (EC, 2016a)

a = acceleration in [m s^{-2}]

v = velocity [km h^{-1}]

N_t = number of samples

4.1.6.3 Boxplots

Many of the results in this chapter are presented in boxplots. Each point on the plot represents an individual vehicles' measurement. Large red triangles represent the mean of each category/bin. The thick horizontal line in the middle of each box represents the median (middle data point) of each category/bin. The box represents the middle half (i.e. the second and third quarter) of the data points. The difference between the highest value data and lowest value data point within the box is known as the interquartile range (IQR). The whisker extends to data points within 1.5 x IQR. Any data point more than 1.5 x IQR is an outlier. In this study the width of the box has no significance. Red dashed horizontal lines represent type approval limits.

4.1.6.4 Cold starts

The cold starts emissions (defined as the first 300 seconds of each PEMS test (Weiss *et al.*, 2011a)) were not included in this analysis. This was due to a lack of continuity between the PEMS tests, the majority were from warm start though some engines were soaked overnight (left outside overnight to ensure after treatment system, engine coolant and engine were completely cooled to ambient temperature). Cold starts will be discussed in more depth in Chapter 6.

4.2 Results

This section first presents the results relating to PEMS measurements, then analysis of the separate urban and motorway sections and effect of driving mode and acceleration followed by the comparison with COPERT speed dependent emissions factors.

4.2.1 PEMS measurements

The PEMS measured average NO_x and NO_2 emissions from each trip is presented in **Figure 4-8**. Results are presented in g km^{-1} on the left-hand y axis and deviation ratio on the right y axis. The Euro 6 current type approval limit is marked in red and the Euro 6d-TEMP NTE RDE limit is marked in blue. The numerical results from **Figure 4-8** are presented in **Table 4-4**.

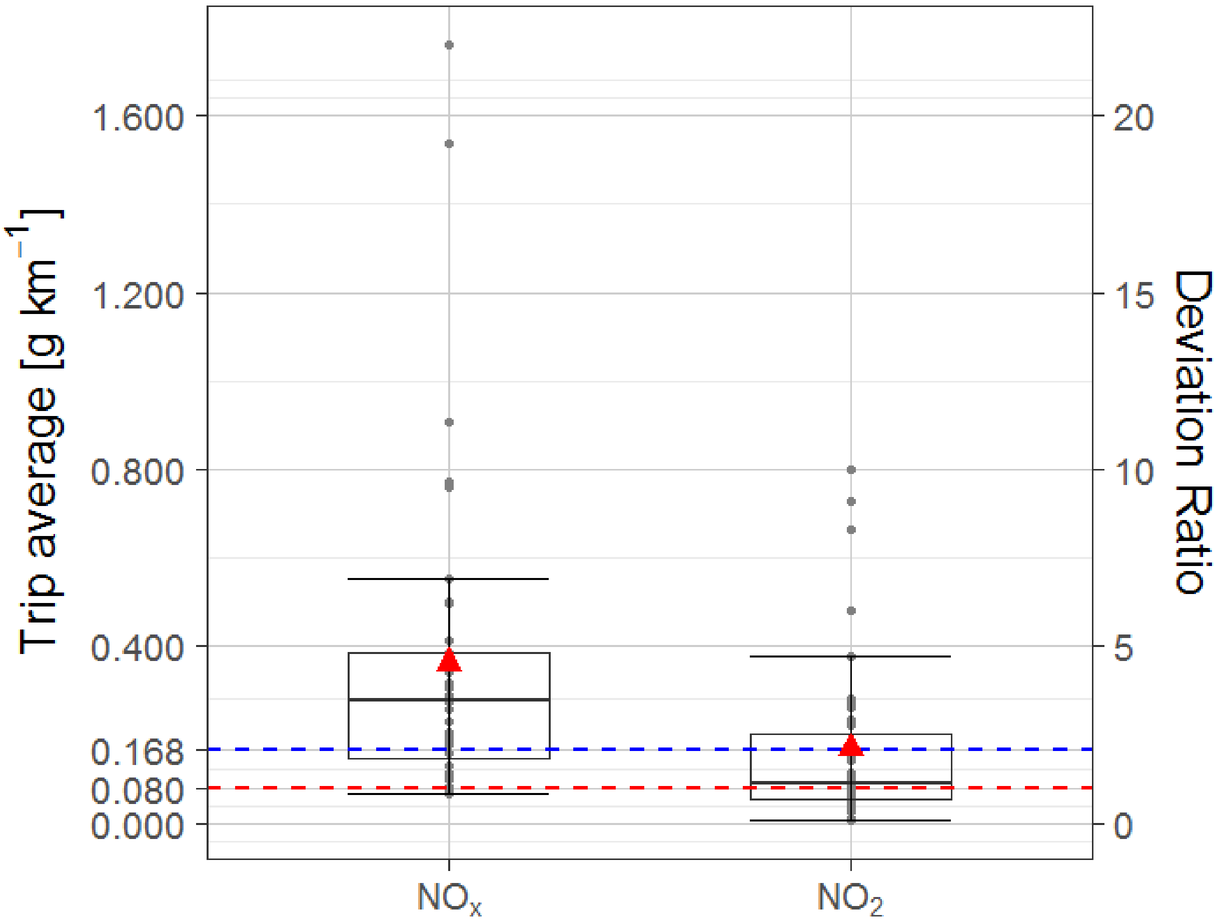

Figure 4-8. Trip average NO_x and NO_2 emissions in g km^{-1} with Euro 6 type approval limit (red) and Euro 6d-TEMP RDE type approval (blue)

Table 4-4. Average trip NO_x, Deviation Ratio, NO_2 and fNO_2

NO_x [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2** [%]
0.36 (sd. 0.36)	4.5 (sd. 4.5)	0.17 (sd. 0.19)	44 (sd. 20)

The average NO_x emission of 0.36 (sd. 0.36) g km^{-1} was 4.5 times the Euro 6 type approval limit. Furthermore, it was 1.4 times the Euro 4 type approval limit. The average NO_2 emission of 0.17 (sd. 0.19) g km^{-1} was twice the Euro 6 type approval limit for total NO_x. The vast majority of vehicles did not achieve the Euro 6 type approval limit during real world driving, only 2 of the 39 vehicles (5%) met the limit. The average emissions of both NO_x and NO_2 exceeded the Euro 6d-TEMP limit, though 11 vehicles (28%) were able to meet it. A high proportion of NO_x was emitted directly as NO_2 with an average of 44 (sd. 20) %. The high levels of NO_2 in modern diesel vehicles is attributed to the presence of oxidising catalysts (DOCs) in the emissions control devices that oxidise NO to NO_2 (DfT, 2016d). Higher levels of NO_2 in exhaust gases also enhance DPF regeneration and improve the efficiency of SCR processes, reducing total NO_x emissions (Wang *et al.*, 2015).

There was huge variability in both NO_x and NO_2 emissions. The highest NO_x emission was 26 times higher than the lowest and the highest NO_2 emission was over 100 times the lowest. Five vehicles were classed as outliers (outside of 1.5 x the IQR). For both NO_x and NO_2 the mean was higher than the median. This indicated that the 5 outliers were having a substantial effect on the group mean and removing them would deliver a substantial benefit, this idea is explored further in the Discussion section of this chapter.

The highest NO_x emitter was vehicle S3.0h with an average emission of 1.76 g km^{-1}, a deviation ratio of 22. The highest NO_2 (g km^{-1}) and fNO_2 were from vehicle L2.0j, 0.80 g NO_2 km^{-1} and 88% fNO_2 respectively.

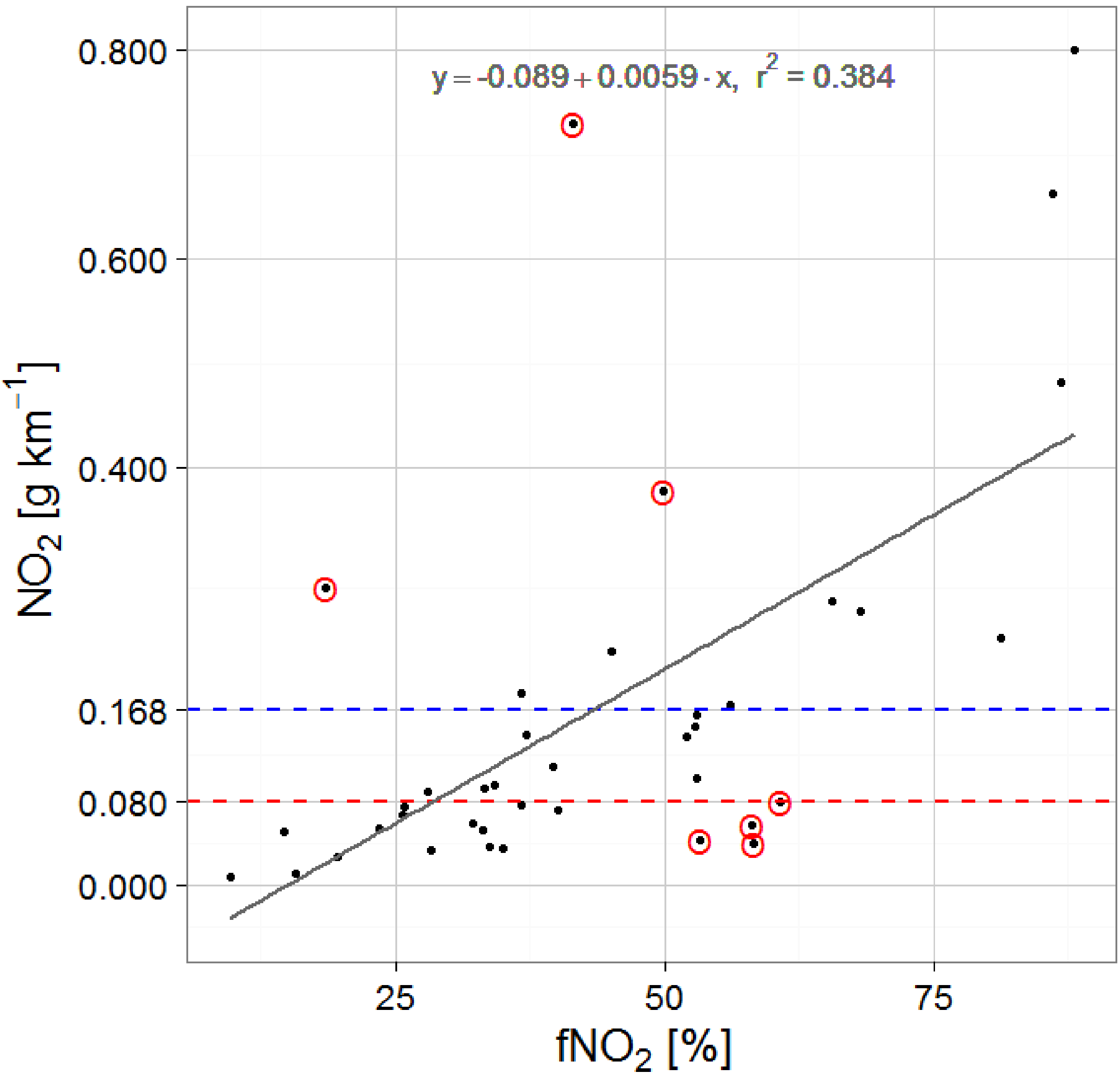

Figure 4-9. Trip average NO_2 against ratio fNO_2

Figure 4-9 illustrates the relationship between fNO_2 and NO_2 in g km^{-1}. There was moderate correlation, though high variance resulted in a relatively low R^2 value. This was due to instances of high absolute NO_2 emission at below average fNO_2 and some relatively low absolute NO_2 at above average fNO_2 (circled in red). This highlights the importance of discussing NO_2 in terms of g km^{-1} rather than solely as a fraction of NO_x.

4.2.1.1 NO_x abatement technology

This section compares average emissions from the different NO_x abatement technologies in the test fleet; Exhaust Gas Recirculation (EGR), Lean NO_x Traps (LNT) and Selective Catalytic Reduction (SCR).

Figure 4-10 compares the NO_x, NO_2 and fNO_2 of the three abatement technologies, the numerical results are presented in **Table 4-5**. With the exception of fNO_2 for SCR vehicles there was no difference between emissions from the different technology vehicles. This finding is in keeping with a remote sensing study by *Carslaw & Rhys-Tyler (2013)* that found SCR no better than non-SCR technology in reducing NO_x. There was huge variation in emissions between vehicles using the same technology. For example both the highest (88%, vehicle L2.0j) and lowest (10%, vehicle L2.0a) measurements of fNO_2 were vehicles using LNT.

Table 4-5. Average trip NO_x, Deviation Ratio, NO_2 and fNO_2

NO_x control	**NO_x** [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2** [%]
ALL	0.36 (sd. 0.36)	4.5 (sd. 4.5)	0.17 (sd. 0.19)	44 (sd. 20)
EGR	0.44 (sd. 0.47)	5.5 (sd. 5.9)	0.12 (sd. 0.08)	31 (sd. 11)
LNT	0.31 (sd. 0.24)	3.9 (sd. 3)	0.16 (sd.0.23)	41 (sd. 24)
SCR	0.39 (sd. 0.45)	4.9 (sd.5.6)	0.20 (sd.0.19)	55 (sd. 13)

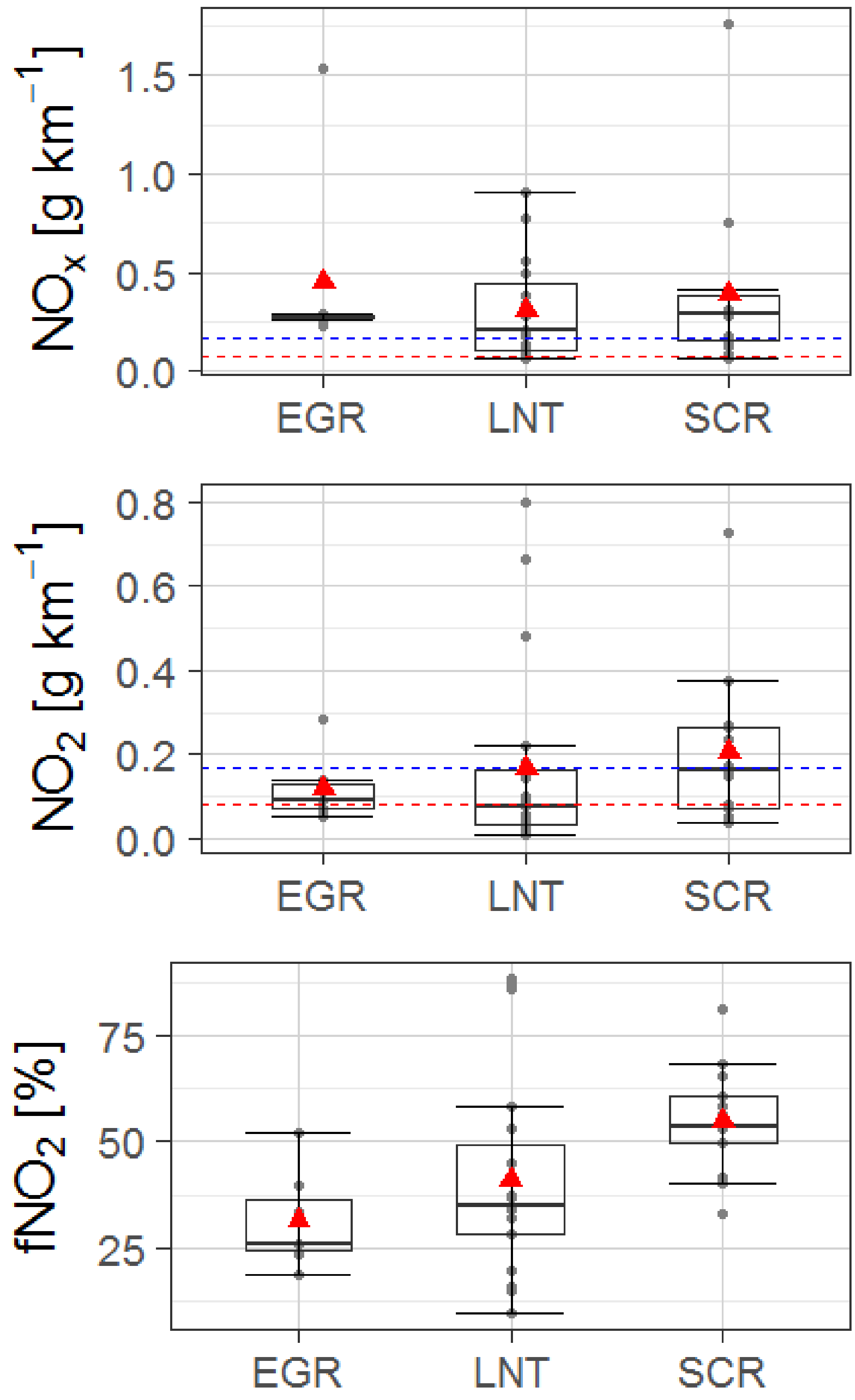

Figure 4-10. Trip average NO_x, NO_2 and fNO_2 by control technology

Analysis of Variance (ANOVA) was performed for NO_x, NO_2 and fNO_2. The only statistically significant difference between the three abatement technologies was in fNO_2. The mean fNO_2 for SCR (55 sd. 13 %) was ~1.5 times the combined EGR + LNT mean of 38 (sd. 21) % with a p value of 0.01. Though LNT had a lower mean than EGR the means were not statistically different due to high variance within the two groups.

The increase in fNO_2 for SCR did not result in a statistically significant difference in NO_2 in g km^{-1}. Though the mean NO_2 in g km^{-1} was higher for SCR than LNT and EGR combined, again high variance within the groups meant that statistically the means were not different.

There was no statistically significant difference in NO_x emissions between the abatement technologies. However, one vehicle fitted with SCR and another fitted with LNT met the Euro 6 type approval limit whereas no vehicle fitted with only EGR was able to. None of the EGR vehicles tested had NO_x emissions within the RDE type approval limit but neither did they have the highest NO_2 emissions. The 5 highest emitters of NO_2 were all SCR or LNT.

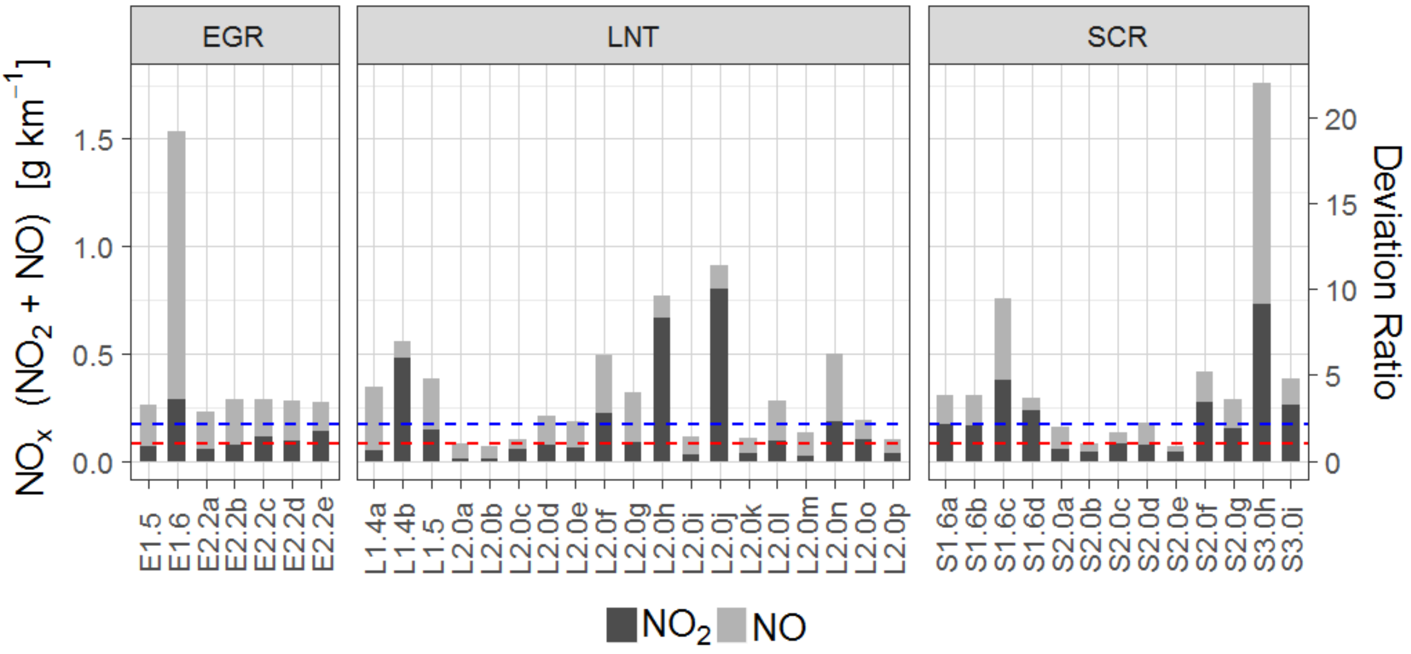

Figure 4-11. Bar chart showing average trip NO_x and NO_2 emissions in g km^{-1} by NO_x control technology (red dashed line = Euro 6b limit, blue dashed line = Euro 6c limit)

Figure 4-11 shows the NO and NO_2 components of each vehicles total NO_x. Many vehicles had NO_2 components (dark grey) far exceeding the limit for total NO_x and forming the majority of the emission. For each abatement technology the vehicles are plotted in order of engine displacement along the x axis.

4.2.1.2 NO_x emissions by engine displacement

There was no direct correlation between engine displacement and NO_x emissions, this is shown by the low R^2 value in **Figure 4-12**. However, 2 ℓ engines had lower NO_x emissions than all other sizes. The mean NO_x emission of the 2 ℓ engines was 0.26 (sd. 0.22) g km^{-1}. This was half the mean NO_x emission of non 2 ℓ engines. Similarly the mean NO_2 emissions of non 2 ℓ engines was 50% higher than for 2 ℓ engines. The lowest 15 NO_x emissions came from 2 ℓ engines as did 12 of the lowest 15 NO_2

emissions. However, this may be because 2 ℓ engines were the most common in the sample (23 out of 39 vehicles).

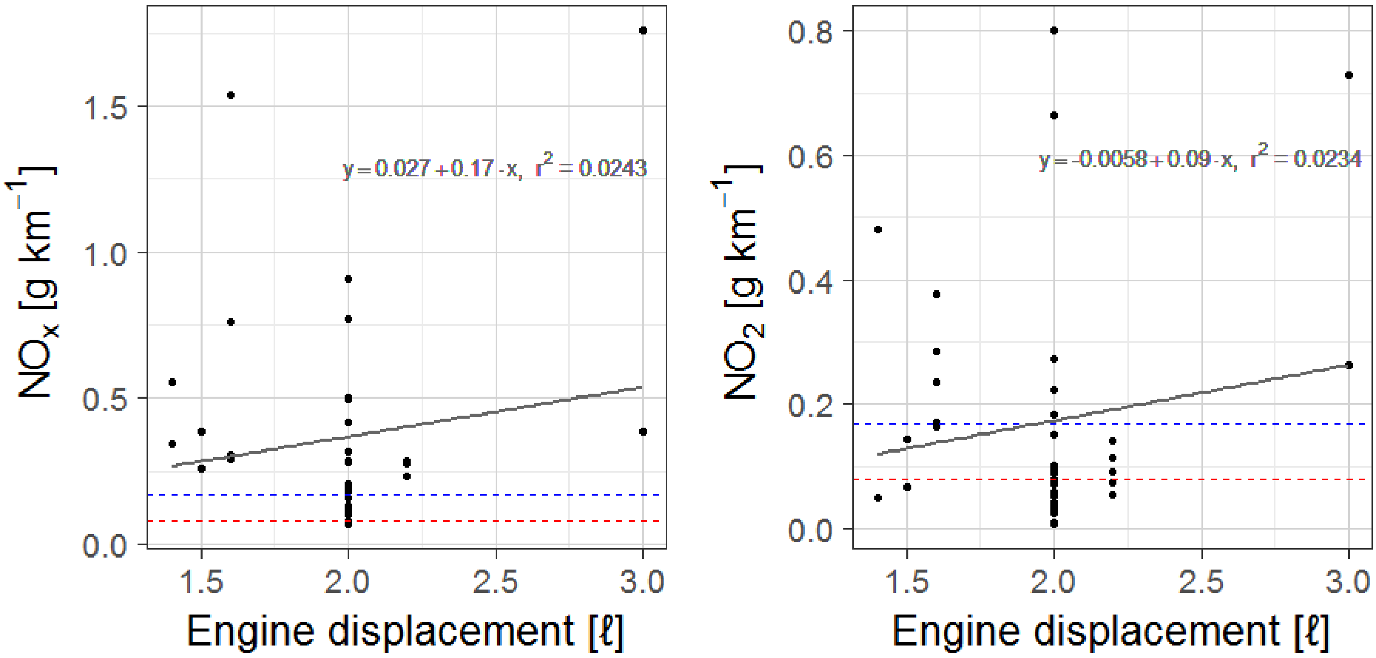

Figure 4-12. NO_x and NO_2 emissions by engine displacement

4.2.1.3 NO_x emissions by temperature

As discussed previous studies have found NO_x varied significantly with temperature, this was thought to be due to NO_x controls being disabled at low temperatures. **Figure 4-13** shows NO_x and NO_2 emissions by ambient temperature. There was no relationship between ambient temperature and NO_2 (**Figure 4-13c**).

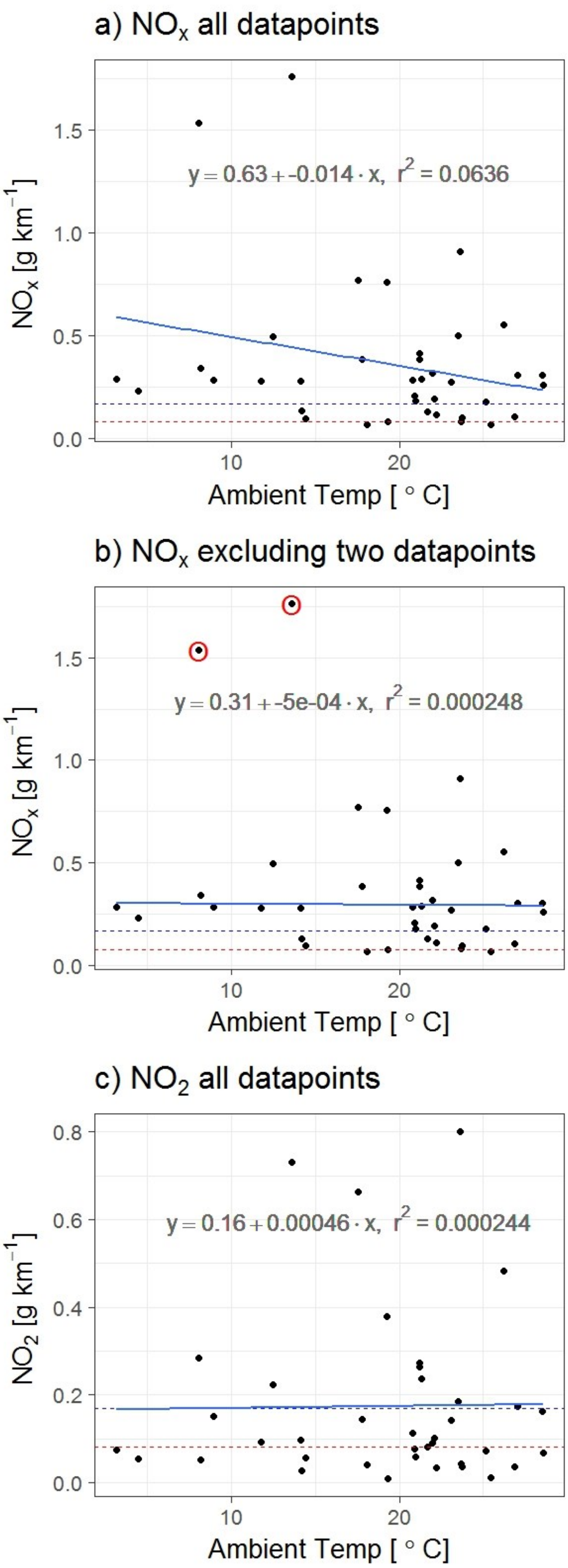

Figure 4-13. NO_x and NO_2 emission by temperature

For NO_x the regression line followed the expected trend, NO_x increased as temperature decreased though there was little correlation and the R^2 value was very low (**Figure 4-13a**). The slope in the regression line was due entirely to two anomalous high NO_x measurements towards the lower end of the temperature scale. The two highest NO_x emissions (E1.6, S3.0h) occurred between 8 – 14 ° C. When these two values were excluded from the analysis (**Figure 4-13b**) the regression line became flat and there was no relationship between NO_x and temperature.

In a report issued by the German government it was found some manufacturers disabled the NO_x abatement technology at temperatures as high as 18 °C (BMVI, 2016). It is likely that the two anomalous measurements in this study are an example of this "thermal window" phenomena. This is worrying for countries such as the UK where the average temperature is 9 °C.

4.2.2 Urban and motorway sections

This section compares the average NO_x emissions from the urban and motorway constituent parts of each trip and analyses the relationship between driving mode and emissions. As discussed previously emissions in urban areas are a key consideration in air quality policy. The majority of air quality limit value exceedances occur at the roadside in urban locations where public exposure is highest. Unfortunately urban driving is also where the PEMS recorded the highest NO_x emissions.

Table 4-6. Average trip, urban and motorway emissions

	NO_x [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2**[%]
Urban	0.43 (sd. 0.42)	5.4 (sd. 5.3)	0.20 (sd. 0.24)	44 (sd. 22)
Trip	0.36 (sd. 0.36)	4.5 (sd. 4.5)	0.17 (sd. 0.19)	44 (sd. 20)
Motorway	0.31 (sd. 0.37)	3.9 (sd. 4.6)	0.14 (sd. 0.18)	45 (sd. 21)
Increase trip to urban	19%	19%	18%	0%
Increase motorway to urban	39%	39%	43%	-2%

Table 4-6 lists the average emissions of the urban and motorway sections of the trip. The trip averages are also included for comparison. On average urban emissions of both NO_x and NO_2 were ~20% higher than for the trip as a whole and 40% higher than the motorway. Primary NO_2 was consistent throughout.

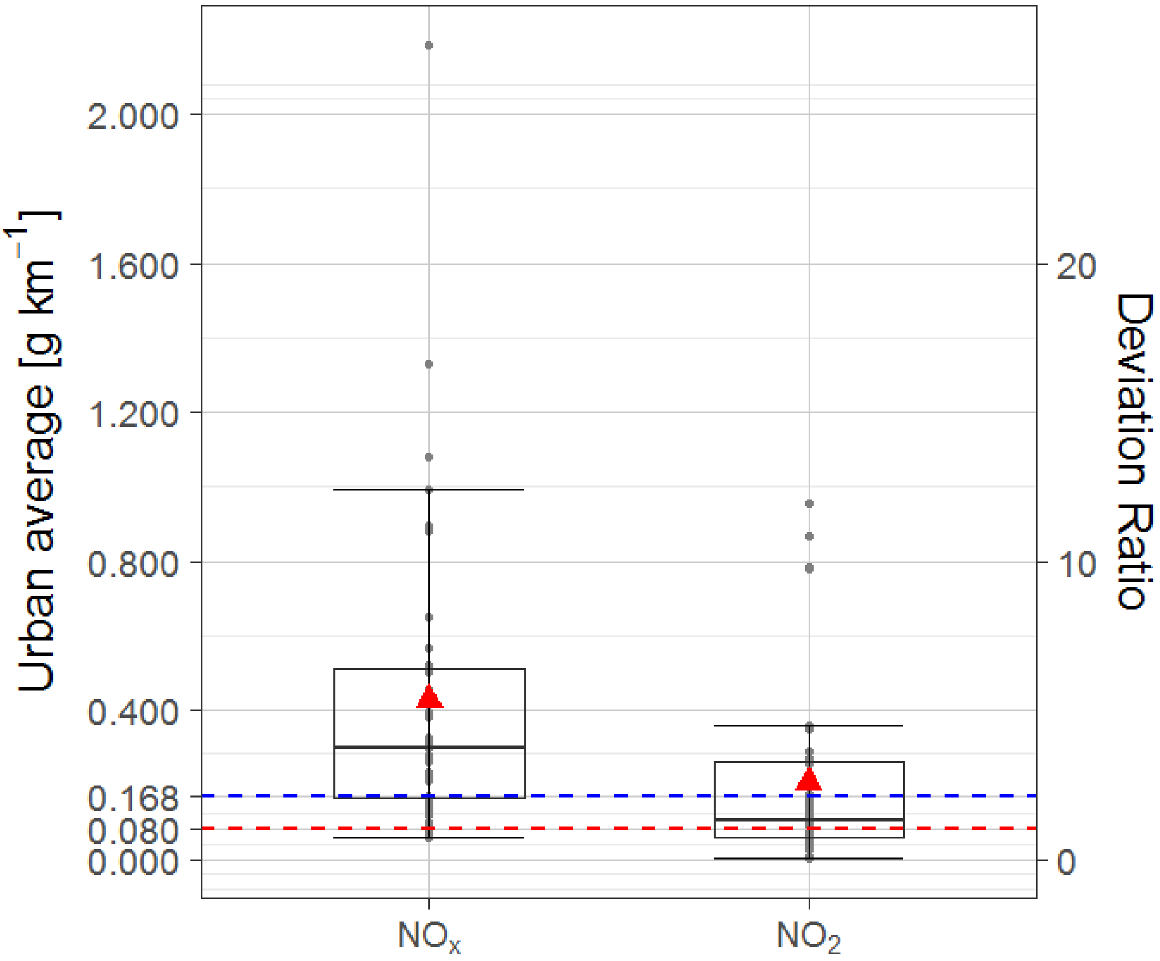

Figure 4-14. Urban average NO_x and NO_2 emissions in [g km^{-1}]

Figure 4-14 shows the urban average NO_x and NO_2 emissions in g km^{-1} with the study mean marked by a red triangle. The average urban NO_x emission of 0.43 (sd. 0.42) g km^{-1} corresponded to a deviation ratio of 5.4. The highest NO_x emission (again from vehicle S3.0h) was 2.18 g km^{-1}, a deviation ratio of 27. This is 4 times the first ever Euro standard type approval limit for NO_x (Euro 3, 0.5 g NO_x km^{-1}). The average urban NO_2 emission was 0.20 (sd. 0.24) g km^{-1}, this is higher than the Euro 5 type approval limit for total NO_x. This has worrying implications for roadside air quality limit value exceedances and urban air quality in general.

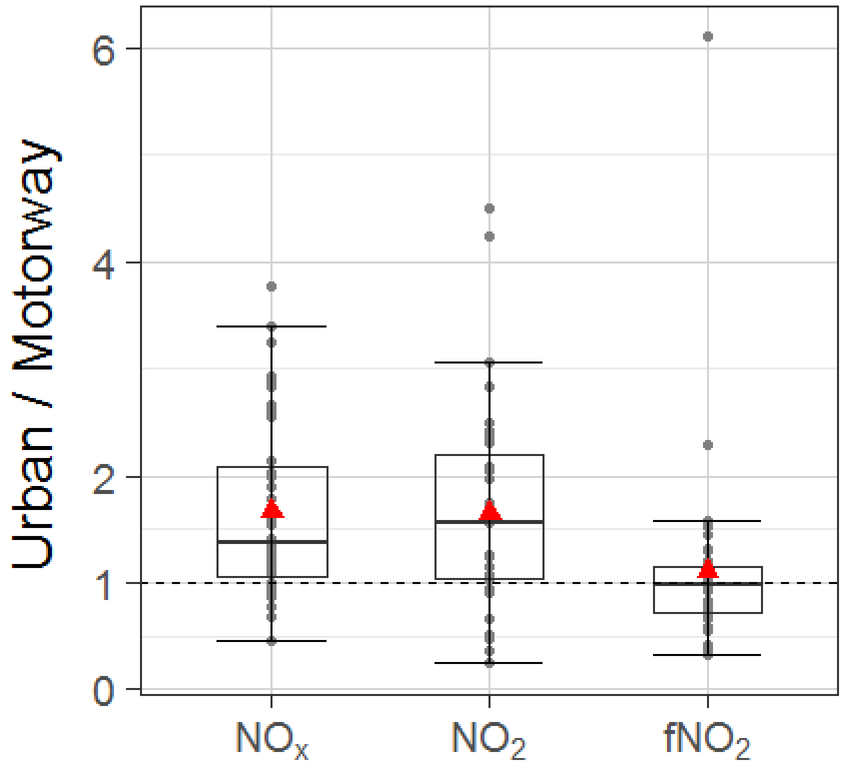

Figure 4-15. Ratio urban/ motorway trip average emissions of NO_x [g km^{-1}], NO_2 [g km^{-1}] and fNO_2 [%]

Figure 4-15 shows the ratio of urban emissions to motorway emissions for each individual vehicle. A ratio of 1 means urban emission = motorway emission. A ratio > 1 means urban emission > motorway emission. For both NO_x and NO_2 urban emissions were higher than motorway for the majority of vehicles by an average of 67% for NO_x, 65% for NO_2. There was a slight increase of 10% for fNO_2. The increase between motorway and urban emissions for individual vehicles was higher than the increase in the test fleet average.

At urban roadside locations the NO_x emitted directly as NO_2 makes a large contribution to ambient NO_2 concentrations. An increase in NO_2 emissions during urban driving of between 43 – 65% has negative implications for air quality objectives and the protection of human health.

4.2.2.1 Emission by driving style

This section explores why emissions are higher during urban driving. The data was divided into different driving modes (as defined by Frey *et al.*, (2003)). A description of the four driving modes and the % of time spent in each during urban and motorway sections is given in **Table 4-7**.

Table 4-7. Driving mode definitions (Frey et al (2003)) and % time of urban and motorway sections spent in each mode.

Mode	**Vehicle speed** [ms^{-1}]	**Acceleration* range** [ms^{-2}]	**% urban section**	**% motorway section**
Idle	< 0.5	± 0.1	12 %	0 %
Cruise	> 0.5	± 0.1	27 %	70 %
Acceleration		> 0.1	32 %	16 %
Deceleration		< - 0.1	29 %	14 %

*Acceleration calculated by **Equation 4-4** (not RPA)

The majority of time from motorway sections (70%) was spent in cruise, with no time spent in idle and the remainder split evenly between acceleration and deceleration. In comparison only 27% of the urban section was spent in cruise, with idle accounting for 12% and acceleration accounting for 32%. To assess the impact of driving mode on emissions each of the 39 trips were segmented into the four driving modes and the average NO_x and NO_2 emission calculated for each segment. **Figure 4-5** showed how instantaneous NO_x emissions were delivered in peaks coinciding with acceleration. This is corroborated by the results plotted in **Figure 4-16**.

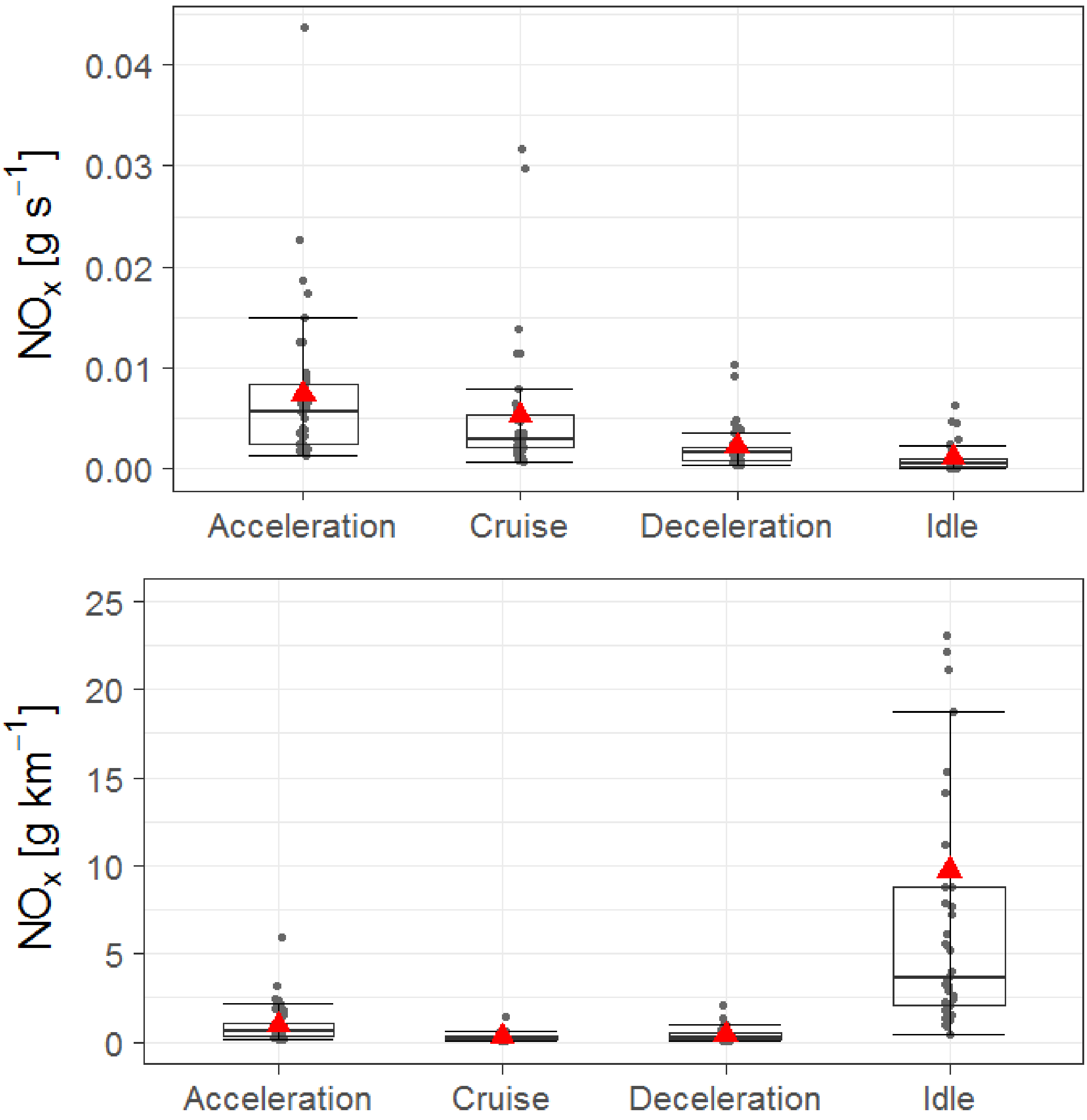

Figure 4-16. NO_x and NO_2 emissions by driving mode in g s^{-1} and g km^{-1}

Figure 4-16 is a boxplot of the averages for the four driving modes from each of the 39 vehicles. The mean of each driving mode (marked by red a triangle) is listed in

Table 4-8. Whilst there was high variability within the driving modes acceleration produced the highest emissions for both NO_2 and NO_x. In **Figure 4-16** emissions are presented in both g s^{-1} and g km^{-1}. It is important to note that whilst idle had the highest emissions in g km^{-1} it had the lowest in g s^{-1}. This is because even though the emission rate during idle driving was the lowest of all the modes, the distance accumulated was also very low. As a result the total grams accumulated were divided by a very small distance, resulting in high distance specific emissions (higher than 20 g km^{-1}). To allow for this results have been reported in both g s^{-1} and g km^{-1}.

