School of Engineering

Investigating the Impacts of Increased Uptake of Electric Vehicles on Air Quality and Health

Saad Almutairi

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Abstract

Globally, nine million deaths per year are attributed to exposure to air pollution, as estimated by the Lancet Commission on Pollution and Health (Landrigan *et al.*, 2018). In the UK, approximately 40,000 deaths per annum are attributed to exposure to $PM_{2.5}$ and NO_2, costing society nearly £20 billion annually from the health-related consequences of people suffering diseases and early deaths (Royal College of Physicians, 2016). Road transport emissions are a major source of air contaminants, and in 2016 they contributed to 12.4% of $PM_{2.5}$, 11.7% of PM_{10} and 33.6% of NO_x (DEFRA, 2018); the latter contributing 80% of NO_2 concentrations at roadsides (DEFRA and DfT, 2017a). Additionally, vehicular emissions account for 24% of greenhouse gas (GHG) emissions (BEIS, 2018a).

To mitigate air quality pollutants and GHGs, the UK government's Road to Zero strategy plans to limit the sale of new cars and vans to ultra-low emissions vehicles (ULEVs), mainly focusing on electric vehicles (EVs), by 2040 with the aim of forming an entire stock of ULEVs by 2050. Currently, the government is investing £1.5 billion in measures dedicated to increasing the penetration of ULEVs and optimising their manufacturing and infrastructure. These measures would result in changes in the vehicle fleet mix and consequently reductions in emissions and pollutant concentrations. A detailed investigation is needed to quantify their impact. In this research, the impact of changes in the vehicle fleet with the increased adoption of EV, on air quality and health was investigated via scenarios that consider different levels of future EV uptake replacing conventional vehicles in Newcastle and Gateshead.

Road transport network data for 2010 for the study area was acquired and updated to provide the 2014 Baseline, considering traffic growth for each vehicle class. The Baseline traffic model was validated following the Design Manual for Roads and Bridges criteria. The resulting emissions rates were calculated using an emissions model. The dispersion of pollutants was modelled taking into consideration the effect of meteorological factors. The air quality model was validated following DEFRA Technical Guidance.

The 2014 Baseline traffic was updated to business-as-usual (BAU) for 2030. Six future scenarios were developed based on this BAU. These scenarios include:

1. 'CCC': Committee on Climate Change proposal for 30% of cars and 38% of vans being electric;

2. 'E-Bus': electrification of all buses;

3. 'E-Car': electrification of all cars;

4. 'E-Car_E-Bus': electrification of all cars and buses;

5. 'E-Car_E-LGV': electrification of all cars and LGVs; and

6. 'All-EV': electrification of all vehicles.

Emission and dispersion models were applied to determine changes in air quality in response to the BAU and the six scenarios. The results indicate that pollution concentrations in 2030 would be reduced to varying extents compared to the 2014 Baseline. The annual mean reductions at the 66 General Practitioner (GP) sites were averaged for all 2030 scenarios across the study and showed a drop of 8 $\mu g/m^3$ in NO_2 levels and 3 $\mu g/m^3$ in PM_{10} and $PM_{2.5}$ levels.

The Department of Health recommended dose-response coefficients, which describe the association between exposure to a certain amount of pollutants and the probabilities of being admitted to hospital and early mortality, were applied to the pollutant reductions at each GP site to estimate the number of respiratory hospital admissions at each GP location. Disease burden estimates suggest that the 2030 BAU will reduce hospital admissions by 1,297, representing 13% of the 9,693 cases recorded in 2014. It was noted that a large reduction in hospital admissions would occur due to decreases in NO_2 concentrations. In the All-EV

scenario, hospital admissions are expected to be reduced by 1,377, which could also nearly be achieved either by electrifying all cars and all buses or electrifying all cars and LGVs with a lower cost in relation to All-EV. Reducing premature mortality is estimated to account for 14 to 16 incidents. This study shows that the EV uptake scenarios will result in significant reductions in air pollution emissions and concentrations and consequent hospital admissions compared to BAU taking into consideration the relatively small population of Newcastle and Gateshead.

Declaration

The content of this thesis is solely my responsibility, and the original work herein is my own, except where specified otherwise in the text. Neither the thesis nor any of the original work included it has been submitted to this or any other institution for consideration for a higher degree.

Signed……………………………

Date..……December 2020…..

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Glossary of Terms

Term	Definition
AADT	Annual Average Daily Traffic
ADMS-Urban	Atmospheric Dispersion Modelling System
AIRVIRO	Air quality dispersion modelling system
All-EV	Electrification of all vehicles scenario
ALOHA	Areal Locations of Hazardous Atmospheres
AM	Morning Peak
ANPR	Automatic Number Plate Recognition
AQD	Ambient Air Quality Directive
AQMA	Air Quality Management Area
AURN	Automatic Urban Rural Network
BAU	Business as Usual
BEIS	Department for Business, Energy and Industrial Strategy
BEVs	Battery Electric Vehicles
BLP	Buoyant Line and Point Source Dispersion Model
CAFE	Clean Air For Europe
CALINE	California Line Source Dispersion Model
CAZ	Clean Air Zones Plan
CCC	Committee on Climate Change scenario
CNG	Compressed Natural Gas
CO	Carbon monoxide
CO_2	Carbon dioxide
COMEAP	Committee on the Medical Effects of Air Pollutants
COPD	Chronic obstructive pulmonary disease
COPERT	COmputer Program for Emissions from Road Transport
CRF(s)	Concentration-response function(s)
CTDM	Complex Terrain Dispersion Model
CV	Conventional Vehicle
DEFRA	Department for Environment, Food and Rural Affairs
DfT	Department for Transport
DMRB	Design Manual for Roads and Bridges
DOC	Diesel Oxidation Catalyst
E-Bus	Electrification of all buses scenario
E-Car	Electrification of all cars scenario
E-Car_E-Bus	Electrification of all cars and buses scenario
E-Car_E-LGV	Electrification of all cars and LGVs scenario
EEA	European Environment Agency
EFT	Emissions Factors Toolkit
EGR	Exhaust Gas Recirculation
EMFAC	EMission FACtor
E-REV	Extended-Range Electric Vehicle
ERG	Exhaust Gas Recirculation

Term	Definition
EU	European Union
Euro Standard	Refers to the emissions standards enforced in the European Union
EV(s)	Electric Vehicle(s)
EXPLOLIS	Air Pollution Exposure Distribution within Adult Urban Populations in Europe
FORGE	Fitting on Regional Growth Effects
GDP	Gross Domestic Product
GEH	Geoffrey Edward Havers index
GHG(s)	Greenhouse Gas(es)
GP(s)	General Practitioner(s)
HDV(s)	Heavy Duty Vehicle(s)
HEV(s)	Hybrid Electric Vehicle(s)
HGV(s)	Heavy Goods Vehicle(s)
ICE	Internal Combustion Engine
ICEV	Internal Combustion Engine Vehicle
I_e	Exposed population
IER	Integrated exposure-response
IGT	Information Governance Toolkit
IP	Inter Peak
ISC	Industrial Source Complex Model
I_u	Unexposed population
Kt	Kilotonne
LDV(s)	Light Duty Vehicle(s)
LGV(s)	Light Goods Vehicle(s)
LNT	Lean NO_x Traps
LPG	Liquid Petroleum Gas
LSOA(s)	Lower super-output area(s)
MFM	Multi-model Freight Model
MODEM	MODelling EMissions and fuel consumption in urban areas
MSOA(s)	Middle Super Output Area(s)
$MtCO_2e$	Million tonnes of carbon dioxide equivalent
NAEI	National Atmospheric Emissions Inventory
NECD	National Emissions Ceiling Directive
NEDC	New European Driving Cycle
NGHGI	National Green House Gas Inventory
NO	Nitrogen monoxide
NO_2	Nitrogen dioxide
NO_x	Nitrogen oxides, generic name for nitrogen monoxide and nitrogen dioxide
NTEM	National Trip End Model
NTM	The National Transport Model
NTS	National Transport Survey

Term	Definition
O_3	Ozone
OBR	Office for Budget Responsibility
OCD	Offshore and Coastal Dispersion
OEM	Original Equipment Manufacturer
OLEV	Office for Low Emission Vehicles
ONS	Office for National Statistics
OP	Off Peak
PCU	Passenger Car Unit
PEMS	Portable Emissions Measurement System
PHEM	Passenger car and Heavy-duty Emission Model
PHEV(s)	Plug-in Hybrid Electric Vehicle(s)
PITHEM	Platform Integrated for Transport, Health and Emissions Modelling
PM	Evening Peak
PM_{10}	Particulates with aerodynamic diameter $\leq$ 10 µm
$PM_{2.5}$	Particulates with aerodynamic diameter $\leq$ 2.5 µm
Ppb	Parts per billion
R	Correlation coefficient
R^2	Coefficient of determination
RDE	Real-driving emissions
RMSE	Root Mean Square Error
RR(s)	Relative risk(s)
RSD	Remote sensing detectors
RTFs	Regional Traffic Growth and Speed Forecasts
SCR	Selective Catalytic Reduction
SMMT	Society of Motor Manufacturers and Traders
SO_2	Sulphur dioxide
TADU	Tyne and Wear Accident Data Unit
TAG	Technical Analysis Guidance
TEE	Traffic Emissions and Energy
TEMPro	Trip End Model Presentation Program
TEOM	Tapered Element Oscillating Microbalance
TG16	Local Air Quality Management Technical Guidance
TPM	Transport Planning Model for 2010
TRL	Transport Research Laboratory
UKCES	UK Commission for Employment and Skill
UKEF	UK emission factors
ULEV	Ultra-low emissions vehicles
VERSIT+	Vehicle Emission factors and Standard calculation method
VKT	Vehicle-Kilometres Travelled
VOC	Volatile organic compounds
WebTAG	Web-based Transport Analysis Guidance
WHO	World Health Organization

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CHAPTER 1

1. Introduction to the Research

1.1 Introduction

This chapter presents the background and motivation for the study and the novelty of the research. The aim and objectives are then described, followed by the structure of the thesis.

1.2 Background and Motivation for the Study

Human health is at risk from serious diseases associated with exposure to air pollution, such as increased concentrations of fine particulate matter ($PM_{2.5}$: particulates with aerodynamic diameter ≤ 2.5 µm), coarse particulate matter (PM_{10}: particulates with aerodynamic diameter ≤ 10 µm) and nitrogen dioxide (NO_2) levels. In 2015, air pollution contributed to the premature deaths of nine million people, forming nearly a sixth of total deaths globally, which is 15 times greater than the number of deaths attributed to all wars and other violence, and triple the number of deaths related to AIDS, tuberculosis and malaria, as estimated by the Lancet Commission on Pollution and Health (Landrigan *et al.*, 2018). Another estimate made by the World Health Organization revealed that 90% of the world's inhabitants breathe contaminated air; this caused the premature death of 4.2 million people due to outdoor pollution and 3.8 million due to indoor air pollution in 2016 (WHO, 2018c). Additionally, air pollution significantly contributes to 24% of deaths worldwide resulting from heart disease, 25% from strokes, 43% from chronic obstructive pulmonary disease (COPD), and 29% from lung cancer (WHO, 2018c). In 28 European countries in 2014, it was estimated that 399,000 premature deaths could be attributed to $PM_{2.5}$ exposure, 75,000 to NO_2, and 13,600 to Ozone (O_3) exposure (EEA, 2017). Air pollution in European cities has been associated with increasing numbers of admissions to hospital for respiratory diseases (Halonen *et al.*, 2008; Sauerzapf *et al.*, 2009; Belleudi *et al.*, 2010; Cirera *et al.*, 2012; DeVries *et al.*, 2017). The Royal College of Physicians (2016) suggested that combined exposure to both $PM_{2.5}$ and NO_2 causes approximately 40,000 early deaths per annum in the UK. Although complex to quantify, it is estimated that air pollution costs society nearly £20 billion annually, due to the health-related consequences of people suffering disease and early death (Royal College of Physicians, 2016). Different estimates in relation to the consequences of air pollution on health are not consistent and the lack of general agreement might be attributed to differences in the methodology used. For example, it is estimated that, without the policies outlined in the Clean Air Strategy 2019 to mitigate air pollution, the annual cost of the population's health impact would be £1.7 billion and £5.3 billion by 2020 and 2030 respectively (DEFRA, 2019, p. 98).

In the same context, exposure to motorised road emissions significantly contributes to increased mortality and morbidity. For example, lung cancer, respiratory infection and cardiovascular illnesses are associated with short- and long-term exposure to road-transport related air pollution (Künzli *et al.*, 2000; Tonne *et al.*, 2007; Beelen *et al.*, 2008; Chatterton, 2011; Gan *et al.*, 2011; Hoek *et al.*, 2013; Raaschou-Nielsen *et al.*, 2013b; Hitchcock *et al.*, 2014, p. 8; WHO, 2014; Hamra *et al.*, 2015; Stieb *et al.*, 2016; Howell *et al.*, 2018). Furthermore, some burdens are associated with motorised pollution, such as high blood pressure (de Paula Santos *et al.*, 2005; Weichenthal *et al.*, 2014), diabetes (Krämer *et al.*, 2010), and dementia (Chen *et al.*, 2017b).

The Department for the Environment, Food and Rural Affairs (DEFRA, 2019b, pp. 10, 19) reported the contribution of road transport to air pollution in 2017. This is estimated at 12% of total $PM_{2.5}$, coarse particulate matter (PM_{10}: particulates with aerodynamic diameter ≤ 10μm diameter), and 32% of total nitrogen oxides (NO_x) (generic name for nitrogen monoxide (NO) and NO_2) which contribute 80% of NO_2 concentrations at roadsides (DEFRA and DfT, 2017a, p. 30; DEFRA, 2019a, p. 45).

Road transport is not only the primary source of air pollution, but also the second largest source of greenhouse gas (GHG) emissions in the European Union (EU) (Serradilla *et al.*, 2017), and the largest emissions of GHGs in the UK (BEIS, 2018b, p. 8). The transport sector's contribution to GHG emissions increased from 16% in 1990 to 27% in 2016. The transport sector was responsible for 125.3 $MtCO_2e$ in 1990 and 124.4 $MtCO_2e$ in 2016 of GHG emissions (BEIS, 2018b, p. 8), where road transport accounted for 90% of those emissions (ibid.). Furthermore, GHG emissions released by light goods vehicles (LGVs) increased by 65% in 2016 compared to the 1990 level (BEIS, 2018a). Success in maintaining nearly the same level of GHG emissions in 2016 compared to 1990 is likely to be attributable to improvements in vehicle technology, as they are generally cleaner than in 1990 given that a vast growth in vehicle numbers in the UK occurred during this period. This was due to developments in road networks and infrastructure, the expansion in vehicle production and access to free navigation systems, given that passenger cars are a convenient mode of transport (Steg, 2003).

Regarding the mitigation of air pollution and GHG emissions, sales of new cars and vans will be limited to ultra-low emissions vehicles (ULEVs) by 2040, with the aim for ULEVs to form the entire car stock by 2050. Currently, the UK government is investing £1.5 billion in measures dedicated to increasing the penetration of ULEVs, and the optimisation of their manufacturing and infrastructure, as set out in the strategies of the Automotive Sector Deal, NO_2 Plan, Clean Growth Plan, the Road to Zero (DfT, 2018a, p. 10) and most recently the Clean Air Strategy 2019 (DEFRA, 2019a, p. 40).

Carbon dioxide (CO_2) is the most dominant GHG. European emissions standards known as 'Euro' standards, have established limits on emissions produced by new vehicles since 1992 (European Commission, 1991), and regulations that set binding targets to reduce average CO_2 emissions released by new cars and vans were created in 2009 and 2011 (European Commission, 2009; European Commission, 2011b). These regulations set average targets for reducing the CO_2 emissions released by new cars to 130 g/km by 2015 and those released by new vans to 175 g/km by 2017. Fortunately, the required reductions were attained in the UK, and levels of CO_2 emissions are now under these thresholds (SMMT, 2018). Further targets related to average CO_2 emissions of 147 g/km by 2020 were set for vans, with a target of 95 g/km for cars by 2021 (European Commission, 2009; European Commission, 2011b). However, delivering these targets, which the UK is not compelled to achieve, requires annual reductions in CO_2 emissions of 6% until 2021 for cars, and 4% until 2020 for vans (SMMT, 2018, pp. 7, 17). These targets might be accomplished through the adoption of vehicles equipped with alternative technologies.

Sales of vehicles in Europe are subjected to Euro standards which have had specified thresholds on exhaust emissions since the early 1990s. Euro 1 (symbolised using Arabic numerals) is the first standard to control emissions from cars and light goods vehicles (LGVs), whilst Euro I (symbolised using Roman numerals) is the first standard that regulates a maximum limit for emissions released by trucks and buses. Emissions of NO_x from all diesel vehicles in the Euro 6/VI legislation have been reduced significantly to approximately half compared to previous standards, as Euro standards that legalise limits regarding vehicle emissions are becoming stricter with every progressive directive (European Commission, 2011a; European Commission, 2012). However, emissions released in real-world driving conditions have not decreased through the Euro 1/I –5/V emission standards, particularly for diesel vehicles (Moody and Tate, 2017; Dey *et al.*, 2018a). Regrettably, the sixth Euro diesel models were discovered to have had their NO_x emissions manipulated in the UK. An independent committee commissioned by the Department for Transport (DfT) established that NO_x service emissions of light-duty Euro 5 and 6 vehicles were six times higher than the legal binding thresholds (Department for Transport, 2016, pp. 22, 23). Recently, two large

European vehicle manufacturers were discovered to have manipulated NO_x emissions data for their diesel vehicles under testing conditions (Dey *et al.*, 2018b). One manufacturer sold more than one million fraudulent vehicles in the UK, and approximately 8.5 million fraudulent vehicles in 23 countries in Europe, including the UK, between 2009 and 2015. This deceitful act was reported by Oldenkamp *et al.* (2016), who estimated that practically 500 kilo-tonnes of NO_x emissions were illegally released into the atmosphere across Europe, with an estimated 44,000 disability-adjusted life years (DALYs) and a value of life lost of £28 billion.

There is the potential to lower the emissions of various GHGs and air pollution in the transport sector via the increasing adoption of electric vehicles (EVs), which promise reductions in emissions and improvements in air quality (Serradilla *et al.*, 2017; Dey *et al.*, 2018a). Some of these emissions will be released during electricity generation at plant sites and their quantities, such as of CO_2 emissions, depend on the intensity of electricity demand and power sources used including coal, natural gas and nuclear (Robinson *et al.*, 2013). A study estimated that replacing 40% of internal combustion engine vehicles (ICEVs) with EVs in Madrid, Spain will mitigate NO_x and carbon monoxide (CO) emission concentrations by 17% and 22% respectively (Soret *et al.*, 2014). Buekers *et al.* (2014) conducted a study across 27 countries in the European Union, and the findings indicated that in 2010 an annual benefit of €30.3 million was expected in the UK from avoiding external costs to tackle subsequent cost in relation to health and climate impacts. This is for only a 5% EV market penetration rate, making an annual travelling distance of 10,000 km. The benefit was predicted to escalate to €46.6 million by 2030. However, this study does not focus on local road network locations and meteorological conditions, given that it was completed not only in the UK, but in 26 other countries, and was based on replacing vehicle kilometres travelled (VKT) by ICEVs with EVs.

Increasing EV adoption not only decreases vehicle emissions, but also reduces noise levels related to engines (Ibarra *et al.*, 2017) and the corresponding adverse health effects linked to traffic noise (Münzel *et al.*, 2014; Stansfeld, 2015). Ibarra *et al.* (2017) established a minimum reduction in traffic noise of 10 dBA if ICEVs are replaced with EVs in urban areas. Tobollik *et al.* (2016) demonstrated that an EV adoption of 50% in Rotterdam, Netherlands, by 2020 would lead to a reduction in DALYs of 26 (confidence interval (CI): 13–161) due to less noise annoyance, and 41 (24–60) due to reduced noise-induced sleep disturbance for the population. This is approximately equivalent to a 10% decrease in private VKT on minor-urban roads.

Exposure to bus emissions raises serious public health issues. Beatty and Shimshack (2011) found the retrofitting of school buses to be associated with a decrease in asthma, bronchitis and pneumonia in children and adults. Among transport commuting modes, bus commuters were discovered to be more exposed to $PM_{2.5}$ than those using other modes of transport such as travelling by car, walking and cycling (McNabola *et al.*, 2008). This exposure occurs not only during waiting times at bus stops, but also while seated in the cabin, with approximately 30% of pollutants inside being emitted by the bus itself (Behrentz *et al.*, 2004; Adar *et al.*, 2008; Beatty and Shimshack, 2011). Thus, the electrification of buses is likely to reduce passenger exposure to pollution. Furthermore, doses of inhaled pollutants such as $PM_{2.5}$ and soot for passengers using electric buses are lower compared to passengers in diesel buses, petrol and diesel cars who had also been exposed to pollutants. This was shown in a study conducted by Zuurbier *et al.* (2010), who also confirmed that inhaled doses of PM_{10} for electric bus passengers are lower than for diesel bus passengers. A strict evaluation of the relationship between reducing bus emissions and morbidity was conducted in the United States, specifically in New York City, by merging bus $PM_{2.5}$ and NO_x spatial concentrations at residential centroids with hospitalisation data for those residents who live near those centroids. It was found that stricter transport bus emissions standards are associated with reduced emergency department visits for respiratory illnesses such as asthma and bronchitis (Ngo, 2015).

Across 161 cities in China, 4.4% of the estimated 652,000 deaths associated with $PM_{2.5}$ exposure will be reduced if targets for plans to tackle air pollution are met by 2020 (Maji *et al.*, 2018). More locally, a study conducted in Newcastle upon Tyne and North Tyneside reported that an increase of 10 $\mu g/m^3$ in the annual mean PM_{10} level led to a 1% increase in salbutamol prescribed to help patients to expand their narrow lung airways (Sofianopoulou, 2011).

An EV is heavier than the equivalent modern ICEV by approximately 24%. Nonetheless, an EV causes nearly the same amount of PM_{10} and 1–3% less $PM_{2.5}$ than an ICEV (Timmers and Achten, 2016). This might be attributed to the smooth driving style regarding the EV and less braking, which saves as much battery charge as possible. Additionally, it can be argued that improvements in EV manufacturing in terms of battery technology will reduce its weight and undoubtedly decrease abrasion rates for tyres, brakes and road surfaces, which are the dominant contributor to non-exhaust emissions, as the higher vehicle mass requires greater energy to decrease vehicle momentum. This increases friction between the brake pads and wheels, and, moreover, the tyres and road surface (Timmers and Achten, 2016).

It seems that investigating the impact of increasing the proportion of EVs in vehicle fleets on air quality and health, in terms of decreasing hospital admissions and premature mortality, has not yet been comprehensively studied. This thesis seeks to ascertain the influence of EV adoption on changes in emissions, air quality and health, which is a novel aspect of this research.

1.3 Aim and Objectives

1.3.1 Aim

The primary aim of the study is to investigate the air quality and health impacts of scenarios that consider different levels of future EV uptake.

To achieve this aim, a set of key objectives has been developed as outlined below.

1.3.2 Objectives

1) To establish a Baseline in relation to traffic, emissions, air quality and health status for the year 2014 in Newcastle and Gateshead;
2) To establish a trend for 2030 traffic growth and to develop the business-as-usual (BAU) scenario;
3) To develop scenarios for 2030 with a range of EV uptake levels in the vehicle fleet, taking into consideration which conventional vehicles the EVs are replacing;
4) To investigate emissions and the air quality impacts of future BAU and EV scenarios; and
5) To investigate the health impacts of the developed scenarios, considering dose-response coefficients developed by the Committee on the Medical Effects of Air Pollutants (COMEAP).

1.4 Thesis Overview

In general, the thesis demonstrates a complete sequence using multiple vehicle traffic flow scenarios with a range of different levels of EV penetration, their emissions, the dispersion of these emissions and health data, in an attempt to evaluate the impact of these emissions on the health of residents.

The relationship between air pollution and human health is presented in chapter 2. In addition, the effect of emissions released by the transport sector is fully demonstrated. Additionally, the effect of dominant transport-related pollutants on human health is revealed, along with how exposure to these pollutants in both the short- and long-term might damage health and increase the rate of premature mortality and hospitalisations.

Types of ULEVs and their related infrastructures are illustrated in chapter 3, in addition to the implications of ULEVs for energy and air quality and how they influence the release of CO_2.

The research methodology describing the framework of the study sequence linked to traffic flow, traffic emissions and emissions dispersion, along with their impact on human health, is presented in chapter 4. Similarly, the parameters required and the sources of data in relation to running the framework are clarified.

In chapter 5, development of the Baseline traffic model and its calibration and validation are explained, which is the first objective, whilst the second and third objectives are achieved in chapter 6 which provides a rationale for the development of future traffic flow scenarios.

The estimation of the impact of the scenarios' emissions levels on human health is revealed in chapter 7 to fulfil the fourth and fifth objectives.

Finally, chapter 8 includes a summary of the thesis, discussion of the findings, conclusions of the study and its contribution to academic research and practice. Additionally, future research is proposed that might be considered.

CHAPTER 2

2. Literature Review of Vehicle Emissions, Air Pollution and Health

2.1 Introduction

This literature review begins by describing the risk of air pollution to health in order to show the danger of exposure to it. The following section outlines the adverse health effects associated with exposure to road transport emissions and noise, in an attempt to characterise the potential for improving health by means of lowering vehicular emissions and concentrations. Different types of vehicular emissions are subsequently identified. Regulatory instruments which have been formed to regulate air quality pollutants are reviewed in Section 2.5, including the Euro standards, National Emissions Ceiling Directive, Ambient Air Quality Directive and Climate Change Act. Finally, Section 2.6 defines the coefficients that used to describe the dose-response relationship between long- and short-term exposure to $PM_{2.5}$, PM_{10} and NO_2 and the probability of premature death and being admitted to hospital. It is crucial that those coefficients are applied to changes in pollution concentrations in order to evaluate the disease burden associated with reductions in exposure to air pollution.

2.2 Risk to Health of Exposure to Air Pollution

The World Health Organisation has defined the term health as "*a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity*" (WHO, 1946). According to Vallero (2014, p. 44), air pollution is defined as "*the presence of contaminants or pollutant substances in the air that interfere with human health or welfare, or produce other harmful environmental effects*", whilst Ott (1982), defined human exposure as "*the event when a person comes into contact with a pollutant of a certain concentration during a period of time*". Air pollution can be divided into air quality pollutants (e.g. NO_2) and GHGs.

Strong associations between exposure to air pollution and human health have been clearly established, and several epidemiological studies have confirmed strong correlations between daily levels of exposure to poor air quality and death, particularly due to cardiovascular and respiratory diseases (Dai et al., 2014; Pope et al., 2015; Qiu et al., 2015). Air quality has a considerable impact on peoples' health; approximately 6.5 to 9 million people die worldwide per annum due to the inferior quality of outdoor air (IEA, 2016; Landrigan *et al.*, 2018; WHO, 2018c). In the UK, more than 50,000 people die prematurely with a reduction in life expectancy of on average 7 to 8 months (EAC, 2010, p. 7), whilst exposure to $PM_{2.5}$ caused 29,000 deaths in 2008 (COMEAP, 2010, p. 90). The Royal College of Physicians (2016, p. 82) recently suggested that combined exposure to both $PM_{2.5}$ and NO_2 causes approximately 40,000 deaths per annum in the UK. Although complex to quantify, it is estimated that air pollution costs society nearly £20 billion annually, due to the health-related consequences of people suffering disease and early death.

More locally, Public Health England suggested that, in Gateshead, 4.9% of deaths in 2010 were attributable to air pollution, with long-term exposure contributing 99 deaths amongst those aged 25 years and over and 962 life years lost, whereas in Newcastle 4.9% of deaths in 2010 were traceable to air pollution, with long-term exposure causing 124 deaths amongst those aged 25 years and over and 1,320 life years lost (Gowers *et al.*, 2014, p. 10).

Similarly, increasing exposure to air pollution leads to an increase in hospitalisation rates (Chang *et al.*, 2005). A growing number of studies have explored the correlation between air pollution and hospital admissions in recent years. These epidemiological studies have described links between daily variations in air pollution and hospital admissions for cardiovascular and respiratory diseases (Wilson *et al.*, 2004; Chang *et al.*, 2005; Cao *et al.*, 2009; Chen *et al.*, 2010a; Gurjar *et al.*, 2010; Kalantzi *et al.*, 2011; Li *et al.*, 2011; Di Ciaula, 2012; Hansen *et al.*, 2012; Lin *et al.*, 2013; Rodopoulou *et al.*, 2014; Tao *et al.*, 2014).

2.3 Air Pollution Due to Road Traffic Emissions

Over the last 20-30 years, the number of epidemiological studies illustrating traffic-related air pollution as a public health problem has grown considerably (Health Effects Institute, 2010). Research reveals a definite correlation between traffic-related air pollution and premature mortality (Health Effects Institute, 2010; Hoek *et al.*, 2013; Beelen *et al.*, 2014; Héroux *et al.*, 2015). Moreover, traffic-related air pollution has been linked to a broad range of diseases, comprising, though not restricted to, cardiovascular disease (Cesaroni *et al.*, 2014); lung cancer (Health Effects Institute, 2010; Raaschou-Nielsen *et al.*, 2013a); diabetes (Eze *et al.*, 2015); adverse birth outcomes such as premature births, low birth weight and perinatal mortality (Health Effects Institute, 2010; Sapkota *et al.*, 2012; Pedersen *et al.*, 2013), and adverse respiratory outcomes specifically in childhood, for instance COPD, asthma, respiratory infections and reductions in lung function (Health Effects Institute, 2010; Anderson *et al.*, 2013; Gehring *et al.*, 2013), in addition to autism spectrum disorder (ASD) (Volk *et al.*, 2013), given that the ASD global index was 6 per 1,000 (WHO, 2018b).

In their research, Bhalla *et al.* (2014, p. 23) reported that air pollution from motor vehicles caused 184,000 premature deaths globally, including 91,000 deaths from coronary heart disease, 59,000 from strokes and 34,000 from lower respiratory infections, COPD and lung cancer. Lelieveld *et al.* (2015), who utilised more complex source models, estimated that road traffic emissions at country level are responsible for in the region of one-fifth of deaths via ambient $PM_{2.5}$ and ozone in the UK, the United States and Germany, whilst comprising approximately 5% of the 3.3 million annual premature deaths worldwide due to outdoor air pollution. It is also worth stating that, if the health impacts of NO_x are added, the numbers may double (Walton *et al.*, 2015).

Approximately 3,000 anthropogenic gaseous groups have been identified over the last few decades (Fenger, 1999) and motor vehicles have been established to cause in excess of 700 gases (EPA, 2006). Moreover, it has been estimated that a large proportion of these particular substances have a detrimental effect on human health and cause several respiratory diseases (Lepeule *et al.*, 2014; Sinharay *et al.*, 2018).

The capacity to observe and examine emissions comprising such a significant amount of pollutants and gases which have such a destructive effect on the environment is perceived to be financially or physically unfeasible. The most common gases produced are known as indicators and are indicative of the pollutants discovered within the atmosphere. For instance, NO_x is the common name for NO_2 and NO and is an indicator of these two gases (Wild *et al.*, 2017). Furthermore, hydrocarbon (HC) is the name given to a substantial number of chemical groups consisting of volatile organic compounds (VOCs), which is a term used to describe a considerable number of chemicals, for instance benzene (C_6H_6), methane (CH_4), 1,3-butadiene (C_4H_6), toluene (C_7H_8), formaldehyde (CH_2O) and polycyclic aromatic hydrocarbons (PAHs) (Franco *et al.*, 2015; Zhang *et al.*, 2017). Moreover, the title designated for particulate matter is the term particulates, which consists of a range of chemicals such as sodium chloride, trace metals, water, black carbon, VOCs and secondary particles. Particulate matter is generally symbolised by means of size (AQEG, 2005); hence, particles which have diameters of less than 10 µm (i.e. PM_{10}) and 2.5 µm (i.e. $PM_{2.5}$) are characteristic of particles that belong in a specific size range (AQEG, 2005).

Most vehicular emissions are released by exhaust pipes; however, other pollutants (e.g. particles) are emitted by braking systems, worn out and wearing tyres, abrasive road surfaces (Omstedt *et al.*, 2005), leaks in tubing and engine casings and, furthermore, due to the process of evaporation (e.g. VOCs) (Boulter *et al.*, 2009c, p. 33). Moreover, indirect emissions released into the atmosphere by vehicles go through chemical transformations and become particles or gases that have a detrimental effect on the environment (Timmers and Achten, 2016). Vehicular emissions are discussed in detail in the following sections.

In the same context, considerable amounts of some GHGs such as CO_2, methane (CH_4) and Nitrous oxide (N_2O) are discovered in exhaust fumes produced by motor vehicles, and these therefore have the ability to cause climate change (Gallo, 2011). In 2017, 34% of CO_2 emissions in the UK were attributed to the transport sector, followed by the energy sector at 29% (BEIS, 2018b, p. 17). Representatives from numerous other countries including the UK met in Japan in 1997 to establish the Kyoto Protocol to mitigate the emissions of six GHGs: CO_2, CH_4, N_2O, sulphur hexafluoride (SF_6) and two groups of gases, hydrofluorocarbons (HFCs) and perfluorocarbons (PFCs) (United Nations, 1998). GHGs are expressed in CO_2 equivalents by means of their global warming potential (BEIS, 2016) to ease the comparison between these GHGs. In general, CO_2, is employed as the principal GHG environmental indicator. It is important to recognise that the transport sector is the main emitter of CO_2 as other sectors have reduced their emissions more rapidly.

2.3.1 Emissions Emitted from Vehicle Exhausts

The complex combination of chemicals discharged by exhaust fumes is created by the incomplete combustion of fuel within a vehicle's engine (Myung and Park, 2011). Furthermore, the combination of exhaust gases depends on the type of engine (e.g. four-stroke spark ignition, diesel) and the vehicle's operating conditions (Kašpar *et al.*, 2003). In most cases, exhaust gases principally consist of HCs, sulphur (S), CO, PM, NO_x, lead (Pb) and unburnt fuel (Van der Westhuisen *et al.*, 2004). With regards to contemporary motor vehicles, Pb and S emissions have been regulated in each type of fuel (European Commission, 1998), whereas CO, NO_x and HC exhaust emissions are managed by catalytic converters (Boulter *et al.*, 2009a).

Catalytic converters were initially launched in the United States during the 1970s and subsequently in Europe in the 1980s (Moldovan *et al.*, 2002). The reactions that arise in current converters involve reduction and oxidation processes. A number of these reactions can be observed in Table 2-1 (Kašpar *et al.*, 2003). As regulations related to new vehicles become stricter, exhaust emissions are being reduced. Counter to this, reduction reactions transform NO_x groups into water vapour (H_2O) and comprise (N_2) and groups consisting of nitrogen in addition to CO_2. Additionally, groups comprising nitrogen discharged from a vehicle exhaust following catalytic reduction include N_2O, which is a GHG (Graham *et al.*, 2009).

Table 2-1: Reactions of oxidation and reduction occurring on a vehicle catalyst

Reaction Type	Equation
Oxidation	$2CO + O_2 \rightarrow 2CO_2$
	$HC + O_2 \rightarrow CO_2 + H_2O$[a]
Reduction	$2CO + 2NO \rightarrow 2CO_2 + N_2$
	$HC + NO \rightarrow CO_2 + H_2O + N_2$[a]
	$2H_2 + 2NO \rightarrow 2H_2O + N_2$

Source: Kašpar *et al.* (2003)
[a] Unbalanced equations

In recent years, several vehicle technologies have been created with the aim of reducing the release of toxic air pollutants into the atmosphere via motor vehicles (Uherek *et al.*, 2010). Diesel particulate filters (DPFs) are ceramic appliances which accumulate particulate matter in the exhaust stream (DEQ, 2012). The high temperature in the exhaust heats up the ceramic structure, and therefore enables the particles contained inside to oxidise into components that are less damaging (U.S. EPA, 2010). DPFs reduce particulate matter emissions discharged from exhaust pipes to 0.5mg/km, which is the value applied in the UK 2009 emissions factors in relation to Euro 5 and Euro 6 vehicles (Boulter *et al.*, 2009b). Moreover, exhaust gas recirculation (EGR) systems decrease emissions of NO_x from the exhaust stream; however, fuel consumption is increased (Dong *et al.*, 2008). On the other hand, selective catalytic reduction (SCR) lessens NO_x emissions to N_2 by utilising urea or ammonia. This takes place without consuming further fuel (Sjövall *et al.*, 2006).

2.3.2 Emissions Emitted Indirectly

Indirect emissions arise because of condensation processes and chemical reactions in the environment, subsequently converting motor vehicle emissions into groups that are extremely deleterious to the environment. It is also worth noting that aerosols and secondary particulates form in the atmosphere because of the condensation found in exhaust pipe gases. For instance, NO_x, SO_2 and VOCs are transformed into minute droplets (Quah and Boon, 2003). Besides this, NO_x in conjunction with HC and CO groups participate in photochemical reactions in the atmosphere that form ozone (O_3), which is a strong GHG and exceedingly poisonous to humans (Inoue *et al.*, 2008). In the presence of NO_x, the production of O_3 in the troposphere involves hydroxyl radical oxidation of CH_4, CO and other HCs. Additionally, a coupled reaction between O_3 and NO_x generates a photostationary state involving NO, NO_2 and O_3 that has a time scale of approximately 100 seconds (Mannschreck *et al.*, 2004). It has been noted that, from 1993-2016, background levels of O_3 in the UK increased from 42µg/m^3 to 56 μg/m^3 (DEFRA, 2018) owing to increased emissions of anthropogenic NO_x and VOCs principally caused by road traffic.

2.3.3 Evaporative Emissions

It is predominantly VOCs that are released into the atmosphere as a result of evaporative processes from both moving and parked vehicles (Boulter *et al.*, 2009c, p. 33). VOCs can be dispersed from vehicles while refuelling and as a result of faults in fuel caps; and/or cracks in engine casings and tubing (Dong *et al.*, 2015). In addition, evaporative VOC emissions produced by diesel vehicles are minimal because of the extremely low volatility of diesel fuel and the presence of heavier HCs (Boulter *et al.*, 2009c, p. 33). Conversely, evaporative emissions generated by petrol vehicles have been demonstrated to contribute considerably to ambient VOC concentrations (Boulter *et al.*, 2009c, p. 33). According to Batterman *et al.* (2005), extreme cases during the summer, up to 62.7 mg/h of non-methane hydrocarbons (NMVOCs) might be produced by vehicles with broken or loose fuel caps.

2.3.4 Emissions from Tyres, Brakes and the Road Surfaces

Non-exhaust-derived vehicle pollution created by wear and tear to tyres and brakes occurs by means of mechanical and abrasive friction and primarily results in the emission of particulate matter that varies in size from a few hundred nanometres to tens of micrometres (Thorpe and Harrison, 2008). Additionally, emissions from tyres and brakes that are worn down have been revealed to make a significant contribution to ambient pollution concentration levels. For instance, in Stockholm, non-exhaust pipe mechanically generated emissions comprised 90% of the total traffic's contribution in relation to PM_{10} (Omstedt *et al.*, 2005). Moreover, Grigoratos and Martini (2014) demonstrated that exhaust and mechanical emissions contribute uniformly to ambient PM_{10} levels in European cities. Nonetheless, the rate of brake wear is closely dependent upon the linings employed and the type of driving the brakes are subjected to by individual drivers (Thorpe and Harrison, 2008). It is also worth noting that particulate matter emissions produced by exhausts tend to make up most $PM_{2.5}$, whilst non-exhaust emissions are primarily made up of PM_{10}, although a significant proportion of emissions also consist of fine $PM_{2.5}$ (Timmers and Achten, 2016). It can be argued that motorised roads emit significant amounts of particulate matter, at least at roadsides.

2.4 Traffic-Related Noise

Levels of surrounding noise are correlated with the road network, traffic flow, speed and load, junctions, acoustics and meteorological conditions in cities (Foraster *et al.*, 2011; Bell and

Galatioto, 2013; Zuo *et al.*, 2014). Values of L_{50} (noise level for 50% of the measurement duration) range from approximately 54 dB in acoustic shadows in residential tertiary streets up to 74 dB on roads carrying heavy traffic; at which levels over 55 dB are potentially dangerous to health (Bell and Galatioto, 2013). The health effects of traffic-related noise are increasingly being acknowledged as responsible for a large burden of disease that may be equivalent to that of air pollution (Hänninen *et al.*, 2014). It is estimated that in the region of one million healthy life years are lost every year from traffic-related noise in parts of Western Europe (World Health Organization, 2011).

Ambient noise has been linked to a range of distinctive auditory and non-auditory outcomes (Basner *et al.*, 2014), as well as all-cause premature mortality (Halonen *et al.*, 2015), cardiovascular mortality and disease (Ndrepepa and Twardella, 2011; Van Kempen and Babisch, 2012; Babisch *et al.*, 2014; Basner *et al.*, 2014; Münzel *et al.*, 2014), irritation and sleep disturbances (Omlin *et al.*, 2011; Laszlo *et al.*, 2012; Basner *et al.*, 2014), adverse reproductive outcomes (Ristovska *et al.*, 2014), cognitive problems in children (Stansfeld *et al.*, 2005; Basner *et al.*, 2014), high blood pressure in children (Paunović *et al.*, 2011), type-2 diabetes (Dzhambov, 2015), psychological and health problems (World Health Organization, 2011) and strokes (Sørensen *et al.*, 2011). Furthermore, it has been claimed that ambient noise may also produce increased stress and aggression in people (Geurs *et al.*, 2009).

Although the long-term effects of traffic-related air pollution and noise can be mutually confounded, cardiovascular effects by way of surrounding noise have been shown to be independent of exposure to air pollution (Van Kempen and Babisch, 2012; Liu *et al.*, 2014). Research has suggested that low-income individuals and ethnic minorities tend to be located in areas that are polluted the most by road traffic noise (Brainard *et al.*, 2004; Nega *et al.*, 2013; Carrier *et al.*, 2016).

2.5 Air Quality Regulations

The purpose of designated policy which sets limits for rates of emissions and pollutant concentrations is to protect humans and the environment. Currently, three main regulatory instruments have been formed to regulate the emissions of air quality pollutants in the European Union (EU). These main instruments are the European Emission Standards, the National Emissions Ceiling Directive (NECD), and the Ambient Air Quality Directive (AQD). Alongside these instruments, the UK Climate Change Act 2008 was established with the objective of reducing greenhouse gases by 2050 by 80% compared to the baseline 1990 emission levels. Although, the Climate Change Act which was enacted into law on 26th

November 2008 (HM Government, 2008) does not address air quality issues, it encourages the penetration of ultra-low emission vehicles which will certainly lead to reductions in vehicular emissions in urban areas where most people are exposed to air pollution. These instruments aim to improve air quality, given that it is a vital component of public health and well-being and is important for all living organisms. If the quality of air improves, the quality of life will be improved and life expectancy extended (Walters, 2010) and, as a result, there will be an enhancement in economic development (Autrup, 2010).

To reduce road emissions and improve air quality, the UK in 2011 was the first country which recommended that ULEVs should be the only available car for sale by 2040. There is another target to make all cars in the UK ULEVs by 2050. To do this, the sale of all new cars will be limited to ULEVs (DEFRA and DfT, 2017b, p. 4). Lately, 'The Road to Zero' strategy and Clean Air Strategy 2019 confirm this proposal and it is estimated that the percentage of ULEVs could reach 40 to 70% of new car and van sales by 2030 and the intention is to replace all governmental fleets with ULEVs by 2030. The latter plan will start by limiting the purchase of new cars to ULEVs aiming for them to represent a quarter of the government's vehicle stock in 2022. Indeed, the UK government is investing more than £2.7 billion to provide better air quality and cleaner transport (DEFRA and DfT, 2017a).

2.5.1 European Emissions Standards

The European emission standards (Euro standards) have set corresponding limits on exhaust emissions from road vehicles. In 1970, legal thresholds were applied for the first time on carbon monoxide (CO) and unburnt hydrocarbon emissions from passenger cars in Europe (EC, 1970), while successive reductions to these thresholds were introduced, such as in Directives 83/351/EEC and 88/76/EEC (Tiwary and Colls, 2010, p. 472). Limit values were successively reduced with more stringent regulations for passenger cars, light duty vehicles (LDVs) and heavy-duty vehicles (HDVs) from 1988 onwards. The Euro standards were initially launched in the UK in stages, beginning in 1992 with Euro 1. The measures are currently stipulated for Euro 6/VI vehicles (SMMT, 2016). As mentioned in Section 1.2, Arabic numbers are used for Euro standards for LDVs and Roman numbers when considering Euro standards for HDVs. The existing emissions standard for LDVs is Euro 6, which was introduced in 2014 for vehicles under 1305 kg and in 2015 for vehicles weighing between 1305 to 1760 kg under Regulation 459/2012/EC (EC, 2012). Euro 6 was introduced with the aim of reducing NO_x emissions from diesel cars from 180 mg/km to 80 mg/km. In the same way, Euro VI, the current edict implemented regarding Heavy Duty Vehicles (HDVs), aims to lower NO_x emissions by 95% in comparison to Euro I. Additionally, the Euro VI standard requires diesel HDVs to conform to a particulate matter threshold to half of what was legislated for previously in Euro V. In the meantime, these standards also set thresholds for CO, total hydrocarbons (THC), non-methane hydrocarbons (NMHC), hydrocarbons and oxides of nitrogen (THC + NO_x), PM, number of particles (PN), and NO_x. Table 2-2 demonstrates the limits of harmful emissions cited in the first to sixth Euro standards.

Table 2-2: Emission limits for M_1 (passenger cars)

Engine type	**Date (new)**	**Date (all)**	**CO (g/km)**	**THC (g/km)**	**NMHC (g/km)**	**(THC + NO_x) (g/km)**	**PM (g/km)**	**PN (# /km)**	**NO_x (g/km)**
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

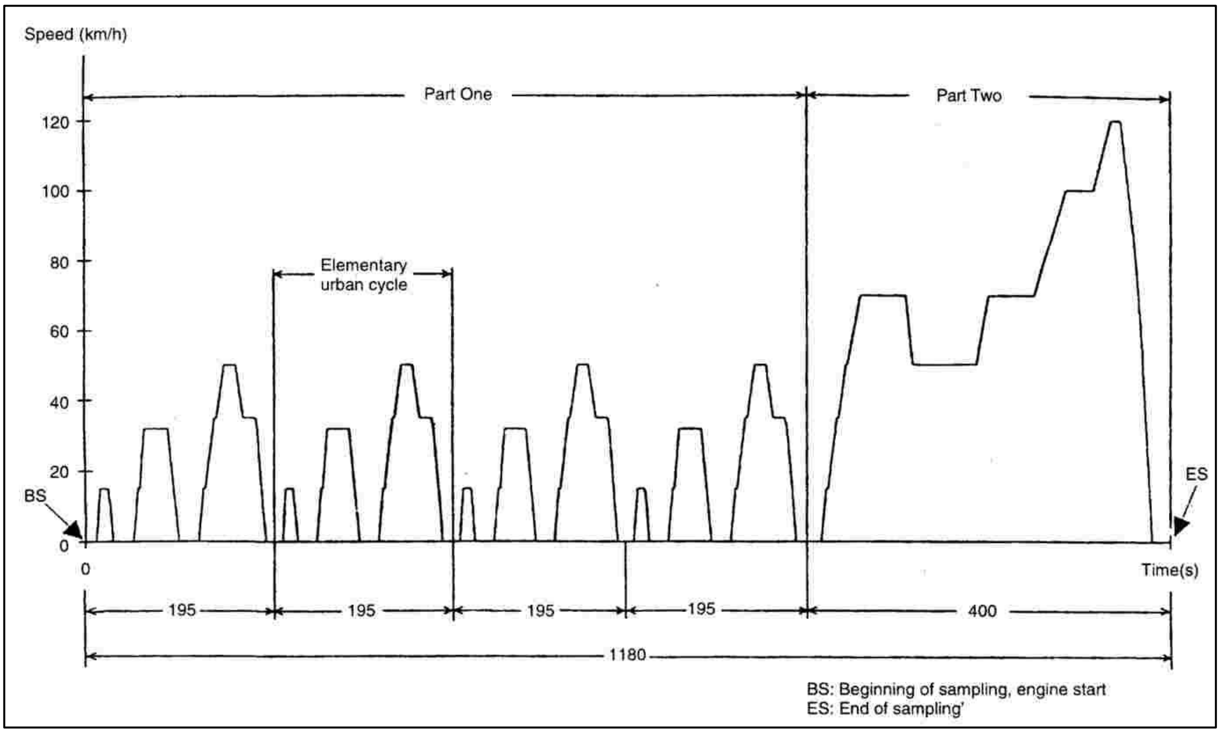
*applies to gasoline direct injection (GDI) only

2.5.1.1 Type Approval Tests

Motor vehicle regulations determine type approval procedures that are applied to specimen example vehicles before general sale is allowed (SMMT, 2016). This is to guarantee that the criteria for the production requirements prescribed by the EU in relation to environmental, safety and conformity regulations are complied with (DfT, 2016). The type approval test is a standardised test called the New European Driving Cycle (NEDC). According to Marotta *et al.* (2015), the NEDC was originally used to measure air quality pollutant emissions. Moreover, in 2009, it became the yardstick for CO_2- type approval emissions to meet the EU's mandatory CO_2 regulations (Mock *et al.*, 2012). Initially, in 1970, the Urban Driving

Cycle (UDC/ECE-15) came up with the first European emissions legislations, and subsequently, to simulate more aggressive, high-speed driving modes called the Extra Urban Driving Cycle (EUDC) in 1990. Shown in Figure 2-1 are the four UDCs and one EUDC as required for the NEDC.

The MOT test is conducted on vehicles in the UK to guarantee that Euro standards are adhered to and to confirm emissions compliance. It should be noted that the MOT test is compulsory for most vehicles in the UK which are over three years old. To conform to Euro standards, manufacturers must also make certain that recent technology fitted to vehicles related to the reduction of emissions design thresholds of vehicle emissions, should remain as it is for a mileage of 160,000 km (Boulter *et al.*, 2009b, p. 29).



Source: European Commission (1998)

Figure 2-1: Speed profile of New European Driving Cycle

However, considerable evidence has emerged of incompatibility between type approval limits and real-world NO_x emissions (Kågeson, 1998; Weiss *et al.*, 2011). In relation to CO_2 emissions, discrepancies between real-world emissions and type approval were observed at a growing rate from 9% in 2001 to 28% in 2012, rising to 42% in 2015 (Archer, 2016b). Because of these discrepancy issues, particularly for the NEDC not being representative of real-world emissions, the United Nations Economic Commission for Europe formulated the UNECE Regulation No. 83 (UNECE, 2015). This recommended that the test must be conducted with a surrounding temperature within 23 ºC ± 5 ºC. In the UK, the average surrounding temperature is 9°C, and according to Kwon *et al.* (2017) NO_x emissions increase at low temperatures because emission controls are disconnected with the intention to protect the engine.

2.5.1.2 Introduction of Real Driving Emissions (RDE) Type Approval

In the intervening time, to address the incongruity of the NEDC and real-world driving emissions, the EU has introduced the real-driving emissions (RDE) type approval test which is carried out by the use of a Portable Emissions Measurement System (PEMS). The PEMS is light, small, and therefore convenient and can be carried and moved with the vehicle while being driven during the testing, unlike the fixed rollers of a dynamometer mimicking real-world driving. Aimed at cost and time reduction, in matters of initiating mobile emissions, the United States Environmental Protection Agency (USEPA), EU, private agencies, and diverse states have begun to promote the PEMS as an effective vehicle emission testing tool. The efficiency of the PEMS in terms of the validity and reliability of results during emissions testing was reported in a study by Weiss *et al.* (2011) on Euro 3-5 diesel vehicles. The NO_2 emission values were found to exceed emission limits using this mobile equipment, contrary to the NEDC testing which showed the vehicles to have complied with the legislation. Moreover, the PEMS can determine where the emissions occur and how the vehicle was being driven at the time.

2.5.1.3 Defeat Devices (Diesel Cars)

Between 2008 and 2015, defeat devices were installed in 11 million Volkswagen cars (Brand, 2016). This illegal device was discovered by the USEPA in 2015, via a “switch code” which Volkswagen wrote into their diesel vehicles’ electronic control module (ECM). The code could spot type approval conditions through the vehicle’s behaviour, such as the steering wheel position, barometric pressure, and duration of engine operation (Cruden *et al.*, 2018, p. 126). Once the type approval condition is detected, the ECM would fully execute the NO_x emission controls. Even so, the emission controls are only temporary when vehicles are being tested in the real-world, resulting in higher NO_x emissions. Archer (2016a) referred to this as the “cycle detection” defeat device. This defeat device can be deadly, as evidenced by a recent study which reported that 1,200 early deaths in Europe might be attributed to the presence of a defeat device in Volkswagen cars sold in Germany alone (Chossière *et al.*, 2017).

A cold start is when the engine has cooled to ambient temperature to below 30°C for 12 to 36-hours before the engine is switched on (Heimrich, 1990). The low temperatures in the first minutes of engine operation results in incomplete combustion, causing higher emissions than in normal operation to be released, as found by Weilenmann *et al.* (2005). The occurrence of higher emissions is aggravated by the catalytic converter running below its ideal operating temperature at ~400°C as this prevents the removal of HC, CO, PM, and NO_x (Mathis *et al.*, 2005; Chang *et al.*, 2014). Cold start emissions are exceptionally sensitive to ambient temperature, except for NO_x from petrol-driven vehicles (Weilenmann *et al.*, 2009; Reiter and Kockelman, 2016). Favez *et al.* (2009) stated that most HC, CO, and NO_x emissions are from the cold start phase for the first 1-7 km. It can be said that, for short journeys, cold start emissions are the overriding source of total pollutant emissions and can even persist throughout the journey. Miller and Franco (2016, p. 8) calculated the cold start journey comprise 8% of driven VKT.

A defeat device gives fully secure emission control only after a cold start but switches off or reduces the effectiveness of emissions control in 'hot restart' conditions. This phenomenon was found in 32 out of 38 diesel vehicles in the UK and 48 out of 53 vehicles in Germany (ICCT, 2016). Among defeat devices, the most common is the 'thermal window', which turns off or decreases the effectiveness of emissions controls at temperatures below the laboratory test (23 - 29°C) to protect the engine, given that the average temperature in Europe is 9°C (Archer, 2016, p. 23). For example, Opel and Renault switch off their emissions controls below 17°C, Daimler below 10°C and Peugeot below 5°C (Archer, 2016a, p. 23). Finally, certain defeat devices switch off emissions control after 22 minutes because the NEDC lasts for 20 minutes, as found in some Fiat models (Archer, 2016a, p. 24).

2.5.2 National Emissions Ceiling Directive (NECD)

The NECD determines the permissible ceilings on total pollutants emissions measured in kilotonnes. These ceilings are issued to member states on an annual basis, decreasing yearly relative to the 2005 baseline target level. Two directives were decreed from 2010–2020 and from 2020–2030; namely, Directive 2001/81/EC and Directive 2016/2284/EU respectively. Limits were established for NO_x, $PM_{2.5}$, NMVOCs, sulphur dioxide (SO_2), and ammonia (NH_3) and these have been fully binding from 2010. Annual reporting to the European Environment Agency (EEA) on compliance monitoring is conducted by member states via their national emission inventories.

Fortunately, the UK has met all emissions ceilings in relation to 2015 (DEFRA, 2017a, p. 6). However, without new policies, breaching would most likely take place in emissions ceilings for $PM_{2.5}$ and NH_3 by 2020 and all five of our emissions ceilings by 2030, as estimated in the

Clean Air Strategy 2019 (DEFRA, 2019a, p. 98). The UK national emissions reduction targets for five key pollutants by 2020 and 2030 are presented in Table 2-3.

Table 2-3: Future NECD targets

	2005 baseline (kt)	Reduction required by 2020	Reduction required by 2030	2020 ceiling (kt)	2030 ceiling (kt)
NO_x	1,714	55%	73%	771	463
SO_2	773	59%	88%	317	93
NMVOCs	1,042	32%	39%	709	636
$PM_{2.5}$	127	30%	46%	89	69
NH_3	288	8%	16%	265	242

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

In the UK, the Environment Protection Act 1995 and the National Air Quality Strategy 1997 introduced an approach to decreasing the air pollution that was recognised as a threat to public health (Longhurst *et al.*, 2016). This approach was ratified in a series of laws and air quality standards, as shown in Table 2-4, which firm the basis of air quality legislation in the UK.

Table 2-4: Parts of the Environment Protection Act 1995 relating to air quality

Clause of the Environment Protection Act	Key features
81	Environmental agencies are mandated to consider strategy.
82	Within their areas, Local Authorities must frequently review air quality and identify any part where standards may be breached.
83	Air Quality Management Areas (AQMAs) must be assigned in areas where air quality limit values are not achieved by LAs.
84	Frequent assessment of AQMAs and producing action plans must be taken by LAs to enhance air quality to acceptable level.
85	The Secretary of State has the power to give directions to LAs.
86	County Councils may make recommendations to the district on air quality and action plans.

In 2001, the European Commission introduced the Clean Air for Europe (CAFE) programme so as to create a strategy that was long-lasting and integrated with the intention of addressing the issue of air pollution and to guard against its effects on the environment and people's health (EC, 2001). The aims of the initiative were to set target values in relation to air pollution and national emissions ceilings in order to increase pollution-reduction plans in areas that had been targeted and to specify measures for raising product standards and restricting emissions. Subsequently, the European Commission created the Thematic Strategy on Air Pollution which is based on the framework pertaining to the CAFE Programme (AEA, 2005). This Strategy created temporary objectives with regards to air pollution in the EU and set out suitable measures which could be applied so as to attain them. In 2010, the Strategy was reconsidered.

Due to the CAFE Programme, the 2008 European Commission Ambient Air Quality Directive (2008/50/EC), which is known as the Air Quality Directive (AQD), established limit values regarding pollutant concentrations in outdoor air which were legally binding (EC, 2008). The principal legislative change implemented in the AQD was in relation to an exposure reduction approach for $PM_{2.5}$, and a comparatively high limit was established concerning $PM_{2.5}$. However, it requires that those areas which have the most air pollution have to accomplish greater reductions. The AQD was an amalgamation of virtually all the previous EU air quality laws and became legislation in England by means of the Air Quality Standards Regulations 2010 (HM Government, 2010). These Regulations put in place air quality limit values for pollutants such as Lead (Pb), Benzene, PM_{10}, $PM_{2.5}$, SO_2, NO_2, PAH, 1, 3-butadiene and CO (see Figure 2-2) (DEFRA, 2017d). The Regulations require that all competent Secretaries of State constantly examine air quality in their areas of authority (EC, 2008). This requirement was handed over to local authorities who are accountable for conducting assessments and reviews regarding air quality in the UK (HM Government, 2010). Furthermore, areas which were noted to be uncooperative in terms of the EU limit values were declared as AQMA, for which the local authorities are required to formulate air quality action plans that aim to diminish harmful air pollution to below EU limit levels (DEFRA, 2015, p. 29). As per instructions from the Directive (2008/50/EC), if and when member states could not comply with the NO_2 limit value by January 2010, application for an extension to January 2015 could be filed on a zone by zone basis. It was in 2009 when the UK filed an extension request for 24 zones and subsequently received approval for 9 zones. The UK at this point had succeeded in 37 out of the 43 zones (DEFRA, 2017a, p. 43). Regarding the hourly limit, the UK has yet to succeed in complying for the Greater London Urban area and South Wales (ibid.).

While it can be said that, without new policies, the UK might not able to meet the NECD commitments, as mentioned in the previous section; this also holds true when it comes to the AQD. The NECD sets limits concerning the total emissions of pollutants. Meanwhile, the AQD is resolute in its Air Quality Limit Values, in restricting or curbing pollutant concentrations in ambient or surrounding air. The UK has not been successful in this regard.

Pollutant	Applies	Objective	Concentration measured as[10]	Date to be achieved by and maintained thereafter	European obligations	Date to be achieved by and maintained thereafter	New or existing
Particles (PM_{10})	UK	50µg.m^{-3} not to be exceeded more than 35 times a year	24 hour mean	31 December 2004	50µg.m^{-3} not to be exceeded more than 35 times a year	1 January 2005	Retain existing
	UK	40µg.m^{-3}	annual mean	31 December 2004	40µg.m^{-3}	1 January 2005	
	Indicative 2010 objectives for PM_{10} (from the 2000 Strategy and 2003 Addendum) have been replaced by an exposure reduction approach for $PM_{2.5}$ (except in Scotland – see below)						
	Scotland	50µg.m^{-3} not to be exceeded more than 7 times a year	24 hour mean	31 December 2010			Retain existing
	Scotland	18µg.m^{-3}	annual mean	31 December 2010			
Particles ($PM_{2.5}$) Exposure Reduction	UK (except Scotland)	25µg.m^{-3}	annual mean	2020	Target value 25µg.m^{-3} [12]	2010	New (European obligations still under negotiation)
	Scotland	12µg.m^{-3}		2020	Limit value 25µg.m^{-3}	2015	
	UK urban areas	Target of 15% reduction in concentrations at urban background[11]		Between 2010 and 2020	Target of 20% reduction in concentrations at urban background	Between 2010 and 2020	
Nitrogen dioxide	UK	200µg.m^{-3} not to be exceeded more than 18 times a year	1 hour mean	31 December 2005	200µg.m^{-3} not to be exceeded more than 18 times a year	1 January 2010	Retain existing
	UK	40µg.m^{-3}	annual mean	31 December 2005	40µg.m^{-3}	1 January 2010	
Ozone	UK	100µg.m^{-3} not to be exceeded more than 10 times a year	8 hour mean	31 December 2005	Target of 120µg.m^{-3} not to be exceeded more than 25 times a year averaged over 3 years	31 December 2010	Retain existing

Source: DEFRA (2017d)

Figure 2-2: Air quality limit values implemented in the European Union and the UK

Pollutant	Applies	Objective	Concentration measured as	Date to be achieved by and maintained thereafter	European obligations	Date to be achieved by and maintained thereafter	New or existing
Sulphur dioxide	UK	266µg.m-3 not to be exceeded more than 35 times a year	15 minute mean	31 December 2005			Retain existing
	UK	350µg.m-3 not to be exceeded more than 24 times a year	1 hour mean	31 December 2004	350µg.m-3 not to be exceeded more than 24 times a year	1 January 2005	
	UK	125µg.m-3 not to be exceeded more than 3 times a year	24 hour mean	31 December 2004	125µg.m-3 not to be exceeded more than 3 times a year	1 January 2005	
Polycyclic aromatic hydrocarbons	UK	0.25ng.m-3 B[a]P	as annual average	31 December 2010	Target of 1ng.m-3	31 December 2012	Retain existing
Benzene	UK	16.25µg.m-3	running annual mean	31 December 2003			Retain existing
	England and Wales	5µg.m-3	annual average	31 December 2010	5µg.m-3	1 January 2010	
	Scotland, Northern Ireland	3.25µg.m-3	running annual mean	31 December 2010			
1,3- butadiene	UK	2.25µg.m-3	running annual mean	31 December 2003			Retain existing
Carbon monoxide	UK	10mg.m-3	maximum daily running 8 hour mean/in Scotland as running 8 hour mean	31 December 2003	10mg.m-3	1 January 2005	Retain existing
Lead	UK	0.5µg.m-3	annual mean	31 December 2004	0.5µg.m-3	1 January 2005	Retain existing
		0.25µg.m-3	annual mean	31 December 2008			

Figure 2-2: (Continued)

2.5.4 Climate Change Act

The Climate Change Act was brought into law in 2008, putting in place the commitments of the Kyoto protocol in UK law (HM Government, 2008). Clearly, the commitment stipulated that greenhouse gases should be reduced by 80% of 1990 baseline levels by 2050. With this legislation came the creation of the Committee on Climate Change (CCC) which, to achieve the 80% reduction goal, established 5 ‘carbon budgets’. Each has a target to meet increased reductions up to 2050 so as to keep up with the Kyoto commitment. It is worth noting that the UK had achieved half of this 80% target by 2016, when annual emissions had dropped by 41%, and most of the GHG sources witnessed a decrease in emissions. For example, GHG emissions from the energy supply sector were reduced from 242.1 $MtCO_2e$ in 1990 to 113.7 $MtCO_2e$ in 2016 (BEIS, 2018b, p. 8).

2.6 Relative Risk Coefficients

Numerous epidemiological studies have been conducted to estimate the association between short- and long-term exposure to air pollution and a range of health endpoints in different populations (Anenberg *et al.*, 2010; Li *et al.*, 2015a; Anenberg *et al.*, 2018). Acute effects including asthma and all-cause or cause-specific mortality can be short-term and due to time-varying exposures (COMEAP, 1998; WHO, 2006). Conversely, chronic or long-term effects, for instance lung cancer, cardiopulmonary diseases and mortality are due to the long-developing impacts of exposure.

Typically, epidemiological studies use statistical models to quantify the association between health outcomes such as mortality or hospitalisations and the contaminant concentration to which the population is exposed. These are described as concentration-response functions (CRFs) which are then used to derive the relative risks (RRs) (Gowers *et al.*, 2014, pp. 6, 29). The RRs are the ratio of the occurrence of a health outcome in the exposed population (I_e) to the health outcome occurring in the unexposed population (I_u) (i.e. $RR = I_e / I_u$) (Last, 2001, p. 156). In general, recommendations for CRFs are provided as RRs. However, some studies used to establish the CRFs provide odds ratios, which approximate to RRs under certain assumptions, such as for rare events or small concentration increments (WHO, 2013a, p. 3). The use of odds ratios might be more appropriate for larger concentration increments, such as in calculations of burden or the impact of a new policy quantified as the difference in total effects with and without the policy (ibid.).

Often, epidemiological studies report results in terms of an increase in the risk of an adverse health outcome such as mortality and hospitalisation related to a certain increment of concentration of air pollution; for instance, an RR increased in mortality of 6% for each 10 $\mu g/m^3$ increase in $PM_{2.5}$ (or similarly for any other pollutant) (Gowers *et al.*, 2014, p. 4). A 95% confidence interval is typically provided for each RR.

The choice of an optimal epidemiological study design depends on the research question posed as well as the availability of data, because no single study design is best for all applications. The aim of each study design is to target specific types of effects, outcomes and exposure sources; hence, the choice of optimal design relies on its efficiency in detecting the influence of exposure (WHO, 2013b, p. 41). This in turn relies on the scale of the study and the variability of the exposure. In this thesis, RRs developed by UK affiliated organisations were used to quantify the health impact.

CRFs are developed from two main epidemiological study types: cohort studies and time series studies (COMEAP, 1998, p. 7). In a cohort study, a group of people are followed over a long period of time and air pollution concentration data is collected during this period (e.g. Beelen *et al.* (2008), Lipsett *et al.* (2011), Faustini *et al.* (2012), Abbey *et al.* (1999) and Carey *et al.* (2013)). Thus, cohort studies represent combined acute and chronic i.e. short and long-term effects. The occurrence of a health outcome (such as respiratory mortality) in people exposed to air pollution (e.g. NO_2 concentrations) is then compared to the health outcomes in people who are not exposed; the RR.

In contrast, time-series studies analyse the association between daily changes in pollution concentration and daily outcome counts. Thus they are ideal for the analysis of short-term or acute effects (such as Atkinson *et al.* (2014)). For these types of studies, individual-level variables such as smoking and body mass index might typically not be controlled, as these factors are not likely to change substantially from day-to-day. However, other variables that vary daily and influence both air pollution and health, such as the weather, can be accounted for; these are termed confounders (Jiménez *et al.*, 2011). Particular studies account for the effect of one pollutant and thus use single pollutant models. Other models referred to as multipollutant models are used when investigating the effect of more than one pollutant (Burnett *et al.*, 2004).

Furthermore, specific studies which review the findings of available epidemiological studies (e.g. cohort or time-series studies), are called meta-analysis studies. This type of study targets the use of data from epidemiological studies that have addressed the same research questions (e.g. analysing the all-cause mortality effects of long-term exposure to $PM_{2.5}$) and similar study designs such as time series studies (e.g. Atkinson *et al.* (2014), Atkinson *et al.* (2018), Adar *et al.* (2014) and Mills *et al.* (2015)). A combined statistical analysis is then conducted and a single summary result is delivered.

2.6.1 Concentration-response Functions (CRFs) and Relative Risks (RR)

As mentioned briefly above, concentration-response functions (CRFs) and relative risks (RRs) are derived from different epidemiological studies. The magnitude of the RR or CRF relies on the exposure timescale which may be long- or short-term, and on the health outcome being researched; for instance, cause-specific mortality. Additionally, the CRF holds for the range of pollutant concentrations measured in the epidemiological study, because information on the association between the health outcome and the RR outside the recorded pollutant concentration range would not be valid. This might in turn rely on the period of measurement, such as the warm season or the whole year. Thus, CRFs are regularly associated with a threshold. This is the lowest concentration of pollutant recorded in the respective epidemiological study. Another limitation of the CRF is that the health outcome might be associated with individuals of a certain age; for example, those aged 65 or above.

The CRF curve can be linear or non-linear in shape. The Harvard Six Cities Study from 1974-2009 illustrated that the association between $PM_{2.5}$ exposure and lung cancer mortality was statistically significant, with a linear concentration-response relationship (Lepeule *et al.*, 2012a). The non-linear relationship, which can be used for long- and short-term NO_2, $PM_{2.5}$ and PM_{10} health impacts, commonly uses a log-linear function (e.g. Cohen *et al.* (2004)). The power function (e.g. Pope III *et al.* (2011)) and the integrated exposure response (IER) function (Burnett *et al.* (2014), Geng *et al.* (2015) and Cohen *et al.* (2017). The latter is only used to estimate cause-specific mortality associated with long-term exposure to $PM_{2.5}$.

The integrated exposure-response (IER) function combines evidence from epidemiological studies for air pollution, as well as passive and active smoking, to estimate the level of disease risk such as strokes at different $PM_{2.5}$ levels (Burnett *et al.*, 2014). Hence, the same measure is employed to estimate the risk of, for instance, heart disease from $PM_{2.5}$ due to outdoor air pollution as that of passive smoking or indoor air pollution. In addition, the IER function was developed to allow for non-linear patterns in the relationship between increases in $PM_{2.5}$ concentrations and the corresponding causes of diseases.

Many major epidemiological studies have been conducted in the USA to estimate the association between long-term exposure to $PM_{2.5}$ and health outcomes, and have only reported RRs across the $PM_{2.5}$ concentration range from 5 to 30 μg/m^3 (such as Krewski *et al.* (2009)). Hence, these studies might not provide knowledge concerning the association between mortality and long-term exposure to $PM_{2.5}$ at high ambient exposure levels, which is common in other areas in Asia as annual average exposures could reach higher concentrations (e.g. 100 μg/m^3) compared to other regions of the world. For this reason, the IER function was developed to estimate the RR of cause-specific mortality associated with long-term $PM_{2.5}$ exposure covering the entire range of ambient annual mean $PM_{2.5}$ concentrations globally. An IER function that accounts for changes in the shape of the CRF at high $PM_{2.5}$ concentrations to represent the RR of IHD mortality was developed by Burnett *et al.* (2014). Recently, Cohen *et al.* (2017) have developed further IER functions related to long-term $PM_{2.5}$ exposure for IHD, COPD, cerebrovascular disease, lung cancer and lower respiratory infections (LRI) using risk estimates from meta-analysis studies on ambient and indoor air pollution as well as exposure to passive and active smoking (Burnett *et al.*, 2014; Shin *et al.*, 2016).

2.6.2 Epidemiological Studies, Risk Estimates and Thresholds Used in this Thesis

2.6.2.1 Risk of Long-term Exposure to Air Pollution

In this thesis, premature death measures associated with long-term exposure to $PM_{2.5}$ and NO_2 were quantified using risk estimates from COMEAP (2009) and COMEAP (2015a) respectively and Carey *et al.* (2013) for PM_{10}-related mortality. However, the risk of hospital admission related to long-term exposure to NO_2, $PM_{2.5}$ and PM_{10} was quantified using estimates developed by Lee and Sarran (2015).

A relative risk of 1.025 (95% confidence interval 1.01, 1.04) associated with mortality for long-term exposure to an increase of 10 μg/m^3 in NO_2 concentration according to COMEAP (2015a) was based on an evaluation of cohort studies such as those by Fischer *et al.* (2015), Carey *et al.* (2013), Cesaroni *et al.* (2013) and Beelen *et al.* (2014). In addition, a 1.07 (-0.99, 1.16) relative risk of premature death linked to exposure to PM_{10} was developed by Carey *et al.* (2013) from their study of the health of a total of 835,607 patients aged over 40 years registered with 205 general practices (GP) during the period 2003 to 2007 in England. In relation to the relative risk of exposure to $PM_{2.5}$, COMEAP (2009) recommended an estimate of 1.06 with a 95% confidence interval (CI: 1.02, 1.11) for the individual coefficients that express the relative risks of early death associated with a 10 μg/m^3 increase in $PM_{2.5}$ concentration.

The latter RR was adopted by COMEAP (2009) because it was considered the most reliable source of the coefficient linking long-term exposure to air contaminants with mortality suitable for application in the UK, and was developed by the American Cancer Society (ACS) in the USA, taking into account a review by the Health Effects Institute (2000) on the ASC study and published extensions to the ACS study (Pope *et al.*, 2002). The ACS study has several major benefits in relation to the transferability of its coefficient to the UK. For example, it was a large-scale study, across the USA, which includes an extremely wide range of climate conditions, pollution mixtures and sub-populations. In addition, the ACS study is a general population study of people actually exposed to ambient air pollution, including $PM_{2.5}$ in background concentrations which are very relevant to UK conditions and policies. This supports the basis for its use in the UK.

In relation to the association of hospitalisation with long-term exposure to air pollution, it is relatively expensive and time-consuming to follow-up cohort number's details. Instead of the evaluation of cohort data in developing risk coefficients of hospitalisation related to long-term exposure to NO_2, $PM_{2.5}$ and PM_{10}, Lee and Sarran (2015) used spatial ecological studies which utilised geographical contrasts in air pollution concentrations and population-level disease risks. Analysis of these data, typically conducted using Poisson log-linear models, indicates the relative risk of hospital admissions associated with long-term exposure to increase of 10 μg/m^3 in NO_2, $PM_{2.5}$, PM_{10} concentrations, which were 17%, 32% and 8% respectively.

2.6.2.2 Risk of Short-term Exposure to Air Pollution

In this thesis, all-cause mortality associated with short-term exposure to NO_2 was estimated from systematic reviews and meta-analyses of 204 time series studies, giving an RR of 1.0071 (95% CI: 1.0043, 1.0100) per 10 μg/m^3 increase in 24-hr mean NO_2 developed by Mills *et al.* (2015). The mortality associated with short-term exposure to $PM_{2.5}$ was estimated from the random effects summary of the analysis of air pollution epidemiology databases, such as the results of 12 single-city time-series studies and the one multicity study on all-cause mortality for all ages. The result indicated a risk from short-term exposure to $PM_{2.5}$ of 1.0123 (95% CI: 1.0045, 1.0201) per 10 μg/m^3 increase in 24-hr mean $PM_{2.5}$ concentration (WHO, 2013a, p. 19). Furthermore, COMEAP (1998, p. 16) pointed out that the estimate of risk of mortality associated with short-term exposure to PM_{10} reported by the World Health Organisation is a suitable figure for quantifying acute mortality effects in the UK. Hence, COMEAP (1998, p. 56) recommend a RR of 1.0075 per 10 μg/m^3 increase in 24-hr mean PM_{10}.

In relation to the risk of hospital admission, a RR of 1.008 (1.0048, 1.0112) of hospitalisation from short-term exposure to an increase of 10 µg/m^3 (24 hour mean) in PM_{10} concentration developed by the WHO based on studies conducted in four cities in the US, one in Canada and one in Paris was adopted in COMEAP (1998, p. 17) recommendations. Whilst for the same increase in NO_2 level, COMEAP recommend a relative risk of a 0.5% increase in hospital admission. Moreover, the WHO (2013a, p. 22) evaluated studies in Europe and the US; specifically, three single-city studies conducted in Prague by Braniš *et al.* (2010), Madrid by Linares and Díaz (2010) and the West Midlands (United Kingdom) by Anderson *et al.* (2001); the random effects summary estimate calculated for hospital admissions for all-age respiratory disease was 1.90% (95% CI = -0.18%, 4.02%) per 10 µg/m^3 increase in $PM_{2.5}$ (WHO, 2013a, p. 22).

2.6.3 Health Impact Assessments

In health impact assessments (HIAs), the concentration-response function CRFs derived from epidemiological studies are used to translate estimates of population exposure in terms of air pollutant concentrations into health impact estimates (Briggs *et al.*, 2009). A description of the method used for calculating HIAs by different studies is presented next (Section 2.6.4) followed by the uncertainties associated with HIAs in Section 2.6.5.

2.6.4 Methods for Calculating HIAs

The number of deaths due to long-term exposure to, for example anthropogenic $PM_{2.5}$ concentrations, has been estimated in a number of studies such as those by Anenberg *et al.* (2010) and Ghosh *et al.* (2016) using the following equation:

$$\text{Mort} = y_0 \times \text{AF} \times \text{Pop} \qquad (1.1)$$

where:

Mort: attributable mortality due to long-term exposure to the air pollutant

y_0: baseline mortality rate

AF: attributable fraction of the disease (i.e. mortality)

Pop: size of the exposed population

The number of attributable deaths is a metric that can also be expressed as a rate and is commonly used in the literature. However, this metric does not simply represent the number of people whose length of life was shortened by air pollution (COMEAP, 2010, p. 3). For instance, long-term exposure to air pollution may be a contributory cause of deaths from respiratory and cardiovascular diseases but is unlikely to be the only factor causing these deaths (Gowers *et al.*, 2014, p. 6).

The attributable fraction (AF) of the disease is frequently expressed as a percentage (COMEAP (2010, p. 61), Gowers *et al.* (2014, p. 6) and Anenberg *et al.* (2010)) as follows:

$$AF = \frac{RR-1}{RR} \qquad (1.2)$$

where the following log-linear relationship between the RR and pollutant concentrations, as defined by Anenberg *et al.* (2010), for $PM_{2.5}$-related health impacts is used:

$$RR = \exp(\beta(x-x_o)) \qquad (1.3)$$

In Equation 1.3, β is the estimated slope of the log-linear relationship between air pollutant concentration and mortality (i.e. the CRF) and x is the population-weighted pollutant concentration with a threshold of x_o.

Health impact assessments are performed using air pollutant concentration data sourced by monitoring station observations, satellite remote sensing data and from chemistry-climate models (Butt *et al.*, 2017). For instance, $PM_{2.5}$ concentrations can be simulated using global chemical transport models (Anenberg *et al.*, 2010; Lelieveld *et al.*, 2013; Silva *et al.*, 2016), or through a combination of modelling, satellite remote sensing data, and measured levels of pollution at ground stations, in addition to land-use regressions (Van Donkelaar *et al.*, 2010; Brauer *et al.*, 2012; Brauer *et al.*, 2015; Jerrett *et al.*, 2016).

Countries in South and East Asia typically witness higher $PM_{2.5}$ concentrations compared to the US and Europe where annual mean $PM_{2.5}$ concentrations are below 30 $\mu g/m^3$. Kushta *et al.* (2018) implemented a different definition of the RR. They use the CRF developed by Burnett *et al.* (2014), as described in the equation below, where (a) and (p) are factors derived from statistical models to account for large $PM_{2.5}$ concentrations.

$$RR = 1 + a\,\{1-\exp[-\beta(x-x_o)^P]\} \qquad (1.4)$$

Where β, x and x_o are as defined previously, a and P are parameters that define the overall shape of the concentration–response relationship (Burnett *et al.*, 2014).

Other research has estimated the global burden of disease; for instance, Cohen *et al.* (2017). Global and regional trends such as those described by Butt *et al.* (2017) make use of IER functions for IHD, COPD, lung cancer and lower respiratory infections to estimate the RR for each cause of death using non-linear CRFs covering the range of exposure to pollutant concentrations across the globe.

Additionally, statistical tools to carry out assessments of health impact are available, such as the extensively used Benefit Mapping and Analysis Program (BenMAP) developed by the US Environmental Protection Agency (EPA) (Punger and West, 2013; Thompson *et al.*, 2014). This tool is a geographical information systems program that combines US census-level population and incidence data at country-level resolution with user-supplied air pollution data (from monitors, satellite or models) to estimate health effects (Abt Associates, 2010). Hence, all of the calculations are conducted using the single package BenMap to ensure coherence at every step of the health impact assessment.

Health impact assessments of short-term exposure to air pollution are performed in a similar way to long-term HIAs; nonetheless, short-term HIAs are conducted daily and the terminology used for the health burden is ‘deaths brought forward’. For instance, Macintyre *et al.* (2016) estimated the mortality associated with $PM_{2.5}$ episodes over the UK in spring 2014. The total all-cause mortality related to short-term exposure to $PM_{2.5}$ summed over each day of the air pollution episode is estimated by the following Equation 1.5:

$$\mathrm{Mort} = \sum D_i \times AF_i \qquad (1.5)$$

where D_i is an estimate of the total regional daily baseline mortality. The daily AF of mortality is calculated following Equation 1.2 and using the definition of RR given in Equation 1.6:

$$RR_i = \exp(\beta x_i) \qquad (1.6)$$

where β is the slope of the log-linear relationship between concentration and mortality (i.e. the CRF) and x_i is the daily mean population-weighted $PM_{2.5}$ concentration.

2.6.5 Uncertainties Associated with HIAs

Several areas of uncertainty are associated with the estimation of mortality burdens attributable to long- or short-term exposure to air pollution, including the following:

- The use of a CRF derived from epidemiological studies based in only one geographical location;
- The use of CRFs derived from single-pollutant statistical models;
- Uncertainties related to population and baseline mortality rates; and
- The representativeness of pollutant exposure estimates or pollutant concentrations derived from modelling or observations.

The quantification of the health burden for one pollutant from single-pollutant statistical models might include effects attributable to other pollutants. Consequently, the recommendations by the WHO (2013a, p. 43) in their 'Review of Evidence on Health Aspects of Air Pollution', suggested that for any particular health outcome and exposure period, estimated $PM_{2.5}$ and NO_2 related to health burdens should not be added without acknowledging that this will lead to some overestimation of results. Additionally, the impact estimated for only one pollutant is believed to underestimate the true impact of the pollution mixture, given that other pollutants also affect the same health outcome. For instance, Williams *et al.* (2014) demonstrated that separate analyses of O_3 and NO_2 in epidemiological studies would underestimate the combined effects from exposure to both pollutants on the population.

Different factors can affect the estimation of health impacts related to the uncertainty of associated with pollutant exposure estimates, as well as short- and long-term changes in air pollutant levels. These involve the impact of both the horizontal resolution of the model and the effect of air pollution episodes.

2.6.6 Impact of Model Horizontal Resolution on Pollutant Concentrations and Associated Health Impacts

Differences in horizontal resolution present benefits and drawbacks. Global chemistry-climate models (CCMs) have coarse resolutions (~50-100 km in the horizontal) to enable long-term integration for large areas. Nevertheless, coarse resolutions could lead to a lack of accuracy in representing local or urban effects; for instance, high levels of emissions at pollution hotspots.

Several studies have been conducted to evaluate the effect of changes in the horizontal resolution of a model on measures of different pollutant levels (De Ridder et al., 2014; Thompson et al., 2014). Nevertheless, only a few studies have analysed the impact of changes in the model horizontal resolution on estimated human health burdens (Punger and West, 2013; Thompson *et al.*, 2014; Li *et al.*, 2015a; Kushta *et al.*, 2018). The majority of these studies find that mortality associated with long-term exposure, for example to O_3, is higher since simulated O_3 concentrations are often larger when using coarse compared to finer resolution (Thompson and Selin, 2012; Punger and West, 2013; Thompson *et al.*, 2014).

Less agreement between available studies is found for $PM_{2.5}$-related health estimates, and certain studies have suggested that the attributable deaths associated with short-term exposure to $PM_{2.5}$ are higher at coarse resolutions larger than 100 km grid squares (Punger and West, 2013; Li *et al.*, 2015a). A further study ascertained that using horizontal resolutions lower than 36 km has a negligible effect on changes in $PM_{2.5}$ levels and associated health impacts (Thompson *et al.*, 2014). The uncertainties in ambient $PM_{2.5}$-related mortality in Europe were evaluated by Kushta *et al.* (2018) in relation to model simulations of $PM_{2.5}$ concentrations at 20 km and 100 km. This study suggested that simulated $PM_{2.5}$ levels and corresponding health impacts at the fine resolution of 20 km are higher compared to estimates at the 100 km resolution for most countries in Europe. According to Kushta *et al.* (2018), uncertainties related to mortality estimates are dominated by the estimated CRFs derived from epidemiological studies rather than the representation of annual mean $PM_{2.5}$ concentrations by air quality models which have different horizontal and vertical resolutions. Nevertheless, most of the relevant studies have been carried out in the USA with only a limited number conducted in Europe.

2.6.7 Impact of Air Pollution Episodes on Pollutant Concentrations and Associated Health Impacts

Meteorology has an immense influence on air quality as it influences the chemical reaction rates through changes in temperature; the deposition of $PM_{2.5}$ due to changes in precipitation, and stagnation of air and long-range transport by means of changes in wind direction and speed. Under anticyclonic weather conditions with low wind speeds, the resulting stable conditions can frequently create an inversion of the temperature profile, thus trapping pollutants in the shallow boundary layer close to the ground, resulting in high levels of air pollutants and producing an air pollution episode (Rebetez *et al.*, 2009). In summer, these air pollution episodes could coincide with heatwaves (Solberg *et al.*, 2008; Schnell and Prather, 2017). Additionally, synoptic flows across the UK could lead to the transport of pollutants from across Europe (e.g. Francis *et al.* (2011)) or clean air from the Atlantic Ocean depending on the orientation of the prevailing meteorological system.

Numerous studies have estimated the impact of short-term exposure to particulate matters on human health during air pollution episodes. In particular, a number of studies estimated the mortality burden associated with all-cause mortality due to air pollution during the heatwave in Europe in the summer of 2003. Stedman (2004), estimated that there were 207 additional deaths brought forward due to PM_{10} in the UK during the first two weeks of August 2003, in comparison with the same period in 2002.

Episodes of $PM_{2.5}$ air pollution have also been studied, and Macintyre *et al.* (2016) estimated that a spring-time air pollution episode in 2014 was associated with 600 daily deaths brought forward from short-term exposure to $PM_{2.5}$ in the UK. It was suggested that the mortality burden was 2.0 to 2.7 times that associated with periods when $PM_{2.5}$ levels were more typical.

2.7 Pollutant Concentration Measurements

The commitment of the UK Government to monitor pollution and policy outcomes have led since 1998 to a substantial investment in the measurement of air pollution in all large towns and cities by the establishment of the Automatic Urban and Rural Network (AURN). These monitoring sites use precision monitors which are frequently calibrated and all data is ratified on a three-monthly period (Bell *et al.*, 2013). Levels of pollution concentration of $PM_{2.5}$, PM_{10} and NO_2 are measured at monitoring sites by the means of AURN or can be estimated by employing air quality models such as the ADMS-Urban program.

2.7.1 Automatic Urban and Rural Network (AURN)

In the UK, the AURN is the largest automatic monitoring network, which includes automatic air quality monitoring stations to measure oxides of nitrogen (NO_x), sulphur dioxide (SO_2), ozone (O_3), carbon monoxide (CO) and particles ($PM_{2.5}$, PM_{10}) with high resolution and reported as hourly information. The main aim of the network is for use for compliance reporting against the Ambient Air Quality Directives. Additionally, to increase public awareness of local air quality status.

The role of the Quality Assurance and Control Unit (QA/QC Unit) for the entire AURNs is undertaken by Ricardo Energy and Environment. In addition, the role of the Management Unit for AURNs in London is contracted to the Environmental Research Group (ERG) at

King's College London at the time of the research carried out in this thesis, whilst Bureau Veritas was appointed as the Management Unit for the AURNs in places outside London. The latter responsibility is contracted for fixed periods.

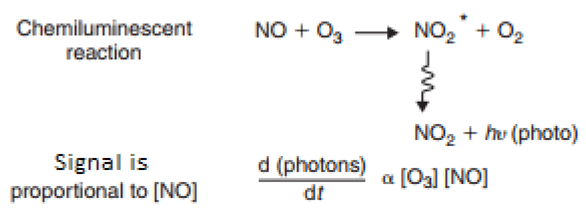
In 2005, after extensive discussion and research across the EU, the European Commission published a series of standard methods for monitoring to ensure the quality of data. These include:

- EN14211:2012 Nitrogen Oxides.
- EN12341:2014 PM_{10} and $PM_{2.5}$.

Standard methods are used as a template for instrument performance testing. The standards specify a series of tests and requirements for analysts to achieve based on both laboratory and field-based studies. The process of such a test is known as Type Approval. Some of the techniques used for monitoring within the UK's national compliance monitoring network, the AURN, are summarised below.

2.7.2 Measurement of Nitrogen Monoxide (NO) and Nitrogen Dioxide (NO_2)

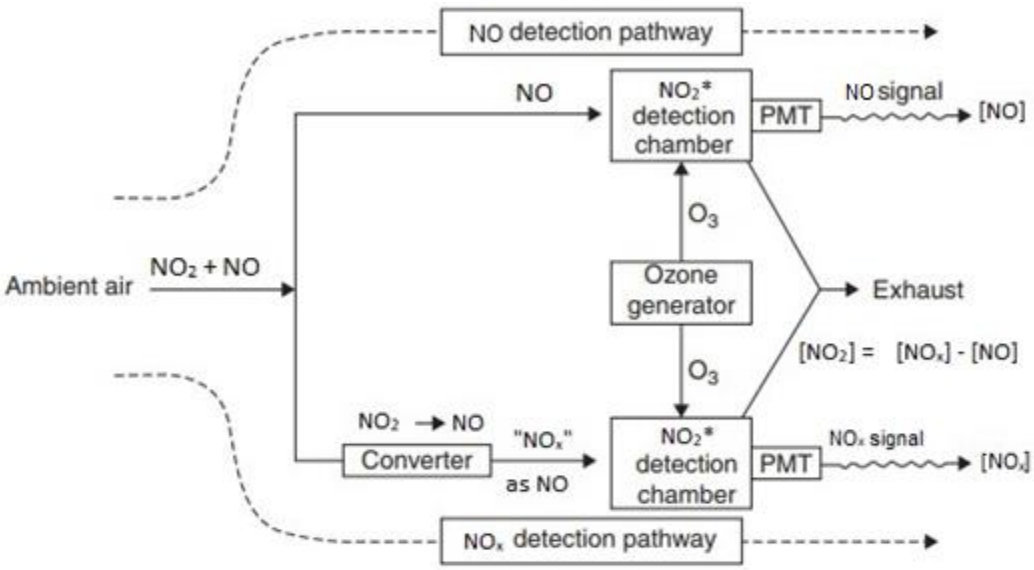
Chemiluminescence is the production of light from a chemical reaction. Two chemicals react to form an excited (high-energy) intermediate, which breaks down releasing some of its energy as photons of light. According to Vallero (2008, p. 476), the principal method used to measure NO_2 is based on chemiluminescence, as shown in Figure 2-3. NO_2 concentrations are determined indirectly from the difference between the concentrations of NO and NO_x (NO + NO_2) in the atmosphere. These concentrations are determined by measuring the light emitted from the chemiluminescent reaction of NO with O_3.



Source: Vallero (2008, p. 476)
Figure 2-3: NO_2 chemiluminescent detection principle based on reaction of NO with O_3

An analytical technique is shown in Figure 2-4 where two pathways are represented. Air passes through the 'NO-pathway' to enter the reaction chamber, where the NO present reacts with generated O_3. The light produced is measured by the photomultiplier tube and converted into NO concentration. The NO_2 in the air stream in this pathway is unchanged.

In the NO_x-pathway, the NO-laden and NO_2-laden air enters the converter, where the NO_2 is reduced to form NO. All of the NO_x exits the converter as NO and enters the reaction chamber. The NO reacts with O_3 and the output signal is the total NO_x concentration. The NO_2 concentration in the original air stream is the difference between the NO_x and NO concentrations.



PMT: photomultiplier tube
Source: Vallero (2008, p. 477)
Figure 2-4: Schematic diagram of chemiluminescent detector for NO_2 and NO

The worthy to note is there are two forms of interference may occur in the chemiluminescent sampler; a reduction in intensity of the chemiluminescence by quenching in the reaction chamber, and bias due to conversion of various N-species to NO taking place in the NO_2-to-NO converter. Whereas a several compounds such as H_2O, O_2, CO_2, CO, and H_2 might give rise to quenching effects, only water vapour is likely to give rise to significant effects. The installation of permeation driers at the sample inlet should prevent such problems occurring and are a common feature of most modern samplers (Saeger *et al.*, 2002). Another method used to measure NO_2 concentration employs diffusion tubes which are passive samplers. They consist of small plastic tubes containing a chemical reagent which absorbs the pollutant to be measured directly from the air. This method is low-cost and takes measurements universally. However, it measures the total amount of pollutant in the air measured during the period of exposure and requires adjustments against bias.

2.7.3 Measurement of Particulate Matter

The separation of the size of particle distribution can be achieved by applying two methods: the filter-based gravimetric method or Tapered Element Oscillating Microbalance (TEOM). The TEOM analyser where a filter is mounted on a tapered glass tube that vibrates continually is the type used at the majority of AURN sites. As the particles collect on the filter, the vibration becomes slower. The change in vibration can be measured very precisely without suspending sampling, giving a continuous measurement of the amount of particulate matter being collected. In the UK, the TEOM analyser is commonly used because it gives results that can be made available immediately. Nevertheless, this method requires that the air passing into the TEOM analyser is heated, which leads to the loss of semi-volatile material such as some organic compounds, ammonium nitrate and water. It is worth mentioning that it is essential to heat the air in the filter-based gravimetric method. As a result, it takes more than 24 hours before continuous measurements can begin (AQEG, 2005, p. 10). Additionally, a default adjustment factor (1.03 × TEOM reading + 3 $\mu g/m^3$) should be applied to TEOM readings. All TEOM analysers in the UK, both for PM_{10} and $PM_{2.5}$ are set up with this default factor.

Several environmental studies have been conducted based on pollution concentrations monitored at fixed sites, such as those by Adar *et al.* (2008) in Seattle, Washington, Abbey *et al.* (1999), Ostro *et al.* (2001) and Blanchard *et al.* (1999) in California; Chen *et al.* (2017a) and Kim *et al.* (2006) in Canada, , An *et al.* (2013) in China, Raaschou-Nielsen *et al.* (2001) in Denmark, Ballester *et al.* (2001) in Spain; as well as Anderson *et al.* (1996) and Atkinson *et al.* (2010) in London and Carey *et al.* (2013) in England.

Although the AURN sites serve as important sources of information in relation to field pollution concentrations through more than 1500 sites across the UK deploy to monitor air quality, such coverage is insufficient to monitor street pollution (Chen *et al.*, 2008). According to Briggs *et al.* (2009), most air pollution monitoring networks are controlled according to the demands of regulatory compliance. They focus on obtaining time-series data and target known or suspected pollution hotspots. Additionally, the AURN sites could provide biased assessments of exposure.

Therefore, many epidemiological studies have turned to the use of modelling to assess human exposure, since it is difficult if not impossible to carry out direct measurements for every individual person over the full exposure of a period of interest. It can be argued that most epidemiological studies involve air quality models used to assess human exposure to air pollution (Briggs *et al.*, 2009). Likewise, assessments of risk and health impact depend on modelling to generalise monitored data across the whole population as well as to predict future exposure assessing the effect of different scenarios (for example offering different policy options for intervention measures).

2.8 Personal Exposure to Air Pollution

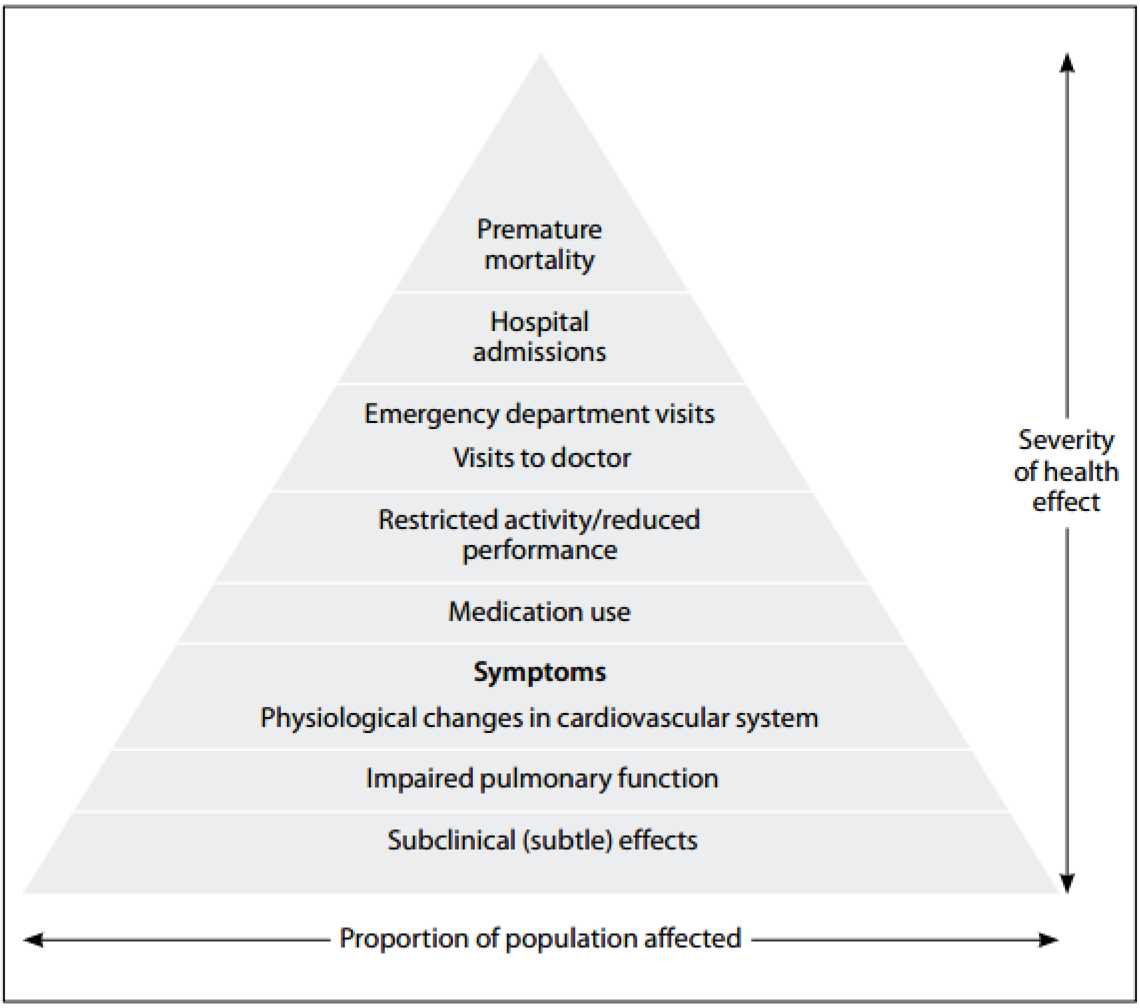
The survival of humankind is essentially based on continuous supplies of air, water and food. Demand for air is continuous and it is estimated that one adult at rest consumes an average of 12 m^3 of air on a daily basis (WHO, 2010; U.S. EPA, 2011, pp. 6-11). Thus, it is crucial that the air which is breathed is pure. However, due to the growing demand for energy and transport activities, air contamination has reached significant levels and occasionally breaches national air quality objectives, particularly in urban areas. Hence, exposure to contaminated air is inevitable, in particular in urban areas where traffic flow is the main contributor to air pollution (Chatterton, 2011; Hitchcock *et al.*, 2014, p. 8). Exposure to air pollution refers to a particular time in a place where the pollution exists, which triggers an interaction between air contaminants and the interior of the human body through breathing and/or externally by means of irritation to the skin and eyes (Ott, 1982). Thus, exposure depends on other factors, such as physical activity that increases breathing rates, and exposure duration and intensity, which have short and long-term consequences. According to the WHO (2006, p. 89), health consequences range from losing working days to death. Usually, the effects are attributed to long-term and chronic exposure to air pollution such are DALYs and COPD, whilst studies on short-term exposure correlate the acute effects of air pollution with occurrence of daily mortality and hospital admissions (WHO, 2006, p. 89). Table 2-5 demonstrates the negative health outcomes of short- and long-term exposure to air pollution.

These outcomes range according to their severity. They start with the most widespread minor symptoms among the population. Subsequently, as the severity of the exposure increases, less of the population is affected. Finally, the severity reaches its limit causing early death. Fortunately, this is less common, as the pyramid in Figure 2-5 illustrates.

Table 2-5: Health effects due to long-term exposure to air pollution

Effects attributed to short-term exposure	
1	Daily mortality
2	Respiratory and cardiovascular hospital admissions
3	Respiratory and cardiovascular emergency department visits
4	Respiratory and cardiovascular primary care visits
5	Use of respiratory and cardiovascular medications
6	Days of restricted activity
7	Work absenteeism
8	School absenteeism
9	Acute symptoms (wheezing, coughing, phlegm production, respiratory infections)
10	Physiological changes (e.g. lung function)
Effects attributed to long-term exposure	
1	Mortality due to cardiovascular and respiratory disease
2	Chronic respiratory disease incidence and prevalence (asthma, COPD, chronic pathological changes)
3	Chronic changes in physiologic functions
4	Lung cancer
5	Chronic cardiovascular disease
6	Intrauterine growth restriction (low birth weight at term, intrauterine growth retardation, small for gestational age)

Source: WHO (2006, p. 89)

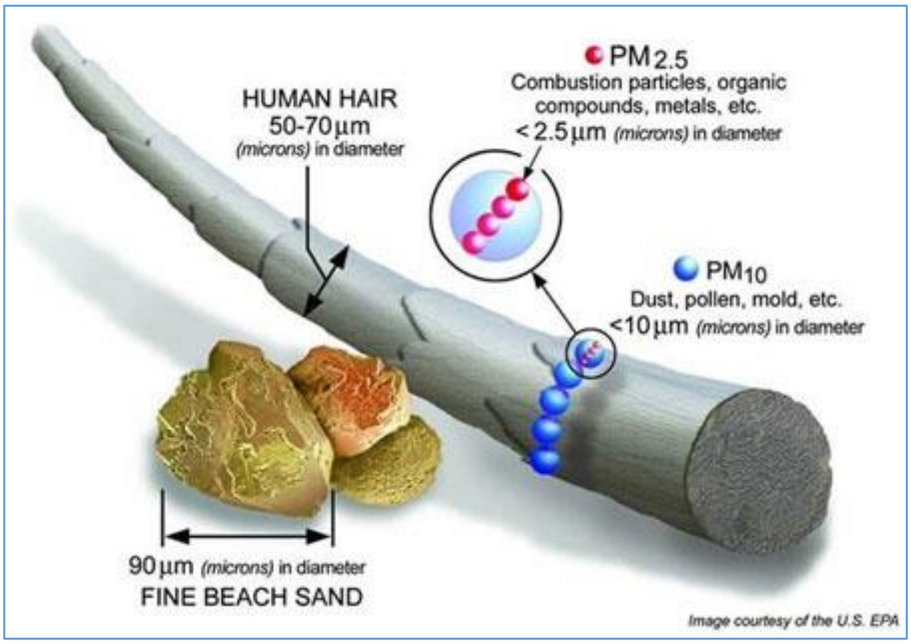


Source: WHO (2006, p. 91)

Figure 2-5: Pyramid of health effects related to air contamination

2.8.1 Dose-Response Exposure Coefficients for Selected Pollutants

Particulate matter comprises fine liquid droplets which are naturally and anthropogenically created. Wind-blown dust, volcanic ash, forest fires, sea salt and pollens originate from natural sources, whilst energy plants, industry, commercial and residential facilities, in addition to road transport, produce anthropogenic particulates. As previously explained, a particulate that has an aerodynamic diameter of 2.5 microns or less is defined as a fine particle (i.e. $PM_{2.5}$), whereas a particulate that has an aerodynamic diameter of 10 microns is known as a coarse particle (i.e. PM_{10}), while the mean thickness of a human hair is from 50 to 70 microns. Figure 2-6 compares the sizes of particulate matter with human hair.



Source: U.S. EAP (2017)

Figure 2-6: Comparison of particulate matter sizes

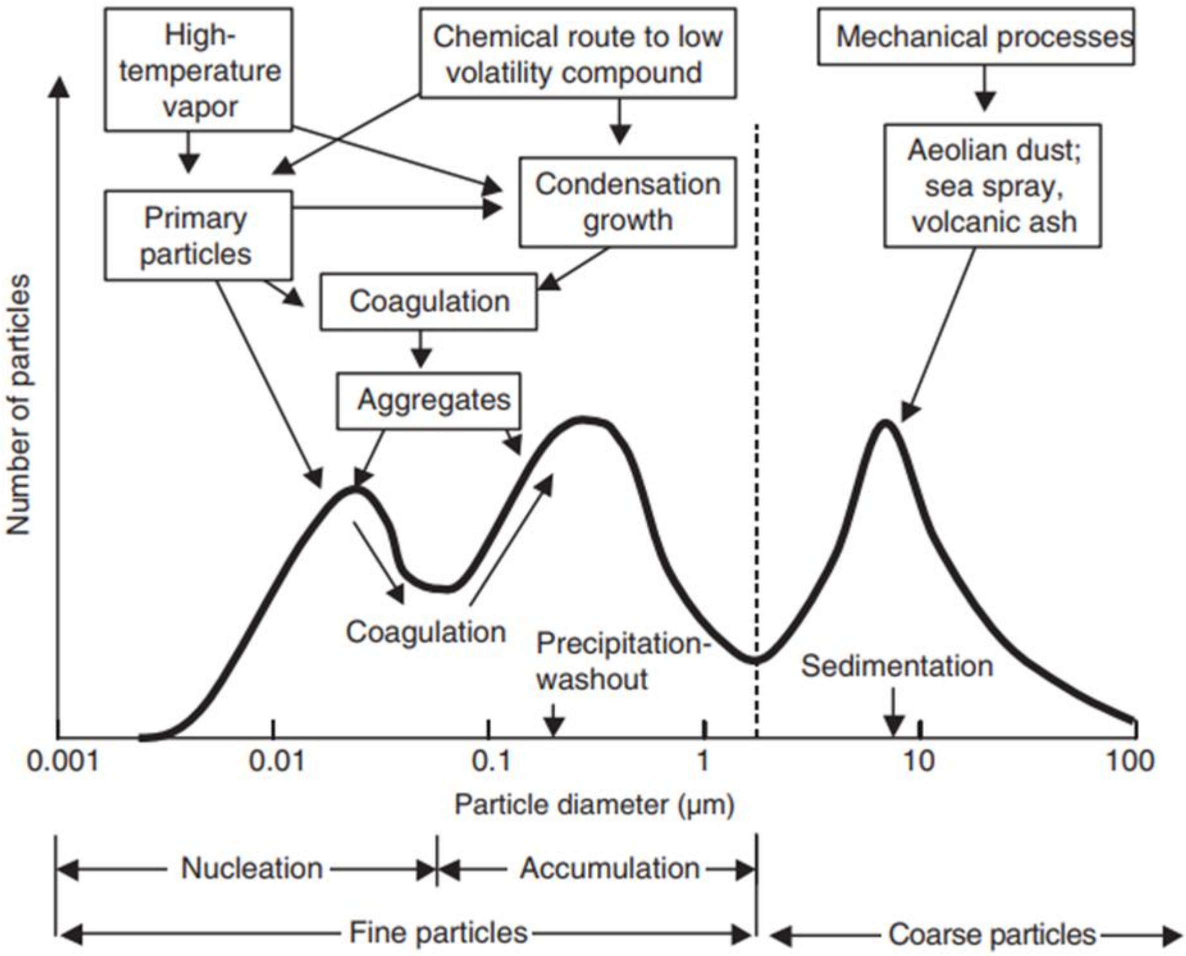
Particulate matter is minute in size, and causes medical issues for humans due to its ability to penetrate deeply into the respiratory system. Some may pass into the bloodstream and accumulate over time. This accumulation makes particulate matter a danger that is waiting to happen (U.S. EAP, 2017). NO_x is a species comprising NO and NO_2, besides other compounds such as N_2O which is a GHG, dinitrogen trioxide (N_2O_3), and nitrogen pentoxide (N_2O_5). This species is emitted into the air naturally from lightning, volcanic eruptions, and the action of bacteria in the soil. In addition, the combustion of fuels in internal combustion engines, thermal power plants, industrial and heating facilities, and incinerators are considered anthropogenic sources. Generally, both NO and NO_2 are collectively known as NO_x which is primarily released by traffic flow emissions. NO is the dominant contributor to this type of emission and emitted directly from the combustion of fuel, whilst most NO_2 forms arise indirectly by means of the oxidation of NO (Onursal and Gautam, 1997, p. 17). Together particulate matter and NO_x are principally emitted by road transport and can accumulate in the human body over time. Their relevance is discussed in the following sections.

2.8.2 Differences between Fine $PM_{2.5}$ and Coarse PM_{10} Particulate Matter

The United Nations Framework Convention on Climate Change defines an aerosol as '*particulate matter, solid or liquid, larger than a molecule but small enough to remain suspended in the atmosphere'* (UNFCCC, 2014). Particulate matter contains a complex mixture of inorganic ions, metallic compounds, elemental carbon, organic compounds and crustal substances (Vallero, 2008, p. 60) that differ in size ranging from a few nano-metres to tens of micro-metres in diameter, as shown in Figure 2-7. In addition, aerosols can be emitted directly as particles (primary aerosol) from stationary sources such as factories, power plants, and open burning, and from mobile sources by direct emissions resulting from burning of fossil fuel in internal combustion engines. Also, particulate matter can be released from non-exhaust emissions as a consequence of wear and tear regarding brakes and tyres and road dust resuspension (Woodcock *et al.*, 2009). Secondary aerosol can be produced in the air by gas-to-particle conversion processes.

Once airborne, particles can form and change in their composition and size due to a number of processes such as nucleation, which is the growth of clusters of molecules that become a thermodynamically stable nucleus. Nucleation and gas-to-particle conversion are examples of secondary aerosol formation processes. Additional processes are by condensation and coagulation. Condensation is the result of collisions between a gaseous molecule and an existing aerosol droplet when supersaturation exists, while the coagulation of aerosols is a process by which discrete particles come into contact with each other in the air and remain joined together due to surface forces (Vallero, 2008, p. 431). Particles are removed from the atmosphere through dry deposition on the Earth's surface and wet deposition by means of precipitation. The composition and concentration of tropospheric aerosols vary extensively over the Earth, while residence times for particles in the troposphere also vary from a few days to a few weeks (Seinfeld and Pandis, 2006, p. 57).

The size of components in atmospheric aerosols varies. Particles with a diameter of less than 2.5 μm are typically referred to as fine particulate matter or $PM_{2.5}$ and those less up to 10 μm in diameter are referred to as coarse particulate matter or PM_{10}. In general, fine particles are divided into nucleation and accumulation modes, as shown in Figure 2-7. Particles in nucleation mode are typically small in size with a diameter between 0.005 to 0.1 μm and hence do not often account for more than a few per cent of the total mass of airborne particles. These particles are formed due to the condensation of hot vapours during combustion processes and from the nucleation of atmospheric gaseous species such as sulphur and nitrogen oxides. Meanwhile fine particles in the accumulation mode (ranging from ~ 0.1 to 2.5 μm in diameter), account for most of the aerosol surface area as well as a significant part of the aerosol mass in the troposphere. Particles in the accumulation mode are produced from the coagulation of particles in the nucleation mode and from the condensation of vapours onto existing particles, which causes them to grow in size (see Figure 2-7).

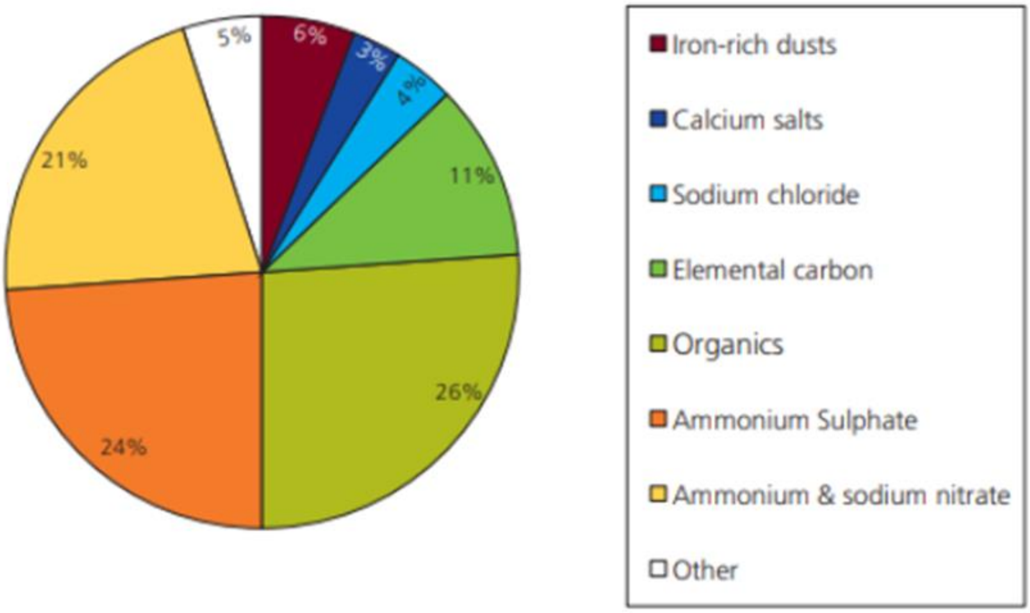


Adapted from: Vallero (2008, p. 61)
Figure 2-7: Schematic of the distribution of particle surface area of an atmospheric aerosol with the dashed line representing a diameter of approximately 2.5 µm

Generally, coarse mode particles (> 2.5 µm in diameter) are formed by mechanical processes and typically consist of anthropogenic and natural dust particles including sea spray. In addition, particles in nucleation mode rapidly coagulate and grow larger because of the condensation of vapour species. Moreover, coarse particles are removed from the atmosphere over a relatively short time due to sufficient large sedimentation velocity. Hence particles in the accumulation mode have a considerably longer atmospheric residence time (Seinfeld and Pandis, 2006).

2.8.2.1 Chemical Composition and Sources of $PM_{2.5}$

Fine particulate matter ($PM_{2.5}$) is made up of a number of chemical species or components; for instance, sulphate (SO_4^{2-}), sodium (Na^+), ammonium (NH_4^+), elemental carbon (EC), nitrate (NO_3^-), organic carbon (OC), sea salt and mineral dust. Figure 2-8 shows an example of the chemical composition of $PM_{2.5}$ at a central background site in Birmingham in the UK between May 2004 and May 2005.



Source: Yin and Harrison (2008)
Figure 2-8: Contribution of chemical composition to total $PM_{2.5}$ at a central urban background site in Birmingham, UK

The primary pollutants are generated from natural and anthropogenic sources. Natural sources of primary $PM_{2.5}$ comprise ash from volcanic activity, mineral dust, sea salt and forest fires.

The form of secondary inorganic particles can be formed from the nucleation of new particles and gas-to-particle conversion through the oxidation of primary gases such as NO_2 and sulphur dioxide (SO_2) into nitric acid (HNO_3 - gas) and sulphuric acid (H_2SO_4 - liquid) (Jacob, 1999, p. 251). Ammonia (NH_3) gas is primarily released from the agricultural sector and on reaction with HNO_3 and H_2SO_4 forms ammonium nitrate and ammonium sulphate; however, it reacts more readily with sulphate aerosol due to its lower vapour pressure.

Secondary organic aerosol formation predominantly takes place via the oxidation of VOCs. This formation can be highly complex and involves biogenic, anthropogenic and biomass burning VOC species.

2.8.2.2 Toxicity of PM

The toxicity of particles can be determined from the size and number of particles and their chemical composition and age. One of the main theories on the determination of particle toxicity is the 'ultrafine hypothesis' (Seaton *et al.*, 1995) which proposes that the number of particles is the driving factor as opposed to their mass. This is because fine particles dominate the particle number distribution but not the particle mass distribution, and they have high surface area. Evidence that some individual $PM_{2.5}$ components (such as elemental carbon) are more toxic than others has also been reported (Levy *et al.*, 2010). It should be noted that there is a dearth of studies comparing the toxicity of fine $PM_{2.5}$ and coarse PM_{10} particles (WHO, 2013b, p. 16).

2.8.3 Personal Exposure Due to Variable Traffic Flows

Vehicular emissions in ambient air play a significant role in air pollution, causing adverse health outcomes, and growing evidence suggests a strong association between traffic pollution and mortality and morbidity related to residents who live near heavy vehicle traffic flows (Künzli *et al.*, 2000; Adar and Kaufman, 2007; Beelen *et al.*, 2008; Delfino *et al.*, 2008). In a Canadian study, people who live less than 100 metres from highways or less than 50 metres from major urban roads are subjected to relative risks (RRs) of between 1.35 and 1.40 regarding cardiovascular disease mortality, and RRs of between 1.79 and 1.85 for cerebrovascular mortality (Finkelstein *et al.*, 2005). In a German study, Gehring *et al.* (2006) determined RRs of 1.70 for cardio-pulmonary mortality in a cohort of females in their 50s who lived within 50 metres of roads with high traffic flow. According to Ghosh *et al.* (2016), living in close proximity to roads that have high traffic densities is likely to increase the risk of coronary heart disease (CHD) mortality and/or morbidity. Even though more clean vehicles are expected by 2035, the risk of CHD tends to be higher because of the expected increase in vulnerable elderly people among the population (ibid.). It should be noted that Type 2 diabetes is likely to be attributed to long-term exposure to particulate matter related to local traffic rather than other sources. Weinmayr *et al.* (2015) established that a higher proportion of diabetic patients lived in proximity to urban roads than among the general population. Meanwhile Volk *et al.* (2011) suggested that there is an association between children who are autistic and mothers who lived close to freeways during pregnancy. The development, persistence and exacerbation of asthma in children is strongly attributed to vehicular emissions (Gowers *et al.*, 2012; Guarnieri and Balmes, 2014). In addition, living in proximity to roads is linked to decreased lung growth and the development of asthma and, as a consequence, hospitalisation (Eder *et al.*, 2006).

A Dutch study that collected 10 years of valuable data estimated that the relative risk of living in the vicinity of a major road is 1.19 (95% CI, 0.91-1.56) in relation to causing respiratory mortality (Beelen *et al.*, 2008). Table 2-6 presents the association of different causes of mortality with traffic variability.

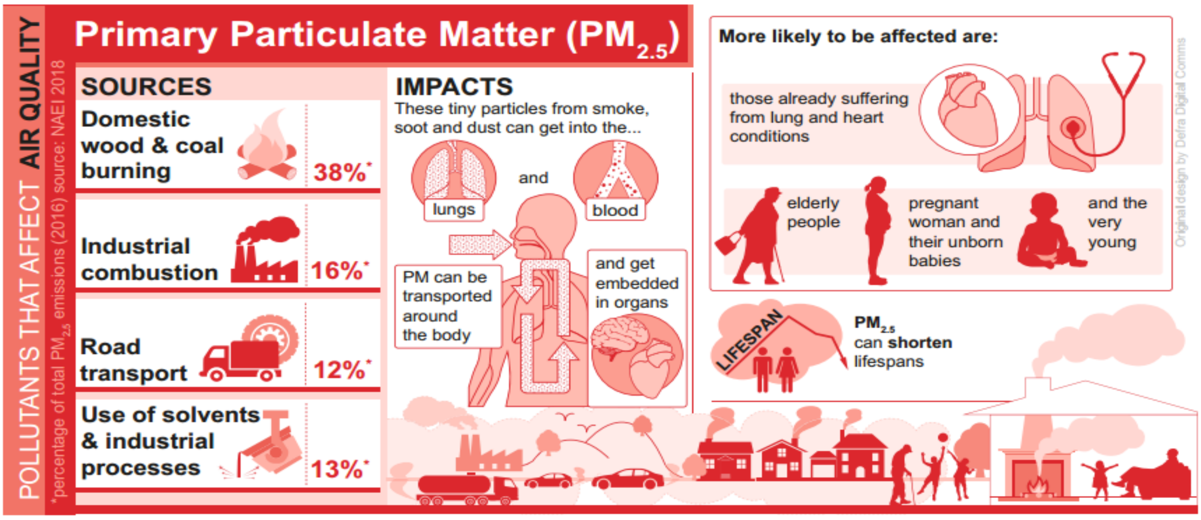
Table 2-6: Relative risks (95% CIs) for the relationship between intensity of traffic and cause-specific mortality

Exposure model	Full cohort	Case cohort
Natural-cause mortality		
Traffic intensity on nearest road	1.03 (1.00–1.08)	0.99 (0.88–1.11)
Traffic intensity in a 100-m buffer	1.02 (0.97–1.07)	0.98 (0.85–1.13)
Living near a major road	1.05 (0.97–1.12)	0.92 (0.74–1.15)
Cardiovascular mortality		
Traffic intensity on nearest road	1.05 (0.99–1.12)	1.03 (0.90–1.17)
Traffic intensity in a 100-m buffer	1.00 (0.92–1.08)	0.98 (0.82–1.16)
Living near a major road	1.05 (0.93–1.18)	0.93 (0.72–1.21)
Respiratory mortality		
Traffic intensity on nearest road	1.10 (0.95–1.26)	0.94 (0.71–1.25)
Traffic intensity in a 100-m buffer	1.21 (1.02–1.44)	1.23 (0.89–1.68)
Living near a major road	1.19 (0.91–1.56)	0.85 (0.50–1.43)
Lung cancer mortality		
Traffic intensity on nearest road	1.07 (0.96–1.19)	1.03 (0.87–1.22)
Traffic intensity in a 100-m buffer	1.07 (0.93–1.23)	1.10 (0.85–1.43)
Living near a major road	1.20 (0.98–1.47)	1.07 (0.70–1.64)
Other mortality		
Traffic intensity on nearest road	1.00 (0.94–1.06)	0.93 (0.82–1.06)
Traffic intensity in a 100-m buffer	0.99 (0.93–1.06)	0.93 (0.80–1.07)
Living near a major road	0.98 (0.88–1.09)	0.85 (0.68–1.07)

Source: Beelen *et al.* (2008)

2.8.4 Fine Particulate Matter ($PM_{2.5}$) and its Effect on Health

Sources of $PM_{2.5}$ emissions may be natural such as sea spray and pollen or human-made including road transport which accounts for 12% of $PM_{2.5}$ emission in the UK (DEFRA, 2019a, p. 16). Current annual $PM_{2.5}$ emissions in the UK are 107.9 kilotons, as estimated in 2016 (DEFRA, 2018). These emissions should decrease to 89 kilotons and 69 kilotons by 2020 and 2030 respectively (DEFRA, 2019a, p. 97). Figure 2-9 shows the sources of $PM_{2.5}$ and the associated health impact.