Table 4-8 lists the mean NO_x and NO_2 emission for each driving mode in g s^{-1} and g km^{-1} and the ratio of all other modes to acceleration. For both NO_x and NO_2 the rate of emission during acceleration was higher than any other driving mode by as many as 7 times. This is in agreement with Daham *et al.*, (2005) who found traffic calming measures that increased the amount of acceleration/ deceleration (such as speed bumps) also increased NO_x emissions by 195%.

Table 4-8. NO_x and NO_2 emissions by driving mode in $g\ s^{-1}$ and $g\ km^{-1}$

	Acceleration	**Cruise**	**Deceleration**	**Idle**
NO_x [g s^{-1}]	0.007 (sd. 0.008)	0.005 (sd. 0.007)	0.002 (sd. 0.002)	0.001 (sd. 0.001)
NO_2 [g s^{-1}]	0.003 (sd. 0.004)	0.002 (sd. 0.003)	0.001 (sd. 0.001)	0.0005 (sd. 0.0007)
NO_x [g km^{-1}]	0.97 (sd.1.10)	0.30 (sd. 0.32)	0.39 (sd. 0.39)	9.7 (sd. 16.8)
NO_2 [g km^{-1}]	0.46 (sd. 0.58)	0.15 (sd. 0.16)	0.19 (sd. 0.20)	4.2 (sd. 5.8)
Increase between other modes and acceleration				
NO_x [g s^{-1}]	-	x 1.4	x 3.5	x 7.0
NO_2 [g s^{-1}]	-	x 1.5	x 3.0	x 6.0
NO_x [g km^{-1}]	-	x 3.2	x 2.5	x 0.1
NO_2 [g km^{-1}]	-	x 3.1	x 2.4	x 0.1

These results indicate prevalence of acceleration in urban sections is a dominant factor in the 39% increase in average NO_x emissions. Therefore substantial reduction in urban emissions could be achieved by traffic management and junction redesign. Traffic schemes that reduce congestion and ease flow are cheaper and quicker to implement than more complex policy tools (such as Low Emission Zones) and can be effective at tackling air pollution (Chin, 1996; Carslaw & Beevers, 2005).

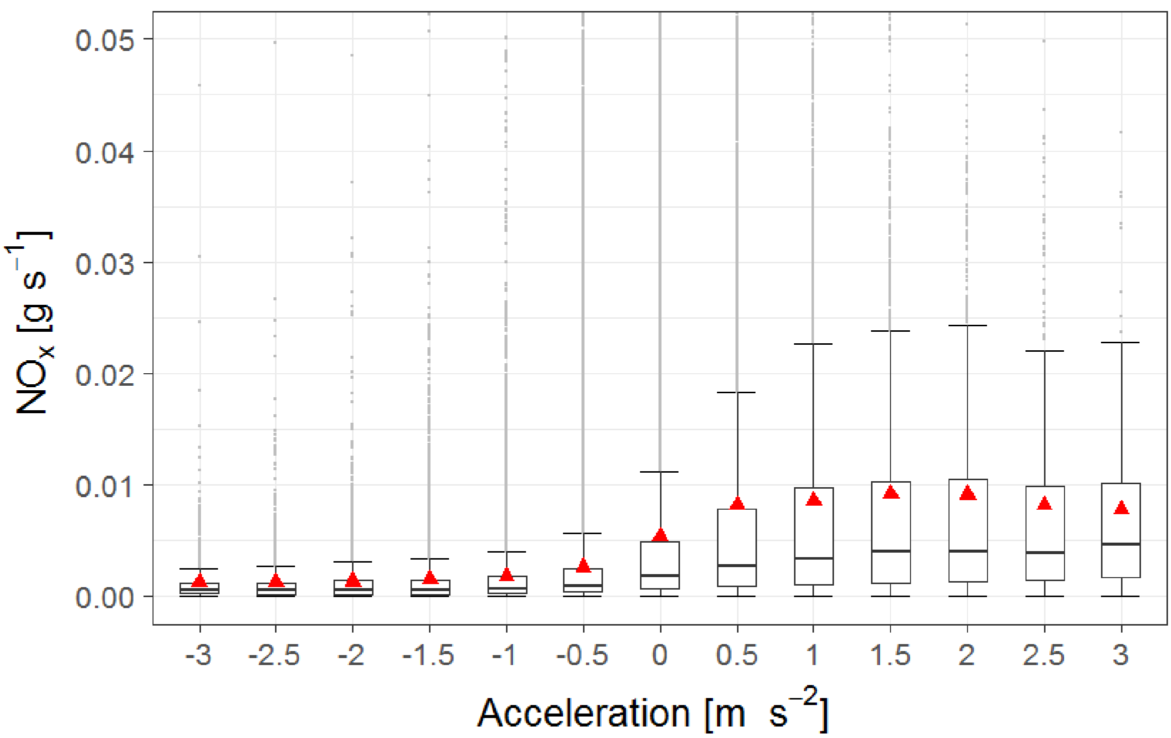

Figure 4-17. NO_x emission in g s^{-1} by acceleration

Figure 4-17 is a boxplot of the 220,000 instantaneous accelerations (all trips combined) and NO_x measurements from all 39 vehicles divided into 12 acceleration bins each 0.5 m s^{-2} wide. Approximately half of the data points had $a>0$ and half $a<0$. Both the mean and median NO_x emissions were substantially higher for $a>0$. The mean NO_x, range and prevalence of data points for each bin is listed in **Table 4-9**. The majority of data points fell within the range $-0.5 < a < 0.5$.

Table 4-9 Cuts, distribution of data points and mean NO_x for acceleration bins

Bin	Range	Mean NO_x [g s^{-1}]	% of data points
-3	[-3,-2.5]	0.0013	0.41%
-2.5	(-2.5,-2]	0.0012	0.78%
-2	(-2,-1.5]	0.0013	1.51%
-1.5	(-1.5,-1]	0.0015	3.06%
-1	(-1,-0.5]	0.0018	6.41%
-0.5	(-0.5,0]	0.0026	46.03%
0	(0,0.5]	0.0053	28.38%
0.5	(0.5,1]	0.0082	7.84%
1	(1,1.5]	0.0086	3.26%
1.5	(1.5,2]	0.0091	1.39%
2	(2,2.5]	0.0091	0.58%
2.5	(2.5,3]	0.00808	0.26%
3	(3,3.5]	0.00773	0.09%

4.2.3 Comparison between PEMS and COPERT (4v11)

In this section PEMS measurements for NO_x and NO_2 are analysed for speed dependency and compared with COPERT 4v11 speed dependent emissions factors. The trip average NO_x, deviation ratio, NO_2 and fNO_2 of the PEMS measurements and COPERT estimates are reported and compared in **Table 4-10**.

Table 4-10. Comparison of PEMS and COPERT trip averages

	NO_x [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2** [%]*
PEMS	0.36 (sd. 0.36)	4.5 (sd. 4.5)	0.17 (sd. 0.19)	44 (sd. 20)
COPERT	0.23 (sd. 0.01)	2.9 (sd. 0.1)	0.07(sd. 0.003)	30 (sd. 0)
Ratio	x 1.6	x 1.6	x 2.4	x 1.5

*fNO_2 calculated by mass

COPERT estimates were considerably lower than the average emissions measured by the PEMS. Comparatively the average PEMS NO_x emission was 1.6 times higher than the COPERT average estimate. COPERT's assumption of 30% fNO_2 was also an underestimate. The combination of the underestimate in NO_x and fNO_2 resulted in an even larger underestimate for NO_2 emissions. Real world NO_2 emissions were 2.4 times higher than COPERT estimates.

In contrast to the PEMS measurements there was very little variation in COPERT estimates. This was expected as COPERT estimates are speed dependent and there was little variation in the speed profiles between the trips (as seen in **Figure 4-2**).

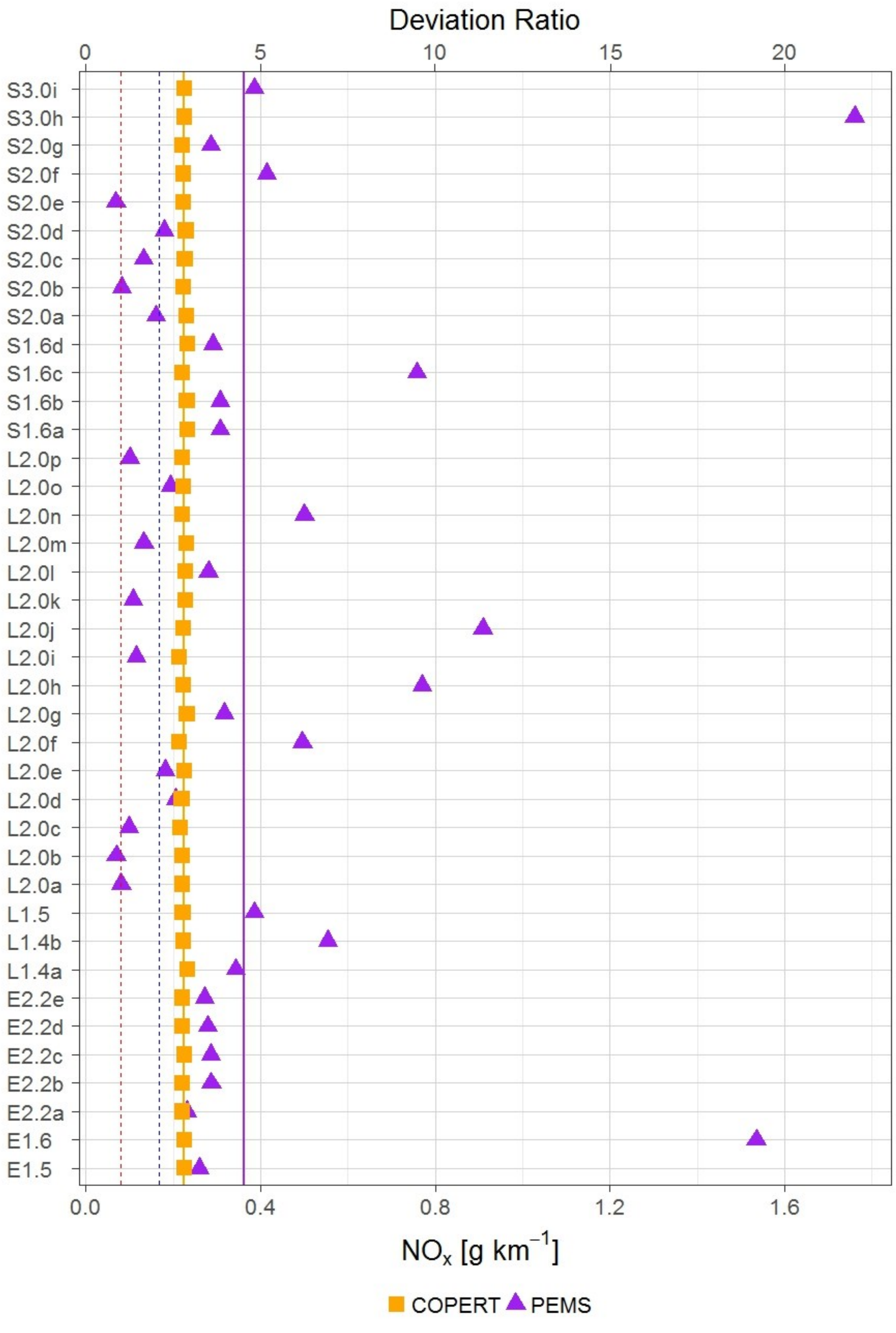

Figure 4-18. Comparison of PEMS measurements and COPERT estimates for NO_x (orange line = COPERT mean, purple line = PEMS mean, red dashed line = Euro 6b limit, blue dashed line = Euro 6c limit)

Figure 4-18 compares the PEMS measurements to the COPERT estimates for each vehicle. The orange line represents the COPERT mean, the purple line represents the PEMS mean. PEMS measurements were higher in some instances and lower in others but the overall trend was an increase from COPERT to PEMS. 24 vehicles (62%) had PEMS emissions higher than the COPERT estimate, some by a factor of over 10.

To investigate the cause of the discrepancy between the PEMS measurements and COPERT estimates the speed dependency of the real world emissions was analysed. **Figure 4-19** is a boxplot of the 220,000 instantaneous speed and NO_x measurements from all 39 vehicles divided into 11 speed bins each 10 km h^{-1} wide. The mean NO_x, range and prevalence of data points for each bin is listed in **Table 4-9**.

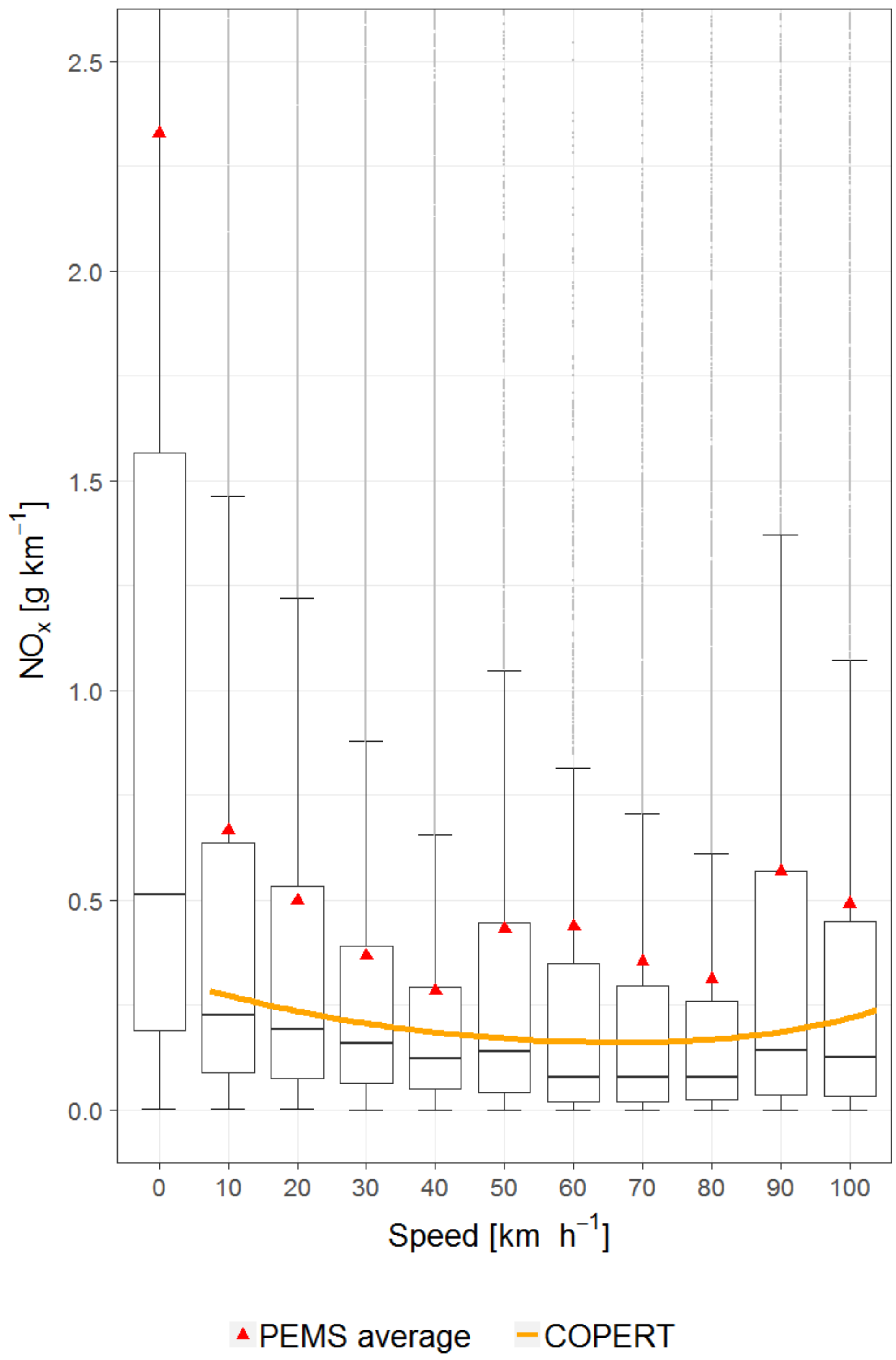

Figure 4-19. Comparison of instantaneous PEMS measurements and COPERT 4v11 speed dependent emissions factors

In **Figure 4-19** the orange curve marks the COPERT speed dependent emissions factors, the boxplots are the instantaneous PEMS measurements (220,000 data points), the red triangle is the mean of each PEMS speed bin and the grey dots are outliers (some of which have been cropped out). COPERT does not provide emissions factors for speeds <10 km h^{-1}.

The COPERT emission curve was close in value to the PEMS median for each speed bin (thick horizontal line across centre of box). However, the curve was much lower than the PEMS means. This was due to the large number of outliers with emissions far above the interquartile range. These outliers had a significant effect on trip average emissions and contributed to the deviation between COPERT estimates and PEMS measurements. As illustrated by **Figure 4-20** the majority of these outliers were during acceleration.

It should be noted that the gas analysers of the SEMTECH-DS are high resolution, and the span and zero tests before and after each PEMS test (tests with error over 3% are not valid) mean it is extremely unlikely these high measurements were due to experimental or measurement error.

Table 4-11. Cuts, distribution of data points and mean NO_x for speed bins

Bin	Range	PEMS NO_x [g km^{-1}]	% of data points
0	[0,10]	2.33	20%
10	(10,20]	0.665	11%
20	(20,30]	0.499	15%
30	(30,40]	0.368	19%
40	(40,50]	0.283	20%
50	(50,60]	0.432	2%
60	(60,70]	0.438	1%
70	(70,80]	0.354	3%
80	(80,90]	0.311	4%
90	(90,100]	0.568	2%
100	(100,110]	0.492	3%

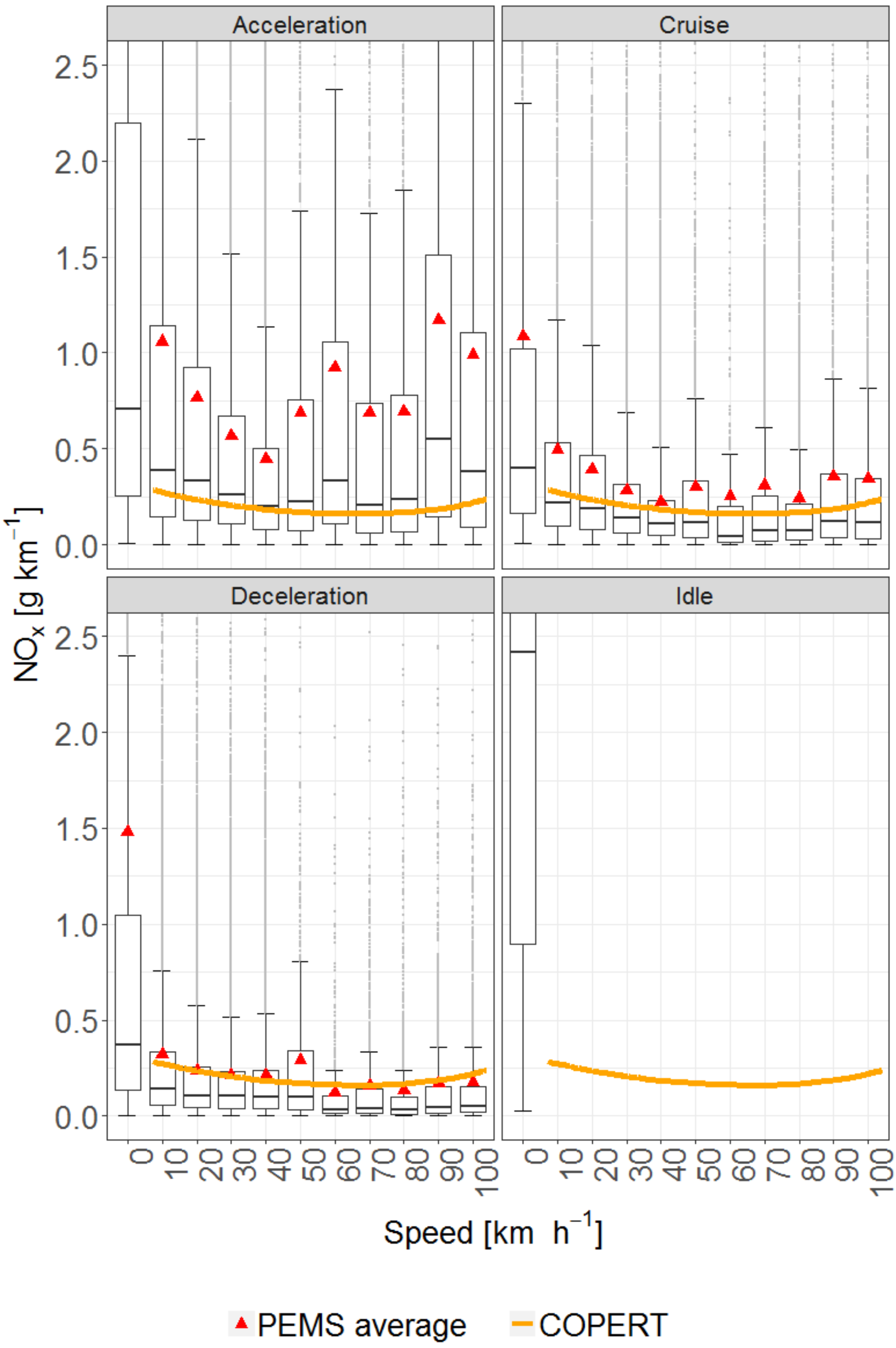

Figure 4-20. Comparison of instantaneous PEMS measurements and COPERT emissions factors for different driving modes

Figure 4-20 compares the PEMS instantaneous emissions measurements to the COPERT emissions factors for the different driving modes. The means of the acceleration data points exceeded the COPERT curve by a factor of >2. The means of the cruise data points followed the COPERT curve almost exactly for speeds >30 km h^{-1}. Acceleration events accounted for some of the difference between COPERT estimates and PEMS measurements but did not account for it all. The mean NO_x emission for the cruise segments (0.30 (sd. 0.32) g km^{-1}) was still 30% higher than the average COPERT estimate.

4.2.3.1 COPERT estimates for urban and motorway sections

As seen in **Figure 4-7** the COPERT 4v11 Euro 6 emissions factor for NO_x was not very sensitive to speed. PEMS measurements for NO_x increased 39% between motorway and urban sections. As a result the divergence between COPERT estimates and PEMS measurements was greater for urban driving.

Table 4-12. Comparison of urban PEMS and COPERT urban averages

	NO_x [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2** [%]
PEMS	0.43 (sd. 0.42)	5.4 (sd. 5.3)	0.20 (sd. 0.24)	44 (sd. 22)
COPERT	0.24 (sd. 0.01)	3 (sd. 0.1)	0.07 (sd. 0.003)	30 (sd. 0)
Ratio	x 1.8	x 1.8	x 2.9	x 1.5

The trip average PEMS measurements for NO_x were 1.6 times higher than the COPERT estimates. During urban driving this rose to 1.8 times for NO_x and 2.9 times for NO_2. Most air quality policies focus on reducing emissions in urban areas where the public exposure is highest. It is also at urban roadside locations where the proportion of NO_x emitted directly as NO_2 becomes the dominant factor in ambient air concentrations. These result shows that in these key areas the COPERT model underestimated NO_2 by a factor of ~3. This has serious implications for air quality policy makers who rely on COPERT 4v11 to model scenarios. Particularly scenarios for the near future when the percentage of Euro 6 in the fleet mix will be much higher. For example, the NAEI predicts by 2020 ~60% of diesel passenger cars will be Euro 6. This will be discussed further in the next chapter.

For motorway driving the PEMS measurements exceeded the COPERT estimate by 1.4 times for NO_x and 2.2 times for NO_2.

4.3 Discussion

In this section the results from this chapter are put in the context of existing PEMS studies and discussed from a policy perspective. This section will focus mainly on urban emissions as these are the most relevant to air quality policy.

4.3.1 Comparison with other studies

The results from this study have been compared with previous PEMS studies of Euro 6 diesel passenger cars. Whilst in recent years the number of passenger car PEMS studies has slowly increased there are still relatively few and often the sample sizes are small. With 39 vehicles this study was the largest published PEMS passenger car study to the time of its publication. Test conditions, drivers, measurement equipment, route and routine vary between studies. The comparisons made in this section aim to put this study in the context of the wider field of PEMS measurements. **Figure 4-21** is a boxplot of the results from this study and previous studies which together amount to

173 vehicles. The names, references, year, sample size and mean NO_x emission of the previous studies are listed in **Table 4-13.**

Results from previous studies are plotted in green, results from this study are plotted in purple. The mean from each study is marked with a red triangle. The red dashed line marks the Euro 6 type approval limit. The green dashed vertical line marks the average of all the previous studies means (not including this study).

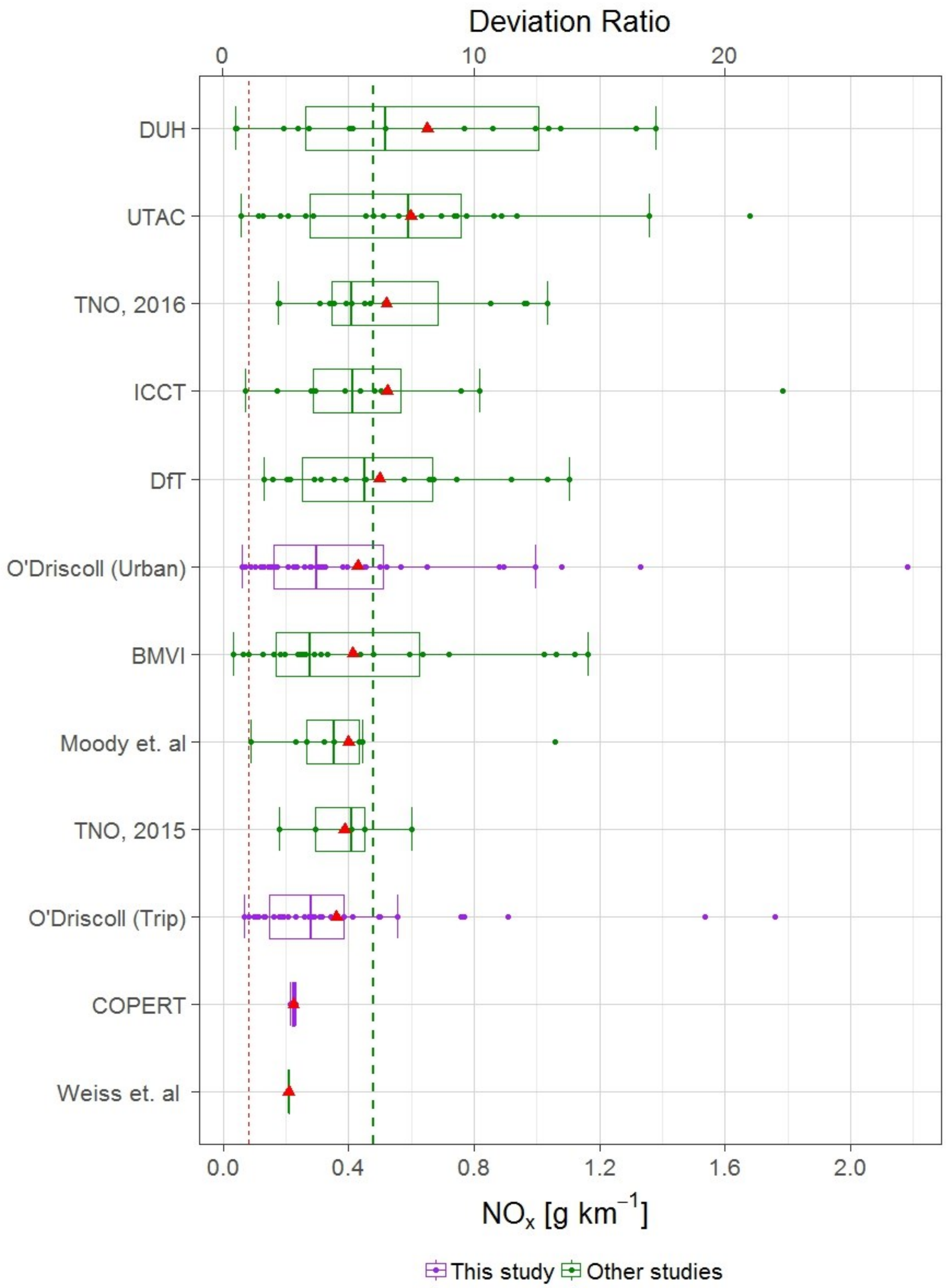

Figure 4-21. Comparison of PEMS measured NO_x emissions from this study with other studies (Euro 6 diesel)

The results from this study were in good agreement with previous studies. All studies found huge variability in the real world performance of Euro 6 diesel cars. The average of the means of the previous studies (marked by green dashed line) was 0.47 (sd. 0.13) g km^{-1}. This was higher than the trip average of this study (0.36 g km^{-1}) though much closer to the urban average (0.43 g km^{-1}). Given these studies were performed independently across Europe there is a remarkable level of consistency. With the exception of Weiss et al. (which had a limited sample size of 1) the COPERT estimates were much lower than the PEMS study averages.

The existing studies together included 134 vehicles. The average NO_x emission for these 134 vehicles was 0.51 (sd. 0.35) g km^{-1}, a deviation ratio of 6.3. This was higher than the urban deviation ratio found in this study (5.4). A potential reason for this may be that the test fleet in this study included more vehicles from the premium range than the economy. Economy range vehicles are cheaper, use cheaper abatement technologies and as a result have higher emissions. Another reason for lower emissions in this study may be the prevalence of 2 ℓ engines in the sample (59%) which were found to have lower emissions than non 2 ℓ engines. Additionally results from this study may be lower than other studies due to the removal/ absence of cold starts emissions. Lastly German studies include autobahn driving at much higher speeds than UK motorways, resulting in higher NO_x emissions.

7 of the 134 vehicles (5%) met the Euro 6 limit, this is the same percentage as found in this study. However only 13% of the vehicles in previous studies met the Euro 6d-TEMP limit whereas 26% did in this study. This being said the results from this study results were within the same range of previous studies.

Table 4-13. Previous PEMS studies including Euro 6 vehicles

Name (Location)	Reference	Year	# of vehicles	Mean NO_x [g km^{-1}]	Deviation Ratio
DUH (Germany)	(DUH, 2016)	2016	20	0.65	8.1
UTAC (France)	(Ministre de l'environnement, 2016)	2016	23	0.60	7.5
TNO, 2016 (Netherlands)	(TNO, 2016)	2016	15	0.52	6.5
ICCT (UK)	(Franco *et al.*, 2014)	2014	12	0.52	6.5
DfT (UK)	(DfT, 2016d)	2016	19	0.50	6.3
BMVI (Germany)	(BMVI, 2016)	2016	30	0.41	5.1
Moody et. al (UK)	(Moody & Tate, 2017)	2017	9	0.40	5.0
TNO, 2015 (Netherlands)	(Kadijk, Mensch & Spreen, 2015)	2015	5	0.39	4.9
Weiss et. al (Italy)	(Weiss *et al.*, 2012)	2012	1	0.21	2.6
Trip	This study	2016	39	0.36	4.5
Urban	This study	2016	39	0.43	5.4

Of the six previous studies only two reported NO_2 separately from total NO_x. The results are listed in **Table 4-14**.

Table 4-14. NO_2 and fNO_2 from previous studies

	# of vehicles	**NO_2** [g km^{-1}]	**fNO_2**
Weiss et. al	1	0.10	51%
Moody et. al	5	0.10	36%
Trip	39	0.17	44%
Urban	39	0.20	44%

Weiss et. al and *Moody et. al* reported NO_2 within the range found in this study. The lower % fNO_2 reported by *Moody et. al* is likely because the 5 vehicles in their sample were 3 LNT and 2 SCR. The 3 LNT vehicles sampled by *Moody et al*. had an average fNO_2 of 30%, the 2 SCR had an average 40%. This is in good agreement with findings relating to NO_x abatement technologies stated earlier in this chapter (that SCR has a higher % fNO_2 than LNT and EGR).

The average fNO_2 of 44 (sd. 20) % was also higher, though within the range of, a remote sensing study that found fNO_2 of 34.0 ± 9.8 % (Carslaw *et al.*, 2016). This is potentially because the remote sensing study was carried out in 2012/13 whereas the PEMS testing in this study was performed later, in 2015/16. This could be a continuation of the trend of increasing fNO_2 (Beevers *et al.*, 2012).

4.3.2 Discussion of variability

A key challenge facing policy makers tackling air pollution from diesel vehicles is the variability in performance between vehicles of the same Euro standard. For example, in urban driving the vehicles in this study had deviation ratios of between 0.7 and 27. Increasingly air quality policy makers are depending on schemes such as Low Emission Zones to counter the problem of diesel emissions. However, Low Emission Zones (LEZ) discriminate by Euro standard and do not take into account real world emissions. For example, the Ultra-Low Emission Zone (ULEZ) to be introduced in London in 2019 will bar all but Euro 6 diesel passenger cars (Euro 4 for petrol). This will mitigate against the oldest and theoretically worst polluting vehicles but it makes no allowance for Euro 6 diesels that that emit up to 27 times the current type approval limit.

Figure 4-22 illustrates the potential of discriminating by RDE as opposed to Euro classification. Our results indicate a LEZ that bans all but Euro 6 diesel vehicles would have an urban average NO_x emission (from the diesel proportion of the fleet) of 0.43 g km^{-1} (red line). Euro 5 diesel vehicles have real driving NO_x emissions of ~0.7 g km^{-1} (grey dashed line). If the LEZ were to discriminate on the basis of RDE i.e. instead of "*only Euro 6 diesel cars allowed*" the rule was "*only diesel cars with RDE below Euro 5 (< 0.7 g km^{-1}) allowed"* the resulting average NO_x emission of the diesel proportion of the fleet within the LEZ would be 0.28 g km^{-1} (purple line). This is a 35% reduction in average NO_x by removing only 15% of the Euro 6 diesel vehicles. The 6 vehicles that would be barred from a RDE dependant LEZ are circled in red, vehicles that have NO_x < Euro 5 RDE are plotted as purple triangles. Similarly in the RDE dependent LEZ average urban NO_2 emissions would fall by 38% to 0.13 g km^{-1}.

The COPERT estimates were included in **Figure 4-22** for comparison (orange). When the worst 6 vehicles were removed the new PEMS average was in much better

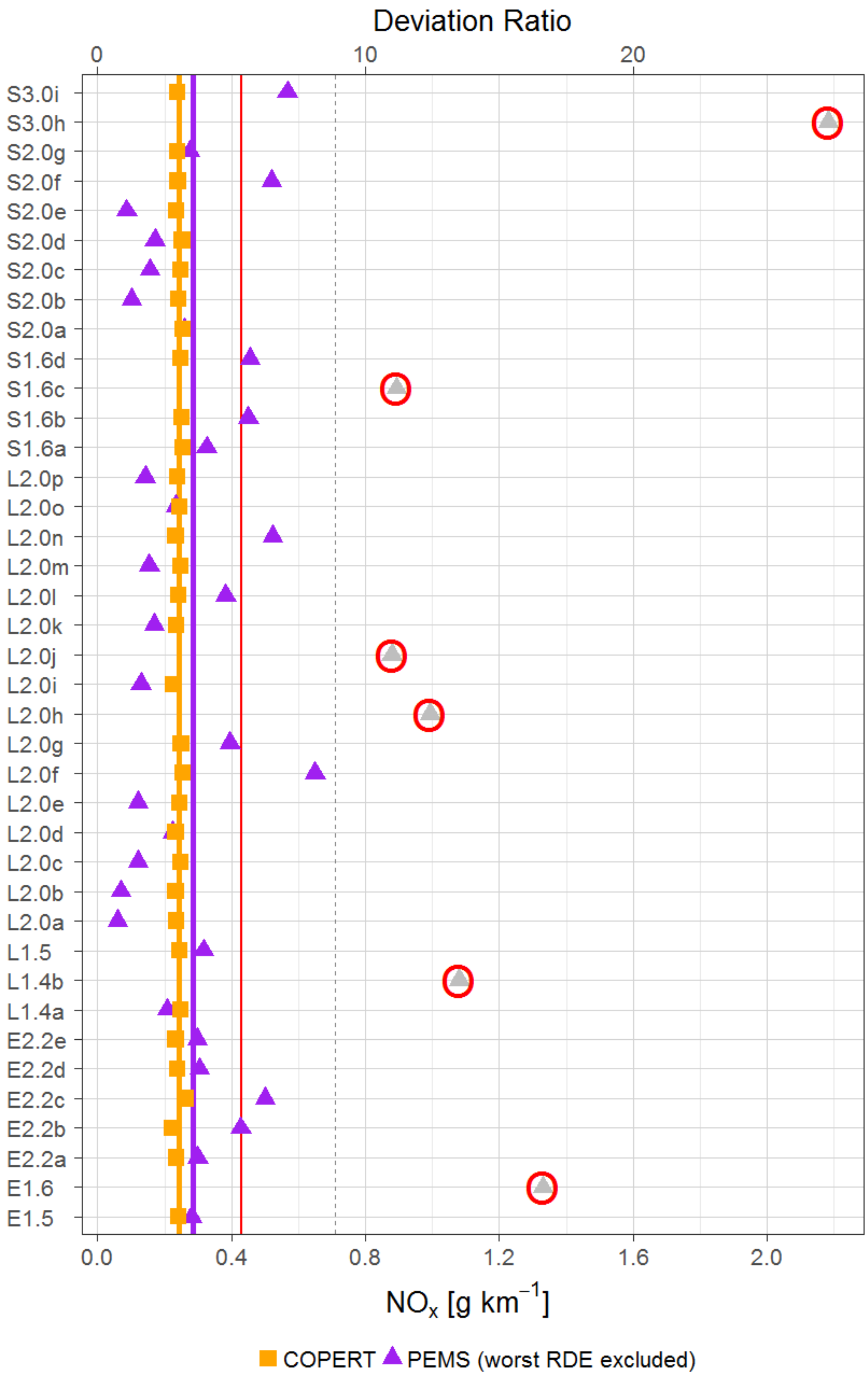

Figure 4-22. Comparison with COPERT with < RDE Euro 5 (circled in red) removed for the urban section (grey dashed line = RDE Euro 5 (0.7 g km^{-1}), red line = old PEMS mean, purple line = new PEMS mean, orange line = COPERT mean

agreement with the COPERT average estimate. This means that proposed LEZs modelled using COPERT 4v11 emission factors (such as in the DEFRA's 2015 Air Quality Action Plan (DEFRA, 2015b)) would deliver the proposed results if vehicles were selected by RDE as opposed to Euro standard.

A recent review of the efficacy of LEZs in 5 European countries found mixed results, in all cases air quality benefits were less pronounced than expected (Holman, Harrison & Querol, 2015). The best results were recorded in Germany (the only LEZs to include cars and HGVs) though the introduction of LEZs was accompanied by a scrappage scheme which accelerated the fleet turn over making initial success hard to attribute to any one scheme. The variation and underestimate of diesel RDE is a potential reason why modelling of LEZs has often been over optimistic.

4.3.3 Euro 6d-TEMP real driving type approval limit

10 vehicles in this study achieved the Euro 6d-TEMP type approval limit of 0.168 g NO_x km^{-1} during urban driving. **Figure 4-23** is a bar chart showing the NO and NO_2 composition of these 10 vehicles. One vehicle (S2.0c) achieved the Euro 6c limit for total NO_x with NO_2 emissions of 0.1 g km^{-1}, 25% above the Euro 6 limit for total NO_x. This indicates the introduction of Euro 6d-TEMP may not be as effective as hoped in reducing NO_2 concentrations in urban areas. Given that in urban areas the amount of NO_x emitted directly as NO_2 dominates ambient concentrations these results indicate an additional dedicated NO_2 type approval limit should be considered.