Source: DEFRA (2019a, p. 16)

Figure 2-9: Sources of $PM_{2.5}$ and related health impact

The effects of long-term exposure to $PM_{2.5}$ were quantified by COMEAP (2010) as causing 29,000 deaths and 340,000 life years lost. The increase in exposure to $PM_{2.5}$ causes diseases that are extremely costly for the NHS and extra social care is required to deal with the consequences. In England, the cost was estimated to be £76 million. This amount was divided into £10.45 million for primary care, £36.83 million for secondary care, £18.81 million for medication and £10.01 million for social care in 2017 (PHE, 2018, pp. 28, 31). If an appropriate plan is not made to reduce levels of $PM_{2.5}$, cumulative expenditure between 2017 and 2035 will reach £9.41 billion, representing 1.3 million new patients in England which is equivalent to 2,250 per 100,000 of the population (ibid.).

Many epidemiological studies have investigated the level of $PM_{2.5}$ and have showed that it is a strong indicator of adverse health effects. In a Dutch study of data collected from 1987 to 1996, Beelen *et al.* (2008) estimated the long-term association between exposure to an increase of 10 µg/m^3 in $PM_{2.5}$ levels and cause-specific mortality. They estimated a relative risk of 4% [95% confidence interval (CI), -10%, 21%] for cardiovascular mortality, 7% (-25%, 52%) for respiratory mortality and 6% (-18%, 38%) for lung cancer mortality. Their estimation of the all-cause mortality relative risk was 6% (-3%, 16%). A study conducted on an English cohort population aged over 40 estimated that an annual increase of 10 µg/m^3 in mean $PM_{2.5}$ concentration is associated with an increase in mortality of 13% (95% CI: 0, 27%) (Carey *et al.*, 2013), whilst COMEAP (2009) estimates that deaths would increase by 6% (2%, 11%). According to Jerrett *et al.* (2005), who controlled 44 individual variables in Los Angeles, an increase in all-cause mortality of 17% (5%, 30%) is associated with a 10 µg/m^3 increase in $PM_{2.5}$ concentrations. Additionally, a study undertaken in Massachusetts for the years 2000 to 2008 reported a higher RR of a 1.6% (1.5%, 1.8%) increase in deaths linked to10 µg/m^3 of $PM_{2.5}$ (Kloog *et al.*, 2013). Table A-1 in the Appendices shows the relative risk of mortality per 10 µg/m^3 of long-term exposure to $PM_{2.5}$ found in various studies.

Meanwhile, Kloog *et al.* (2013) indicated that the short-term exposure to 10 µg /m^3 on a daily basis of $PM_{2.5}$, is likely to cause an increase of 2.8% (2%, 3.5%) for all causes of mortality. The WHO (2013a, p. 6) estimated a lower RR of 1.23% (95% CI: 0.45%, 2.01%), whilst Atkinson *et al.* (2014) reported that a daily increase of 10 µg/m^3 of $PM_{2.5}$ is likely to be associated with a RR of a 1.04% (95% CI: 0.52%, 1.56%) increase in mortality. They conducted a meta-analysis of previous studies of the relative risk of short-term exposure to $PM_{2.5}$ in terms of mortality and hospital admissions. Table A-2 in the Appendices shows the relative risk of mortality per 10 µg/m^3 of short-term exposure to $PM_{2.5}$ in selected studies.

Moreover, short-term exposure to an increment of 10 µg/m^3 of $PM_{2.5}$ is likely to be linked to a 0.96% (-0.63%, 2.58%) increase in hospital admissions (Atkinson *et al.*, 2014). Nevertheless, the WHO (2013a, p. 7) estimated a higher dose-response of 1.90% (-0.18%, 4.02%) for the same daily dose. Also, short-term $PM_{2.5}$ concentrations have been positively associated with an increase in the risk of hospital admission with respiratory disease of 2.1% (95% CI: 1.2%, 2.95%) (Zanobetti *et al.*, 2008). Table A-3 in the Appendices shows the relative risk of hospital admissions per 10 µg/m^3 of short-term exposure to $PM_{2.5}$ in some studies.

Increasing long-term exposure to $PM_{2.5}$ is likely to increase hospitalisation for respiratory disease, according to Pannullo *et al.* (2017) study in England between 2007-2011. Their research noted that the burden of respiratory hospital admission increased by 0.8% (-0.3%, 2.4%) per 4.2 µg/m^3. Lee and Sarran (2015) found that exposure to an increase of only 1 µg/m^3 will increase the probability of being admitted to hospital by 3.2% (0.5%, 6%).

$PM_{2.5}$ not only causes direct adverse health effects, but also indirect effects via vehicle accidents. A reduction in road visibility (haze) while driving is principally attributed to the presence of $PM_{2.5}$ (U.S. EAP, 2017).

It is worth mentioning that the UK's national objective is to lower $PM_{2.5}$ levels below an annual average of 25 µg/m^3 and 12 µg/m^3 in Scotland by 2020, whereas Newcastle and Gateshead have not recorded any case of exceeding the $PM_{2.5}$ threshold from 2011 to 2015 (Gateshead Council, 2016, p. 30; Newcastle City Council, 2016, p. 12). However, any decrease in the current level of $PM_{2.5}$ is likely to enhance the population's current health status.

2.8.5 Coarse Particulate Matter (PM_{10}) and its Effect on Health

Exposure to PM_{10} has an influence on health. Fortunately, Newcastle and Gateshead have not witnessed any breaches in relation to the threshold of national air quality objectives for annual concentrations of PM_{10}. Therefore, Gateshead Council stopped recording PM_{10} concentrations in 2012.

A short-term study conducted in the Middle East suggests an association of RR of 3.9% (95% CI: 3.3%, 4.5%) for all causes of mortality, 4.2% (95% CI: 2.7%, 9.05%) for cardiovascular mortality and 6.2% (95% CI: 4.2–16.9%) for respiratory mortality, with an increase in exposure to PM_{10} of 10 μg/m^3 (Khaniabadi *et al.*, 2017). Nevertheless, the above-mentioned RRs differ from the findings reported by the WHO (2004) of 1.006 (1.004,1.008) in all-cause mortality for European cities. This agrees with findings from the Middle East where it was reported that, during 2012-2015, an increase in the daily mean exposure of 10µg/m^3 led to an increase of 0.6% (0.4%-0.8%) in total mortality (Nikoonahad et al., 2017), which suggests that Khaniabadi's findings may be slightly exaggerated. COMEAP (1998, p. 56) estimate that this relative risk is 0.75%. Table A-4 in the Appendices shows the relative risk of mortality per 10 µg/m^3 of short-term exposure to PM_{10} in various studies.

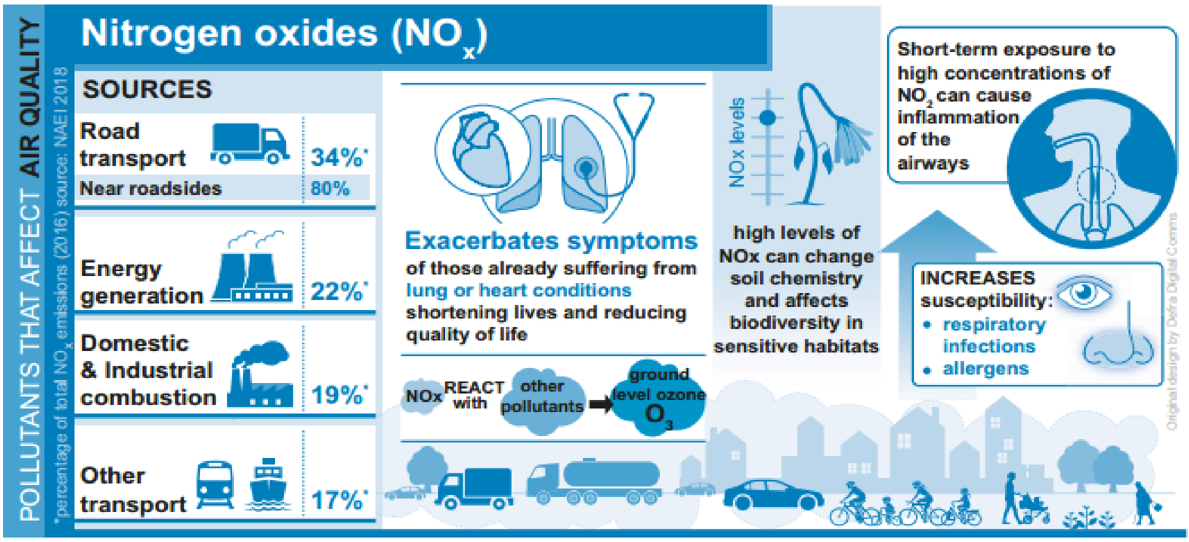
With regard to hospitalisation due to short-term exposure, Nikoonahad *et al.* (2017) estimated a RR of 0.8% (0.48%, 1.12%) of respiratory disease hospitalisation, which is similar to the estimation from COMEAP (1998, p. 56). Romeo *et al.* (2006) examined time series and panel studies of the short-term effects of PM_{10} concerning increases in respiratory illnesses in childhood. The results revealed that exposure to PM_{10} was linked to an increase in hospitalisations for treatment for asthma. Additionally, the exposure of children who suffer from asthma to PM_{10} was linked to the frequency of asthmatic symptoms (coughing and wheezing) and the use of anti-asthma medication, as well as regular therapy and a reduction in lung function. Hajat *et al.* (1999) established a connection between daily General Practitioner (GP) consultations for asthma and other lower respiratory diseases and air pollution in London. The most noteworthy correlations were detected in children with NO_2, CO and SO_2 identified as significant contributors. In adults, the only consistent correlation was observed with regard to PM_{10}. It should also be mentioned that in São Paulo, Brazil, visits to COPD emergency departments were associated with daily ambient concentrations of PM_{10}, SO_2, NO_2, CO and O_3 (Arbex *et al.*, 2009). Furthermore, PM_{10} and SO_2 readings were associated with both acute and lagged effects regarding visits to COPD emergency department, Medina-Ramón *et al.* (2006) conducted a study in 36 US cities of hospital admissions for COPD and pneumonia and their associations with O_3 and PM_{10} levels. The research established an increased risk of COPD and pneumonia admissions linked with ambient concentrations of PM_{10} and O_3. It is worth noting that their study covered a significant number of cities and scrutinised more years of follow-up than previous multi-city studies examining the respiratory effects of PM_{10} and O_3 (Medina-Ramón *et al.,* 2006). Table A-5 in the Appendices summarises several RRs in relation to hospital admissions related to short-term exposure to PM_{10}.

Long-term exposure to an increase in the annual mean of 10 μg/m^3 for PM_{10}, is linked to an RR of 3.5% (95% CI: 0.4%, 6.6%) of mortality, according to Hoek *et al.* (2013), whilst Carey *et al.* (2013) estimated an RR 7% (95% CI: -1%, 16%) in an English population above 40. Table A-6 in the Appendices shows values of RR in relation to mortality for an increase of 10 μg/m^3 for long-term exposure to PM_{10}.

Long-term exposure to PM_{10} contributes to increasing hospitalisation rates. In one study conducted in Scotland, hospital admissions associated with lengthy exposure to $PM_{10,}$ revealed a 28.8% (95% CI: 27.2%, 29.9%) increase in hospital admissions per 10 μg/m^3 increase in PM_{10} in annual mean concentrations (Huang, 2016). Lee and Sarran (2015) established that an increase in long-term exposure to an incremental 1 μg/m^3 of PM_{10} would increase the probability of being admitted to hospital by 0.8% (-1.1%, 2.7%). In the study area of this thesis, a study carried out on people registered in 64 GP practices in Newcastle and North Tyneside discovered a 1% increase in prescription of salbutamol correlated with a 10 μg/m^3 increase in ambient PM_{10} levels (Sofianopoulou, 2011). This confirms the potential of long-term exposure to pollutant concentrations in increasing respiratory disease incidence.

2.8.6 Nitrogen Dioxide (NO_2) and its Effect on Health

In the UK, emissions from the transport sector, and especially NO_2, are a particular hazard to people's health. Most AQMAs have been declared because NO_2 levels breached the national objectives. This pollutant causes serious health problems. It is estimated that 34% of NO_x pollution can be attributed to road transport, as shown in Figure 2-10 which also shows the health impact from exposure to NO_x (DEFRA, 2019a, p. 18).



Source: DEFRA (2019a, p. 18)

Figure 2-10: Sources and health impact of NO_x

In a Dutch study, Beelen *et al.* (2008) estimated the relative risk of being exposed in the long-term to 10 µg/m^3 increases in NO_2 concentrations to be 1.02 [95% confidence interval (CI), 0.98–1.07] for cardiovascular mortality, 1.12 (1.00–1.29) for respiratory mortality and 1.03 (1.0-1.05) for all-cause mortality. Whereas the WHO (2013a, p. 10) suggested an RR per 10 µg/m^3 for NO_2 of 1.055 (1.031-1.080) for all causes of mortality for populations aged 30 and older. In an English cohort with a population aged 40 and older, the RR is 3% (95% CI: 0, 5%) (Carey *et al.*, 2013), which is similar to the prediction of 2.5% made by COMEAP (2015b). Table A-7 in the Appendices shows the relative risk of mortality per 10 µg/m^3 of long-term exposure to NO_2 found in selected studies.

According to Mills *et al.* (2015), mortality associated with short-term exposure to a daily mean increment of 10 µg/m^3 in NO_2 concentrations is associated with a 0.71% RR (95% CI 0.43% to 1%), whilst the WHO (2013a, p. 11) predicted 0.27% (0.16%, 0.38%) although this was on an hourly average basis. Table A-8 in the Appendices shows the relative risk of mortality per 10 µg/m^3 of short-term exposure to NO_2 found in various studies.

Furthermore, hospital admissions would increase by 0.5% for an increase in increments of 10 µg/m^3 in NO_2 concentrations (COMEAP, 1998, p. 56). A similar RR of 0.57% (95% CI: 0.33%, 0.82%) has also been estimated (Mills *et al.*, 2015). Furthermore, the WHO (2013a, p. 11) provided an RR of 0.15% (-0.08%, 0.38%) on an hourly basis and 1.8% (1.15%, 2.45%) on a daily mean basis. Mills *et al.* (2015) reviewed 204 time-series studies up to 2011 of the association between short-term exposure to NO_2 and hospital admission and mortality for several diagnosis and age groups. Table A-9 in the Appendices shows relative risk of hospitalisation per 10 $\mu g/m^3$ of short-term exposure to NO_2 in selected studies. Regarding the increases in hospitalisation with respiratory disease as a result of increased long-term exposure to NO_2, a dose-response coefficient of 8.5% (5.2%, 11.8%) per an increase of 5 $\mu g/m^3$ dose of NO_2 concentration has been found (Lee and Sarran, 2015). For the same dose, Pannullo *et al.* (2017), suggest a relative risk of 8.5% (5.2%, 11.8%) in their study in England, whilst another estimate per 10 $\mu g/m^3$ increase in long-term exposure to NO_2 suggest a relative risk of 11.25% (3.75%, 18.75%) (Lee *et al.*, 2009).

It is also worth noting that increasing exposure to NO_2 causes diseases that are very expensive for the NHS and social care organisations to deal with. In England, the cost was estimated to be £81 million in 2017 (Pimpin *et al.*, 2018). Unless a feasible plan is made to reduce levels of NO_2, this expenditure will reach £9.16 billion between 2017 and 2035. Moreover, around £3.7 billion will need to be spent to cover the social care cost (Pimpin *et al.*, 2018), with cases of new diseases increasing from 109 per 100,000 of the population in 2017 to 1,933 per 100,000 of the population in 2035 (PHE, 2018, p. 30). A study conducted in eight cities in North America explored the connection between ambient concentrations of five pollutants and the exacerbation of asthma (daily symptoms and the use of inhalers) among 990 children (November-September 1995). It was determined that PM_{10} and O_3 had no association with such exacerbation, although noticeable correlations were discovered with respect to CO and NO_2 (Schildcrout *et al.*, 2006). Research focusing on COPD patients living in rural areas of England between 2006 and 2007 was unable to establish any positive relationship between PM_{10}, O_3 and COPD, although positive relationships were detected between CO, NO_2 and COPD admissions (Sauerzapf *et al.*, 2009).

2.9 Summary of Dose-Response Coefficients for Exposure to Air Pollutants

A summary of important relative risk estimates due to short-term exposure is provided in Table 2-16, whilst Table 2-7 shows relative risk due to long-term exposure. These were determined to be suitable for consideration in this work because they were developed by UK governmental bodies or affiliations or based on UK research. The coefficients sited are used in chapter 7 to evaluate the disease burden resulting from changes in pollutant concentrations.

Table 2-7: Relative risk coefficients associated with short-term exposure to PM_{10}, $PM_{2.5}$ and NO_2

Pollutant	Health Outcome	Coefficient for exposure to daily mean of an incremental 10μg/m^3	Reference
PM_{10}	Death	+0.75%	(COMEAP, 1998)
	Respiratory hospital admission	+0.80%	(COMEAP, 1998)
$PM_{2.5}$	Death	1.23% (95% CI: 0.45%, 2.01%)	(WHO, 2013a)
	Respiratory hospital admission	1.90% (95% CI: -0.18%, 4.02%)	(WHO, 2013a)
		2.07% (1.20%, 2.95%) [2-day averaged]	(Zanobetti *et al.*, 2008)
NO_2	Death	0.71% (95% CI 0.43%, 1.00%)	(Mills *et al.*, 2015)
	Respiratory hospital admission	0.50%	(COMEAP, 1998)

Table 2-8: Relative risk coefficients associated with long-term exposure to PM_{10}, $PM_{2.5}$ and NO_2

Pollutant	Health Outcome	Coefficient for exposure to annual mean of an incremental 10µg/m^3	Reference
PM_{10}	Death	7% (95% CI: -1%, 16%), age 40+	(Carey *et al.*, 2013)
	Respiratory hospital admission	1.4% (0.3%, 2.4%) % per 1.872µg/m^3	(Huang, 2016, p. 159)
		7% (1%, 13%) % per 1.7µg/m^3	(Lee *et al.*, 2009)
		1.57 % (-0.2%, 3.92%)	(Pannullo *et al.*, 2017)
		8% (-11%, 27%)	(Lee and Sarran, 2015)
$PM_{2.5}$	Death	6.2% (4%, 8.3%), age 30+	(WHO, 2013a)
		6% per 10µg/m^3 (2%, 11%)	(COMEAP, 2009)
		13% (0, 27%), age 40+	(Carey *et al.*, 2013)
	Respiratory hospital admission	1.9% (-0.71%, 5.69%)	(Pannullo *et al.*, 2017)
		32% (5%, 60%)	(Lee and Sarran, 2015)
NO_2	Death	2.5% (1%, 4%)	(COMEAP, 2015a)
		5.5% (3.1%, 8%), age 30+, Rang > 20 µg/m^3	(WHO, 2013a)
		2% (0, 5%), age 40+	(Carey *et al.*, 2013)
	Respiratory hospital admission	11.25% (3.75, 18.75%)	(Lee *et al.*, 2009)
		1.67% (0.84%, 2.93%) 8.5% (5.2%, 11.8%) per 5µg/m^3	(Pannullo *et al.*, 2017)
		17% (10.4%, 23.6%) per 10µg/m^3	(Lee and Sarran, 2015)

2.10 Summary

Information from the main regulatory organisations which formulate air quality regulation has been highlighted in this chapter. It was shown that most of the regulations were legislated by the European Union. Ambient Air Quality Directives (AQD) are usually a combination of all the previous EU air quality laws were enshrined in legislation in England by means of the Air Quality Standards Regulations 2010. It was outlined that the UK has failed to comply with AQD standards but succeeded in lowering total harmful emissions, according to the National Emissions Ceiling Directive (NECD) and the Climate Change Act. The governmental target of converting the entire car stock to be made up of ultra-low emission cars by 2050, commencing with the restriction of new car sales to only ultra-low emission cars, was also pointed out.

The impact on health in terms of all-cause mortality and respiratory hospital admissions was introduced as functions per 10 μg/m^3 incremental increases in concentrations of $PM_{2.5}$, PM_{10} and NO_2. These functions which describe the probability of premature death and being admitted to hospital are essential in evaluating the disease burden as a result of changes in pollutant concentrations as discussed in chapter 7.

CHAPTER 3

3. Literature Review of Low Carbon Cars

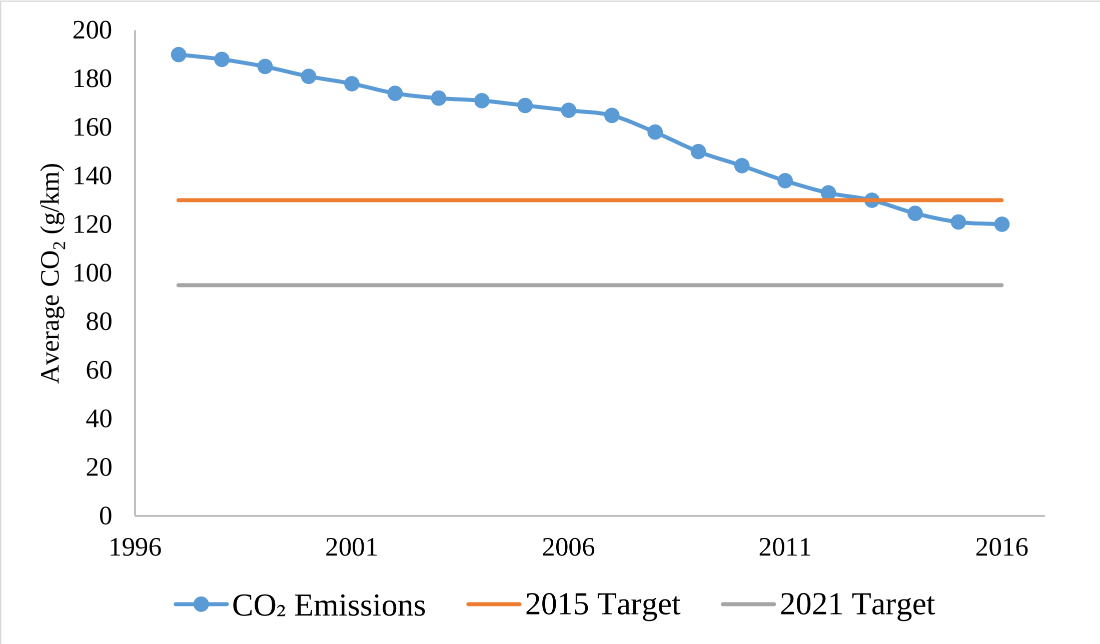
3.1 Introduction

This chapter provides an overview of low carbon cars. It starts with the required targets for reductions in average CO_2 emissions caused by new cars in Europe. Subsequently, the differences between diesel and petrol engines are examined in Section 3.4, followed by a description of technologies used to abate harmful emissions. An overview is also provided of alternative fuel vehicles focusing on electric vehicles and the related infrastructure. Finally, the effects of electric vehicles on energy, carbon dioxide and air quality are explained in Section 3.9 to give an idea of the possible role of vehicle policy in reducing emissions and improving air quality.

3.2 Target of Car Fleet Average Carbon Dioxide (CO_2)

From the beginning of the 1990s, new developments in technology have advanced due to the implementation of legislation and regulation. Emissions regulations ensured that catalytic converters and fuel injection had to be installed in vehicles. Thus catalytic converters were commercially installed in most cars in Europe (King, 2007, p. 42), which has therefore decreased harmful emissions. These initiatives have succeeded, and CO_2 emissions released by the transport sector in 2016 (124.4 $MtCO_2e$) have stayed at 1990 levels (125.3 $MtCO_2e$) (BEIS, 2018b, p. 8), despite the vast increase in vehicle registrations from 25 million in 1994 to 37 million in 2016 in Great Britain as published in DfT statistics in Table VEH0101 (Department for Transport, 2018). However, the contribution of the transport sector in releasing GHG emissions increased from 16% of the total in 1990 to 27% in 2016, where road transport accounted for 90% of those emissions. This is because of the big reductions in GHG emissions released by other sectors. For instance, a reduction took place in CO_2 emissions in the energy sector from 242 $MtCO_2e$ in 1990 to 113.7 $MtCO_2e$ in 2016 (BEIS, 2018b, p. 8).

Average new car discharges registered in the UK were 121 g/km of CO_2 in 2017 (SMMT, 2018, p. 4), which is below the 2015 European target of 130 g/km and 0.8% greater than the 2016 level. This was reinforced by an increase of 34.8% in the registration of alternatively-fuelled vehicles (AFVs) (SMMT, 2018, p. 10). Also the share of sales of new diesel car reduced by 17.1% compared to the previous year, given that diesel cars typically emit 15%-20% lower CO_2 compared to their equivalent petrol cars (SMMT, 2018, p. 4). Attaining the European target of reducing average CO_2 emissions below 95 g/km by 2021 for new cars is not an obligation, given that as the UK is due to leave the European Union. Nevertheless, annual reductions of 5.9% should be achieved via this target (SMMT, 2018, pp. 7, 26). Figure 3-1 shows the CO_2 emissions of average new cars in the UK from 1997 to 2016.



Source: SMMT (2017, p. 4)

Figure 3-1: UK average new car CO_2 emissions - registration weighted

The parameters used to calculate the limit curve for each car model are presented below.

$$CO_2 = \text{Target} + a\,(M - M_o)$$

where,

Target: Target fleet average (130 g/km in 2015; 95 g/km in 2020).

a: Gradient of the line (0.0457 in 2015; 0.0333 in 2020).

M: Mass of vehicle.

M_o: Average mass of new passenger cars in previous 3 years (1372 Kg in 2015; 1392.4 Kg in 2016).

Furthermore, car emissions of CO_2 should be significantly reduced by 2050 if the entire car fleet is formed by ultra-low emission cars only, starting from limiting the sales of new cars to ultra-low emission cars in 2040 (DEFRA and DfT, 2017b, p. 4). The initiatives introduced by the UK government aim not only to reduce emissions of CO_2 only but also of other air pollutants. The UK government is investing more than £1.5 billion between 2015 and 2021 to provide better air quality and cleaner transport; for example, in its strategy for Road to Zero (DfT, 2018a, p. 45). Achieving 95 g/km average CO_2 new car emissions by 2021 might be difficult but not as difficult, as making all car sales in 2040 of ultra-low emission cars.

3.3 The UK's Passenger Cars

The number of passenger cars has increased globally in recent decades. For example, in Europe, car manufacturing reached 18.5 million cars in 2015, representing an increase of 18% on what was recorded 5 years previously (OCIA, 2018). Consequently, more than 252 million passenger cars were running on the roads of the European Union countries in 2015, an increase by 4.5% on 2010 (ACEA, 2017, p. 3). This increase means that passenger cars contribute 12% of CO_2 emissions (EC, 2015), even though the fuel consumption of passenger cars has significantly improved in the European Union over the past few years (Fontaras and Dilara, 2012).

In 2017, statistics published by the DfT (see Table VEH0203) indicate that there were 32 million licensed passenger cars in the UK. Petrol cars represented 58.3% and diesel cars 40.1%, with the remainder comprising hybrid (1.3%); gas (0.1%) and electric (0.1%) vehicles (DfT, 2018i). In the same year, the DfT's Table NTSS0901 reveals that diesel vehicles travel an average of 10,100 miles annually, which is 55% higher than that of a petrol vehicle (6,500 miles) (DfT, 2018c). This might be attributed to the fact that many work cars and vans are diesel-powered and able to achieve greater mileage. Furthermore, in the UK, the total VKT in 2017 was 526 billion vehicle kilometres, 78% of which were car- and taxi-driven, as revealed in Table TRA0206 of the DfT statistics (DfT, 2018f). Although the registration of petrol-cars is higher than that of registration of diesel-cars, they achieve nearly the same VKT.

3.4 Differences Between Petrol and Diesel Fuel

Basic differences exist in the chemical nature of petrol and diesel fuels and the engineering of the engines. Petrol- and diesel-run vehicles both use fossil fuel in an internal combustion engine (ICE). Fractional distillation is a process by which crude oil is split up into several fractions or portions with accompanying differences in boiling points in each component part (Babich and Moulijn, 2003). Some of these components are diesel and petrol, which differ in their CO_2 emissions, mainly due to their distinctive chemical composition. Petrol is lighter than diesel, and thus less diesel is required to produce the same amount of energy, and consequently lower CO_2 emissions are emitted from diesel vehicles. Also, diesel engines have a more efficient fuel combustion process which requires higher temperatures; but, it is at higher temperatures that NO_x is produced, although with lower CO_2 emissions (Schipper *et al.*, 2002).

3.4.1 Combustion in Diesel Engines

Also referred to as the compression ignition (CI) engine, a diesel engine contains a mixture of air and fuel during combustion, which is the mechanism wherein fuel is vaccinated into the combustion chamber under high pressure at >2000 bars. What follows is that the fuel heats up to a high temperature, and ignites. However, again due to the diesel's dense composition, if at one point the air and fuel do not easily blend, this could result to irregular combustion, thereby allowing the formation of particulates due to incomplete combustion (Überall *et al.*, 2015).

It has been said that, for complete combustion to occur, an exact combination of diesel and air is needed according to a certain ratio termed 'stoichiometric'. If the diesel engine operates with a higher quantity of air (where the engine runs 'lean'), this combustion results in lower CO_2 emissions and more heat. Although contributing to the efficiency of diesel engine, this leads to the formation of NO_x due to the reaction of nitrogen in the air with oxygen (Zheng *et al.*, 2004).

3.4.2 Combustion in Petrol Engines

Petrol engines are called spark or positive ignition (PI) engines. There are two different types of petrol engine, namely conventional port fuel injection (PFI) engines and a more recent type, the gasoline direct injection (GDI) engine.

3.4.2.1 Port Fuel Injection (PFI)

When the fuel is injected through an intake track at low pressure and is made to start by an external energy source (a spark) which supplies a localised high temperature, this is referred to as a PFI petrol engine. Where diesel is dense, petrol fuel is considerably lighter and more readily evaporates, allowing an efficient mixing of the fuel with air in the combustion chamber. A consequent small spark then produces a smooth combustion in the well-mixed combustion chamber. Unlike diesel, petrol operates at a much lower air-fuel ratio, allowing the production of smaller amounts of particulates. This is due to the complete burning of the fuel.

3.4.2.2 Gasoline Direct Injection (GDI)

In theory, the GDI type of engine has higher fuel efficiency and, for that reason, it has lower CO_2 emissions in contrast to the PFI (Überall *et al.*, 2015). In consequence, Saliba *et al.* (2017) have recently reported a high market share of GDI engines at ~50% of new petrol vehicles.

Similar to the diesel engine, GDI creates a lean air-fuel mixture, hence producing a higher air ratio. The petrol fuel is ignited after it is injected directly to the combustion chamber at a higher pressure of 200 bars. Since a higher air ratio is produced, less fuel is consumed coupled with lower levels of CO_2 emissions. According to Überall *et al.* (2015), GDI engines provide fuel consumption savings of 5–10% compared to the equivalent PFI engines.

Yet, still, compared to PFI, GDI emits higher amounts of particulates, but lower than a diesel vehicle without a DPF (Überall *et al.*, 2015). Apart from these particulates, GDI also produces a greater amount of smaller ultrafine particles <100-nm diameter, which are damaging to human health (Wang *et al.*, 2014a).

Nevertheless, the introduction of DPF has successfully caused reductions in diesel particulate emissions (Mathis *et al.*, 2005). In parallel, the introduction of the gasoline particulate filter (GPF) will have a similar reduction effect for GDI engines (Chan *et al.*, 2012).

3.5 Technologies to Abate Harmful Emissions

Certain common treatments of exhaust emissions are used to reduce harmful emissions from both petrol and diesel cars. Among them, five prominent examples are discussed here.

3.5.1 Three Way Catalyst (TWC)

A three-way catalyst (TWC) is a catalytic converter fitted into modern petrol vehicles which uses the chemical processes of oxidation and reduction to turn destructive pollutants into non-disruptive by-products. A reduction of ~95% in CO, NO_x, and HC emissions is caused by the reactions made possible by the precious metals platinum, rhodium, and palladium distributed over a 3-D ceramic honeycomb structure to augment or take advantage of the surface area (Santos and Costa, 2008).

The combustion process is completed with a TWC through the reactions as follows:

- NO_x is reduced to nitrogen and oxygen ($2NO_x \rightarrow O_2 + N_2$)
- CO is oxidized into CO_2 ($2CO + O_2 \rightarrow 2CO_2$)
- Unburnt HC is oxidised into CO_2 and H_2O

It is worth noting that the ideal conditions for all three of the aforementioned reactions to take place is when the air-fuel ratio vacillates around the stoichiometric ratio of air-fuel in a petrol engine. Incidentally, the TWC will only be functional for stoichiometric engines where the O_2 concentration is <1%.

3.5.2 Diesel Oxidation Catalyst (DOC)

As the name implies, a DOC works only in diesel engines. Because TWC does not work in diesel engines, NO_x emissions tend to be at higher levels. In addition, since diesel engines run lean, operating considerably above the stoichiometric ratio, the exhaust gases from diesel-run vehicles emit higher levels of oxygen. In effect, the second and third reactions taking place in a petrol engine are favoured at the expense of the first reaction; and to this extent, the process is damaging. While the DOC can effectively reduce CO and HC, it is not capable of removing NO_x. The reason why diesel vehicles have lower CO and HC emissions is due to the oxidising effect of the lean exhaust gases. Because DOC also oxidises some of the NO to NO_2, diesels produce a higher percentage of primary NO_2, as found by Carslaw *et al.* (2016).

3.5.3 Exhaust Gas Recirculation (EGR)

EGR may be said to be another abatement technology deployed by diesel passenger cars satisfying Euro 5 and some later models to reduce NO_x formation (Weiss *et al.*, 2012). As pointed out above, high temperature expedites the oxidation of nitrogen, thus resulting in a rise in NO_x formation. The role of the EGR is to lower the temperature of combustion in order to reduce NO_x formation, which is accomplished by taking a proportion of the exhaust gas and returning it into the combustion chamber. Since the exhaust gas is inert (void of oxygen), some of the heat energy created during combustion is absorbed by the exhaust gas resulting in a decrease in peak combustion temperature and subsequently the formation of less NO_x (Maiboom *et al.*, 2008).

Nevertheless, EGR, on its own is inadequate to reduce the formation of NO_x so as to meet Euro 6 requirements (Weiss *et al.*, 2012).

3.5.4 Selective Catalytic Reduction (SCR)

Only in recent years has SCR been used in diesel passenger cars (Malpartida *et al.*, 2012). It is installed after the DOC and DPF in passenger cars.

SCR was initially used in large municipal waste boilers and later in large diesel engines on ships and trains, and then buses and HGVs have made use of SCR.

The primary reaction in the SCR process is the formation of NH_3 and CO_2 when AdBlue, the most common diesel exhaust fluid (DEF), is mixed with exhaust gases and is rapidly hydrolised. AdBlue is 30% high purity urea dissolved in deionised water. Ofoli (2014) stated that, when NH_3 and NO_x pass into the SCR catalyst, a reaction is activated by a honeycomb structure of precious metals, turning the NO_x into nitrogen and water. In the final phase, the gases traverse an oxidation catalyst, turning the remaining NH_3 into N_2 and H_2O. Because the ammonia catalyst cuts down only the NO_x in an oxidising environment, this explains why SCR is termed selective reduction.

It is important to note that the SCR process is susceptible to some degree of error, in that when a higher quantity of DEF is injected, NH_3 is likely to be released into the atmosphere. However, when the amount is low, a deficiency in NO_x conversion is to be expected. This is termed the ‘ammonia slip’. The occasional refilling of the DEF tank is another problem. SCR is also said to be more effective at removing NO_x when a higher ratio of primary NO_2 is present (Malpartida *et al.*, 2012).

3.5.5 Lean NO_x Trap (LNT)

The LNT is the most contemporary NO_x reduction technology on the market. It literally traps NO_x emissions from a lean burn engine. Exhaust gases pass through filtering over alkali or alkaline-earth metal oxides which remove the NO_x and store it as nitrites and nitrates, resulting in a periodic release and reduction of the accumulated NO_x. Larson *et al.* (2008) further mentioned that the intermittent release and reduction are stimulated by the creation of reducing conditions, with lowering levels of O_2 only for a short interval in the rich engine operation, generating reductants like CO, H_2 and HC, and therefore reducing the NO_x to N_2 and O_2. However, it is worth mentioning that the formation of detrimental by-products such as N_2O and NH_3 is still possible.

3.5.6 Particulate Filters (DPF and GPF)

Since 2009, all Euro 5 and after diesel vehicles, have been fitted with a DPF to lower particulate matter emissions. PFI vehicles do not require a particulate filter because, during stoichiometric combustion, only low counts of particulates are produced. It is only GDI petrol vehicles that need a GPF in order to pass the Euro 6 PN threshold. Mayer *et al.* (2002) and Liu *et al.* (2003) reported that particulate emissions were reduced by up to 99% using DPFs. However, according to the same authors, because of an increase in back pressure which needs additional mechanical repair, a slight increase in CO_2 emissions by 2–5% would be likely to occur. The primary concern with DPFs is the formation of high levels of ultrafine particulate emissions which usually happens during regeneration (Hawker *et al.*, 1998; Giechaskiel *et al.*, 2007). Regeneration occurs without warning when the DPF reaches >600°C, which usually occurs during motorway driving. Even so, DPFs could be successful in filtering the majority of soot particles, since these filtration devices have a honeycomb structure made of microscopic channels through which exhaust gases pass. These channels block the soot particles which are stored on the microscopic channel walls, but they must be burned off on a regular basis during occurrences of regeneration (Mathis *et al.*, 2005).

3.6 Alternative Fuel Vehicles

As a result of the lower carbon content of diesel fuel, diesel conventional vehicles (CVs) have lower CO_2 emissions in comparison to petrol CVs. Nevertheless, diesel CVs are associated with greater f-NO_2, NO_x, and PM_{10} emissions compared to petrol CVs (Rhys-Tyler *et al.*, 2011). In contrast to petrol fuel, diesel fuel contains more potential energy and lasts longer in CV engines; consequently, most European vehicle fleets have become more dependent on diesel vehicles (Schipper, 2011). Therefore, research continues to be conducted into alternative fuels that reduce both CO_2 emissions and poisonous air pollutants.

Currently, several fuels are considered alternatives to petrol and diesel. Fuels that can be employed in CVs with very few adjustments, including liquid petroleum gas (LPG), hydrogen and biofuels, are able to directly compete with petrol and diesel. Conversely, other fuels, for instance compressed natural gas (CNG), for which engines are extremely expensive to convert, remain reasonably uncompetitive with CVs (King, 2007, p. 25).

3.6.1 Ultra-Low Emission Vehicles (ULEVs)

Ultra-low emission vehicles (ULEVs) are defined as vehicles that produce CO_2 emissions, via their exhausts, less than 75 g/km (OLEV, 2013a, p. 16; DfT, 2018a, p. 24; SMMT, 2018, p.

9). Regarding expected advancements in vehicle technology, by 2021, the DfT (2018a, p. 24) define ULEVs as cars or vans that emit less than 50 gram of CO_2 from the exhaust per kilometre measured against the relevant test cycle. Ultra-low emissions vans are eligible for incentives of 20% discounts up to £8,000 if their range is more than 10 miles with less than 75 g/km of CO_2 emissions (OLEV, 2018a). However, only passenger cars that cause less than 50 g/km of CO_2 emissions for at least 70 miles are eligible to have 35% discounts on the purchase cost up to £3,500, which increases to 20% up to £7,500 if the car is licensed as a taxi (OLEV, 2018a). In addition, an incentive of 75% of the price up to the £500 cost of setting up the home-charger to recharge vehicles equipped with a rechargeable battery is offered to customers who wish to install home-charging (OLEV, 2016, p. 8).

3.6.1.1 Liquefied Petroleum Gas (LPG) Vehicles

Liquefied petroleum gas (LPG) has been extensively used in commercial vehicles (Li *et al.*, 2007). In the UK, there are approximately 150,000 LPG vehicles and more than 1,400 filling stations that provide LPG (UKLPG, 2017). LPG consists of three major constituents: iso-butane, n-butane and propane, which are known as LPG alkanes (Lai *et al.*, 2009). The higher butane content of LPG in relation to diesel fuels produces lower NO_x emissions, and likewise LPG comprises fewer carbon molecules than petrol or diesel which results in a decrease in CO_2 emissions (Saleh, 2008). Generally, quantities of particulate matter and NO_x emissions released by LPG are similar to those of petrol fuel (DfT, 2018a, p. 125). Furthermore, LPG is low-cost, easily stored and has a high combustion efficiency, making it an attractive alternative to traditional fuels (Gumus, 2011). Nevertheless, the efficiency of LPG engines is lower than that of diesel engines in term of producing energy, which might influence the total GHG released by LPG engine fuel (DfT, 2018a, p. 125).

3.6.1.2 Biofuel Vehicles

Biomass is defined by the EU Renewable Energy Directive as the biodegradable fraction of products, waste and residues of biological origin from agriculture, including vegetable and animal substances and those from forestry and related industries including fisheries and aquaculture, as well as the biodegradable fraction of industrial and municipal waste. Moreover, biomass can be utilised to manufacture biofuels that can be employed as an alternative fuel to petrol or diesel (Acquaye *et al.*, 2012). Biofuels comprise four principal types. Biodiesel is prepared from the esterification of vegetable oils and methanol; alcohols (e.g. methanol, ethanol) are created from fermented sugar crops (grain crops); natural gas (methane) is manufactured from the digestion of energy crops; and second generation biofuels, for instance dimethyl ether (DME) and synthetic biodiesel, are produced from synthetic processes and the gasification of lignocellulosic biomass (Biofuels Research Advisory Council, 2006). The latter development was as a consequence of increased deforestation and rising food prices owing to the production of biofuels from food crops, and on account of these pressing issues, methods were developed that generated biofuels from non-feed stock biomass (Pandey *et al.*, 2011, p. 74). These biofuels are known as second generation and are produced in a different way to other biofuels.

Emissions from biofuels tend to fluctuate depending on the make-up of the fuel (Hammond *et al.*, 2008). A synopsis of the emissions levels from the burning of biofuels in relation to those from traditional petrol/diesel is shown in Table 3-1. Even though CO_2 emissions from vehicles which utilise alcohols or biodiesel as fuels are usually lower than those of standard vehicles, these liquid biofuels include less energy than petrol or diesel (Biofuels Research Advisory Council, 2006). Consequently, additional fuel has to be burnt in order to cover the same distance as an equivalent conventional vehicle. This indicates that biodiesel or alcohols have comparable CO_2 emissions to those of diesel- or petrol-fuelled vehicles (Gaffney and Marley, 2009).

Table 3-1: Emissions from biofuel combustion corresponding to emissions from the combustion of conventional petrol/diesel fuel

Biofuel	Regulated Pollutant Emissions	Non-regulated Pollutant Emissions	CO_2 Emissions from Exhausts
Alcohols (Ethanol/methanol)	Reduce except for NO_x which may rise	Increase considerably, particularly aldehydes	Similar to petrol/diesel
Natural Gas (Methane)	Reduce CO, NO_x and HC	-	Lower CO_2 (although methane is a powerful GHG)
Biodiesel	Up to 80% increase in NO_x, PM_{10} and benzene, 60% fewer CO	A rise in aldehydes	Identical to diesel

Source: Gaffney and Marley (2009)

3.6.1.3 Hydrogen Vehicles

Oxygen and hydrogen can be combined in a fuel cell to produce power which can be employed to move a vehicle, releasing only water vapour (Bento, 2010), and these types of vehicles are known as hydrogen fuel cell vehicles (HFCVs). Hydrogen can be produced in large quantities and an HFCV can be quickly refuelled compared to petrol/diesel vehicles and has high energy efficiency by weight, which makes it a highly economical fuel. Additionally, HFCVs have zero exhaust emissions and if they are fuelled with hydrogen generated with electricity from renewable sources, subsequently do not produce any emissions when the required energy is generated (Martin *et al.*, 2009). Nonetheless, only managed use has resulted in HFCVs entering into the worldwide vehicle fleet (Martin *et al.*, 2009), due to several obstacles which are restricting their spread, the principal factors being storage and a lack of refuelling infrastructure (Hu and Green, 2011). Moreover, given that hydrogen fuel is highly flammable and susceptible to leaks, concerns have also been raised regarding safety hazards (Houf *et al.*, 2012; Sáinz *et al.*, 2012). Additionally, questions have also been asked because the low density of hydrogen restricts on-board storage in a vehicle (Mori and Hirose, 2009). Thus, development in storage technologies and refuelling are necessary to make sure that hydrogen becomes a more sustainable fuel (King, 2007, p. 28; Park *et al.*, 2011).

3.6.2 Electric Vehicles (EVs)

According to the technology used, EVs can be categorised into four classes, which are:

- The battery electric vehicle (BEV) relies purely on electricity for propulsion that is stored in the battery pack. The battery pack with an electric motor replaces the ICE and fuel tank (OLEV, 2011, p. 13). The BEV has more range and battery capacity than other electric vehicles. The Nissan Leaf and Renault Kangoo Express ZE are examples of BEVs (Element Energy, 2013, p. 13).

- The hybrid electric vehicle (HEV) uses both an ICE and an electric powertrain, including a battery pack that can be recharged by converting kinetic energy such that from braking to electricity (Alvarez *et al.*, 2010).

- The plug-in-hybrid electric Vehicle (PHEV), such as the Toyota Prius Plug-in (Element Energy, 2013, p. 13) is driven by battery power until reaching depletion mode and then the ICE takes over to propel the vehicle (Element Energy, 2010, p. 70). Generally, distances driven by the electric motor are approximately 50-100 km (Li *et al.*, 2017), which is sufficient for the average journey length travelled by UK drivers (OLEV, 2013a, p. 37). In addition, the battery pack can be recharged externally by connecting it to any outlet charging point whether at home, work or in public. Internal charging is possible through regenerative braking and decelerating.

- The extended-range electric vehicle (E-REV) is similar to the PHEV but, when the battery is depleted, an ICE run by fossil fuel is used to feed the electric motor with electricity and to charge the battery pack simultaneously (Li *et al.*, 2017). The Opel

Ampera and Chevrolet Volt Renault Kangoo Express ZE are examples of E-REVs, (Element Energy, 2013, p. 13).

3.6.3 Battery Electric Vehicles (BEVs)

The BEV is propelled solely by an electric motor and hence it releases zero exhaust emissions. This makes it an ideal solution to improve the environment and mitigate climate change, particularly if it is introduced in core urban areas where the stop-go pattern is dominant (Li *et al.*, 2017; Mahmoudzadeh Andwari *et al.*, 2017) due to traffic congestion or for bus journeys. The BEV runs only on electricity stored in its battery, and therefore the deployment of large numbers of charging stations, in particular the rapid type, are crucial for substantial BEV sales to take place (Levinson and West, 2017). Furthermore, individuals concerned about the daily maximum range of the BEV, are likely to be encouraged to own a BEV after experiencing driving one for a few miles to decide on the suitability of the BEV range (Franke *et al.*, 2017). Studies that evaluated the relevant demographic factors reveal that well-educated middle-aged males, particularly individuals who work in technical professions, tend to adopt BEVs (Li *et al.*, 2017).

3.6.4 Hybrid Electric Vehicles (HEVs)

Hybrid electric vehicles (HEVs) are different from CVs in that they employ an ICE powertrain in addition to an alternative powertrain (Alvarez *et al.*, 2010). Due to the wide availability of ICE infrastructures, it is expected that HEV penetration will be likely to be higher than that of the fuel cell vehicle (Chi *et al.*, 2016). Additionally, HEVs can be categorised into different levels, for instance micro, full, mild or plug-in, depending on the hybrid components and technology involved. A comparison of the technology used and vehicle characteristics associated with the various hybridisation levels is illustrated in Table 3-2 (Fontaras *et al.*, 2008).

Table 3-2: Hybrid electric vehicle (HEV) classifications relating to hybridisation levels

Vehicle operation	Conventional vehicle	Belt/muscle/micro hybrid	Mild hybrid	Full hybrid	Plug-in hybrid
Engine shut-off	Yes	Yes	Yes	Yes	Yes
Regenerative braking		Yes	Yes	Yes	Yes
Smaller IC engine compared to conventional			Yes	Yes	Yes
Electric drive				Yes	Yes
Electric grid battery recharge					Yes

Source: Fontaras *et al.* (2008)

3.6.5 Plug-in Hybrid Electric Vehicles (PHEVs)

The PHEV is distinct from a traditional HEV as a result of its higher battery capacity, the way the two powertrains work together, and the electrical output via the plug which enables batteries to be recharged (Silva *et al.*, 2009). The way PHEV powertrains work together relies on their arrangement; therefore, PHEVs can be categorised as a parallel configuration or series (Doucette and McCulloch, 2011). In a parallel configuration, an electric motor helps the ICE to generate power for the vehicle; in addition, it recaptures energy during the deceleration stage (Zhai *et al.*, 2011). In a series configuration, the mechanical output of the ICE is transformed to electricity, which is stored in an on-board rechargeable battery that is used by an electric motor to drive the vehicle (Adly *et al.*, 2006). Additionally, when an ICE is employed, a PHEV is noted to be in charge-sustaining mode (Silva *et al.*, 2009). The ICE of a parallel PHEV can be discontinued when required (in charge-depleting mode), permitting stop-start driving and battery power to be utilised at lower speeds (Eyre *et al.*, 2002, p. 50). Consequently, this can assist in improving air quality, particularly in urban areas where traffic congestion regularly takes place (Sioshansi *et al.*, 2010; Soret *et al.*, 2014). The ICE can be downsized in both parallel and series configurations PHEV (King, 2007, p. 48; Doucette and McCulloch, 2011), which reduces both CO_2 and pollutant exhaust emissions and considerably enhances fuel efficiency.

The charging of PHEVs from the electricity grid means that emissions savings from the exhaust are transferred to the stage of energy production. This is an advantage to human health, given that power plants are normally located in isolated areas away from heavily populated places (Sioshansi *et al.*, 2010), which undoubtedly reduces the risk of exposure to harmful contaminants.

Emissions from PHEVs rely on the energy generation mix (Sioshansi *et al.*, 2010). If PHEVs are charged from electricity, the generation mix is primarily dependent on fossil fuels, such as in China where an increase occurs in CO_2 emissions compared to CVs (Doucette and McCulloch, 2011). In contrast, Silva (2011) reported that, if PHEVs use electricity generated from renewable sources, such as in Brazil, a decrease of CO_2 emissions takes place. Similarly, if the electricity to charge PHEVs is produced from nuclear power, such as in France, this cuts CO_2 emissions; however, there is an increase in the emissions of other GHGs like water vapour. Additionally, producing electricity using natural gas generators can cut CO_2 emissions (Sioshansi and Miller, 2011). It is also important to mention that energy generation mixes vary between regions and depend on the peak demand time during the day, so that when a PHEV is charged has an influence on emissions (Axsen *et al.*, 2011), and this also applies to other electric vehicles.

3.7 Charging Infrastructure

Electric vehicles can be charged at home, work or in public areas. The charging process requires, cables/connectors and a communication protocol between a vehicle and charger unit. An additional protocol is required between the unit and electricity grid. A charger unit has three main characteristics. The level describes the power output from the charger outlet such as 230 volts; type refers to the socket and connector; and mode describes the communication protocol between the vehicle and charging unit (IEA, 2017b, p. 29). Currently, common types of connectors are CHAdeMO, Type 2, CCS and Tesla. Some organisations are working on the standardisation of charger standards worldwide, in particular in relation to rapid charging, such as CharIN (CharIN, 2015) of which Tesla (a significant electric vehicle manufacturer) is a member. In addition, battery swapping stations have been established in China where more than 800 stations have been deployed to swap electric bus batteries and some other models (Hall and Lutsey, 2017; IEA, 2017b, p. 36).

3.7.1 Charging Infrastructure in the European Union

The current number of recharging units globally, including those which are public and non-public (both domestic and work), was 2.3 million in 2016 (IEA, 2017a). According to the European Commission (2014), EU member countries were required to announce 2030 targets for charging point numbers by 2016. Consequently, countries such as France identified a target of 7 million by 2030 (IEA, 2017b, p. 34).

In 2011, the Green eMotion project was setup. This is €42 million project financed partially by the European Commission (EC) as part of the development of the recharging infrastructure in Europe, was set up (Green eMotion, 2011) carried out in twelve regions in Europe (Rodríguez *et al.*, 2014) with the aim to create common standards for electric vehicle charging across the continent (Hall and Lutsey, 2017).

3.7.2 The UK Government's EV Recharging Infrastructure Plans

In 2011, the Office for Low Emission Vehicles (OLEV, 2011) identified three perspectives for improving charge points infrastructure, as follows:

- Recharging at home: where users can benefit from using smart meters that detect off-peak periods to utilise low tariffs when demand is low, such as overnight charging which is likely to reduce driving costs.

- Recharge at work: where company vehicles can be parked and recharged overnight. This might attract employees who do not have access to home charging or where home charging is insufficient.

- Recharging in public spaces: which can play an important part in advertising campaigns for electric vehicles and may increase awareness of their importance in improving air quality, since most recharging otherwise occurs at home and/or work. Additionally, those users who lack charging units at home will take advantage of public units. Deployment in service areas on motorways that link cities might also be useful for individuals travelling over longer distances. It takes just 30 minutes to recharge 80% of electric vehicle batteries using a rapid charger (Blythe *et al.*, 2012; Wardle *et al.*, 2014).

Currently, according to the IEA (2017b, p. 52), there were approximately 11,000 and 1,500 slow and fast charging points respectively in the UK in 2015 for public use. Meanwhile the EAFO (2017) reported that, up until June 2017, 2400 fast chargers had been deployed in the UK. Additionally, a contribution of 75% of the cost of installing home chargers, including VAT, was capped at £500 (OLEV, 2016).

3.7.3 The UK Plugged-in-Places Programme

In 2010, the UK government committed £30 million to the Plugged-in-Places (PiP) programme to support the development of the national public recharging infrastructure (OLEV, 2011, p. 26). This programme offered funding to businesses, local authorities and groups to subsidise the establishment of EV recharging points. It was rolled out across eight regions in the country, as illustrated in Figure 3-3.

Source: OLEV (2011, p. 26)

Figure 3-2: Regions utilising the Plugged-in-Places programme

The proposal was for 8,500 recharging units to be deployed countrywide in workplaces, car parks and in homes. The PiP programme got underway in 2010, primarily in the North East of England, London and Milton Keynes (OLEV, 2011, p. 26). By March 2013, approximately 4000 recharging units had been installed by means of the PiP programme. Of this number, 65% were publicly accessible. Furthermore, it was also stated that an additional 5000 recharging units had been established by private organisations (OLEV, 2013b). The EAFO (2017) reported that, until June 2017, 13,524 charging units had been deployed of which 2,407 are rapid charging units in the UK.

3.7.4 Plugged-in-Places (PiP) in the North East

The North East of England is the second smallest region in the UK, where more than two million inhabitants live in urban areas which represent 27% of the 8,600 km^2 of the total area of the region. The North East hosts the Nissan vehicle plant which has manufactured the Nissan LEAF and lithium-ion batteries since 2013. This makes the region a major hub for developments in low carbon vehicles (Herron and Wardle, 2015).

The PiP scheme began in April 2010 in the North East of England (NE PiP). Here, funding was offered to homeowners who were able to access off-street parking to have standard recharging units positioned. In this case, 100% of the installation cost was covered by this specific grant. Additionally, it should be noted that funding was offered for recharging units to be installed in public or workplace settings, on condition that the host of this recharging infrastructure provided parking and electricity for NE PiP members. The programme employed for the NE PiP scheme enabled EV drivers to pay an annual membership fee of £100 or a monthly fee of £10. The PiP scheme also offered users unrestricted access to all public recharging units and several workplace units by means of active maps. Furthermore, users from outside the North East of England could use ‘pay as you go’ facilities available via their mobile phones to make use of the recharging units scattered across the region (Herron and Wardle, 2015). More than 700 public recharging units had been established in the NE PiP network (Herron and Wardle, 2015; Zap-Map, 2017).

3.8 Implications of EVs

Increasing EV penetration will positively influence air quality, particularly in urban areas where road traffic contributes 22% of emissions of CO_2 (Mahmoudzadeh Andwari *et al.*, 2017) and other harmful contaminants in the UK. The energy suppliers and their distribution grids will suffer from the extra demand for electricity; however, smart cards have the ability to utilise off-peak periods when the tariff is comparatively low.

3.8.1 Energy

Meeting electricity demand in any period and delivering electricity through the distribution network are the principal roles of the National Grid, which is also responsible for the maintenance of plant and distribution networks in the UK. As electric vehicle numbers increase, the power needed to recharge them will be placed at the top of current demand for electricity, and this undoubtedly will have direct influence on current power production and distribution grids (Papadopoulos *et al.*, 2011). The energy required to recharge electric vehicles depends on the number of EVs, duration and occurrence time, and the efficiency of the batteries adopted in EVs (Papadopoulos *et al.*, 2011). Concerns have been raised that the capacity of the existing power and distribution networks may not be able to cope with the additional electricity demand in charging electric vehicles (Oliveira *et al.*, 2013), particularly if EVs are recharged during peak periods (Oliveira *et al.*, 2013). However, Papadopoulos *et al.* (2011) have reported that the estimated expansion in electricity production by 2030 in the UK is capable of managing the increase in demand for domestic charging for an EV penetration level of only 7% within the car fleet, if tariff appropriate regimes are established.

Consequently, it was ascertained that, with a market penetration for EVs of 20% in the UK, the power distribution system would need to be increased in scale to support demand (Huang and Infield, 2010). Pudjianto *et al.* (2013) reported that a 10% market penetration for EVs would require the investment of roughly £36 billion to increase grid capacity should EVs be recharged during peak hours.

By 2030, more than 30% of electricity in the UK might be produced from renewable energy sources (Grau *et al.*, 2009; Papadopoulos *et al.*, 2011). As electricity generation by means of wind increases overnight, this source can play a significant role in recharging EV, with electricity at night (Grau *et al.*, 2009). Likewise, Grau *et al.* (2009) highlighted the opportunity to store electricity through a massive integration of the grid with EV batteries.

It is also recognised that there are concerns regarding future demand for electricity due to the uncertainty associated with future energy sources, the commitment to sources of new electricity production and the optimisation of existing sources.

3.8.2 Carbon Dioxide (CO_2)

Electricity is produced from different sources such as renewables (wind, hydro), nuclear, coal and petroleum. Carbon content is different in relation to each fuel type. For example, carbon is not released when using wind as an energy source. Additionally, compared to coal and oil, only a small amount of carbon is released with the use of natural gas.

3.8.3 Air Quality

Human health is threated by poor air quality in urban areas where exposure to pollution is unavoidable. In the UK, national objectives related to pollutant concentration thresholds have been legislated. Nevertheless, local authorities have declared approximately 600 Air Quality Management Areas AQMAs within their jurisdictions (Hitchcock *et al.*, 2014). Virtually all of the declarations concerning AQMAs are due to excessive emissions attributed to road traffic (Chatterton, 2011).

Electric vehicles can play a significant role in reducing air pollutants and consequently improving air quality. Li *et al.* (2016a) have pointed out that electrifying all motorcycles and cars of 2010 fleet in Taiwan will remarkably influence air quality. They developed a scenario where electricity is generated from thermal plants (mainly coal-fired), as it was estimated that road transport emissions would be reduced. A significant decrease would occur for CO of 85%, VOCs of 79%, NO_x of 27% and $PM_{2.5}$ of 27%. It is worth mentioning that emissions of NO_x attributed to road transport are estimated to decrease by 33.9 Gg/year, although at power plants they will increase by 20.3 Gg/year. Similarly, $PM_{2.5}$ emissions from transport are estimated to decrease by 7.2 Gg/year; nevertheless, at the plant site they will increase by 0.8 Gg/year. In addition, the study estimated reductions in harmful pollutant concentrations in urban and rural areas. In urban areas, NO_x, VOC, CO, $PM_{2.5}$ and O_3 and SO_2 would be reduced by 18%, 21%, 65%, 6%, and 3% respectively.

Moreover, according to DeLuchi *et al.* (1989), the complete replacement of conventional cars with EVs would result in reductions in road transport emissions, the virtual elimination of CO and HC emissions, and considerable NO_x reductions. Conversely, it should be noted that SO_2 emissions may escalate if the energy generation is from the coal-fired power stations which were then the principal source of electricity in the UK.

3.9 Summary

This chapter has outlined targets for the reduction of average CO_2 emissions caused by new cars. It outlined the target of 130 g/km attained in 2016, and future challenging targets of 95 g/km by 2021 and ultra-low CO_2 emissions (i.e. less than 50 g/km) for all new cars by 2040. Differences between petrol and diesel engines have been highlighted, followed by an overview of harmful emissions-abating technologies. Alternative fuel vehicles were presented in Section 3.6, focusing on electric vehicles and their effect on energy, CO_2 emissions and air quality. It has been shown that the demand for electricity required for charging electric vehicles will add to current electricity consumption, causing extra emissions to be released from power plants. Nonetheless, these extra emissions will be shifted from the substantial numbers of vehicles in urban areas, where exposure to humans is the greatest, to power plants where exposure to humans is much lower. As a result of the role of electric vehicles in reducing harmful emissions and the subsequent improvements in air quality, scenarios for 2030 vehicle fleet are developed in chapter 6 with a range of proportions of electric vehicle in order to quantify the impact of increasing electric vehicles on emissions, air quality and disease burden in chapter 7.

CHAPTER 4

4. Methodology

4.1 Introduction

The aim of this research is to investigate the impact of 2014 Baseline and 2030 future traffic scenarios on vehicular emissions and air quality. This is estimated in terms of the effect of changes in concentrations of $PM_{2.5}$, PM_{10} and NO_2 on the disease burden for residents in Newcastle and Gateshead. Hence, traffic networks and related flows in Newcastle and Gateshead are chosen to undertake the research.

The standard pollutants monitored in the UK include NO_2, $PM_{2.5}$, PM_{10}, SO_2, Benzene (C_6H_6), 1,3-Butadiene (C_4H_6), CO, Lead (Pb) and VOC. In addition to monitoring, local authorities employ the Emission Factors Toolkit (EFT) published by DEFRA to model all significant point area, and line (road) sources. Devolved Administrations are responsible for continual review and assessment of local air quality. The EFT is used to calculate road vehicle pollutant emission rates for pollutants such as NO_x, PM_{10}, $PM_{2.5}$ and CO_2 (DEFRA, 2016c; DEFRA, 2017b; DEFRA, 2019c) using a combination of monitoring and modelling local authorities including Newcastle and Gateshead as part of their duties under the Environment Act 1995, have showed that pollutant concentrations of either NO_2, $PM_{2.5}$ or PM_{10} exceeded the limit values and declared AQMAs (Air Quality Management Areas). The major sources associated with transport along with their source categories are presented in Table 4-1.

Table 4-1: Emission sources and relevant pollutants to be considered during Assessment by local authorities as part of LAQM (DEFRA, 2016d)

	Road Transport Emission Sources	Relevant Pollutants
1	Narrow congested streets with residential properties close to the kerb	NO_2
2	Busy streets where people may spend 1 hour or closer to traffic	NO_2
3	Roads with a high flow of buses and/or HGVs	NO_2, $PM_{2.5}$, PM_{10}
4	Junctions	NO_2, $PM_{2.5}$, PM_{10}
5	New roads constructed since the last round of Review and Assessment	NO_2, $PM_{2.5}$, PM_{10}
6	Roads with significantly changed traffic flows	NO_2, $PM_{2.5}$, PM_{10}
7	Bus and coach stations	NO_2

This clearly shows that road transport is the biggest source of urban air pollution exceedance in the UK. In fact 90% of AQMAs across the UK have been declared because of NO_2 released from road transport related pollution (Chatterton, 2011; Bell *et al.*, 2013). More locally, the main sources of $PM_{2.5}$ in the city of Newcastle are road traffic emissions (comprising engine exhaust, road and tyre/brake abrasion) (Newcastle City Council, 2019, p. 17). Therefore, given that transport is the major source of pollution across the UK, the research reported in this thesis chose to study $PM_{2.5}$, PM_{10} and NO_2.

According to the Royal College of Physicians, exposure to particulates causes roughly 29,000 deaths in the UK per annum. This number may rise to around 40,000 deaths when considering exposure to NO_2 (Royal College of Physicians, 2016). In addition, the UK faced legal challenges from the EU in 2010 due to breaches of Directive 2008/50/EC for PM_{10} and NO_2 (Chatterton, 2011; Bell *et al.*, 2013). In its strategy to tackle roadside nitrogen dioxide concentrations, the UK government is already committed to investing over £2.7 billion overall in air quality and cleaner transport (DEFRA and DfT, 2017a).

In this chapter, the methodology used for the research is explained and several of the methods applied are described in detail in subsequent chapters. The methodology consists of several steps, such as the modelling of traffic flows, emissions rates and emissions dispersion, and the final step, to quantify the diseases burden resulting from a change in pollution concentrations between traffic scenarios and the Baseline scenario.

The first step was to build a transport model for the 2014 Baseline scenario (objective one), which was derived from the Transport Planning Model (TPM) in relation to network and traffic flows for cars, LGVs, HGVs and buses in 2010. This was completed by means of utilising DEFRA guidelines that describe procedures for projecting traffic flow for a future year based on a traffic flow for a given year (DEFRA, 2010). Subsequently, establishing the Business as Usual (BAU) scenario for 2030 (objective two) and six other future scenarios (objective three).

The second step was to calculate the rates and dispersion of emissions released in each scenario (objective four). For each scenario, emissions rates were calculated by means of the Platform for Integrated Transport and Health Emissions Modelling (PITHEM) (Namdeo and Goodman, 2012), which can read GIS files that include data on the volume and speed of traffic for each vehicle class, and moreover can read road network geometry. The PITHEM emissions model has the ability to convert the calculated emission rates and road network data, including street canyons, into comma-separated value (CSV) files, which can be extracted by ADMS-Urban software to estimate emissions dispersions and hence estimate emissions concentrations at selected receptors.

The final step was to utilise the dose-response coefficients developed by the Department of Health and other recommended affiliates, which describe the association of the probability of premature death and respiratory hospital admissions with exposure to incremental doses of $PM_{2.5}$, PM_{10} and NO_2. These coefficients were applied to changes in concentrations between the Baseline and 2030 scenarios in order to quantify the predictions of disease burden and health gain (objective five). Figure 4-1 demonstrates the general steps of the methodology and links with the proposed research objectives.

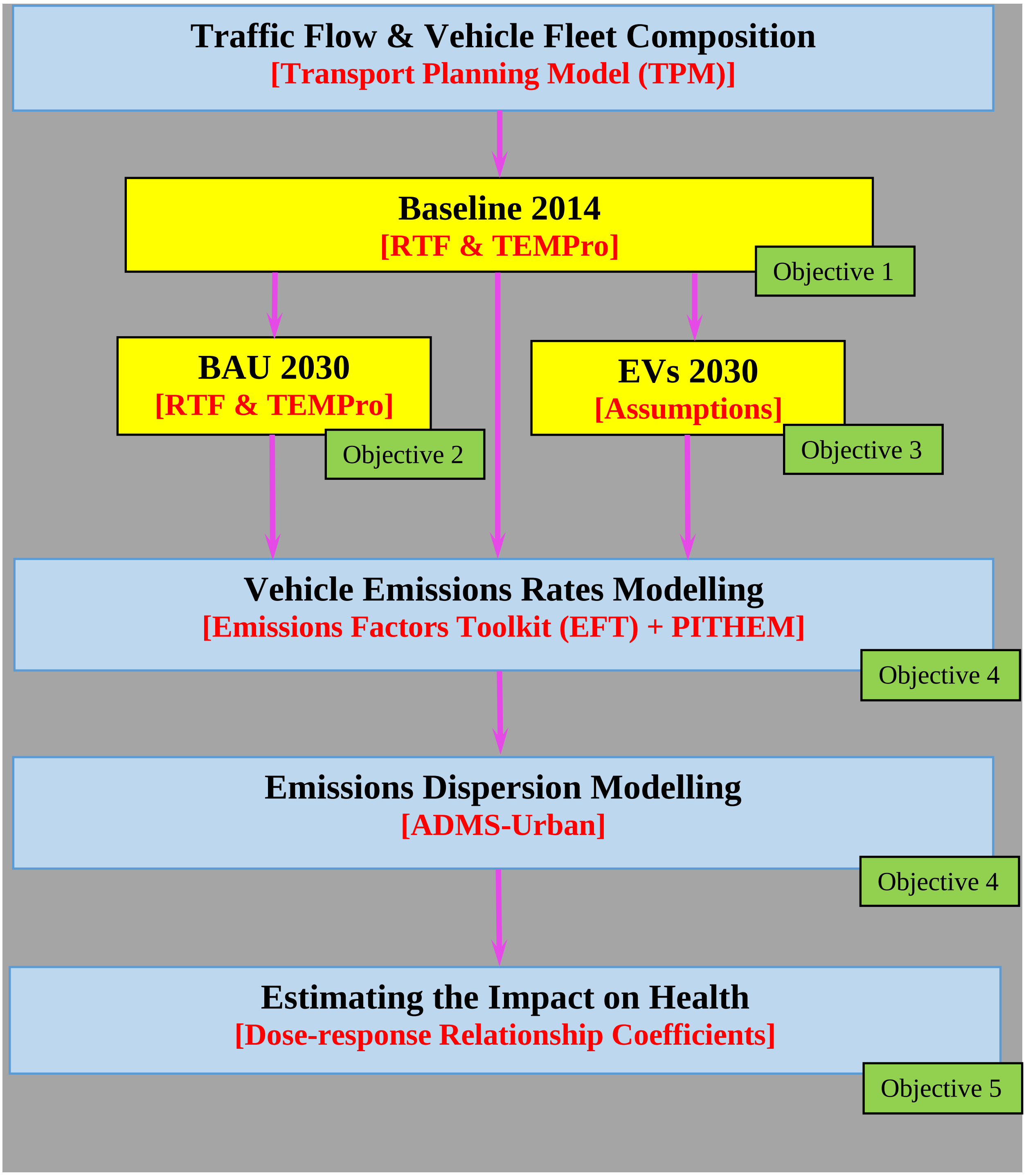
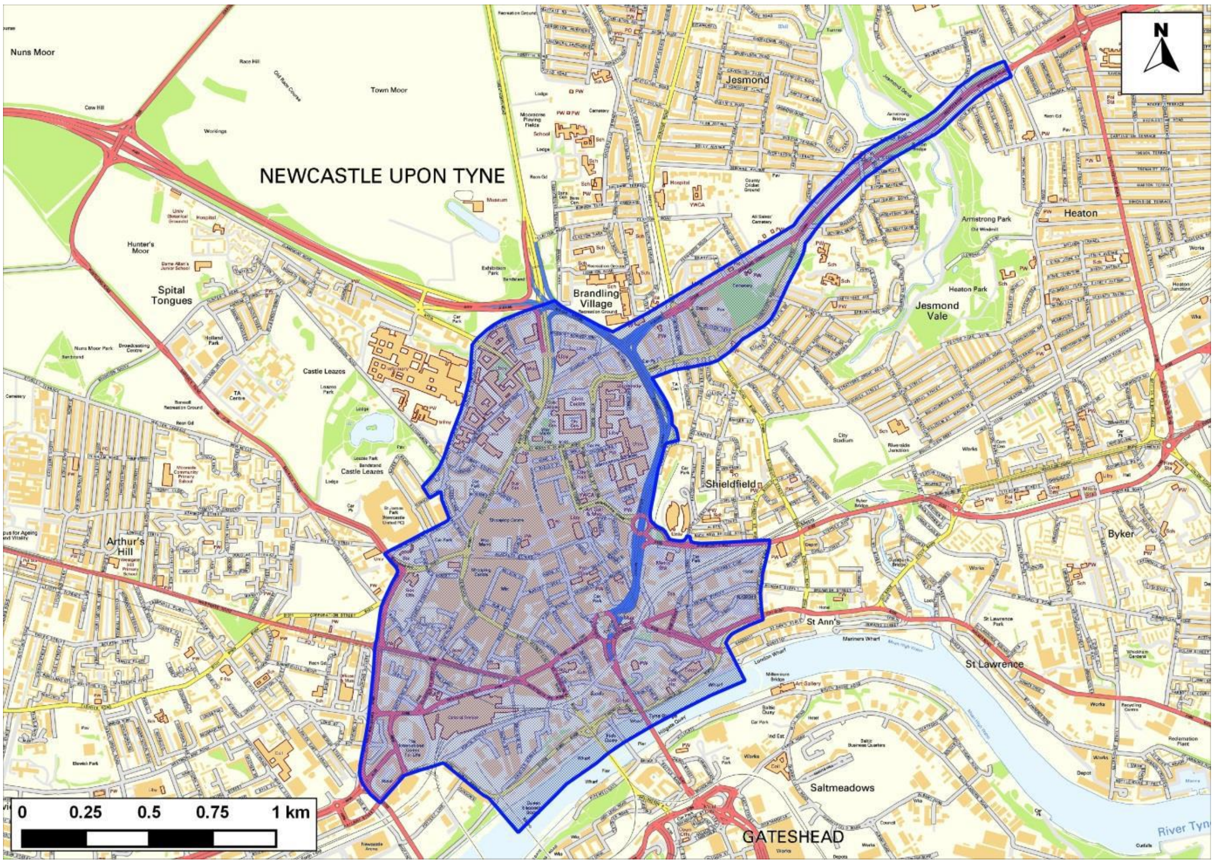


Figure 4-1: Method flow diagram

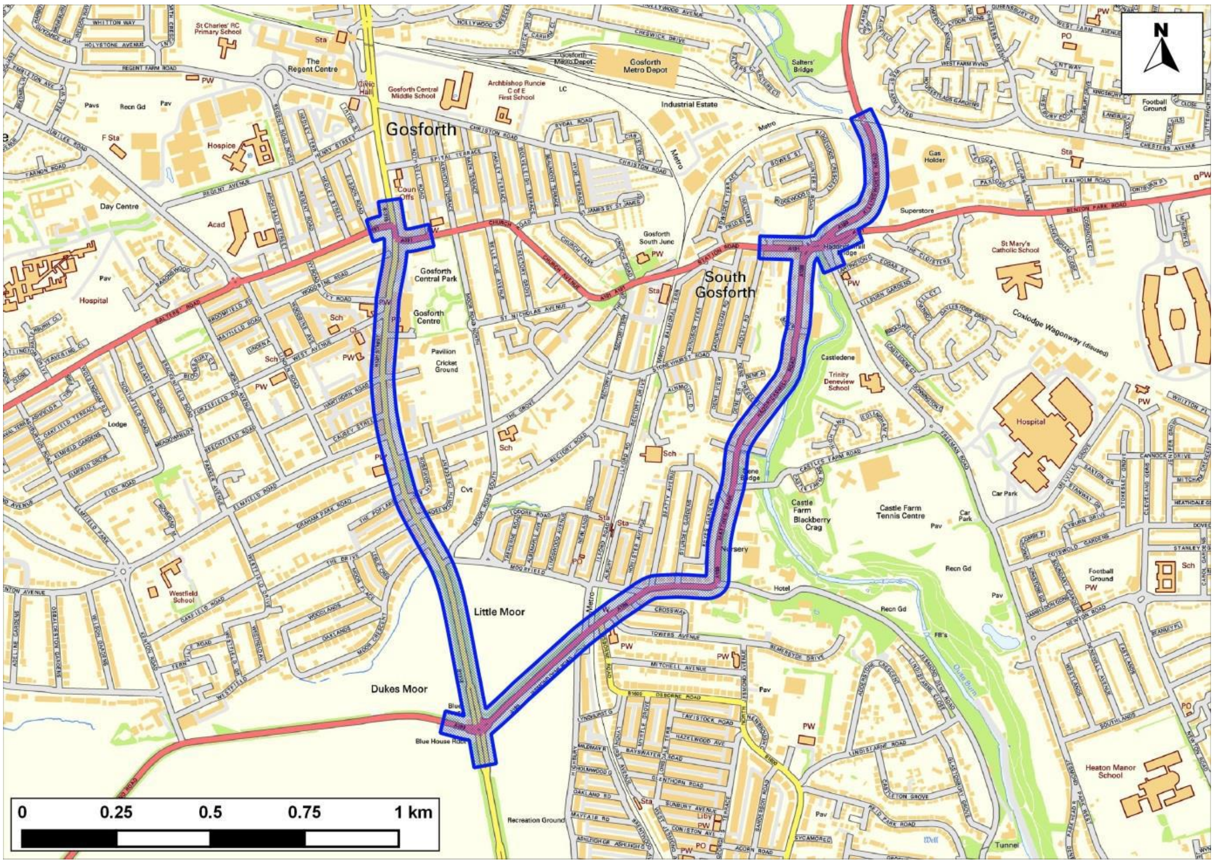
4.2 Selection of Study Area

Newcastle upon Tyne and Gateshead were selected as the study area to test the suitability of the methodology and its applications in demonstrating the air quality and health benefits of the introduction of low carbon vehicles, including EVs. Both are areas declared as AQMAs, where the annual mean concentrations of NO_2 have not met national air quality objectives. In Newcastle, there are two AQMAs; covering the city centre as shown in Figure 4-2 and

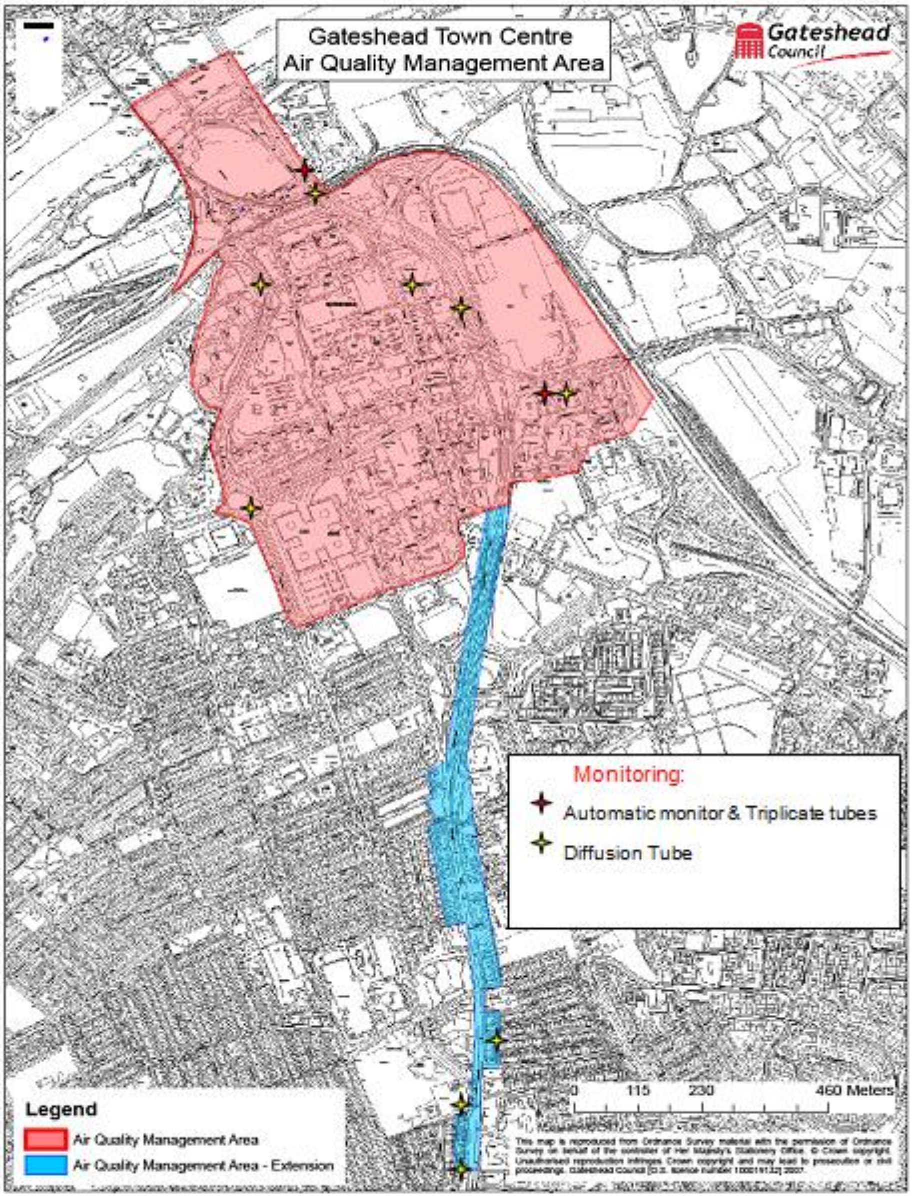
Gosforth High Street 4-3, whilst the town centre is the AQMA in Gateshead as Figure 4-4 displays. The Birtley site was declared AQMA in 2008 because the annual mean concentration of NO_2 reached 43 µg/m^3. This AQMA was revoked in 2012 as levels fell below the NO_2 annual mean objective level of 40 µg/m^3 for a sustained period of time (Gateshead Council, 2018, p. 4).



Source: Newcastle City Council (2015, p. 8)
Figure 4-2: Map of Newcastle upon Tyne City Centre AQMA



Source: Newcastle City Council (2015, p. 9)
Figure 4-3: Map of Gosforth AQMA



Source: Gateshead Council (2016, p. 39)
Figure 4-4: Map of Gateshead Town Centre AQMA Boundaries and Monitoring Locations

The work carried out in this study focused on road transport in Newcastle upon Tyne and Gateshead, which were chosen as a case study for both academic and technical reasons. The former are that Newcastle and Gateshead are medium-sized UK urban conurbations whose characteristics are typical of other such sites in the UK, for example in featuring dense urban infrastructure, high traffic flows and heavy congestion in city/town centres. Moreover, various different public travel modes operate in the area including the Metro, buses and planes. More specifically as with many other cities bus flows are high in city centre streets being the main destinations of services from the outer suburbs, satellite towns and villages. In addition, the UK mandates standard procedures for monitoring and modelling which ensure that air quality is administrated and evaluated consistently across all other local authorities where Air Quality

Management Areas (AQMAs) were declared. Thus, using both these cities as a case study enables the outcomes from this research to be applicable throughout the UK. Furthermore, Newcastle and Gateshead are major cities located each side of a rivers on the North East coast close to the border with Scotland. With more similar weather patterns may possibly increase the transferability to air quality management in Scotland.

The technical reasons for choosing Newcastle upon Tyne and Gateshead as a case study for this research are as follows:

1) The existence of comprehensive historic traffic data, given that the Traffic and Accident Data Unit (TADU), which is based in Gateshead, records traffic accident, cycle flow and traffic data for all Tyne and Wear councils;

2) Good air quality monitoring network in place;

3) Historic traffic, air quality and meteorological data are available; and

4) Traffic and air quality models for both boroughs are available for use in this research.

A general description of Newcastle and Gateshead is now given. Newcastle upon Tyne is a city on the north bank of the River Tyne. Together with its twin city Gateshead on the south of the Tyne is the largest conurbation in the North East England. The populations of Newcastle and Gateshead are 280,000 and 200,000 respectively, and they are located approximately 270 miles north of London by rail.

According to the RAC Foundation (2012), the 2011 census revealed that approximately 60% of households in Newcastle and Gateshead owned a car or van. Table 4-2 shows that in 2016, roughly 54~67% of people travelled to work via this mode of transport, 18~23% of commuters by bus, whilst 8~12% walked (Labour Force Survey, 2017). Road transport was the dominant pollution source and has had a substantial impact on the air quality emissions in Newcastle and Gateshead.

Additionally, the Tyne and Wear Metro serves 60 stations along 78 km of track and carried 41 million passengers in 2009–2010 (TWITA, 2011, p. 33). The Metro system underpins transport provision in the region serving 25% of households (TWITA, 2011). Moreover, it should be noted that the trunk road network around Tyne and Wear is typically congested (TWITA, 2011).

Table 4-2: Travel modes adopted by inhabitants of Newcastle and Gateshead in 2016

Travel Mode	Newcastle (%)	Gateshead (%)	Great Britain (%)
Car, van, minibus, works van	54	67	70
Bicycle	4	1	3
Bus, coach, private bus	23	18	7
Train	5	2	6
Underground train, light railway, tram	2	4	4
Walk	12	8	10

Source: Labour Force Survey (2017)

As part of the assessment process, Newcastle City Council undertake detailed air quality monitoring. It established that, in 2018, the annual EU air quality limit value for NO_2 (40 $\mu g/m^3$) was not met in the two declared AQMAs (Newcastle City Council, 2019, p. 31), with the primary source being from road vehicle exhaust. In addition, the main source of airborne particulate matter in the city of Newcastle is road traffic emissions, including engine exhaust as well as road and tyre/brake abrasion (Newcastle City Council, 2019, p. 17).

The two local authorities have the potential to increase EV use and the related infrastructure due to the location in the North East of England, which hosts the manufacturer of the Leaf EV and lithium-ion batteries at Nissan's Sunderland plant. This places the region at the forefront of research and development, manufacturing and training facilities in relation to vehicles and batteries throughout the EV source chain (Wardle *et al.*, 2014). Additionally, the Office for Low Emission Vehicles funded the Plugged-in-Places project in the North East to the tune of £7.8 million between 2010 and 2013, aiming to create a comprehensive connected network of EV charging points across the region (Blythe *et al.*, 2012). All public access charge points are run by a single network operator, Charge Your Car, which is the scheme brand for the NE PIP project. Furthermore, Newcastle University was commissioned by the Technology Strategy Board to undertake a programme known as the North East's Switch to Electric Vehicles (Switch-EV) trial. This project ran from November 2010 until May 2013 (Wardle *et al.*, 2014; Neaimeh *et al.*, 2015). The programme's primary objective was to understand the delays regarding the mass adoption of ultra-low carbon vehicles such as electric cars. In this programme, trips, recharging data and locations were recorded using an in-vehicle logger and on-board GPS device for 44 new EVs driven on the region's roads. The data was transferred and stored in computer servers hosted by Newcastle University for the purposes of analysis (Blythe *et al.*, 2012; Wardle *et al.*, 2014).

Furthermore, traffic flow data on a local network of 2,887 roads for cars, LGVs and HGVs links and 10,608 roads for bus links in Newcastle and Gateshead are available for this research, as Newcastle University has undertaken a project to test the feasibility of low-emission zones in both boroughs (Goodman *et al.*, 2014). Traffic flow data per hour for cars, LGVs, HGVs and buses corresponding to each link for the year 2010 were available. Hence, the necessary elements to conduct this research were in hand.

4.3 Source of Traffic Data

This thesis relies primarily on the availability of traffic data related to the study's area network in order to model emissions rates and concentrations (air quality) and the impact of changes in air quality on disease burden.

4.3.1 Tyne and Wear Transport Planning Model (TPM) 2005 and 2010

The initial version of the Transport Planning Model (TPM) was a large scale, strategic, multi-modal transport model with an O-D matrix and network data for 2005. It covers the five metropolitan boroughs of Newcastle, Gateshead, Sunderland, North Tyneside and South Tyneside in the county of Tyne and Wear. The network traffic flows of 2005 along with its calibration and validation have been reported elsewhere (Jacobs Consultancy, 2010).

The 2005 TPM was updated to 2010 to cover Newcastle and Gateshead only and provided the supporting information for the practicability and potential of implementing Low Emissions Zones (LEZs) to address identified issues of air quality within the Newcastle City and Gateshead Metropolitan Borough areas. Newcastle University was commissioned to produce the following elements towards delivering the overall goal of the feasibility study of low emission zone:

a) A traffic emission inventory by vehicle type and fleet age for nitrogen dioxide, particulate matter and carbon dioxide for the Newcastle and Gateshead Air Quality Management Areas (AQMAs);

b) An assessment of existing air quality across the Newcastle and Gateshead AQMAs for nitrogen dioxide and particulate matter;

c) A source apportionment analysis of emissions within the AQMA areas;

d) The remodelling of air-quality to show the effectiveness of the proposed LEZ for a future year (i.e. 2021).

The 2005 TPM was modified by updating general traffic flows to reflect the 2010 profile by means of the following measures:

a) The geometric network for the revised layout and zones of the 2010 model was assumed to be the same as for the network for 2005.

b) The general traffic flow of cars and freight transport within the TPM were updated using automatic traffic monitoring data received from TADU across the Tyne and Wear, Northumberland and Durham regions. Flow data for both the TPM original year of 2005 and 2010 were received;

c) A separate model for bus transport was produced based on public transport information provided by Newcastle City Council;

d) The geometries of both the 2005 and 2010 TPMs were linked to the Ordnance Survey's MasterMap Integrated Transport Network map layer; and

e) Hourly speed values from the council-held TrafficMaster link-speed dataset were assigned to the 2010 TPM.

In addition, the updating of the 2005 TPM to 2010 primarily involved changes to overall traffic flow levels. Further changes were made to reflect the general trend of reduced numbers of heavy goods vehicles in the centre of Newcastle over the period 2005-2010 also were made. Traffic count information (primarily inductive loop information) was processed for use in calibrating and validating the 2010 TPM.

The validation of the 2010 TPM was carried out by comparing real and modelled traffic flows. The average GEH was found to be <6 for over 85% for all links (Goodman *et al.*, 2014, p. 45), although the Web-based Transport Analysis Guidance (WebTAG) states that 85% of the links should have a GEH value of 5 or below.

The TPM model was updated to the 2014 Baseline following DEFRA (2010) guidelines. Subsequently, it was validated against observed traffic flows by means of figures acquired from the Tyne and Wear Accident Data Unit (TADU), which is a body unit comprising councils in Tyne and Wear formed to support the delivery of transport solutions and economic development across Newcastle upon Tyne, Gateshead, Sunderland, North Tyneside and South Tyneside (TADU, 2014).

4.4 Source of Pollution Concentration Data

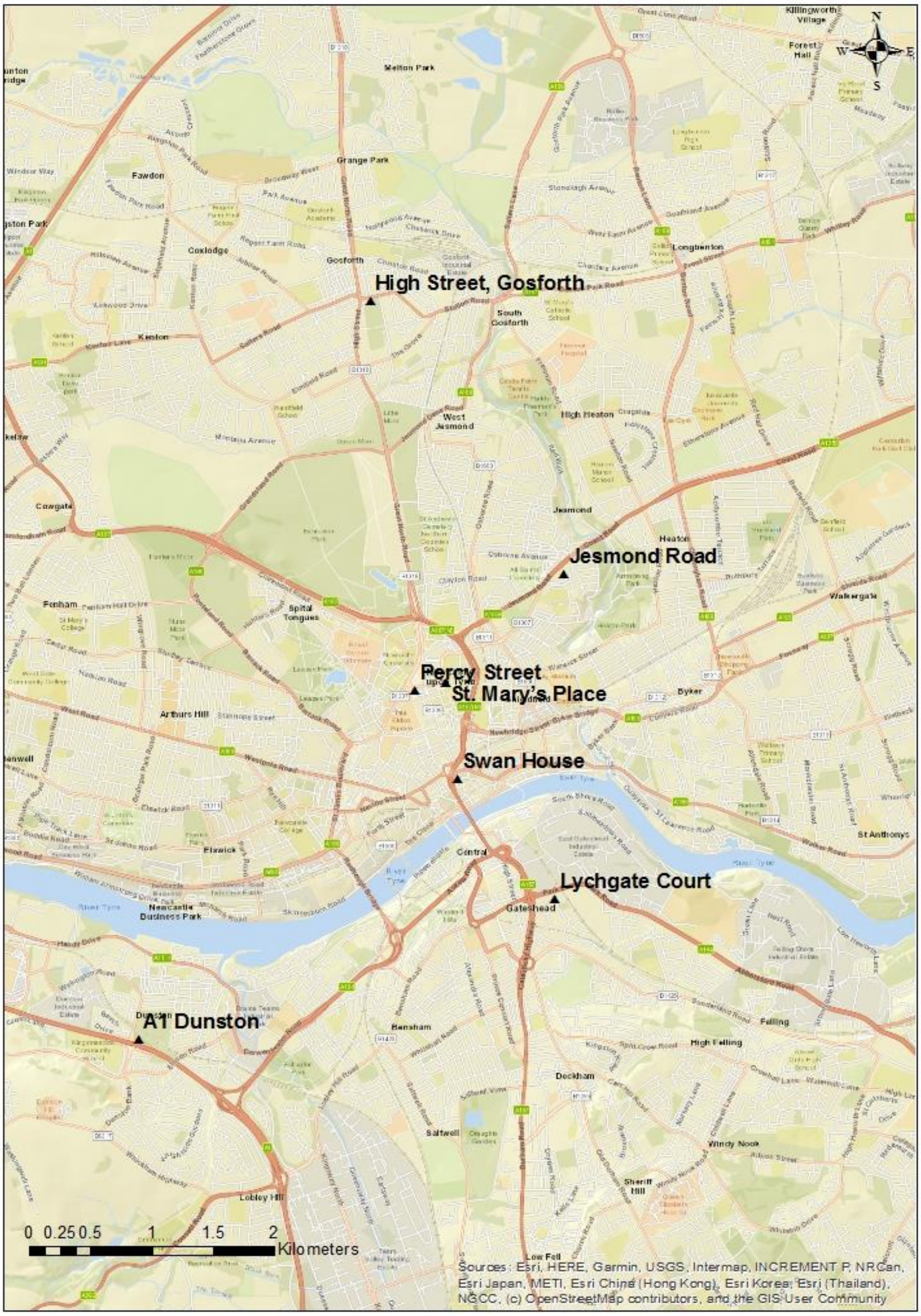
Monitored pollutant concentrations are required to determine the effectiveness of the Baseline model in estimating $PM_{2.5}$, PM_{10} and NO_2 concentrations. Newcastle and Gateshead authorities are obliged to fulfil the requirements of the Local Air Quality Management process as set out in Part IV of the Environment Act (1995), the Air Quality Strategy for England, Scotland, Wales and Northern Ireland 2007, and the relevant Policy and Technical Guidance documents. Both councils are required to publish an Annual Status Report showing pollution results obtained from the monitoring tools used by the councils and identifying potential sources of air pollution (Gateshead Council, 2016; Newcastle City Council, 2016). Newcastle City and Gateshead Councils conducted automatic monitoring at seven sites during 2014. Table 4-3 characterises the monitoring sites and Figure 4-5 shows a map of the monitoring site in the study area.

Table 4-3: Details of Automatic Monitoring Sites

Site Name	Site Type	X coordinate	Y coordinate	Pollutants Monitored
St. Mary's Place (AURN[1])	Urban Centre	425029	564916	NO, NO_x, NO_2, PM_{10}, $PM_{2.5}$, O_3
Jesmond Road, Cradlewell	Roadside	425992	565831	NO_2, PM_{10}, O_3
Percy Street	Roadside	424776	564861	NO_2
Swan House, Pilgrim Street	Roadside	425124	564112	NO_2
High Street, Gosforth	Roadside	424411	568115	NO_2, PM_{10}
Lychgate Court	Roadside	425912	563108	NO_2, $PM_{2.5}$
A1 Dunston	Roadside	422510	561928	NO_2, $PM_{2.5}$

[1]AURN: Automatic Urban Rural Network

Pollutant concentrations in all proposed scenarios were estimated based on released vehicular emissions only. Thus, to complete the pollution concentrations profile, information on background pollutant levels from non-transport sources were taken directly from the DEFRA source-apportioned background maps (DEFRA, 2016b). Maps of 1km grid squares for Gateshead and Newcastle for NO_x, NO_2, PM_{10} and $PM_{2.5}$ for 2014 and 2030 were downloaded from the DEFRA web site. Only contributions from non-road sources and minor roads in pollutant concentrations were used in the calculations.



Source: Newcastle City Council (2015, p. 11) and Gateshead Council (2016, p. 39)

Figure 4-5: Map of monitoring sites in Newcastle and Gateshead

These monitoring sites were assigned as receptors in the Baseline model, in order to compare modelled concentrations against recorded concentrations at the monitoring sites for validation purposes.

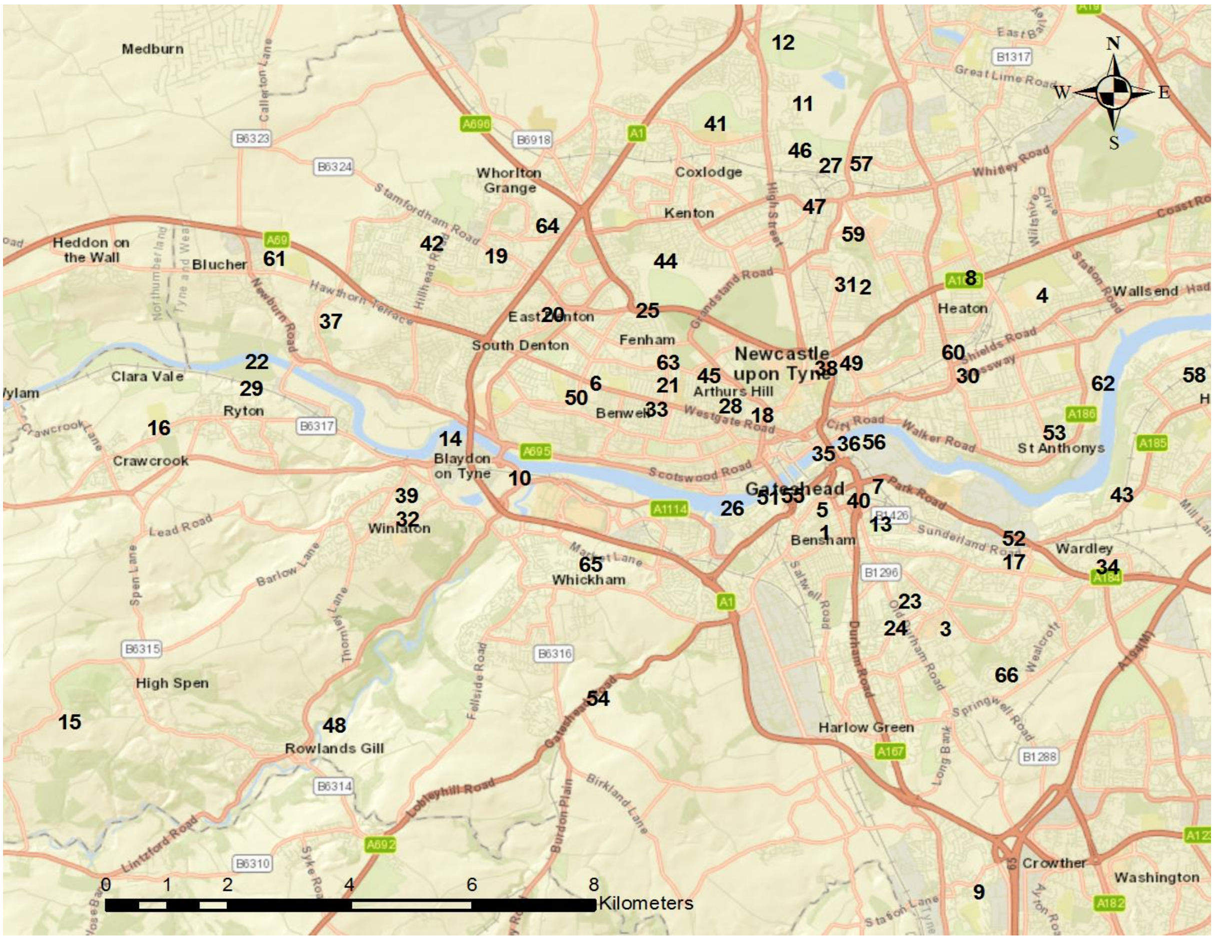
4.5 Source of Health Statistics

Health status, in terms of figures concerning hospital admissions and mortality due to respiratory disease burden for residents living in Newcastle and Gateshead, need to be linked with levels of pollution at the 2014 Baseline, so as to investigate the impact of changes in pollution concentrations on the rates of hospitalisations and premature deaths. Those figures related to hospitals throughout Newcastle and Gateshead are available from NHS Digital, which maintains records of all hospital episodes.

Therefore, an application form was submitted to NHS Digital, previously known as the 'Health and Social Care Information Centre', to purchase figures describing numbers of deaths and numbers of admissions to hospital that are attributed to respiratory illnesses and are specifically under the categorisation of J00 to J99 (e.g. codes from J00 to J06 which indicate acute upper respiratory infections), according to the International Classification of Diseases (ICD-10). The request was accepted by NHS Digital, and data was provided pertaining to patient registration at each of the 66 General Practitioner (GP) practices across the Newcastle and Gateshead area, as shown in Figure 4-6. The data were received in an aggregated format, with the patients' details, such as age range, gender and home address suppressed. Those specific data purchased for a cost of £1,800 formed a significant part of the budget dedicated to this research. Furthermore, NHS Digital requires that their data be stored and processed in a secure network to maintain confidentiality in line with data protection laws. Thus, an Information Governance Toolkit (IGT) was installed in the researcher's desktop by the Information Security Team at Newcastle University. In addition, obtaining the IGT required training modules to be undertaken prior to setting it up. The training included the following modules:

- Introduction to information governance.
- Access to information and information sharing in the NHS.
- The importance of good clinical record keeping.

Prior to requesting the above-mentioned aggregated health figures, a significant amount of time was taken with NHS Digital to acquire more detailed information, such as date and time of event occurring and each patient's residential information at postal or sectoral code level, so as to allow time series analysis to be conducted and for the results to be related to the patients' residences. Thus, a preliminary application was submitted to NHS Digital. This was accepted and categorised as Tier 2. However, the nature of the data is personal and identifiable, and it requires a legal basis for its acquisition (i.e. Section 251, 'Obtaining patient data without their consent'). Hence, a completed application of 28 pages comprising 10,356 words was submitted to the Confidentiality Advisory Group (CAG) via the Integrated Research Application System to apply for the Section 251 to be submitted as a legal basis for the intention to purchase sensitive patient details from NHS Digital. Unfortunately, a medical purpose must be clearly declared and attached to support the CAG application, regardless of its strength. As a result of legal issues, the only way to obtain health data is to suppress identifiable information. Therefore, NHS Digital was asked, as mentioned earlier, to provide numbers of patients distributed geographically by GP practice registrations, assuming that patients have a tendency to register with a GP practice close to where they live. This tendency has been recently confirmed (Santos *et al.*, 2017; Beghelli, 2018). The constraints set by NHS Digital prevented high resolution analysis from being performed by spatially linking residential and pollution hotspots, or time series analysis.



Source: Own Work (ArcMap 10.5)

Figure 4-6: Sites of 66 GP practices across Newcastle and Gateshead

The purchased data reported 9,693 hospital admissions distributed among 8,058 emergency and 1,635 elective admissions, and 702 in-hospital deceased recorded in hospitals across Newcastle and Gateshead. No in-hospital deaths were recorded for people who registered in 19 practices because no deaths occurred. These hospital admissions and mortality cases which were recorded by GPs in the pilot study during 2014 are demonstrated in Table 4-4. Therefore, the GP sites were used as receptors to determine corresponding pollution concentrations at these sites for all scenarios.

4.5.1 Population Registration with GP Practices in the Study Area

The population of patients registered at GP practices is broken down by gender and age range and updated frequently by means of the NHS Digital website. Patient registrations as of January 2015 recorded for 66 GP practices across Newcastle and Gateshead have been extracted from a spreadsheet available on the web and are included in Table 4-4. A total population of 502,838 people were registered with 66 GP practices in Newcastle and Gateshead (NHS Digital, 2015). The Scotswood GP Practice (GP_50) was associated with the smallest number of patient registrations, at 1,340. In contrast, the Saville Medical Group (GP_49) in Newcastle accounted for the largest portion of patient registrations, virtually double the number of people registered with Whickham Cottage Medical Centre (GP_65). Given that GP_49 is located in the city centre and has a branch that serves different areas in Newbiggin Hall (West Newcastle Centre), where several of the main GPs have branches, the number of patients at the branches is added to its related main GP.

Table 4-4: Mortality and hospital admissions GP registration (IMD: Index of Multiple Deprivation Decile)

General Practitioner Details		Number of Patients			End Point (count)		IMD
No.	Practice	Male	Female	All	Hospital admission	Mortality	
GP_01	108 Rawling Road Practice	877	776	1,653	21	0	3
GP_02	Avenue Medical Practice	1,543	1,495	3,038	24	0	9
GP_03	Beacon View Medical Centre	2,318	2,230	4,548	106	10	4
GP_04	Benfield Park Medical Group	4,291	4,207	8,498	193	0	6
GP_05	Bensham Family Practice	2,410	2,133	4,543	84	9	1
GP_06	Betts Avenue Medical Group	5,146	5,105	10,251	246	14	1
GP_07	Bewick Road Surgery	2,939	3,089	6,028	92	6	3
GP_08	Biddlestone Health Group	4,960	4,639	9,599	128	15	7
GP_09	Birtley Medical Group	7,799	8,103	15,902	356	34	4
GP_10	Blaydon GP Led Practice	649	699	1,348	14	0	6
GP_11	Broadway Medical Centre	1,374	1,269	2,643	29	0	6

General Practitioner Details		Number of Patients			End Point (count)		IMD
No.	Practice	Male	Female	All	Hospital admission	Mortality	
GP_12	Brunton Park	2,144	2,326	4,470	47	0	10
GP_13	Central Gateshead Medical Group	5,269	4,989	10,258	218	10	3
GP_14	Chainbridge Medical Partnership	5,458	5,793	11,251	254	18	6
GP_15	Chopwell Primary Healthcare Centre	1,384	1,302	2,686	60	0	3
GP_16	Crawcrook Medical Centre	3,655	3,783	7,438	123	14	8
GP_17	Crowhall Medical Centre	3,505	3,174	6,679	171	20	2
GP_18	Cruddas Park Surgery	5,189	4,517	9,706	262	24	1
GP_19	Denton Park Medical Group	3,399	3,757	7,156	189	17	1
GP_20	Denton Turret Medical Centre	4,031	4,214	8,245	174	11	1
GP_21	Dilston Medical Centre	4,267	3,283	7,550	117	0	5
GP_22	Elvaston Road Surgery	1,207	1,084	2,291	43	6	8
GP_23	Fell Cottage Surgery	4,009	4,397	8,406	183	13	10
GP_24	Fell Tower Medical Centre	3,531	3,791	7,322	151	13	8
GP_25	Fenham Hall Surgery	4,033	4,218	8,251	143	7	2
GP_26	Glenpark Medical Centre	4,479	4,753	9,232	216	16	2
GP_27	Gosforth Memorial Medical Centre	4,242	4,371	8,613	126	0	6
GP_28	Grainger Medical Group	3,533	3,021	6,554	161	9	1
GP_29	Grange Road Medical Practice	1,691	1,916	3,607	65	0	6
GP_30	Heaton Road Surgery	3,784	3,471	7,255	139	10	1
GP_31	Holly Medical Group	4,373	4,530	8,903	103	8	9
GP_32	Hollyhurst Medical Centre	1,245	1,240	2,485	34	0	2
GP_33	Holmside Medical Group	4,593	4,583	9,176	209	31	1
GP_34	Longrigg Medical Centre	5,293	5,546	10,839	260	18	3
GP_35	Metro Interchange Surgery	2,188	1,793	3,981	129	8	1
GP_36	Millennium Family Practice	1,881	1,492	3,373	74	6	1
GP_37	Newburn Surgery	2,615	2,757	5,372	102	10	4
GP_38	Newcastle Medical Centre	7,015	6,475	13,490	34	0	5
GP_39	Oldwell Surgery	2,569	2,677	5,246	127	10	2
GP_40	Oxford Terrace & Rawling Rd Med. Group	7,777	7,387	15,164	368	40	3
GP_41	Park Medical Group	5,805	6,061	11,866	148	12	2
GP_42	Parkway Medical Group	3,861	4,131	7,992	127	0	8
GP_43	Pelaw Medical Practice	2,610	2,703	5,313	111	8	7
GP_44	Ponteland Road Health Centre	1,421	1,420	2,841	52	0	2
GP_45	Prospect Medical Centre	8,285	7,206	15,491	299	25	1
GP_46	Regent Medical Centre	1,931	1,839	3,770	64	0	10
GP_47	Roseworth Surgery	2,526	2,566	5,092	77	11	10
GP_48	Rowlands Gill Medical Centre	3,312	3,513	6,825	185	16	9

General Practitioner Details		Number of Patients			End Point (count)		IMD
No.	Practice	Male	Female	All	Hospital admission	Mortality	
GP_49	Saville Medical Group	15,682	14,962	30,644	375	24	5
GP_50	Scotswood GP Practice	688	652	1,340	40	0	1
GP_51	Second Street Surgery	1,719	1,010	2,729	43	0	1
GP_52	St. Albans Medical Group	4,000	4,259	8,259	225	18	2
GP_53	St. Anthony's Health Centre	3,051	3,056	6,107	161	8	1
GP_54	Sunniside Surgery	1,583	1,670	3,253	44	0	10
GP_55	Teams Medical Practice	2,699	2,439	5,138	125	10	3
GP_56	The Bridges Medical Practice	2,238	2,321	4,559	83	9	1
GP_57	The Grove Medical Group	6,207	6,451	12,658	126	10	10
GP_58	The Park Surgery	1,465	1,590	3,055	97	10	1
GP_59	The Surgery-Osborne Road	2,585	2,480	5,065	46	0	8
GP_60	Thornfield Medical Group	6,165	5,603	11,768	274	24	1
GP_61	Throckley Primary Care Centre	3,217	3,333	6,550	125	13	3
GP_62	Walker Medical Group	5,759	5,545	11,304	259	15	1
GP_63	West Road Medical Centre	4,819	4,325	9,144	174	15	3
GP_64	Westerhope Medical Group	5,991	6,410	12,401	317	20	4
GP_65	Whickham Cottage Medical Centre	8,023	8,336	16,359	327	29	5
GP_66	Wrekenton Medical Group	4,994	5,271	10,265	214	18	1
	Total	**253,301**	**249,537**	**502,838**	**9,693**	**702**	

Source: Purchased data from NHS Digital (2015) and MHCLG (2017)

Resident registrations vary by age range; half of the population lie within the 15-49 age range, 16% within the 0-14 age range, 29% within the 50-79 age range, while a small number, 4%, are in the 80+ age range. Newcastle Medical Centre (GP_38), which does not have branches, is associated with the highest proportion of patients who fall in the 15-49 age range (94%) and the lowest proportion of 3% in the 0-14 age range, 3% among the 50-79 age range and 0.10% within the 80+ age range. This GP site is located in the city centre, where most of the registered patients are most likely students at Newcastle and Northumbria universities. This might explain the zero deaths and minimal rate of hospital admissions, as seen in Figures 4-7 and 4-8, which present distributions of patients by age range and hospital admissions and the rates of mortality with respect to each practice’s registrations.

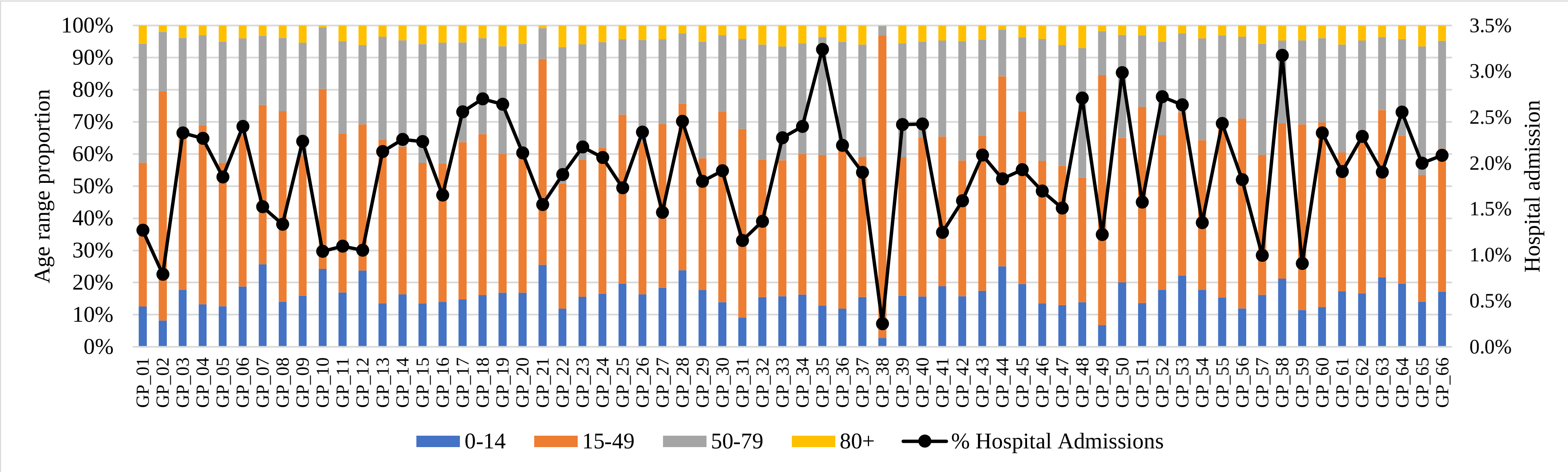


Figure 4-7: Age range proportions and hospital admissions rates for patients by GP registration

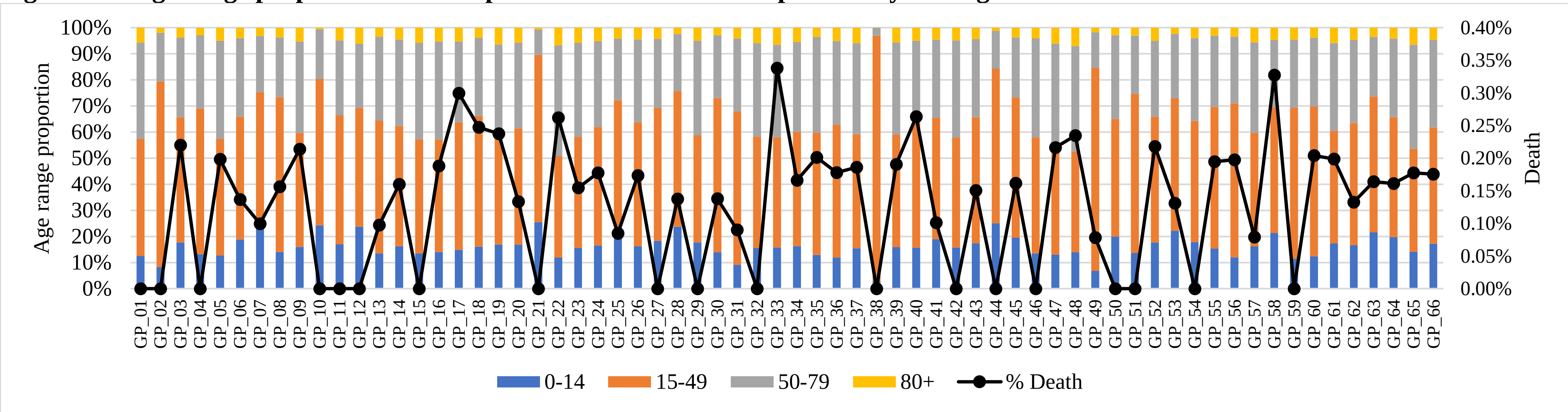
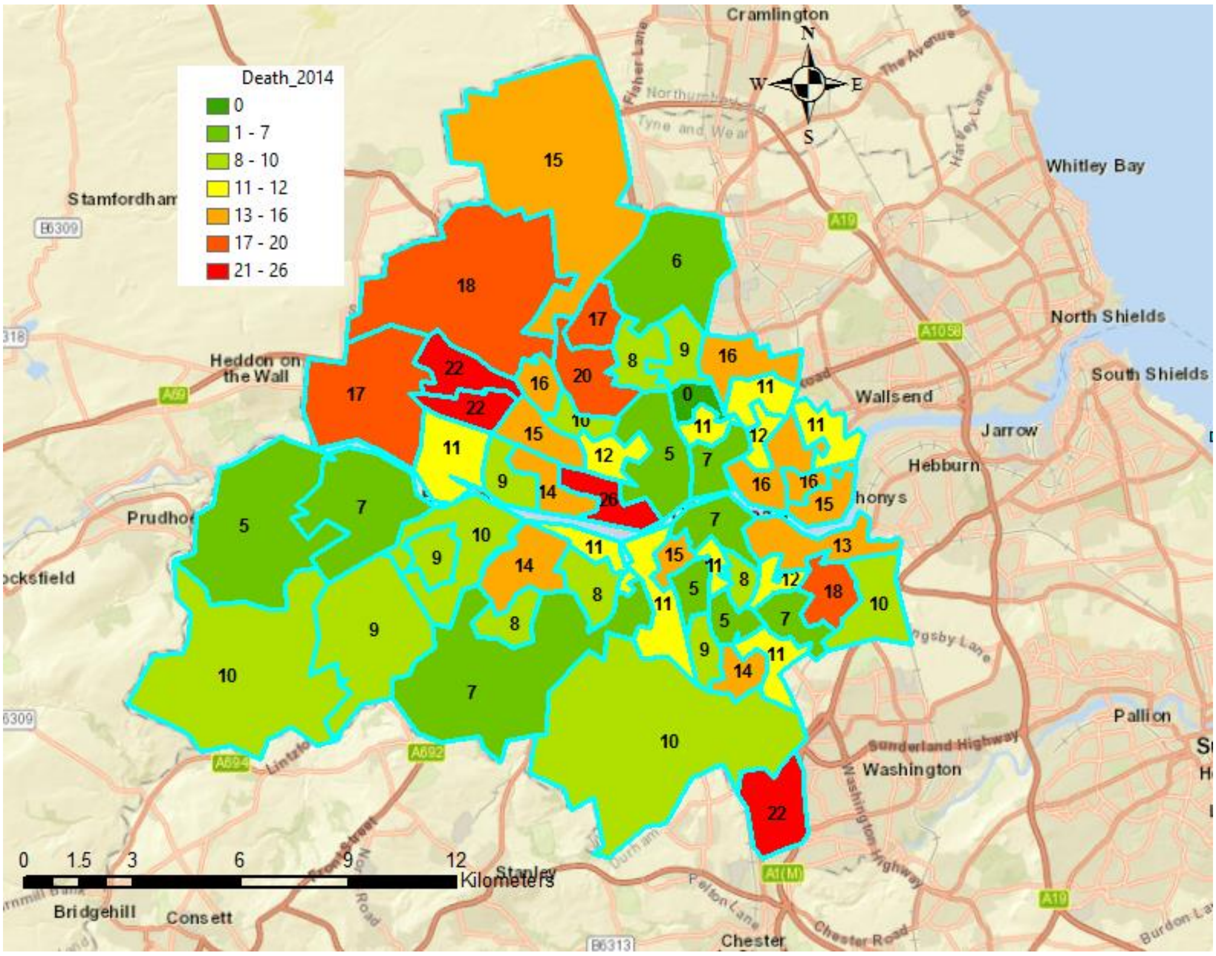


Figure 4-8: Age range proportions and death rates for patients by GP registration

4.5.2 Mortality Data Sourcing: ONS

An additional source of mortality figures is available from the Office for National Statistics (ONS) website, where mortality statistics are published frequently for the deceased by means of their residential address described generally in terms of Middle Super Output Areas (MSOAs). Moreover, underlying cause of death, sex and age range are included (ONS, 2018). Figure 4-9 shows the number of deaths in 2014 attributed to respiratory illness in each MSOA in Newcastle and Gateshead. According to the ONS figures, total mortality in the pilot area was 662, which differs slightly from the figures provided by NHS Digital, which totalled is 702 deaths due to diseases classified as J00 to J99.



Source: Extracted from ONS (2018)

Figure 4-9: Numbers of deaths in 2014 distributed geographically over MSOAs

4.5.3 Introduction to Methods of Exposure Assessment

Human exposure to air pollution can be estimated using direct and indirect methods. Direct approaches use measurements of air pollution in each microenvironment, whereas indirect approaches integrate population models with air quality models (Ott, 1990).

4.5.3.1 Direct Methods

Direct methods tend to be expensive and time-consuming to employ in exposure studies. The main types of direct methods are personal sampling and biological markers. In personal sampling, human exposure to air pollution is estimated from direct measurements of individual exposure to air pollution using personal monitors. Passive samplers are the most common and are relatively inexpensive. These samplers depend on the principle of the passive diffusion of a gas over periods of days and the concentration in air calculated in accordance with Fick's law of diffusion (Palmes et al., 1976) and represents the total pollution over the period of exposure. A number of studies have used other small, lightweight devices for measuring personal exposure to particles (Monn, 2001).

Biological markers can be grouped into exposure markers and effect markers. Generally, a marker of an effect is a pre-clinical indicator of abnormalities which also can include a medical diagnosis (Grandjean, 1995). An exposure marker reflects the concentration of the analyte which has passed a human boundary. Biological markers can be collected from breath, urine, hair, nails, nasal lavage, blood or fluids from bronchoalveolar lavage. The use of biomarkers is most widespread in occupational studies with known specific exposures such as for solvents (Lowry, 1995).

4.5.3.2 Indirect Methods of Exposure Assessment

Although the importance of the direct approach in determining the exposure and its sources for a specific population sample is acknowledged, extrapolation to a larger population is made possible by use of an exposure model which complements the results from direct studies extending and extrapolating findings to other locales and other situations (Ott, 1990). The movement of air pollution from source to a receptor can be estimated by dispersion modelling which employs mathematical models. Dispersion models can estimate both temporal and spatial variations in air pollution at high resolution (e.g. hourly, 200 metres). Common dispersion models use Lagrangian, Eulerian or Gaussian methods. Both Lagrangian and Eulerian models are similar and investigate the chemical and physical processes that occur within an air mass as it moves through a given trajectory. They only differ in terms of the way that the mass of air is represented in space and time. The Lagrangian approach follows the air mass continuously over a trajectory; the Eulerian approach looks at fixed points over specific time intervals (Harrison, 2001, p. 261). The Gaussian distribution is used to model the plume of pollution and is comparatively less computationally intensive; therefore, Gaussian models are more commonly used. In this thesis, concentrations of vehicular emissions were estimated by modelling their dispersion utilising ADMS-Urban programme which is based on the Gaussian model.