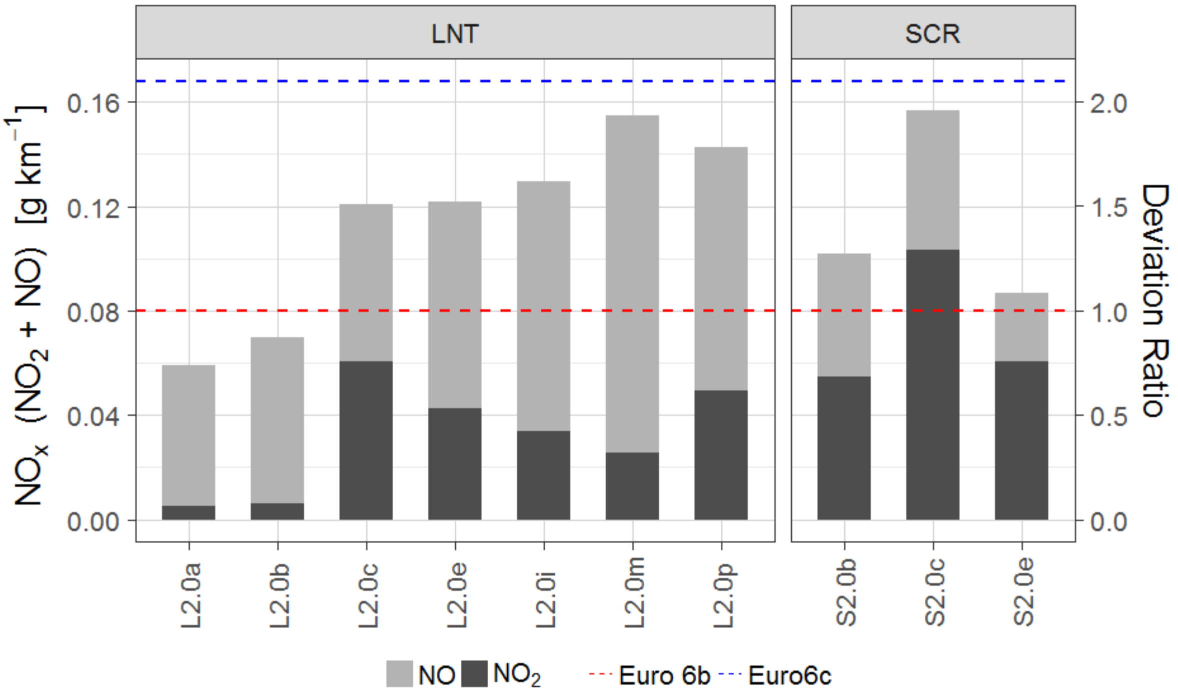

Figure 4-23. NO and NO_2 emissions of vehicles that met the 0.168 g km^{-1} Euro 6c NTE limit for NO_x (blue dashed line = Euro 6c limit, red dashed line = Euro 6b limit)

4.4 Summary

This chapter described the results of a Portable Emissions Measurements System (PEMS) study containing 39 Euro 6 diesel passenger cars and compared the derived real world emission factors to COPERT speed dependent emissions factors. The key results are listed in **Table 4-15.**

Table 4-15. Key results from Chapter 4, average emissions for PEMS and COPERT

	NO_x [g km^{-1}]	Deviation Ratio	NO_2 [g km^{-1}]	fNO_2 [%]
Urban (PEMS)	0.43 (sd. 0.42)	5.4	0.20 (sd. 0.24)	44 (sd. 22)
Trip (PEMS)	0.36 (sd. 0.36)	4.5	0.17 (sd. 0.19)	44 (sd. 20)
Motorway (PEMS)	0.31 (sd. 0.37)	3.9	0.14 (sd. 0.18)	45 (sd. 21)
COPERT average	0.23 (sd. 0.01)	2.9	0.07(sd. 0.003)	30 (sd. 0)

It was found that during urban driving (when public exposure is highest) real driving emissions exceeded the Euro 6 type approval limit (0.08 g NO_x km^{-1}) by 5.4 times and emissions of NO_2 were over twice the limit for total NO_x. Real world urban emissions were found to be 1.8 and 2.9 times COPERT 4v11 speed dependent emissions factors for NO_x and NO_2 respectively.

Analysis of instantaneous PEMS data found NO_x to be delivered in peaks that coincided with acceleration. Both NO_x and NO_2 emissions in g km^{-1} were three times higher during acceleration than deceleration. Urban driving contained twice as much acceleration ($a > 0.1$ ms^{-2}) as motorway driving and which contributed to average NO_x emissions being 40% higher during urban sections.

4.5 A note on COPERT 5

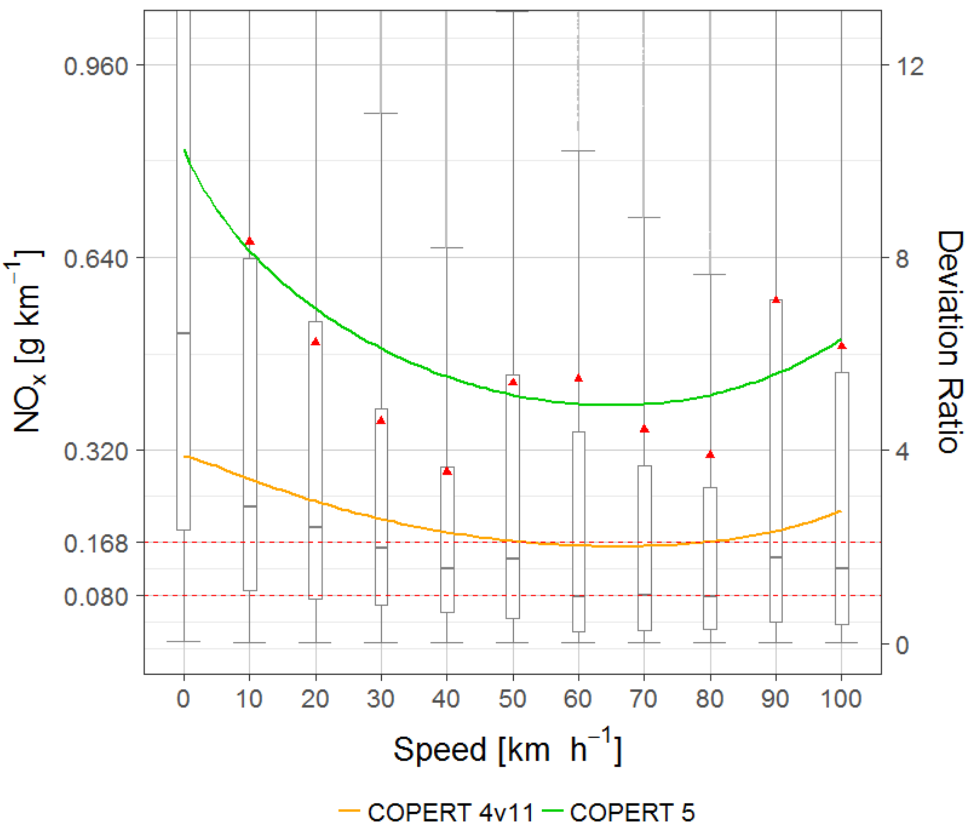

Figure 4-24. COPERT 5 and 4v11 emission factors comparison with PEMS

COPERT 5 (an update to the COPERT 4v11) was published by Emisia in September 2016. It included updated emission factors for Euro 6 diesel cars and LGVs and Euro 5 diesel LGVs (DEFRA, 2017a). The updated COPERT 5 Euro 6 diesel emission factor is plotted (green line) in **Figure 4-24** and compared with 4v11 and instantaneous PEMS measurements from this study. The COPERT 5 emission factor is in much better agreement with the PEMS measurements. It is more sensitive to speed, increases by a greater amount at lower speeds and has a deviation ratio between 5 – 10. COPERT 5 has been used in the latest DEFRA air quality action plan (DEFRA, 2017b), this will be discussed further in the next chapter.

Chapter 5. Scenario analysis of 2030 Euro 6 diesel NO_x emissions

In the previous chapter real world emissions factors were derived for Euro 6 diesel passenger cars using PEMS data. In this chapter these emissions factors are used to inform five scenarios for 2030. These scenarios are then modelled by the UKIAM to assess the potential implications of changing Euro 6 emission factors on UK total NO_x emissions (in tonnes), annual average roadside concentrations of NO_2 (in µg m^3), change in Population Mean Weighted Concentration of NO_2 (also in µg m^3) and damage cost (in Billion £).

5.1 Background

The NAEI projects that by 2030 over 90% of diesel passenger cars in the UK fleet will be Euro 6 (NAEI, 2014b). The real driving emissions type approval process being introduced in September 2017 (Euro 6d TEMP) is designed to bring down the deviation ratio of Euro 6 diesel cars. However, it will not address the deviation ratio of vehicles already in circulation. Without the introduction of policy measures such as a national scrappage scheme there will be a time delay in realising any benefits of the new type approval regime as the average age of a passenger car in Europe is 9.73 years (ACEA, 2017a).

Current plans are to introduce the Euro 6d TEMP with a limit of 0.168 g km^{-1} to apply to newly approved vehicle models from September 2017, extending to all new vehicles sold in September 2019. In 2020 the RDE type approval Euro 6d will be introduced, bringing down the conformity factor to 1.5 (0.12 g km^{-1}) for newly approved models (applicable January 2021 for all new vehicles sold).

The International Council on Clean Transportation (ICCT) have comprehensively modelled potential Euro 6 emissions factors for various scenarios relating to the enforcement of the RDE test procedure up to 2030 (Miller & Franco, 2016). They used PEMS data and emissions modelling to calculate the component of real world driving that will be covered by the RDE type approval procedure, and the component that will not (e.g. cold start, more extreme driving styles and “defeat devices”). They then made projections of the real world deviation ratios of new Euro 6 cars from 2017 onwards.

The ICCT modelling took a large sample of 1 Hz resolution PEMS data from 32 pre-RDE Euro 6 diesel cars. They then used the guidelines of the new RDE procedure to identify which driving conditions will be covered during the tests (described as the "normal" driving component) and which will not. They estimated that 80% of real world driving fell within the "normal" driving component captured by the RDE test. Conditions not covered in the test were grouped into three components: "cold start", "extended driving" and "defeat device". The "cold start" component related to additional emissions as a result of SCR being below optimum temperature in a cold engine, estimated to affect 8% of driving. "Extended driving" included events such as DPF regeneration, aggressive driving and driving at altitude, which occur in the real world but will not be captured by the RDE test. These conditions were estimated to account for 12% of total driving. "Defeat device" refers to any additional emissions that may be present due to the presence of legally questionable defeat devices such as driving cycle identification, thermal windows, or timers. Some form of defeat device was thought to be implemented in 30% of vehicles in the sample.

The percentage of total driving time spent in each component was then multiplied by the deviation ratio measured during these components from the PEMS data. For example, the deviation ratio was 6 during "cold starts" and 7.6 during "extended driving". Vehicles using "defeat devices" were assumed to have a deviation ratio of 7.6 across all driving. For Euro 6d- TEMP vehicles "normal" driving was assigned a deviation ratio of 2.1 (with a safety margin of 30%). The various projected fleet average real world emissions factors were then devised by reducing these component deviation ratios and the percentage of driving not included in test, in line with evolution of the RDE test procedure.

The ICCT projected that the implementation of Euro 6d TEMP (2017/19) will bring the real world deviation ratio of new Euro 6 vehicles down to 4. With the introduction of

Euro 6d the deviation ratio was projected to fall to 2 by 2022. The ICCT concluded that EU must introduce additional RDE components to the type approval process (for example an extension to include cold starts) to eventually bring the Euro 6 diesel deviation ration down to 1.

Given the average age of a car in Europe is ~10 years it is likely, if the ICCT projections are accurate, that the average deviation ratio in 2030 will be somewhere between ~6 (as it is now) and 1 (best case scenario). The scenarios modelled in this chapter cover the full range of the ICCT modelled Euro 6 emissions factors for 2030.

5.1.1 DEFRA "*Draft UK Air Quality Plan for tackling nitrogen dioxide*" (2017)

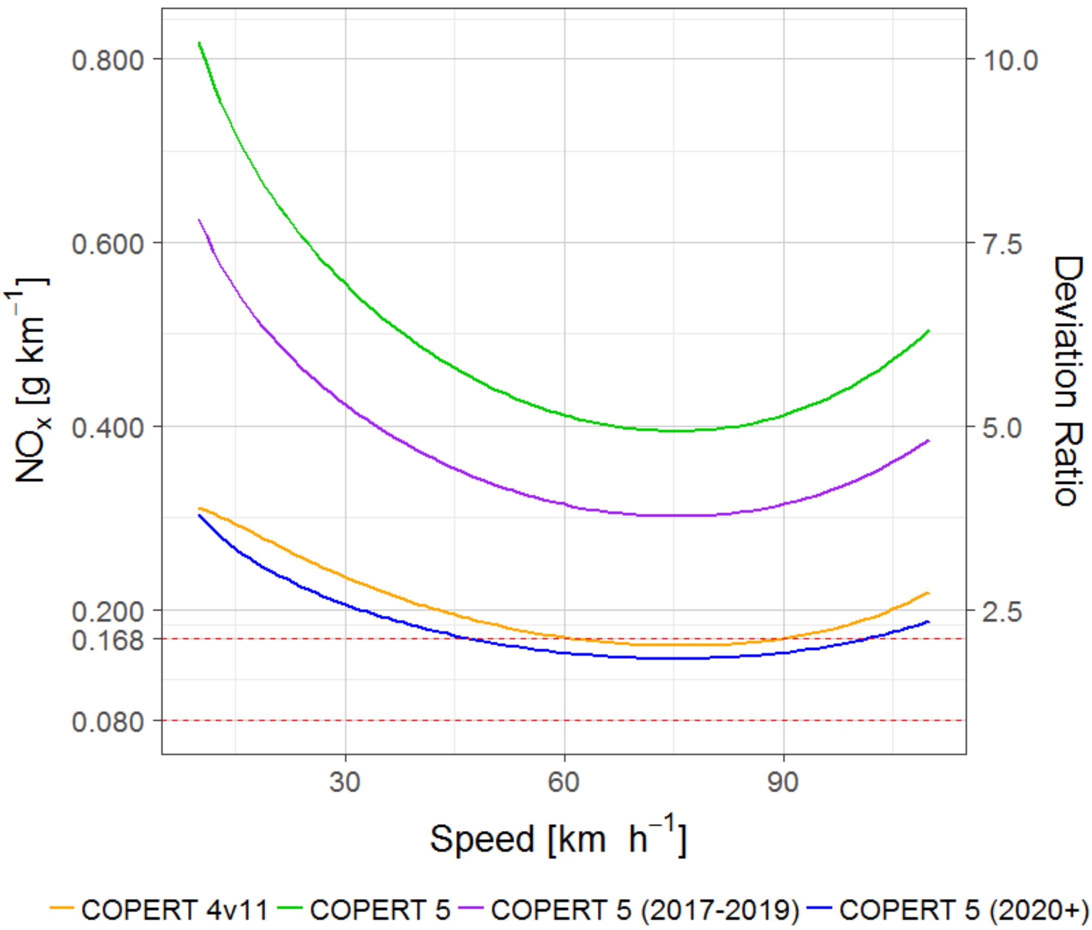

Figure 5-1. Scaled factors of COPERT 5 used by DEFRA

As mentioned at the end of the previous chapter, the updated (third) DEFRA air quality action plan used the new COPERT 5 emissions factors for Euro 6 diesel passenger cars. DEFRA's modelling also accounted for the reduction in Euro 6 diesel emissions factors as a result of the introduction of Euro 6d TEMP and Euro 6d. These reduced emission factors are plotted in **Figure 5-1.** DEFRA assumed a deviation of ~6.7 for 2016, falling to ~5.1 for 2017–19 and eventually ~2.5 for 2020 onwards. This is similar to the reduction in deviation ratio projected by the ICCT.

Figure 5-1 also illustrates by how much the COPERT 4v11 (orange line) emissions factors underestimated compared to the latest version (green line).

5.2 Methods

The real driving emissions factors from Chapter 4 were used to create five scenarios for 2030 with varying NO_x and NO_2 emission factors for Euro 6 diesel vehicles. These scenarios were then modelled by the UK Integrated Assessment Model. The UKIAM generated background emissions that remained constant for each scenario, only Euro 6 diesel NO_x emissions factors were changed. The key model outputs were the total NO_x in kilo-tonnes produced by Euro 6 diesel passenger cars (and comparison to UK NEC Directive ceilings for 2030), and annual mean roadside concentrations of NO_2 in $\mu g\ m^{-3}$ (and comparison to annual mean limit value). The total NO_x emission in tonnes was also used as an input for the Abatement Impact Monetisation (AIM) model to calculate the change in Population Weighted Mean Concentration (PWMC) of NO_2 and annual damage cost in Billion £. Damage costs were also calculated using the DEFRA damage costs £/tonne method and comparisons made.

5.2.1 Scenarios

Each scenario had an “a” and “b” component relating to the fraction of primary NO_2 assumed. For “a” scenarios the COPERT fNO_2 emission factor of 30% was used, for “b” scenarios fNO_2 of 44% as measured by PEMS was used.

The scenarios are described in **Table 5-1,** with emissions factors listed in **Table 5-2** and plotted in **Figure 5-2**. The deviation ratios stated in the tables refer to the average deviation ratio. Each scenario uses a scaled version of COPERT’s speed dependent emissions factors (as shown in **Figure 5-2**).

The ICCT report and DEFRA projections indicate that the most likely fleet average deviation ratio by 2030 is somewhere between S3 and S4.

Table 5-1. Description of scenarios

Scenario 1 (S1) – By 2030 all Euro 6 diesel vehicles have real world emissions in compliance with the Euro 6 type approval limit (i.e. fleet average emission factor for NO_x of 0.08 g km^{-1}, deviation ratio = 1). This is the best case scenario, and given the long phase in time for new vehicles, quite unlikely without some form of national scrappage scheme.
Scenario 2 (S2) – By 2030 all Euro 6 diesel vehicles have real world emissions in compliance with the Euro 6d type approval limit (i.e. emission factor for NO_x of 0.17 g km^{-1}, deviation ratio = 2.1). It is likely all new Euro 6 vehicles will be compliant with Euro 6d by 2030 but due to the age of fleet it is unlikely the average deviation ratio will fall to 2.1 by 2030.
Scenario 3 (S3) – By 2030 all Euro 6 diesel vehicles have real world emissions in line with COPERT 4v11 Euro 6 emissions factors, a deviation ratio of 2.4. This is the more ambitious of the two most likely scenarios.
Scenario 4 (S4) – By 2030 all Euro 6 diesel vehicles have real world emissions with trip average emissions factors derived from the PEMS study in the previous chapter, a deviation ratio of 4.5. This is the more pessimistic of the two most likely scenarios.
Scenario 5 (S5) - By 2030 all Euro 6 diesel vehicles have real world emissions derived from the PEMS study in the previous chapter, applying the urban average to urban roads and the motorway average to motorways, an urban deviation ratio of 5.4. This scenario assumes no improvement in Euro 6 diesel deviation ratios; it is a worst case scenario and given the introduction of Euro 6d quite unlikely.

Table 5-2. Emissions factors used in scenarios

Name	Average Euro 6 diesel NO_x [g km^{-1}]	Deviation Ratio	fNO_2 [%]	
			a	b
S1	0.08	1.0	30	44
S2	0.17	2.1	30	44
S3	0.19	2.4	30	44
S4	0.34	4.5	30	44
S5	Motorway (0.31) Urban (0.43)	3.9 5.4	30	44

Figure 5-2 shows the 2030 Euro 6 diesel speed dependent NO_x emission factors for each scenario. These are scaled versions of the COPERT 4v11 curve.

The UKIAM does not model the clear air zones proposed in DEFRA's air quality action plans (DEFRA, 2015b, 2011, 2017a) or the Mayor of London's ambitious plans for a Greater London wide Ultra Low Emission Zone. S4 / S5 (using the real world emissions factors) can be seen as pessimistic Business As Usual scenarios (2030 emissions if no action is taken to reduce Euro 6 diesel emission factors or implement new air quality policies). S1 – S3 can be seen as more optimistic, best case scenarios.

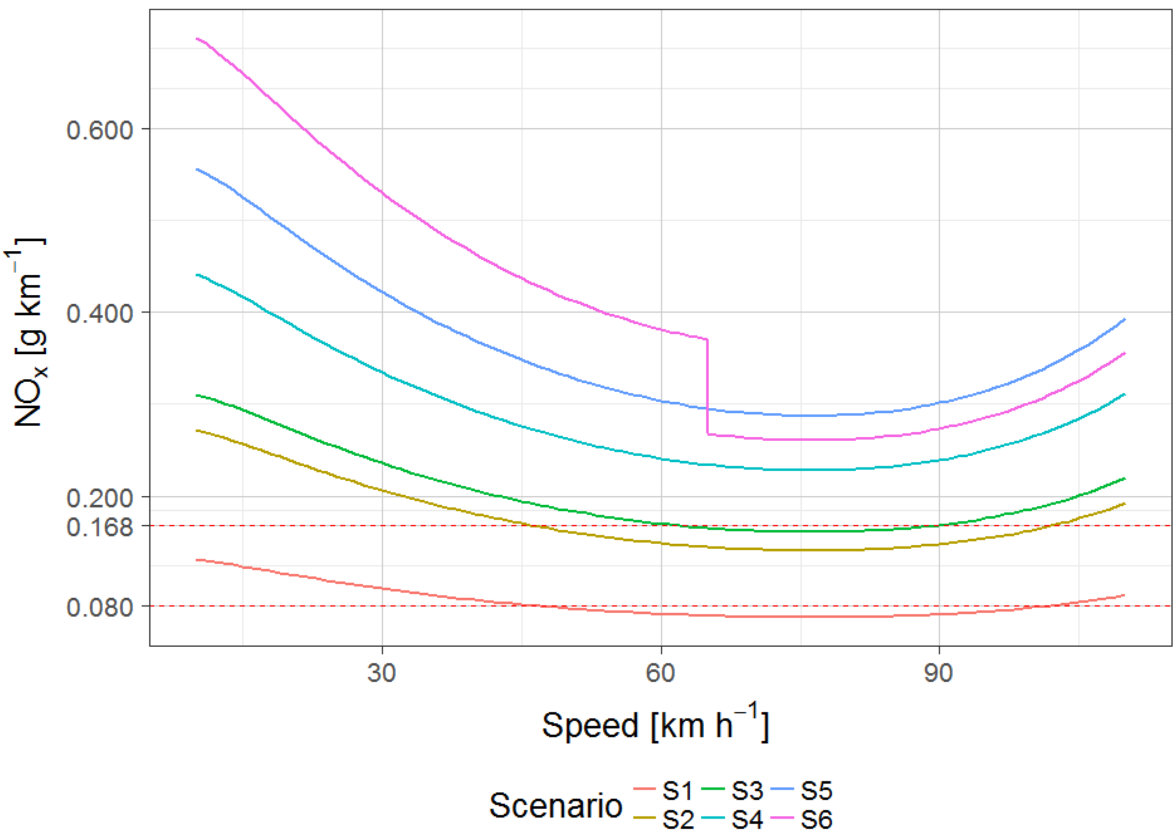

Figure 5-2. Speed dependent NO_x emissions factors from scenarios

5.2.2 UKIAM and BRUTAL

Transport emissions in the UKIAM are simulated by the BRUTAL model. A description of the UKIAM and BRUTAL model has been given in Chapter 2. In this analysis Euro 6 diesel emission factors were isolated, all other inputs were kept constant between scenario runs; the traffic mix, flow, and emissions factors of all other vehicles were kept constant. Results stated relate only to Euro 6 diesel emissions.

5.2.2.1 Fleet mix

The vehicle fleet mix in BRUTAL is taken from NAEI projections. It is projected that by 2030, 92% of diesel passenger cars will be Euro 6 standard (**Table 5-3**) (NAEI, 2014b). The 2030 petrol car projected fleet composition is also stated for comparison.

Table 5-3. 2030 percentage of petrol and diesel cars by technology (NAEI, 2014b) **(ICE = Internal Combustion Engine)**

	Diesel	**Petrol**
Euro 5 (ICE)	1%	1%
Euro 6 (ICE)	92%	78%
Full hybrid	7%	10%
Euro 6 plug in hybrid	-	11%

The fleet split by fuel type again comes from NAEI projections and varies by region and road type. The lowest projected proportion of diesel passenger cars is 28.1% in Central London, the highest is 51.2% on Welsh motorways. The majority of roads are projected to be between 36-44 % diesel passenger cars. These fleet projections were calculated assuming that diesel will continue to make up ~50% of UK passenger car sales. They do not allow for the most recent developments in UK sales, which indicate the diesel share is in decline, however this will be discussed further in the following chapter. This is another large uncertainty. It is possible that a market shift away from diesel will result in fewer diesel cars on the road but a higher average deviation ratio across the Euro 6 fleet. This is because the fleet will contain fewer new cars (meeting Euro 6d) and a higher proportion of older cars with higher deviation ratios (Miller & Franco, 2016).

5.2.3 Damage costs

In this Chapter damage costs will be used from two sources; the Abatement Impact Monetisation (AIM) model and DEFRA guidance.

5.2.3.1 Abatement Impact Monetisation (AIM) model

The AIM model is a spreadsheet model designed by the Integrated Assessment Unit at Imperial College for the Department of Environment Food and Rural Affairs. AIM is a simplified version of the UKAIM that estimates the effects of abatement measures on the exposure of the UK population to NH_3, NO_x, SO_2 and primary $PM_{2.5}$. The main purpose of which is to perform cost benefit analysis for abatement measures listed in the Multi Pollutant Measurement Database (MPMD). AIM uses impact factors calculated by the UKIAM to give the change in population weighted mean exposure per 1 tonne reduction in each pollutant from each source category. The benefit of each measure can then be monetised.

5.2.3.2 Impact Factors

The impact factors in the AIM model are calculated by the UKIAM. The Source Receptor (SR) matrices from the FRAME model are combined with the SR matrix from the UKIAM (for NO_x) to calculate the change in population weighted mean concentration (Δ PWMC n gm^{-3}) per unit change in total UK emission in kilotons of emission by source. The impact factor used in this analysis is listed in **Table 5-4**. This essentially means that for every additional kiloton of NO_x emitted by diesel cars annually the PWMC of NO_2 increases by 30.263 ng m^{-3}.

Table 5-4. Impact factor per kiloton of NO_x

SID	Source Name	ΔPWMC [ng m^{-3}] per 1 kiloton reduction of NO_x
47	07_Road_Transport_Diesel_Cars	30.263

5.2.3.3 Cost per unit of NO_2 exposure for the UK population

The UKIAM and AIM both include low, medium and high cost scenarios. These are taken from existing literature and are described below. The costs come from internal communication with Mike Holland, an economist who is part of the Committee On the Medical Effects of Air Pollution (COMEAP) and are described below. The costs corrected for 2016 per million people are listed in **Table 5-5**.

LOW – The "low" cost scenario is valued at £385 Million £ per year per unit change [μg m^{-3}] in PWMC. This is derived from the Inter Departmental Group on the Costs and Benefits of Air Quality (IGCB (A)) valuation based COMEAP preliminary report that assigned the value of a life year lost at £35,000. This valuation only includes chronic effects on mortality.

MEDIUM – The "medium" cost scenario is valued at £505 Million £ per year per unit change in PWMC. This cost is derived from the DEFRA guidance report.

HIGH – The "high" cost scenario uses the same exposure-response functions (from COMEAP) as the "low" scenario but an alternative valuation of mortality based on a paper for the Interdepartmental Group on Valuation of Life and Health (IGVLH) (Franklin, 2014). A higher value of life year lost (£60,000) is also assumed.

Table 5-5. AIM model damage cost £ million per ΔPWMC NO_2 [μg m^{-3}] per million people

	Low	Medium	High
Million £ per year per ΔPWMC NO_2 [μg m^{-3}] per million people (2016 price)	6.244	8.191	12.245

The change in PWMC NO_2 is assumed to be 70% of the change in PWMC NO_x. This is an approximation and does not account for the non-linear relationship between NO_2 and NO_x, however, it is consistent with the average ratio of background concentration in the UKIAM.

5.2.4 DEFRA damage costs per tonne of NO_x

These values come from DEFRA report "*Damage costs by location and source*" (DEFRA, 2015a). These costs most commonly used to assess national policies, programmes and projects and are stated in £ per tonne of NO_x (**Table 5-8**). The "Travel average" (underlined below) was used for this analysis.

Table 5-6. Damage cost per tonne of NO_x

	Low Central Range [£]	Central Estimate [£]	High Central Range [£]
Transport average	10,101	25,252	40,404
Agriculture	2,020	5,050	8,080
Waste	4,343	10,858	17,373
Energy Supply Industry	505	1,263	2,020
Industry	5,253	13,131	21,010
Domestic	5,859	14,646	23,434
Transport central London	46,162	15,5405	184,648
Transport inner London	47,475	118,688	189,901
Transport outer London	31,010	77,526	124,041
Transport inner conurbation	24,546	61,365	98,184
Transport outer conurbation	15,253	3,8191	61,010
Transport urban big	18,182	45,455	72,728
Transport urban large	14,647	3,6617	58,587
Transport urban medium	11,545	28,788	46,061
Transport urban small	7,273	18,182	29,091
Rural	3,131	7,829	12,526

Table 5-6 highlights the importance of location when assessing the impact of NO_x emissions, the central estimate of damage costs per kiloton of transport related NO_x in central London is ~20 times higher than transport emissions in rural areas. This indicates that reducing emissions in some areas (i.e. urban) is more important than in others. The “Travel average” is calculated by weighting these various damage costs according to their prevalence. The range in the DEFRA low to high estimates reflects the uncertainty in risk coefficients, whereas the range in AIM damage costs reflects differences in the monetisation of health impacts and takes only a central risk coefficient.

The damage costs listed above do not allow for the fact that there is substantial double counting in health effects from emissions of NO_x and $PM_{2.5.}$ Recent discussions with the COMEAP group gave the best estimate of this double counting to be 33%. The “Travel average” damage costs for NO_x once double counting for $PM_{2.5}$ has been removed are listed in **Table 5-7**.

Table 5-7. Travel average damage cost per tonne of NO_x after removing double counting from $PM_{2.5}$

Location and source	Low	Central	High
Travel Average (considering PM)	£6,734	£16,835	£26,936

These values differ from those stated in the 2015 DEFRA guidance. This is because in 2015 DEFRA’s costs considering PM were calculated as the average between the cost not considering PM and the advised COMEAP reduction of 33% (i.e. DEFRA reduced the cost by 1/6 instead of 1/3).

5.3 Results

This section first presents the total NO_x emissions from Euro 6 diesel vehicles from the 5 scenarios and then the associated damage cost estimates. This is followed by analysis of the effect on annual mean NO_2 concentrations.

5.3.1 Total NO_x emissions

Total NO_x emissions increased significantly with the Euro 6 emissions factor. In **Figure 5-3** the light grey section is the NO component and the dark grey is NO_2. The total NO_x emission is constant between the 'a' and 'b' scenarios but the amount of NO and NO_2 varies.

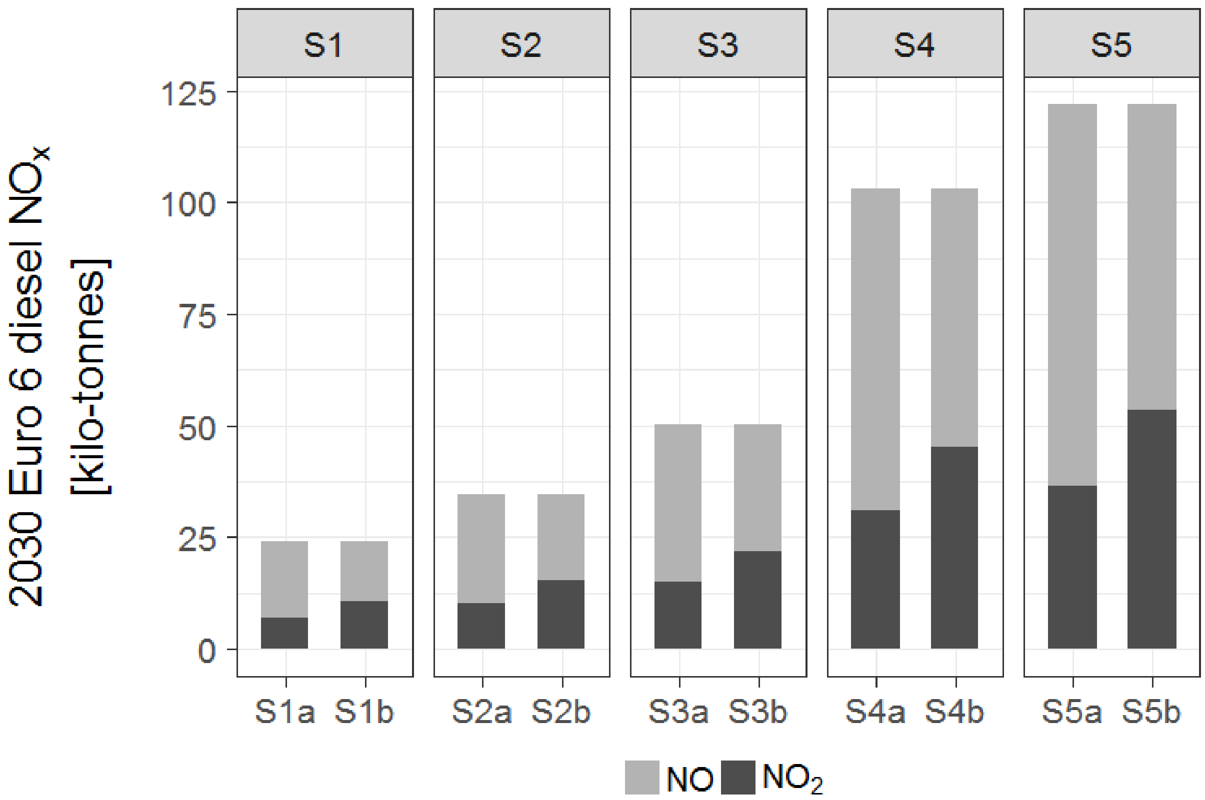

Figure 5-3. Total Euro 6 diesel 2030 NO_x and NO_2 emissions in kilo-tonnes (Scenario a- fNO_2=30%, Scenario b- fNO_2=44%)

The difference between S5 (worst case scenario) and S1 (best case scenario) was 97.9 kilotons. This represents the potential additional amount of NO_x emitted in the year 2030 as a result of diesel passenger cars not meeting the type approval limit during real world driving if the deviation ratio is not reduced.

Table 5-8. 2030 total Euro 6 NO_x emissions and comparison to National Emissions Ceiling

Scenario	NO_x [kilotons]	% of 2030 NEC [429.8 kilotons]
S1	24.0	5.6 %
S2	34.8	8.1 %
S3	50.3	11.7 %
S4	102.9	23.9 %
S5	121.9	28.4 %

The 2030 UK National Emission Ceiling for NO_x is 429.8 kilo-tonnes. The difference between the best and worst case scenarios in this analysis makes up a significant proportion of this ceiling. **Table 5-8** lists the % of NO_x in kilo-tonnes that Euro 6 diesel cars (~90% of all diesel cars by 2030) would make of the 2030 UK emission inventory if the National Emission Ceiling (429.8 kilo-tonnes) was met. For context, in 2014 ~13% of the UK's total NO_x emissions came from diesel passenger cars. This is similar to the 11.7% predicted by S3. However, for S5 diesel passenger cars would produce 28.4% of the 2030 ceiling. The results in **Table 5-8** highlight the need for greater certainty in projected Euro 6 diesel emissions factor reduction if the UK is to meet its 2030 national emission reduction commitment.

5.3.2 Damage costs

In this section the total Euro 6 NO_x emissions calculated for 2030 by the UKIAM are used to derive the relevant damage cost from each scenario using the costs described in the Methodology earlier in this chapter.

5.3.2.1 AIM damage cost

Total Euro 6 NO_x emissions from the UKIAM were multiplied by impact factors and costs from the AIM model to calculate the cost of the scenarios due to the change in PWMC of NO_2. The results are listed in **Table 5-9**.

Table 5-9. ΔPWMC NO_2 and damage costs calculated using AIM model

Scenario	NO_x [kilotons]	Δ PWMC NO_2 [μg m^{-3}]	Cost [Billion £]		
			Low	Medium	High
S1	24.0	0.51	0.23	0.31	0.46
S2	34.8	0.74	0.34	0.45	0.67
S3	50.3	1.07	0.49	0.64	0.96
S4	102.9	2.18	1.00	1.32	1.97
S5	121.9	2.58	1.19	1.56	2.33

Table 5-9 shows the difference in 2030 PWMC between S1 and S5 (best and worst case scenario) was 2.07 μg m^{-3} with a cost of between 0.96 – 1.87 Billion £. The difference in 2030 PWMC NO_2 between Euro 6 emissions as modelled by COPERT

4v11 (S3) and real world driving (S5) was 1.51 μg m^{-3} with a cost of between 0.7 – 1.37 Billion £.

Using the AIM model the damage costs per tonne for diesel cars for 2030 were £9,752 (low), £12,793 (medium), £19,125 (high).

5.3.2.2 DEFRA damage costs

The damage costs associated with the 2030 modelled Euro 6 NO_x emissions using DEFRA 2015 "Travel Average" costs (not considering PM) are listed in **Table 5-10**.

Table 5-10. DEFRA damage costs not considering PM

Scenario	NO_x [kilotons]	Cost [Billion £]		
		Low [£10,101 £/tonne]	Medium [£25,252 £/tonne]	High [£40,404 £/tonne]
S1	24.0	0.24	0.61	0.97
S2	34.8	0.35	0.88	1.41
S3	50.3	0.51	1.27	2.03
S4	102.9	1.04	2.60	4.16
S5	121.9	1.23	3.08	4.93

The damage costs calculated from the DEFRA 2015 "Travel Average" cost per tonne were consistently higher than the AIM model estimates. This is mostly because the DEFRA damage costs do not allow for double counting of health effects between PM and NO_2 (approximately 1/3). **Table 5-10** shows that using the DEFRA damage costs the difference in 2030 between S1 and S5 (best and worst case scenario) was between 0.99 – 3.69 Billion £.

The damage costs associated with the 2030 Euro 6 NO_x emissions using the DEFRA 2015 “Travel Average” damage costs for NO_x, removing double counting for PM, are listed in **Table 5-11.** These are more in line with the AIM damage costs listed in **Table 5-9**.

Table 5-11. DEFRA damage costs removing double counting for PM

Scenario	**NO_x** [kilo-tonnes]	**Cost [Billion £]**		
		Low [£6,734 £/tonne]	**Medium** [£16,835 £/tonne]	**High** [£26,936 £/tonne]
S1	24.0	0.16	0.40	0.65
S2	34.8	0.23	0.59	0.94
S3	50.3	0.34	0.85	1.35
S4	102.9	0.69	1.73	2.77
S5	121.9	0.82	2.05	3.28

5.3.2.3 Combination and comparison of damage costs

Figure 5-4 is a boxplot showing the low, medium and high estimates from the three different damage costs for each scenario. The red triangle marks the mean of the 9 different costs, the value of the mean for each scenario is listed in **Table 5-12**.

Table 5-12. Mean damage cost by scenario

Scenario	Mean damage cost Billion £
S1	0.45 (sd. 0.26)
S2	0.65 (sd. 0.37)
S3	0.94 (sd. 0.54)
S4	1.92 (sd. 1.10)
S5	2.28 (sd. 1.31)

To put these costs in context, the Royal College of Physicians currently value the total economic cost to the UK from the impact of air pollution at £20 Billion a year (RCP, 2016). This is similar to the annual national cost of obesity of £27 Billion (Morgan & Dent, 2010). The 2015 estimated annual tax revenue from diesel cars was £5.6 Billion (Brand, 2016).

Given that the ICCT predicted a Euro 6 real world deviation ratio between S3 and S4, this modelling indicates 2030 UK damage costs from Euro 6 diesel vehicles are likely to be between 0.95 – 1.92 Billion £. The additional annual cost to the UK by 2030 of Euro 6 vehicles not meeting type approval limits in the real world (S3/4 – S1) would be between 0.49 – 1.47 Billion £.