Exposure data is obtained in many epidemiological studies based ambient monitoring networks where people living in defined areas (such as a particular city) are assigned the same pollution level. In this type of study, the units of analysis are populations or groups of people rather than individuals, such as in ecological analysis (Last, 2001, p. 56).

Fixed-site monitors are the most common source of data on air pollution in exposure assessment and epidemiological studies. Typically, exposure estimates are made on the basis of air quality data from the nearest site in a national network of monitoring stations. It should be mentioned that the Automatic Urban and Rural Network (AURN) is the UK's largest and main automatic monitoring network for compliance reporting against Ambient Air Quality Directives and measures for NO_x, SO_2, O_3, CO, PM_{10} and $PM_{2.5}$. This network consists of automatic air quality monitoring stations that provide high-resolution hourly information. The use of fixed-site monitors might lead to misclassifications of exposure in areas where there is significant spatial variation in air pollution. Harrison and Deacon (1998) suggested that the number of monitors should be sufficient to cover all the spatial variation within cities. However, practically it is impossible to establish adequate stations to monitor sufficiently. Alternatively, passive samplers which employ test tubes, are widely used for indicative monitoring of ambient NO_2 because they give an indication of longer-term average concentrations, and for highlighting areas of high concentrations such as traffic emissions, and where installation of an automatic analyser is not feasible. However, measurements of NO_2 diffusion tube need to be re-calculated using bias adjustment. This adjustment factor is estimated by comparison with the chemiluminescent analyser located at a fixed site; or for example, a bias adjustment factor of 0.89 was applied to NO_2 diffusion tube results in Newcastle (Newcastle City Council, 2019, p. 55).

4.5.3.3 Likelihood that People are Registered at their Nearest GP Centre and Distance from the Receptor Site

In England, GP practices are responsible for providing residents registered within each practice boundary area with health care. Since 2015, all GP practices in England have been free to register new patients who live outside their practice boundary. However, these arrangements are voluntary for GP practices, which may not provide home visiting services due to the greater distance to a patient's home. In addition, if the capacity of the practice is limited, or it is not clinically appropriate or practical to be registered so far away from home, the practice can refuse registration. However, the practice should explain the reason for refusing any registration.

Aspects that affect choosing with which GP practice to register were evaluated in the UK. Santos *et al.* (2017) used data on the choices made by nearly 3.4 million adults aged 25 and over, from amongst nearly 1,000 family doctor practices. They calculated the straight-line distance between the centroid of 2,875 lower super-output areas (LSOAs) and all GP surgeries in the East Midlands Strategic Health Authority. The mean distance of the centroid of the LSOAs where patients resided to the chosen practice was found to be 1.9 km in the East Midlands (Santos *et al.*, 2017). This implies that a circle with a radius of 2 km that surrounds a GP practice can be considered as a catchment area to identify those people exposed to air pollution.

In addition, it can be argued that older people, who are most vulnerable to health issues related to exposure to air pollution (Spix et al., 1998), have a tendency to register with the closest GP practices, whereas individuals who are employed might prefer to register with a GP practice near their workplace.

4.5.3.4 Implications and Limitations of Using GP Centres as Receptor Sites to Reflect Exposure Levels of Local Populations

In this thesis, is was assumed people chose the nearby GP practices offering health services. GP sites were used as receptors to estimate levels of air pollution in order to evaluate exposure to air pollution of those patients who registered with these practices to receive their health care. This approach is consistent with that used in a PhD thesis recently published at King's College London (Beghelli, 2018). However, this assumption might contain limitations as not all people register with the closest GP practice. Obtaining people's residence details would be more accurate in terms of estimating people's exposure to air pollution. However, this type of information is considered sensitive and an invasion of an individual's privacy and therefore not available for use in this thesis. Therefore, the most appropriate method for this thesis is to assume that people are exposed to the similar level of air pollution as measured at the closest GP practice.

As mentioned previously, the mean distance between the LSOA centroids where patients resides and the registered practice was 1.9 km in the East Midlands (Santos *et al.*, 2017). Levels of air pollution are not expected to have significant variations, unless there is a heavy flow of vehicle traffic and congestion in street canyons. This particularly is the case for NO_2 concentrations.

For example, Milojevic *et al.* (2014) studied the association of hospital admissions and mortality with a range of cardiovascular diseases in England and Wales as well as the short-term effects of pollution. Cardiovascular events of 2 million emergency hospital admissions and 600,000 deaths were linked to exposure to daily mean concentrations of CO, NO_2, PM_{10}, $PM_{2.5}$ and SO_2, and a daily maximum of 8-hourly running mean of O_3 measured at the nearest air pollution monitoring site, which was obtained from the monitoring stations run by the UK National Air Quality Information Archive. They set 50 km as the maximum distance away from the monitoring station to characterise exposure. Similarly, Spix *et al.* (1998), used measured pollutant concentrations at the monitoring stations to assess exposure to air pollution in London and other European cities. In Huddersfield, UK, Kingham *et al.* (2000) investigated small-area variations concerning pollutants. Spatial variations in pollutant concentrations were only modest and they found no significant association either with distance from road or modelled NO_2 concentrations.

Blanchard *et al.* (1999) studied spatial variations in PM_{10} levels within the San Joaquin Valley in California. PM_{10} levels varied by one-fifth over distances from 4 to 14 km from the core sites. In an intra-urban study Burton *et al.* (1996) measured PM_{10} and $PM_{2.5}$ at eight sites ranging from 0.6 km (City Centre Station) to 28.8 km (Valley Forge Station) from the centre of Philadelphia. In general, significant correlations were found for $PM_{2.5}$ and PM_{10} concentrations measured at the eight sites. Pearson correlation coefficients between the sites were stronger for $PM_{2.5}$ ranging from 0.70 to 0.96, and slightly lower for PM_{10} ranging from 0.62 to 0.96. The authors proposed that concentrations at a central monitoring site might be used to characterise exposure concentrations across the city, as well as in other similar cities in the north-eastern United States. Likewise, a study by Wilson and Suh (1997) found that site-to-site correlations of 24-h $PM_{2.5}$ and PM_{10} concentrations among sites distributed across urban areas in Philadelphia and St. Louis were high, with an average correlation coefficient of R=0.90 for $PM_{2.5}$ and R=0.97 for PM_{10} concentrations.

Bari *et al.* (2003) correlated $PM_{2.5}$ hourly and longer-term averages at two monitoring sites in Manhattan and the Bronx, New York city, which are 11 km apart. Correlations between the daily average concentrations of $PM_{2.5}$ at the sites were high (R^2=0.92, with a slope=0.95). Although linear regression of the hourly data at the two sites has an R^2 of only 0.62, the slope is 0.97. Annual absolute concentration levels at the sites were 15.2 μg/m^3 at the Bronx and

15.5 μg/m^3 at Manhattan. DeGaetano and Doherty (2004) measured hourly $PM_{2.5}$ concentrations in New York city using a high-density monitoring network of 20 stations and found similarly low spatial variations in concentrations across the city. Pearson correlations between a central site and all but one of the other sites in lower Manhattan were greater than 0.85. Overall, the between-station correlation of $PM_{2.5}$ observations is relatively high.

Researchers in Washington DC measured $PM_{2.5}$ and PM_{10} concentrations at six sites across the metropolitan area. Pearson correlation coefficients were high and significant for PM_{10}, ranging from 0.64 to 0.98, while those for $PM_{2.5}$ ranged from 0.69 to 0.98. It was concluded that a central stationary monitoring site was sufficient to estimate ambient exposures for PM_{10} and $PM_{2.5}$ (Suh *et al.*, 1997). Li *et al.* (1999) measured PM_{10} at 11 monitoring sites in Vancouver, Canada, and found high temporal correlation between sites and relatively small spatial variation.

Roorda-Knape *et al.* (1998) measured pollution concentrations near motorways in Holland. Black smoke and NO_2 levels decreased with distance from the roadside; however, no concentration gradient was observed for PM_{10} and $PM_{2.5}$. In the Emilia-Romagna region of Italy, a study of 10 urban environments determined that Pearson correlation coefficients between sites for PM_{10} were high with a mean of 0.89 and for NO_2 with a mean of 0.77, were also high (Sajani *et al.*, 2004). The results suggested that a single fixed-site monitoring station may not accurately characterise the spatial nature of air pollution at deep street canyons, high traffic densities or very low ventilation. However, the study found evidence that the intra-urban spatial variability of particulate levels was low. A study in Basel, Switzerland, at six mobile sites and one fixed site within 3.3 km of each other by Röösli *et al.* (2000) found a relatively homogeneous annual mean of PM_{10} mass concentrations ranging from 27.6 to 32.0 μg/m^3. This marked spatial homogeneity in long-term mean PM_{10} levels clearly implies that less error will result from assigning data from one fixed monitoring site to all study subjects living in Basel. In a related study in the same city, Röösli *et al.* (2001) discovered PM_{10} mass concentrations to be uniformly distributed (±10%) across the city at seven sites with the exception of one site, in a street canyon next to traffic signal control.

Using data from the European EXPOLIS (Air Pollution Exposure Distribution within Adult Urban Populations in Europe) study, Oglesby *et al.* (2000) found that, for regional air pollution, fixed-site concentrations are valid surrogates for population exposure in the city. In China, Ye *et al.* (2003) conducted an analysis of weekly $PM_{2.5}$ concentrations over one year at two sites in Shanghai that were 4 km apart. A linear regression between the $PM_{2.5}$ mass concentrations at the two sites had a high correlation (R^2=0.94 and a slope=0.95). Annual mean concentrations at the sites were 57.9 and 61.4 μg/m^3, suggesting regionally homogeneous sources. He *et al.* (2001) measured long-term average concentration variations of $PM_{2.5}$ at two sites 10 km apart in Beijing. Annual mean concentrations of $PM_{2.5}$ at the sites were 115 and 127 μg/m^3 respectively.

The literature has correlated fixed site measurements in urban areas in cities in USA, Asia and Europe for different pollutants at different spatial and temporal resolutions. Correlations ranged from R^2=0.62 to 0.97 but despite difference in pollutants for longer exposures can be argued that in exposure assessment studies it is reasonable to use fixed-site fine particle mass to estimate exposures to regional air pollution, acknowledging that air pollution can be higher in street canyons and heavy traffic flow.

4.5.3.5 Typical Modes Travelled

Ott (1982) defined human exposure as an "*event when a person comes into contact with a pollutant of a certain concentration during a period of time*", which means that exposure requires both the pollutant and the person to be present in a microenvironment. The definition of a microenvironment is *"a small space in which human contact with a pollutant takes place, and which can be treated as a well-characterized, relatively homogenous location with respect to pollutant concentrations for a specified time period*" (USEPA, 2017). The microenvironment could be indoors or outdoors at home, school, work, whilst travelling and in other locations. The exposure time in these microenvironments varies depending on the pattern of time activity. Watson *et al.* (1988, p. 214) illustrated that general form of the personal time-weighted pollutant exposure can be expressed by the following equation:

$$E_i = \sum_j^J C_j t_{ij}$$

where

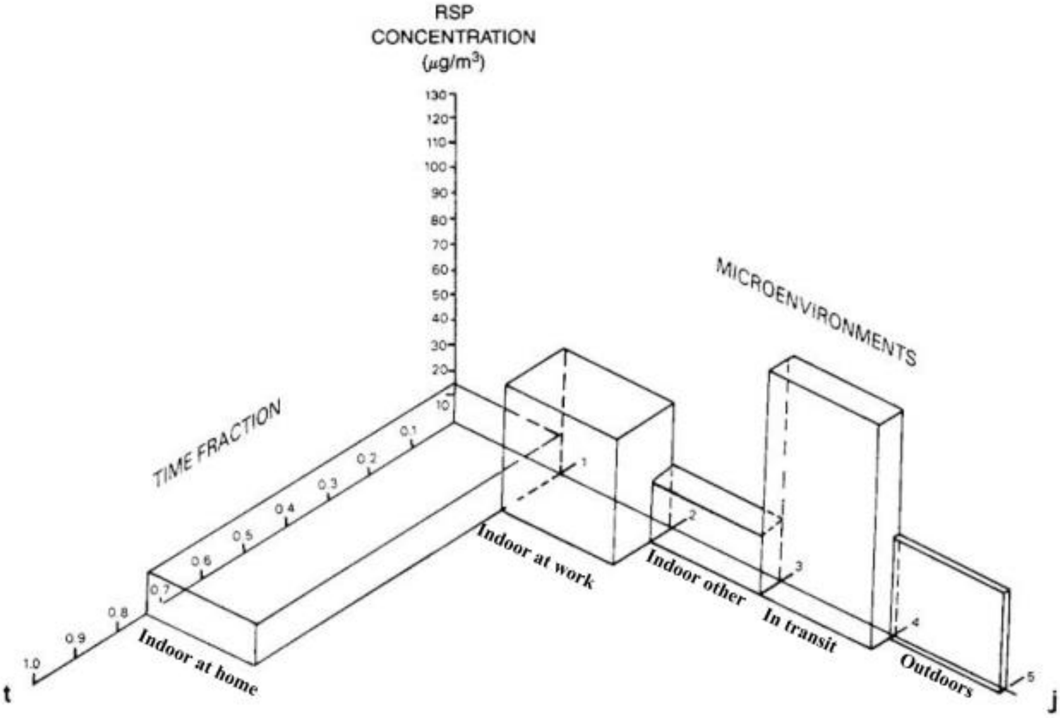
E_i: the time-weighted integrated exposure for person 'i' over the specified time period.

C_j: the pollutant concentration in microenvironment 'j'.

J: the total number of microenvironments that person 'i' visited.

Furthermore, Watson *et al.* (1988, p. 216) illustrated the concept of time-weighted integrated exposure as shown in Figure 4-10. A unit width is indicated on the j-axis for each of five microenvironments with a unit width represented on the j-axis. These are: indoors at home, indoors at work, indoors in other locations, in transit and outdoors. On the Y-axis, the concentration of respirable suspended particles (RSP) is shown, whilst the t-axis illustrates the fraction of time that person i spends in each microenvironment over a 24-hr period. Contributions for each of the five microenvironments to time-weighted integrated exposure are represented as volumes in the Figure.

In addition, the share of each microenvironment is illustrated mathematically in the table at the bottom of the Figure. Although the RSP concentration was small inside the home, it contributed substantially to time-weighted exposure since this person spent 75% of 24 hours indoors at home. Conversely, the graph shows activities distributed over 24 hours. People are exposed to high concentrations of air pollution outdoors; however, as they spend a shorter time outdoors, the high concentration there represents a minor contribution to total time-weighted exposure.



Microenvironment Type	RSP Concentration (C_j, $\mu g/m^3$)	Time Fraction[a] (t_{1j})	$C_j \times t_{1j}$ ($\mu g/m^3$)	Microenvironment Contribution[b] to E_i (%)
Indoors at Home	15	0.75	11.25	47
Indoors at Work	50	0.15	7.5	31
Indoors, Other	25	0.04	1	4
In Transit	90	0.04	3.6	15
Outdoors	40	0.02	0.8	3

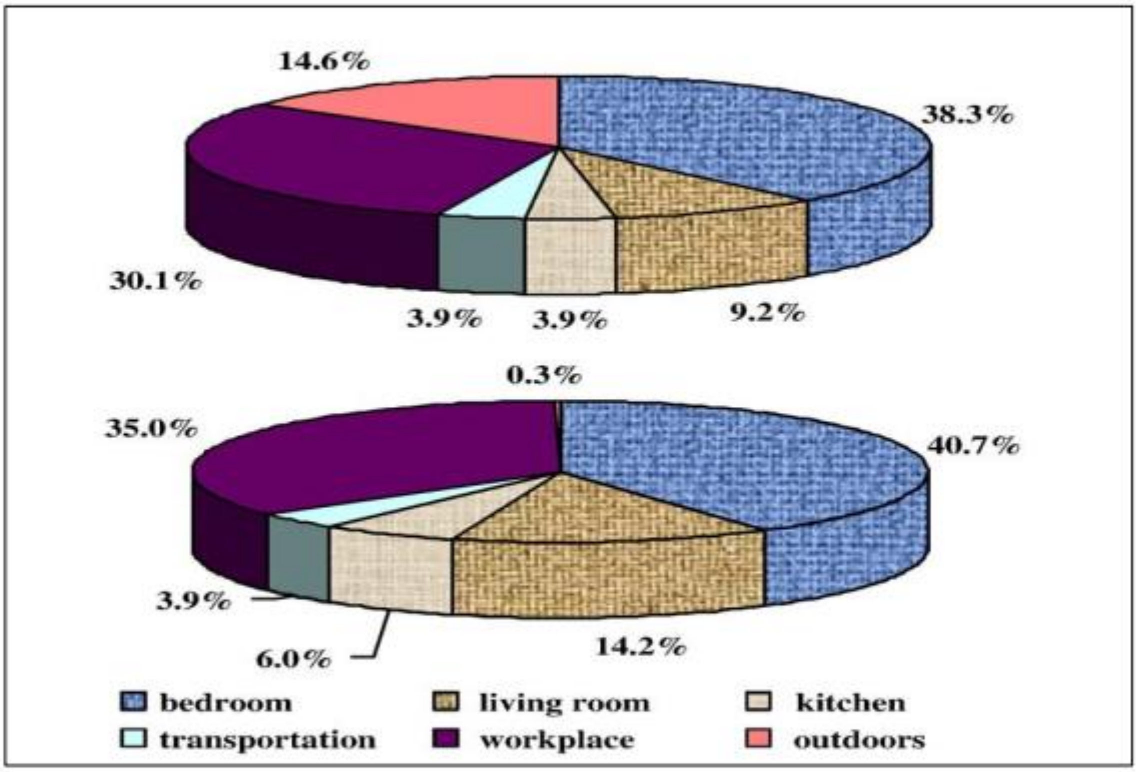
$$E_i = \sum C_j \times t_{ij} = 24.15\ \mu g/m^3$$

a Fraction of 24 hr spent in each microenvironment.
b Percentage that each microenvironment contributes to the 24-hr, time-weighted, integrated exposure (E_i).

Source: Watson et al. (1988, p. 216)

Figure 4-10: Example of the relative contributions from specific microenvironments to an individual's time-weighted, integrated exposure to RSP

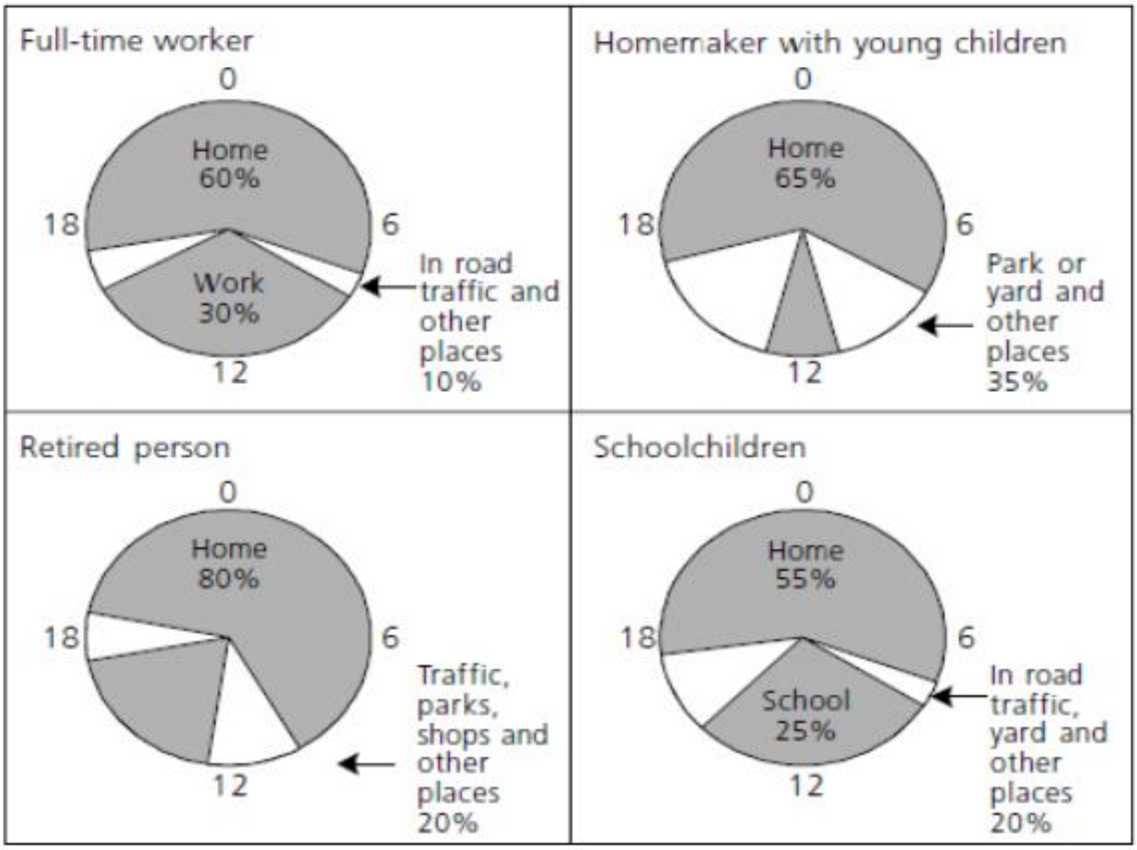
In some regions, microenvironment-associated exposure is affected by the seasons. In a study undertaken in north London in 2009 during summer a person's exposure can reach 15% outdoors, whilst in the winter it typically may only be 0.3%, as reported by Kornartit *et al.* (2010). Nevertheless, personal exposure during transportation remains at 4% in both summer and winter, as shown in Figure 4-11.



Source: Kornartit et al. (2010)
Figure 4-11: Personal exposure distribution in different microenvironments during summer (above) and winter (below)

Levels of air pollution can be used in connection with time-activity patterns in different microenvironments to estimate a measure of total exposure for humans (WHO, 2006)

The actual exposure level relies on the personal activities of the person (WHO, 1999). Figure 4-12 displays different patterns of time-activity profiles for typical days of a full-time worker, homemakers with young children, retired person and schoolchildren (WHO, 1999).



Source: WHO (1999, p. 19)
Figure 4-12: Time-activity profiles for typical 24-hour days

Concentrations of outdoor air pollutants have been used as a surrogate for human exposure in various epidemiological studies, such as those by Autrup (2010), Briggs (2005), Ashmore and Dimitroulopoulou (2009) and Avery *et al.* (2010). Several studies depend on ambient fixed-site measurement stations as proxies for personal exposure (Briggs, 2005; Violante *et al.*, 2006; Tsai *et al.*, 2008; Gerharz *et al.*, 2009; Avery *et al.*, 2010). In fact, measurements provided by fixed-site monitoring stations alone might not reflect good estimations of personal exposure because individuals could spend most of their time indoors (Ashmore and Dimitroulopoulou, 2009; Gerharz *et al.*, 2009; Avery *et al.*, 2010; Kornartit *et al.*, 2010), whereas they are exposed to different air pollutant sources which are different from those outdoors (Gerharz *et al.*, 2009; Avery *et al.*, 2010). Many studies conducted in Europe and North America have highlighted that on average, people spend time ranging from 81% to 94.5% indoors, 3.4% to 14% outdoors and 2.5% to 8% travelling as Table 4-5 shows. Thus, total personal exposure is formed from a combination of personal experience in different microenvironments (WHO, 2006).

Table 4-5: Selected studies presenting total time spent indoors, outdoors and in transport

Author	Location (city/country)	Total Time spent (%)		
		Indoor	Outdoor	Transport
Michikawa *et al.* (2014)	6 Japanese cities	84	-	-
Wang *et al.* (2014b)	Guangzhou, China	81.5	-	-
Mohammadyan (2012)	Bradford, UK	90.8	3.4	4.7
Braniš and Kolomazníková (2010)	Prague, Czech Republic	84.3	10.6	5.1
Johannesson *et al.* (2007)	Gothenburg, Sweden	94.5	4	2.5
Kim et al. (2006)	Toronto, Canada	88.6	5.3	6.1
Wu *et al.* (2005)	Alpine-CA, USA	82.6	14.1	3.3
Lai *et al.* (2004)	Oxford, UK	89.5	3.8	6.7
Jantunen *et al.* (1998)	6 European cites	88	4	8
Burke *et al.* (2001)	Philadelphia, USA	83.0-91.0	10.8	6.2
Overall mean (range)		87 (81.5-94.5)	7 (3.4-14.1)	5 (2.5-8.0)

Adams *et al.* (2001) carried out a multi-modal study of personal and microenvironment exposure to $PM_{2.5}$ in London. A total of 465 journeys were completed by 61 volunteers during both summer and winter. The modes used were bicycle, bus, car and the London Underground at four different times of the day for one central and two outside routes. The study established that levels of exposure to $PM_{2.5}$ were nearly similar inside buses and cars in both seasons, whilst exposure for cyclists was substantially lower than that inside either the bus or car in summer, and marginally lower than both bus and car exposure in winter.

Lower levels of exposure for cyclists were explained by the position of the cyclist on the road, and their ability to avoid in congested traffic. It is worth mentioning that the intake dose for cyclists generally is not the same as the levels of exposure for people travelling by bus and car due to the higher breathing rates associated with cycling. The same can be said to a lesser extent for pedestrian exposure. Pedestrians are, nevertheless, likely to be exposed for longer journey periods than cyclists; hence, the target dose for pedestrians and cyclists might be similar over an equivalent route and distance.

4.6 Creation of the BAU and 2030 Traffic Scenarios

The aim of this research is to investigate the impact of improvements in vehicle propulsion technology, such as the increasing penetration of EVs, on emissions, air quality and the disease burden in 2030 compared to 2014. The Baseline year 2014 was selected at the time this research commenced due to the availability of recent measurements of traffic data and actual pollution concentrations, which are the data required for the validation of the model created in this study.

The road network and traffic flow data for 2010 for the study area, was acquired from Newcastle University and was updated to provide the 2014 Baseline, taking into consideration traffic growth for each vehicle class, as stated in DEFRA's guidelines (DEFRA, 2010). The Baseline traffic model was validated following the Design Manual for Roads and Bridges criteria (DMRB, 1997). The released emissions rates were calculated using an emissions model (PITHEM) and the dispersion of emissions was modelled using the air quality programme (ADMS-Urban) taking account of the effect of meteorological factors. The air quality results were validated following DEFRA Technical Guidance (TG16) (DEFRA, 2016d).

Moreover, the 2014 Baseline traffic was updated to BAU for 2030 and six future scenarios were developed based on this BAU. These scenarios include:

1. 'CCC': Committee on Climate Change proposal for 30% of cars and 38% of vans are electric;
2. 'E-Bus': Electrification of all buses;
3. 'E-Car': Electrification of all cars;
4. 'E-Car_E-Bus': Electrification of all cars and buses;
5. 'E-Car_E-LGV': Electrification of all cars and LGVs; and
6. 'All-EV': Electrification of all vehicles.

Modelling of emissions rates and dispersion for 2030 scenarios was performed by following the same procedure for modelling emission rates and dispersion for the Baseline.

4.7 Software Used

This project requires the use of several software packages; most are either free or are relatively inexpensive. The traffic flow and the geometry of the road links are available in ‘shape’ file format, as the TPM files were received in this format. Hence, the use of ArcGIS software was crucial to read and update the traffic flows and identify hotspots relating to areas of high pollution. Moreover, PITHEM, which is able to read ‘shape’ files, was utilised to calculate emission rates and generate output files that are compatible with ADMS-Urban.

4.7.1 PITHEM

The Platform for Integrated Traffic, Health and Emission Modelling (PITHEM) software was developed at Newcastle University to calculate traffic related parameters such as vehicle kilometres travelled and rates of pollutant emissions.

The PITHEM software allows the development of bespoke mappings between user classes defined in a traffic model such as SATURN, CUBE and AIMSUN with the hierarchical vehicle fleet information within the UK National Atmospheric Emissions Inventory (NAEI) (Namdeo and Goodman, 2012). PITHEM comprises an integral emissions model which calculates emissions and particulates levels based on vehicle flows and network speed outputs using UK National fleet emissions factors which are determined as a function of vehicle type, age, emissions control standard, engine size and fuel used. These factors are applied via PITHEM to twenty-four hour traffic counts and traffic speed data obtained for each link in the network (O’Brien *et al.*, 2012). The PITHEM software outputs emissions information in both vector format for import into GIS and in a text-based format compatible with the suite of ADMS air quality models used for dispersion calculations (Tiwary *et al.*, 2013).

According to Goodman *et al.* (2016), the PITHEM software is used to provide the assessment of key environmental criteria based on the output from the traffic assignment model. The environmental criteria used are divided into four broad categories:

a) Greenhouse gas emissions (tailpipe ‘ultimate CO_2’ or ‘uCO_2’ emissions), which are calculated based on the speed-emissions curves presented in Boulter *et al.* (2009b) and which form part of the UK Emissions Factor Toolkit, Version 5.1.3 (EFT 5.1.3). The emissions function forms a ‘U-shaped’ curve, which is considered to be valid over a type-specific speed range (typically 5–120 km/h for light vehicles and 10–90 km/h for heavy vehicles). Total emissions for a link are calculated from the summation of individual contributions for all traffic types active in the period.

b) Local air quality emissions (tailpipe NO_x, HC and primary NO_2 emissions, PM_{10}, $PM_{2.5}$ including brake and tyre wear components). Similar to the uCO2 calculation methodology, these air quality emissions are calculated with additional fuel quality and vehicle mileage scaling correction factors from the EFT 5.1.3 applied after calculation of the base emissions rate. For particulate matter and hydrocarbons, the emissions functions have the same form as those for uCO_2 described above, with coefficients from Boulter *et al.* (2009b). For NO_x a variety of functions are used based on those found in the COPERT4 (COmputer Program for Emissions from Road Transport) (EEA, 2013).

c) Noise levels at the roadside

d) Standardised axle loading applied to the road.

As noise and axle loadings are outside the scope of the research presented in this thesis, no further details are provided. However, it is recognised that the PITHEM method leads to significant underestimation of emissions on particular streets and junctions where congestion and queues build-up and prevail for a high proportion of the day (O'Brien *et al.*, 2012). Another limitation is that the PITHEM is not updated to the latest ninth version of the EFT.

Vehicular emission rates were calculated by PITHEM, and then emission dispersion was modelled by ADMS-Urban. The following steps describe how is PITHEM run:

a) The first step is to assign vehicle user classes, which then serve as a template for the PITHEM internal library format. The Baseline traffic model comprises user classes as car, LGV and HGV; therefore, three user classes need to be defined in this step. From the 'Classes' tab, UC1, UC2 and UC3 are added and assigned to cars, LGVs and HGVs respectively, as shown in the snapshot of the PITHEM 'Classes' tab in Figure 4-13.

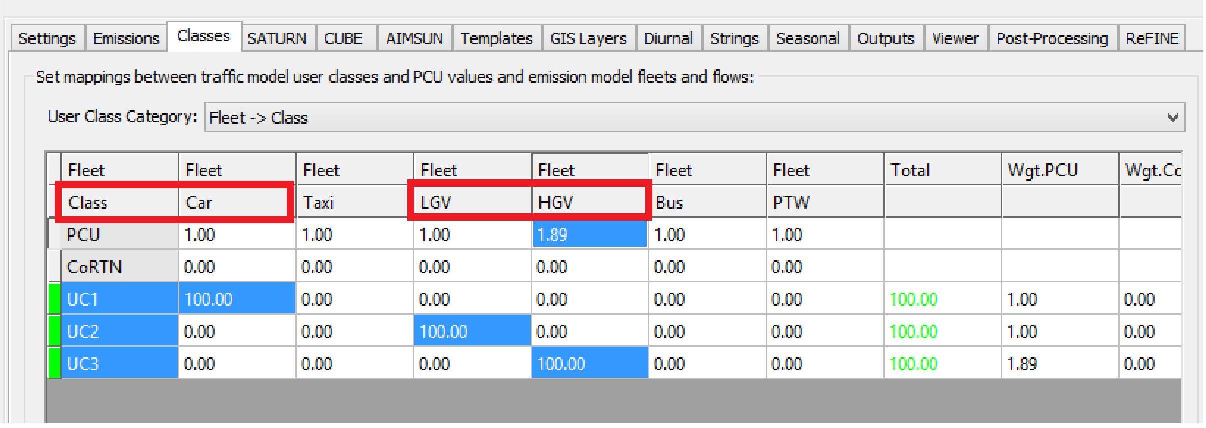


Figure 4-13: Assessment of vehicle classes and PCU equivalency

b) The second step is to add a template by means of the ‘Templates’ tab. The template parameters are UC1_F, UC2_F and UC3_F, which were defined in the previous step representing the flow of cars, LGVs and HGVs; and UC1_S. UC2_S and UC3_S representing the speed of cars, LGVs and HGVs. Mapping combinations need to be built between the PITHEM parameters (e.g. UC1, UC2) and the corresponding ones from files among the model data files. A database file (.dbf) that is “Gen-Traf_Merge_0”, containing the model traffic details for the period 00:00 to 01:00, is used in this step. A snapshot of the ‘Templates’ tab from PITHEM is shown in Figure 4-14. It is crucial to complete the combinations by clicking on the ‘Save to Manager’ button, in order to ensure compatibility between the PITHEM internal data format and the input model file arrangement.

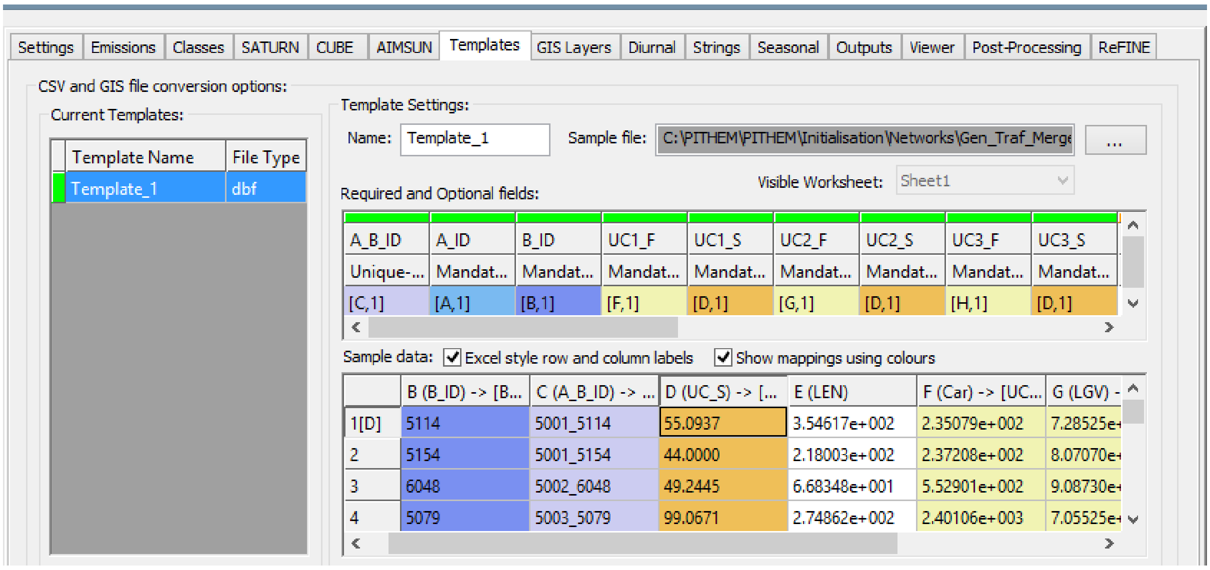


Figure 4-14: Mapping combinations between an input model file and template internal data

c) All of the 24 shape file layers (representing 24 hours of the day) for the model (e.g. Gen-Traf_Merge_23 represents traffic flow at 11pm) need to be loaded into the template through the 'GIS Layers' tab, as the snapshot in Figure 4-15 demonstrates. It can be noticed that only 12 layers can be seen in the Figure, where the rest of the files do not appear due to limitations of space.

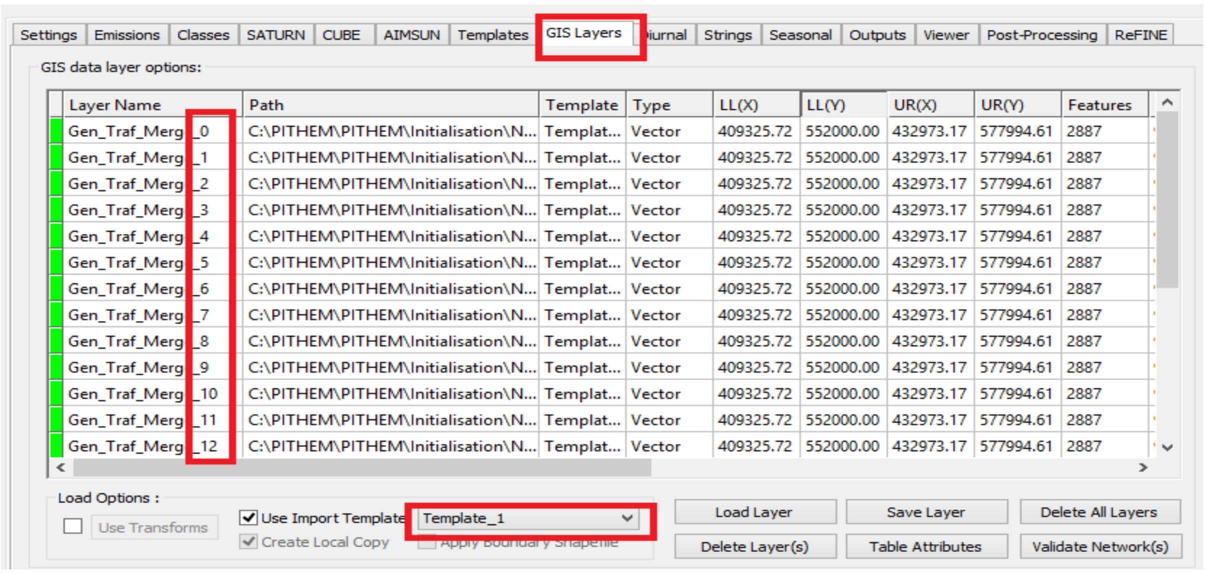


Figure 4-15: Loading of the Baseline model shape files to the template layers

d) In this step, the 'Diurnal' tab is used to link the model GIS layers to each time period. In other words, PITHEM requests the use of traffic data from the file named "Gen-Traf_Merge_0" for the period 00:00 to 01:00 and "Gen-Traf_Merge_1" for the period 01:00 to 02:00 and so on until "Gen-Traf_Merge_23" for the period 23:00 to 24:00.

Flow details in the Baseline model are divided into three time periods: Morning Peak (AM) (07:00-09:59), Inter Peak (IP) (10:00-15:59, 19:00-06:59) and Evening Peak (PM) (16:00-18:59), as can be observed in Figure 4-16.

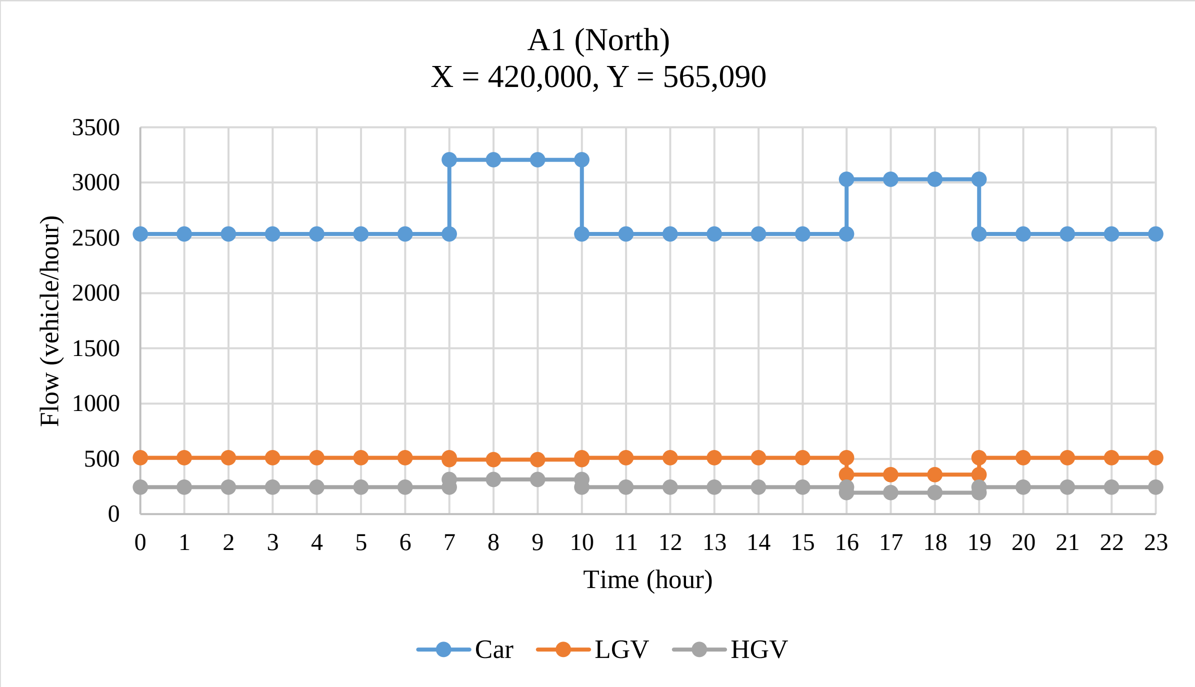


Figure 4-16: Three flow periods modelled in the Baseline model

Figure 4-17 illustrates the diurnal profile for the traffic flow after applying 'Flow % Multiplier' (scaling) factors to the AM, IP and PM peaks (see Figure 4-16). The distribution of traffic flow over the 24-hour period were found to be more realistic than what was depicted in the Transport Planning Model (TPM) for the year 2010. This is mainly because of the assumption in TPM that traffic flows remain constant throughout the duration of the morning, inter and evening peak hours. For example, the traffic flow is considered to be constant throughout the three hour AM peak period embracing 8am, 9am and 10am which is not realistic. These scaling factors were proposed in the study areas to update traffic activities from 2010 to 2021 (Goodman *et al.*, 2014, p. 155).

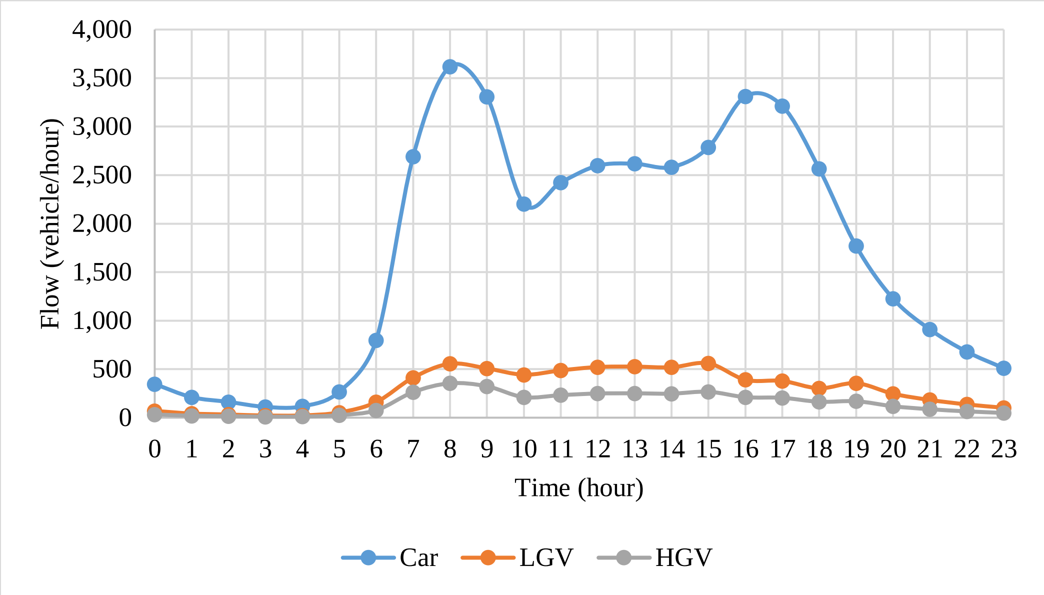


Figure 4-17: Diurnal profile of traffic flow

Therefore, it is important in this step to assign flow distributions across the hours of the day by defining 'Flow % Multiplier' factors, in order to scale the diurnal flow profile. An image of the 'Diurnal' tab is presented in Figure 4-18.

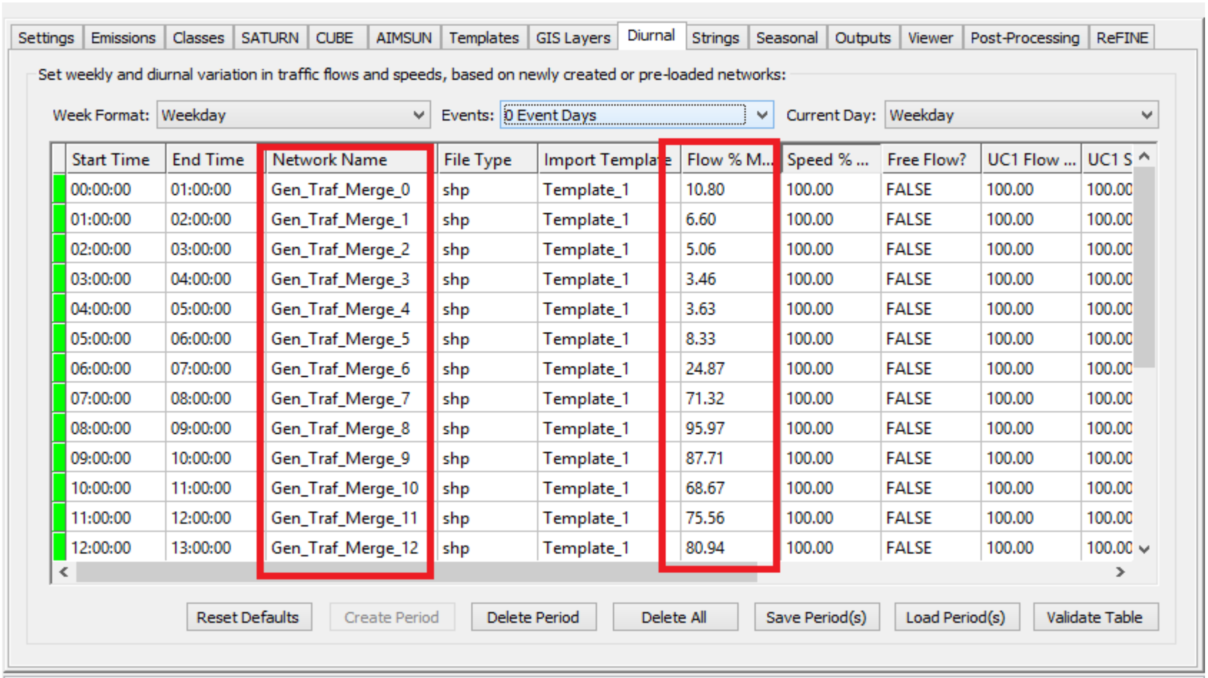


Figure 4-18: Scaling the model traffic flow by assigning hourly "Flow % Multiplier"

e) The next step is to set up calculation parameters in the 'Outputs' tab; thus, the road type and the year are set to 'Urban (England)' and 'year' (e.g. 2014 for Baseline). In addition, the name of the output folder is entered wherever PITHEM will store the generated files, as revealed in Figure 4-19.

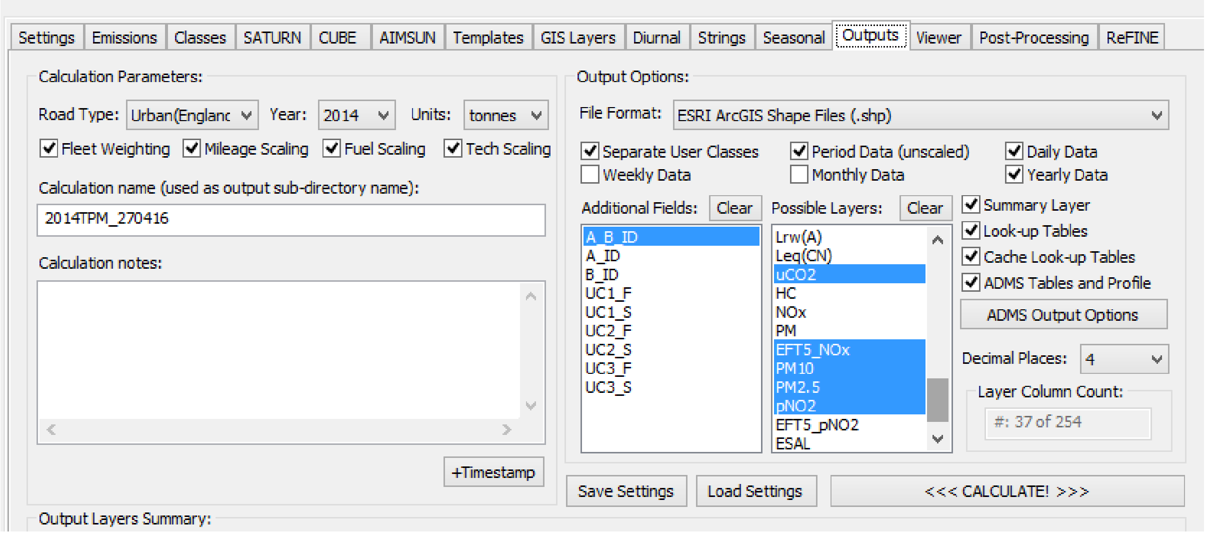


Figure 4-19: Setting up the calculation parameters in the PITHEM 'Outputs' tab

In the same tab, in order to allow PITHEM to generate the (.csv) file to be extracted later by ADMS-Urban, some settings need to be selected on the 'ADMS Output Options' section. The 'Target Version' drop-down menu should be set to the current available version of ADMS-Urban; and 'Output as ADMS 3.2 .csv, .eit, .tft and .vgt' files' must be verified, as seen in Figure 4-20.

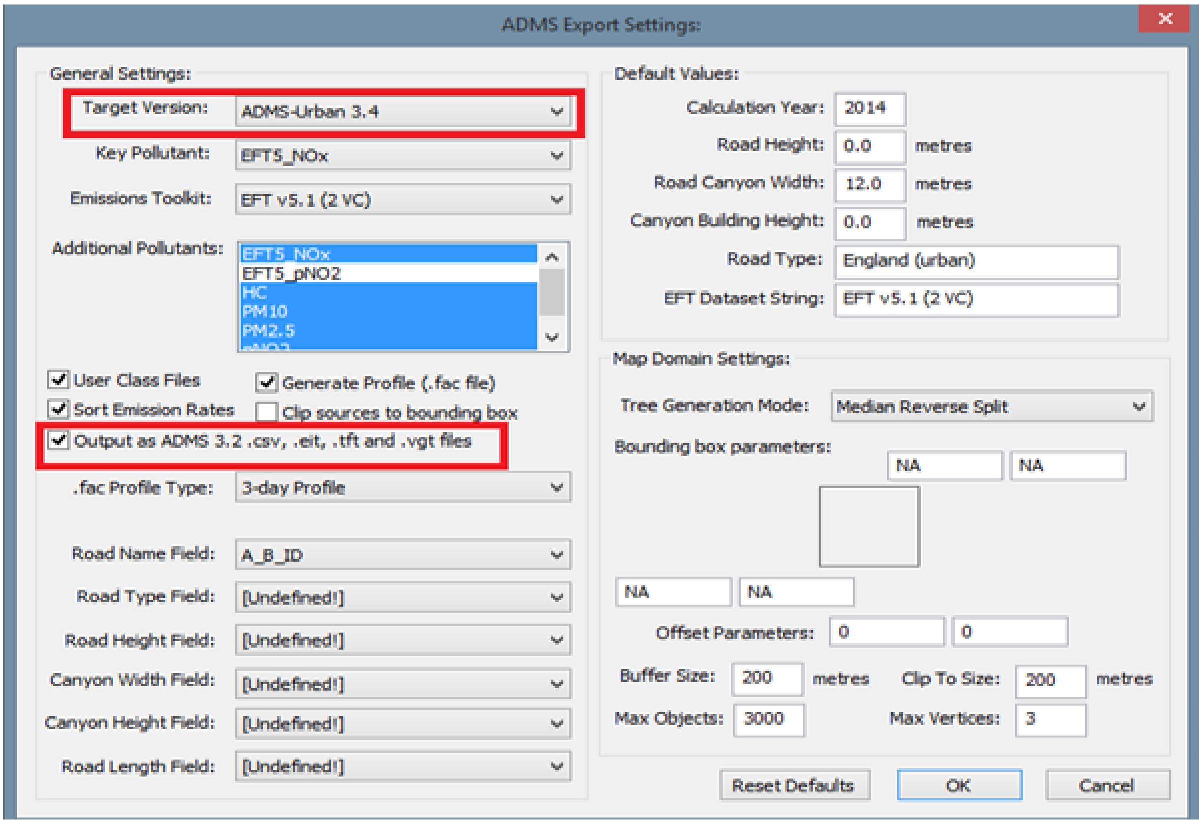


Figure 4-20: Setting up of the ADMS output options

Finally, by clicking the 'CALCULATE!' icon, PITHEM will start the calculation of the traffic emissions rates. The generated files comprise several 'csv' outputs that are compatible with ADMS-Urban use such as:

- File containing NO_x, PM_{10} and $PM_{2.5}$ rates (or other pollutants as selected)
- File containing the geometry of roads and locations of street canyons
- File containing time-varying emissions factors

4.7.2 Atmospheric Air Quality Dispersion Model

Spatial dispersion of various pollutants released into the atmosphere can be predicted effectively by the application of air quality models. In many aspects of air pollution control and overall air quality management system, these models are considered significant (Elbir *et al.*, 2010). Air quality models are used to complement continuous monitoring in order to carry out air quality assessment, forecasting and planning interventions across Europe, at both national and local levels, and this use is supported by the Council Directive 2008/50/EC, on Ambient Air Quality and Cleaner Air for Europe. The measurements of pollution concentrations from ambient air quality monitoring are the essential part of air quality assessment; however, it gives information of air pollution level only representative of a very limited area of the city, under certain meteorological conditions and of a specific situation without the capability to evaluate temporal and spatial scales (Dėdelė and Miškinytė, 2015b). Models of air pollution dispersion are able to provide estimates of the complete spatial coverage of air quality in areas with insufficiently dense monitoring networks (Zou *et al.*, 2009; Johns *et al.*, 2012).

Mathematical equations are used in dispersion models to describe the atmosphere, dispersion as well as chemical and physical processes within the plume, to calculate pollutant concentrations at various sites. The relative cost of advanced dispersion modelling, such as Lagrangian and Eulerian models is likely to take longer computation times because of the large number of variables associated with these models compared to Gaussian models (Bluett *et al.*, 2004, p. 20; Lagzi *et al.*, 2014, p. 111). Therefore, Gaussian models are commonly employed in decision support software if a robust model set-up and fast response time to a problem is a crucial priority (Leelőssy *et al.*, 2014).

Among Gaussian models, the ADMS-Urban has an intuitive graphical interface which is integrated with ESRI's ArcView Geographical Information System (GIS). Microsoft Access is used for an emissions inventory database and output from the model is easily imported into other standard applications such as Microsoft Excel for the post-processing of results.

According to CERC (2017), the main features of the ADMS-Urban are summarised below:

• An advanced dispersion model in which the boundary layer structure is characterised by the height of the boundary layer and the Monin-Obukhov length, a scale length is dependent on the friction velocity and the heat flux at the surface.

• A non-Gaussian vertical profile of concentration in convective conditions which allows for the skewed nature of turbulence within the atmospheric boundary layer that can lead to high concentrations near the source.

• A meteorological pre-processor which calculates boundary layer parameters from a variety of input data: e.g. wind speed, date, time, cloud cover or wind speed, surface heat flux and boundary layer height. Meteorological data may be statistically analysed or raw hourly averaged data for the whole year.

• Point, line, area and volume sources

• An integrated street canyon model

• Realistic calculation of flow and dispersion over complex terrain

• Modelling of chemical reactions involving NO, NO_2 and O_3

• A unique ability to model odours using short-term concentration fluctuations

• Easy to use graphical user-interface

• Integration with Geographical Information Systems (GIS) and an Emissions Inventory Database. Table 4-6 summarises the features of several Gaussian dispersion models.

Table 4-6: Summary of selected Gaussian models' features

ADMS	Atmospheric Dispersion Modelling System	The impact of complex terrain or convective boundary layer turbulence is parameterised to provide more accurate prediction in environmental applications. ADMS provides a range of modules specified for different locations such as urban, coastal or mountain areas, Calculates deposition and radioactive doses (Leelőssy *et al.*, 2014). ADMS allows for plume channelling caused by topography (Bluett *et al.*, 2004, p. 10). Moreover, simulates the effects of coastal fumigation when modelling the dispersion of pollutants from a source located near the coast (Bluett *et al.*, 2004, p. 33). It performs well at receptors < 50 m from source (Bluett *et al.*, 2004, p. 61).
CTDM	Complex Terrain Dispersion Model	The impact of complex terrain or convective boundary layer turbulence is parameterised to provide more accurate prediction in environmental applications. It provides a range of modules specified for different locations including urban, coastal or mountain areas, Calculates deposition and radioactive doses (Leelőssy *et al.*, 2014). Performs well in complex terrain where there are receptor locations above stack tops (Vallero, 2008, p. 590).
CALINE3	California Line Source Dispersion Model	CALINE can model roadways, intersections, street canyons, parking areas, bridges and underpasses (Bluett *et al.*, 2004, p. 30). However, complex terrain should be approached with care (Bluett *et al.*, 2004, p. 31). CALINE is a steady-state model and is not designed to emulate the changing rate of emissions from decelerating, idling and accelerating vehicles (i.e. the emissions rate for each roadway element in the model is an hourly average) (Bluett *et al.*, 2004, p. 31). CALINE does not allow gridded receptors to be used. The user can define a maximum of 20 receptors (Bluett *et al.*, 2004, p. 31). In CALINE, line sources are divided into strings of point sources (Bluett *et al.*, 2004, p. 41).
BLP	Buoyant Line and Point Source Dispersion Model	Designed to handle unique modelling problems associated with aluminium reduction plants, besides other industrial sources where plume rise and downwash effects from stationary line sources are important (Vallero, 2008, p. 606).
ISC	Industrial Source Complex Model	For industrial sites (Leelőssy *et al.*, 2014). ISC was replaced with AEROMOD (Vallero, 2008, p. 604). AERMOD is a source-oriented model ideal for examining the impact of terrain and meteorological conditions and source factors such as stack height when evaluating exposure to all pollutants.
ALOHA	Areal Locations of Hazardous Atmospheres	ALOHA is most commonly used as an accidental release model and used worldwide for response, planning, training and academic purposes (Leelőssy *et al.*, 2014). ALOHA can predict rates of chemical release from broken gas pipes, leaking tanks and evaporating puddles and also can model the dispersion of both neutrally buoyant and heavier-than-air gases. ALOHA is intended for use during hazardous chemical emergencies and was designed to be easy to use so that inexperienced responders can use it during high-pressure situations (Bluett *et al.*, 2004, p. 37).
OCD	Offshore and Coastal Dispersion model	For coastal areas, the model estimates the overwater dispersion by use of wind fluctuation statistics in the horizontal and the vertical measured at the overwater point of release. Lacking these measurements, the model can make overwater estimates of dispersion using the temperature difference between water and air (Vallero, 2008, p. 590).
AIRVIRO	Air quality dispersion modelling system	Airviro has a plume model based on a Lagrangian–Gaussian formulation, recommended for areas where the topography is reasonably flat (Namdeo *et al.*, 2002).

4.7.2.1 Model Selection

The main issues to be taken into account when selecting the most suitable dispersion model were identified by Bluett *et al.* (2004, p. 8) as follows:

a) The potential scale and significance of potential effects, including the sensitivity of the receiving environment (e.g., human health versus amenity effects).

b) The complexity of dispersion (e.g., terrain and meteorology effects).

The description of the fate of an emission in the atmosphere from a point, area or line source is the qualitative aspect of dispersion theory (Tiwary and Colls, 2010, p. 273). It should be stated that there are three types of air pollution dispersion model, along with various combinations of those types.

4.7.2.2 Gaussian Model

The Gaussian model is one of the oldest types of dispersion models (dating back to ca. 1936) and remains in use to this day. It is possibly the most accepted computational process used to calculate the concentration of a pollutant at a certain point. There are a number of versions of the Gaussian plume model. A well-known and highly respected equation is the Pasquill-Gifford model. Pollution concentrations that are at distances <10 km from their sources of emission can be approximated via Gaussian plume models. However, it should be noted that the Gaussian plume may not always be the best model. The following criteria should be applied to decide whether to use a Gaussian-plume model or a more advanced model (Bluett *et al.*, 2004, p. 10):

a) Plume models are typically only applicable to near-field (within 10km from the source) calculations. It is risky to assume the meteorological conditions will be the same at a distance greater than 10 km away from the source.

b) The weather may not be consistent in such situations, owing to various meteorological phenomena in addition to sea breezes or slope and valley flows. Most Gaussian-plume models do not allow for the plume channelling generated by topography, although ADMS3 and CTDM are exceptions.

c) Plume models treat SO_x and NO_x chemistry in terms of simple exponential decay; therefore, it fails to address comprehensively the mechanisms involved in atmospheric chemistry. On the other hand, they can simulate particular chemical processes (such as the production of NO_2 from NO_x) as a post-processing step. Advanced models can manage SO_x, NO_x and organic chemistry and aqueous-phase chemistry, as well as secondary aerosol production.