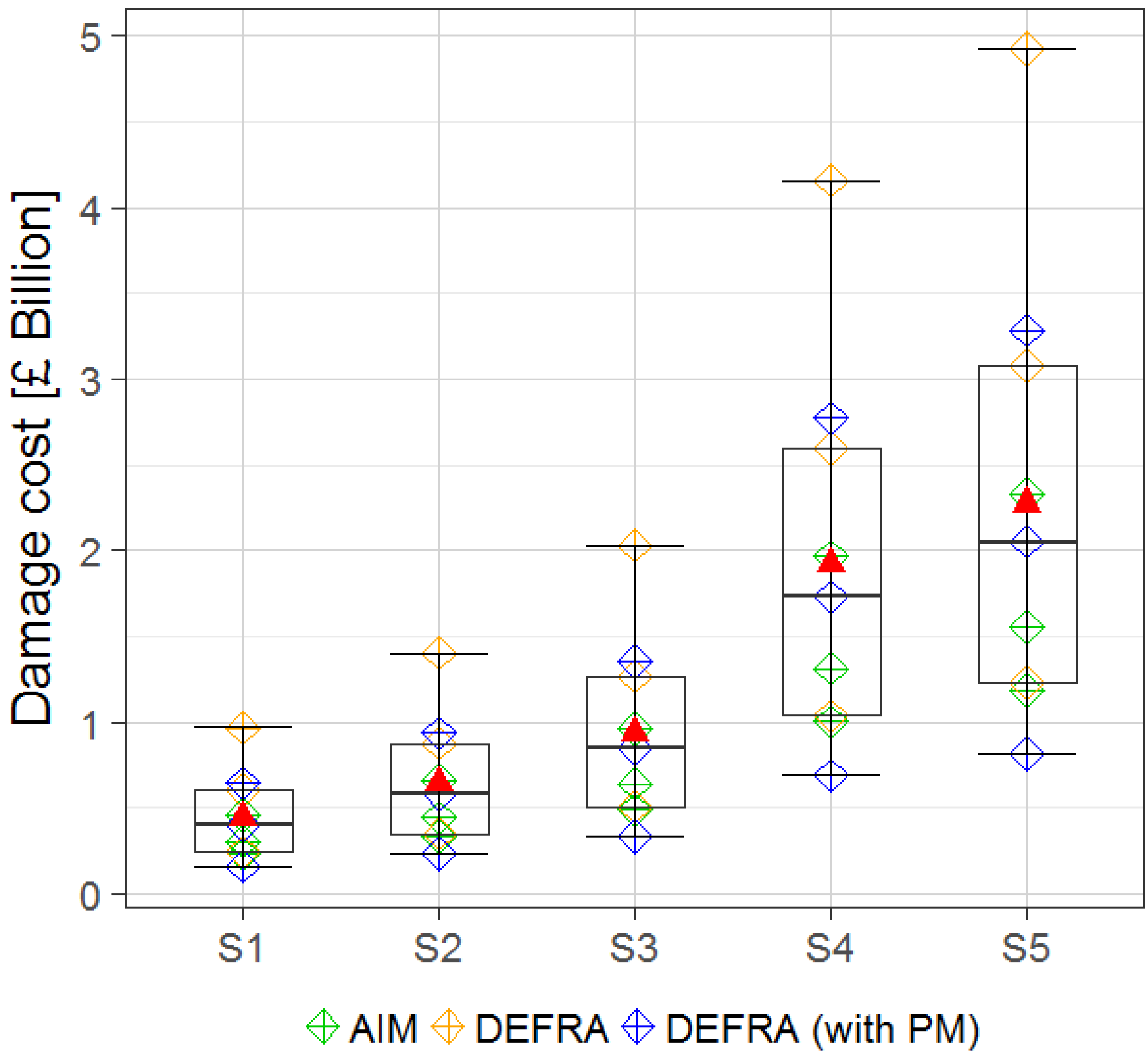

Figure 5-4. Total cost in Billion £ by scenario using all damage costs

Figure 5-4 highlights the large amount of uncertainty there is relating to the true cost of NO_x emissions. It also shows that the DEFRA costs that do not allow for the double counting of NO_2 and PM (orange) are far higher and in less agreement than the AIM model and the DEFRA study that allows for 33% double counting.

5.3.3 Annual mean concentrations of NO_2

Table 5-13 lists the number of grid-squares in 2030 projected to contain roads with annual mean NO_2 emissions in exceedance of 40 μg m^{-3}. The exceedances are listed for each scenario for the whole of the UK and for Greater London (excluding Heathrow). As mentioned previously, this modelling does not include the clear air zones proposed in DEFRA's air quality action plans (DEFRA, 2015b, 2011, 2017a).

Table 5-13. Number of grid squares in 2030 with NO_2 roadside exceedances (excluding Heathrow)

	UK		London	
	a	**b**	**a**	**b**
S1	0	1	0	1
S2	0	2	0	1
S3	9	18	3	4
S4	43	79	13	17
S5	67	136	16	36

The locations of the grid-squares with roads at risk of exceedance are plotted in **Figure 5-5**: ~30% were in London. As discussed earlier, this modelling did not account for the Mayor of London's new T-charge or Ultra Low Emission Zone (ULEZ). It is hoped that these policies when implemented will mitigate most if not all exceedances in the Greater London area.

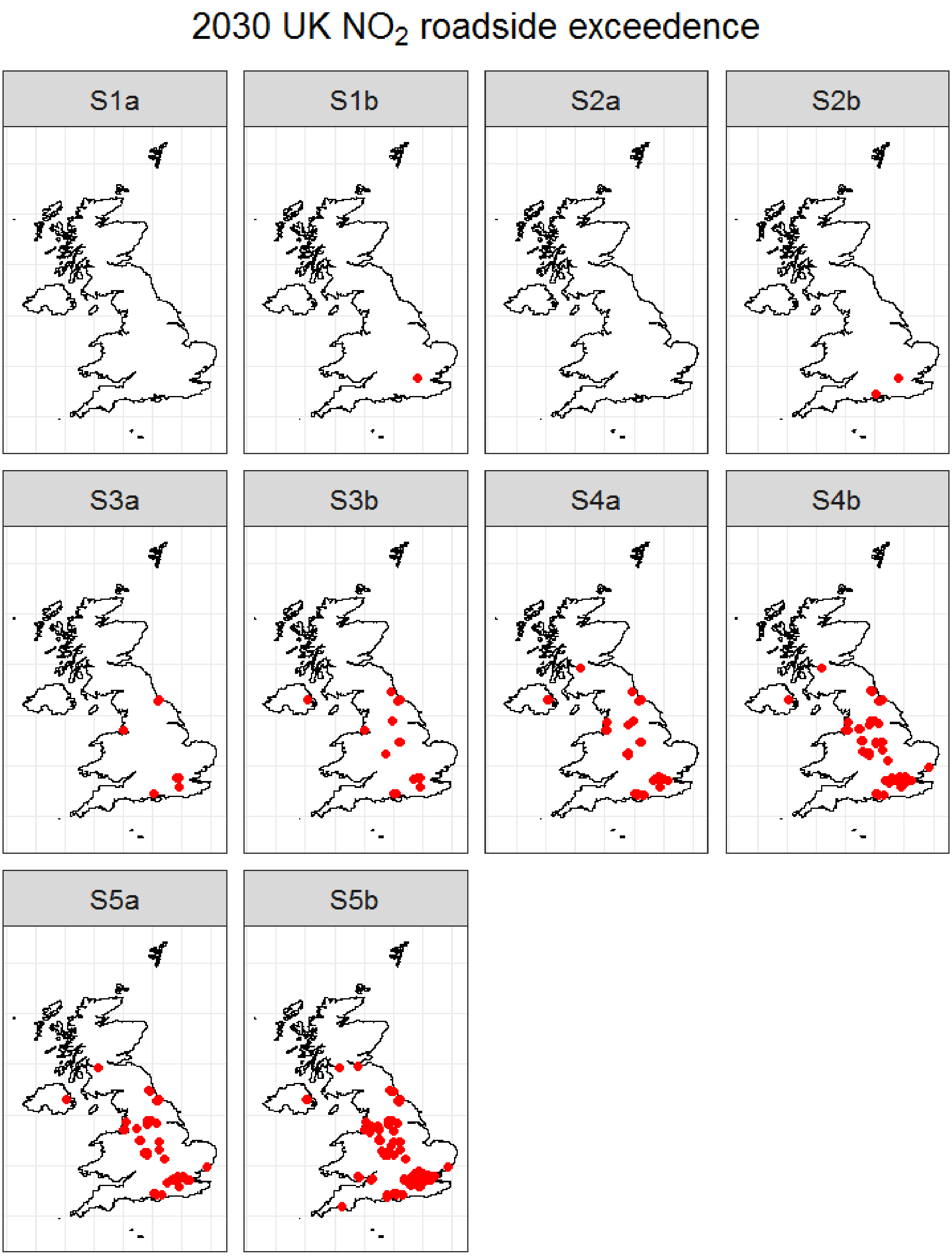

Figure 5-5. 2030 NO_2 roadside exceedances by location (UK)

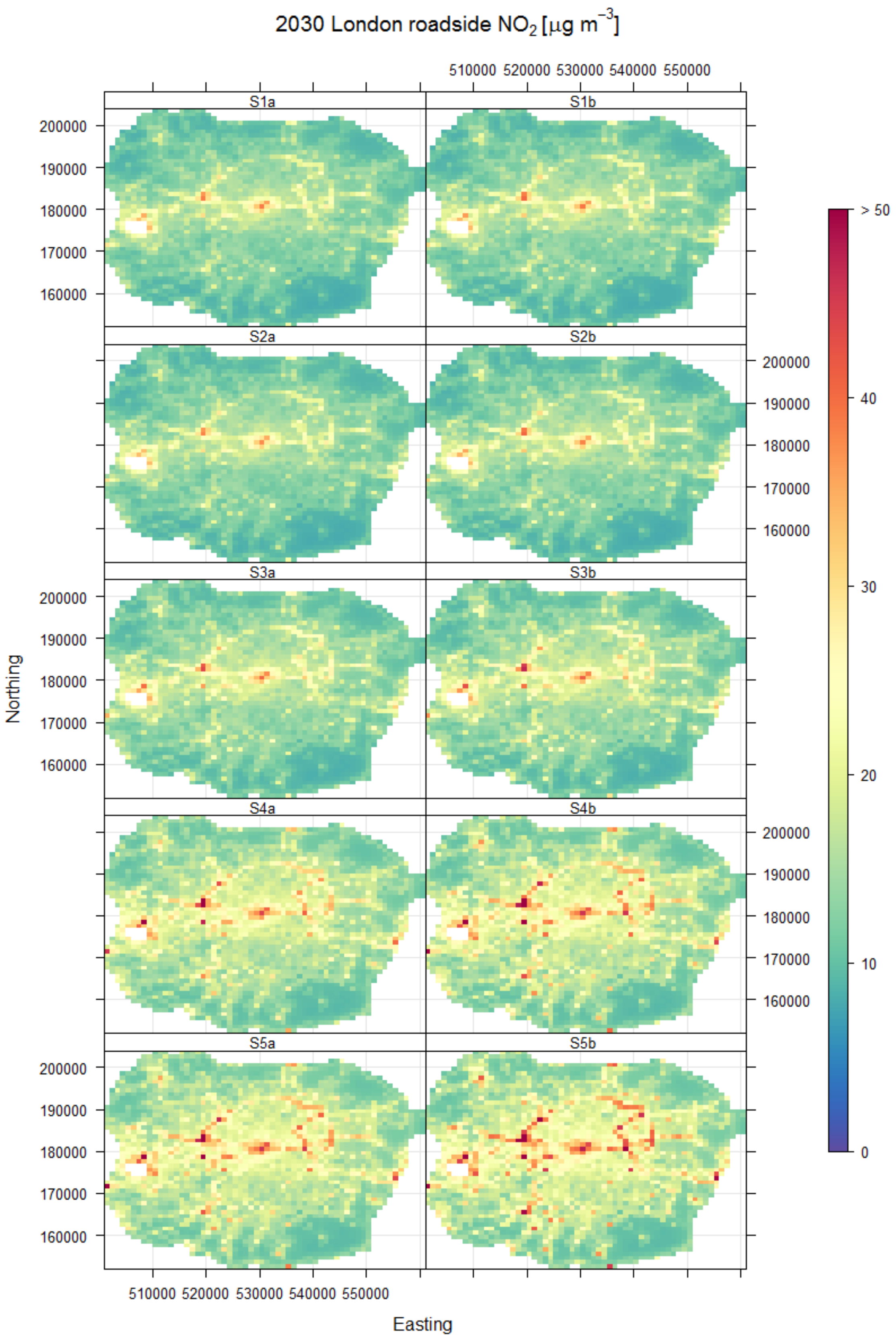

Figure 5-6. 2030 Greater London NO_2 (μg m^{-3}) (excluding Heathrow)

Figure 5-6 is a concentration map of 2030 annual mean NO_2 for Greater London. Heathrow has been removed for reasons discussed in Chapter 2. **Figure 5-6** shows that without the ULEZ and a reduction in deviation ratio there may still be roadside exceedances of the annual mean limit in London in 2030. S1a and S2a were the only scenarios with no exceedances. This shows that reducing the Euro 6 fleet average deviation ratio is essential if the UK is to be in compliance with limit values.

Table 5-13 (number of grid squares in 2030 with NO_2 roadside exceedances) showed that whilst fNO_2 did not affect total NO_x (in tonnes), it did affect the number of grid-squares with roads at risk of exceedance. An increase in fNO_2 from 30 – 44% (47% increase in fNO_2) resulted in a national increase in roads at risk of exceedance of between 84 – 103%. There was an increase in annual mean NO_2 concentrations between the two scenarios for all grid-squares. The increase in fNO_2 between 'a' and 'b' led to an increase in roadside concentrations as seen in **Figure 5-7**. The discrepancy between 'a' and 'b' scenarios increased with the NO_x emission factor. This is expected, as fNO_2 is a fixed ratio, therefore as NO_x emissions increased NO_2 emissions increased at the same rate.

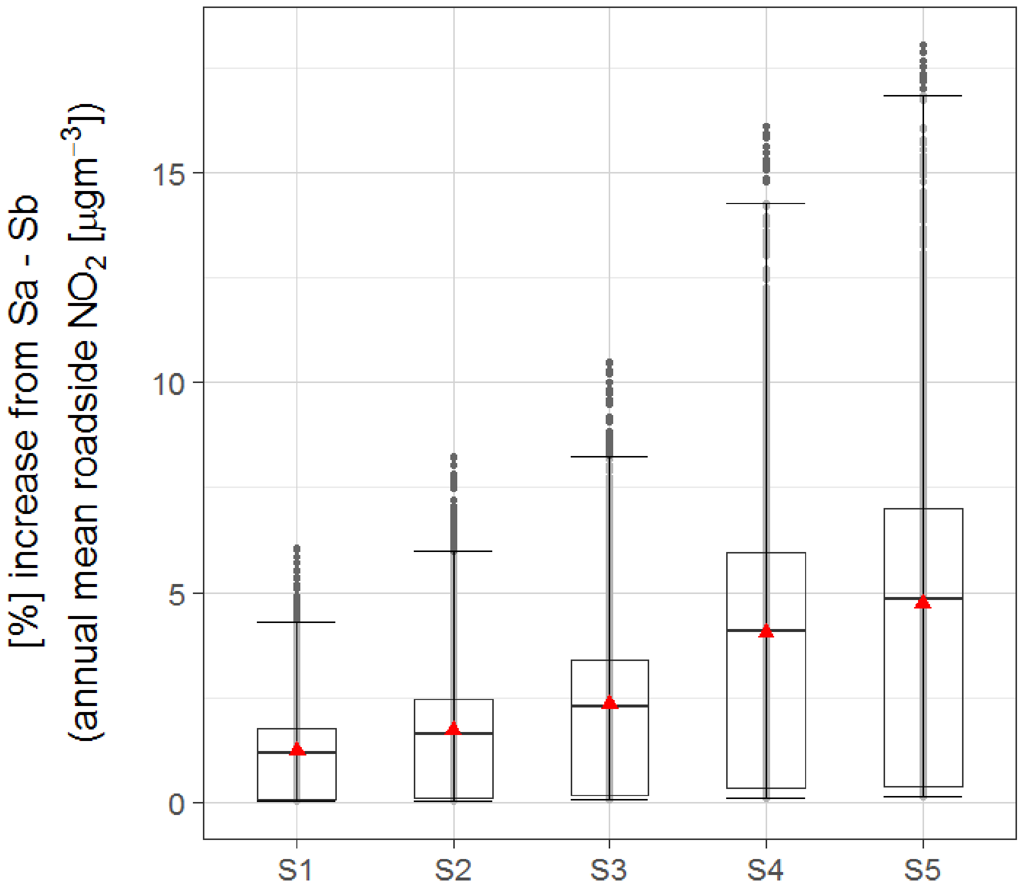

Figure 5-7. Ratio of Sb (fNO_2 = 44%) to Sa (fNO_2 = 30%) across Greater London

Figure 5-7 is the ratio of annual mean roadside concentration of NO_2 from the 'a' and 'b' components from each scenario for the ~2500 grid squares in London. The majority of concentration increases did not result in a compliant grid-square being forced into exceedance, though for some the increase was substantial (>15%). Any increase in ambient NO_2 concentrations, even if small, poses a risk to public health. For S3, the 'b' scenario annual mean roadside NO_2 concentrations were on average 2.3% (equating to 0.4 µg m^{-3}) higher than 'a' scenario, and the highest increase was 10.5% (equating to 3.5 µg m^{-3}). For S4, the average increase was 4.0% (equating to 0.8 µg m^{-3}) and the highest increase was 16.1% (equating to 7.2 µg m^{-3}).

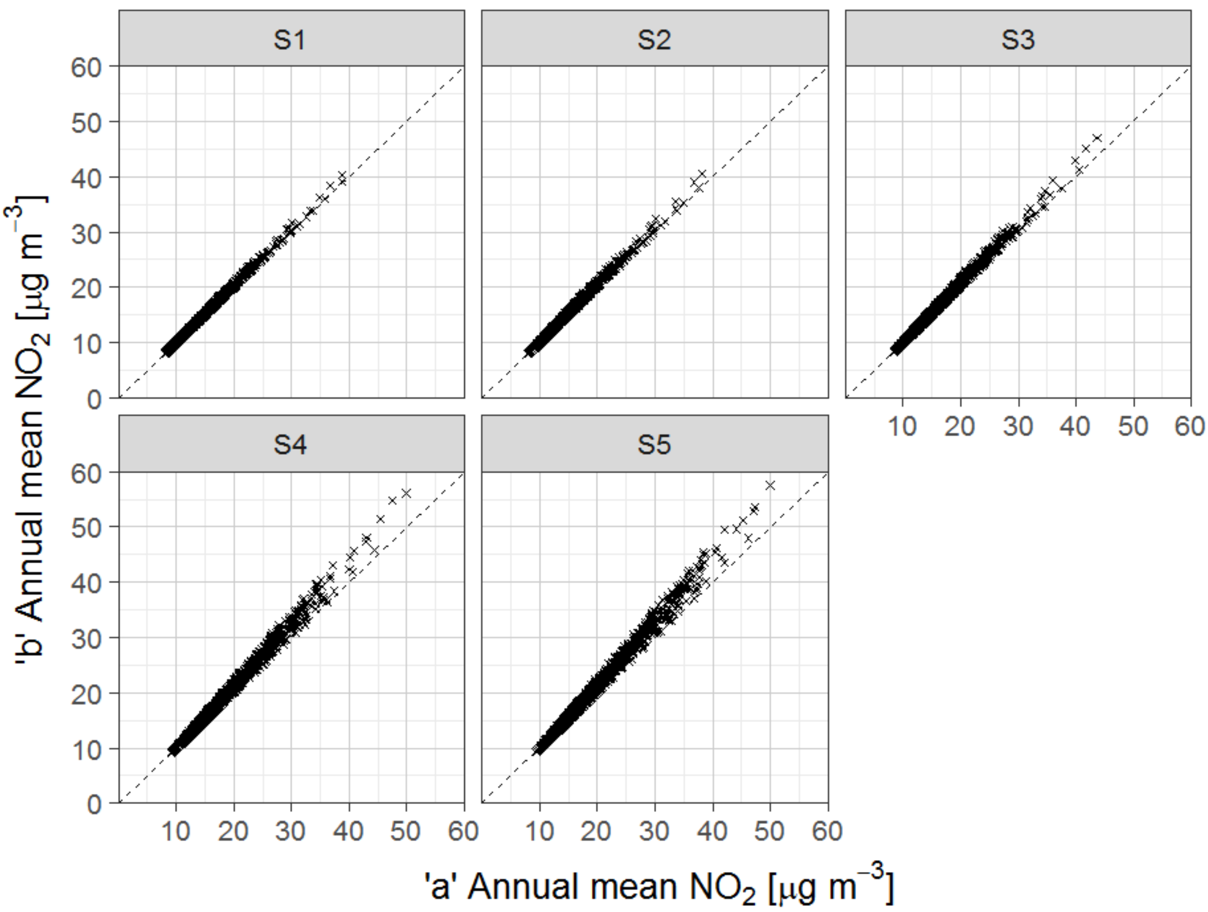

Figure 5-8. Annual mean NO_2 'a' component vs. 'b' component by scenario

The increase in annual mean NO_2 concentrations between 'a' and 'b' scenarios was greater at roadside locations with higher concentrations of NO_2, as seen in **Figure 5-8**. This is because higher concentrations occurred on busier roads with taller buildings causing a street canyon effect. The BRUTAL model assigns a higher "street canyon" factor to those grid squares, therefore the roadside increment was higher. This is representative of real world roadside locations in urban areas where often the background O_3 has already been depleted, limiting the amount of O_3 available for the fast chemistry reactions described in Chapter 2. In these circumstances NO_x emitted directly as primary NO_2 becomes a dominant factor in ambient concentrations (Degraeuwe *et al.*, 2015; Carslaw *et al.*, 2016).

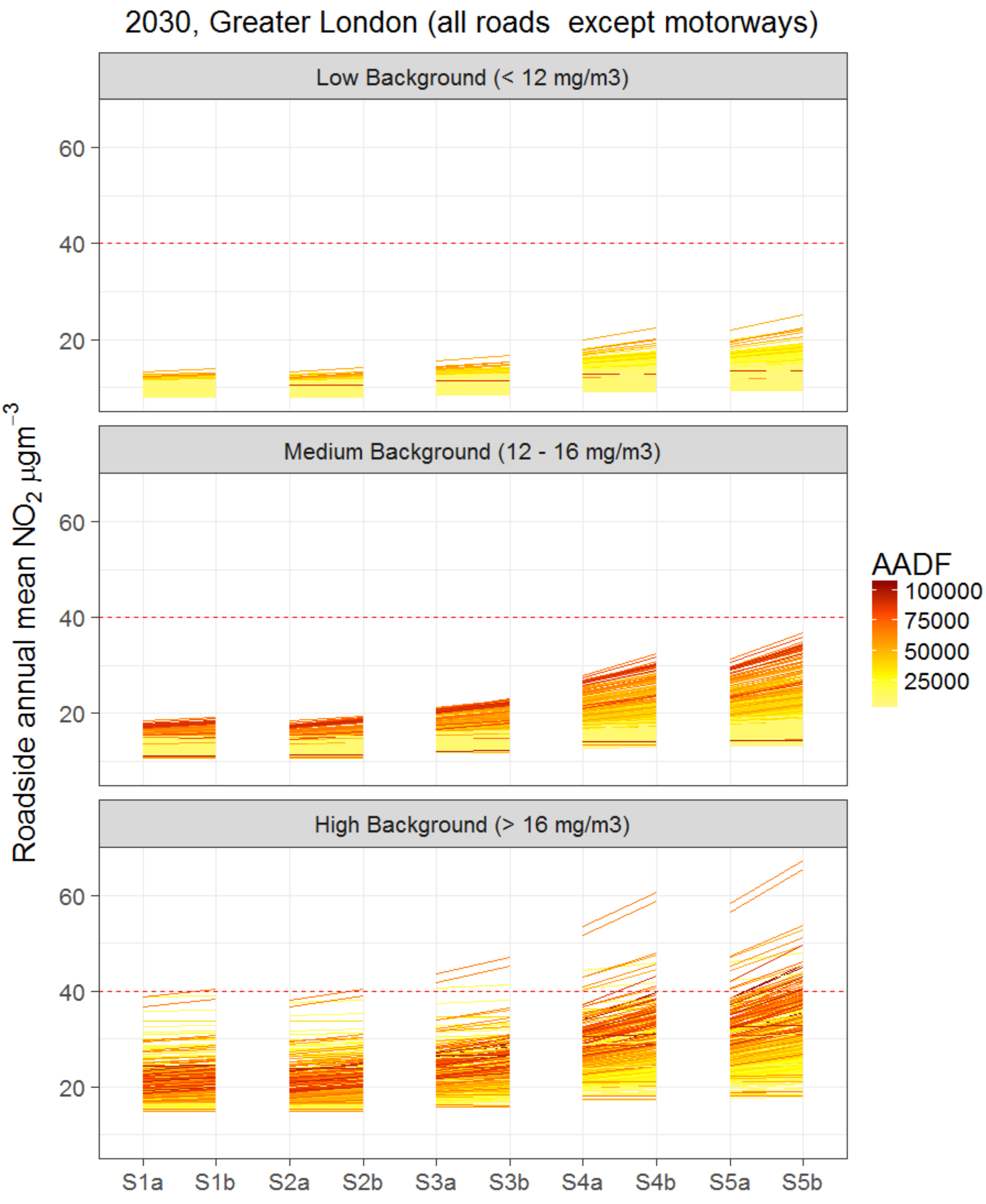

Figure 5-9. Roadside annual mean NO_2 by background concentrations and traffic flow

Figure 5-9 shows the effect of background concentrations and traffic flow on annual mean roadside NO_2. AADF stands for Annual Average Daily Flow. Each diagonal line connects an 'a' component of a scenario for an individual grid square to the 'b' component for the same location. The slope of the line represents the magnitude of the increase from 'a' to 'b', the colour of the line represents the AADF (red/ orange the highest, yellow the lowest). **Figure 5-9** again shows that the increase in annual mean NO_2 from 'a' to 'b' is greater for scenarios with higher deviation ratios and higher background concentrations. This indicates, similar to the findings of Degraeuwe, Thunis, Clappier, *et al.*, (2015), that if the Euro 6 deviation ratio is reduced to compliance (S1), then fNO_2 will be less of a cause for concern. However, if Euro 6 diesels continue with higher deviation ratios (S4/S5) fNO_2 will have a much greater impact on annual mean roadside concentrations of NO_2.

Figure 5-9 highlights the importance of the background concentration's contribution to roadside exceedances. The vast majority of roads with low background concentrations also had low AADF. There were also many roads with high AADF that had medium background concentrations and were still in compliance with Air Quality Limit Value for NO_2 (marked as red dashed line). The highest annual mean roadside concentrations of NO_2 were recorded at locations with high background concentrations and high AADF.

5.4 Discussion

The scenarios used in this analysis were carefully chosen to cover the best and worst cases for the evolution of Euro 6 diesel NO_x emissions by 2030. However, these projections are subject to huge uncertainties and limitations. In this section these uncertainties and limitations will be outlined briefly and discussed.

5.4.1 Emissions factors

Firstly, there is huge uncertainty in how the emissions factors for Euro 6 will evolve with new type approval. Thanks to a growing body of PEMS evidence there is now a clear picture developing of the current real world emissions from Euro 6 diesels. Failure to account for how deviation ratios will evolve, and the assumption that type approval will translate to proportional real world reductions, was a mistake of policy makers regarding the introduction of Euro 4 and Euro 5 standards (Beevers *et al.*, 2012). This analysis ensured that all likely scenarios were covered by consulting existing literature from the ICCT and DEFRA. For this reason, S3 and S4 have been used to provide a most likely range of total NO_x and costs. However, learning from previous mistakes, policy makers should err on the side of caution and deploy the precautionary principle.

5.4.2 Market share diesel

There is currently huge uncertainty in the future market share of diesel in the UK fleet. Early evidence seems to indicate changes in public opinion and Vehicle Excise Duty are driving consumers away from diesel, but at the time of writing it was still too soon for certainty. Given the slow turnover of the passenger car fleet (without introduction of a national scrappage scheme, not proposed at the time of writing) these changes will take approximately a decade to become significant. The ICCT project the lowest likely diesel market share percentage by 2030 to be 20%. The 36 – 44 % in this study is the highest likely diesel share, but given current emerging trends it is very unlikely that the market share of diesel will increase. It is therefore likely that the 2030 Euro 6 diesel activity will be less than what is modelled in this study. However, as previously discussed, a slowdown in the renewal of the Euro 6 fleet will also slow down the reduction in deviation ratio.

5.4.3 Comparison with other studies

Following on from this there are several key policies (ULEZ, CAZ) that were not modelled in these scenarios. As a result, the projections from this study present a more pessimistic view than other studies. However, within the range of scenarios there is a level of agreement. The latest DEFRA action plan projects that there will be compliance in all but 1 zone by 2030 (DEFRA, 2017b). This result is in agreement with the S1 and S2 scenarios modelled in this study. This is because, as discussed previously, DEFRA project that after 2020 the Euro 6 deviation ratio will be 2.5. Rather than a direct comparison between the two studies, it may be more useful to view the analysis in this chapter as an assessment of what could potentially happen if the Euro 6d type approval process fails to effectively bring down the Euro 6 average deviation ratio.

5.4.4 Other pollutants

It should also be noted that this analysis focused on the costs relating to NO_x alone. There are also damage costs associated with emissions of CO_2, PM, PAH and CO, VOCs and SO_2. Consideration particularly of CO_2 emissions is important to frame this discussion because a reduction in CO_2 emissions (and the air quality / climate change trade off) was the initial argument for the mass introduction of diesel vehicles. This will explored further in the next chapter.

5.5 Summary

Five scenarios assuming various deviation ratios for Euro 6 diesel passenger cars were modelled for 2030 using the UKIAM. The scenarios (S1 – S5) started with the most optimistic (S1) assuming service conformity to the type approval limit (0.08 g NO_x km^{-1}), and ended with most pessimistic (S5), which used deviation ratios devised in the previous chapter and assumed no improvement between 2016 and 2030. Each scenario had an ‘a’ and ‘b’ component relating to fNO_2 (30% for ‘a’, 44% for ‘b’).

Uncertainty in both the deviation ratio and fNO_2 of the Euro 6 diesel passenger fleet creates uncertainty in the UKs ability to comply with both the National Emissions Ceiling Directive and the Air Quality Framework Directive. The difference between the best and worst case scenarios amounted to a substantial proportion of the entire UK 2030 NO_x allowance, and the number of grid squares with roads at risk of exceedance varied from 0 for scenario 1a to 136 for scenario 5b.

The total NO_x modelled in the scenarios ranged from 24 – 121.9 kilotons, though the most likely range was between 50.3 –102.9 kilotons. Using nine different damage cost estimates, this study calculated that the most likely range for annual damage cost for 2030 from Euro 6 diesel cars was between 0.95 – 1.92 Billion £.

The fraction of NO_x emitted as NO_2 (fNO_2) was found to have a significant impact on roadside concentrations, and was more significant for scenarios with higher deviation ratios.

Chapter 6. CO_2 and NO_x emissions from diesel and petrol passenger cars

The previous two chapters focused on NO_x emissions from Euro 6 diesel passenger cars. This chapter extends the scope to include Euro 5 and Euro 6 diesel, petrol and hybrid passenger cars as well as CO_2 and CO emissions. The aim is to present an accurate representation of emissions from the current Euro 5 and 6 passenger cars in order to inform policies relating to both air quality and climate change objectives.

6.1 Background

In Europe the majority of passenger cars are fuelled by either petrol or diesel internal combustion engines. The market share of diesel varies between member states, though in recent years the European average has been ~50% (ICCT, 2016a). In the UK diesel accounts for ~40% of the currently licenced passenger car fleet (DfT, 2016c). As discussed emissions from petrol and diesel vehicles have different exhaust compositions due to differences in energy density and engine mechanics. Previous studies have found diesel engines produce between 20 – 30 % less CO_2 but emit many times more NO_x (Suzuki & Matsumoto, 2004; Moody & Tate, 2017; Weiss *et al.*, 2012).

In the mid 1990's the vast majority of the European passenger fleet was fuelled by petrol. The Kyoto Protocol of 1997 committed signatories (of which the UK was one) to reducing their CO_2 emissions by 8% over the next 15 years (UNFCCC, 1998). Whilst America and Japan focused CO_2 reduction efforts on hybrid and electric vehicles, the EU opted to promote diesel fuel. Diesel was touted as the environmentally friendly alternative to petrol and promoted through tax incentives. This led to a peak in the EU wide market share diesel of 52% in 2015 (ICCT, 2016). However, recent trends indicate that the VW emissions scandal combined with the growing body of evidence relating to the adverse health effects associated with diesel fumes have started a decline in diesel sales (RCP, 2016; COMEAP, 2010; WHO, 2016; EEA, 2015; FT, 2016). This is illustrated by **Figure 6-1**.

In **Figure 6-1** the bars represent the difference in total passenger car sales in the UK between 2016 and 2017. A positive value represents an increase from 2016- 17, a negative value represents a decrease. The green section of each bar relates to diesel vehicles. Since June 2016 the majority of months have seen a decrease in the number of diesel cars sold compared to the same month in the previous year. The increase in March 2017 was due to a rush to buy vehicles before the UK government's change to the Vehicle Excise Duty (VED, a form of tax). The VED changes introduced in April 2017 considerably increased the annual cost of keeping diesel cars on the road and applied to all vehicles registered after March 2017. These changes were introduced (in part) to disincentivise diesel. Early results indicate this was a success, with April 2017 diesel passenger car sales -27% lower than April 2016 and May 2017 sales 20% lower than May 2016.

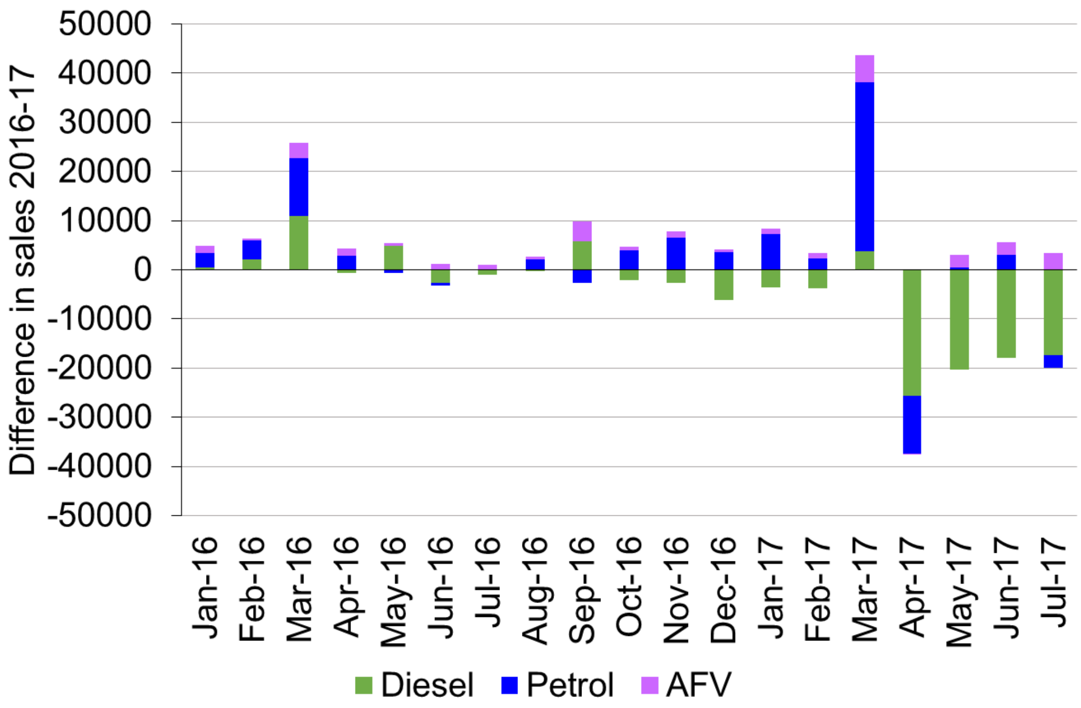

Figure 6-1. Comparison of UK passenger car sales to previous year by fuel type (SMMT, 2017) (AFV= Alternative Fuel Vehicle)

However, total passenger car sales in the UK are increasing annually. The annual average VKM driven are also increasing (DfT, 2014), which is the case across the EU. Whilst technological advancements have reduced vehicles' on-road CO_2 emissions, increased activity has outweighed carbon intensity improvements. This has resulted in transport being the only major sector in the EU for which greenhouse gas emissions continue to rise (CCC, 2015; Fontaras, Zacharof & Ciuffo, 2017). It is therefore a matter of concern that (as shown in **Figure 6-1**) the majority of consumers moving away from diesel are switching back to petrol instead of alternative fuel vehicles (AFV). Across Europe the fall in diesel sales since 2015 has been entirely offset by an increase in petrol sales, whilst from 2015 – 2016 the share of AFV fell from 4.5% to 4.2% (ACEA, 2017b).

AFV refers to vehicles that are powered by sources other than traditional petroleum fuels using internal combustion engines. This includes electric vehicles (EV), plug in hybrid- electric vehicles (PHEV), hybrid- electric vehicles (HEV) and hydrogen fuel cells. Currently in the UK over half all AFVs are petrol- electric hybrids, these have been found to deliver fuel economy savings between 40- 60% relative to conventional petrol vehicles (Fontaras, Pistikopoulos & Samaras, 2008). In the UK AFVs are the fastest growing passenger car sector, and this growth has accelerated since the VED change in April 2017. May 2017 represented a 47% increase in sales compared to May 2016, however, the market share (4.4%) was still relatively low.

6.2 Methodology

The testing regime undertaken by Emissions Analytics followed exactly the same procedure as described in Chapter 4. The Euro 6 diesel vehicles were the same as used in Chapter 4, however the urban and motorway section selection method was different. In Chapter 4, sections were selected by road type and GPS, whereas in this chapter sections were selected following "*EU Commission Regulation (EU) 2016/646 of 20 April 2016 amending Regulation (EC) No 692/2008 as regards emissions from light passenger and commercial vehicles (Euro 6)".* Regulation (EU) 2016/646 dictates the procedure for the RDE test component of Euro 6d TEMP. The sections selected in this chapter followed this guidance including the minimum and maximum dynamic boundary conditions relating to relative positive acceleration (RPA) (which takes a different definition from that used in Chapter 4) and $v.a_{pos}$_[95] (defined below). However, some guidance from (EU) 2016/646 (e.g. speed binning) was not followed. Areas where the selection method used in this chapter differs from (EU) 2016/646 are discussed below.

6.2.1 Test fleet

The test fleet contained 37 Petrol Euro 5 (P5), 35 Petrol Euro 6 (P6), 36 Diesel Euro 5 (D5), 39 Diesel Euro 6 (D6), 1 Euro 5 petrol- electric Hybrid (H5) and 1 Euro 6 petrol-electric Hybrid (H6). The vehicle models in the test fleet accounted for 56% of all passenger cars sold in Europe in 2016 and included 27 different manufacturers. **Table 6-1** lists the main characteristics of the 149 vehicles in the test fleet.

The Euro 6 diesel vehicles were the same as those used in Chapter 4.