Several other Gaussian models are available to be used in research, such as CALINE3 for highway air pollution, BLP and ISC for industrial sites, ALOHA for accidental and heavy gas releases as well as OCD for coastal areas (see Holmes and Morawska (2006)) for further details). These models are commonly used by environmental protection organisations, local authorities and industry to conduct health risk investigations and impact studies (Krishna *et al.*, 2005; Silverman *et al.*, 2007). Their short runtime permits users to produce statistical simulations over a long-term period (Athanassiadou *et al.*, 2010; Leelőssy *et al.*, 2011) or detailed sensitivity studies (Sriram *et al.*, 2006; Bubbico and Mazzarotta, 2008). They also enable immediate first-guess information to be provided in cases of accidental release. They are frequently used in combination with GIS software to produce an efficient decision support tool for risk management (Zhang *et al.*, 2000; Namdeo *et al.*, 2002).

The ADMS model is also commonly used in relation to air quality simulations. It provides a range of modules required for different locations such as urban, coastal or mountain areas, whilst it also has the ability to calculate deposition and radioactive doses (Carruthers *et al.*, 1994). The ADMS model is popular in Europe for use in environmental impact studies and urban air quality prediction (Carruthers *et al.*, 2003a). As regards urban air quality prediction, ADMS-Urban includes pre-defined emission scenarios, in addition to a chemistry model used to calculate interactions between plumes of several point and line sources in an urban area (McHugh *et al.*, 1997).

ADMS-Urban was employed to compare three different sets of road traffic emission factors released by the UK government for use in air quality review and assessment in the Greater Manchester area (Peace *et al.*, 2004).

In addition to ADMS-Urban, AirVIRO is a Gaussian air quality dispersion model which has been applied to study the development of asthma and related symptoms longitudinally over the first 12 years of life in relation to exposure to PM_{10} and NO_x from road traffic in Sweden (Gruzieva *et al.*, 2013).

4.7.2.3 Eulerian model

The Eulerian method describes the behaviour of pollutant species relative to a fixed coordinate system (Piver, 1987). This model is appropriate for estimating the dispersion of non-inert pollutants over complex terrain. Reynolds *et al.* (1973) used an Eulerian model to investigate episodic O_3 in urbanized areas. Williams (2003) has described the application of various Eulerian models relating to long-range transport modelling and improving air quality.

The most significant difference between Eulerian and Lagrangian models is that the former method treats the particle phase as a continuum and develop conservation equations on a control volume basis, in a similar form as that for a fluid-phase system. On the other hand, the Lagrangian method considers particles as a discrete phase and tracks the path of each individual particle (Zhang and Chen, 2007).

Researchers in the field of air pollution have recently begun to focus more on Eulerian models. Yet, as with other models, this process also has some disadvantages, including high computational costs, insufficient mass conservation, and numerical diffusion, besides the requirement for substantial computer and modelling expertise (Moschandreas *et al.,* 2002; Holmes and Morawska, 2006; Zhang and Chen, 2007). Sodemann and Zubler (2010) and Zhang and Chen (2007) provided a detailed discussion on the benefits and drawbacks of Eulerian and Lagrangian methods.

4.7.2.4 Lagrangian Model

The Lagrangian dispersion model mathematically explains pollution plume parcels (also termed particles), as they move through the atmosphere using a random walk process. Lagrangian modelling is regularly employed to cover longer time periods (Builtjes, 2001). Nevertheless, as with other models, Lagrangian models have various advantages and disadvantages (Williams, 2003; Gertler *et al.*, 2006). These benefits and shortcomings are shown below.

Benefits:

a) Relatively inexpensive in terms of computer time.

b) Effective in relating emissions from sources to concentrations at receptor sites.

c) Can be traced back to the source of the pollution.

d) Rapidly evaluate the effect of emissions inventories on pollutant levels.

Shortcomings:

a) Complex wind fields can cause incorrect predictions concerning pollutant location.

b) Difficulty in accounting for chemical interactions between different air parcels.

c) Cannot deal with complex chemistry.

d) Overestimate pollutant concentrations.

4.7.2.5 Air quality Modelling by ADMS_Urban

While driving along roads, vehicle flows cause emissions which are dispersed because of meteorological elements such as the speed and direction of the wind. In order to estimate the concentrations of these dispersed emissions, ADMS-Urban was used to specify emissions concentrations at selected receptors.

The calculation of the emissions rates related to the Baseline and 2030 traffic model was performed by the PITHEM which has prepared (.csv) files for use by ADMS-Urban use. Moreover, the extraction and then conversion of those files were executed by means of ADMS-Urban to produce a compatible file (.upl) that can be run later by ADMS-Urban. The steps to run ADMS-Urban are listed below:

a) From the ‘Utilities’ drop-down menu, clicking on “Extract sources from CSV file” will allow ADMS to extract the (.csv) file and convert it to an (.upl) file which is compatible for ADMS use. A snapshot representing the first step in ADMS is shown in Figure 4-21.

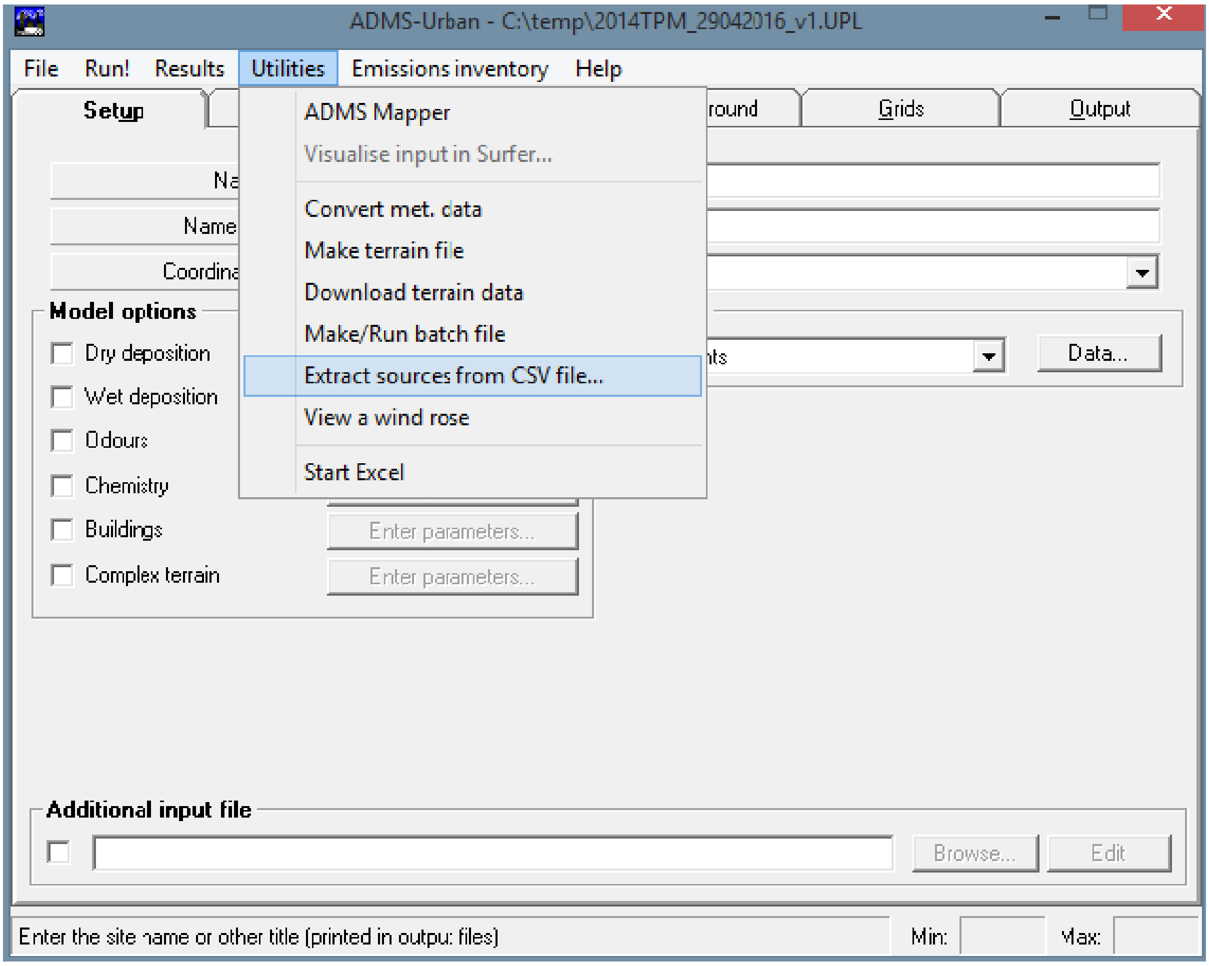


Figure 4-21: Extracting and converting a (.csv) file

b) The second step is to open the (.upl) file which is a conversion of the (.csv) file that was extracted in step (a). This file contains the geometry of 2887 roads and their related hourly emission rates. Time-varying emission factors should be defined manually by inputting the required file, which is in (.fac) format and was generated through the use of PITHEM (see Figure 4-22). Those factors identify the variations in emissions through the day.

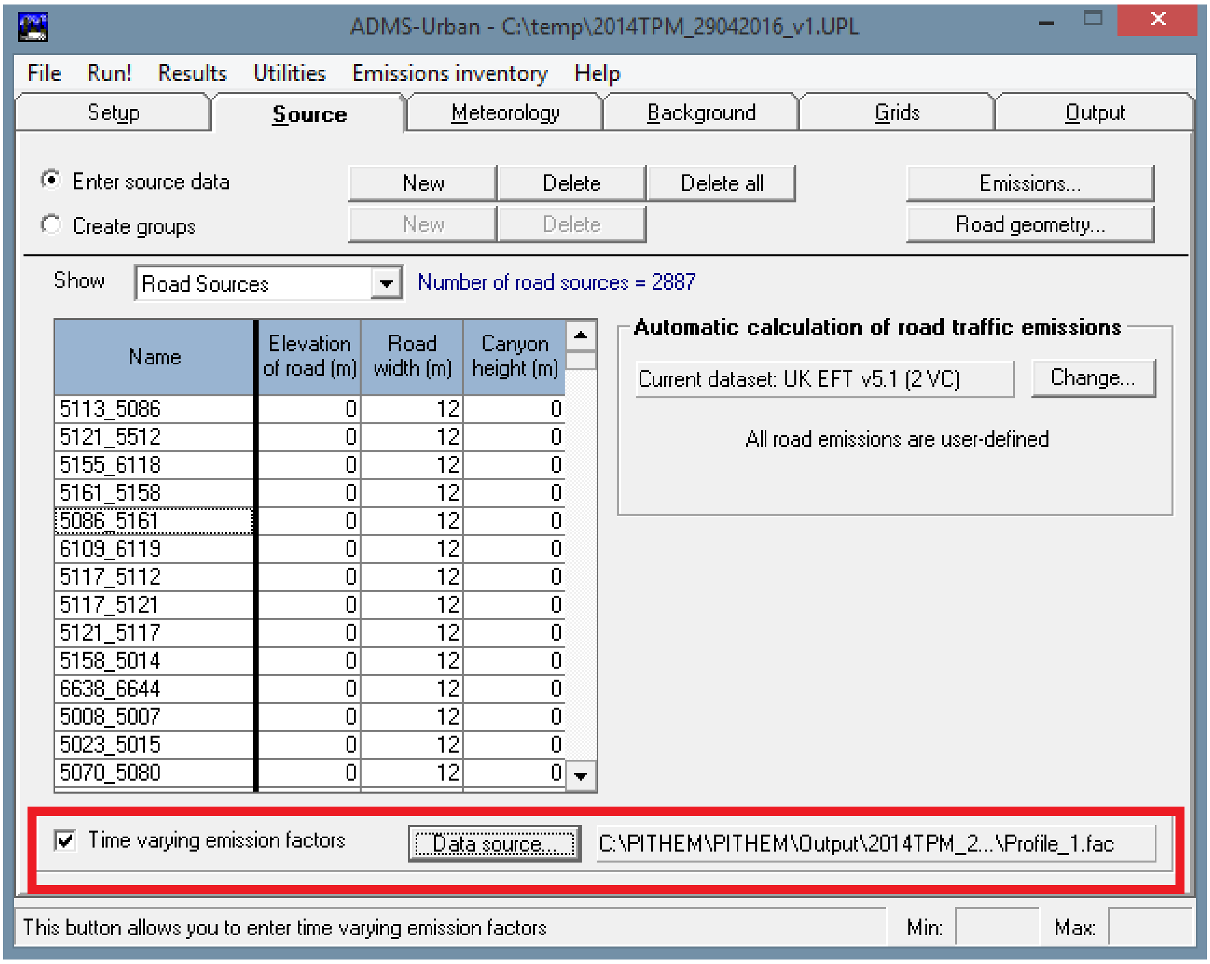


Figure 4-22: Defining the time-varying emission factors

c) The third step is to enter the meteorological data associated with the study area by clicking on the 'Meteorology' tab and setting the 'Latitude' for Newcastle to 55.0330 and 'surface roughness' to 1.5 (i.e. large surface areas). In the 'Met. Data' section, the 'from file' button is clicked to allow the manual entering of meteorological data. The 'Meteorology' tab is shown in Figure 4-23. This is the data for Newcastle for the year 2014 which was acquired from the Meteorological Office. Access was gained to the Met Office's archived records, where the required meteorological data was extracted and prepared to be read by ADMS-Urban. This data pertains to Woolsington Station as detailed below:

Name: NEWCASTLE/WOOLSINGTON

ID no.: 18931

Geographic area: TYNE & WEAR

Latitude: 55.033 (WGS 84 value: 55.0331); and Longitude: -1.68393

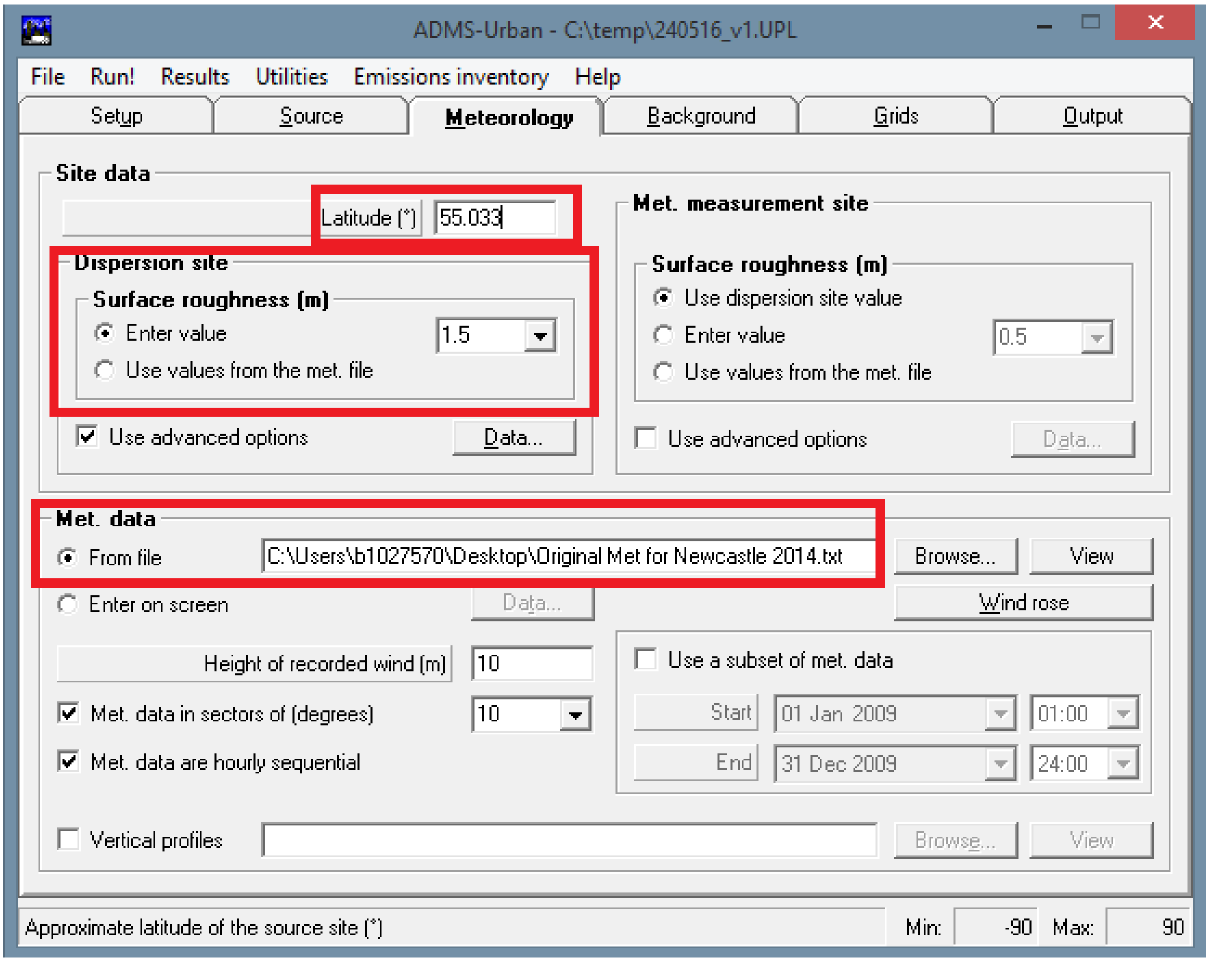


Figure 4-23: 'Meteorology' tab

d) Assigning receptors wherever ADMS-Urban is asked to calculate the pollutants concentrations or when defining grid spacing points can be performed through the "Grids" tab. Receptors names and their co-ordinates are assigned in this step, as shown in Figure 4-24.

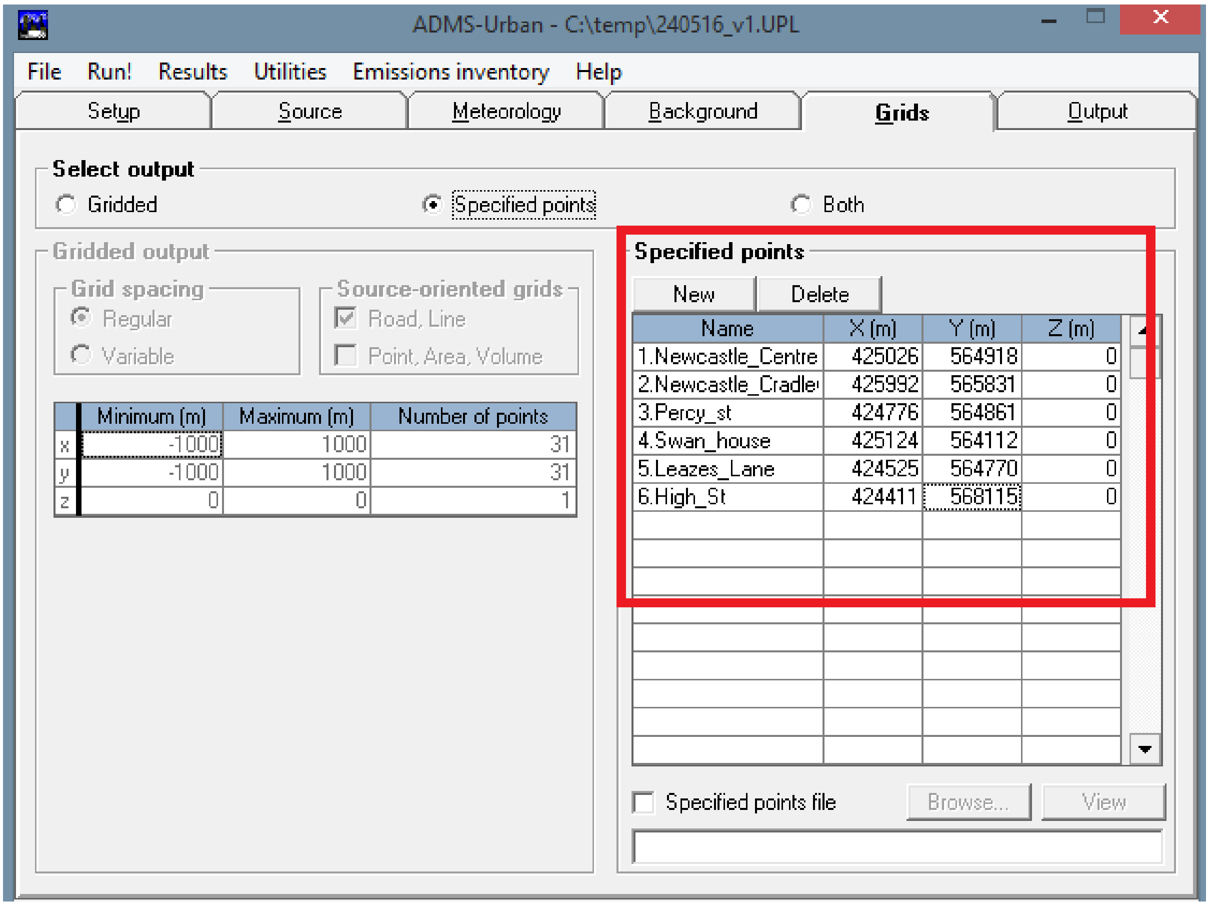


Figure 4-24: Defining receptor names and their co-ordinates

e) By clicking on the 'Output' tab, the calculations in relation to short- and long-term pollutants are selected, as revealed in Figure 4-25.

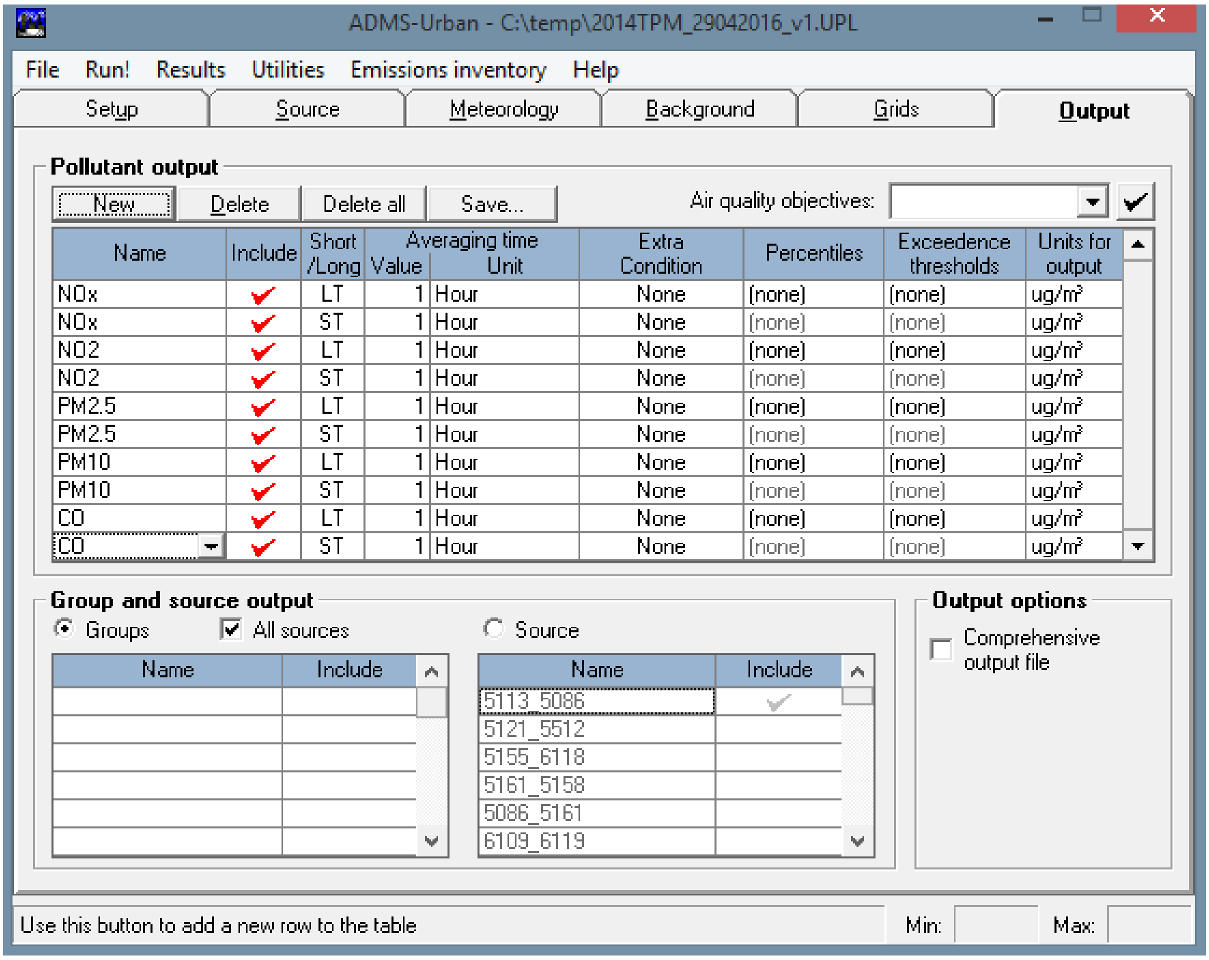


Figure 4-25: Selection of pollutant calculations

The output files will contain pollutant concentrations at the selected receptors and will be presented in notepad format which can be read in Excel for analysis.

4.8 Dose-Response Functions to Link Changes in Air Quality and Health

In order to estimate the impact of air quality change on hospital admissions and death, the exposure to changes in concentrations of the pollutants (between the Baseline and 2030 scenarios) was quantified. Coefficients for short- and long-term exposure to several pollutants in the UK were developed, as described in chapter 3 and presented once again in Tables 4-7 and 4-8. For example, an increase in long-term exposure of 10 µg/m^3 of PM_{10} (annual mean) led to an increase of 7% in premature deaths (Carey *et al.*, 2013) and 8% in respiratory hospital admissions (Lee and Sarran, 2015). COMEAP (1998) has developed coefficients for short-term exposure to PM_{10}, in which the excess risk per 10 µg/m^3 (daily mean) exposure might cause a 0.75% increase in premature deaths and a 0.80% increase in respiratory emergency hospital admissions.

Table 4-7: Dose-response coefficients associated with an increase in daily mean pollutant levels

Pollutant		Short-term exposure to 10µg/m^3	Reference
PM_{10}	Death	0.75% (24 hour mean)	COMEAP (1998)
	RHA	0.80% (24 hour mean)	COMEAP (1998)
$PM_{2.5}$	Death	1.23% (95% CI: 0.45, 2.01%)	WHO (2013a)
	RHA	1.90% (95% CI: -0.18%, 4.02%)	WHO (2013a)
NO_2	Death	0.71% (95% CI 0.43%, 1%)	Mills *et al.* (2015)
	RHA	0.50%	COMEAP (1998)

Table 4-8: Dose-response coefficient associated with an increase in annual mean pollutant levels

Pollutant		Long-term exposure to 10µg/m^3	Reference
PM_{10}	Death	7% (95% CI: -1%, 16%)	Carey *et al.* (2013)
	RHA	8% (-11%, 27%)	Lee and Sarran (2015)
$PM_{2.5}$	Death	6% (2%, 11%)	COMEAP (2009)
	RHA	32% (5%, 60%)	Lee and Sarran (2015)
NO_2	Death	2.5% (1%, 4%)	COMEAP (2015a)
	RHA	17% (10.4%, 23.6%)	Lee and Sarran (2015)

4.9 Comments on the Methodology

The research seeks to model traffic flows, emission rates and dispersion based on the 2014 Baseline scenarios to quantify the resulting ill health or disease burden of the population and then to assess the effect of the uptake of electric vehicles as six scenarios. DEFRA (2010) suggested the use of the English and Welsh Regional Traffic Growth Speed Forecasts (RTFs) and Trip End Model Presentation Program (TEMPro) to model changes in traffic flows into the future. In the Transport Planning Model (TPM), the final growth factor is represented as:

Final Growth Factor = RTF factor × (TEMPro Local/TEMPro regional)

In considering findings of this research, it is important the reader remains aware of the fact that the DEFRA methodology was used consistent with that used by Goodman *et al.* (2014) and commissioned by local authorities of Newcastle and Gateshead. However, the research carried out in this thesis acknowledges that there are inherent limitations within the DEFRA methodology and its underlying assumptions. Some of the shortcomings of DEFRA methodology can be attributed to criticisms of the TPM, ADMS-Urban and Gaussian plume models. These are discussed in this section.

The assumption here is that TEMPro will provide local data that complements or adjusts the RTF's national and regional data, which will then give reliable future local traffic projections. DEFRA's suggestion to researchers, who wish to use their methodology, is to avoid using TEMPro or the RTF data on their own to provide growth projections when either of them is absent to complement the other's results. Although Newcastle and Gateshead are covered in these national models, RTF data does not provide UK wide coverage, it excludes traffic flows in Scotland, Wales and Northern Ireland the North East proximity to Scotland may have some impact. Similarly, TEMPro data does not cover Northern Ireland. The representativeness and/or reliability of the DEFRA methodology when calculating for future traffic flows in any of the uncovered areas is questionable.

In addition, there are a number of concerns with the use of this methodology in academic research. The RTF data records growth in national road traffic for cars, LGVs, and HGVs (but excluding buses) in England and Wales and also calculates traffic flow data for air quality management but does not focus on local growth levels. One problem here is associated with extrapolating from national or regional data to represent a local event or data. How representative is it? Furthermore, data from the trip end model presentation program (TEMPro), does not include information from LGVs and HGVs. This leaves a massive gap in the analysis which may skew the results. According to Clark (2016), TEMPro growth factors are prone to consistent over-forecasting of up to 20% (or more) compared to observed growth because the model is built to assume constant and cumulative growth. The truth, however, is that observed growth is rather flat (Clark, 2016). A potential reason for this is that the capacity of urban roads is limited and many towns and cities are managing demand for private vehicle use through parking charges and encouraging the use of more sustainable modes.

TEMPro's underlying assumption is that an annual increase in population growth or in the level of development within an area correlates with an increase in traffic. But this may not be the case in the light of the effects of lifestyle changes and sustainable travel modes such as walking and cycling as well as using EVs; the introduction of new technology that encourages choices relating to shorter trips or not travelling at all (for example, services such as Deliveroo, Netflix and virtual mobility), or events such as fuel crises or major lifestyle altering situation such as financial crisis. Figure 4-26 clearly demonstrates the effect of such and the assumption and reason to be cautious when relying on model forecasts for policy decisions.

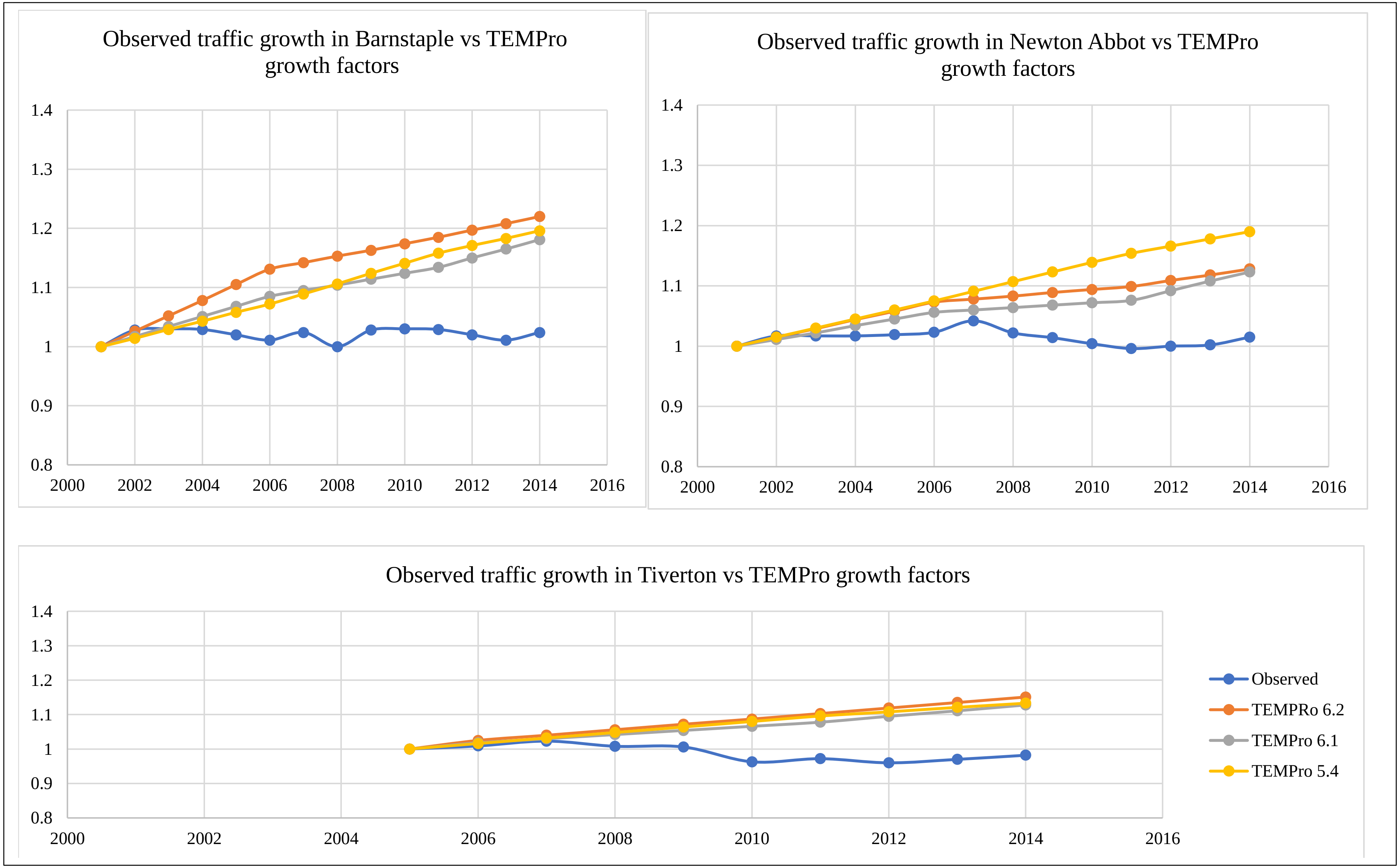


Figure 4-26: Observed traffic growth factors (Barnstaple, Newton Abbot and Tiverton) vs TEMPro, source: Clark (2016)

The figure shows a huge chasm between TEMPro growth factors versus the observed traffic growth in selected UK towns (Barnstaple, Newton Abbot and Tiverton) as reported by Clark (2016). This criticism of TEMPro's over-estimation has been confirmed by Hertfordshire Transport Planning (2017) and Marsden *et al.* (2018) who stated that over the last 25 years people have been travelling less, making 16% and 14% fewer trips than in 1996 and 2002 respectively and 10% fewer miles travelled per person than in 2002. Therefore, the results yielded by TEMPro should be used with caution and consideration given as to how appropriate it is to draw conclusions at local, regional or national scale. It would seem reasonable to deduce that any sort of framework (such as the DEFRA methodology) that incorporates TEMPro will carry with it the assumptions, errors and inaccuracies associated with the model and these will propagate through to the output of its application.

A prevailing cause of error within TEMPro growth forecasts is their inability to incorporate time-sensitive information to act as a time-series model able to use information from preceding years to forecast future traffic patterns. Clark (2016) suggests that TEMPro with a time-series model characteristic, would allow earlier trends to contribute towards the future growth factors. In other words, it will eliminate the potential for error propagation by correcting for the local trends. For TEMPro or RTF to be legitimate or reliable tools for future traffic levels estimations, they would need to be equipped with a mechanism which allows them to use trends from preceding years to inform predictions.

Moreover, the forecast models are riddled with underlying assumptions that do not take into consideration human behaviour, travel choices, key drivers of travel demand and changes in government policies, in addition to a multitude of variables which may affect changes in traffic flows. Therefore, TEMPro forecasts should be considered as part of a logical exercise considering other evidence to assist predictions of future traffic growths. In the quest for the efficient management of future road networks, all diligent planners understand the difficulty of accurately forecasting future traffic growth, as patterns are affected by a multitude of variables including an unstable economic climate, government policies etc. Nonetheless, this uncertainty should be used as an incentive for regular updates and a comprehensive assessment of the best available industry predictive models. On a final note, it is imperative to remember that, according to Clark (2016):

"In reality, no one can predict the future of travel demand accurately, but we must aspire to ensure that all likely scenarios are considered, and critique any values arising that seem to be counterintuitive given the historical and recent data available."

4.10 Summary

This chapter has presented the methodology used for this research. The main steps are building the traffic model for the 2014 Baseline and 2030 scenarios, modelling vehicular emissions and their dispersion, and quantifying the disease burden resulting from changes in pollution concentrations between the Baseline and 2030 scenarios.

The reasons for selecting the study area were highlighted and followed by details of the sources of traffic and pollution concentrations data which are required to validate the Baseline model. Legal issues that prevented death and hospitalisation figures corresponding to patient residency from being obtained, have been clarified. Obtaining patient addresses is not possible because this sort of information is considered individually identifiable and sensitive. It has been suggested that patients are exposed to the same pollution that is occurring at the GP sites where they are registered, assuming that people are usually registered at the GP practice closest to where they live.

The procedure for updating the traffic model for the 2014 Baseline from the 2010 traffic model was described. The creating of the future traffic model for BAU and other 2030 scenarios was explained. The calculation of emissions rates for each scenario utilising the PITHEM has been clearly demonstrated. The dispersion of those emissions as affected by meteorological factors in terms of hourly data via an air quality model ADMS-Urban, was described in detail. Finally, the investigation of the disease burden because of changes in pollution concentrations through dose-response coefficients was revealed.

CHAPTER 5

5. Development of the Baseline Traffic Model for 2010 and 2014

5.1 Introduction

In this chapter, the development of the updating of the 2010 Transport Planning Model (TPM) to the 2014 Baseline are presented. The TPM is updated to the 2014 Baseline and then calibrated and validated against TADU real-world traffic data. Subsequently, vehicular emissions associated with the Baseline were calculated and their dispersions estimated and validated against pollution levels recorded at the seven monitoring sites over the study area, taking into consideration variations in background emission concentrations. This creates a framework model able to simulate vehicular emissions and to estimate pollution concentrations at any selected receptor in Newcastle and Gateshead.

5.2 Observed Traffic Data

The Baseline year selected for this research was 2014, when the study commenced. The reason for this is that it was the most recent year for which published real-world traffic information was available. The observed flows of vehicles for 2014 were obtained from the Tyne and Wear Road Traffic and Accident Data Unit (TADU) at Gateshead City Council, the Department for Transport (DfT) and Urban Traffic Management Control (UTMC) at Newcastle City Council. TADU (2014) has published hourly traffic profiles for 2014 for 109 links in Gateshead, and 58 sites in Newcastle. It is worth mentioning that sites with a number starting with 9700 to 9999 are cycle counters only and are not included in the calibration and validation analyses. Meanwhile, the DfT has produced Annual Average Daily Traffic (AADT) data, which can be found via the count points on its website. There are 60 count points in Gateshead and 78 in Newcastle. Furthermore, Automatic Number Plate Recognition (ANPR) records were received from UTMC that comprised 63 units of ANPRs distributed throughout Tyne and Wear.

Observed traffic data are insufficient to cover the full transport network relating to the study area. Hence, transport modelling is crucial to undertake the research, since models generally aim to represent reality in a simple form in order to investigate the consequences of existing, changes to or the introduction of new policies or strategies in a short time and at low risk (O'Flaherty and Bell, 1997, p. 103).

5.3 Transport Planning Model (TPM)

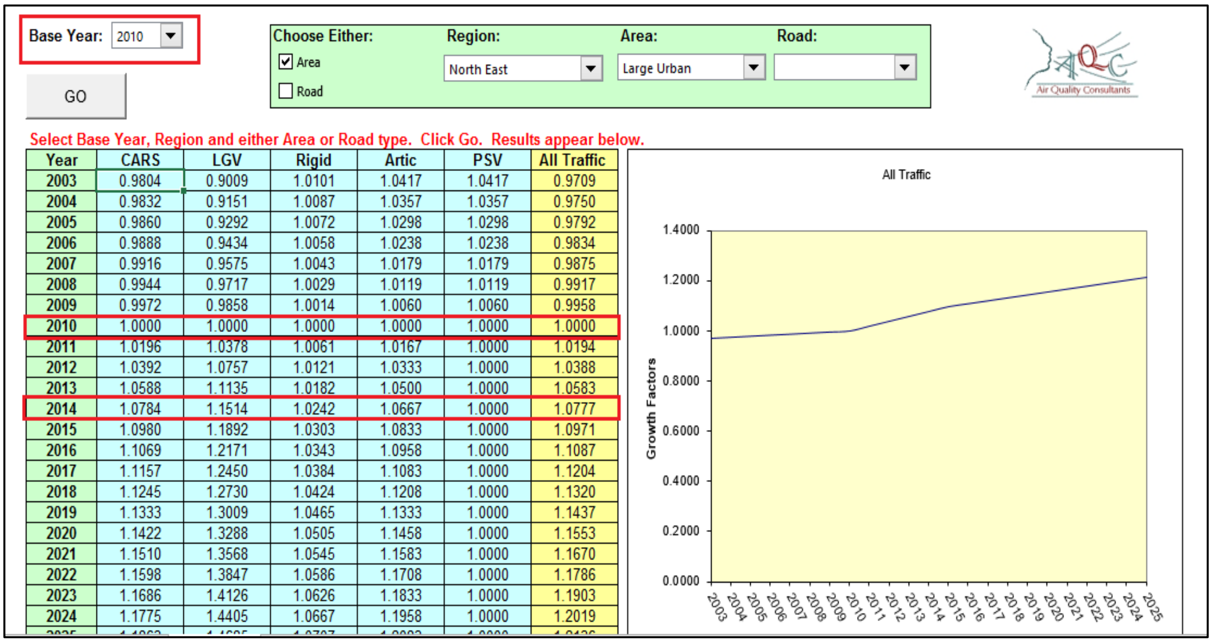
The TPM is a traffic model on a large scale set up for 2010 traffic flows that covers Newcastle upon Tyne and Gateshead and extends to nearby towns. It was previously used by Newcastle University, which was commissioned by Newcastle and Gateshead City Councils in a project funded by the DEFRA in 2014 in order to assess the feasibility of low-emission zones by 2020 in Newcastle upon Tyne and Gateshead. This model is considered to be fit for purpose by the DfT and Highways Agency (Goodman *et al.*, 2014, p. 32). In addition, the model is extensive and distributed in numerous folders and files, and traffic flow is estimated from daily averages (Monday to Friday) for 2,887 links. Details of each link are provided, such as length, hourly flow and approximate speed corresponding to cars, LGVs and HGVs. Similarly, the networks and their data on bus journey activities were obtained in a separate set, which also contains the same traffic parameters for 10,608 links for the hourly bus flow patterns besides speed and link lengths.

For the present research, the TPM was assumed to be appropriate as a ready-to-use source of data on traffic characteristics and the required traffic information over the study area.

5.4 Traffic Growth Forecast

As mentioned in the previous section, the TPM provides information on traffic activities pertaining to 2010, whilst the Baseline for the research in this thesis was set to 2014. Thus, a series of growth factor generators were required in order to update the TPM to reflect Baseline conditions. Traffic growth factors can be generated by utilising the 'English Regional Traffic Growth and Speed Forecasts' (RTFs) and Trip End Model Presentation Program (TEMPro), and those factors can subsequently be applied to the TPM as suggested by DEFRA (DEFRA, 2010). Worthy of note is that the TPM model provides separate flows for cars, LGVs, HGVs and buses. The need for sensitivity analysis regarding vehicle compositions specifically was not performed because previous research using the same network when updating the TPM from 2010 to 2021 in Newcastle and Gateshead, Goodman *et al.* (2014) showed insensitivity of the model to vehicle compositions over anticipated changes; this is mainly because the traffic flow of each vehicle type was specified separately as inputs to the emissions model; for example, actual bus flows and services in the network, which were defined separately in a traffic model.

The RTFs produce a national view of forecasts relating to the growth in road traffic, congestion and vehicular emissions across England and Wales, derived from the latest results from the Department for Transport's National Transport Model (DfT, 2015, p. 8). Air Quality Consultants Ltd has developed the 'Automated Traffic Growth Calculator' (RTFs' calculator) to support the scaling of traffic flow data for Local Air Quality Management purposes (DEFRA, 2010). The RTFs Calculator allows the user to estimate growth factors in future traffic flows by inputting either the intended region, such as the North East, and its area type (such as 'Large Urban'), or the road type, such as a motorway. The outputs will consist of growth factors for each vehicle class (for example cars, LGVs, rigid HGVs). In order to update the TPM to provide the 2014 (Baseline) conditions, the RTFs Calculator was utilised to anticipate regional traffic projections by setting the attributes of the study area which are 'Base Year', 'Region' and 'Area' to '2010', 'North East' and 'Large Urban', as illustrated in Figure 5-1.



Year	CARS	LGV	Rigid	Artic	PSV	All Traffic
2003	0.9804	0.9009	1.0101	1.0417	1.0417	0.9709
2004	0.9832	0.9151	1.0087	1.0357	1.0357	0.9750
2005	0.9860	0.9292	1.0072	1.0298	1.0298	0.9792
2006	0.9888	0.9434	1.0058	1.0238	1.0238	0.9834
2007	0.9916	0.9575	1.0043	1.0179	1.0179	0.9875
2008	0.9944	0.9717	1.0029	1.0119	1.0119	0.9917
2009	0.9972	0.9858	1.0014	1.0060	1.0060	0.9958
2010	1.0000	1.0000	1.0000	1.0000	1.0000	1.0000
2011	1.0196	1.0378	1.0061	1.0167	1.0000	1.0194
2012	1.0392	1.0757	1.0121	1.0333	1.0000	1.0388
2013	1.0588	1.1135	1.0182	1.0500	1.0000	1.0583
2014	1.0784	1.1514	1.0242	1.0667	1.0000	1.0777
2015	1.0980	1.1892	1.0303	1.0833	1.0000	1.0971
2016	1.1069	1.2171	1.0343	1.0958	1.0000	1.1087
2017	1.1157	1.2450	1.0384	1.1083	1.0000	1.1204
2018	1.1245	1.2730	1.0424	1.1208	1.0000	1.1320
2019	1.1333	1.3009	1.0465	1.1333	1.0000	1.1437
2020	1.1422	1.3288	1.0505	1.1458	1.0000	1.1553
2021	1.1510	1.3568	1.0545	1.1583	1.0000	1.1670
2022	1.1598	1.3847	1.0586	1.1708	1.0000	1.1786
2023	1.1686	1.4126	1.0626	1.1833	1.0000	1.1903
2024	1.1775	1.4405	1.0667	1.1958	1.0000	1.2019

Figure 5-1: Illustration of the RTFs calculator

RTF outputs indicate that, by setting 2010 which is the TPM year as the base year, traffic flow was expected to grow by 2014 (Baseline) in the North East by 1.0784 for cars, 1.1514 for LGVs and 1.04 for HGV. Nevertheless, the RTFs Calculator offers national and regional data that may not focus on local levels, as traffic volumes are liable to grow at different rates in different localities. Hence, adjusting the RTF's factors for local growth by utilising TEMPro will give a more appropriate reflection of traffic growth factors (DEFRA, 2010).

TEMPro is a presentation tool developed by the DfT to provide projections related to traffic growth over time. It is for use at local authority district level, where it might be required for the local adjustment of growth factors related to traffic. The tool is consistent with the

National Trip End Model (NTEM) dataset that provides a set of predictions pertaining to traffic growth in cars and buses, with the associated planning of data projections at all geographical levels down to local authority districts. More details related to a description of the derivation of TEMPro from NTEM are illustrated in the Design Manual for Roads and Bridges (DMRB, 1997).

In urban areas where congestion occurs it is strongly recommended for the modeller to break down the day into separate model intervals to cover AM, IP, PM and off peak (OP) periods (DMRB, 1996). Traffic flow periods in the received TPM were broken down into periods of three hours from 07:00 to 09:59 for AM, six hours from 10:00 to 15:59 and 19:00 to 06:59 for IP and three hours from 16:00 to 18:59 for PM. These periods were found to be suitable in representing variations in traffic activity in the study area, given that they cover the morning and evening periods when most traffic activity occurs.

Projections regarding traffic growth for vehicles with regard to AM, IP and PM time periods in the study area were extracted from TEMPro. Table 5-1 shows the traffic growth factors in the study area. Unfortunately, up-to-date growth factors for LGVs and HGVs are not provided by TEMPro. Assuming constant increase in traffic growth in all vehicle types over the whole day across all links is unrealistic and inappropriate; however, this approach is consistent with government guidance by investigating the extremes of vehicular emissions quantities, conservative estimates are derived such that the anticipated outcome will certainly be lower. Given that networks have a finite capacity for traffic flows if traffic continues to increase it is likely that the duration of the peaks will increase. This means that the proportion of time networks are at AM and PM flows relative to IP will increase. If evidence that this is occurring emerges, model guidelines will need to be changed accordingly.

Table 5-1: Car and bus growth factors from 2010 to 2014

TEMPro Factors	Car			Bus		
	AM	IP	PM	AM	IP	PM
Newcastle (Local)	1.0374	1.0376	1.03635	1.0	1.0	1.0
North East (Regional)	1.0291	1.0412	1.0309	1.0	1.0	1.0

Traffic growth factors acquired from RTFs should be adjusted associated to factors obtained by TEMPro using the following equation (DEFRA, 2010) :

$$\text{Final growth factor} = \text{RTF} \times \frac{\text{TEMPro}_{\text{Local}}}{\text{TEMPro}_{\text{Regional}}}$$

where

RTF: Traffic growth factor obtained from the RTF spreadsheet

$\text{TEMPro}_{\text{Local}}$: Local traffic growth factors, such as for Newcastle upon Tyne

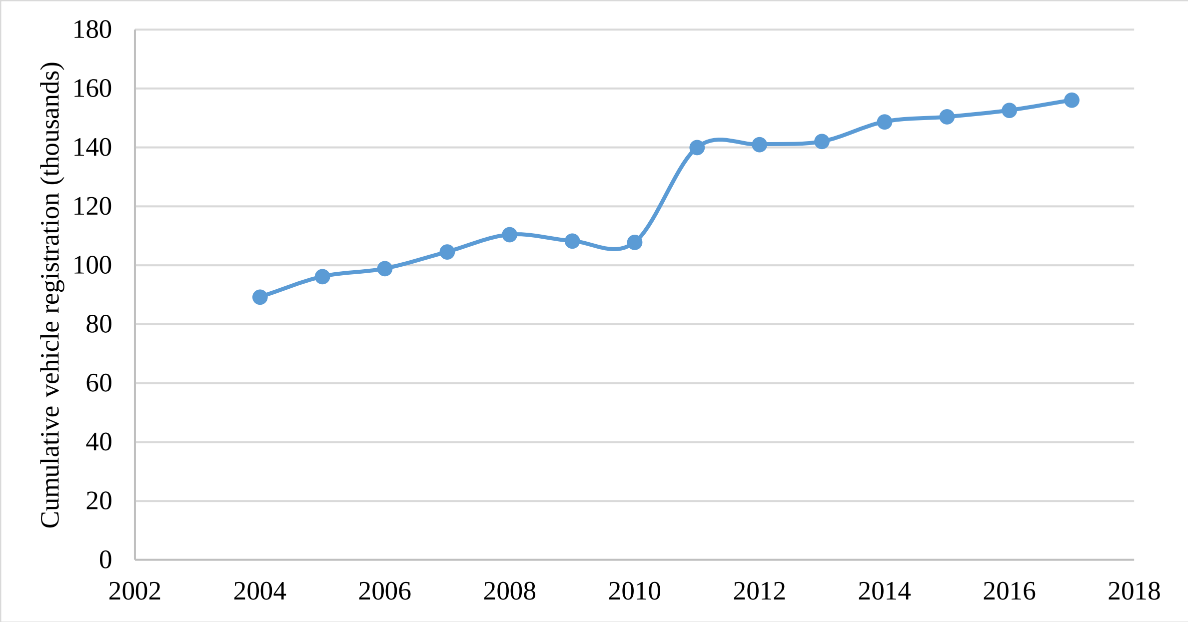
$\text{TEMPro}_{\text{Regional}}$: Traffic growth factors; here, for the North East

The final growth factors for traffic in the pilot area are presented in Table 5-2 after combining the RTFs with TEMPro traffic flow growth factors.

Table 5-2: Final growth factors of traffic flow for cars, LGVs, HGVs and buses

Combined growth factors (2010-2014)	Car	LGV	HGV	Bus
Morning peak period, AM (07:00-09:59)	1.08715	1.1514	1.04	1.0
Inter peak period, IP (10:00-15:59)	1.074723	1.1514	1.04	1.0
Evening peak period, PM (16:00-18:59)	1.084101	1.1514	1.04	1.0

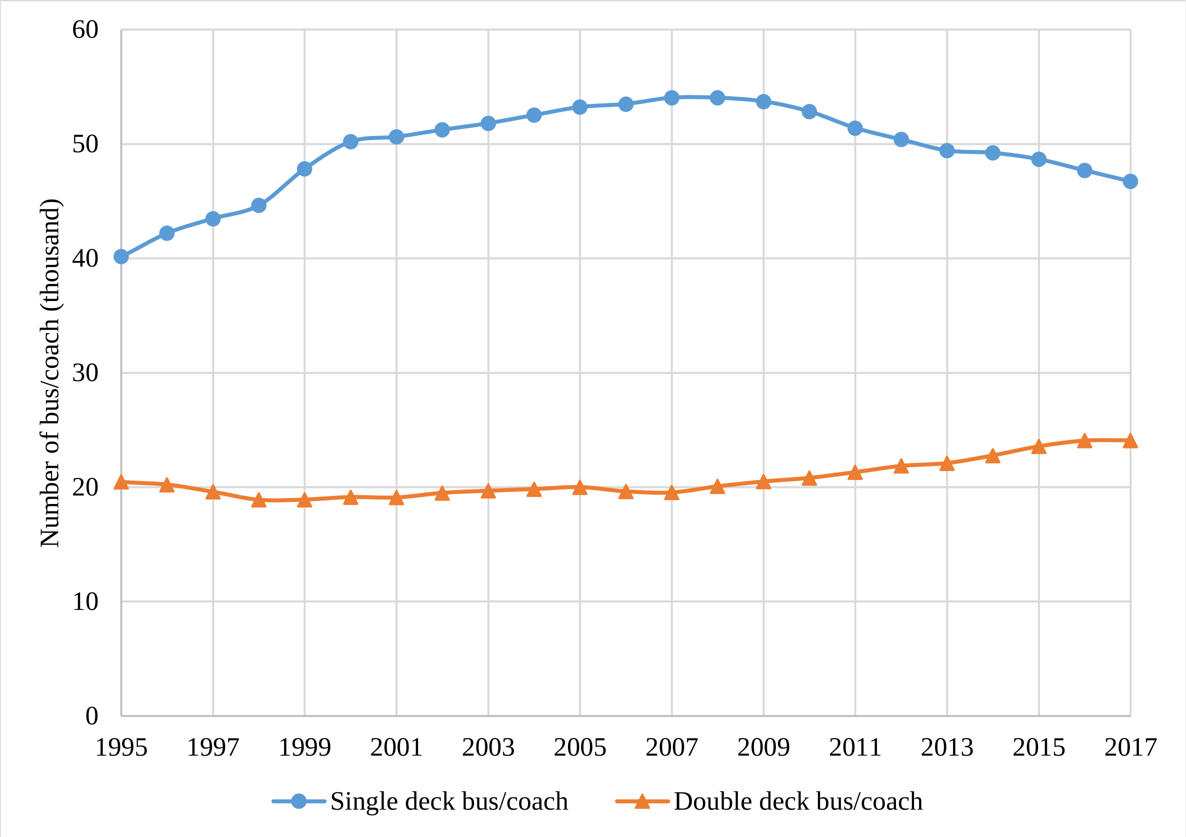
The flow of LGVs is expected to witness the largest growth of 15% by 2014 (the Baseline year) compared to 2010 (the TPM year). This could be attributed to the vast increase in the cumulative registration of LGVs in the North East. Official statistics published by the DfT in Table VEH0404, demonstrates that LGVs registration in the region increased by 38% from approximately 108,000 in 2010 to 149,000 in 2014, with a rapid jump between 2010 and 2011, as seen in Figure 5-2. Part of the increase in LGVs registrations took place in Darlington during 2011 and the jump corresponded with the construction and operation of the Darlington campus of Teesside University. However, generally there was a systematic increase in the use of vans across the UK as freight distribution practices changed (including on-line shopping). Therefore, LGVs accounting for the largest factor among other vehicle classes in relation to growth in flow.



Source: DfT (2018j)

Figure 5-2: Cumulative registration of LGVs in the North East

Conversely, the flow profile of buses may not need to grow over time, even though the population has increased, given that spare capacity in buses could cope with increase demand for bus journeys. Bus operators also may adopt more double-decker instead of single-decker buses to cater for the increase in demand. The trend for registrations of double-decker buses is increasing, whilst single-decker registrations are decreasing in the UK, Figure 5-3 shows the cumulative registrations of buses and coaches over the past ten years, where registrations of double-decker buses increased by 20% whilst single-deckers decreased by 13.5% (DfT, 2018k). This explains why bus journeys in 2014 are expected to witness only slight traffic changes compared to 2010.



Source: DfT (2018k)

Figure 5-3: Cumulative registration of buses and coaches in Great Britain

In relation to differences in vehicular emissions released from single- and double-decker buses, the Emission Factor Toolkit v9 (EFT9) was used to draw comparisons. In EFT v9, single- and double-decker buses were available only as a hybrid bus to perform the comparison. Both types are hybrid Euro VI. They were entered in the EFT v9 so as to model their emissions. The results indicated a slight difference in particulate matters and NO_x emissions between single- and double-decker buses. It appears that this slight difference would not significantly affect the quantity of the vehicular emissions released. In general, emissions from both types of buses were higher when compared with private cars. This finding agrees with that of a study conducted in Hong Kong which established that the emission released from double- and single-decker are 9 and 7.5 times higher that emissions released from a passenger car (Cen *et al.*, 2016). The important issue is that emissions per km travelled by passengers depending on loadings is significantly less than a private car.

Worthy to note is that passengers on the upper deck of a bus are less exposed to air pollution compared to those travelling on the lower deck, Chan et al. (2002) simultaneously measured PM_{10} concentrations on the upper and lower-decks found that PM_{10} concentrations on the upper deck were 16% less in air-conditioned mode and 25% less in non-air-conditioned mode.

5.5 Baseline for 2014

Growth factors of future traffic for a certain year's flow based on available traffic data from the TPM were extracted from RTFs and TEMPro and combined to estimate the traffic projection for the future year (DEFRA, 2010). Therefore, the combined growth factors of traffic flow for cars, LGVs, HGVs and buses were applied to TPM 2010 to produce the Baseline 2014. For example, modelled traffic flows for the AM, IP and PM periods are presented in Figure 5-4 for both TPM 2010 and Baseline 2014 at link numbers 1708 and 2700 in the Baseline model. These links correspond with the Great North Road, north of Forsyth Road (E424690, N566610), where volume flows are combined for both northbound and southbound directions.

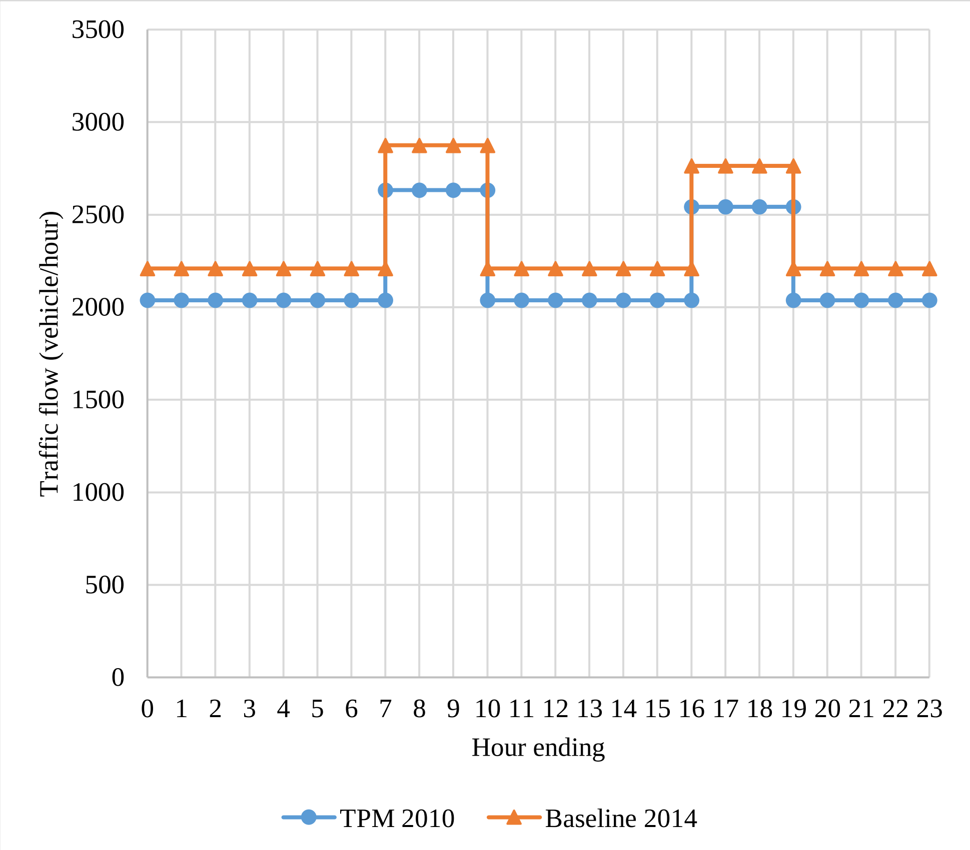
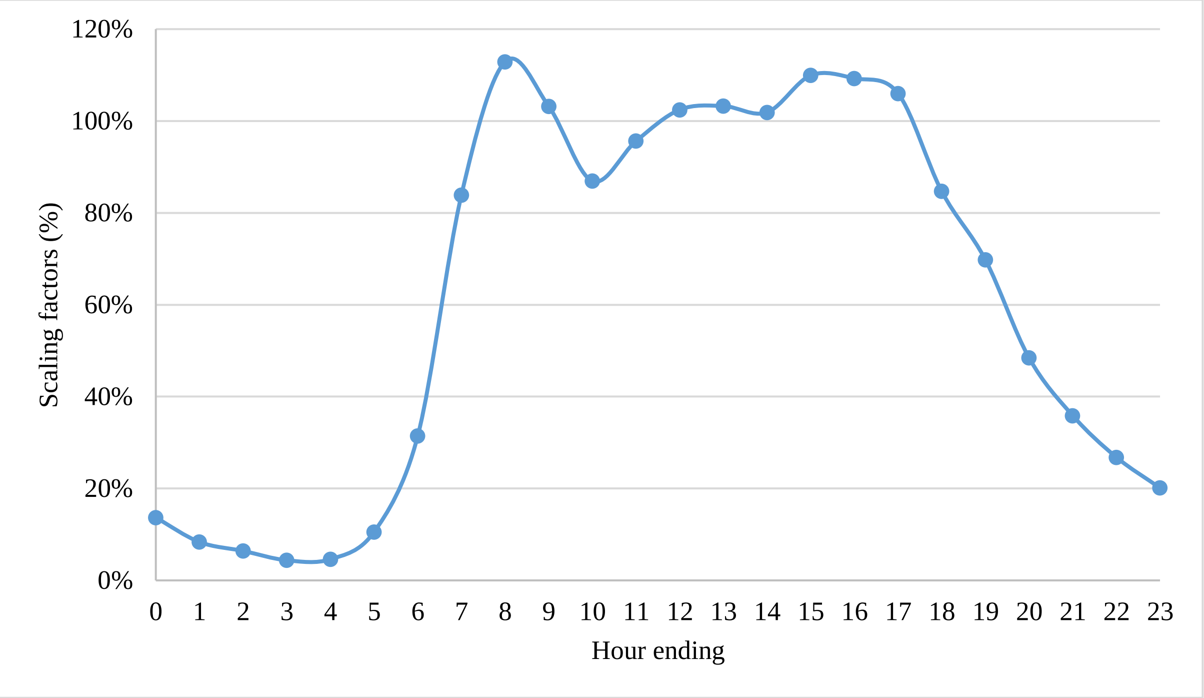


Figure 5-4: Updating TPM to Baseline; AM, IP and PM periods for links 1708 and 2700

Traffic flows for the AM, IP and PM periods in relation to the Baseline have been set in the previous section for weekdays. However, the off-peak (OP) period of weekdays and all periods in relation to weekends need to be determined in order to complete the traffic pattern picture for the seven days of the week. Traffic activities related to the overnight period hours (the OP period), were quantified by reducing the inter-peak (IP) traffic activities by 24%, whilst the weekend diurnal profile for AM, IP, PM and OP was identified by scaling down the weekdays periods to 77% (Goodman *et al.*, 2014, p. 35). Hence, the AM, IP, PM and OP periods are now complete for the week.

Within the same period, such as the six hours of the IP peak (1000-16:00), traffic flow during the first hour (i.e. 10:00-11:00) might differ from that in the second hour (i.e. 11:00-12:00), while at the moment they have the same value, as shown in the previous figure. Owing to the variation in traffic flow between specific hours, associated vehicular emissions may also vary. Therefore, it is important to assign traffic flows for each hour of the day. For the same study area, Goodman *et al.* (2014, p. 155), scaled the AM, IP PM and OP periods to produce hourly traffic flows in the TPM prior to vehicular emissions modelling using the time-varying factors presented in Figure 5-5.



Source: Goodman *et al.* (2014, p. 155)

Figure 5-5: Time-varying factors to scale AM, IP, PM and OP periods

Traffic flow during the AM, IP, PM and OP intervals was scaled by means of the time-varying factors in order to produce the diurnal profile for each hour of the day. Thus, the Baseline model is ready for the calibration and validation process so as to ensure that the Baseline model performs well in reflecting traffic parameters for the year 2014.

5.5.1 Calibration and Validation of the Baseline

The calibration of a model involves any adjustments to it carried out to reproduce observed behaviour; whilst model validation is a comparison between modelled data against real-world data which has not been used previously in calibration processes (O'Flaherty and Bell, 1997,

p. 105). In general, the process of calibrating and validating a traffic model relies on the availability of traffic counts and/or journey times. The former data might be acquired by automatic or manual means and the latter can be collected by moving vehicle observer surveys, or commercial sources such as a GPS-tracked car, in addition to camera recordings from ANPR (DfT, 2014b). In this research, calibration and validation operations were performed based on traffic counts pertaining to the study area in 2014, which were extracted from TADU publications (TADU, 2014). The TADU data were divided into a part dedicated to the calibration process, whilst another part was used to validate the Baseline.

Guidelines for validation criteria are proposed by way of Technical Analysis Guidance (TAG), which are similar to the guidelines suggested by the Design Manual for Roads and Bridges (DMRB, 1996; DfT, 2014b). These guidelines were followed to calibrate and validate the Baseline model. The following subsections demonstrate the validation criteria used to calibrate and validate the Baseline.

According to the DMRB (1996) and DfT (2014b) guidelines, there are two frequently used alternative analytical methods that can be applied to the comparison of modelled and observed values, in this case those values pertaining to 2014 in Newcastle and Gateshead.

5.5.1.1 First Validation Method: GEH Index

The first method is to calculate the Geoffrey Edward Havers (GEH) index, which is a standard measure of the goodness of fit between observed and modelled traffic data (DMRB, 1996; DfT, 2014b). The GEH index incorporates both relative and absolute differences between modelled and observed traffic counts and is defined as follows:

$$\mathrm{GEH} = \sqrt{\frac{2 \times (\mathrm{M} - \mathrm{C})^2}{\mathrm{M} + \mathrm{C}}}$$

where M is the modelled traffic flow volume and C is the observed volume.

If the GEH index is less than five for 85% of the individual links, then there is a good fit and the modelled traffic is considered to be a good match with real-world traffic counts.

In addition, the individual modelled traffic flow should be within 100 vehicles/hour of the observed traffic for less than 700 vehicles/hour; within 400 vehicles/hour for an observed flow greater than 2700 vehicles/hour and within 15% of the observed flow if this is between

700 and 2700 vehicles/hour (DMRB, 1996; DfT, 2014b). Table 5-3 summarises the DMRB and TAG validation criteria.

Table 5-3: Validation criterion and acceptability guidelines

Criteria		**DMRB Acceptability Guidelines**
1	Individual link flows within 15% of counts for flows from 700-2700 vehicles/hour	> 85% of cases
2	Individual link flows within 100 vehicles/hour of counts for flows less than 700 vehicles/hour	
3	Individual link flows within 400 vehicles/hour of counts for flows more than 2700 vehicles/hour	
4	Differences between modelled flows and counts on screen lines should be less than 5% of the counts	All or nearly all screen lines
5	GEH < 5 for individual link flows	> 85% of cases
	GEH statistic based on modelled flows and counts should be less than 4	All or nearly all screen lines
6	Modelled times along routes should be within 15% of surveyed times (or 1 minute, if higher)	> 85% of routes

Source: DMRB (1996)

5.5.1.2 Second Validation Method: Regression Analysis

The second method for validation of a traffic model is described by DMRB (1996) and DfT (2014b), which suggests performing a correlation analysis between the two sets of observed and modelled data. The slope of the best-fit regression line should be between 0.9 and 1.1 and the correlation coefficient (R) should be above 0.95 for an acceptable goodness of model fit.

To implement the DMRB and TAG criteria, observed traffic flows were extracted from TADU (2014), which publishes yearly traffic flow statistics at several sites across Tyne and Wear. The volume of traffic flow is counted in vehicle/hours in both directions for cars, LGVs, HGVs and buses. For example, at Site 1, the Great North Road north of Forsyth Road (E424690, N566610), the volume of traffic flows is presented as a combination of both northbound and southbound directions (TADU, 2014). This road corresponds with links number 1708 and 2700 in the Baseline model. Thus, the flows in these two links were added together in order to be compared with observed traffic flow recorded for both directions at Site 1.

5.5.1.3 Traffic Flow Calibration of the Baseline

According to TADU, observed data are available for 58 sites in Newcastle and 109 in Gateshead (TADU, 2014). However, Baseline traffic links only match 46 sites in Newcastle and 90 in Gateshead. The 46 sites in Newcastle were used to calibrate the Baseline, whilst the

Gateshead sites were kept aside for the validation of the Baseline model. As O'Flaherty and Bell (1997, p. 105) argued, traffic models should be validated using data independent from that used to calibrate the traffic model. Thus, the data independence was considered spatially based on where the data was collected, either in Newcastle or Gateshead.

Time-varying factors proposed by Goodman *et al.* (2014, p. 155), which are demonstrated in the previous figure, were used to run the first iteration of the calibration process at the available individual sites, where the traffic was counted. Modelled values were plotted against counted values to perform a correlation analysis between both groups of data. The slope magnitudes of best-fit regression lines are within the limits set by DMRB at all periods. The results almost all of the correlation coefficients hit the target of 0.95 set by DMRB. Conversely, the ‘GEH < 5’ term values are far from the 85% target as required in the criteria guidelines, as displayed in Table 5-4. Hence, a further iteration to calibrate the model was required.

Table 5-4: Individual link flow calibrations, first iteration

Time period	**AM**	**IP**	**PM**
Number of links with 'GEH < 5'	12	9	11
% of links with 'GEH < 5'	26%	20%	24%
Slope of the best-fit line	0.9	1.0	0.9
Coefficient of determination (R^2)	0.86	0.87	0.84
Correlation coefficient (R)	0.93	0.93	0.92

After several attempts at calibration, it was found that scaling down the traffic flow associated with the AM, IP and PM periods by factors of 0.94, 0.88 and 0.94 respectively led to better performance from the Baseline model compared to the first iteration results. These factors are close to 1; hence, no significant change would occur. Given that between 2010 and 2014 it is expected that traffic flow would increase by 8% for cars, 15% for LGVs, 4% for HGVs and no growth for buses, this increase in traffic volume would be accommodated by road networks in Newcastle and Gateshead since their councils have a joint commitment to tackling transport problems in the local area to help pursue growth and are committed to investment in complementary small, local schemes and in better traffic management (DfT, 2014a, p. 9). Moreover, the government made a commitment to develop new proposals for improving major roads such as the part of the A1 in the study area, as presented in the Road Investment Strategy, of £15.2 billion between 2015 and 2021 (DfT, 2014a, p. 9). This investment plan will help to cope with the future traffic growth in both boroughs.

The GEH index outcomes have been improved, and they vary from 35% to 37%. At the same time, the regression analysis outcomes indicate values at least of 0.9 in the slope at all peak periods and correlation coefficients (R) that range between 0.92 and 0.93, which almost conforms to the DMRB and TAG requirements. The results of the final iterations are displayed in Table 5-5.

Table 5-5: Individual link flow calibrations, final iteration

Time period	**AM**	**IP**	**PM**
Number of links with 'GEH < 5'	17	16	17
% of links with 'GEH < 5'	37%	35%	37%
Slope of the best-fit line	0.9	0.9	0.9
Coefficient of determination (R^2)	0.86	0.87	0.84
Correlation coefficient (R)	0.93	0.93	0.92

5.5.1.4 Traffic Flow Validation of the Baseline

Traffic data observed by TADU in Gateshead in 2014 were dedicated solely for validation purposes, whereas traffic activities recorded in Newcastle were employed in the calibration process.

In the Baseline, modelled traffic flows in the Gateshead network were scaled down by factors proposed by several iterations at the calibration process, as mentioned in the previous section. Subsequently, these modelled traffic values in Gateshead from the Baseline model were compared to corresponding links related to observed values from TADU (2014). The results of the GEH statistics and regression analysis are summarised in Table 5-6.

Table 5-6: Individual link flow validation results (TADU vs Baseline model)

Time period	**AM**	**IP**	**PM**
Number of links with 'GEH < 5'	31	27	23
% of links with 'GEH < 5'	34%	30%	25%

The values of 'GEH < 5' do not achieve the validation requirements. This might be attributed to the fact that most of the flows related to the Baseline links are higher than those from the observed ones. It is worth mentioning that several inner roads are not represented in the TPM. The total length of the minor roads is equivalent to 553 miles in Newcastle, whilst the total road length is 609.6 miles (DfT, 2018d). Hence, flow volumes for the minor roads are more likely be transferred to major links, given that the aim of the TPM is to model vehicle emissions and not the volume of traffic flow. This denotes high traffic volumes compared to real-world links flows. Furthermore, Villa *et al.* (2014), believe that maximising the term “%GEH<5” values is challenging due to the non-differentiability of the numerator, which is the absolute difference between the modelled and observed traffic volume, and that other parameters should be developed in order to maximise “%GEH”. Feldman (2012) highlighted that, notwithstanding the fact that the GEH index is similar to the Chi-squared test, it is not a true statistical test, but rather an empirical formula. Similarly, the GEH index is strongly biased in favour of highway models, whereas the Baseline model comprises numerous road types, such as trunk and principal roads in addition to B and C roads. Moreover, the MVA Consultancy (2011), was commissioned as part of a team’s to support Transport for South Hampshire with the calibration and validation of a Road Traffic Model. The team validation results indicate that “GEH<5” for individual link flows of 47%, 56% and 45% associated with the AM, IP and PM Peaks (MVA Consultancy, 2011).

The second method for validation is described by DMRB (1996) and DfT (2014b). This method performs a correlation analysis between sets of observed and modelled values. By plotting modelled against observed data, the gradients of best-fit lines are within 0.9 and 1.1, which complies with the DMRB criteria for validation. Coefficients of determination (R^2) were derived to obtain correlation coefficients (R); were specifically, 0.9 for the AM, IP and PM periods. They nearly hit the target of 0.95 set by the DMRB. Furthermore, it can be noticed from Figures 5-6. 5-7 and 5-8 which demonstrate the regression analysis of modelled against observed data that the performance of the model is excellent for traffic lower than 10,000 vehicles/hour. However, as the traffic flow increase further, the performance of the model may worsen.

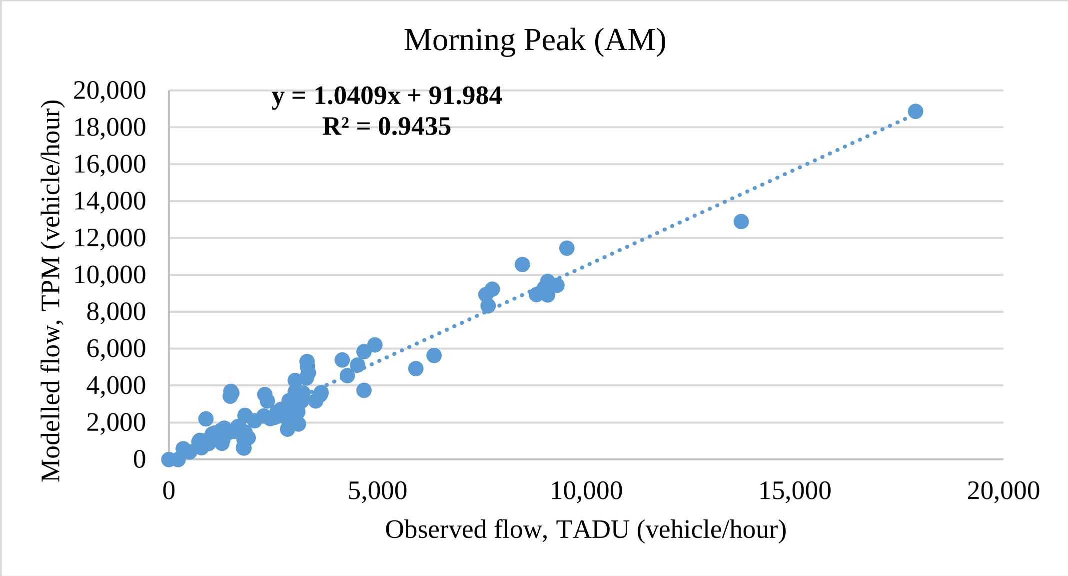


Figure 5-6: Morning peak (AM) validation regression analysis

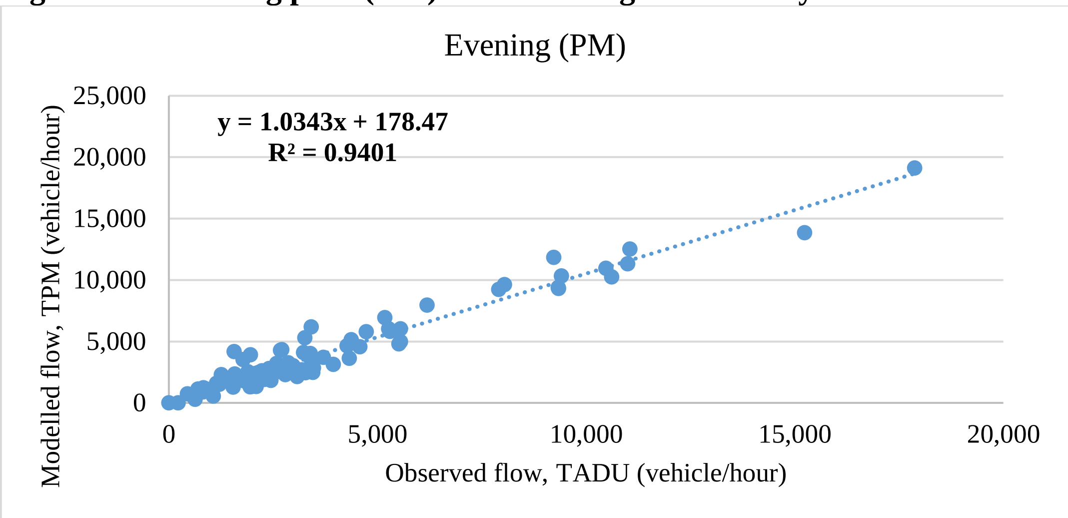


Figure 5-7: Inter peak (IP) validation regression analysis

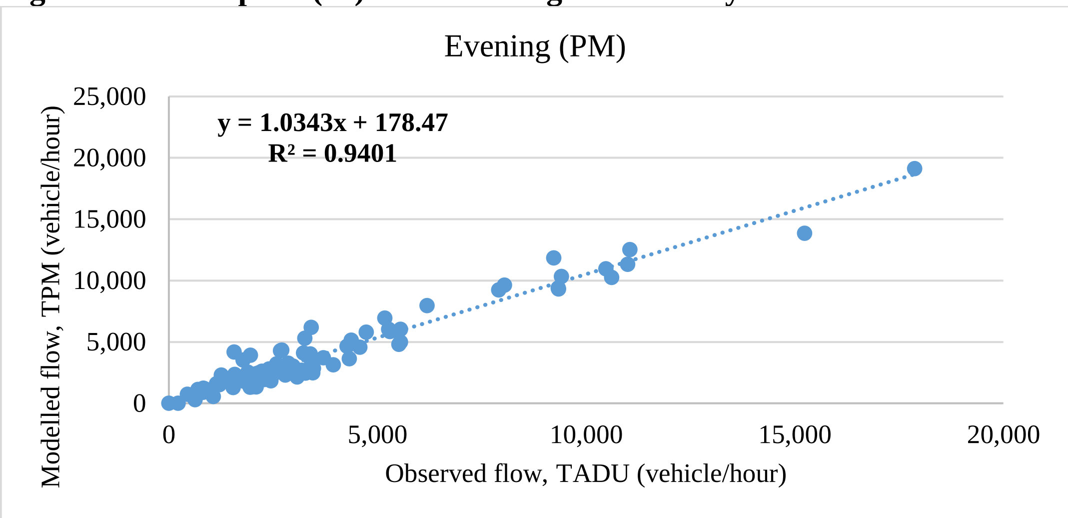


Figure 5-8: Evening peak (PM) validation regression analysis

5.6 Finalised Traffic Modelling of the Baseline

By accomplishing the calibration and validation process, the traffic modelling of the Baseline is finalised. The spatial distribution relating to the vehicle flow of cars, LGVs and HGVs at 8:00 am is illustrated in Figure 5-9 in passenger car units (PCUs), noting that HGV flows were converted to PCU by multiplying by 1.89 (Goodman et al., 2014, pp. 36, 129). It can be seen that the A1 motorway has witnessed heavy traffic demand, at over 4,800 PCU at 8:00 am. Figure 5-10 displays the bus routes and their corresponding flow.

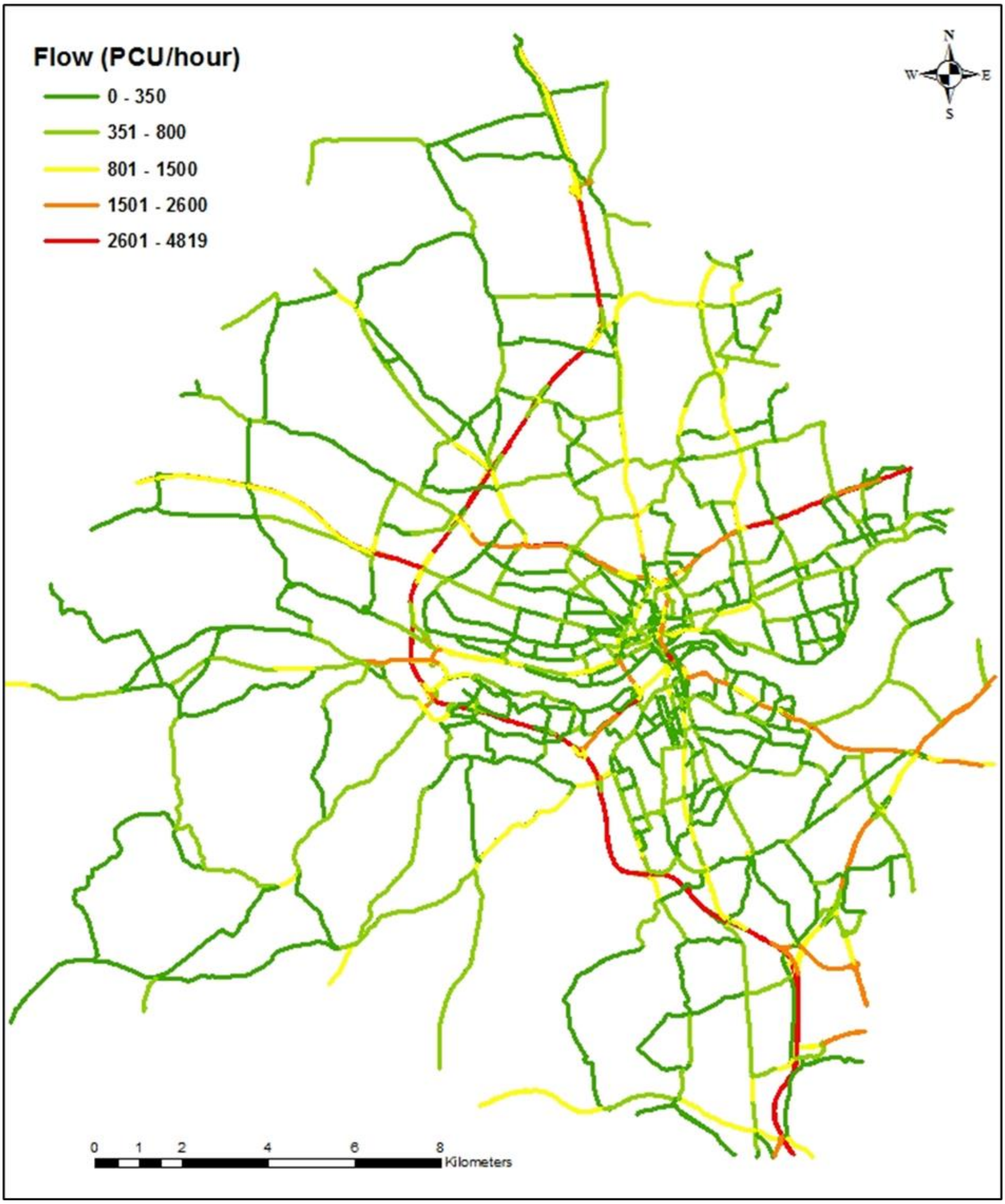


Figure 5-9: Baseline flow of vehicles at 8:00 am in Gateshead and Newcastle

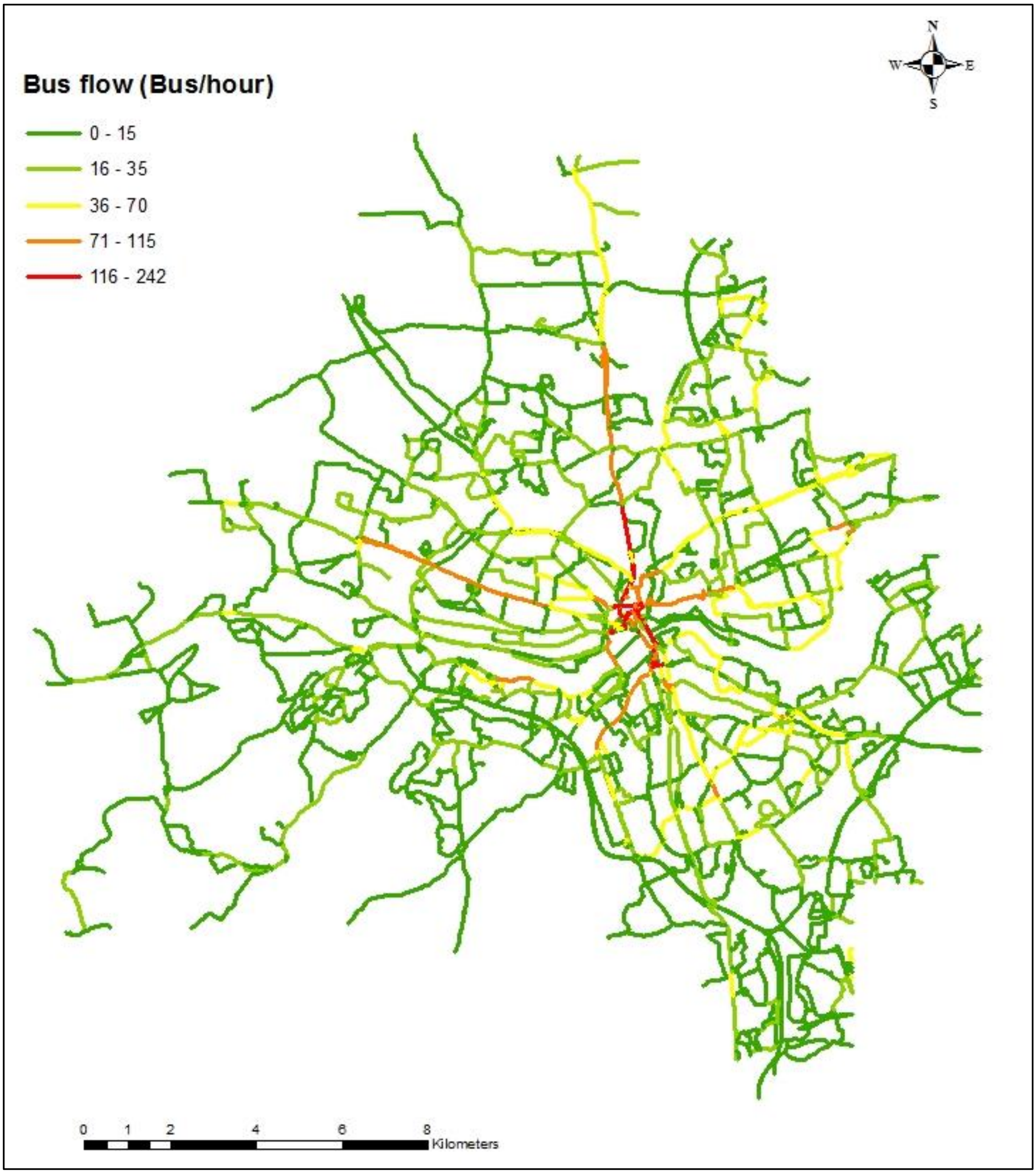


Figure 5-10: Routes and flow of buses at 8:00 am in Newcastle and Gateshead

5.7 Air Quality Modelling of the Baseline Framework

The reason for the development of the Baseline framework is not only to simulate current and future activities on the traffic network in the study area, but also to create a Baseline able to model rates of vehicular emissions and their dispersion in order to identify pollution concentrations at selected receptors. The following sections demonstrate the use of the Baseline framework in estimating pollution levels at the pollution monitoring station sites so as to compare modelled with observed pollution concentrations in order to ensure that the model is capable of simulating future pollution concentrations as a result of the adoption of more EVs within vehicle fleets.

Many factors influence the rates of emissions which are released by vehicles. The most common factors include vehicle speed, given that vehicles tend to emit higher emissions at low speed. For example, CO_2 emissions tend to be higher at a vehicle speed lower than 50 km/h (Barth and Boriboonsomsin, 2008), whilst NO_x emissions rates tend to be higher at low speed and gradually decrease to the lowest at intermediate speed (such as 40km/h to 60km/h) and subsequently increase to higher rates at high speed (Lozhkina and Lozhkin, 2016). In addition, vehicle types (such as cars, LGVs, HGVs and buses) and fuel type (petrol or diesel) play an important role in the quantity of emissions released, because petrol engines tend to emit more greenhouse gases but less NO_x compared to the equivalent diesel engines (DfT, 2018a, p. 11). Additionally, meteorological conditions such as wind direction and speed play a vital role in the dispersion of emissions accumulated in different locations, in particular in street canyons (Kim and Baik, 2004). Therefore, a consideration of meteorological factors, traffic volume and speed, in addition to differences in vehicle characteristics is important when calculating emissions in such a way that reflects reality.

5.7.1 Emission Factors

The Department for the Environment Food and Rural Affairs (DEFRA) Emissions Factors Toolkit (EFT) is an average speed emission model designed to assist local authorities in the UK to evaluate the impact that vehicle transit in their region has as regards local emissions, as part of their responsibilities under the Environmental Act 1995 (DEFRA, 2017b, p. 1). Emissions are determined at a road link level by the input of vehicle fleet composition, average traffic speed and road type. The 8th version of the EFT (v8) that was issued in December 2017, employs exhaust emissions such as NO_x and particulate matter from the European Environment Agency’s (EEA) COPERT (DEFRA, 2017b, p. 5). Emission Factors for Brake, Tyre and Road Abrasion particulate matters are based on NAEI assumptions (DEFRA, 2017b, p. 30).

According to DEFRA (2017b, p. 5), version 8 includes:

1. Updated NO_x and particulate matter speed emission coefficient equations, obtained from the European Environment Agency (EEA) COPERT 5 emission calculation tool (initially released in September 2016), including Euro 6 sub-categories;

2. The ability for the user to define the LGV petrol / diesel percentage split via the Euro Compositions advanced option;

3. Better representation of failure rates of both catalysts and Diesel Particulate Filters (DPFs). The input tables for the Euro Compositions Advanced Option no longer assumed failure rates within the presented proportions (default failure rates were subsequently assumed as part of the calculation procedure). Likewise, when using the Output % Contributions from Euro Classes Advanced Option, the proportion of total emissions attributable to failed catalysts and DPFs was presented separately within brackets alongside the emissions for each Euro category;

4. New NO_x, PM_{10} and $PM_{2.5}$ Annual Emissions Euro Split Advanced Options were added, giving emissions by kg/yr, broken down by vehicle type and Euro emission standard, with contributions from failed catalysts and DPFs;

5. A new Advanced Option allowed the user to output the fraction of primary NO_2 emissions (f-NO_2) for the input data provided;

6. Basic fleet assumptions for 2015-2030 updated in line with NAEI projections;

7. Euro class compositions for 2015-2030 updated in line with NAEI projections and TfL data (inclusive of Euro 6-subcategories);

8. The ‘Alternative Technologies’ Advanced Option were combined with the Traffic Format drop-down menu on the Input Data sheet. However, EFT v8 provided emission rates from 2017 up to 2030. It did not offer emission data beyond 2030.

More recently, the ninth version of EFT (v9) was published in May 2019. In this tool, the gradient of the road can be entered from 0 to 30% up or downhill. Additionally, vehicle loads of either 0, 50% or 100% can be entered. However, the EFT fails to address the effect of congestion on the increasing intensity of emissions released in relation to vehicle traffic.

5.7.1.1 Differences in Underlying Assumptions Between Emissions Factors Toolkits v5, v7 and v8

There are different Emissions Factors Toolkit (EFT) datasets; specifically, the EFT 4, 5, 6, 7, 8 and 9 datasets. With every successive version, minor and/or major changes are made to the EFT dataset as more measurements and associated data are made available. DEFRA (2019c, p. 5) stated that these changes may affect the output generated using the Toolkit and thus increasing the confidence in the more up to date output from the Toolkit, DEFRA recommends employing the latest version of the EFT (DEFRA, 2019c, p. 11). All preceding versions should no longer be used for the calculation of road vehicle pollutant emission rates, as they would not show a true picture of the current year's vehicle fleet changes and events (DEFRA, 2019c, p. 5). Hence, in this research, the latest EFT v8 was utilised to model vehicular emissions for the 2030 scenarios and EFT v7 was employed to simulate the Baseline scenario. The EFT v8 is not compatible with vehicle fleets in the year 2014.

Unfortunately, due to the variability of emission factors, the EFTs do not include any allowance for cold starts. The EXEMPT tool is recommended by DEFRA for the measurement of cold starts at junctions or car parks. This is a limitation with regard to the Emissions Factors Toolkit.

In previous versions of the EFT, i.e. all versions preceding version 7, Hydrocarbons were measured (DEFRA, 2016c, p. 4); however, from version 7 and onwards, there was a streamlining of the specified pollutants to merely include oxides of nitrogen (NO_x), PM_{10}, $PM_{2.5}$ and CO_2 only (DEFRA, 2016c, p. 1). This was due to the enhancements of engine technology and introduction of fuel-filling 'cap' thus reducing HC emissions from vehicles substantially. Furthermore, it is important to note that only the EFT v8 datasets contain emissions for the primary fraction of NO_2 because the data sources for the other versions do not contain emissions for NO_2 (CERC, 2018, pp. 31, 127). Both versions 7 and 8 have the same detailed vehicle categorisation. These categorisations are useful when applying, for example, emissions rates for bus lanes independently or conducting 'source apportionment studies' (CERC, 2018, p. 32). In most cases, a simple vehicle categorisation is sufficient; for example, light vehicles or heavy-duty vehicles.

In EFT v8, the 'Alternative Technologies Advanced Option' has been consolidated into the 'Traffic Format' drop-down menu on the 'Input Data' sheet, which makes it easier to directly input proportions of alternatively fuelled vehicles such as electric vehicles within the traffic fleet mix (DEFRA, 2017b, p. 6). Table 5-7 shows underlying differences in assumptions between EFT 5.3, EFT 7 and EFT 8.

Also, worthy of note is that there are differences between the rates of emissions calculated by EFT v7 and v8 because EFT v8 adopts emissions factors published by COPERT 5, whilst EFT v7 adopts COPERT 4 emissions factors. COPERT is a software tool developed by the European Environment Agency and is used extensively to calculate national emissions from road transport in Europe. According to O'Driscoll (2017, p. 145), real-world urban emissions were estimated to be 1.8 and 2.9 times the emissions calculated by COPERT 4 speed-dependent emissions factors for NO_x and NO_2 respectively, whilst the representatives of real-world urban emissions in COPERT 5 were in much better agreement, as they are more sensitives to speed and increase by greater amounts at lower speeds (O'Driscoll, 2017, p. 146).

Finally, only EFT v9 (not EFT v8, v7, or v6) can model emissions released by HGVs for the three loading cases of fully loaded, 50% loaded and 0% loaded, and takes into account the effect of road gradient on amounts of emissions. This version of EFT was released after the research reported in this thesis had been completed.

Table 5-7: Summary of changes of the EFT versions (DEFRA, 2019c)

Version 5	Version 7	Version 8
'Advanced Options' added to the Input Data worksheet allowing Advanced Users to provide User Defined Euro Compositions and Alternative Technologies and output relative percentage contributions from Euro Classes. NOx emission factor and vehicle fleet information updated. NOx Emissions Factors were taken from the EEA COPERT 4v8.1 emission calculation tool. Emissions Factors for other pollutants were those published by the DfT on 29 June 2009.	NO_x and Particulate Matters speed emission coefficient equations for Euro 5 and 6 vehicles updated, taken from the European Environment Agency (EEA) COPERT 4v11 emission calculation tool, reflecting more recent evidence on the real-world emission performance of these vehicles. Streamlining of pollutants – removal of Hydrocarbons as an option in the EFT. CO_2 tailpipe emissions associated with alternative fuelled vehicles are included in emissions calculations, in addition to those from petrol and diesel vehicles. The applied CO_2 scaling factors for alternative technology vehicles are consistent with those applied in London Atmospheric Emissions Inventory (LAEI). The ability to output CO_2 when Alternative Technologies advanced user input option is selected has also been added. Updated fleet composition data, accounting for updates to traffic and fleet projections in London, based on information from TfL. This includes varying fleet data specific to the Central, Inner, Outer and Motorway areas of London. The ability for user to define Euro compositions individually for the Central, Inner, Outer and Motorway areas of London. The ability to enter up to at least 25,000 rows of input data or up to 200,000 rows of input data, dependent upon selected output options.	Correction of bus and coach split on London roads when entering data using the Alternative Technologies traffic format input option only. Updated NO_x and Particulate Matters speed emission coefficient equations, taken from the European Environmental Agency (EEA) COPERT 5 emission calculation tool, including Euro 6 subcategories. Ability for the user to define LGV petrol / diesel percentage split via the Euro Compositions advanced option. Better representation of failure rates of both catalysts and Diesel Particulate Filters (DPFs). The input tables for the Euro Compositions Advanced Option no longer assume failure rates within the presented proportions (default failure rates are subsequently assumed as part of the calculation procedure). Also, when using the Output % Contributions from Euro Classes Advanced Option, the proportion of total emissions attributable to failed catalysts and DPFs is now presented separately within brackets alongside the emissions for each Euro category. New NO_x, PM_{10} and $PM_{2.5}$ Annual Emissions Euro Split Advanced Options added, giving emissions by kg/yr, broken down by vehicle type and Euro emission standard, with contributions from failed catalysts and DPFs again separated. New Advanced Option that allows the user to output the fraction of primary NO_2 emissions (f-NO_2) for the provided input data. Basic fleet assumptions for 2015-2030 updated in line with DfT (2015) projections. Euro class compositions for 2015-2030 updated in line with DfT (2015) projections and TfL data (inclusive of Euro 6 subcategories). The ‘Alternative Technologies’ Advanced Option has been consolidated into the Traffic Format drop down menu on the Input Data sheet.

5.7.1.2 Development of Emission Factors

The continuous monitoring of emissions from mobile sources at all times and on all roads is at present impractical and financially unfeasible (Smit *et al.*, 2010). Hence, a method is needed to estimate the emissions associated with road traffic that uses readily available data. The agreed procedures and guidelines for reporting emissions released from road traffic worldwide involve a combination of traffic activity data (e.g. km travelled and speed) with vehicle fleet mix data (e.g. vehicle composition by size of vehicle and engine, exhaust treatment technology, Euro emissions standard, fuel type and vehicle use) and emissions factors (Tsagatakis, 2018, p. 2). The term 'emissions factors' was defined by the United Nations Framework Convention on Climate Change (UNFCCC, 2014) as '*a unique value for scaling emissions to activity data in terms of a standard rate of emissions per unit of activity*'. For road traffic, the National Atmospheric Emissions Inventory (NAEI) defines emissions factors as the *'relationship between the amount of a pollutant that is produced and the number of vehicle miles travelled'* (DEFRA, 2014). Hence, an emissions factor is a representative value of the typical rate at which a pollutant is released from a vehicle into the air.

Emissions factors can be grouped, which allows an emissions model to calculate pollutant emissions released from a fleet consisting of a number of vehicle types such as passenger cars, buses and HGVs (Peace *et al.*, 2004). Emissions models are typically employed to estimate total emissions from a given geographical area, including a stretch of road or road network such as those found in Brady and O'Mahony (2011), Soret *et al.* (2014) and Dey *et al.* (2018a). An emissions inventory is a list of emissions by source for a given geographical area and time period which can be compiled using an emissions model. There are many sources of air pollution including household heating, industrial processes, agriculture and traffic (DEFRA, 2014). Several emissions modelling approaches have been adopted for traffic including average-speed, corrected average-speed, multiple linear regression and instantaneous models (Barlow and Boulter, 2009, p. 4), as well as dependence on factors including fuel and engine type and age of the vehicle (Ericsson, 2001; Barlow and Boulter, 2009, p. 3). Examples of methods where emissions factors are developed are presented in the next sections.

There are several methods for estimating emissions factors from road vehicles. Typically, laboratory dynamometer tests are employed to determine emissions factors from road vehicles (Ning *et al.*, 2008; Franco *et al.*, 2013). A few common alternatives to dynamometer experiments include on-board measurement (Huo *et al.*, 2012a; Huo *et al.*, 2012b), remote sensing measurement campaigns (Guo *et al.*, 2007), and tunnel experiments (Colberg *et al.*, 2005). In addition to these approaches, other methods such as inverse modelling (Zárate *et al.*, 2007) and mass balance (Kalthoff *et al.*, 2002) are only rarely employed to estimate emissions, and therefore they are deemed to be beyond the scope of this work.

It should be noted that CO_2 and SO_2 emissions can easily be estimated from road vehicles, since their quantities are directly related to fuel consumption (Thambiran and Diab, 2011). In contrast, the emissions of other pollutants from road vehicles are dependent on many other parameters, including engine load and ambient temperature. Thus, it is difficult to accurately estimate their emissions rates (Franco *et al.*, 2013).

5.7.1.3 Dynamometer Tests

Dynamometer tests are the most commonly employed method to estimate emissions from road vehicles (Barlow and Boulter, 2009, p. 35; Franco *et al.*, 2013). They involve a vehicle being run on a dynamometer under controlled conditions whilst the vehicle exhaust gases are simultaneously collected and subsequently quantified (Prati *et al.*, 2011). In these tests, the vehicles are subjected to various driving cycles, which involve changing the dynamics (speed, engine load etc.) of the vehicle to reflect 'real-world' driving conditions (André *et al.*, 2006; Kamble *et al.*, 2009). During these test cycles, different activity parameters can be measured; for example, engine load, acceleration and deceleration (Prati *et al.*, 2011).

The main benefit of dynamometer tests compared to other emissions factors development approaches is that tests can be performed in a controlled (laboratory) environment under standard conditions, allowing the test procedures to be easily replicated (Franco *et al.*, 2013). Subsequently, the test's outcomes (such as emissions factors produced from different dynamometer tests) can be compared and subsequently compiled in a single database, leading to an increase in the number of vehicles sampled. Conversely, the standardisation of dynamometer test conditions can be a drawback. This is because sample size, maintenance level, vehicle size and age of test cars are all sensitive to uniform measurement methods, test conditions and driving patterns during dynamometer tests (Winther, 1998). Failure to consider the sensitivity of these parameters whilst carrying out laboratory testing could lead to misunderstandings with regards to emissions factors and ultimately under- or over-prediction of emissions model estimates (Boulter *et al.*, 2009c, p. 7).

In dynamometer test driving cycles provide a speed-time profile of driving behaviour in a specific area/region (Newman *et al.*, 1992). Current driving cycles are developed with on-road driving data such as Assessment and Reliability of Transport Emissions Models and Inventory Systems (ARTEMIS), rather than simulation methods such as the Japanese driving cycle and the California-seven cycle and mode-cycle, and are typically stratified by route type (e.g. urban, rural, motorway), vehicle type (HGV, LGV), time period (peak, off-peak) and speed level (Hung *et al.*, 2007). It is important to apply correction factors to the emissions rates resulting from dynamometer tests to take into account deviations between laboratory and on-road conditions; for instance, temperature and average vehicle speed (Grieshop *et al.*, 2012). Nevertheless, it is acknowledged in the literature that there are deviations between the representatives of laboratory dynamometer driving cycles and on-road real-world driving conditions (De Vlieger, 1997; Lau *et al.*, 2011; Smit and Bluett, 2011; Grieshop *et al.*, 2012; Franco *et al.*, 2013). Certain studies have documented these deviations in relation to the application of emissions factors developed from generic, and often national or international, driving cycles known as ‘standard’ driving cycles (Lenaers, 1996; Joumard *et al.*, 1999; Joumard *et al.*, 2000). These cycles are typically legislative cycles performed to test vehicles registered within a country or region, such as in Europe. Given that the characteristics of driving in each city are unique because of different vehicle fleet mixes, driving behaviour and road network topography (Tsai *et al.*, 2005; André *et al.*, 2006), the use of emissions factors developed from these standard driving cycles have been shown to substantially underestimate emissions (Joumard *et al.*, 2000). For instance, Rhys-Tyler and Bell (2012) compared outcomes from remote sensing detectors (RSD) campaign in London with emissions results over the New European Drive Cycle (NEDC) and found that CO emissions from petrol-powered cars older than three years measured using remote sensing were around 1.3 times higher than the NEDC. This was noted to rise to 2.2 times for cars between four and eight years old and to 6.4 for cars between nine and 12 years old. Furthermore, Rhys-Tyler and Bell (2012) found similar results when NO_x, HC and particulate matters emissions were compared. Discrepancies between dynamometer and ‘real-world’ estimates have also been documented by Carslaw *et al.* (2011b). The discrepancies between emissions estimates developed from dynamometer tests and those developed from RSD campaigns are discussed further in section 5.7.1.5.

In specific cases, dynamometer driving cycles have been developed using more local on-road driving data in attempting a better representation of real-world conditions in emissions estimates. For instance, Nutramon and Supachart (2009) compared the more local Bangkok driving cycle to that of the European driving cycle (the adopted legislative cycle for testing vehicles registered in Thailand) and reported that HC and CO emissions from the Bangkok driving cycle were almost two and four times respectively higher and NO_x emissions and 10% greater than those of the European driving cycle. Durbin *et al.* (2002) unearthed similar results as they observed substantially greater emissions estimates from tests using the New York City driving cycle compared to experiments with the Federal Test Procedure (FTP). They concluded that the discrepancies found were attributable to the inability of the standard FTP cycle to consider the more aggressive acceleration behaviour that was present at a local level. Tsai *et al.* (2005) compared a local driving cycle (the Kaohsiung Driving cycle, KHM) to the standard legislative cycle used for testing vehicles in China (the European driving cycle). They reported that the percentage of time spent in acceleration and deceleration modes in the Kaohsiung Driving cycle was significantly lower for the European driving cycle, resulting in completely different emissions factors and fuel consumption. Nevertheless, despite the development of such local cycles, it is not practical to develop and regularly update driving cycles at every locality. Thus, it is unavoidable that the use of 'standard' driving cycles will remain common practice in the future.

Further drawbacks pertaining to dynamometer tests include the use of set ambient temperatures and preconditioning procedures and the absence of road gradients (Franco *et al.*, 2013). Additionally, emissions factors developed from dynamometers tests are usually not representative of the entire vehicle fleet since only a few vehicles of each technology type are typically tested (Boulter *et al.*, 2009a). This can lead to the failure of dynamometer tests to appropriately represent local extremes (including gross emitters) associated with real-world driving (Liaw and Dubarry, 2007). Whilst representing a small number in the vehicle fleet these gross emitters, often associated with poorly maintained vehicles (Muncaster *et al.*, 1996), account for a high percentage of total pollutant gases. This suggests that their emissions behaviours are not frequently captured in dynamometer tests (Lau *et al.*, 2011; Grieshop *et al.*, 2012; Huo *et al.*, 2012a).

5.7.1.4 Instrumented Vehicles

An additional approach by which emissions factors from vehicles can be estimated is conducting on-board measurement or instrumented vehicle approaches (Lenaers, 1996; Chen *et al.*, 2007a; Hung *et al.*, 2007; Lopez *et al.*, 2009). In this type of method, a device is fitted to a vehicle which directly measures the rates of emissions and other activity parameters; for instance, the speed, engine load and gear changes of the vehicle whilst it is driven in real-world conditions (Lau *et al.*, 2011). Often GPS positioning also is reported to take account of gradient location at junction mid-link etc (Franco *et al.*, 2013). The main benefit of this method over dynamometer tests is that emissions are collected under real-world conditions and as such, the influence of external variables such as temperature, are accurately reflected in emissions estimates (Lau *et al.*, 2011). Moreover, vehicle dynamics during on-road experiments are more representative of real-world driving (De Vlieger, 1997; Lau *et al.*, 2011; Huo *et al.*, 2012a; Huo *et al.*, 2012b) than dynamometer tests, although it is important to choose routes on the network that are 'typical' of the journeys for which an attempt is being made to describe the relevant emissions factors.

The main drawback of instrumented vehicle experiments is related to restriction of sample size, as these tests are generally conducted using a limited number of vehicles of a specific type. For instance, de Haan and Keller (2000) carried out extensive on-board measurements of Euro-I petrol passenger cars during real-world driving to produce a set of emissions factors and subsequently an instantaneous emissions model. The model was shown to predict emissions more accurately than models using dynamometer tests over standard driving cycles. However, the instantaneous emissions model was restricted by sample size and vehicle type, having been developed based on a small number of Euro-1 petrol cars. Thus, the model could not be applied to a wider vehicle fleet.

5.7.1.5 Remote Sensing

Remote sensing detectors (RSD) involve the use of ultraviolet and infrared beams of light which, when passing through a vehicle exhaust plume, are absorbed by its constituent gases and particles. This allows volume concentrations of HC, CO, CO_2 and NO to be estimated simultaneously (Guo *et al.*, 2007). In contrast to on-board measurement methods, the use of RSD has enabled emissions data to be gathered from large samples of vehicles driving in real-world conditions. For instance, a study conducted in Hong Kong by Chan *et al.* (2004) involved a series of remote sensing exhaust emissions measurements resulting in the collection of emissions data from a total of 10,781 petrol vehicles. Kuhns *et al.* (2004) compiled an emissions inventory in Las Vegas of 61,207 gasoline and 1,180 diesel vehicles based on RSD studies, while Sjödin and Andréasson (2000) presented an RSD study involving the monitoring of 30,000 emission measurements in Gothenburg, Sweden. Moreover, an RSD measurement campaign in the UK by Carslaw *et al.* (2011b) resulted in the recording of a large sample size of 84,269 valid records emission measurements. RSD studies, such as those by Smit and Bluett (2011) and Beevers *et al.* (2012) acknowledged a lack of agreement between RSD data and dynamometer-based emissions inventories. The discrepancies are attributed to the fact that RSD data provides more detailed measurements that can represent a better reflection of real-world driving dynamics compared to dynamometer studies (Rhys-Tyler *et al.*, 2011).

There are a number of limitations associated with RSD experiments; in particular, their dependence on appropriate meteorological conditions since high winds cause the rapid dispersion of exhaust plumes, the inability to capture vehicle emissions emitted from exhausts at varying heights, and restriction of use on specific road types (such as single lane, urban roads that are on a gradient) (Carslaw *et al.*, 2011a, p. 32). Another limitation is due to the fact that specific road-types have limited range of speeds.

Furthermore, for the purpose of maintaining a consistent and acceptable level of accuracy, RSD equipment requires daily multi-point calibration with real wet exhausts which in some tests might not be commonly adopted practice (Sjödin and Andréasson, 2000).

5.7.1.6 Tunnel Experiments

In a typical tunnel study, pollutant concentrations are sampled at the entrance and exit of a tunnel (Kuhns *et al.*, 2004). The difference between pollutant concentrations at the entrance and exit points is assumed to be directly associated with the mass emitted from the vehicles passing through the tunnel during the sampling period (Mancilla and Mendoza, 2012). Calculation of average emissions factors that are representative of the total vehicle fleet passing through the tunnel is possible from this data (Colberg *et al.*, 2005). Two types of tunnel study are available, the first type is concerned with the development of emissions factors and the other type with the air quality levels within a tunnel and compliance with legislation (El-Fadel and Hashisho, 2000). Both types can be considered useful compared to other emissions factor development methods because they can determine the average emissions of a large number of vehicles, reflect vehicle exhaust in real-world traffic, are inexpensive compared to laboratory-based tests, and provide estimates that are representative of exhaust and non-exhaust emissions (Hung-Lung and Yao-Sheng, 2009).

Relatively, tunnel studies have been demonstrated to provide accurate estimates of local emissions (Weingartner *et al.*, 1997; El-Fadel and Hashisho, 2000; Touaty and Bonsang, 2000; Ho *et al.*, 2007); however, they are subject to a number of limitations. They are restricted in application because they provide emissions factors specific to a tunnel and its particular driving conditions as well as road geometry and topography (Ning *et al.*, 2008). Moreover, the vehicle composition travelling through the tunnel might not be representative of the urban vehicle fleet in the surrounding area, suggesting that the information provided is often not broadly applicable to the wider road network (Ning *et al.*, 2008). It is possible to subsequently estimate emissions factors by vehicle type using different calculations and regression analysis, although these techniques are subject to assumptions and limitations of their own (Grieshop *et al.*, 2006).

5.7.1.7 Summary of Emissions Factors Development:

Although dynamometer tests do not accurately represent real-world driving conditions, they have the benefit of reproducibility. Real-world driving studies allow variability in vehicle dynamics to be represented in emissions factors, nonetheless these studies can be restricted by vehicle sample size, such as when instrumented vehicles are used, sensitivity to meteorological conditions at the time of measurement such as remote sensing, or lack of applicability to the wider road network for instance with tunnel studies.

Dynamometer experiments due to their high reproducibility enable governing organisations to have a legislative driving cycle in place that guarantees new vehicles are meeting the required standards prior to putting new vehicles on general sale. This means that dynamometer tests will remain fundamental to emissions factor development. Nevertheless, a combination of dynamometers test data with real-world data would provide a more robust approach to estimating and controlling emissions from road vehicles (Rhys-Tyler *et al.*, 2011) and the application of several methods to estimate emissions factors on a national scale is vital if air quality limit values are to be met in the UK.

5.7.1.8 Emissions Models

During the development of emission factors, different activity parameters can be recorded and subsequently can be used in conjunction with emissions rates, to develop emissions models. Many parameters affect emissions rates. Some details of them are now presented. Bearing in mind that these models are greatly affected by the emissions factors they include, emissions factors are in turn governed by the method from which they have been developed. Therefore, emissions models often share the same benefits and drawbacks as the emissions factors they use because it is challenging to incorporate the uncertainties associated with emissions factors into the models (Hung-Lung and Yao-Sheng, 2009).

Average-speed emissions models such as MOBILE, EMFAC and COPERT are the most commonly used (Smit *et al.*, 2008a). This is primarily because their data requirements of traffic flow, vehicle types and their average-speed, are more often than not readily available, the models are comparatively easy to use and several are available free of charge. Also, these types of models are some of the oldest approaches still in use (Barlow and Boulter, 2009, p. 5). The emissions factors employed in these models are based upon the principle that the average emissions for a particular pollutant and a given type of vehicle varies due to continuous changes in the average-speed during a trip (Barlow and Boulter, 2009, p. 6). Therefore, the model assigns an average emissions factor according to the user assigning a specified average-speed and vehicle type.

A major concern with average-speed models is that trips can have very different vehicle dynamics and emissions even with the same average speed (Barlow and Boulter, 2009, p. 5). For instance, an average-speed of 60 km/h on an arterial road would represent uncongested free-flowing conditions, while the same speed on a motorway could represent much more congested conditions, such as involving stop and go dynamics (Smit *et al.*, 2008a). This is particularly relevant in the case of modern catalyst equipped vehicles, for which a large proportion of total emissions during a trip could be as a result of such congested, stop-start conditions (Huo *et al.*, 2012b). Very short and sharp increases in emissions may be attributed to such conditions (Lau *et al.*, 2011; Grieshop *et al.*, 2012). Average-speed models fail to explicitly capture these peaks in emissions and, as such, it could be difficult to determine their representativeness.

One method used to overcome the limitations of average-speed models to accurately represent the impact of changing vehicle dynamics is the corrected average-speed emissions modelling approach. This approach, such as in the TEE model (Negrenti, 1999), employs average-speed, green time, percentage of traffic signals, link length and traffic density variables to estimate a congestion correction factor which is subsequently applied to an average-speed model (Smit *et al.*, 2010). By applying these variables, the model can identify vehicle time spent in different modes (such as idling, acceleration, cruising etc) (Smit *et al.*, 2010) and adjusts emissions profiles where they better represent the short-sharp peaks in emissions that have likely taken place during a trip (Smit *et al.*, 2008b).

In traffic situation models, specific emissions factors are assigned to specific traffic situations such as ‘stop and go’ or ‘free flow’ (Colberg *et al.*, 2005). The user is required to input a textual description for each road that directly relates to these descriptions. The model depends on the assumption that the user can relate to the traffic situations defined in the model (Boulter and Latham, 2009a, p. 12).

Several linear regression models such as VERSIT+ (Smit *et al.*, 2007) are based on the completion of a large number of dynamometer test cycles during which a number of descriptive parameters are recorded (Barlow and Boulter, 2009, p. 7). Regression analysis is employed to fit continuous functions to average emissions and each variable, as this allows the variables that best describe the emissions to be recognised (Smit *et al.*, 2008b). The model predicts a range of variables and assigns the most appropriate emissions factors according to the vehicle driving cycle data entered by the user (Smit *et al.*, 2007).

Instantaneous emissions models such as MODEM and PHEM relate emissions to vehicle operation over a small time period (such as minutes or seconds). An emissions rate is calculated for each time period and the sum of the emission rates for all time periods is used as the overall link emissions value (Barlow and Boulter, 2009, p. 7). Moreover, Boulter *et al.* (2007, p. 23) documented a comprehensive evaluation of instantaneous emissions models and concluded that, given the level of detail required, instantaneous models are not suitable for use at larger scales such as nationally.

5.7.1.9 Road Transport Emissions in the UK

Emissions from road traffic in the UK are estimated using the UK NAEI on a national scale, which includes a set of average-speed emissions factors specific to the general UK vehicle fleet in conjunction with national road traffic activity (flow) data from the Department for Transport (DfT) traffic census. The inventory includes emissions factors for a large number of pollutants, including $PM_{2.5}$, PM_{10} and NO_x which are of primary concern here. The National Green House Gas Emissions Inventory (NGHGI) forms a part of the NAEI and similarly includes emissions factors for a large number of GHGs, including CO_2. The emissions factors are revised and updated regularly, the most recent factors were published in DEFRA (2019c).

The Transport Research Laboratory (TRL) was commissioned by the DfT in 2009 to review the NAEI methodology used to estimate emissions factors. This work resulted in reports that reviewed NAEI methodologies for estimating hot exhaust emissions factors (Boulter *et al.*, 2009a), cold start emissions (Boulter and Latham, 2009a), fuel properties (Boulter and Latham, 2009b), exhaust emission factors for road vehicles in the UK (Boulter *et al.*, 2009b) and evaporative emissions (Latham and Boulter, 2009) for road vehicles. Additionally, a report was published documenting deterioration factors and other modelling assumptions for road vehicles (Boulter, 2009). The main outputs from the work comprised a driving cycle reference book, the subsequent revision of the average-speed approach for estimating hot exhaust emissions and the publication of fuel and mileage scaling factors (Barlow and Boulter, 2009).

Despite revisions of the UK emissions factors, discrepancies between real-world emissions and those in the NAEI have been documented, and a significant amount of research has focused on trends in NO_x emissions from road vehicles (Latham *et al.*, 2001, p. 37; Jenkin, 2004; Carslaw and Beevers, 2005; Rhys-Tyler *et al.*, 2011). This has occurred due to the fact that, although emissions standards established in the UK have shown a reduction, a synergistic decrease in ambient NO_2 concentrations has not been recorded (Carslaw *et al.*, 2011a; Carslaw *et al.*, 2011b). Furthermore, in various places and particularly at roadside locations, increases in NO_2 concentrations have been found (AQEG, 2007). The reason for this lack of synergy between emissions standards and concentrations of NO_2 can be attributed to an increase in the proportion of NO_x emitted as f-NO_2 in vehicle exhaust fumes (Carslaw and Beevers, 2004). Levels of f-NO_2 increased in the UK from 5-7% in 1996 to 15-16% in 2009, whilst in London the increase has been greater at 5-7% in 1996 to 21-22% in 2009 (Carslaw *et al.*, 2011a, p. 3). Research based on UK data indicates that the increase in f-NO_2 is due to the increased penetration of diesel vehicles into the fleet and an increase in the number of vehicles primarily buses equipped with diesel particulate filters (DPF) and oxidation catalyst as exhausts after treatment technologies (AQEG, 2007). Hueglin *et al.* (2006) documented similar findings for road traffic in Europe.