Table 6-1. Characteristics of vehicles in test fleet

Vehicle ID	Fuel	Euro	Engine size [ℓ]	Segment	Kerb weight [kg]	Year of manufacture	Mileage [km]	Fuel injection
D5.1.5a	Diesel	Euro 5	1.5	C	1500-2000	Dec-13	5123	-
D5.1.5b	Diesel	Euro 5	1.5	B	1500-2000	Jun-13	5057	-
D5.1.5c	Diesel	Euro 5	1.5	C	1500-2000	Dec-13	5087	-
D5.1.5d	Diesel	Euro 5	1.5	C	1500-2000	Jan-14	1585	-
D5.1.6a	Diesel	Euro 5	1.6	I	1500-2000	Oct-14	2268	-
D5.1.6c	Diesel	Euro 5	1.6	I	1500-2000	Nov-13	946	-
D5.1.6d	Diesel	Euro 5	1.6	B	1000-1500	Apr-14	4741	-
D5.1.6e	Diesel	Euro 5	1.6	C	1500-2000	Jan-14	2794	-
D5.1.6f	Diesel	Euro 5	1.6	H	2000-2500	Jul-14	2548	-
D5.1.6g	Diesel	Euro 5	1.6	H	1500-2000	Jul-14	5333	-
D5.1.6h	Diesel	Euro 5	1.6	C	1500-2000	Nov-11	10240	-
D5.1.6i	Diesel	Euro 5	1.6	C	1500-2000	Jan-14	6442	-
D5.1.6j	Diesel	Euro 5	1.6	D	1500-2000	Feb-14	2937	-
D5.1.6k	Diesel	Euro 5	1.6	D	1500-2000	Mar-13	2831	-
D5.1.6l	Diesel	Euro 5	1.6	C	1500-2000	Oct-13	3174	-
D5.1.6m	Diesel	Euro 5	1.6	I	1500-2000	Feb-14	2200	-
D5.1.6n	Diesel	Euro 5	1.6	C	1500-2000	Jan-14	5789	-
D5.1.6o	Diesel	Euro 5	1.6	B	1500-2000	Jul-12	6185	-
D5.1.6p	Diesel	Euro 5	1.6	C	1500-2000	Jan-14	4519	-
D5.1.6q	Diesel	Euro 5	1.6	C	1500-2000	Mar-14	4216	-
D5.1.6s	Diesel	Euro 5	1.6	C	1500-2000	Mar-13	1294	-
D5.1.6t	Diesel	Euro 5	1.6	B	1500-2000	Nov-12	5113	-
D5.1.7a	Diesel	Euro 5	1.7	C	1500-2000	Jul-12	12754	-
D5.1.7b	Diesel	Euro 5	1.7	H	1500-2000	May-13	4176	-
D5.2.1	Diesel	Euro 5	2.1	E	1500-2000	Dec-13	4807	-
D5.2.2a	Diesel	Euro 5	2.2	H	2000-2500	Oct-13	3589	-
D5.2.2b	Diesel	Euro 5	2.2	H	2000-2500	Sep-12	17489	-
D5.2.3c	Diesel	Euro 5	2.3	C	1500-2000	Jul-13	7495	-
D5.2b	Diesel	Euro 5	2.0	D	1500-2000	May-13	21129	-
D5.2c	Diesel	Euro 5	2.0	D	2000-2500	Jul-13	1242	-
D5.2d	Diesel	Euro 5	2.0	D	2000-2500	Jan-14	6233	-
D5.2e	Diesel	Euro 5	2.0	H	1500-2000	Oct-13	3438	-
D5.2f	Diesel	Euro 5	2.0	C	1500-2000	Feb-13	7173	-
D5.2g	Diesel	Euro 5	2.0	H	1500-2000	Dec-11	2829	-
D5.3a	Diesel	Euro 5	3.0	D	2000-2500	Mar-13	7669	-
D5.3b	Diesel	Euro 5	3.0	H	3000-3500	Aug-12	2205	-
D6.1.4a	Diesel	Euro 6	1.4	B	1000-1500	Nov-14	3613	-
D6.1.4b	Diesel	Euro 6	1.4	B	1500-2000	Jun-14	2354	-
D6.1.5a	Diesel	Euro 6	1.5	C	1000-1500	Apr-15	2033	-

D6.1.5b	Diesel	Euro 6	1.5	B	2500-3000	May-15	2696	-
D6.1.6a	Diesel	Euro 6	1.6	I	1000-1500	Jun-14	3872	-
D6.1.6b	Diesel	Euro 6	1.6	D	1000-1500	Sep-14	875	-
D6.1.6c	Diesel	Euro 6	1.6	C	1000-1500	Jul-15	3803	-
D6.1.6d	Diesel	Euro 6	1.6	I	2500-3000	Oct-13	3505	-
D6.1.6e	Diesel	Euro 6	1.6	D	1000-1500	Aug-14	3264	-
D6.2.0a	Diesel	Euro 6	2.0	D	2000-2500	Jul-15	1704	-
D6.2.0b	Diesel	Euro 6	2.0	D	2000-2500	Sep-15	4027	-
D6.2.0c	Diesel	Euro 6	2.0	E	2000-2500	Apr-14	4131	-
D6.2.0d	Diesel	Euro 6	2.0	C	1500-2000	Jul-14	4133	-
D6.2.0e	Diesel	Euro 6	2.0	G	1500-2000	Sep-14	1199	-
D6.2.0f	Diesel	Euro 6	2.0	D	2000-2500	May-14	3368	-
D6.2.0g	Diesel	Euro 6	2.0	E	2000-2500	Jun-14	8481	-
D6.2.0h	Diesel	Euro 6	2.0	D	1500-2000	May-15	726	-
D6.2.0i	Diesel	Euro 6	2.0	H	2000-2500	May-15	2111	-
D6.2.0j	Diesel	Euro 6	2.0	D	1500-2000	Dec-13	3249	-
D6.2.0k	Diesel	Euro 6	2.0	C	1500-2000	Jul-14	1030	-
D6.2.0l	Diesel	Euro 6	2.0	C	1500-2000	Feb-15	4683	-
D6.2.0m	Diesel	Euro 6	2.0	D	1500-2000	Jul-14	4125	-
D6.2.0n	Diesel	Euro 6	2.0	D	1500-2000	Apr-13	6536	-
D6.2.0o	Diesel	Euro 6	2.0	C	1500-2000	Sep-14	2401	-
D6.2.0p	Diesel	Euro 6	2.0	D	1500-2000	Aug-14	1609	-
D6.2.0q	Diesel	Euro 6	2.0	C	1000-1500	Feb-15	1905	-
D6.2.0r	Diesel	Euro 6	2.0	I	1000-1500	Jan-14	6183	-
D6.2.0s	Diesel	Euro 6	2.0	E	1500-2000	Jul-15	2729	-
D6.2.0t	Diesel	Euro 6	2.0	I	1500-2000	Jul-15	6882	-
D6.2.0u	Diesel	Euro 6	2.0	C	1000-1500	Jul-15	6746	-
D6.2.0v	Diesel	Euro 6	2.0	C	1000-1500	Mar-14	1194	-
D6.2.0w	Diesel	Euro 6	2.0	E	1500-2000	Nov-14	7010	-
D6.2.2a	Diesel	Euro 6	2.2	D	1500-2000	Nov-12	9677	-
D6.2.2b	Diesel	Euro 6	2.2	D	1500-2000	Feb-15	950	-
D6.2.2c	Diesel	Euro 6	2.2	D	1500-2000	Dec-12	362	-
D6.2.2d	Diesel	Euro 6	2.2	D	1500-2000	Jan-13	1873	-
D6.2.2e	Diesel	Euro 6	2.2	H	1000-1500	Mar-15	855	-
D6.3.0a	Diesel	Euro 6	3.0	F	1500-2000	Oct-13	2242	-
D6.3.0b	Diesel	Euro 6	3.0	F	2500-3000	Apr-13	2995	-
P5.1.0a	Petrol	Euro 5	1.0	B	1500-2000	Nov-12	4030	PFI
P5.1.0c	Petrol	Euro 5	1.0	B	1500-2000	Nov-12	9608	GDI
P5.1.0d	Petrol	Euro 5	1.0	A	1000-1500	Jun-12	3640	GDI
P5.1.0e	Petrol	Euro 5	1.0	C	1000-1500	Dec-13	7052	GDI
P5.1.0f	Petrol	Euro 5	1.0	B	1000-1500	Sep-12	2939	GDI
P5.1.2a	Petrol	Euro 5	1.2	A	1000-1500	Jan-14	1885	GDI
P5.1.2b	Petrol	Euro 5	1.2	A	1000-1500	Jan-14	4130	GDI

P5.1.2c	Petrol	Euro 5	1.2	B	1000-1500	May-14	2371	PFI
P5.1.2d	Petrol	Euro 5	1.2	B	1500-2000	Jan-15	1040	GDI
P5.1.2e	Petrol	Euro 5	1.2	B	1000-1500	Jul-13	1688	PFI
P5.1.2f	Petrol	Euro 5	1.2	C	1500-2000	Jan-14	5042	GDI
P5.1.2g	Petrol	Euro 5	1.2	C	1500-2000	Jan-14	5258	PFI
P5.1.2h	Petrol	Euro 5	1.2	B	500-1000	Apr-14	803	GDI
P5.1.2i	Petrol	Euro 5	1.2	B	1000-1500	Jan-13	5415	GDI
P5.1.2j	Petrol	Euro 5	1.2	B	1000-1500	Oct-15	565	PFI
P5.1.3a	Petrol	Euro 5	1.3	B	1000-1500	Jul-14	7229	GDI
P5.1.3b	Petrol	Euro 5	1.3	B	1000-1500	Jul-14	504	PFI
P5.1.4a	Petrol	Euro 5	1.4	C	1000-1500	Jul-13	2628	PFI
P5.1.4b	Petrol	Euro 5	1.4	C	1500-2000	Jun-13	6404	PFI
P5.1.4c	Petrol	Euro 5	1.4	C	1000-1500	Jul-12	2383	GDI
P5.1.4d	Petrol	Euro 5	1.4	B	1500-2000	Jul-14	2673	PFI
P5.1.6a	Petrol	Euro 5	1.6	B	1500-2000	Mar-13	4606	PFI
P5.1.6b	Petrol	Euro 5	1.6	H	1500-2000	Nov-12	2400	PFI
P5.1.6c	Petrol	Euro 5	1.6	C	1500-2000	Jan-14	7408	GDI
P5.1.6d	Petrol	Euro 5	1.6	B	1500-2000	Jan-13	6396	GDI
P5.1.6e	Petrol	Euro 5	1.6	D	2000-2500	Jun-15	4154	PFI
P5.1.6f	Petrol	Euro 5	1.6	G	2000-2500	Apr-13	1905	PFI
P5.1.6g	Petrol	Euro 5	1.6	C	1500-2000	Oct-12	5995	PFI
P5.1.6h	Petrol	Euro 5	1.6	B	1000-1500	Mar-13	4437	GDI
P5.1.6i	Petrol	Euro 5	1.6	B	1500-2000	Mar-13	3000	PFI
P5.1.8	Petrol	Euro 5	1.8	C	1500-2000	Apr-14	7636	PFI
P5.2.0b	Petrol	Euro 5	2.0	C	1500-2000	May-12	5483	PFI
P5.2.0c	Petrol	Euro 5	2.0	C	1500-2000	Jun-14	6787	GDI
P5.2.0d	Petrol	Euro 5	2.0	C	1000-1500	Oct-13	6690	GDI
P5.2.0e	Petrol	Euro 5	2.0	C	1500-2000	Oct-13	11309	PFI
P5.2.5a	Petrol	Euro 5	2.5	C	1500-2000	May-14	4620	GDI
P5.2.5b	Petrol	Euro 5	2.5	G	1500-2000	Jul-12	3174	PFI
P6.1.0a	Petrol	Euro 6	1.0	C	1000-1500	Mar-12	2131	PFI
P6.1.0b	Petrol	Euro 6	1.0	B	1000-1500	Mar-16	1954	GDI
P6.1.0c	Petrol	Euro 6	1.0	A	500-1000	Apr-15	3658	PFI
P6.1.0d	Petrol	Euro 6	1.0	A	500-1000	May-15	2103	GDI
P6.1.2a	Petrol	Euro 6	1.2	C	1000-1500	Feb-16	2015	GDI
P6.1.2b	Petrol	Euro 6	1.2	B	1000-1500	Nov-14	2366	PFI
P6.1.2c	Petrol	Euro 6	1.2	B	1000-1500	May-15	1221	GDI
P6.1.2d	Petrol	Euro 6	1.2	B	1500-2000	Sep-15	2073	PFI
P6.1.2e	Petrol	Euro 6	1.2	B	1000-1500	Feb-15	7281	PFI
P6.1.2f	Petrol	Euro 6	1.2	B	1000-1500	Oct-14	1621	PFI
P6.1.4a	Petrol	Euro 6	1.4	B	1000-1500	Jan-15	7775	PFI
P6.1.4b	Petrol	Euro 6	1.4	B	1500-2000	Feb-15	1917	GDI
P6.1.4c	Petrol	Euro 6	1.4	C	1500-2000	May-14	8383	GDI

P6.1.4d	Petrol	Euro 6	1.4	C	1500-2000	Jun-15	3259	PFI
P6.1.4e	Petrol	Euro 6	1.4	C	1500-2000	Jun-14	2937	PFI
P6.1.4f	Petrol	Euro 6	1.4	B	1000-1500	Sep-15	3803	PFI
P6.1.4g	Petrol	Euro 6	1.4	I	2000-2500	Nov-15	1318	PFI
P6.1.4i	Petrol	Euro 6	1.4	C	1500-2000	Jul-05	-	PFI
P6.1.5a	Petrol	Euro 6	1.5	C	1000-1500	Sep-15	1790	PFI
P6.1.5b	Petrol	Euro 6	1.5	B	1500-2000	Mar-14	4307	PFI
P6.1.6a	Petrol	Euro 6	1.6	C	1500-2000	Jul-15	-	PFI
P6.1.6b	Petrol	Euro 6	1.6	C	1500-2000	Jul-15	-	PFI
P6.1.6c	Petrol	Euro 6	1.6	C	1000-1500	Mar-15	1978	GDI
P6.1.6d	Petrol	Euro 6	1.6	C	1500-2000	Jan-16	2100	GDI
P6.1.6e	Petrol	Euro 6	1.6	H	1500-2000	Feb-16	1746	GDI
P6.1.6f	Petrol	Euro 6	1.6	C	1000-1500	Jan-16	1936	PFI
P6.1.6g	Petrol	Euro 6	1.6	B	1000-1500	May-15	3621	PFI
P6.1.8a	Petrol	Euro 6	1.8	D	2000-2500	Jul-15	-	PFI
P6.2.0a	Petrol	Euro 6	2.0	C	1000-1500	Jun-13	1658	PFI
P6.2.0b	Petrol	Euro 6	2.0	B	1000-1500	Mar-14	6882	GDI
P6.2.0c	Petrol	Euro 6	2.0	C	1500-2000	Mar-14	3330	PFI
P6.2.0d	Petrol	Euro 6	2.0	C	1500-2000	Apr-14	2321	GDI
P6.2.0e	Petrol	Euro 6	2.0	C	1500-2000	Jul-15	-	PFI
P6.2.0f	Petrol	Euro 6	2.0	I	2000-2500	Dec-13	4484	PFI
P6.3.0a	Petrol	Euro 6	3.0	D	1000-1500	Jun-14	5110	PFI
H5.1.8	Hybrid	Euro 5	1.8	C	1500-2000	Nov-12	-	PFI
H6.1.8	Hybrid	Euro 6	1.8	C	1500-2000	Dec-16	-	PFI

6.2.1.1 Hybrid vehicles

This analysis includes a limited sample of 2 petrol- electric hybrid passenger cars. Due to the limited sample size a greater level of caution should be used when drawing conclusions from these results. It should also be noted that these results relate only to petrol- electric hybrids deploying kinetic energy recovery technology, and both vehicles sampled were made by the same manufacturer. No inferences can be made for other types of hybrids (e.g. diesel or plug-in) and results may only be indicative of this particular manufacturer.

6.2.1.2 Engine displacement

In Chapter 4 it was shown engine displacement was not a significant factor for NO_x emissions. However, engine size is known to correlate closely with CO_2. In this chapter vehicles have been divided into categories relating to their engine displacement; <1.4 ℓ = Extra Small [XS], 1.4 ℓ - ≤1.55 ℓ = Small [S], 1.55 ℓ - ≤2ℓ = Medium [M] and >2 ℓ = Large [L]. The number of vehicles in each engine displacement category is listed in **Table 6-2**.

Table 6-2. Engine displacement of test fleet

	<1.4 ℓ [XS]	**1.4ℓ - ≤1.55 ℓ [S]**	**1.55ℓ - ≤2 ℓ [M]**	**>2 ℓ [L]**
Petrol	27	14	28	3
Diesel	-	8	54	13

Following the general European trend, diesel engines in the test fleet were on average larger than petrol engines, an average of 1.9 ℓ for diesel compared to 1.5 ℓ for petrol. The distribution of engine displacements in the test fleet was representative of the UK as shown in **Table 6-3**.

Table 6-3. Comparison of size distribution of vehicles in study and UK fleet (DfT, 2015c)

	≤1 ℓ	1ℓ to ≤1.55 ℓ	>1.55ℓ to ≤ 2 ℓ	>2 ℓ
Petrol				
UK 2015 sales petrol cars (%)	12%	44%	39%	5%
Test fleet share petrol cars (%)	12%	45%	38%	5%
Diesel				
UK 2015 sales diesel cars (%)	0.1%	12%	65%	23%
Test fleet share diesel cars (%)	0	11%	72%	17%

6.2.1.3 Euro car segment

Table 6-4 describes the test fleet by European market segments. The most represented segments in the test fleet were B, C, D and H. These were also the most common in the EU passenger car market, in the UK in 2015 they made up 83% of new vehicles registered (SMMT, 2015).

Table 6-4. Comparison of segments of vehicles in study and EU 2015 sales

Segment		% new cars sold in EU 2015 (ACEA, 2016)	# in test fleet			
			D5	D6	P5	P6
A	Mini	31%	-	-	3	2
B	Supermini/ Small		4	3	16	12
C	Lower Medium	22%	14	9	14	17
D	Upper Medium	10%	6	14	1	2
E	Executive	3%	1	4	-	-
F	Luxury Saloon		-	2	-	-
G	Specialist Sport		-	1	3	-
H	Dual Purpose (SUV)	23%	8	2	2	1
I	Multi-Purpose Vehicle	11%	3	4	0	2

6.2.1.4 Mileage

The vehicles in the test fleet were all relatively new, with an average start mileage of 4105 (sd. 3000) km. As a result emission degradation (usually observed > 50,000 km (Borken-Kleefeld & Chen, 2015)) is not considered in this analysis. It is still too soon for there to be substantial evidence relating to emission degradation from Euro 5 and 6 cars, though it is a fair assumption that emissions stated here will not remain constant over the lifetime of the vehicles (Chen & Borken-Kleefeld, 2016).

6.2.1.5 After treatment technologies

All petrol vehicles were fitted with a three way catalytic converter (TWC). TWCs effectively control emissions of NO_x, CO and total hydrocarbons. Like all diesel vehicles Euro 5 and above the D5 cars were fitted with a Diesel Oxidation Catalyst

(DOC), Diesel Particulate Filter (DPF), and Exhaust Gas Recirculation (EGR). As discussed in Chapter 4 the majority of Euro 6 diesels were also fitted with additional NO_x abatement technology (either LNT or SCR).

In terms of CO_2 abatement, half of the test fleet was fitted with fuel saving stop- start technology. This is in line with the 60% of European new passenger cars that now use stop- start technology (Gross, 2015). Previous studies found stop- start delivers fuel savings of between 3 – 5 % (Bishop *et al.*, 2007), meaning any CO_2 benefit from stop-start fell within the natural variability of the PEMS testing and were not detectable in this study.

6.2.2 Ambient temperature

For reasons previously stated in Chapter 4, the results were not corrected for ambient temperature and pressure. However, as this analysis made comparisons between the different vehicle categories, it was important to ensure a level of consistency in local ambient temperatures between categories to avoid bias. **Table 6-5** shows that the mean ambient temperature of tests for each category fell within the same range.

Table 6-5. Average temperature by category

	D5	**D6**	**P5**	**P6**
Average temp. [°C]	15.0 (sd. 5.5)	17.1 (sd. 6.2)	15.1 (sd. 5.9)	14.0 (sd. 5.4)

6.2.3 Test sections

Table 6-6. Motorway and urban cycle characteristics

	Duration [s]	**Distance** [km]	**Average speed** [km h^{-1}]	**% idle***
Urban	2368 (sd. 105)	16.1 (sd. 0.1)	24.5 (sd. 1.1)	17.9 (sd. 3.1)
Motorway	580 (sd. 10)	16.1 (sd. 0.1)	99.8 (sd. 1.5)	0.02 (sd. 0.1)

*Vehicle speed < 0.5 ms^{-1}, acceleration between ± 0.1 ms^{-2}

The average characteristics of the motorway and urban sections in this analysis are presented in **Table 6-6**. As discussed, the urban and motorway sections were selected following Regulation (EU) 2016/646. This was done by creating a moving 16 km window (Regulation (EU) 2016/646 stipulates 16 km is the minimum allowed trip distance) and evaluating the characteristics of each window to assure the dynamic boundary conditions were met. This was done using purpose built software written in the statistical package R.

This analysis differs from Regulation (EU) 2016/646 guidelines in that urban and motorway sections were not identified by binning for speed (urban < 60 km h^{-1}, motorway > 90 km h^{-1}). Regulation (EU) 2016/646 also gives a wide range for the average speed of the urban section (15 – 40 km h^{-1}), however to ensure comparability between the vehicles in this analysis all urban sections had an average speed of ~25 km h^{-1}. Regulation (EU) 2016/646 states there must be over 150 data points with acceleration higher than 0.1 ms^{-2}, all sections in this analysis had many more than 150. These are examples of rules in Regulation (EU) 2016/646 that allow for different driving characteristics that will impact on the average emissions. This is discussed further in the Discussion section of this chapter.

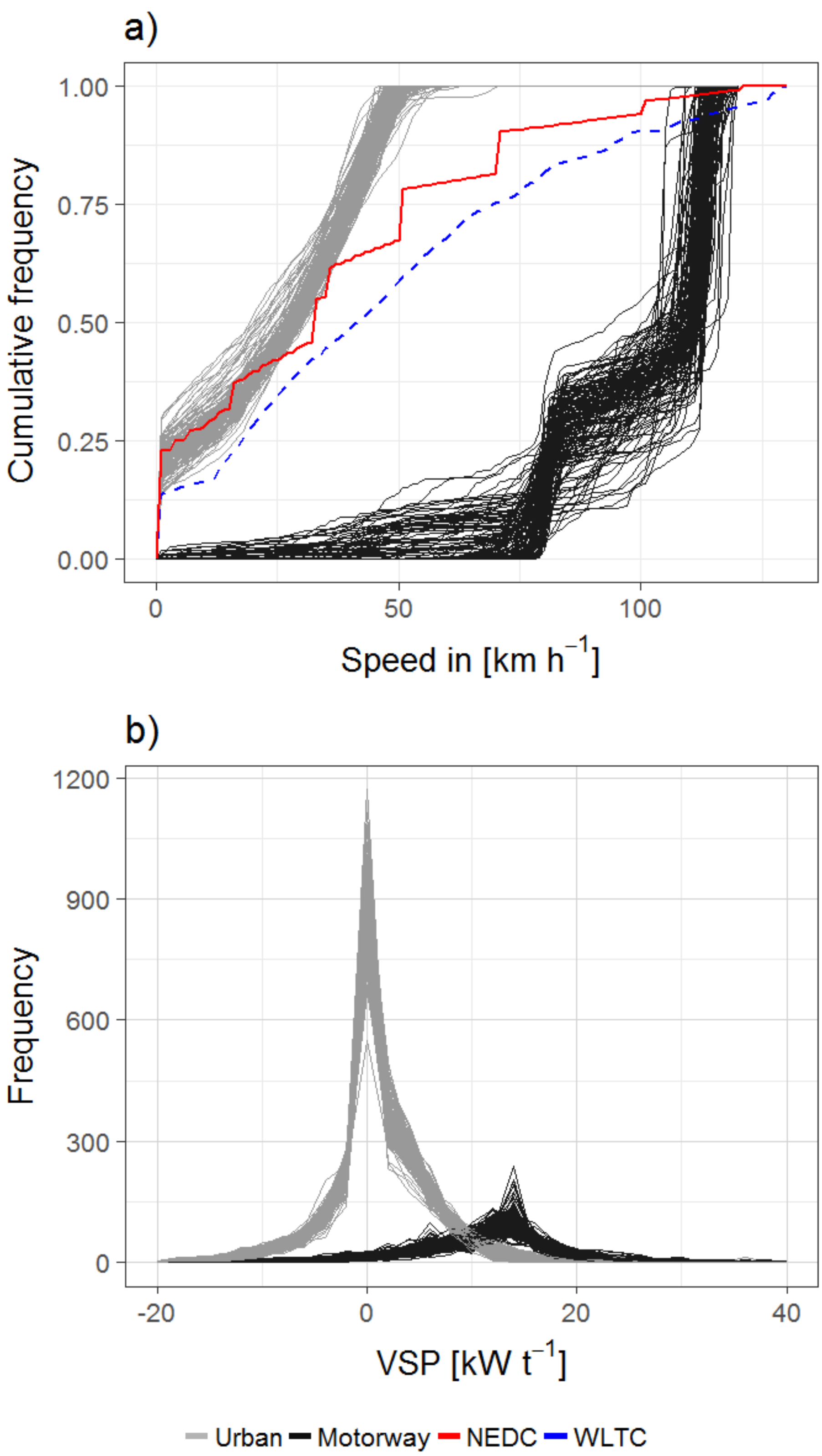

Figure 6-2. a) Cumulative frequency of speed b) Frequency diagram of VSP

The analysis here differs from Regulation (EU) 2016/646 guidelines in that urban and motorway sections were not identified by binning for speed. However, guidelines relating to minimum, maximum and speed ranges were followed. As with the previous analysis in Chapter 4, urban sections were selected from A, B and C roads with a speed limit < 50 km h^{-1}, motorway sections were selected from M roads with a speed limit < 110 km h^{-1}.

Figure 6-2 shows the speed and Vehicle Specific Power (VSP, defined in Chapter 4) distribution of the urban and motorway sections. The data fell into distinct groups for both parameters, indicating a high level of consistency between the tests. **Figure 6-2a)** compares the speed distributions of the urban and motorway sections selected for this analysis to the NEDC and WLTC. Urban sections covered the range 0 - 50 km h^{-1}, motorway sections mainly covered the range > 75 km h^{-1}. **Figure 6-2b)** shows the frequency distribution of VSP for urban and motorway sections. As with the analysis in Chapter 4, urban sections were characterised by lower VSP and motorway sections higher. The difference in the height of the peaks reflects the duration spent in each section; the average duration of an urban section was 4 times the average duration of a motorway section.

$v.a_{pos}$_[95] is defined fully in section 6.2.5 Data analysis (below). It is the metric that determines the maximum dynamic boundary condition of a test trip in (EU) 2016/646. $v.a_{pos}$_[95] refers to the 95th percentile of speed*positive acceleration. For a test section to be valid it must have a value of $v.a_{pos}$_[95] below a certain value. This value is proportional to the average speed of the test section, and the upper limits for each average speed are marked in **Figure 6-3a)**. There are two limits for $v.a_{pos}$_[95], one for low speeds and one for high. All the test data points fall below the relevant dashed lines indicating that all tests met the $v.a_{pos}$_[95] maximum dynamic boundary condition.

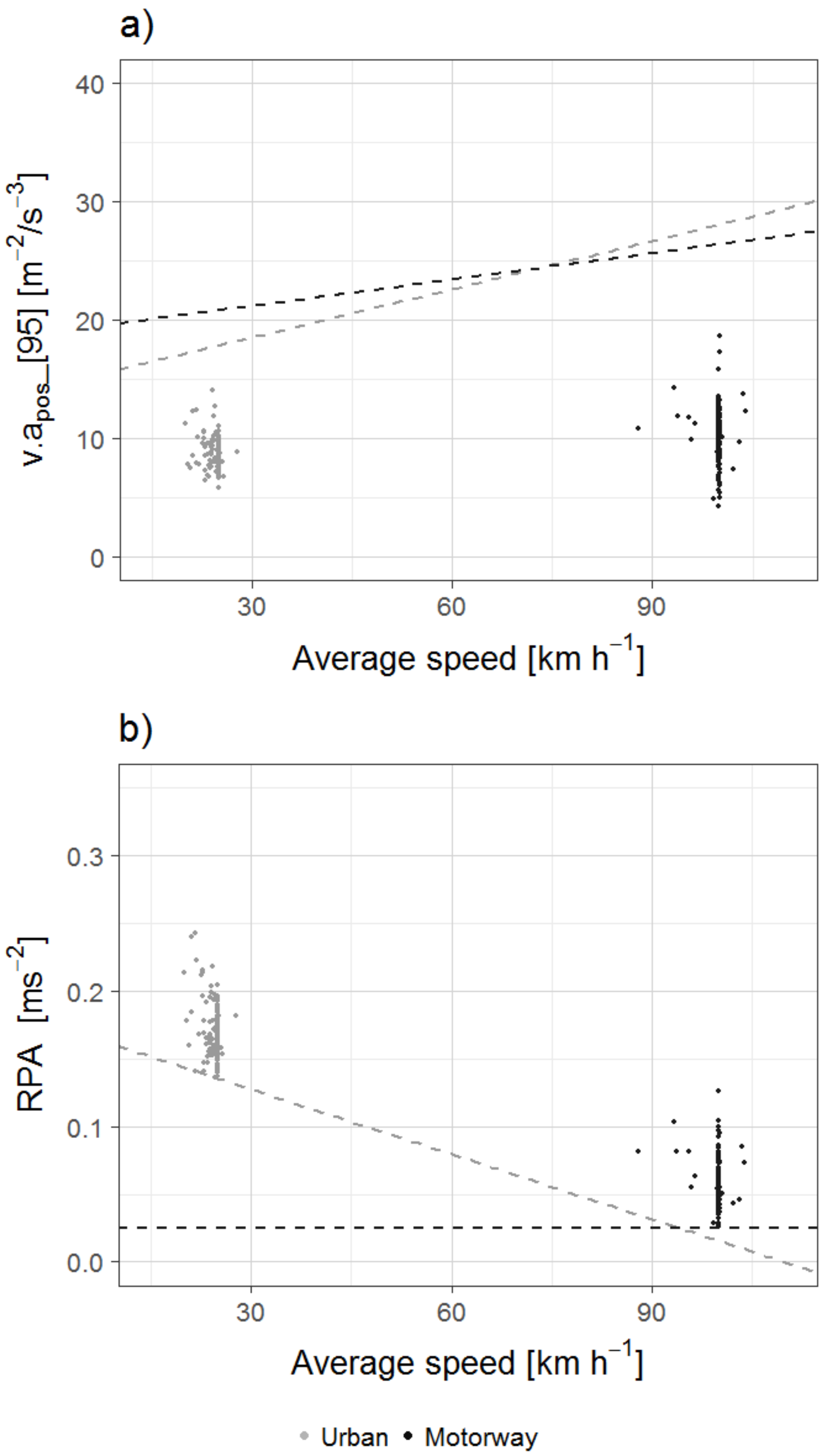

Figure 6-3. Dynamic boundary conditions from (EU) 2016/646

The metric for assessing the minimum dynamic boundary condition in (EU) 2016/646 is relative positive acceleration (RPA). The definition of RPA used by (EU) 2016/646 differs from the definition of Weiss, Bonnel, Hummel, *et al.*, (2011) (used in Chapter 4). Weiss et. al broke each trip down into sub-trips whereas (EU) 2016/646 calculates the RPA of an entire test section (this is described further in the Data analysis section below). As with $v.a_{pos}$_[95] the minimum RPA boundary is proportional to average speed (marked on **Figure 6-3b)** by dashed lines). For a section to be valid, the RPA must be above a certain value. All section data points fell above the relevant dashed lines, showing that they met the minimum dynamic boundary condition.

6.2.4 Cold starts

The urban and motorway sections selected for this analysis were part way through a test and therefore did not include a cold start. Furthermore, the majority of test trips were from warm start (as opposed to cold start) and there was no uniform engine rest period before each test trip. Because of this, cold start analysis does not form a core part of this research. However, using the exhaust temperature as a proxy, several cold starts were identified. Analysis of two identified cold starts will be presented in the results section of this chapter, one petrol and one diesel.

6.2.5 Data analysis

Data analysis such as emissions factor calculations, VSP and acceleration were calculated as in Chapter 4. Additional parameters of $v.a_{pos}$_[95] and an alternative RPA are detailed below.

6.2.6 v.a_{pos}_[95]

The value v.a_{pos}_[95] must be below a certain value for the trip to be valid in the RDE test procedure. It was calculated as follows:

First the product of vehicle speed per acceleration was calculated using **Equation 6-1**:

Equation 6-1. Vehicle speed per acceleration

$$(v * a)_i = \frac{v_i * a_i}{3.6} \qquad i = 1 \; to \; N_t$$

a_i = instantaneous acceleration in [m s^{-2}]

v_i = instantaneous velocity [km h^{-1}]

N_t = number of samples

To calculate v.a_{pos}_[95] values of $(v*a)_{i,k}$ with $a_{i,k} \geq 0.1$ m s^{-2} are sorted in ascending order and assigned a rank 1 to k, with 1 being the lowest value and k assigned to the highest. The highest value $(v*a)_{i,k}$ is denoted as M_k .

Percentile values were assigned to $(v \cdot a_{pos})_{j,k}$ values with $a_{i,k} \geq 0.1$ m s^{-2}. The lowest v*a_{pos} value was assigned the percentile 1/M_k , the second lowest 2/M_k etcetera, the highest value (M_k/M_k) represented 100 %. The value of $(v*a_{pos})_k$ _[95] was the $(v*a_{pos})_{j,k}$ value, with j/M_k equal to 95 %.

6.2.6.1 Validity of v*a_{pos}_[95]

$\overline{v_k}$ = average speed of entire section.

For $\overline{v_k}$ ≤ 76.6 km h^{-1} a trip was not valid if $(v \cdot a_{pos})_k$ _[95] > (0.136*$\overline{v_k}$ + 14.44)

For $\overline{v_k}$ > 76.6 km h^{-1} a trip was not valid if $(v \cdot a_{pos})_k$ _[95] > (0.0742*$\overline{v_k}$ + 18.966)

6.2.6.2 Relative Positive Acceleration

The maximum metric of the dynamic boundary conditions in (EU) 2016/646 is Relative Positive Acceleration. The value RPA must be above a certain value for the trip to be valid in the RDE test procedure. RPA was calculated using **Equation 6-2**.

Equation 6-2. RPA as defined by (EU) 2016/646

$$RPA = \frac{\sum_j \Delta t * (v.a_{pos})_{jk}}{\sum d_{i,k}} \qquad j = 1 \text{ to } M_k,\ i = 1 \text{ to } N_k,$$

Δt = time difference (1 second)

M_k = samples in test section (i.e. urban / motorway) with $a_{i,k} \geq 0.1$ m s^{-2}

N_k = samples in entire test section (i.e. urban / motorway)

6.2.6.3 Validity of RPA

For $\overline{v_k}$ ≤ 94.05 km h^{-1} a trip was not valid if $RPA_k < (-0.0016 * \overline{v_k} + 0.1755)$

For $\overline{v_k}$ > 94.05 km h^{-1} a trip was not valid if $RPA_k < 0.025$

6.3 Results

This section presents analysis for individual pollutants (CO_2, NO_x, NO_2 and CO) followed by cold start emissions, an evaluation of the (EU) 2016/646 selection method and a discussion.

6.3.1 CO_2 emissions

Figure 6-4 compares average CO_2 emissions by category (engine size) from petrol, diesel and hybrid vehicles for urban and motorway sections. The increase in CO_2 from diesel to petrol vehicles of the same engine size ranged between 13 – 66%. ANOVA statistical analysis was performed using the software package R, no statistically significant difference was found in CO_2 emissions from Euro 5 and 6 technology vehicles for either petrol or diesel. The sample size for hybrids was too small for ANOVA analysis, but there was little difference between the H5 and H6 vehicles. CO_2 varied significantly with engine size and urban emissions were much higher than motorway for diesel and petrol vehicles, the reverse was true for hybrids.

The average CO_2 emission for each category by fuel type for the motorway and urban sections is listed in **Table 6-7.** The UK weighted average was calculated by weighting the vehicles in the study relative to the distribution of petrol and diesel engines in the UK using the 2015 new car sales data presented in **Table 6-3**.

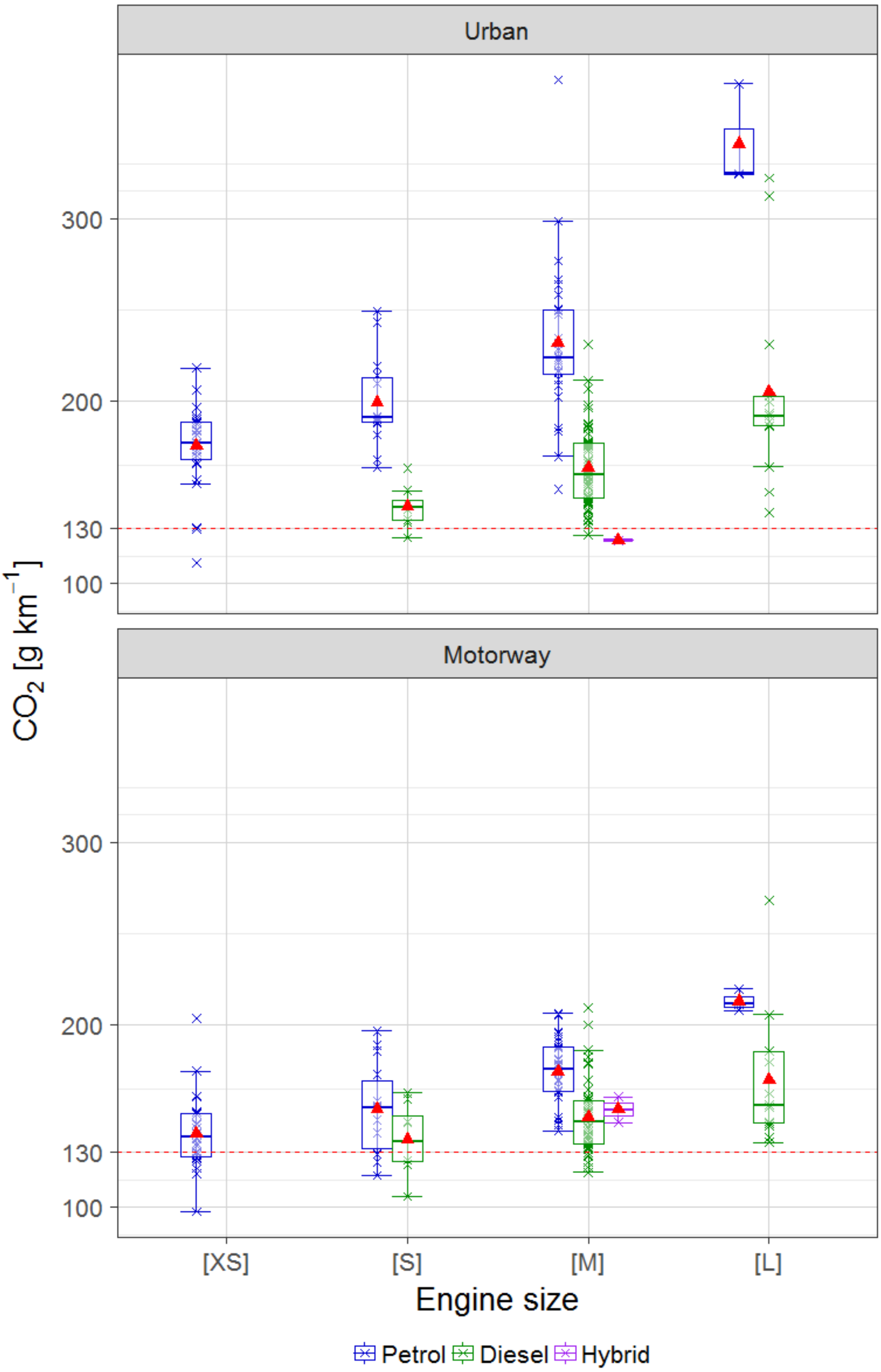

Figure 6-4. Urban and motorway average CO_2 emissions by engine size, red dashed line = 2015 fleet average target

Table 6-7. Average CO_2 emission by engine displacement, section and fuel type [g km^{-1}]

Urban	[XS]	[S]	[M]	[L]	UK weighted average
Petrol	175.2 (sd. 23.3)	199.2 (sd. 25.2)	231.5 (sd.42.3)	340.9 (sd. 28.6)	210.5 (sd. 47)
Diesel	-	141.9 (sd.11.6)	163.4 (sd. 21.6)	205.1 (sd. 55.1)	170.2 (sd. 34)
Hybrid	-	-	117.4 (sd. 12.4)	-	-
Motorway	**[XS]**	**[S]**	**[M]**	**[L]**	**UK weighted average**
Petrol	140.6 (sd. 20.3)	154.3 (sd. 24.9)	174.4 (sd. 17.9)	213.0 (sd.6.0)	160.2 (sd. 29)
Diesel	-	137.1 (sd. 19.8)	149.0 (sd. 18.9)	170.0 (sd. 36.4)	152.3 (sd. 22)
Hybrid	-	-	150.9 (sd. 36.3)	-	-

The petrol- electric hybrids were by far the best group, particularly during urban driving. Hybrids were the only group to have an average (117.4 (sd. 12.4) g km^{-1}) below the 2015 CO_2 fleet average target of 130 g km^{-1}. This equated to a 49% reduction compared to conventional petrol vehicles of the same size. For the motorway section the average CO_2 (150.9 (sd. 36.3) g km^{-1}) was 13% below the conventional petrol average for that category and on a par with diesel. Hybrids were the only group for which motorway emissions were higher than urban. These results show that whilst the hybrid technology performed well in both sections it is most effective in urban driving. This is because the vehicles tested deployed kinetic energy recovery technology, which is utilised most effectively when there are regular acceleration and deceleration events. Further work should include analysis of other types of hybrid.

Table 6-8 lists the % increase in average CO_2 from diesel to petrol by category. The increase was between 12.5 – 66.2%, stated another way this equated to a reduction from petrol to diesel of between 11.1 – 39.8%.

Table 6-8. Increase [%] average CO_2 from diesel to petrol

	[S]	**[M]**	**[L]**	**UK weighted average**
Urban	40.4%	41.7%	66.2%	23.7%
Motorway	12.5%	17.0%	25.3%	5.2%

Engine size is a key factor when considering the replacement of diesel vehicles with petrol. CO_2 emissions increased significantly with engine size and the increase was greatest for petrol vehicles. For example [L] vehicles' urban average CO_2 was 71.1 % higher than [S] for petrol cars and only 44.5 % higher for diesel. This indicates there are potential CO_2 savings to be made by downsizing petrol engines. However, as shown in **Table 6-3** the majority of petrol engines are already small, with [L] engines accounting for only 5% of the UK fleet.

As consumers move away from diesel and towards petrol, a determining factor in the CO_2 penalty incurred will be the size of the new petrol engines. **Table 6-7** shows that an [M] diesel, the most common size in the UK, replaced by an [M] petrol would result in a 42% increase in urban CO_2. The same [M] diesel replaced with an [S] petrol would result in a 22% increase. The lowest CO_2 penalty (7%) would be incurred by substituting the [M] diesel with an [XS] petrol.

2.1.1.1 CO_2 fleet average target

Figure 6-5 shows CO_2 average emissions by vehicle weight for urban and motorway sections. The diagonal red dashed line is the weight dependent 2015 limit curve, the horizontal red dashed line is the fleet average target of 130 g CO_2 km^{-1}.

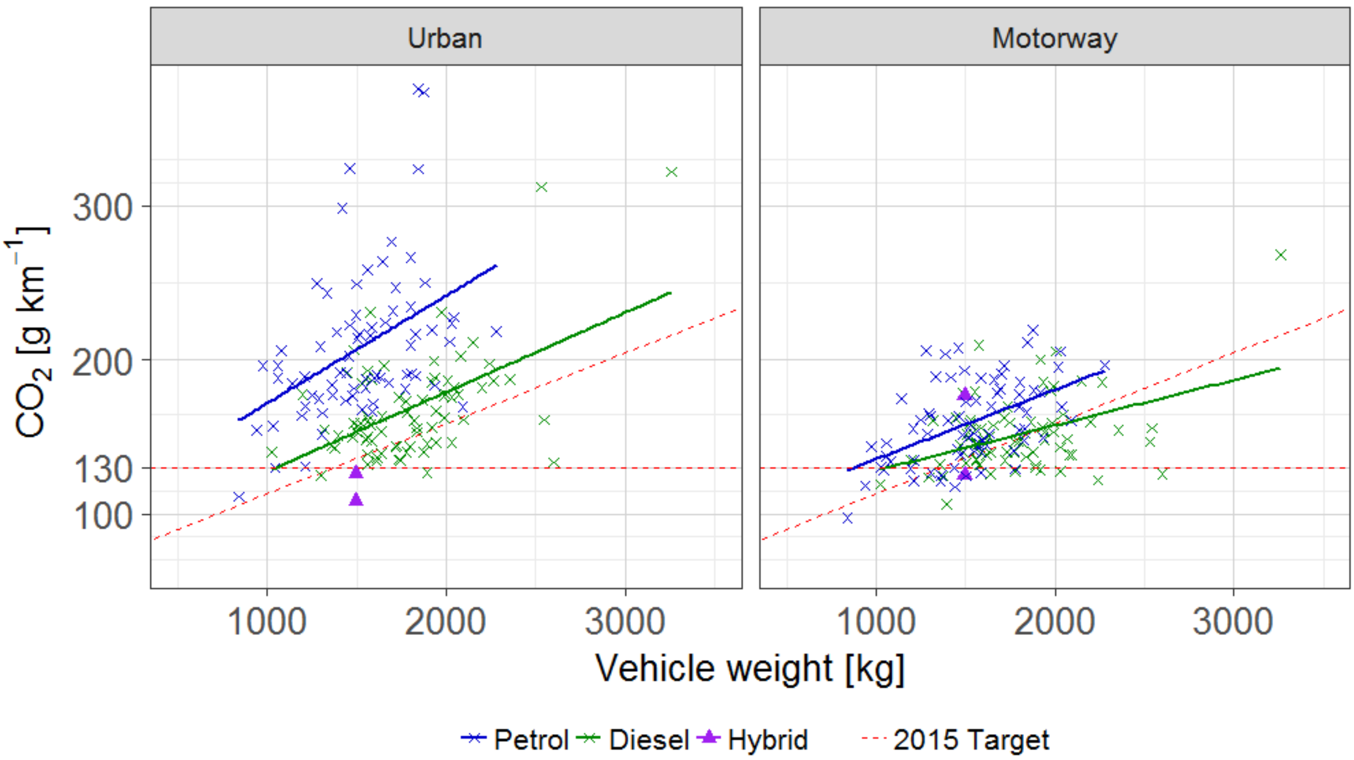

Figure 6-5. CO_2 by vehicle weight and comparison to 2015 fleet average target

Figure 6-5 shows that for both urban and motorway driving a number of diesel vehicles had emissions below the weight dependent limit curve. For urban driving, 6 vehicles (2 diesel, 2 petrol, 2 hybrid) met the fleet average target of 130 g CO_2 km^{-1} and the average exceedance was +31% for diesel and +62% for gasoline. This increased to 25 vehicles (10 diesel, 13 gasoline, 2 hybrid) for the motorway section, and the average exceedance fell to +16% for diesel and +23% for petrol.

2.1.1.2 Comparison with manufacturers' stated emissions

The PEMS average CO_2 measurements were also compared to the manufacturers' official estimates, recorded over the NEDC. **Table 6-9** lists the average percentage by which real driving emissions exceeded the manufacturers' official estimates.

Table 6-9. Percentage by which PEMS measurements exceeded manufacturers' official estimates

	Urban	Motorway	Average
D5	35.1 %	29.9 %	32.5 %
D6	46.7 %	25.1 %	35.9 %
P5	60.0 %	20.0 %	40.0 %
P6	61.1 %	26.7 %	43.9 %
H5	13.1 %	83.9 %	48.5 %
H6	80.2 %	78.9 %	79.6 %

On average hybrid vehicles exceeded the manufacturer's estimates by the largest percentage, though they also had the lowest CO_2 emissions. In general petrol vehicles exceeded the manufacturers' official estimates by a higher percentage than diesel. The results stated in **Table 6-9** are in good agreement with previous studies, which found RDE of CO_2 exceeded manufacturers' estimates by ~40% (Fontaras & Samaras, 2010; Fontaras *et al.*, 2014; T & E, 2015). These results were also in agreement that the discrepancy between manufacturers' estimates and real world emissions is increasing. For diesel, petrol and hybrid vehicles on the motorway and urban roads the increase was greater for Euro 6 vehicles than for Euro 5. This is because manufacturers have reported a reduction in CO_2 that was not evident in the PEMS measurements.

2.1.1.3 CO_2 urban vs. motorway emissions

As discussed, for petrol and diesel vehicles average CO_2 emissions were higher during urban driving, with the reverse being true for hybrids. The UK weighted average urban CO_2 emission for petrol vehicles was 31.4% higher than the motorway sections, for diesel the increase was only 11.8%.

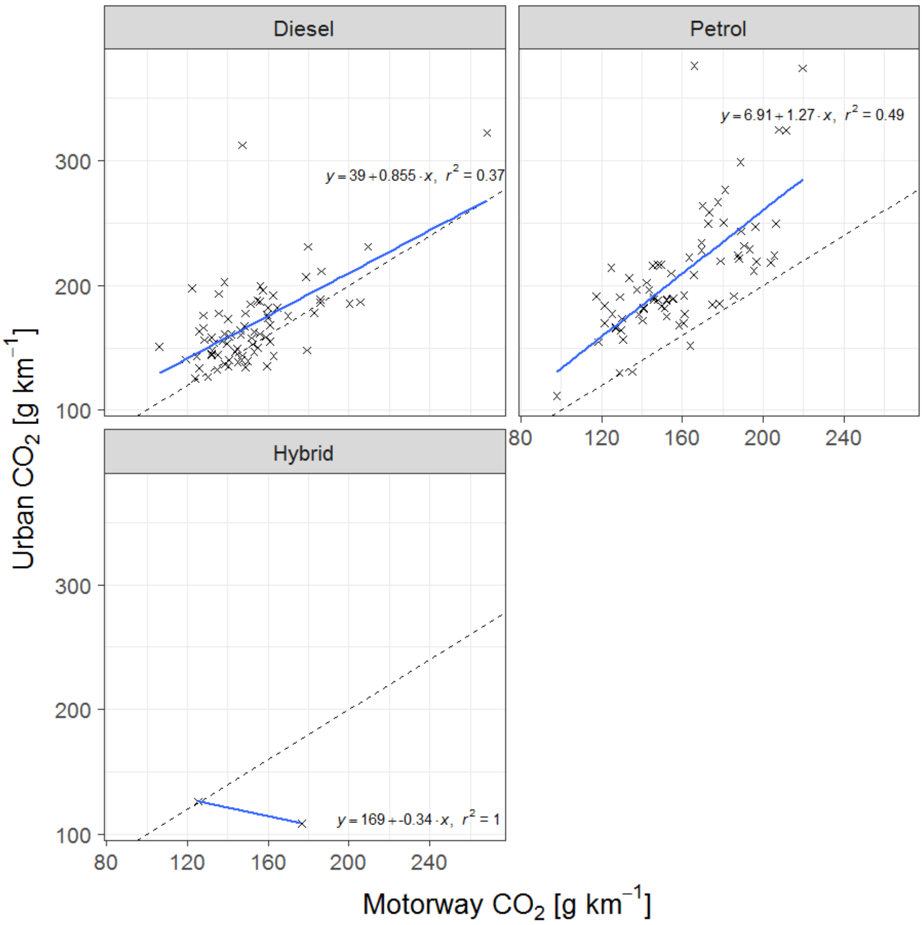

Figure 6-6. Motorway vs. Urban CO_2 by fuel

The results stated previously related to the average increase in CO_2 by fuel type or category. When considering the increase for individual vehicles between motorway sections and their urban counterparts the results are very similar, an average increase of 31.1% for petrol and 12.0 % for diesel.

Previous studies have found the increase in CO_2 during urban driving is due to driving behaviour at these speeds (i.e. increase in stop/ start due to traffic lights and congestion) and not something inherent to emissions at low speeds (Barth & Boriboonsomsin, 2008; Daham *et al.*, 2005).

2.1.1.4 CO_2 from GDI engines

The average urban and motorway CO_2 emissions from petrol GDI and PFI are listed in **Table 6-10** along with the % reduction in CO_2 delivered by GDI.

Table 6-10. CO_2 in g km^{-1} for GDI, diesel and PFI

URBAN	**[XS]**	**[S]**	**[M]**
GDI	164.3	198.6	233.4
PFI	187.0	199.5	230.3
% reduction	*12.1%*	*0.5%*	*-1.3%*
MOTORWAY	**[XS]**	**[S]**	**[M]**
GDI	133.9	151.4	177.9
PFI	147.8	156.0	172.1
% reduction	*9.4%*	*2.9%*	*-3.4%*

Half the GDI engines in the test fleet were [XS] and this was the only category for which there was a significant improvement from PFI. The urban [XS] average CO_2

emission was 12.1% less than for PFI, though still higher than the [S] and [M] diesel averages as seen in **Figure 6-7**.

Whilst the results for [XS] GDIs were promising, they still emitted more CO_2 than both the diesel and hybrid vehicles tested in this study. There are also questions surrounding GDI not addressed in this analysis relating to particulate emissions, specifically ultrafine particles and cold starts.

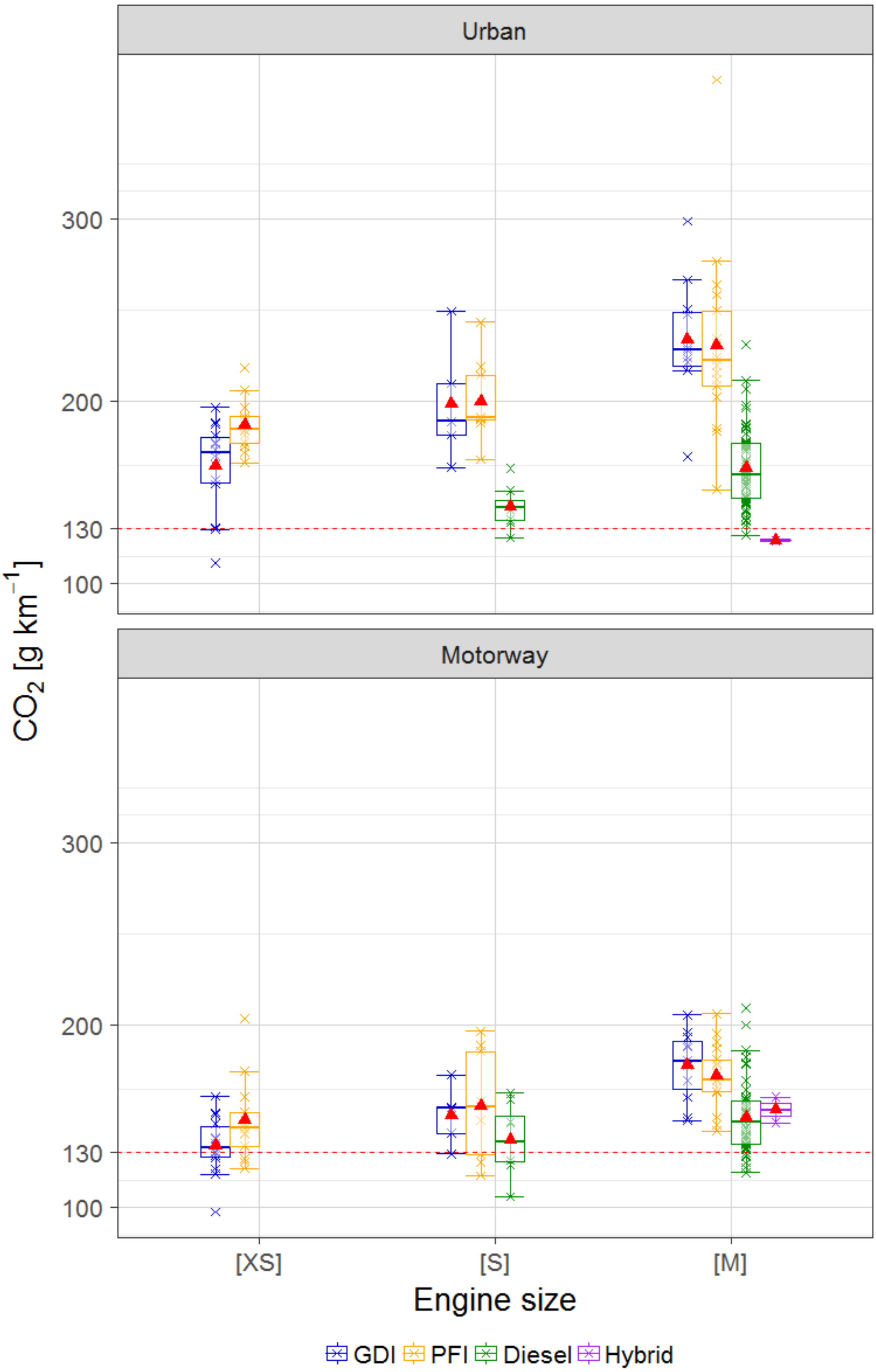

Figure 6-7. Urban and motorway average CO_2 emissions for GDI, PFI, diesel and hybrid

6.3.2 NO_x emissions

Compared with CO_2 emissions there was greater variation within vehicle categories for NO_x and a greater divergence between petrol and diesel. The highest recorded CO_2 emission in the study was 4 times the lowest, whereas the highest NO_x emission was over 4000 times the lowest. Unlike CO_2 there was a significant improvement in NO_x from Euro 5 to Euro 6 for both petrol and diesel vehicles. This is a potential indication that carbon intensity improvements moving from Euro 5 to 6 were partly cancelled out by additional NO_x abatement strategies (which often incur a carbon penalty).

Figure 6-8 shows the average urban and motorway NO_x emissions from the different vehicle categories. The red dashed horizontal lines represent the type approval limits; 0.18 g km^{-1} for Euro 5 diesel, 0.08 g km^{-1} for Euro 6 diesel, 0.06 g km^{-1} for Euro 5 and 6 petrol. As the hybrids in this study were petrol- electric, the relevant type approval limit for comparison is the petrol limit. Red triangles mark the mean of each category. **Figure 6-8** illustrates that, as with CO_2, hybrid vehicles were the best performing group, with every measurement far below the type approval limit. The NO_x emissions measured from hybrid vehicles in this study were very low, within the error range of the PEMS system used. Whilst this may impact the numerical accuracy, the key finding to be drawn from these results is that NO_x from these hybrids was extremely low, far lower than any other group.

With the exception of P5 urban sections the majority of petrol vehicles met their type approval limit in the real world. In contrast the majority of diesels failed to meet the even more lenient Euro 5 diesel limit.

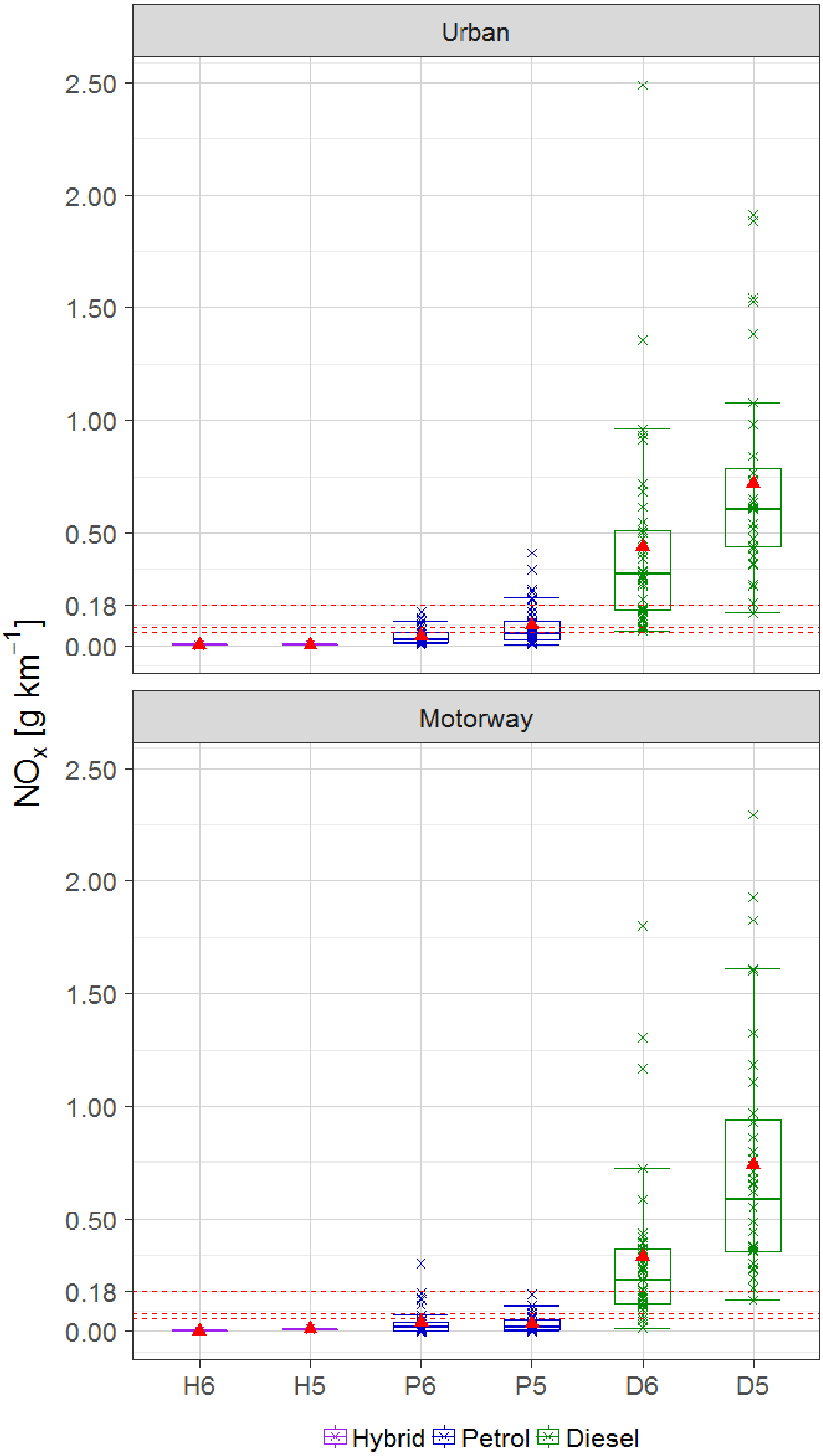

Figure 6-8. Urban and motorway NO_x emissions

Table 6-11 lists the mean NO_x emission by vehicle category. Also stated is the reduction in average NO_x from diesel to petrol vehicles of the same Euro standard. As with CO_2, the average NO_x emission for D6 and P5 was higher for urban sections than for motorway and the reverse was true for hybrids. However, for P6 the average urban and motorway emissions were the same and for D5 motorway emissions were slightly higher.

Table 6-11. Mean NO_x emission by sections and category in [g km^{-1}]

	NO_x [g km^{-1}]						Reduction diesel to petrol	
	H6	**H5**	**P6**	**P5**	**D6**	**D5**	**Euro 6**	**Euro 5**
Urban	0.002	0.003	0.04 (sd. 0.04)	0.09 (sd. 0.1)	0.44 (sd. 0.44)	0.72 (sd. 0.45)	91%	86%
Motorway	0.003	0.010	0.04 (sd. 0.06)	0.03 (sd. 0.04)	0.33 (sd. 0.36)	0.74 (sd. 0.54)	88%	96%
	Deviation ratio							
Urban	0.03	0.05	0.7	1.5	5.5	4		
Motorway	0.05	0.17	0.7	0.5	4.1	4.1		

In agreement with previous studies it was found that the deviation ratio increased from D5 to D6, as has been the case with successive Euro standards (Carslaw *et al.*, 2011a; Franco *et al.*, 2014). In contrast, the urban section petrol deviation ratio halved from P5 to P6. As discussed in the previous section, the gap between real world and manufacturers' estimates has also been expanding for CO_2. However, the ratio between real world diesel and lab measurements for CO_2 (~35%) is an order of magnitude smaller than for NO_x (~400%).

Table 6-11 also lists the reduction in average NO_x emissions from diesel to petrol: the reduction ranges between 86 – 96%. This can also be stated as the increase in NO_x between petrol and diesel, which was between 11 – 25 times.

P5 was the only petrol category with an average NO_x emission above the petrol type approval limit. The urban P5 sections had an average NO_x emission of 0.09 (sd. 0.1) g km^{-1}, 1.5 times the petrol limit. 16% of the P5 vehicles in the test fleet (6 out of 37) had urban emission above the Euro 5 diesel type approval limit.

6.3.2.1 NO_x emissions by temperature

The relationship between urban NO_x emissions and local ambient temperature is plotted in **Figure 6-9**, a similar result was found for motorway sections. There was a trend of increased NO_x at lower temperatures for every category, however the trend was much more profound for Euro 5 vehicles (D5 and P5). This is probably because in recent years, due to pressure from the European Commission, the temperature ranges of "thermal window" engine protection functions have been extended. For example, Renault have extended their thermal window from a narrow 17 -35 °C to a much broader 5 – 40 °C (T & E, 2016). This is the most likely explanation for the decrease in correlation between temperature and NO_x from Euro 5 to Euro 6. A recent remote sensing study in Sweden also found that the temperature dependence of NO_x emissions had declined with Euro standard (Sjödin *et al.*, 2017).

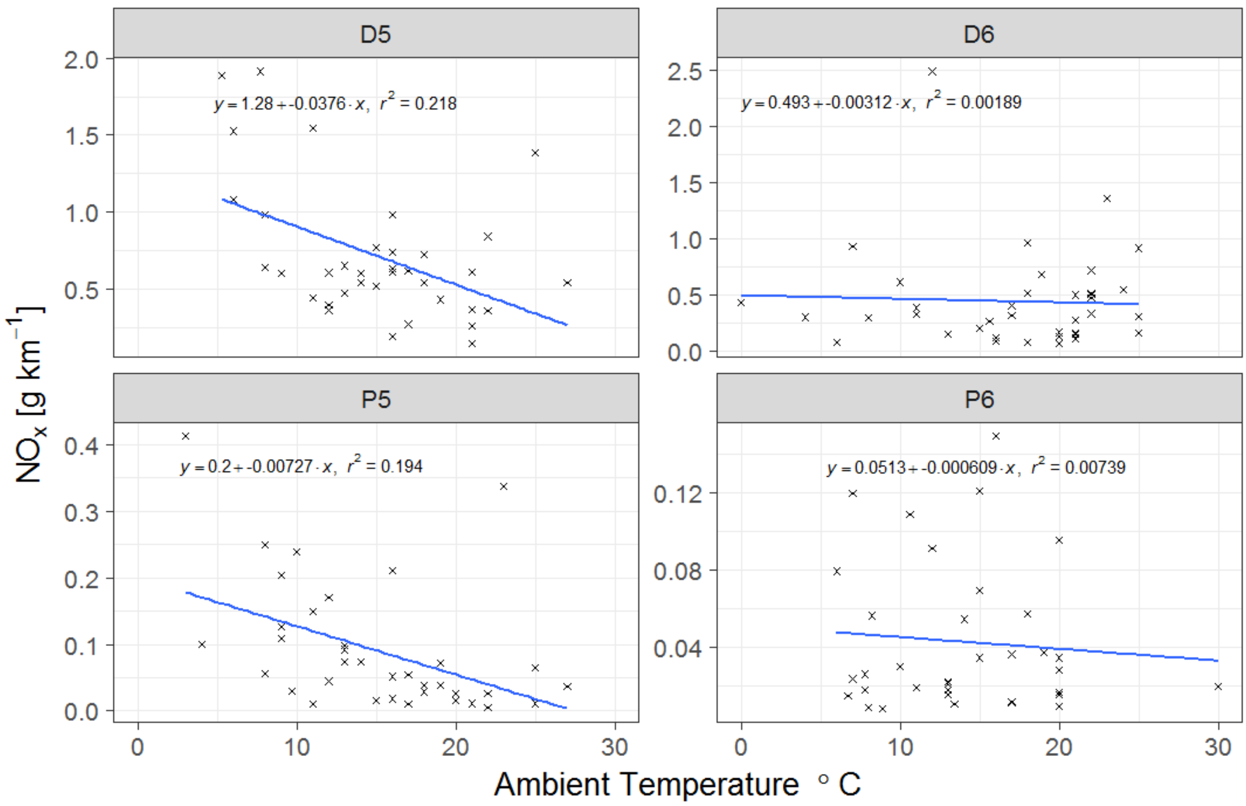

Figure 6-9. Urban section NO_x emissions by temperature

6.3.2.2 NO_x urban vs. motorway emissions

As discussed previously the urban section emissions were higher than motorway for P5 and D6, equal for P6, and lower for D5 and hybrids. **Figure 6-10** shows the ratio between motorway and urban section emissions for individual vehicles. The dashed vertical line represents a ratio of 1 (i.e. urban section average = motorway section average). Data points above the line are vehicles with higher urban section emissions, data points below the line correspond to vehicles with higher motorway section emissions.

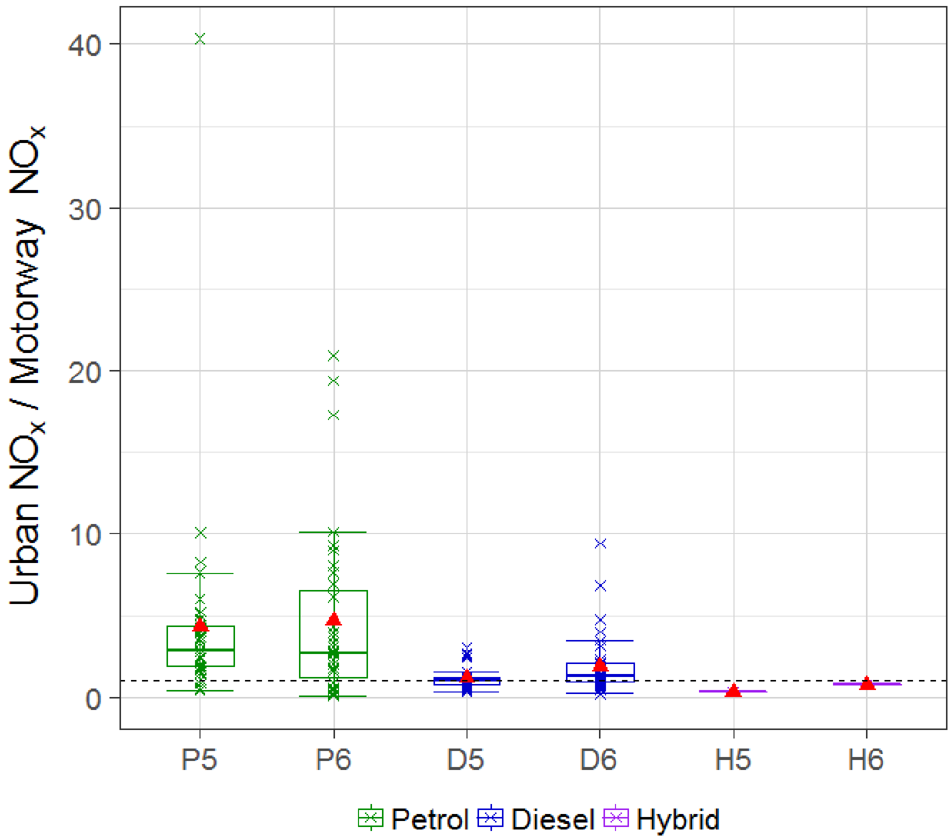

Figure 6-10. Individual vehicle urban / motorway section NO_x emissions by category

When comparing vehicles' urban sections to their own motorway counterparts, the average ratio between urban and motorway was; 4.3, 4.6, 1.2, 1.9, 0.3 and 0.7 for P5, P6, D5, D6, H5, and H6 respectively. Only hybrid vehicles had lower NO_x emissions during urban driving. The emissions increase in urban driving for both petrol and diesel vehicles was much greater for individual cars than for the category averages. This indicates that the vehicles with the highest urban/ motorway ratios had low NO_x emissions in g km^{-1}, meaning that even an increase from the motorway to urban section of up 10 times had little effect on the category averages. As with CO_2 the increase was greater for petrol vehicles than for diesel.

6.3.2.3 NO_x emissions by engine displacement

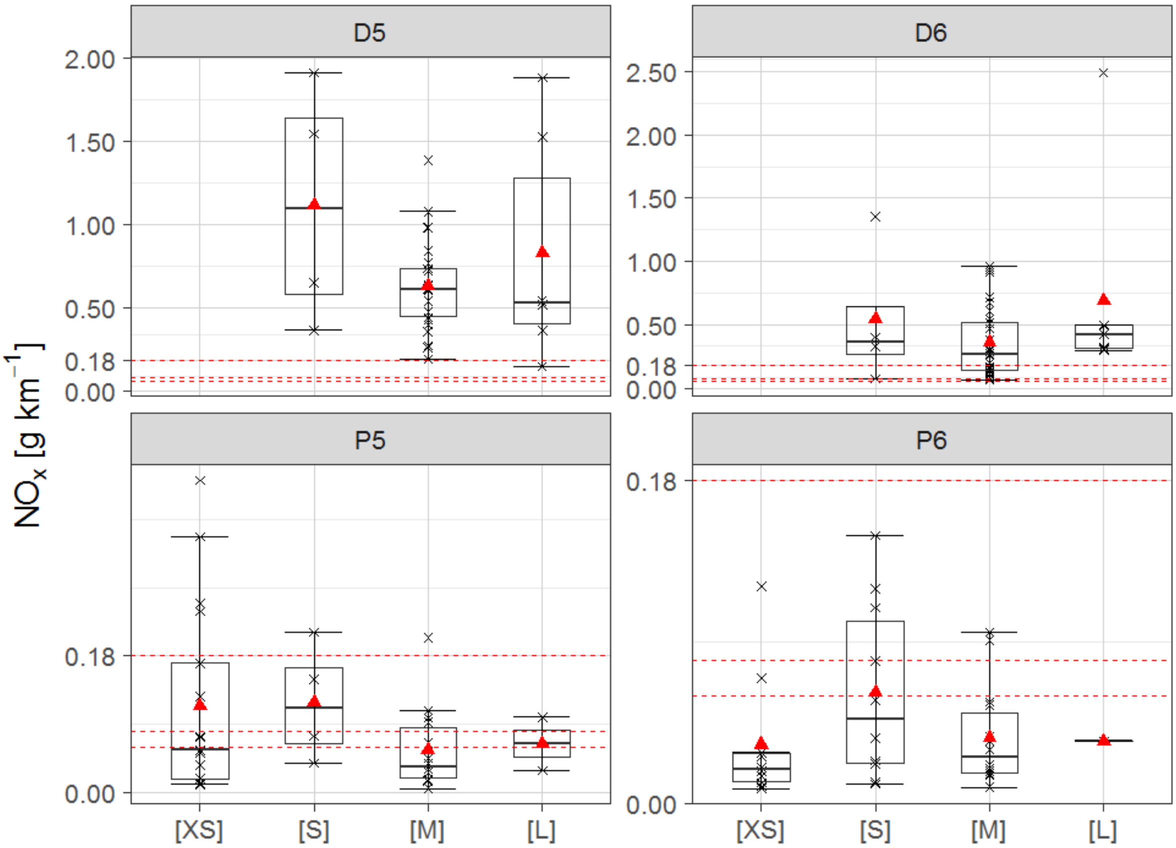

Figure 6-11. Urban NO_x emissions by engine displacement

As illustrated by **Figure 6-11** there was no significant relationship between NO_x emissions and engine displacement.

6.3.3 Primary NO_2 emissions

As well as differences in total NO_x emissions, diesel and petrol engines also differed in the amount of NO_x emitted as primary NO_2, as seen in **Figure 6-12**. Each bar represents a vehicle's total average urban NO_x, with the NO_2 component in dark grey and NO light grey. The diesel vehicles in the study emitted a much higher proportion of fNO_2 than the petrol; 42% and 46% for D5 and D6 respectably compared with 27%

and 17% for P5 and P6. This is similar to the range found by *Weiss, Bonnel, Hummel, et al., (2011)*. The proportion fNO_2 is higher for diesel vehicles due to the presence of the Diesel Oxidation Catalysts (DOC), which has the intended purpose of oxidising CO and THC. The DOCs also oxidise NO to NO_2 increasing the proportion fNO_2. It is also the case that some NO_x abatement technologies (SCR) are more effective at reducing total NO_x if the proportion fNO_2 is higher. Hybrids were not included in this analysis as the measured NO_2 emissions were extremely low, far below the measurement error of the PEMS system used.

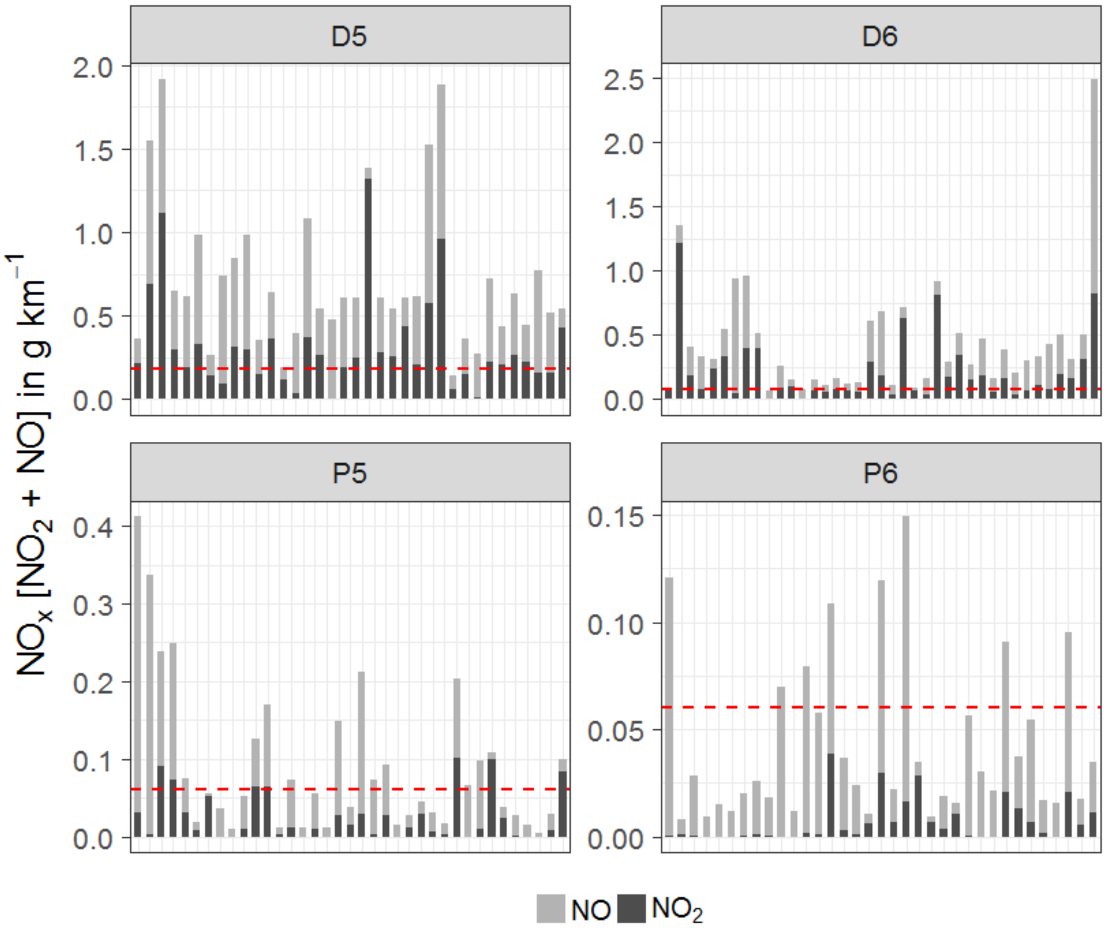

Figure 6-12. NO_x in g km^{-1} as NO and NO_2 components for urban sections (bar represents individual vehicle, red dashed line = NO_x type approval limit)

The average NO_2 emissions in g km^{-1} for motorway and urban sections by category are listed in **Table 6-12**. The average D6 urban NO_2 emission was 2.7 times the Euro 6 diesel type approval limit for total NO_x, the average D5 urban NO_2 emission was 1.8 times the Euro 5 diesel type approval limit for total NO_x.

Table 6-12. NO_2 emission by category [g km^{-1}]

NO_2 [g km^{-1}]					**Reduction from diesel to petrol**	
	D5	**D6**	**P5**	**P6**	**Euro 5**	**Euro 6**
Urban	0.315	0.215	0.025	0.007	92%	97%
Motorway	0.305	0.158	0.013	0.004	96%	97%

Table 6-12 also lists the % reduction from diesel to petrol, which ranged from 92% - 97%. NO_2 emissions from diesel vehicles were between 12.6 – 39.5 times higher than petrol vehicles of the same Euro standard. For all categories the average NO_2 emissions were higher for the urban sections than motorway and, as with NO_x and CO_2, the increase was highest for petrol vehicles.

6.3.4 CO emissions

The Euro standards also set legal limits for emissions of carbon monoxide (CO), 1.0 g km^{-1} for petrol vehicles and 0.5 g km^{-1} for diesel. These limits have been constant since Euro 4. **Figure 6-13** is a boxplot of urban and motorway section average CO emissions grouped by category. As with NO_x there was large amount of variance within the categories. Six D5 vehicles exceeded the diesel CO limit during urban driving, eight P5 vehicles exceeded the petrol CO limit during motorway driving. However, the vast majority of vehicles met both the petrol and diesel limit and every category average (marked with red triangle) was well below the relevant type approval limit. These results indicate that the catalytic converters used by both petrol and diesel vehicles are effective at oxidising CO. The average CO emission were 0.12 g km^{-1}, 0.09 g km^{-1}, 0.63 g km^{-1}, 0.21 g km^{-1}, 0.14 g km^{-1}, 0.00 g km^{-1} for D5, D6, P5, P6, H5 and H6 respectively.

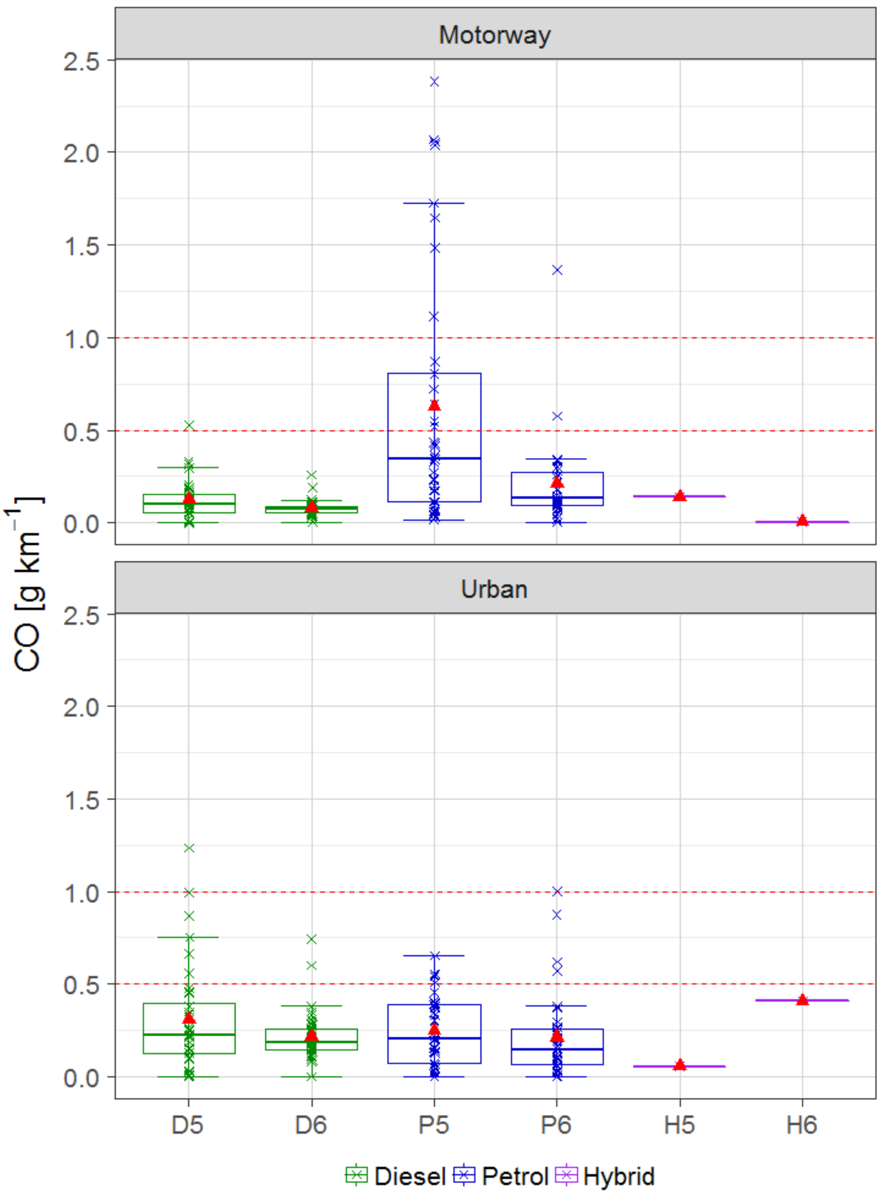

Figure 6-13. Urban and motorway CO emissions

6.3.5 Cold start emissions

As discussed, the majority of trips did not include a cold start. In this section two examples of cold starts recorded at the beginning of a test trips are illustrated. **Figure 6-14** shows the speed, CO and NO_x trace for the first 10 minutes of the trip for vehicle P5.1.4a. This was identified as cold start (as opposed to a warm/ hot start) using the initial exhaust temperature. The initial exhaust temperature for P5.1.4a was 15°C (ambient temperature) whereas the trip average exhaust temperature was 83.1 (sd. 36) °C.

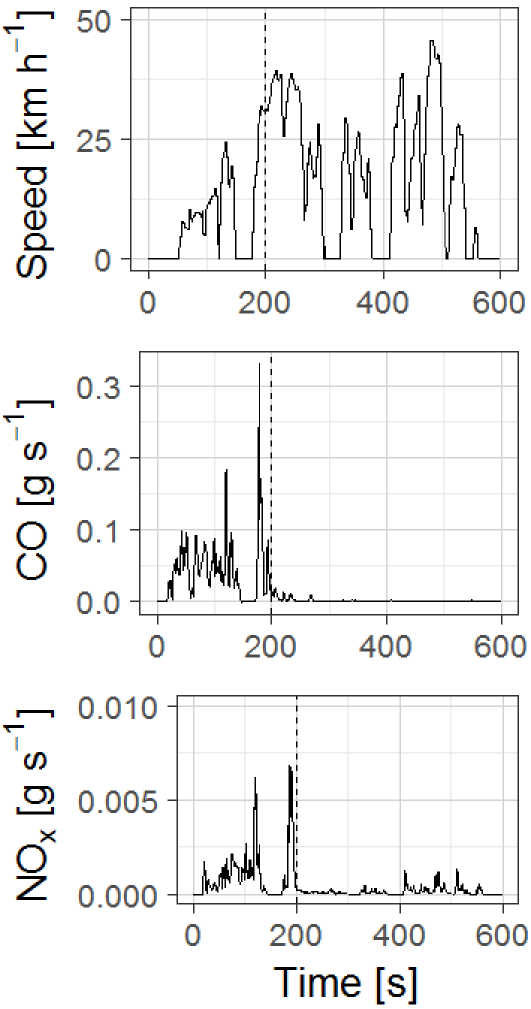

Figure 6-14. First 10 minutes of P5.1.4a trip including cold start. Dashed line marks approximate end of cold start period

Figure 6-14 shows spikes in CO and NO_x emissions that fell away after the first 200s. This is longer than the 120s cold start time period found by Chen *et al.* (2011). The Carbon monoxide (CO) type approval limit is 1.0 g CO km^{-1} for gasoline and 0.5 g CO km^{-1} for diesel. In the first 200s of the trip P5.1.4a had an average CO emission of 19.1 g CO km^{-1}. This was 830 times the urban section average from this vehicle. The first 200s had an average NO_x emission of 0.46 g NO_x km^{-1}. This was 4.6 times the urban section average and 23 times the motorway section average.

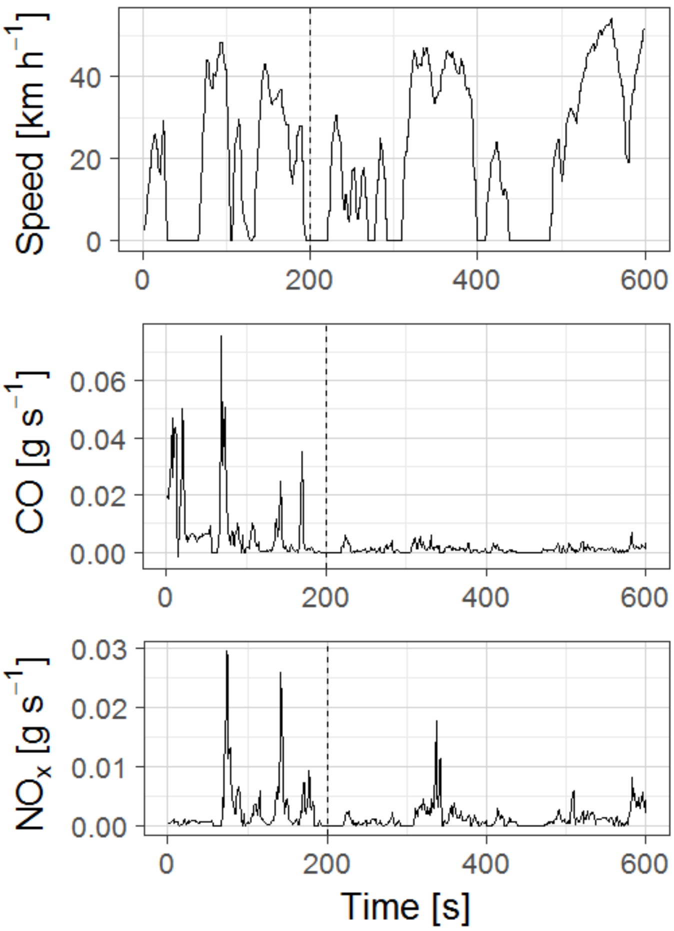

Figure 6-15. First 10 minutes of D6.2.2b trip including cold start. Dashed line marks approximate end of cold start period

Figure 6-15 is the first 10 minutes of emissions from vehicle D6.2.2b. Again the cold start period lasted ~200s. The average CO emission during the cold start period (2.48 g CO km^{-1}) was 15 times higher than urban average. The cold start average NO_x (0.78 g NO_x km^{-1}) was 2.4 times higher than the urban average.

These examples demonstrate how for both petrol and diesel vehicles the cold start emissions can be many times higher than the CO and NO_x emissions during normal driving operation. However the PEMS tests analysed in this study were not specifically designed to monitor cold start emissions. Therefore conclusions cannot be drawn from this PEMS testing regime as to the magnitude and characteristics of cold starts. These results highlight that cold start emissions are an area in which further work is required. This should include a PEMS testing regime specifically targeting cold start emissions.

6.4 Discussion

This section includes a discussion of the results presented in the previous section, with a particular emphasis on Euro 6 diesels. The D6 results from this chapter are compared to results from Chapter 4 and followed by discussion of the Euro 6d RDE type approval procedure. This is followed a basic cost benefit analysis of damage cost per km between the vehicle categories.

6.4.1 Petrol- electric hybrids

The petrol-electric hybrids in this analysis showed a clear improvement compared with convention petrol and diesel vehicles. However, with only two vehicles the sample size was very limited. The average NO_x and CO_2 emissions presented in this chapter are strengthened by the fact they are in good agreement with previous studies. The average NO_x and CO_2 emissions from this analysis (between 0.002 – 0.010 g NO_x km^{-1} and 117.4 – 150.9 g CO_2 km^{-1}) were within the range found by *Wu et al.,* (2015) (0.009 ± 0.005 g NO_x km^{-1}, 136 ± 21 g CO_2 km^{-1}). This again was similar to the findings of Weiss, Bonnel, Hummel, *et al.*, (2011).

6.4.2 Comparing D6 with urban and motorway sections from Chapter 4

Using the selection method from Chapter 4 (sections selected by GPS co-ordinates) the D6 urban deviation ratio was 5.4 and motorway was 3.9. This is very similar to the results from this chapter (where 16 km sections were selected to meet dynamic boundary conditions of Regulation (EU) 2016/646), which had an urban deviation ratio of 5.5 and motorway of 4.1. Results from the two selection methods were in good agreement, as illustrated by **Figure 6-16**, which is a scatter plot of the average NO_x

calculated using the different methods. There was strong correlation ($R^2 > 0.9$) for both urban and motorway sections.

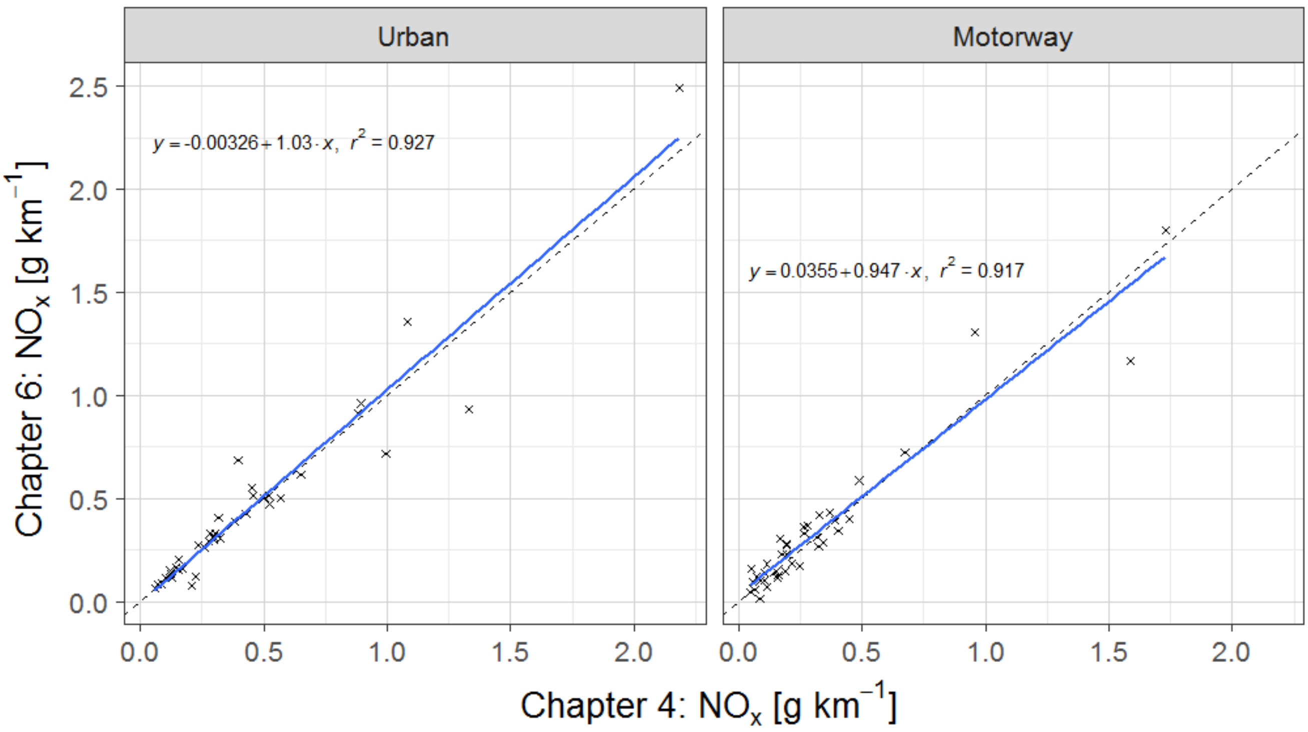

Figure 6-16. Comparing methods of urban and motorway section selection (dashed line y = x)

This result does not however show that the test routes used during the Euro 6d RDE type approval will be completely representative of real world driving. As discussed previously, the sections selected for this chapter met the dynamic boundary conditions but differed from the potential sections for Euro 6d type approval in several ways. Firstly, the results from this chapter were not binned by speed. Secondly, for continuity the sections selected in this chapter had an average speed ~25 km h^{-1}. In the Euro 6d type approval vehicles are permitted a range of average speeds between 15 – 40 km h^{-1}, this gives manufacturers leeway to minimise NO_x emissions. Thirdly, the RDE type approval sets a minimum limit to the number of datasets with a_i > 0.1 ms^{-2} of 150. The majority of NO_x emissions occur during acceleration, so manufacturers could aim for as close to 150 datasets with a_i > 0.1 ms^{-2} as possible in a bid to minimise emissions. In the sections analysed in this chapter the average number of datasets in the urban section with a_i > 0.1 ms^{-2} was 778 (sd. 55), potentially much higher than tests that could be submitted as Euro 6d RDE type approval sections.

2.1.1.5 Lowest possible NO_x emissions from same measurement data

For comparison the urban section for each vehicle with the minimum average NO_x whilst still meeting the Regulation (EU) 2016/646 was found. This was done using purpose built software in the statistical package R. Each trip was binned by speed (< 60 km h^{-1}) and again split into moving 16 km sections. The 16 km section that fulfilled all the requirements listed above and had the lowest average NO_x was then selected for comparison.

Figure 6-17 compares the average NO_x emission from each vehicle calculated in this chapter (16 km section selected using the Euro 6d dynamic boundary conditions and average speed ~ 25 km h^{-1}) to the lowest possible NO_x measured over a 16 km section (within the RDE boundary conditions) from the same test data. The aim of this is to show that even within the same test trip there are sections that meet the urban Euro 6d type approval specifications that have average NO_x emissions much lower than the emissions recorded across the entire urban section of the trip. This shows that even with the introduction of RDE into the type approval process, real world emissions may still exceed emissions recorded during the RDE type approval test.

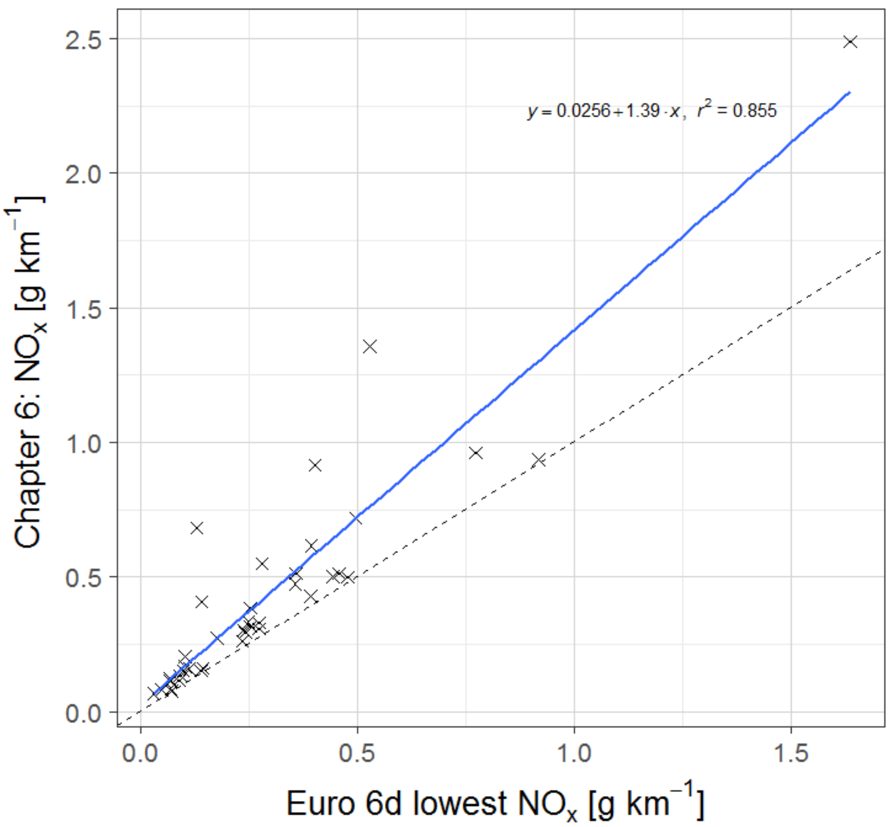

Figure 6-17. D6 comparison of sections with lowest NO_x meeting RDE type approval standards and sections used in this analysis

The lowest urban NO_x sections had a mean deviation ratio of 3.7, and 16 vehicles met the Euro 6d-TEMP type approval limit compared to 13 using the Chapter 6 method. The sections selected in this chapter had NO_x emissions on average 58% higher than the Euro 6d lowest NO_x sections. This is within the range of variability between the Euro 6d RDE type approval and real world predicted by the ICCT (21 – 63%) due to a combination of "*engineering safety margin for RDE*" and "*extended conditions*" (Miller & Franco, 2016). This shows how controlling driving characteristics within the RDE type approval could result in average NO_x emissions ~30% lower than the true real world emissions. Furthermore, neither of the methods compared in **Figure 6-17** included cold starts, as a result they are both likely to be an underestimate.

6.4.3 Damage costs

The results presented in this chapter indicate that a consumer shift towards petrol cars would incur a CO_2 increase per vehicle of between 13 – 66% and deliver a reduction in NO_x of 88 – 96%. Whilst the UK has firm commitments to reduce CO_2 emissions some argue that the benefits to air quality of replacing diesel with petrol far outweigh the potential climate change costs. For example, a recent study by *Brand* (2016) found that in the UK the relative air quality benefits of switching from diesel to petrol outweighed any carbon dis-benefits. NO_x and CO_2 are notoriously difficult to compare, and most studies focus on either air quality or climate change (van der Zwaan, Keppo & Johnsson, 2013). Air pollution presents an immediate threat to human health whereas climate change presents a profound longer-term hazard to the climate and those, like humans, who depend on it.

This section aims to quantify the potential costs/ benefits between diesel and petrol vehicles using a basic damage cost approach similar to that used in Chapter 5. The range of CO_2 and NO_x damage costs presented in the previous chapter were applied to the real world emissions factors from this chapter. This was done in two different ways. First, the damage cost per km was calculated simply by multiplying the average emission factor in g km^{-1} by the various CO_2 and NO_x damage costs in £/tonne to calculate vehicle specific damage costs in £/km. Second, scenarios were devised to 2030 modelling different evolutions of petrol and diesel real world emissions factors and comparing the total projected UK annual 2030 combined damage cost for CO_2 and NO_x.

Note this is not a full Life Cycle Analysis and there are additional costs associated with various pollutants not considered here. There are also additional CO_2 emissions associated with the extraction, refining and distribution of petroleum fuels. The JRC "*Well-to-wheels Analysis of Future Automotive Fuels and Powertrains"* is a well-recognised study analysing well to wheel emissions and energy efficiencies of various fuels. They found that in terms of energy requirement and GHG emissions, diesel is ~10 – 20% more efficient than petrol (Edwards *et al.*, 2011).

6.4.4 Damage cost per km

The NO_x damage costs used in this analysis are the lowest, middle and highest NO_x damage costs from the previous chapter. The lowest NO_x cost (NO_x [L]) was £6,734/tNO_x, the mid cost (NO_x [M]) was £16,853/tNO_x and the highest (NO_x [H]) was £40,404/tNO_x. Damage costs for CO_2 came from the UK Department of Energy and Climate Change (DECC, now BEIS) *Green book supplementary guidance (2015)*. These values are listed in **Table 6-13** and are non-traded carbon values used for UK public policy appraisals (DECC, 2015).

Table 6-13. CO_2 damage costs (DECC, 2015)

Non- traded CO_2 damage costs	**Low CO_2 [L]**	**Central CO_2 [M]**	**High CO_2 [H]**
2015 Baseline costs [£/tonne CO_2]	£31	£62	£94

The DECC damage costs are based on a target-consistent approach as opposed to the Social Cost of Carbon. There is huge uncertainty relating to CO_2 marginal damage costs, with studies presenting values as varied as \$5/tCO_2 to \$200/tCO_2 (Tseng & Hung, 2014). The widely renowned *Stern Review on The Economics of Climate Change* (2007) decided on a mean value of \$85/tCO_2 (approximately £79/tCO_2 in 2017 after allowing for inflation) (Stern, 2007). This Stern figure was derived using a low discount rate (how much less future generations are valued compared to those who are alive today) of 1.4%. However, the UK government guidance recommends a higher discount rate of 3.5% (HM Treasury, 2011).

Damage costs per km have been calculated by converting each vehicle's average NO_x and CO_2 emission in g km^{-1} to tonne/km and then multiplying by the various damage costs listed above and adding the NO_x and CO_2 components. The vehicles were then grouped by category and an average taken. **Figure 6-18** is a boxplot of damage cost per urban km for each damage cost combination and vehicle category.

The label above each plot indicates which combination of damage costs was used in the plot below. The first segment of the title refers to the NO_x damage cost used, the second refers to the CO_2 damage cost used. The mean for each category is marked with a red triangle.

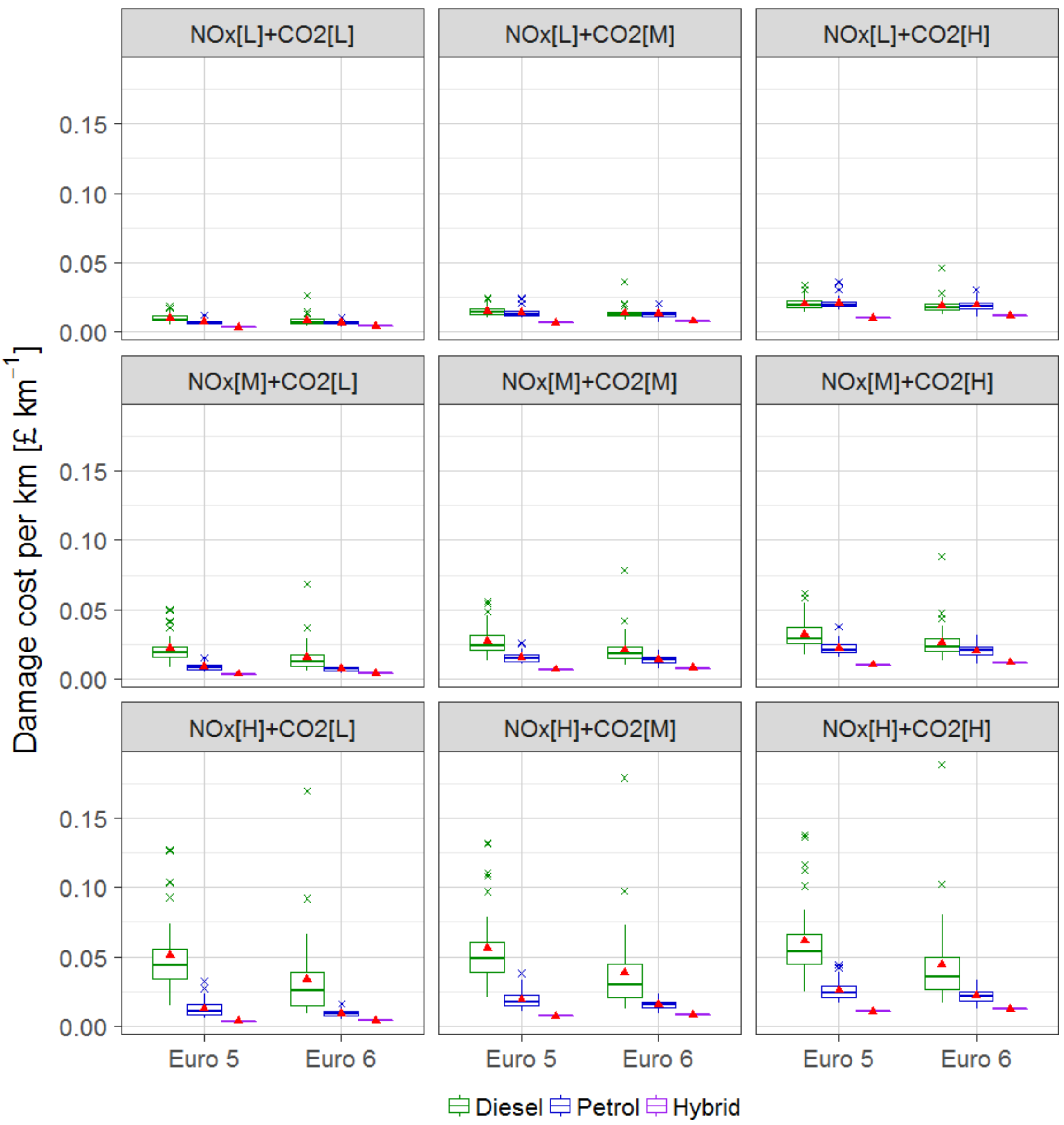

Figure 6-18. Urban total damage cost per km by category using DEFRA and AIM damage costs for NO_x and DECC damage costs for CO_2

Figure 6-18 shows that the relative benefits of each fuel type varied widely depending on the damage cost combination used. For all NO_x [L] scenarios there was little benefit in switching from diesel to petrol, whereas for the NO_x [H] scenarios diesel average damage costs were between 2 – 4 times higher than petrol. Similar results were found for motorway driving. **Table 6-14** lists the urban and motorway £/km using the central cost combination (NO_x [M] + CO_2 [M]) and the mean £/km from all costs for each vehicle category.

Table 6-14. Mean urban and motorway damage costs £ per km

[£ per km]	**Urban central estimate (NO_x[M]+CO_2[M])**	**Urban mean estimate**	**Motorway central estimate (NO_x[M]+CO_2[M])**	**Motorway mean estimate**
D5	0.027 (sd. 0.011)	0.033 (sd. 0.025)	0.027 (sd. 0.013)	0.033 (sd. 0.028)
P5	0.016 (sd. 0.004)	0.016 (sd. 0.008)	0.011 (sd. 0.002)	0.011 (sd. 0.005)
H5	0.007	0.007 (sd. 0.003)	0.011	0.011 (sd. 0.005)
D6	0.021 (sd. 0.012)	0.025 (sd. 0.022)	0.017 (sd. 0.009)	0.019 (sd. 0.017)
P6	0.014 (sd. 0.003)	0.014 (sd. 0.006)	0.010 (sd. 0.002)	0.011 (sd. 0.005)
H6	0.008	0.008 (sd. 0.003)	0.008	0.007 (sd. 0.003)

Hybrid vehicles were the clear best for all damage cost combinations. The mean damage cost reduction per km by replacing a diesel vehicle with a petrol- electric hybrid (of the same Euro standard) ranged between 37 – 93 % (dependant on the damage costs used) with a mean of 68% reduction. Compared with conventional petrol, hybrids reduced damage costs per km in the range between 38 – 71% with a mean of 51%.

As discussed in Chapter 5 the damage cost of NO_x emissions are dependent on where the emission takes place, emissions in densely populated urban areas cause the most damage. The different NO_x damage costs used in **Figure 6-18** could be used as a proxy for location. This indicated that in rural areas where the damage cost of NO_x is low, the total damage cost from using petrol would be higher, whereas in urban areas the reverse is true. **Table 2-5** (Chapter 2) showed the existing trend is for diesel cars to spend ~10% more VKM on the motorway and petrol cars to spend ~10% more VKM on urban roads. Policies that promoted this trend could result in air quality and climate change benefits.

For all but one cost combination petrol had a lower average per km damage cost than diesel. The increase in damage cost per km between petrol and diesel was most substantial for the NO_x [H] scenarios (bottom row of **Figure 6-18**). The reduction in £/km from diesel to petrol (of the same Euro standard) ranged from -2 – 75% with a mean of 38%.

These results indicate that Brand (2016) was right to say the NO_x benefits of petrol outweigh the carbon dis-benefits. However, these results do not take into account what has been a key finding of this research: the huge variability in real world emissions of diesel passenger cars. As seen previously with average NO_x emissions, a few high emitting diesels for both D5 and D6 increased the diesel mean damage costs. When considering performance of vehicles individually there was a lot of overlap between diesel and petrol as shown in **Figure 6-19**.

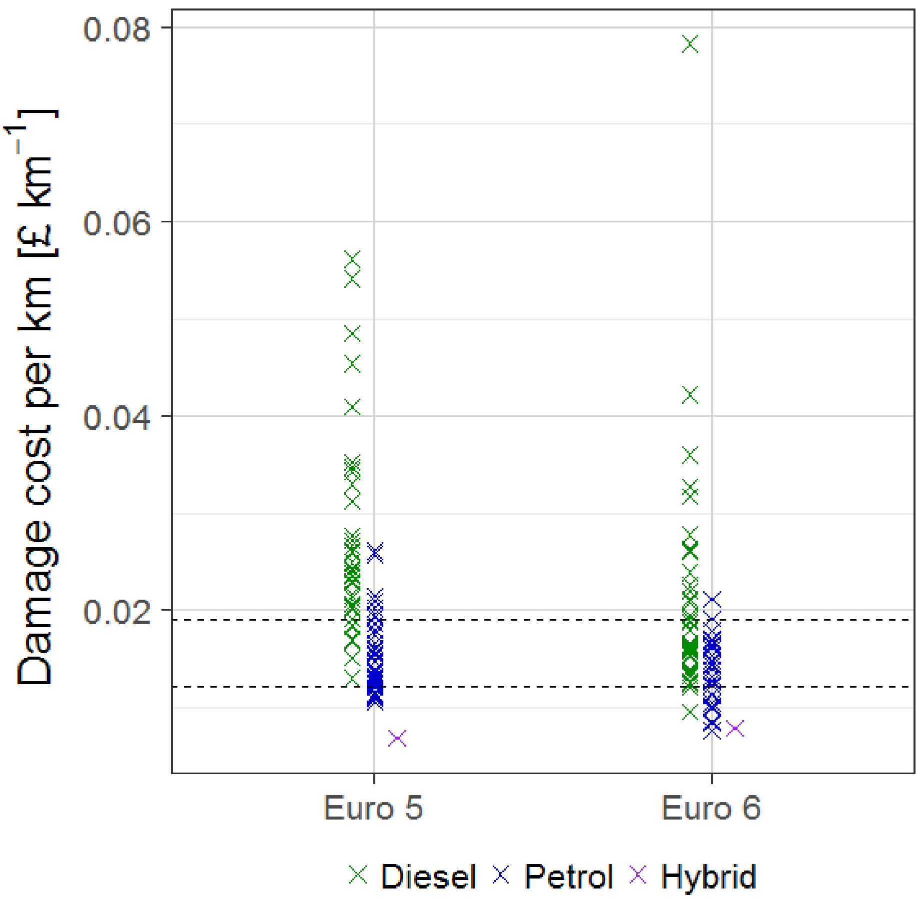

Figure 6-19. Urban damage cost per km for individual vehicles for NO_x [M] + CO_2 [M]

Figure 6-19 uses the central costs for both NO_x and CO_2. When using this central damage cost estimation, the mean reduction in £/km from D6 to P6 was 33%. However, when comparing individual vehicles the majority for D6 and P6 fell within the same range, this range is marked with dashed horizontal lines in **Figure 6-19**. 63% of D6 vehicles and 68% of P6 vehicles fell within this range. As seen throughout this analysis the diesel damage costs were greatly increased by the ~10% of high polluting vehicles that pulled up the mean. The mean D6 damage cost for NO_x [M] + CO_2 [M] was 0.21£/km. When the worst 10% of NO_x emitters were removed this fell to 0.16£/km and was comparable to the P6 cost of 0.14£/km. This highlights the value of discriminating on the basis of real world emissions as opposed to Euro standards and the importance of tackling the worst diesel vehicles.

These results indicate that the potential cost/ benefits of switching from diesel to petrol are hugely dependent on the damage costs assigned to NO_x and CO_2. Given the huge uncertainty surrounding these damage costs and a lack of academic consensus it is difficult to draw firm conclusions. There are also additional Well-to-Wheel emissions (not considered in this analysis), which are higher for petrol cars (Edwards *et al.*, 2011). Additionally there are health effects not factored relating to other pollutants and the effect of cold start emissions which are thought to effect ~8% of VKM (Miller & Franco, 2016). The only consistent result was the reduction in damage cost per km when switching to petrol- electric hybrids.

2.1.1.6 Damage costs projected to 2030

As discussed in Chapter 2 petrol and diesel vehicles are distributed differently throughout the road network. Diesel vehicles have ~65% higher annual average mileage and spend a higher percentage of time on motorways. In this section a simple spreadsheet model used VKM outputs from the UKIAM to calculate the potential 2030 damage costs of 7 different scenarios relating to real world Euro 5 and 6 petrol and diesel emissions of CO_2 and NO_x.

The PEMS measurements for NO_x and CO_2 were combined with UK traffic projections and damage costs to perform a cost benefit analysis. The year 2030 was chosen because by 2030 the vast majority of both petrol and diesel passenger cars will be

Euro 6 standard, with the remainder being Euro 5. The results presented relate only to Euro 5 and Euro 6 passenger car emissions.

Seven scenarios were chosen to represent possible outcomes of various transport emissions policies and incoming emissions regulations. The scenarios have been chosen to highlight the air pollution/ climate change trade off.

Table 6-15. Description of 2030 diesel and petrol CO_2 and NO_x scenarios

Scenario 1 (S1) – No improvement in petrol or diesel emissions from 2016 – 2030. NO_x and CO_2 emissions factors from this PEMS study were applied to 2030 VKM. This is the worst case scenario and unlikely due to introduction of new regulatory limits for both NO_x and CO_2.
Scenario 2 (S2) – No improvement in NO_x emissions but big improvement in CO_2 from 2016 – 2030. NO_x emissions factors from the PEMS study are used but it is assumed the average CO_2 emissions for all vehicle types falls to 130 g CO_2 km^{-1}
Scenario 3 (S3) – No improvement for D5, P5 and P6 vehicles NO_x emissions and no improvement in CO_2 but improvement in D6 NO_x in line with that projected by ICCT. PEMS emissions factors are used for all vehicles except D6 NO_x which is changed to 0.168 g NO_x km^{-1} (the Euro 6d TEMP real driving limit)
Scenario 4 (S4) – No improvement for D5, P5 and P6 vehicles NO_x emissions and no improvement in CO_2 but big improvement in D6 NO_x. PEMS emissions factors are used for all vehicles except diesel Euro 6 NO_x which is changed to 0.08 g NO_x km^{-1}, the Euro 6 type approval limit (deviation ratio = 1)
Scenario 5 (S5) – No D5 or D6 vehicles in the fleet mix. All km driven by diesel cars are replaced by petrol cars of the same engine size (i.e. petrol cars replace all

diesel cars). This models a national diesel scrappage scheme which by 2030 leads to the replacement of all diesel cars with petrol
Scenario 6 (S6) - No D5 or D6 vehicles in the fleet mix. All km driven by diesel cars are replaced by petrol- electric hybrid cars (i.e. hybrid cars replace all diesel cars). This models a national diesel scrappage scheme which by 2030 lead to the replacement of all diesel cars with hybrid
Scenario 7 (S7) – Improvements in CO_2 emissions from all cars and D6 NO_x. The average CO_2 emissions for all vehicle types falls to 130 g CO_2 km^{-1}. Diesel Euro 6 NO_x emission factor is changed to 0.08 g NO_x km^{-1}, the Euro 6 type approval limit (deviation ratio = 1)

The total combined CO_2 and NO_x cost in Billion £ for each scenario was calculated in the five steps illustrated in the flow chart below (

Figure 6-20**).**

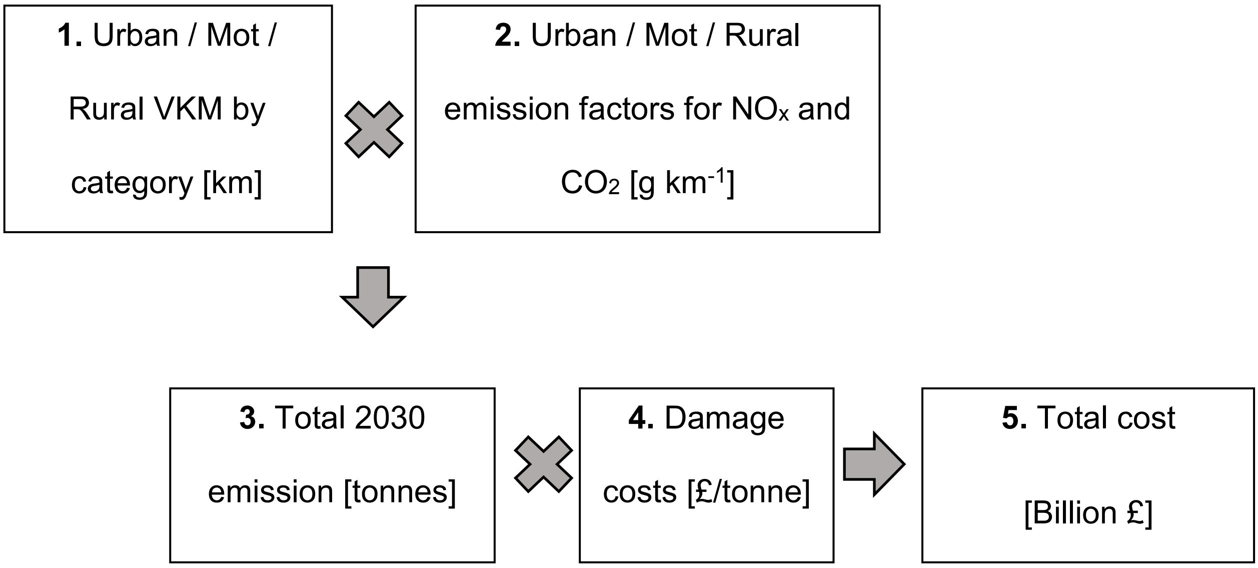

Figure 6-20. Flow diagram of spreadsheet model

The total 2030 VKM for motorway, urban and rural driving were taken from the UKIAM 2030 projections, as used in Chapter 5. The urban and motorway emissions factors were taken from the analysis in this chapter or modified as indicated above. For rural km the average between the urban and motorway emissions factors was taken. This is in keeping with the findings of *Heijne, Ligterink & Stelwagen* (2016) that NO_x and CO_2 was highest in urban driving, followed by rural and lowest in motorway. This is an assumption that should be noted when discussing the results of this modelling. The damage costs used were the same as in the previous section, they were combined in exactly the same way and stated in 2015 prices.

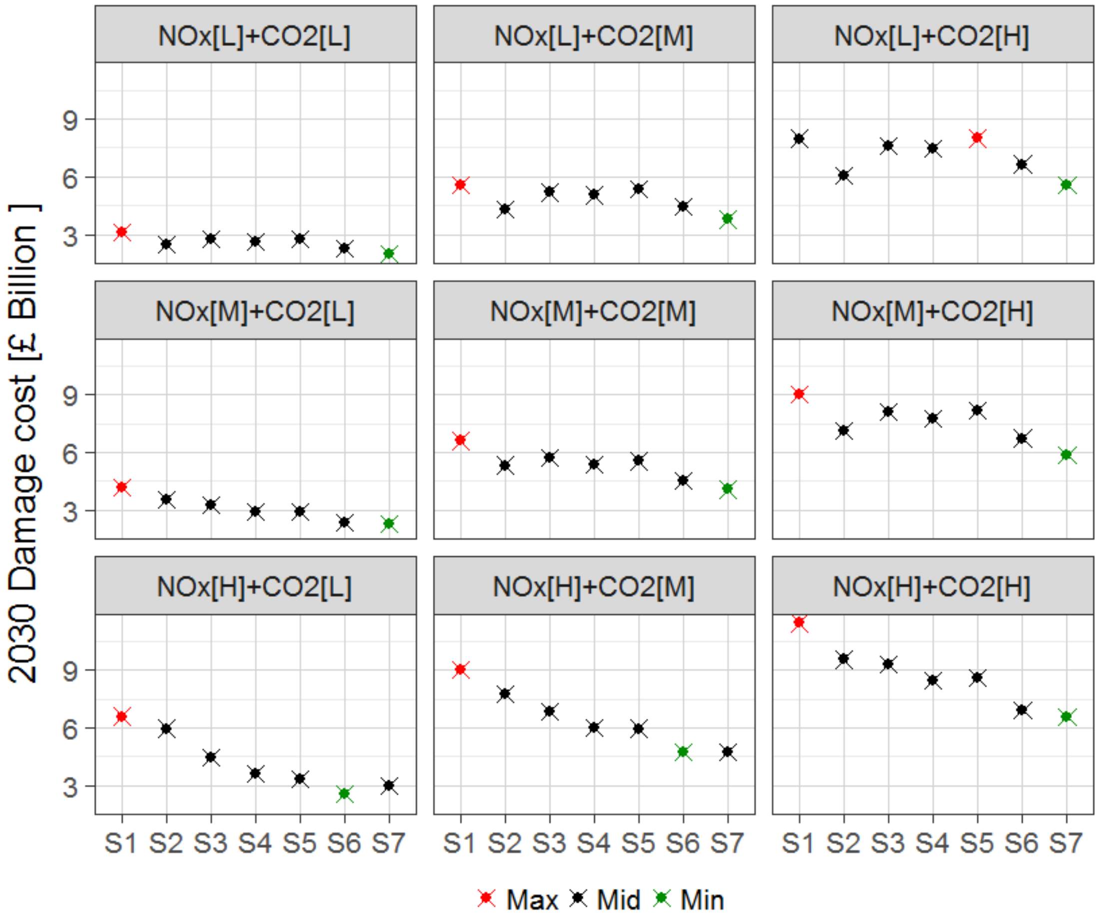

Figure 6-21. 2030 damage costs for the scenarios using various damage cost in [£/tonne] combinations

The damage costs for each scenario by cost combination are plotted in **Figure 6-21**. The highest total damage cost for each cost combination is plotted in red, the lowest in green. The highest cost scenario for all but one damage cost combo was S1 (BAU). The only cost combination for which S1 was not the highest was NO_x [L] +CO_2 [H], for which S5 (all diesel cars replaced with petrol) was marginally higher. This is to be expected, as replacing diesel (S5) with petrol reduces NO_x and increases CO_2, and NO_x[L]+CO_2[H] has the lowest value for NO_x and highest for CO_2. S5 (replace all diesel cars with petrol) was also one of the lowest cost scenarios for many damage cost combinations, especially those with NO_x [H]. This highlights the level of uncertainty surrounding the air quality/ climate change trade off and the need for coherent policies that tackle both. The scenario with the lowest total damage cost for 7 out of 9 cost combinations was S7 (all CO_2 = 130 g km^{-1}, D6 NO_x = 0.08 g km^{-1}). The remaining 2 lowest were scenario S6 (replace all diesel cars with petrol- electric hybrids). This indicates that the best potential outcome (S7) was if both CO_2 and NO_x emissions regulations were properly effective.

Table 6-16 lists the central estimate (NO_x [M] + CO_2 [M]) and mean of all damage costs for the seven scenarios. For the central cost combination scenario, S1 had the highest damage cost. There was little difference between S2 – S5. For both the central estimate and mean of all damage costs, the lowest impact scenario was S7 (all CO_2 = 130 g km^{-1}, D6 NO_x = 0.08 g km^{-1}).

Table 6-16. Damage cost by scenario (red = highest cost, green = lowest cost)

Scenario	**Central estimate** (NOx[M]+CO2[M]) [Billion £]	**Mean estimate of all damage cost combos** [Billion £]
S1	6.60	7.06
S2	5.33	5.79
S3	5.70	5.92
S4	5.34	5.47
S5	5.55	5.63
S6	4.54	4.58
S7	4.08	4.21

Table 6-16 highlights that potential damage costs can be mitigated by addressing NO_x or CO_2, but the most effective course of action would be to tackle both. The potential damage cost saving by 2030 if all vehicles reduced CO_2 and D6 conformed to type approval (compared to BAU i.e. S1 – S7) was 2.52 – 2.85 Billion £.

The main conclusion that can be drawn from both the £/km calculation and 2030 total damage costs is that there is huge uncertainty caused by the wide range in both NO_x and CO_2 damage costs. Within this uncertainty range it is possible that replacing diesel cars with petrol will result in air quality benefits that outweigh climate change dis-benefits. However, it is also possible the two will cancel each other out. It was clear from this basic modelling the optimum approach would be to tackle both CO_2 and NO_x.

6.5 Summary

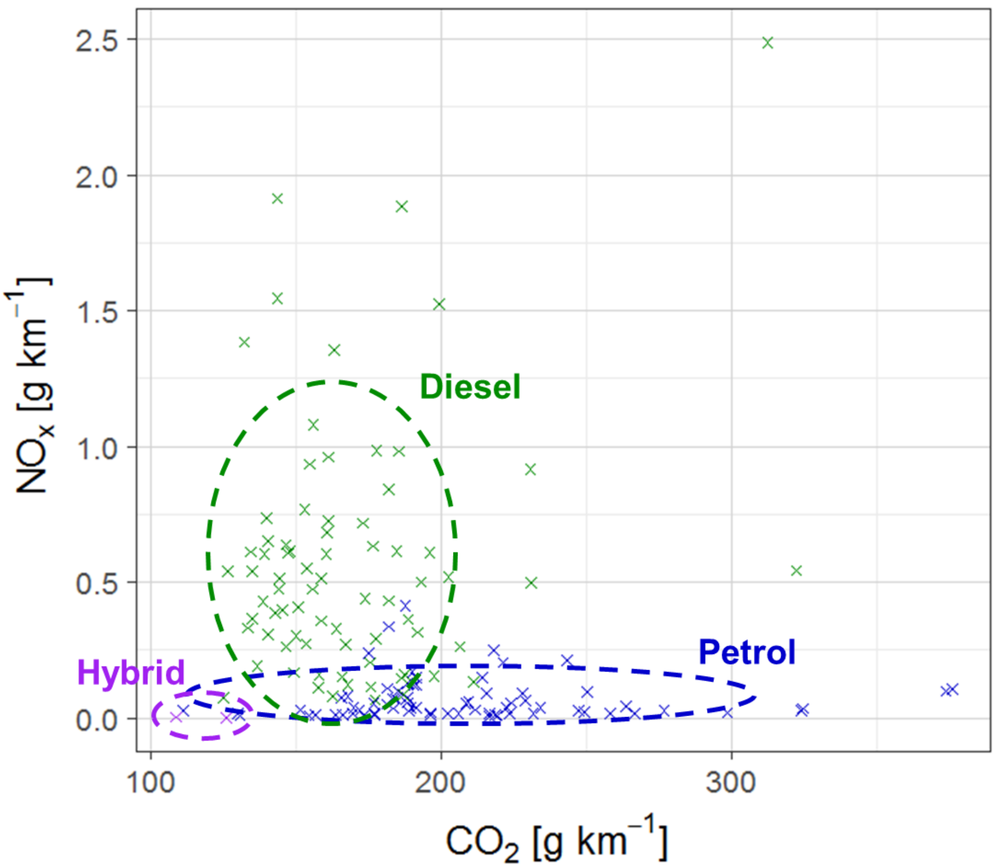

Figure 6-22. Urban NO_x vs. CO_2 by fuel type

In this chapter the results from PEMS measurements of NO_2, NO_x, CO_2 and CO were presented for 149 diesel, petrol and hybrid vehicles. The key findings of this analysis are summarised by **Figure 6-22**, which plots CO_2 against NO_x for the urban sections. There was huge variability within the vehicle categories but in general data followed the expected trends. Petrol vehicles CO_2 emissions were between 13 – 66% higher than diesel, and petrol NO_x emissions were 88 – 96% lower.

For urban driving the two hybrid vehicles delivered a 50% reduction in CO_2 when compared to conventional petrol, as well as reduced NO_x emissions. This is in agreement with existing literature, however it should be noted that the sample size in this study was small and both vehicles came from the same manufacturer. Further work should include an expansion of the petrol- electric sample and an extension to include different types of hybrid technology, about which no conclusions can be drawn from this study.

The vast majority of diesel vehicles exceeded the relevant type approval limits for NO_x many times over. With the exception of urban driving for hybrid cars, all vehicle categories exceeded the fleet target CO_2 limit of 130 g km^{-1}. Most vehicles exceeded the manufacturers' stated CO_2 emissions by ~40%.

A cost-benefit analysis highlighted the uncertainty surrounding the air quality/climate change trade off relating to replacing diesel with petrol. However, scenarios that reduced both CO_2 and NO_x consistently delivered the greatest benefit. This indicates that more should be done to include CO_2 in the discussion as consumers move away from diesel.

Chapter 7. Summary and Discussion

This is the final chapter of the thesis, it pulls together the themes and major findings of the work and discusses them in a wider policy context. The aim of this research was to investigate the uncertainties relating to emissions from passenger cars with a view to mitigating some of them and considering how they affect air quality policy. This chapter summarises the work and indicates how the initial aims and objectives were realised within the thesis.

7.1 Summary and Discussion

This thesis set out to identify the key uncertainties relating to passenger car emissions, explore how they could be minimised and how they affect air quality policy. This section describes how this was achieved using the aims and objectives stated in introductory chapter as guidance.

7.1.1 Develop a framework to assess the possible causes of uncertainty in passenger car emissions and potential risks

The Hazards and Operability (HAZOP) approach described in Chapter 3 provided the framework for the uncertainty analysis presented in this thesis. The initial HAZOP assessment highlighted NO_x emissions from passenger cars as a key uncertainty in applying the UK integrated assessment model (UKIAM) to future air pollution scenarios for the UK. The crucial concern identified was real driving emissions and how they relate to type approval limits and the emissions factors used in the air quality models, particularly the latest Euro 6 vehicles (first sold in 2014). With the introduction of Euro 6 the diesel type approval limit for NO_x was reduced by 56% but there had been little real world testing.

The four step process that forms a HAZOP assessment (identify, define, consider deviations, consider consequences) provided a useful framework for the subsequent research. The “identify” and “define” steps were fulfilled by the work in Chapters 2 and 3, the PEMS studies in Chapters 4 and 6 were part of the “consider deviations” step and the scenario analysis in Chapters 5 and 6 “considered the consequences” of those deviations. The framework provided by HAZOP ensured the thesis was a well-structured and cohesive body of work.

The potential risks identified related to the National Emissions Ceiling Directive (NECD) and the Air Quality Framework Directive (AQFD). For the NECD the risk was possible failure to meet the national NO_x ceilings in future years when the ceilings will be lower and Euro 6 vehicles dominant. For the AQFD the risk was UK exceedance of the air quality limit value for NO_2 continuing for many years to come.

7.1.2 Use Portable Emissions Measurement System (PEMS) data to explore real world emissions of passenger cars

The first PEMS study (presented in Chapter 4) included 39 Euro 6 diesel passenger cars and focused on emissions of NO_x and NO_2. The average urban NO_x emissions were 5.4 times the Euro 6 type approval limit. It was also found that the average NO_2 emission was 2.5 times the type approval limit for total NO_x and that the proportion of NO_x emitted as primary NO_2 had risen to 44%. This increase in the primary NO_2 was due to the introduction of Diesel Oxidation Catalysts and NO_x abatement technologies that either increase primary NO_2 or require higher levels of NO_2 to improve operating efficiency.

The 39 Euro 6 vehicles tested each used one (or two) of the three main NO_x after treatments; Exhaust Gas Recirculation (EGR), Lean NO_x Traps (LNT) and Selective Catalytic Reduction (SCR). There was no significant difference between the NO_x and NO_2 emissions of vehicles using these three technologies. The only significant difference was in the proportion of primary NO_2 emitted, for which SCR had a higher percentage than LNT and EGR. It is possible the lack of distinction in real driving emissions between vehicles using different after treatments was due to the presence of defeat devices. Particularly as all vehicles used EGR, most also used either SCR or LNT in addition. It would be expected that if the LNT and SCR were fully functioning vehicles would produce lower emissions than those using EGR only. This was the case for some vehicles. Cars using both LNT and SCR had real driving emissions below the type approval limit. However, LNT and SCR vehicles also produced some of the highest emissions.

A key finding from this study was the huge variability in NO_x emissions from diesel cars, even from vehicles using the same abatement technologies. Two vehicles were able to meet the Euro 6 type approval limit (0.08 g km^{-1}) whereas some had emissions many times higher, by as many as 23 times. Six vehicles in particular had NO_x emissions far exceeding the others, with the effect of increasing the mean. When these 6 vehicles were removed there was a 35% reduction in the average NO_x emission of the test fleet. The thermal windows used by manufacturers to switch off NO_x controls outside a certain temperature window also add to the variability. The scale of variability creates the need for a grading system based on real driving emissions (such as the EQUA index) to inform consumers and policy makers of the true emissions of the vehicles on the road.

Due to the deviation from type approval limits road transport models are required to develop emissions factors to represent real world driving. The accuracy of these emissions factors was identified as an important uncertainty in the initial HAZOP analysis. This thesis focused on COPERT as it is recommended model of the European Environment Agency. Comparison with the Euro 6 diesel PEMS data found real world emissions were 1.8 times higher than COPERT 4v11 estimates for NO_x and 2.9 times higher for NO_2. COPERT uses speed dependent emissions factors but analysis in this study found emissions correlated more with acceleration than with speed.

Diesel NO_x emissions were found to be highest during urban driving. This is of particular concern as public exposure is highest in urban locations. Analysis of the instantaneous emissions measurements in relation to speed and acceleration found the reason for the increase in emissions during urban driving was the increased prevalence of acceleration events. Both NO_x and NO_2 emissions were three times higher during acceleration than deceleration, and urban driving contained twice as much acceleration as motorway driving.

The second PEMS study (described in Chapter 6) expanded the scope to include Euro 5, petrol and hybrid vehicles with a larger sample of 149 vehicles. This study placed the Euro 6 diesel emissions in a wider context. It was also important because consumer data indicates a decline in diesel sales, the majority of consumers instead choosing petrol, with only a modest increase in the number of alternative fuel vehicles. The key results from the second measurement study are listed in **Table 7-1**.

Table 7-1. Urban real driving emissions recorded in this study

	NO_x limit [g km^{-1}]	**NO_x** [g km^{-1}]	**Deviation Ratio**	**NO_2** [g km^{-1}]	**fNO_2** [%]	**CO_2 limit** [g km^{-1}]	**CO_2** [g km^{-1}]
D6	0.08	0.44	5.5	0.215	46%	130	170.2
D5	0.18	0.72	4	0.315	42%	130	170.2
P6	0.06	0.04	0.7	0.007	17%	130	210.5
P5	0.06	0.09	1.5	0.025	27%	130	210.5
H6	0.06	0.002	0.03	-	-	130	117.4
H5	0.06	0.003	0.05	-	-	130	117.4

Petrol vehicles had CO_2 emissions between 13 – 66% higher than diesel vehicles of the same engine size. Petrol NO_x emissions were 88% lower for Euro 5 and 96% lower for Euro 6. This study included 2 hybrid vehicles, the sample was small and the results only related petrol-electric hybrids using kinetic energy recovery. These 2 hybrids were the best performing group. During urban driving the hybrids emitted 50% less CO_2 than the conventional petrol cars, as well as greatly reduced NO_x emissions. They were the only group to have average emissions below the fleet target CO_2 limit of 130 g km^{-1}. However, these results are only representative of one make of hybrid with unique technology, further work is needed on other types.

As with NO_x emissions CO_2 was found to exceed manufacturers' official estimates. For the newest Euro 6 petrol vehicles the average urban CO_2 emission was 44% higher than officially stated, Euro 6 diesel emissions were 36% higher.

Again it was found a large factor in the uncertainty surrounding diesel NO_x emissions was the huge variability between seemingly similar vehicles. For CO_2 engine size was a good indicator of emissions and there was a clear trend of increasing CO_2 with increased engine size. For NO_x no such trend was found and the variability was much larger. Across all vehicle types the highest CO_2 emission was 4 times the lowest, whereas the highest NO_x emission was over 4000 times the lowest. This research also highlighted the uncertainty in the air quality / climate change trade-off between petrol and diesel vehicles.

7.1.3 Use modelling to project and estimate the impact and risk associated with real world passenger car emissions and surrounding uncertainty

The first use of scenario analysis in this thesis was the modelling of 2030 Euro 6 NO_x emissions in Chapter 5. This was based on the first PEMS study relating to NO_x and NO_2 emissions. The scenarios (S1- S5) assumed varying NO_x emissions factors and proportion of primary NO_2. These ranged from S1 which assumed full Euro 6 conformity with the type approval limit to S5 which assumed no improvement from the current Euro 6 real world emissions. Each scenario also had an 'a' and 'b' component with varying percentage primary NO_2, 30% for 'a', 44% for 'b'.

The UKIAM was used to model the total Euro 6 NO_x emission in tonnes in the year 2030. The difference between the best and worst case scenarios made up a substantial proportion of the entire UK 2030 NO_x national emission ceiling. The most likely range was between 50.3 –102.9 kilotons, between 12 – 24% of the UK's 2030 NO_x ceiling. Using various damage costs this was estimated to cost between 0.95 – 1.92 Billion £.

The modelling found that unless Euro 6 real world emissions factors are reduced to at least the level of the Euro 6d- TEMP type approval (0.168 g km^{-1}), modelled by S2, there will still be roads at risk of exceeding the NO_2 air quality limit value in the year 2030. The number of grid squares containing roads at risk of exceedance varied from 0 for S1a to 136 for S5b. The fraction of NO_x emitted as NO_2 was found to have a significant impact on roadside concentrations. An increase in the proportion NO_2 from 30 to 44% could result in a national increase in roadside exceedances of the NO_2 air quality limit value of between 84 – 103%. In some grid squares it was found an increase in primary NO_2 from 30% to 44% resulted in an increase in ambient NO_2 concentrations of over 15%.

The second scenario analysis in the thesis is found at the end of Chapter 6. It was based on the second PEMS study and again modelled emissions in 2030, but this time including CO_2 from Euro 5 and petrol vehicles. The total emissions were calculated using emissions factors multiplied by kilometres driven, estimated by the UKIAM. These emissions were then multiplied by different combinations of damage costs for CO_2 and NO_x, ranging from low to high. There were 7 scenarios (S1 – S7), each assumed different fleet average emissions factors for petrol and diesel vehicles. Some scenarios also modelled shifts in the fleet composition due to the implementation of certain policies, such as a diesel scrappage scheme.

As expected the business as usual scenario, S1, which had no improvement from current real world emissions, had the highest damage costs. The scenario with the lowest damage costs (S7) assumed all vehicles met the CO_2 fleet average target of 130 g km^{-1} and Euro 6 diesel vehicles had real world NO_x emissions equal to the type approval limit (0.08 g km^{-1}). The difference in 2030 damage costs between the business and usual (S1) and best case (S7) scenarios was ~ 3 Billion £.

The second least damaging scenario S6 replaced all diesel vehicles with petrol-electric hybrids. These results showed that the best way to reduce environmental damage (even given the wide uncertainty in damage costs) is policies that reduce both CO_2 and NO_x. The current trend of diesel vehicles being replaced with petrol was not one of the lowest damage cost scenarios for any combination of damage costs, in fact when NO_x damage costs were low and CO_2 costs high this scenario was the most damaging.

7.1.4 Identify how air quality policies can tackle air pollution from passenger cars given the identified uncertainty

This section discusses the main policies being proposed as options to tackle passenger car emissions in the UK and the impact of the results in this thesis on these policies.

7.1.4.1 The EQUA index

A key finding was the huge variability in NO_x emissions from diesel vehicles of the same Euro standard. Until recently this was not well understood or communicated effectively to the general public. The EQUA index is an accreditation scheme devised by Emissions Analytics to inform consumers of the real world emissions of vehicles on market, it currently has ratings for ~1000 vehicles. An average emission is taken from a repeated sections of the urban test route. This measurement is then assigned a grade from A (<0.08 g km^{-1}) to H (>1.00 g km^{-1}). The grade for all 1000 vehicles is available on the EQUA index website and consumers can search by manufacturer, model and fuel type. There are also grades assigned for carbon monoxide, carbon dioxide and mpg.

The idea has been adopted by the Mayors of Paris and London who will launch and “Clean Vehicle Checker” in autumn 2017 using data from Emissions Analytics and the International Council on Clean Transport. These schemes will have the benefit of informing consumer choice and putting pressure on manufacturers to sell vehicles with low emissions in the real world, not just during type approval. The onus will now be on manufacturers to prove their green credentials in the real world.

7.1.4.2 Clean Air Zones / Low Emission Zones (LEZ)

The current air quality plan put forward by the UK government relies heavily on local councils setting up clean air zones. The benefit of these are they are relatively easy to implement based on number plate recognition, especially in London where the congestion charging zone is already in place. Clean air zones also have the benefit of being easy for the public to understand. Variability in RDE of diesel vehicles could hamper the effectiveness of LEZs.

The introduction of the RDE test procedure in September 2017 is an attempt to address this, although it will have no impact on the millions of vehicles already in circulation. Low Emission Zones, such as that being introduced in London 2019, will discriminate on the basis of Euro standard. The variability evidenced in this thesis shows that more discrimination based on real driving emissions could be beneficial, as illustrated in Chapter 4 when removing the worst Euro 6 diesels reduced the average emission by 35%. The ULEZ in London will allow diesel Euro 6, but not diesel Euro 5.

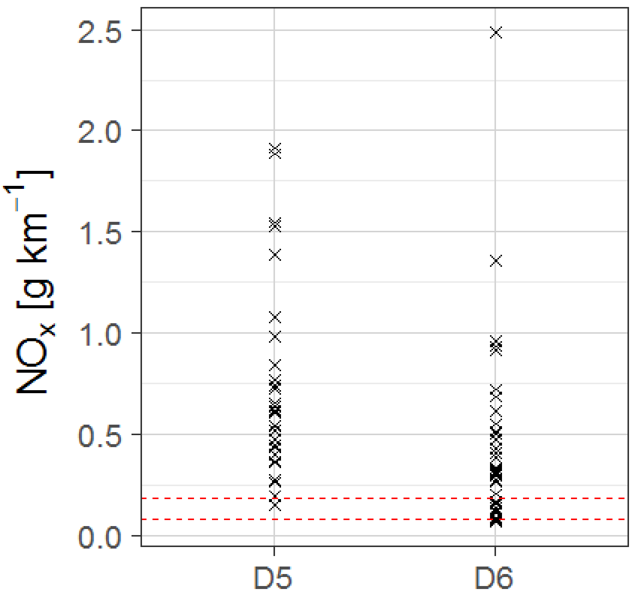

Figure 7-1. Urban NO_x emissions from Euro 5 and 6 diesels (type approval limits marked in red)

Figure 7-1 highlights the huge overlap between Euro 5 and 6. Some Euro 5 diesels had emissions much lower than some Euro 6 vehicles and vice versa. As illustrated in Chapter 4 the average emission of the fleet could be greatly reduced by focusing on RDE. The EQUA index is the first step in moving the discussion away from Euro standards and towards real world emissions.

7.1.4.3 Diesel scrappage scheme

All the modelling in this thesis assumed there would not be a national diesel scrappage scheme. A scrappage scheme was not included in the latest DEFRA air quality plan published July 2017. As public opinion shifts away from diesel and the market share declines there will be clear benefits for urban air quality. However, this shift will do nothing to address problems posed by the diesel passenger cars currently in circulation, which make up ~40% of UK passenger car fleet. This has led many (including the Mayor of London) to call for a National Diesel Scrappage Scheme to accelerate the turnover of the worst diesel vehicles (TfL, 2017).

A national diesel scrappage scheme could be effective at removing the worst polluting vehicles and accelerating vehicle turnover. Without a scrappage scheme the benefits of new type approval procedures will take longer to be realised, given the average age of a passenger car in Europe is 10 years (ACEA, 2017a).

The results in this thesis indicate that any proposed scrappage scheme should have a dual climate change focus (e.g. replacements must be hybrid or EV) as well as consideration of real world emissions. There is the potential to include a real world emissions component (similar to the EQUA index) to identify which vehicles to scrap first, especially if it is a targeted rather than "blanket" scrappage scheme. Focus on real world emissions would allow the scheme to be targeted initially at the most polluting vehicles, maximising the benefits. There may also be more benefit in scrapping newer vehicles with very high real world emissions, as they have a longer remaining lifespan. Whilst pre- Euro 4 cars (12 years old) emit more pollution they would most likely be retired much sooner, scrapping newer cars might prove unpopular but may in the long run be more cost effective. The worst diesel Euro 6 vehicles tested in this study had real world NO_x emissions ~3 times the Euro 4 limit.

A criticism often made of scrappage schemes is they are a subsidy for the middle classes, as the ~£2000 incentive is not enough for people on lower incomes to afford a brand new car. Another is that (as with the UK scrappage scheme in 2009) they provide a boost for the motor manufacture industry. Given the 'diesel gate' scandal and recent unscrupulous behaviour of certain companies this does not currently seem a particularly well earned reward, with the tax payer footing the bill.

7.1.4.4 Measures to disincentivise diesel

Tax incentives can be used to influence consumer behaviour. As discussed in Chapter 6 changes in vehicle excise duty (VED) in April 2017 coincided with a sharp reduction in the sales of new diesel cars. However the new VED puts road tax at £140 for petrol and diesel vehicles and £130 for alternative fuel vehicles (AFVs) does not go far enough to incentivise AFVs. The previous tax free incentive has been removed from hybrids and instead only applies to full electric vehicles. More work is needed but the results of this study indicate certain hybrids can deliver big emission reductions. Emissions from hybrid vehicles measured in this study had both CO_2 and NO_x emissions far below their diesel and petrol counterparts. Early evidence in the UK and across Europe indicates the majority of the decline in diesel sales is being matched by an increase in petrol sales. More thought should be done to incentivise hybrid vehicles instead.

Tax in the first year of the new VED is currently determined by CO_2 emissions. The higher the manufacturers stated CO_2 emission the higher the tax in the first year, with the highest band being £2000 for CO_2 > 255 g km^{-1}. This research found that for Euro 6 vehicles real world CO_2 emissions were 36% higher for diesel cars and 44% higher for petrol cars. None of the vehicles in the test fleet would be charged the maximum £2000 based on their official CO_2 emission. However, if the tax was based on RDE instead 9 petrol and 2 diesel vehicles would pay the maximum.

Another way in which consumer behaviour can be influenced is by the cost of fuel. If someone considering a switch from diesel to petrol was made aware that the petrol running costs are (on average) 44% higher than advertised they may choose to purchase a hybrid or electric vehicle instead. Additionally the tax on diesel fuel could be increased to account for the higher damage cost per km. An increase in the cost of diesel fuel would remove the largest consumer incentive, fuel economy.

7.2 Limitations

As discussed in Chapter 2 a limitation of PEMS studies is always the sample size. PEMS equipment is expensive and testing is time consuming, this is why PEMS and remote sensing studies should be used to complement one another. There was also a lack of repetition in the testing. Real driving emissions depend on a lot of external factors such as wind speed and temperature as well as natural variability in the testing itself. The results would be more robust had each vehicle been tested numerous times in different conditions to compare for continuity. Unfortunately the vehicles used in the testing were often only loaned for one day, so this was not possible with the current Emissions Analytics set up. It would also have been interesting to test the same vehicles on a chassis dynamometer for comparison.

It is thought much of the variability in NO_x emissions comes from thermal windows and defeat devices. PEMS do not include detailed analysis of the on-board software, this would allow identification of when NO_x controls were operating and at what efficiency.

The omission of measurements of particulate matter and non- exhaust emissions is a major limitation of this work, especially when looking at the Gasoline Direct Injection vehicles. With the decrease in exhaust emissions the non-exhaust components are becoming an ever bigger fraction of PM. This will still be a concern for electric vehicles.

As the topic of this thesis was exhaust emissions from passenger cars the policy measures discussed throughout relate only to emissions reduction. It is extremely important to add that the most effective pollution reduction measure is the removal of the car altogether. Promotion of cycling and walking should always be a priority for policy makers. As well as reducing air pollution the co-benefits associated with exercise improve health, happiness and wellness and ultimately could save the NHS billions of pounds per year. Almost half of all car journeys in the UK are less than 5 km in length, many of these could easily be cycled instead or replaced with public transport. Additionally, better city planning and land use has potential to greatly reduce the amount of vehicle kilometres driven by building new residential settlements within walking/ cycling distance of local amenities and schools.

Finally, this work has coincided with enormous changes in the field of air quality, with air pollution becoming part of the public consciousness in a way it hasn't perhaps since the Great Smog. The final year of this work also coincided with a period of political instability in the United Kingdom. The combination of these two circumstances mean air quality policy is constantly changing and developing in ways that make it difficult to judge which decisions future governments will make. All the regulations referenced in this thesis are from the EU, it is not clear how the UK will develop its own air quality regulations after leaving the European Union in 2019.

7.3 Recommendations

There should be extended testing of petrol- electric hybrids, to include vehicles made by a number of different manufacturers. PEMS testing should also be extended to include other types of hybrid vehicles such as plug-in hybrids. A dedicated PEMS study (with sufficient sample size) should be performed to measure emissions during cold starts for petrol, diesel and hybrid vehicles. For comparison there should be continuity between the route followed and speed in all tests.

There should be extensive PEMS testing of vehicles that pass the new real driving type approval test from September 2017 onwards to ascertain what (if any) will be the increase between type approval and real world emissions. There should also be moves to ensure type approval becomes more representative by including cold starts and not allowing manufacturers to choose the ambient temperature on the day of the test.

Lastly more research is required into the real world emissions of light goods vehicles and vans. The NAEI estimate in 2014 they made up 6% of total UK NO_x emissions and with the rise of home delivery and online shopping the miles driven by LGVs are also rising. Many vans use the same engines found in the cars tested in this study but in much heavier vehicles with much heavier loads. This is likely to result in much higher emissions then recorded for passenger cars.

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Appendix 1: HAZOP analysis of the UKIAM

The tables below summarise the full preliminary HAZOP analysis of the UKIAM. This initial HAZOP review provided a framework which could be continually updated, and provided a record of how uncertainties were addressed. This was put together by Rosalind O'Driscoll, Helen ApSimon and Tim Oxley as part of the DEFRA Support for National Air Pollution Strategies (SNAPS) Working Package 1.

Emission projections:

Definition of emissions of specified pollutants (SO_2, NO_x, NH_3, PM_{10}, $PM_{2.5}$, VOCs) for future scenarios up to 2030, for use with the atmospheric modelling module, and in the module defining potential abatement measures. Depend on projected activity data and emission factors as emission per unit activity.

Source of uncertainty	Comment	Action
UK emissions : Activity data (sources within SNAP sectors)	Based on national projections from DECC, DfT, agricultural scenarios etc Or from independent studies e.g. CCC energy and agriculture scenarios	
MORE/LESS	e.g apply to different source/ technology	Variant scenarios including high and low estimates of activity data e.g. as with UEP45 projections

NOT		Sensitivity studies e.g to type of domestic stove for biomass in CCC study
Emission factors		
MORE/LESS	Where possible adopt NAEI emission factors. May be based on legislation rather than specifying technology/ abatement measures in place	Comparison with GAINS Reference to NAEI reports Sensitivity studies, Monte Carlo analysis
NOT	New technologies not covered in NAEI with EFs taken from literature (e.g CCS) or consultation (e.g. CHP plants/biomass)	Specify source of value adopted e.g. in RAPID database, and undertake source apportionment to see how significant source is for model results + conclusions
AS WELL AS	Missing sources e.g. anaerobic digesters in UEP45.	More likely to occur with new activities

	Missing pollutant e.g primary NO2, or black carbon Or more detailed chemical speciation e.g. VOCs	Add pollutant by estimating relative to related pollutant by factors/fractions for most important sources until full inventory available e.g. as for black C in UKIAM
UK road transport emissions (BRUTAL) MORE/LESS (Traffic flows, Traffic mixes, Speed) Emission factors MORE/LESS Emission factors NOT	Modelled on a road by road basis across the UK network From DfT data, road type etc Same as NAEI Problems of Euro standards not delivering expected improvements in real world e.g.	Sensitivity studies, model validation studies e.g. MIE (Comparison with Kings observations- not funded) Sensitivity studies using MIE software NB Kings data from exhaust scanning As above, plus sensitivity studies using GAINS emissions

	Euro 6. NB difference between GAINS and NAEI Other associated sources : tyres and brakes covered but not road abrasion or resuspension	
AS WELL AS	Alternative vehicle technologies	
Shipping emissions	Based on AMEC/ENTEC projections for N Sea and local sea areas plus GAINS for rest. NB based on different assumptions and ship data	? data available from Ricardo-AEA (& Kings) ????
European emissions other countries	Based on GAINS scenarios	

Other aspects: (AS WELL AS)		Note underlying assumptions re projections and include full range of alternative scenarios (e.g. based on national projections)
Spatial mapping	UK emissions on 1x1 km grid plus point sources plus roads. Change over time e.g. power plants.	Currently being updated based on NAEI source footprints & remodelling of roads to include NI. Test atmospheric model sensitivity to grid resolution
Temporal variation	Only annual emissions. Hence uncertainties re episodes or diurnal variations, and with non-linear chemistry	Supporting atmospheric modelling studies (if funded)

Atmospheric modelling:

Purpose- to estimate contributions of different sources to primary and secondary pollutant concentrations, and to sulphur and nitrogen deposition (+ ozone fluxes and PODs in future development). This is used to estimate environmental impacts in response to changes in emissions. The focus here in WP1 is on S-R relationships because other work is being undertaken on uncertainties in atmospheric modelling by CEH, including EMEP4UK as well as FRAME.

Note that UKIAM provides a framework for swapping source-receptor relationships from different models. There is also the potential to use the Model Intercomparison Exercise, MIE software.

Source of uncertainty	Comment	Action
NOx/NO_2 concentrations. BRUTAL model	Emphasis on urban areas. Imported contribution small	

Background NOx MORE/LESS	Background concentrations of annual average in 1x 1 km grid squares. Dispersion based on PPM Gaussian model; detailed treatment of traffic contribution	Participation in MIE, to be repeated with updated emissions
Roadside NOx MORE/LESS	Simple roadside increment to allow for restricted dispersion by buildings and street canyons, depending on urban (population) density	Derived from sensitivity studies with ADMS street canyon model. ? compare with PCM empirical formulae.
NO_2 concentrations MORE/LESS	Derived from NOx using quadratic relationship between annual average NO_2 and NO_x; parameters depend on rural, urban, roadside site characteristics.	Special study by R O'Driscoll. Sensitivity study to model parameters defining quadratic equation. Comparison with Jenkins and Clapp shows good

Exceedance of NO_2 limit value (40 μgm^{-3})	Identified as priority for investigation in MIE, and evidence of overestimation at road side sites Also NB fraction of NO_x emitted as primary NO_2- see emissions	agreement. Focus on roadside using diffusion model for profiles of NO, NO_2 and O_3 near roads to develop parameterisation used in BRUTAL
a) Background MORE/LESS	Background based on 1x1km grid	MIE comparison with more detailed spatial modelling
b) Road-side MORE/NOT	Road lengths at risk of exceedance may be overestimated. BRUTAL takes worst road in each grid square, and if limit	Make clear BRUTAL identifies grid squares including road-side exceedance, and requiring more detailed spatial modelling, rather than total road length of exceedance.

	exceeded assumes all roads in square at risk. Sensitivity to threshold limit value	Estimate exceedance relative to higher and lower thresholds.
Concentrations of PM_{10}/$PM_{2.5}$ Primary PM Secondary SO_4,NO_3 and NH_4	Built up from several components Similar uncertainties to NO_x concentrations based on BRUTAL model. Small imported contributions based on EMEP modelling. Based on EMEP and FRAME models	Swap models and compare (as in previous contract)

	NB Have calibrated and uncalibrated versions for FRAME. But hybrid of EMEP and FRAME avoids need for calibration.	Also comparison with PCM maps interpolated from measurements (which do not show as much spatial variability e.g in NO_3 aerosol due to NH3 availability)
Imported contributions	Currently use EMEP model for imported contributions.	Recently revised data from IIASA-> comparisons with previous model when incorporated in UKIAM
LESS THAN	FRAME model- constant drizzle leads to underestimate.	Refinement of FRAME model to dry and wet periods

National contributions	Use FRAME uncalibrated as standard model. Gives better spatial mapping than EMEP	Scale average to match EMEP.
MORE THAN	Constant cloud leads to excess SO_4 production in FRAME.	Refinement of FRAME to have intermittent cloud
Non- linearity / chemistry: MORE/LESS	Depends on how big a change is made from scenario used to derive S-Rs Fraction of NO_3 aerosol as $PM_{2.5}$	Comparisons of different models undertaken in SSNIP. Sensitivity of total to fraction generally small
Other components of PM :AS WELL AS	Kept constant	
Secondary organic aerosol	Currently from NAME model	Comparison with HARM. Future comparison with EMEP4UK

Water	Based on EMEP. Uncertain how much this reduces in response to changes in SIA	?????
Urban and rural dust	Adopted from PCM modelling.	
AS WELL AS: Temporal	Legislation for PM_{10} based on episode days, (related statistically to annual average in UKIAM as in PCM). Episodes may behave in a different way to abatement strategies from the annual average e.g. episodes of high SIA coinciding with peaks in agricultural NH3 in air from continent.	Further work in this area by Ricardo-AEA? NB Inter-annual variability. Evidence of effect from PCM modelling. Proposed work with NAME to look at this but not funded

Spatial	Primary PM background on 1x1 km grid as for NO_x/NO_2 Patchy effect of NH_3 emissions on SIA	Sensitivity to grid resolution of models
$PM_{2.5}$ and other size fractions of PM_{10} NOT	Primary emissions as fraction of PM_{10} emissions. Assumptions about size distributions. No microphysics for particle processes like coagulation.	Model would not be applicable to smaller particles and ultra-fines.
Chemical composition of PM	Can be derived from source apportionment	See section on health impacts
Sulphur and nitrogen deposition		

Deposition MORE/LESS		
Wet deposition	Constant drizzle in FRAME will deplete imported fluxes leading to underestimation of wet deposition – so use EMEP imported fluxes	May underestimate wet deposition due to imported fractions. Compare with new FRAME with dry and wet periods , as compared with constant drizzle version
Orographic enhancement wet deposition	Difference clearly indicated between FRAME which includes it, and EMEP which does not, Currently assume the EMEP imported fluxes with FRAME spatial distribution.	

Dependence on chemistry	Non-linear chemistry effects on deposition appear a bit less critical than for SO4 and NO3 concentrations, but may affect range	Comparisons EMEP and FRAME. Sensitivity of source apportionment and UK versus imported contributions
Dry deposition	Dependent on assumed deposition velocities which vary with type of land use/cover. No allowance for co-deposition of NH_3 and SO_2	Differentiate contributions from dry and wet deposition, and effect of say 20% increase or decrease in dry portion (will be smaller for NOx deposition)
Occult deposition NOT	Deposition direct from cloud is altitude dependent and is not considered separately from orographic enhancement	Could lead to underestimation of deposition over hills and mountains

Other considerations AS WELL AS		
Spatial	Currently mapped at 5x5 km resolution. Cannot resolve local scale deposition of NH3 round sources with this resolution. Big uncertainties in dry deposition from emissions within local square	Exploratory runs to investigate contribution from dry deposition of NH_3, and role in exceedance of critical loads. Where this is dominated by local square emissions of NH3 local measures may be appropriate.
Temporal	A large proportion of wet deposition occurs in a few concentrated episodes. There are also large inter-annual variations	Compare different years where separate data is available.

Environmental impacts and effects

Purpose: calculation of indicators to quantify environmental impacts and as input to environmental damage costs and CBA.

Impacts on health are derived from population weighted means of atmospheric concentrations of fine particulates (generally $PM_{2.5}$ but also calculated for PM_{10} as that has been used in the Defra damage costs) and NO_2. These are derived from population data and concentrations on the 1x1 km grid, with no contribution from road side increments in exposure. Can be broken down into London or urban v rural, and different regions.

Source of uncertainty	Comment	Action
PM, NO_2 exposure MORE/LESS (also see tables on emissions and concentrations)	Assumes static population with no allowance for travel or indoor exposure	Compare with epidemiological estimates of exposure

Other considerations AS WELL AS		
Differential toxicity between components of PM	Health impacts based on total PM mass eg effects of SIA are same as primary diesel exhaust per unit mass	Break down exposure into different source components . This clarifies for example reductions in exposure to particles in diesel vehicle exhausts as compared with brakes and tyres, or primary versus secondary PM
Uncertainty in direct health impacts of NO_2	There is no agreed relationship between NO_2 exposure and health impacts. Hence impacts of NOx are currently based on exposure to secondary nitrate particles which is a long-range as opposed to a local exposure impact. However tentative	Explore use of HRAPIE proposals and compare with PM based on NO3.

	relationships have been proposed by HRAPIE although not yet accepted.	
Ozone NOT	Ozone is not yet considered although there are plans to do so based on SOMO35 using S-R data from IIASA until data available from EMEP4UK. NB need to allow for urban deficit of ozone which can be derived from NO2 v NOx modelling	Current underestimation of health impacts due to omission. Remedied by model extension.

Protection of ecosystems with respect to acidification and eutrophication is assessed by comparing deposition with the respective critical loads. Where there is no exceedance protection is assumed. For acidification critical loads are determined such that there is no net change in soil acidity. For eutrophication critical loads are empirical and separate data is provided for total ecosystem areas broken down by habitat, and for SSSIs. In the latter case upper and lower bounds for critical loads for each habitat are available from CEH. The criterion for protection is very sensitive to small changes in deposition as well as the critical loads. Large changes in deposition can have relatively little effect on % ecosystems protected unless in the vicinity of the critical load itself, so that areas or %s protected are insensitive indicators of protection. For this reason accumulated exceedance above the critical loads is also calculated as an indicator.

NB No impacts on crops are currently included although there are plans to do so for wheat based on EMEP/IIASA data for POD_6. In the longer term this could be extended to forests.

Source of uncertainty	Comment	Action
Critical loads MORE/LESS (re deposition see previous table on uncertainties)	Sensitivity to critical loads. Since eutrophication is the more difficult problem more emphasis is placed on this.	For SSSIs a different approach has been developed based on different classes of risk re exceedance. This reflects the uncertainty range for the critical load for each habitat, and the ratio of the deposition to the upper limit of critical load where this is exceeded.
Other considerations AS WELL AS		
Spatial	For SSSIs the location of the sensitive ecosystems within the SSSI area is not specified, and so it is assumed that any part of the SSSI area may be sensitive.	May result in pessimistic bias in estimation of protection of sensitive habitats

	Where more than one habitat is present in an SSSI the focus is on the most sensitive.	
NB also comments on small scale variability of NH3 deposition	Limitations of model close to sources of NH3 emissions due to local dry re-deposition of NH3	Examine effect of removing NH dry deposition, which is the component that can be altered most by local measures.
Temporal	Estimated exceedance in a given year gives no indication of temporal effects and accumulated deposition history over time. Nor does it address recovery of damaged areas. More complex dynamic modelling has been developed to address these temporal issues	Explore the use of target loads in order to achieve recovery by a set date derived from dynamic models. Such target loads are smaller than critical loads and more difficult to achieve. Has been tried in past work for freshwater systems, and recovery times may be too long for other systems.

Abatement options and costs

Purpose – for use in deriving potential abatement scenarios to investigate cost effective strategies for reduction of emissions and improved environmental protection. The main source used is the Multi-Pollutant Measures Database, MPMD, provided by AMEC. For each measure the applicability, efficiency and annualised cost per ton reduced are defined; and where possible implications for greenhouse gas emissions are given too. Care has to be taken in combining measures distinguishing incremental measures, alternative measures, and additional measures; in some cases a measure may involve coupled changes in more than one sector, e.g. electric cars. Some of the measures are add-on technical measures or involve changes in technology; whereas others imply changes in activity data (affecting, for example, energy projections) or behavioural change (e,g, eco driving).

Alternative data are abatement measures in the GAINS data base. These are mainly add-on measures although direct comparison with the MPMD is not necessarily straightforward.

Source of uncertainty	Comment	Action
Applicability and efficiency MORE/LESS NB Also refer to tables on uncertainty in emissions	These depend on the assumed technology and abatement in place which is not necessarily defined in the NAEI.	Uncertainty reduced by close collaboration between AMEC, Ricardo AEA and Aether in the NAEI.
Costs	Uncertainties vary greatly between measures, and may reduce over time for new technologies. Can be difficult to define for behavioural change.	Sensitivity analysis for measures with major effect on emissions.
Costs negative NOT	In some cases costs are negative, although may be treated as zero in UKIAM. Care needed with co-benefits to	Note negative costs and any identified barriers to implementation.

	avoid overlap with benefits assessment and double counting in CBA.	
Measures not included NOT	This can be a special problem in applications of the RAPID model, where new technologies such as CCS are specified not specifically targeted at air quality pollutants.	Improve RAPID database including consultation with Imperial colleagues.
New technologies	The role of new technologies may not be clearly defined in the baseline scenario, or in enough detail eg CHP, biomass combustion, or energy from waste. NB Task 1.5.7	

Other considerations: AS WELL AS	Annualised costs do not reflect factors like lifetimes before closure of plant if retrofitting. This is a limitation of the snap-shot in time for a given year. There may be other factors affecting uptake of measures as well as costs.	Critical review of abatement scenarios analysed, and comparison of different target years e.g. 2025 and 2030 Look back at work by N Hasnain on stakeholder considerations in her PhD. Also other work on barriers to implementation such as benefits not accruing to those bearing costs.

NB. New data is being added to the MPMD for measures to reduce agricultural emissions of ammonia which will require specific attention with respect to uncertainties

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Figure 2-2. (a) Trends in the mean concentration of NO_x across 35 roadside sites in Greater London with at least 10 years of data capture and (b) the same of NO_2

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Figure 2-7. Diagram of PEMS

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