A comprehensive investigation by Carslaw *et al.* (2011a, p. 34), involved the analysis of direct measurements of emissions from 72,000 valid emissions measurements in field campaigns using a RSD. The data was compared to current UK emissions factors and alternative emissions factor estimates from the Handbook of Emissions Factors (HBEFA), which was developed on behalf of the Environmental Protection Agencies of Germany, Switzerland and Austria and COPERT 4, which enabled discrepancies between current UK factors to be identified. A summary of the findings from the research conducted by (Carslaw *et al.*, 2011a; Carslaw *et al.*, 2011b) is presented in Table 5-8.

Carslaw *et al.* (2011a, p. 3) and Carslaw *et al.* (2011b) related the discrepancies between the RSD data and other emissions factor data sets observed for petrol cars result from the greater failure of real-world catalysts and emissions degradation rates. For the results observed for HGVs, it was concluded that selective catalyst reduction (SCR) was not effective at urban driving speeds (Carslaw *et al.*, 2011a, p. 4) and therefore emissions were significantly greater under these conditions. Moreover, for discrepancies observed for diesel LGVs and cars, it was established that the LGV test cycles used to develop the UK emission factors (UKEFs) were inadequate to accurately represent real-world conditions, leading to underestimates regarding NO_x emissions (Carslaw *et al.*, 2011a, p. 78).

In a study of data collected during an RSD campaign in London by Southwark Council and Ealing Council, Rhys-Tyler *et al.* (2011) established that Euro 2 diesel cars have NO emissions significantly higher than Euro 1 or Euro 3, whilst Euro 4 diesel cars have NO emissions that are between 6 and 17 times higher than the equivalent Euro 4 petrol car. Additionally, median NO emissions from Euro IV diesel light commercial vehicles (<3.5t) were observed to be approximately 25% higher than emissions from Euro 4 diesel passenger cars.

Table 5-8: Summary of findings from remote sensing detection research conducted by Carslaw *et al.* (2011b) and Carslaw *et al.* (2011a, p. 49)

Dataset	Comparison with	Vehicle Type	Euro Class	Conclusion
UKEF	COPERT 4 and HBEFA	Diesel Cars, LGVs	Euro 3 onwards	UKEF Lower
RSD	UKEF and HBEFA	Petrol Cars	Euro 1 and 2	RSD Higher
RSD	UKEF and HBEFA	Diesel LGVs	Euro 3 onwards	RSD Higher
RSD	UKEF and HBEFA	Diesel Cars	All Euros	RSD Higher
RSD	UKEF and HBEFA	Rigid HGV (Diesel)	Euro I to IV	RSD Higher

5.7.2 Emissions Simulation

The Environmental Act 1995 requires local authorities to complete frequent reviews and assessments of local air quality. Thus, the DEFRA has published the Emissions Factors Toolkit (EFT) to allow the calculation of emissions rates released by road traffic for different road types, vehicle speeds, vehicle fleet mixes and specific years (DEFRA, 2017b). The internal library found in the EFT comprises raw data in relation to vehicle emissions rate factors based on European emission standards (such as Pre-Euro and first Euro) and fleet proportion data at several road types including urban road (Mumovic *et al.*, 2006). Thus, the EFT is able to calculate vehicular emission rates for NO_x, PM_{10}, $PM_{2.5}$ and other pollutants. The EFT is updated frequently based on the availability of data for fleet composition and emission rate factors provided by the National Atmospheric Emissions Inventory (NAEI) and COPERT (DEFRA, 2017b). As the EFT only reads and produces data in Excel format and lacks the ability to read traffic data written in GIS files, which is the format of the Baseline traffic model, finding a program that can read GIS files and utilise the features of the EFT in an efficient way is crucial.

The PITHEM software suite is built on EFT version 5. However, this version lacks the feature of the calculation of emissions rates associated with alternative-fuelled vehicles such as EVs. This vital feature is available in EFT v8 that was utilised to calculate the emissions rates related to 2030 scenarios by amending PITHEM output files that contain emission rate values. Therefore, the PITHEM is used as a tool to transfer emissions rates from the EFT to ADMS-Urban. As EFT v8 does not support emission rates calculation for 2014 or earlier, EFT v7 was used to model vehicular emissions for 2014, whilst EFT v8 was used to calculate vehicular emissions in relation to 2030 traffic flow scenarios. The following paragraphs demonstrate the role of EFT v7 and v8 in amending the PITHEM outputs.

5.7.2.1 Amending the PITHEM Output with EFT Outcomes

The PITHEM generates many files in relation to link length, geometry, traffic activity and associated emission rates. One of these files provides average daily values of NO_x, PM_{10} and $PM_{2.5}$ in g/km/s for each road link. Moreover, the PITHEM generates time-varying emission factors which can be applied to the average daily values of emission rates at a certain link to produce a diurnal profile of emission rates for that link.

These average daily values which are calculated based on EFT v5 should be amended according to the latest version of the EFT. The effectiveness of amending the PITHEM outputs with results produced by EFT v7 must be assured prior to any amendments.

Prior to amending the PITHEM emission rate output with EFT v7 outputs, correlations were performed between the emission rates generated by the PITHEM outputs and average values for 24 hours of emissions rates produced by EFT v5. The traffic model consists of 2887 roads. Each road has different traffic parameters for each hour. This indicates that 69,288 lines of data (2887 links × 24 hours) need to be entered into EFT v5. Unfortunately, the 69,288 lines cannot be entered into EFT v5 in one run, because the entry capacity associated with EFT v5 is restricted. Therefore, EFT v5 was run several times to model the emission rates from the 69,288 lines of traffic data. The results from each run for each pollutant rate were averaged and compared with the PITHEM output. For emissions rates released by cars, LGVs and HGVs, the comparisons indicate excellent correlations between the PITHEM and EFT v5 results, as shown in Figure 5-11.

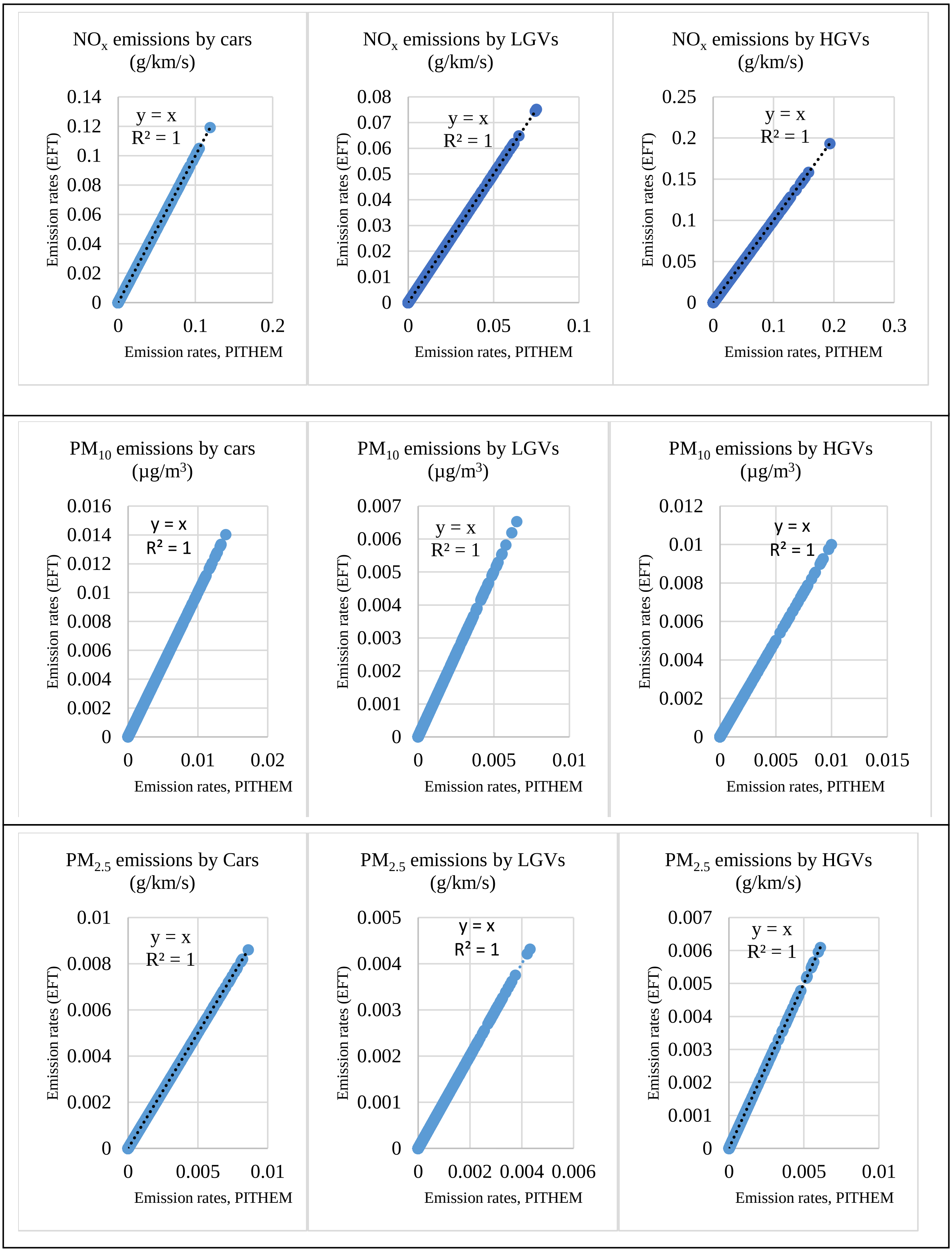


Figure 5-11: Correlations of emission rates calculated by the PITHEM and EFT v5 for cars, LGVs and HGVs

Furthermore, for the bus network, emissions rates calculated by the PITHEM were compared with emissions rates calculated by EFT v5.3. These networks comprise 10,608 road links over 24 hours, forming 254,492 (10,608 link × 24 hour) rows, which need to be entered into EFT v5. The capacity of EFT v5 is only 10,000 inputting rows per one run in the calculation of rates of NO_x, PM_{10} and $PM_{2.5}$. Therefore, EFT v5.3 was run 26 times to cover the entire 254,492 inputting rows. For each link, the calculated emissions rates for 24 hours were averaged and compared with those emissions rates produced by the PITHEM. The results indicate excellent correlations, as Figure 5-12 reveals.

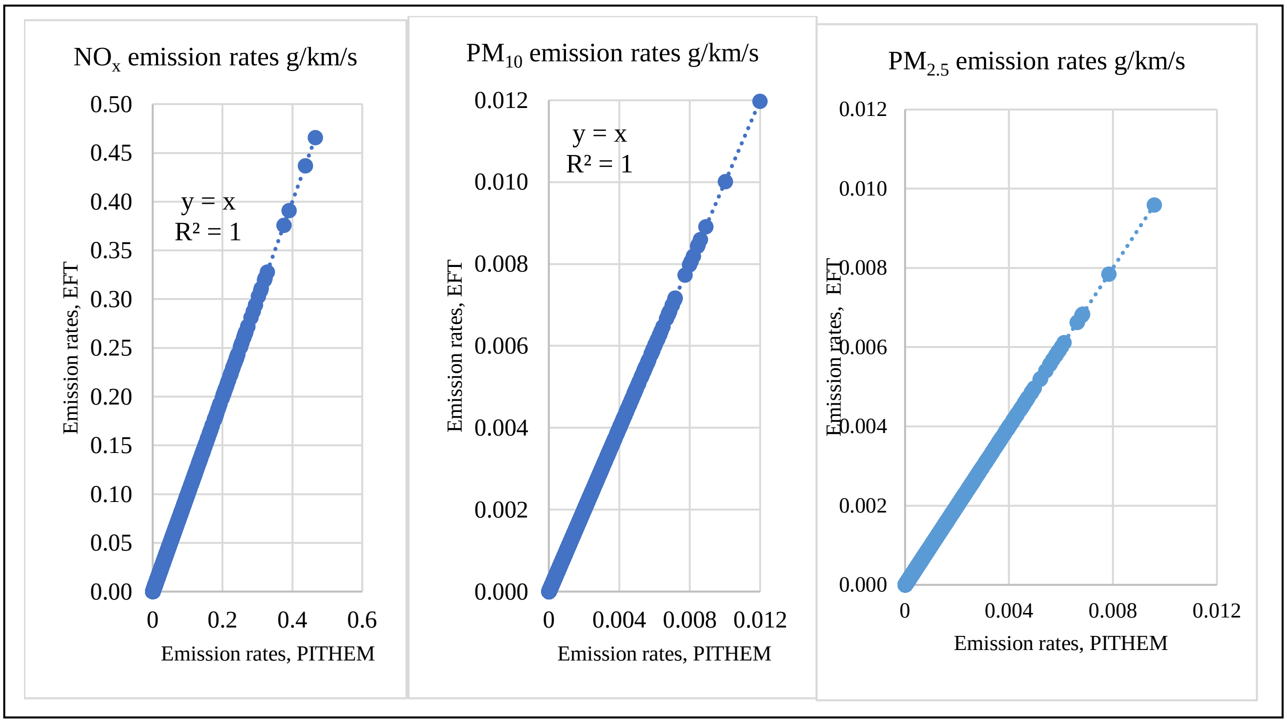


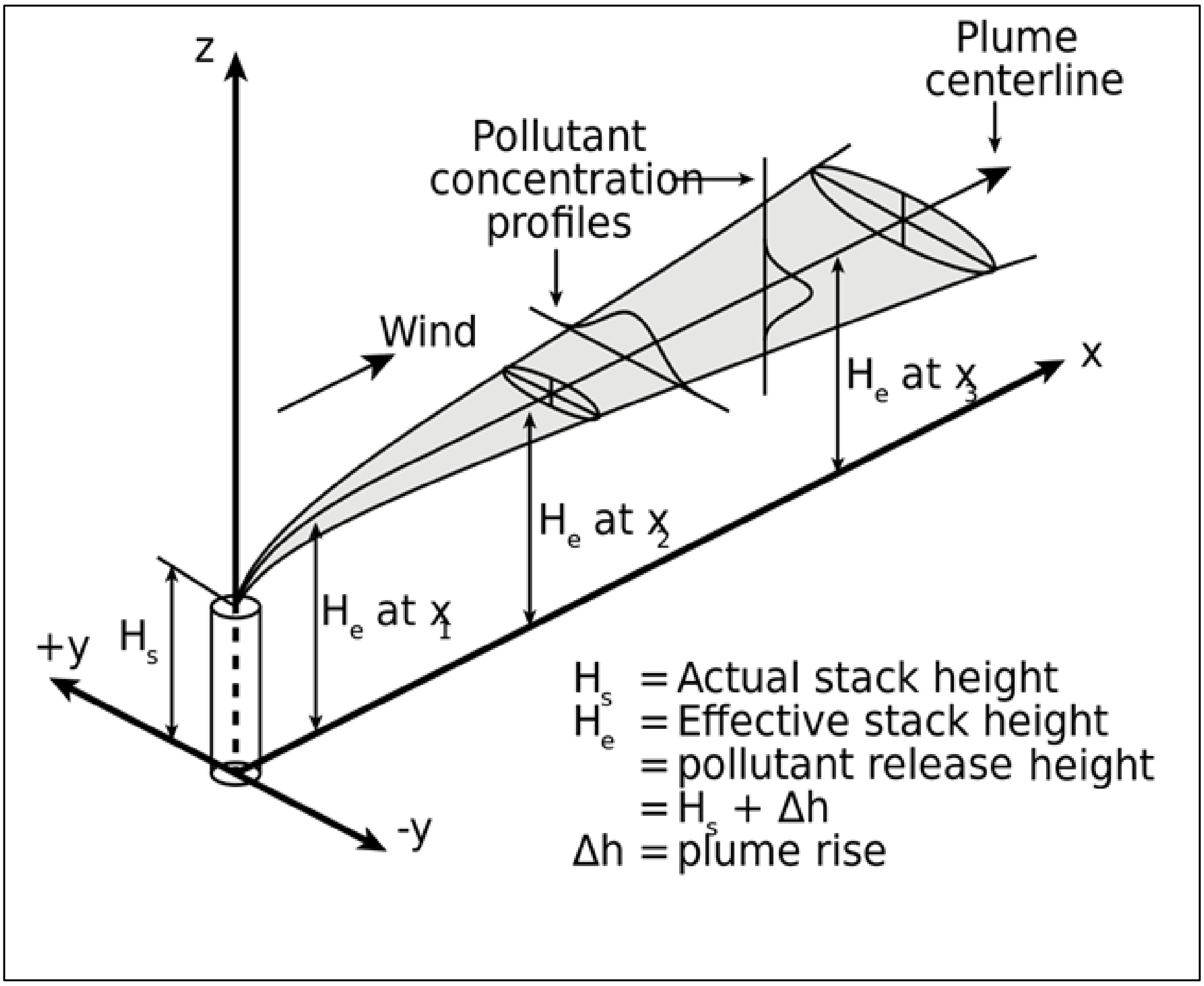
Figure 5-12: Correlations of emission rates calculated by PITHEM and EFT v5 for buses

The high correlations observed between the outcomes of the PITHEM and EFT v5 illustrates excellent performance in replacing the averaged emissions rates modelled by PITHEM with those calculated by EFT v5.3. Therefore, emission rates modelled by the PITHEM were replaced by emission rates calculated by EFT v7 and v8 to model emission rates released by the Baseline and 2030 scenarios. EFT v8 does not support the Baseline year of 2014; it supports years 2015 to 2030. Thus, EFT v7 was needed to model the Baseline year. The following section demonstrates how vehicular emissions were dispersed.

5.7.3 Dispersion Modelling

The mapping of air pollution can be performed by means of approaches such as spatial interpolation and dispersion modelling, which simulates the dispersion of emissions in the air (Briggs *et al.*, 1997). Dispersion simulation involves the creation of a dynamic model that can estimate changes in pollution concentrations over time using information related to the source of pollution, such as the patterns of traffic flow, meteorological conditions and background emissions (Briggs *et al.*, 1997). Outcomes from a dispersion model are concentrations of selected pollutants at selected receptors that vary in spacing. Additionally, the concentrations of dispersed emissions can be expressed for each of the 8760 hours of the year (for short-term exposure), which takes a considerable amount of time, and/or annual concentration (for long-term exposure) with a takes comparatively short time.

In general, estimations of ambient contamination concentrations using dispersion modelling are performed based on the Gaussian plume equation (Holmes and Morawska, 2006), which can give acceptable in a reasonable time frame for the analysis (Briant *et al.*, 2013). The distribution of a Gaussian plume in the horizontal and vertical directions for a point source is shown in Figure 5-13.



Source: Aliyu *et al.* (2015)

Figure 5-13: Gaussian Plume Model scheme for a point source

The concentration of an air pollutant at any given receptor C(x, y, z) is given by the following equation:

$$C(x, y, z) = \frac{Q}{2\pi\sigma_y(x)\sigma_z u} \times e^{-\frac{1}{2}\left(\frac{y}{\sigma_y(x)}\right)^2} \times \left(e^{-\frac{1}{2}\left(\frac{z-H_e}{\sigma_z(x)}\right)^2} + e^{-\frac{1}{2}\left(\frac{z+H_e}{\sigma_z(x)}\right)^2}\right)$$

where:

Q:	Source strength;
σ_y, σ_z:	Deviations that describe the crosswind and vertical mixing of the contaminant which are estimated according to stability class or travel time from the source;
x:	The downwind, where y is the crosswind;
u:	Wind speed at the height h of the release; and
z:	The vertical direction.

In addition, steady-state conditions are assumed in the Gaussian plume model in situations where plumes do not interact with each other (Holmes and Morawska, 2006). Furthermore, the road sources (which are line sources) are separated into smaller line sources so that they will be treated as point sources. The contributions of those points will be summed up to form a line source (Briant *et al.*, 2013).

5.7.4 Dispersion Modelling Using ADMS-Urban

The Atmospheric Dispersion Modelling System (ADMS-Urban) model is a comprehensive system for modelling air quality in large urban areas, cities and towns. ADMS-Urban is able to simulate the atmospheric dispersion of contaminant emissions from different sources such as road traffic, industry and domestic. The first version of ADMS-Urban was developed by Cambridge Environmental Research Consultants in the UK in 1990. Additionally, the ADMS-Urban model is considered to be a useful tool in the UK National Air Quality Strategy process. In the UK, the software is used by over 80 local authorities to undertake reviews and assessments of air quality under the Local Air Quality Management Programme and for the development of air pollution action plans and remedial strategies (CERC, 2018). In addition, an assessment and review of air quality in Newcastle City Centre, the Quayside and Jesmond Road was undertaken in 2005 using the ADMS-Urban by the University of the West of England on behalf of Newcastle City Council (NCC, 2005). Furthermore, Newcastle University owns a valid licence for use of the ADMS-Urban which allowed it to be utilised to model the air quality in the study area.

The modelling of emissions dispersion in the ADMS-Urban model is based on Gaussian plume distribution in estimating contaminant levels. In brief, the inputs are related to:

1) Source parameters, such as road, industrial and grid sources;
2) Time-varying emission factors related to diurnal change in the associated emissions;
3) Meteorology data, including roughness, latitude, wind speed, wind direction, cloud cover;
4) Background pollution concentrations; and
5) The locations of receptor points or grids.

Due to its effectiveness, popularity and availability, the ADMS-Urban was utilised in this research to model the emissions dispersion associated with the Baseline model.

5.7.5 ADMS-Urban's Results for the Emissions Dispersion Modelling of the Baseline

Prior to running ADMS-Urban, vehicular emissions rates were modelled by both the PITHEM and EFT v7. Subsequently, the contents of the PITHEM output file that comprises details of emission rates were replaced with emission rates calculated by EFT v7 because it is the latest version of EFT that supports 2014 (whereas EFT v8 supports 2015 and beyond), whilst PITHEM was built on EFT v5. In addition, the sites the stations where pollution is monitored in the pilot area were specified as receptors in ADMS-Urban. Pollution concentrations recorded by the monitoring stations were compared with their corresponding concentrations as modelled by ADMS-Urban to validate the performance of the Baseline model in estimating pollutant concentrations. Furthermore, background emissions concentrations in ADMS-Urban were set to zero, because the concentrations of background emissions were added uniformly on top of modelled concentrations at all receptors regardless of the variation in the background emissions concentrations over the receptor sites in the study area.

The required inputs were entered into the ADMS-Urban model, such as meteorological data, receptor coordinates, background emissions and the amended PITHEM outputs. Pollution concentrations at the monitoring stations modelled by the ADMS-Urban are illustrated in Table 5-9, where observed pollution concentrations recorded at monitoring stations are used to validate the ADMS-Urban outputs as described in section 5.7.8.

Table 5-9: Modelled annual mean concentrations released by vehicular emissions, ADMS-Urban outcomes

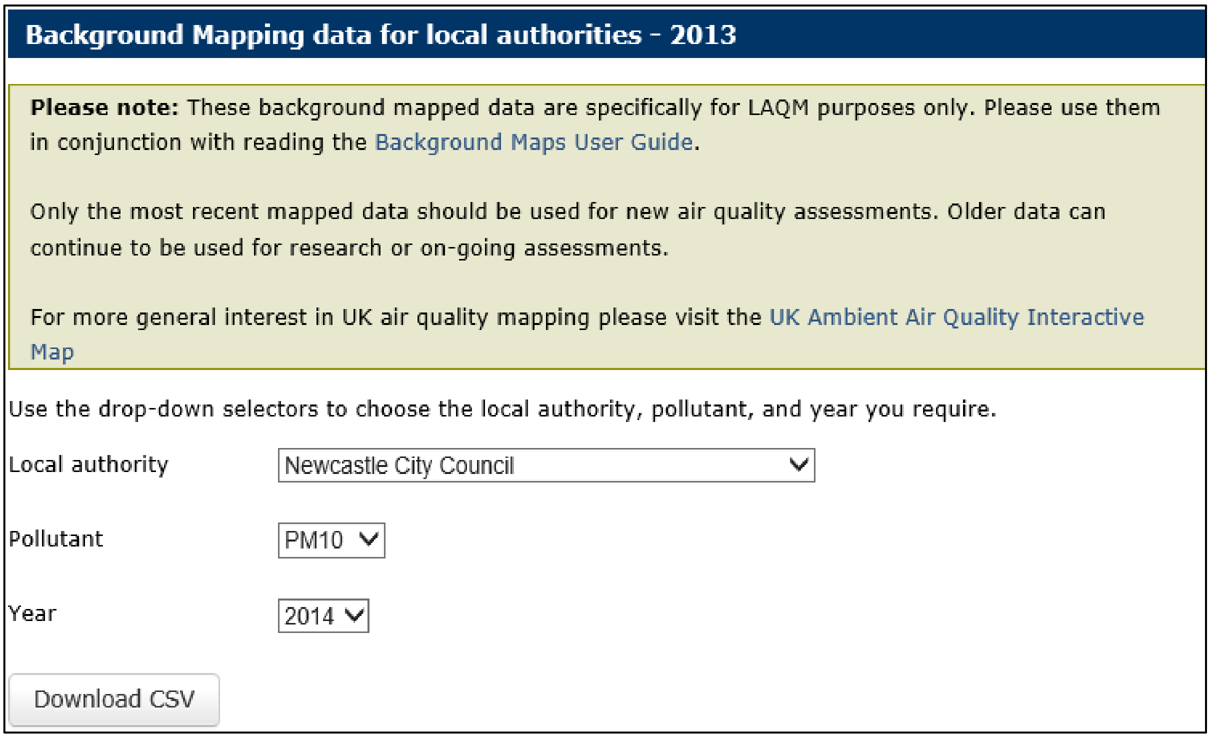
Receptor name	NO_x (μg/m^3)	$PM_{2.5}$ (μg/m^3)	PM_{10} (μg/m^3)
St. Mary's Place (AURN)	8.9	0.3	0.4
Jesmond Road, Cradlewell	26.7	1.0	1.5
Percy Street	39.7	1.0	1.3
Swan House, Pilgrim Street	37.6	1.0	1.5
High Street, Gosforth	16.6	0.5	0.8
Lychgate Court	7.4	0.3	0.5
A1 Dunston	10.6	0.6	0.9

These concentrations were modelled for the monitoring sites representing the contributions of vehicular emissions only as sources of pollution. Other background sources of emissions should then be considered in order to correctly forecast pollution levels.

5.7.6 Background Map of Contaminant Levels

Emissions dispersion related to road traffic was modelled and concentrations at selected receptors estimated via the ADMS-Urban. However, the contributions of other sources including industry, aviation and the domestic sector have not yet been taken into account. Thus, it is important to consider the levels of background emissions when modelling air quality.

The DEFRA (2016b) has published a map of the background concentrations of specific pollutants and the relevant sources of pollution. These concentration maps are published to assist local authorities when undertaking reviews and assessments of local air quality. Details of background pollution levels were obtained directly from the DEFRA website, as the snapshot in Figure 5-14 illustrates.



Source: DEFRA (2016b)

Figure 5-14: DEFRA webpage for the downloading of background emissions concentrations

Background pollution levels are given by DEFRA on grids comprising nodes that are spaced 1 km × 1 km. For example, DEFRA estimates pollution concentrations for a site every 1 km in the horizontal direction. Therefore, for each receptor site assigned in ADMS-Urban, the background emissions concentrations were selected based on the closest node, since DEFRA does not provide the locations of background emissions that exactly match the sites of monitoring stations.

Background emissions concentrations include all sources of pollution, including the vehicular emissions modelled in the ADMS-Urban model. Hence, the contribution of those vehicular emissions must be removed from the background emission levels. Generally, background emissions are broken down into no-road emissions (No-road) and road emissions (Only-road). Given that vehicular emissions related to minor roads are not represented in the Baseline, their contribution should be added to 'No-road' to produce 'No-road + Minor' emissions concentrations. Table 5-10 shows the annual means of the background $PM_{2.5}$ and PM_{10} concentrations.

Table 5-10: Background annual mean concentrations of $PM_{2.5}$ and PM_{10} sourced from 'No-road + Minor' sources

Site Name	$PM_{2.5}$ (μg/m^3)	PM_{10} (μg/m^3)
St. Mary's Place (AURN)	9.5	13.1
Jesmond Road, Cradlewell	9.0	12.5
Percy Street	9.2	12.5
Swan House, Pilgrim Street	9.5	13.1
High Street, Gosforth	8.3	11.4
Lychgate Court	9.3	12.9
A1 Dunston	8.6	12.2

In relation to NO_x and NO_2 background concentrations, DEFRA provides only total concentrations of NO_2, whilst NO_x concentrations are broken down generally into no-road emissions (No-road) and road emissions (Only-road). The contribution of minor road concentration was added to no-road concentration to produce 'No-road + Minor' emissions, as seen in Table 5-11, because emission rates modelled by the ADMS-Urban do include the involvement of the minor road emissions.

Table 5-11: Annual mean of background concentrations of NO_x and NO_2

Receptor name	NO_x (μg/m^3)				NO_2 (μg/m^3)
	Road-only	No-road	Total	'No-road + Minor'	All
St. Mary's Place (AURN)	22.6	21.1	43.7	31.1	28.1
Jesmond Road, Cradlewell	18.6	16.4	34.9	26.0	23.5
Percy Street	18.9	19.2	38.0	31.0	25.1
Swan House, Pilgrim Street	22.6	21.1	43.7	31.1	28.1
High Street, Gosforth	12.2	13.3	25.5	21.8	17.9
Lychgate Court	21.3	19.4	40.8	27.7	26.6
A1 Dunston	15.5	11.8	27.3	17.7	19.1

However, mapped background NO_2 concentrations should be revised, as they represent total background concentrations including vehicular emissions. The background concentration of NO_2 should be related to the contribution of all NO_x sources except for vehicular emissions, which were modelled in the ADMS-Urban. Here, the contribution of road transport in sourcing NO_x, which are the emissions modelled in ADMS-Urban should be subtracted from the total NO_x. Subsequently, the background NO_2 levels were revised based on the removal of the contribution of vehicular emissions for NO_x. In order to assist local authorities to carry out assessments and reviews of air quality, DEFRA has prepared a tool to revise NO_2 background emissions. The 'NO_2 Adjustment for NO_x Sector Removal Tool', can remove the involvement of a NO_x sector and adjust the corresponding level of NO_2. Revised concentrations of background NO_x and NO_2 are presented in Table 5-12 after eliminating the contribution of vehicular emissions concentrations.

Table 5-12: Revised concentrations of background emissions of NO_x and NO_2

Receptor	Revised NO_x (µg/m^3)			Revised NO_2 (µg/m^3)
	Road Traffic Sectors	Non-road Sectors	Total	After NO_x removal
St. Mary's Place	10.0	21.1	31.1	20.9
Jesmond Road	9.6	16.4	26.0	18.0
Percy Street	11.8	19.2	31.0	20.9
Swan House	10.0	21.1	31.1	20.9
High Street, Gosforth	8.5	13.3	21.8	15.5
Lychgate Court	8.3	19.4	27.7	18.9
A1 Dunston	5.9	11.8	17.7	12.8

Modelled concentrations of NO_x estimated by the ADMS-Urban model need to be converted in to NO_2 concentrations. DEFRA has developed the 'NO_x to NO_2 Calculator' to allow the modeller to derive NO_2 from NO_x levels predicted by the modelling of emissions from roads. Thus, the annual mean modelled levels of NO_x produced by ADMS-Urban (given in Table 5-9) were converted to their related NO_2 levels, all the results are presented in Table 5-13 which demonstrates the modelled values of NO_2 due to all sources of emissions and road sources. For example, at St. Mary's Place, ADMS-Urban forecasts that NO_x level is 8.8 µg/m^3. By utilising the 'NO_x to NO_2 Calculator', this level of the NO_x is equivalent to 4.5 µg/m^3 of NO_2. Moreover, taking the revised background concentration into account, the final NO_2 concentration would be 25.3 µg/m^3 at this site. The final modelled concentrations of NO_2 values are then to be compared with the corresponding real-world concentrations measured at the same sites in order to verify the performance of the Baseline in predicting pollution concentrations.

Table 5-13: Final annual mean modelled NO_2 concentrations

Receptor ID	Road increment NO_x (μg/m^3)	Background NO_2 (μg/m^3)	Road NO_2 (μg/m^3)	Final NO_2 (μg/m^3)
St. Mary's Place	8.9	20.9	4.5	25.3
Jesmond Road	26.7	18.0	12.9	31.0
Percy Street	39.7	20.9	18.4	39.4
Swan House	37.3	20.9	17.4	38.3
High Street, Gosforth	16.6	15.5	8.4	23.8
Lychgate Court	7.4	18.9	3.7	22.7
A1 Dunston	10.6	12.8	5.5	18.3

5.7.7 Concentrations of Monitored Pollutant

Local authorities are required to undertake reviews and assessments of air quality in their areas and to take any necessary action to enhance the quality of air, particularly if the national objectives in relation to air quality are not achieved. This is done in order to conform to the requirements for local air quality management set out in Part IV of the Environment Act 1995. Therefore, Newcastle and Gateshead City Councils publish Air Quality Annual Status Reports (ASR), which contain pollution levels recorded at their monitoring sites. The recorded annual mean pollution levels are shown in Table 5-14 (Gateshead Council, 2016; Newcastle City Council, 2016). These monitored values are compared to their corresponding modelled values to ensure the validity of the Baseline framework in predicting pollutant concentrations in the study area.

Table 5-14: Monitored pollutant concentrations in Newcastle and Gateshead in 2014

Site Name	Annual mean NO_2 (μg/m^3)	Annual mean $PM_{2.5}$ (μg/m^3)	Annual mean PM_{10} (μg/m^3)
St. Mary's Place (AURN)	30	9.7	12.6
Jesmond Road, Cradlewell	35.9	-	18.6
Percy Street	42.6	-	-
Swan House, Pilgrim Street	45.7	-	-
High Street, Gosforth	23.2	-	15.3
Lychgate Court	32.2	9.4	-
A1 Dunston	30.6	9.7	-

5.7.8 Validation of Emissions Dispersion

The values of background mapping 'No-road + Minor' concentrations of $PM_{2.5}$ and PM_{10} mentioned in Table 5-10 were added to the modelled concentrations presented in Table 5-9 to represent the final annual mean modelled $PM_{2.5}$ and PM_{10} concentrations, as demonstrated in Table 5-15. This particular Table also shows a comparison of the annual means of observed and final modelled $PM_{2.5}$ and PM_{10} concentrations. The comparison indicates excellent performance in relation to the air quality model, given that the difference between them is less than 5% in predicting $PM_{2.5}$. The model overestimated the value of PM_{10} at St. Mary's Place by 7% and underestimated it at both Jesmond Road and Gosforth High Street with percentage errors of 25% and 20% respectively. All predicted results are within 25% of the monitored concentrations, which conforms to the DEFRA's Local Air Quality Management Technical Guidance (TG16) (DEFRA, 2016d, pp. 7-130).

In addition, the Baseline model gives root mean square error (RMSE) values of 0.3 $\mu g/m^3$ and 3.3 $\mu g/m^3$ in predicting $PM_{2.5}$ and PM_{10} concentrations respectively. These values are within 25% of the air quality objective values of 12 $\mu g/m^3$ in $PM_{2.5}$ (Scotland target) and 40 $\mu g/m^3$ in PM_{10} concentrations (DEFRA, 2016d, pp. 7-134). This error is equivalent to 0.2% and 2.3% of deaths brought forward, and the COMEAP (2009) as well as Carey *et al.* (2013) have documented that exposure to increases of 10 $\mu g/m^3$ of $PM_{2.5}$ and PM_{10} in all probability will increase mortality by 6% and 7% respectively.

Table 5-15: Observed versus final modelled annual mean concentrations of $PM_{2.5}$ and PM_{10}

Site Name	$PM_{2.5}$ Annual mean concentration (µg/m³)		Deviation of modelled from observed $PM_{2.5}$	PM_{10} Annual mean concentration (µg/m³)		Deviation of modelled from observed PM_{10}
	Observed	Modelled		Observed	Modelled	
St. Mary's Place	9.7	9.7	0.41%	12.6	13.5	-7.29%
Jesmond Road	-	-	-	18.6	14.0	24.75%
Percy Street	-	-	-	-	-	-
Swan House	-	-	-	-	-	-
High Street, Gosforth	-	-	-	15.3	12.2	20.33%
Lychgate Court	9.4	9.6	2.04%	-	-	-
A1 Dunston	9.7	9.2	5.38%	-	-	-

N.B. A positive sign indicates overestimation, while a negative sign signifies underestimation

In relation to the validation of the modelling of NO_2 concentrations, a comparison of monitored and modelled NO_2 levels was carried out. At the High Street, ADMS-Urban succeeded in modelling NO_2 concentrations with a difference of only 2.67%. In addition, the ADMS-Urban estimates for NO_2 at all sites, are within 30% compared to the measured concentrations, although they should be below the 25% threshold to be consistent with DEFRA guidelines (DEFRA, 2016d, pp. 7-130). The range difference at the A1 Dunston location of 40% can be explained as there were specific characteristics at this site which caused not only prolonged recurrent congestion but also several accidents. Road works at this site forced vehicles to be driven at a low speed which made levels of emitted NO_2 escalate. Reports from Gateshead confirmed these special characteristics and Gateshead Council (2014, p. 35) reported that:

'*The stretch of A1 from Lobley Hill to Dunston is currently the third most congested link on the national motorway and trunk road network and the most congested regional link in terms of delay. Very heavy congestion occurs daily in both directions during the morning and* *evening peak periods. Accident rates on this road are high; with rear end shunting a prevalent accident type in queuing traffic'.*

Obviously, this explains the higher observed NO_2 compared to the modelled concentrations at the Dunston site.

Given this explanation the majority of results are within 25%, as seen in Table 5-16, which complies with TG16 published by DEFRA, that sets out some of the common steps to be considered to support local authorities in modelling air quality (DEFRA, 2016d, pp. 7-130). Moreover, for the modelling of NO_2 sourced from road traffic, the same methods can be applied to particulate matter in steps:

- No systematic underestimation or overestimation in predictions
- Forecasting at sites where monitoring demonstrates that levels are close to the objective and show a good comparison
- Most results are within 25% (as a minimum - preferably within 10%) of monitored concentrations

In addition, the Baseline model achieves RMSE values of 7.1 $\mu g/m^3$ in modelling NO_2 concentrations, which is within 25% of the air quality objective threshold of 40 $\mu g/m^3$ in NO_2 concentrations (DEFRA, 2016d, pp. 7-134).

These steps confirm the similarity to the results obtained from ADMS-Urban. Thus, the comparison between monitored and observed NO_2 concentrations indicates good validation.

Table 5-16: Performance of modelling of NO_2 concentrations

Receptor ID	Modelled NO_2 ($\mu g/m^3$)	Observed NO_2 ($\mu g/m^3$)	Deviation of modelled from observed
St. Mary's Place	25.3	30	-15.53%
Jesmond Road	31.0	35.9	-13.65%
Percy Street	39.4	42.6	-7.61%
Swan House, Pilgrim Street	38.3	45.7	-16.24%
High Street, Gosforth	23.8	23.2	+2.67%
Lychgate Court	22.7	32.2	-29.63%
A1 Dunston	18.3	30.6	-40.29%

The traffic flow modelling of the Baseline is completed in Section 5.5, whilst the emissions modelling is completed in this section. Additionally, these models finalise the Baseline framework that can simulate vehicular emissions and their dispersion.

5.7.9 Sources of Uncertainty and Differences Between Observed and Modelled Concentrations

The most appropriate method for assessing pollution levels is by direct measurement at Automatic Urban and Rural Network (AURN) stations, which can ensure a high level of precision. However, when measurements of pollution over a large area are essential, particularly in calculating annual hourly concentrations of pollutants, this direct method can be both time consuming and expensive. Therefore, this research has modelled the pollution concentrations sourced from vehicle flow data and other sources taken from the DEFRA database. Thus, for this research a succession of models (for traffic flow, vehicular emissions and the dispersion of emissions) were developed and used to estimate pollution concentrations for the Baseline scenario and therefore to forecast levels of air pollution for future scenarios by means of air quality models. Particular challenges are ensuring that uncertainty and bias are addressed in the modelling stages of this research in order to estimate concentrations of NO_2, PM_{10} and $PM_{2.5}$ from emissions, within a known level of statistical significance.

The first challenge was to estimate parameters related to the traffic flow of vehicles such as composition, speed and volume for the reason that these parameters influence the estimation of emission rates.

Coupled with the traffic flow parameters are the fleet characteristics engine size, fuel, age (Euro class), and for heavy goods vehicles' laden weight and for buses number of passengers, any afterburn treatments and level of maintenance also affect emissions. For example, diesel cars retrofitted with catalytic converters to reduce the amount of primary NO_2 from emissions. Occasionally due to a lack of car maintenance, failure takes place in the performance of catalytic converters in either petrol and diesel vehicles resulting in an increase in the amounts of NO_2, PM_{10} and/or $PM_{2.5}$ emissions.

The second challenge was to consider the impact of road gradient, truck loading degree (such as 50% loading and 100% loading) and road congestion. These factors were not facilitated in the Emissions Factors Toolkit (EFT) which was used to calculate rates of vehicle emissions. Therefore, it can be argued that the gradient in one direction (uphill) on a road would be opposite in the other direction (downhill) on the same road, which consequently can be assumed to cancel each other out through creating an equivalency. This of course does introduce some degree of uncertainty.

The third challenge was to assess the sensitivity of the emissions dispersion model to factors such as meteorological conditions. Meteorological conditions data for 2014 were obtained from the Met Office for Newcastle monitoring station located near to Newcastle Airport. Subsequently, these data were used in ADMS-Urban to model air quality for Newcastle and Gateshead assuming that the data some 4 km distant is representative of the whole of Newcastle and Gateshead. A common practice in local authorities is to perform the model run assuming a previous "dirty" and a "clean" year to produce a range of potential outcomes.

An additional factor that has an influence on emissions dispersion is the efficacy of the ADMS-Urban in modelling the physics of the dispersal of emissions. Generally, ADMS-Urban model tends to under predict pollutant levels when compared to actual measurements as found in literature and summarised in Table 5-17. For example, Briant *et al.* (2013) determined a monthly mean value of 22.5 $\mu g/m^3$ NO_2 compared to 9.6 $\mu g/m^3$ NO_2 as modelled by ADMS-Urban in a summer campaign conducted across 62 sites in Paris. However, in the winter campaign, the researchers observed a monthly mean value of 35.2 $\mu g/m^3$ NO_2 compared to 19.4 $\mu g/m^3$ NO_2 as modelled by ADMS-Urban, which shows that the model has a tendency to under predict air pollutant levels, compared to the passive diffusion tubes given that the emission factors employed in the model were taken from COPERT 3.

Table 5-17: ADMS-Urban validation studies

Study	Site	Vehicle emissions factors used	Validation dataset	R^2	Validation conclusion
Briant *et al.* (2013)	Paris region, France	COPERT 3	62 NO_2 passive diffusion tubes	73%	Model under estimates
Peace *et al.* (2004)	Greater Manchester, UK	TRL/DfT functions	NO_x and NO_2 at 12 continuous fixed-site monitoring stations	88%	Model under estimates particularly at the roadside site
Dėdelė and Miškinytė (2015a)	Kaunas, Lithuania	Design Manual for Roads and Bridges (DMRB 1999)	40 NO_2 Ogawa passives samplers	73% - 79% (depending on season)	Model over estimates
Dėdelė and Miškinytė (2015b)	Kaunas, Lithuania	Design Manual for Roads and Bridges (DMRB 1999)	NO_2 at 4 continuous fixed-site monitoring stations	56% - 91% (depending on site type)	Model under estimates except at the residential site
Carruthers *et al.* (2003b)	London, UK	Not available	NO_x and NO_2 at 24 continuous fixed-site monitoring stations	67%	Model under estimates particularly at the roadside site
de Hoogh *et al.* (2014)	Bradford, UK; London, UK; Barcelona, Spain	DMRB 1999 for Bradford	40, 27 and 40 NO2 Ogawa passives samplers	55%, 72% and 57%	Model under estimates

In Greater Manchester, using the UK emission factors sourced from TRL/DfT functions for 2001, the ADMS-Urban model was set up and validated at 12 continuous fixed-site monitoring stations (Peace *et al.*, 2004). Ten stations were categorised as urban background, one as roadside and one as a sub-urban station. The validation results indicated that the model under-predicted NO_x and NO_2 levels at all sites except the suburban site. The R^2 value of the model was calculated to be 88%. Furthermore, ADMS-Urban was used to model NO_2 levels in Kaunas, south-central Lithuania, and the modelled concentrations were validated against observations from 41 Ogawa passive samplers (Dėdelė and Miškinytė, 2015a). The emissions factors used were taken from the DMRB emissions factor dataset, and built into the dispersion model. However, the vehicle fleet was assigned an average age of 14 years, and a large number of cars on the road did not have catalytic converters. In this research, the ADMS-Urban estimates were lower than the average observed NO_2 concentrations. In most cases the model has a tendency to under-predict maximum concentrations and over-predict minimum concentrations. Likewise, Dėdelė and Miškinytė (2015b) conducted a comparison of modelled NO_2 concentrations and data from four continuous air quality monitoring stations in Kaunas. At two traffic stations and the background station, the modelled average concentrations of NO_2 were lower than the measured value, whilst the opposite trend was shown at the residential site. Carruthers *et al.* (2003b) set up and validated ADMS-Urban against measurements of NO_x and NO_2 from 24 fixed-site monitoring stations in London. The researchers proposed that the model generally tended to under-predict annual average NO_x and to a lesser extent NO_2 concentrations, particularly at the roadside. For instance, observed and modelled annual average NO_x levels at the 10 roadside stations were 115 ppb and 99 ppb respectively. Observed and modelled annual average of NO_2 at the 10 roadside stations were 35 ppb and 33 ppb respectively. In a wider validation study, de Hoogh *et al.* (2014) used and validated 13 dispersion models for different European study areas. At 3 of these areas, Bradford, London and Barcelona, an ADMS-Urban model was used and was validated against 40, 27 and 40 monitoring sites for NO_2 respectively. It was found that the median concentrations were under-predicted. Generally, the observed and modelled NO_2 concentrations correlated well with median Pearson correlation coefficients of 0.74, 0.85 and 0.75, for Bradford, London and Barcelona respectively (de Hoogh *et al.*, 2014).

On the whole when quality assured data is used as input to models good quality outputs are generated. However, as with all models they are not perfect and outputs should be carefully scrutinised. It is important to be aware of, and explain outliers in any model run and err on the cautions side when making policy decisions. Given that in general models underestimate concentration levels when used to test scenarios, provided the metrics of the change are well defined, the estimated differences can be considered reliable.

5.7.10 Source Apportionment of Concentrations at Receptor Sites

The contribution of different vehicle classes to measured concentrations varies depending on the emissions released. The emissions of $PM_{2.5}$ and PM_{10} are higher for buses and cars. Therefore, fleet composition is important since more than 70% of $PM_{2.5}$ and PM_{10} emissions are attributed to bus flows on Percy Street, whilst at Lychgate Court in Gateshead the traffic flow of cars is responsible for more than half of the $PM_{2.5}$ and PM_{10} emissions, as Figure 5-15 demonstrates. Conversely, most NO_x emissions are released by bus flows. For example, on Percy Street in Newcastle City Centre, more than 75% is released by buses, as shown in Figure 5-16.

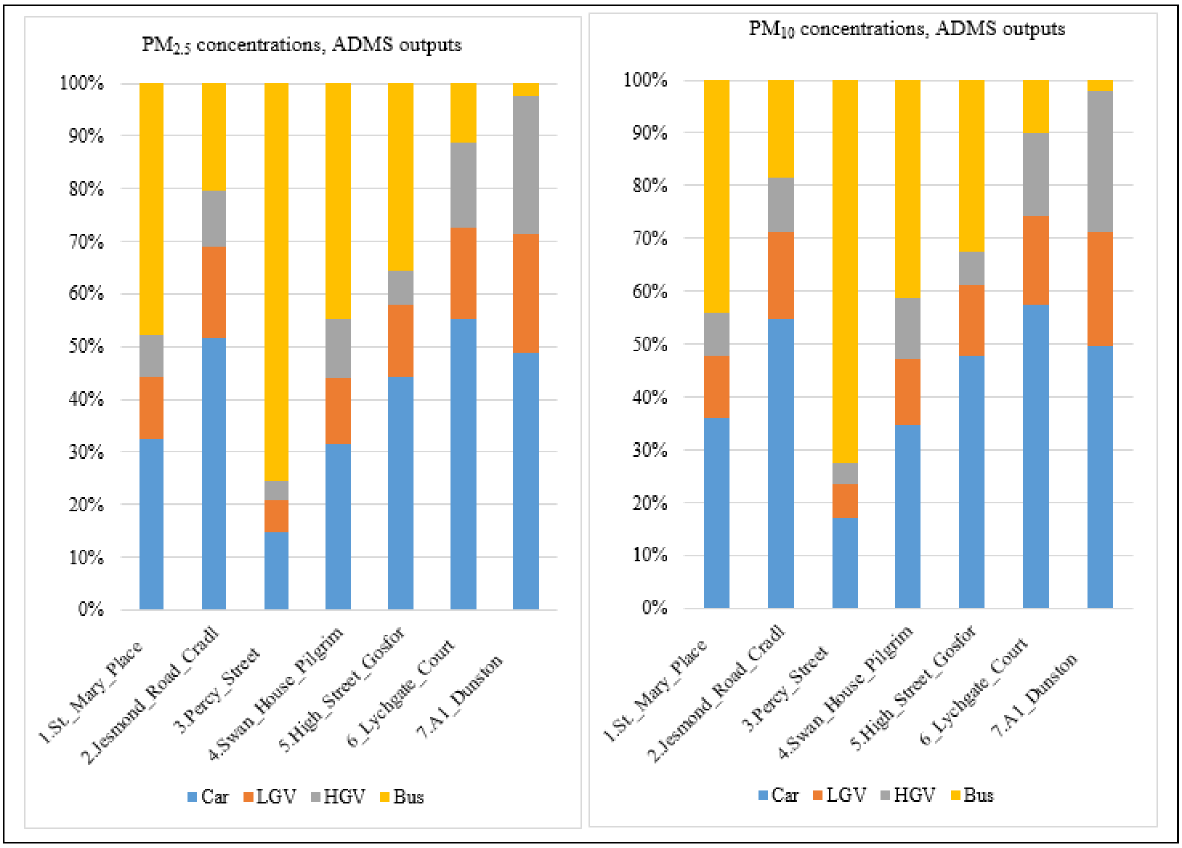


Figure 5-15: Normalised source apportioned concentrations by receptor locations, $PM_{2.5}$ (left) and PM_{10} (right)

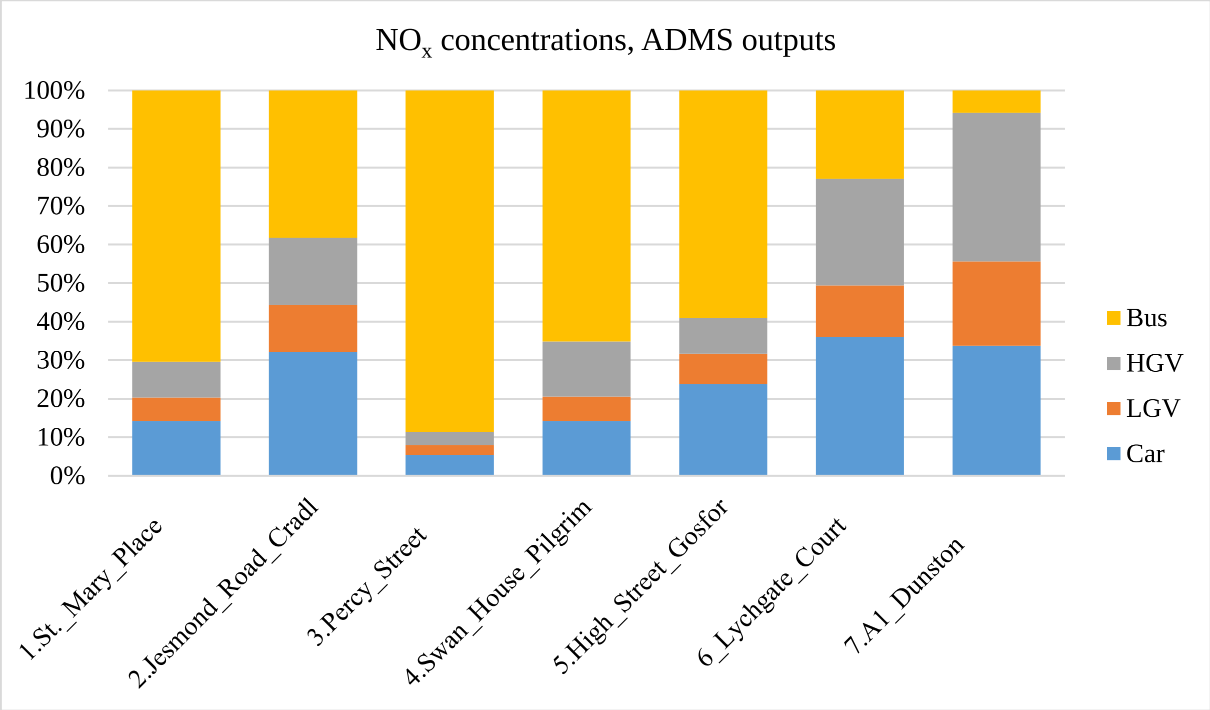


Figure 5-16: Normalised source apportioned concentrations of NO_x by receptor locations

5.8 Dispersion Modelling Issues

Although ADMS-Urban utilises only a small amount of computer memory, desktop memory was insufficient to complete the first trial run of the Baseline within a reasonable time. This is because the Baseline covers the entire study area using an output grid resolution of 100 m. As this run takes a considerable amount of computational time to complete, the boundaries of the study area were reduced to a lower-left OS coordinate of (422200, 558200) and an upper-right coordinate of (428600, 569600), in order to reduce the runtime as seen in Figure 5-17. The new boundaries of the study area cover areas where the most vehicular emissions are mostly released, in particular those areas which are declared Air Quality Managements Areas.

Background emissions published by DEFRA are provided at a large resolution of 1 km, whilst the ADMS-Urban modelling is based on a resolution of 100 m. Thus, background emissions were revised to produce a grid resolution of 100 m. Background emissions were assumed the same for 100 grids in one square kilometre (the spacing between grids is 100m×100m). However, due to the location of sources (other than traffic) the background levels are likely to create differences at the 100×100 grid scale. Whilst creating a limitation the magnitude of the error would be difficult to quantify. Furthermore, the DEFRA (2016d, pp. 7-129), stated: '*In many cases, background is based on national maps or local monitoring, adjustment of this* *component could result in unrepresentative estimates of the background concentrations across the area. Such adjustment could result in unrealistic estimates of different source contributions and may affect the outcome of source apportionment studies undertaken as part of further assessments and action plans*'. In the same context, Goodman et al. (2014, p. 63) highlighted that the choice of interpolation of the DEFRA's background emissions alters the underlying information.

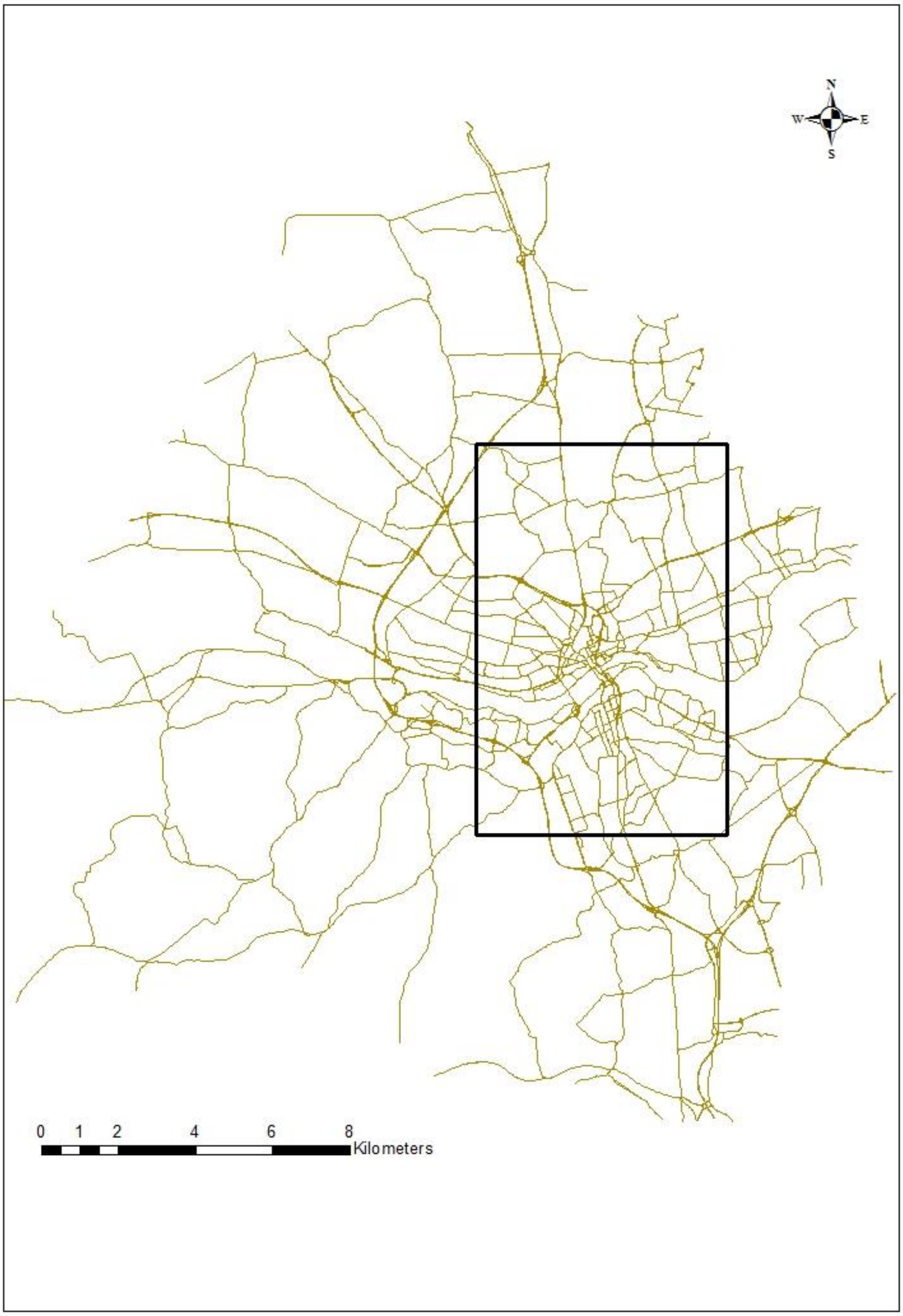
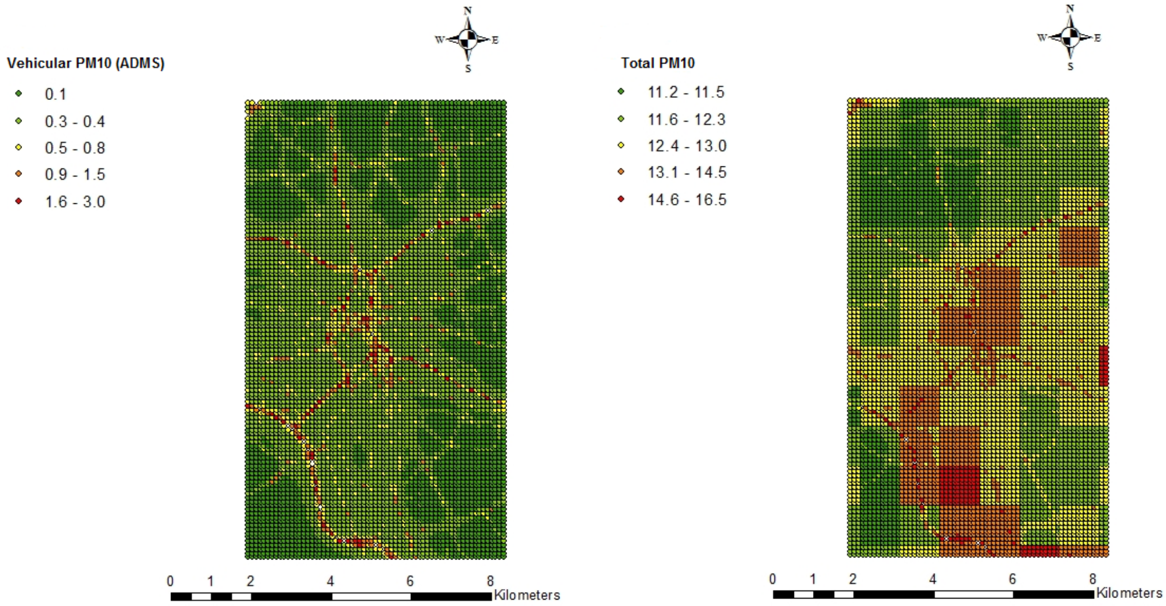


Figure 5-17: Dispersion model domain inside the rectangle

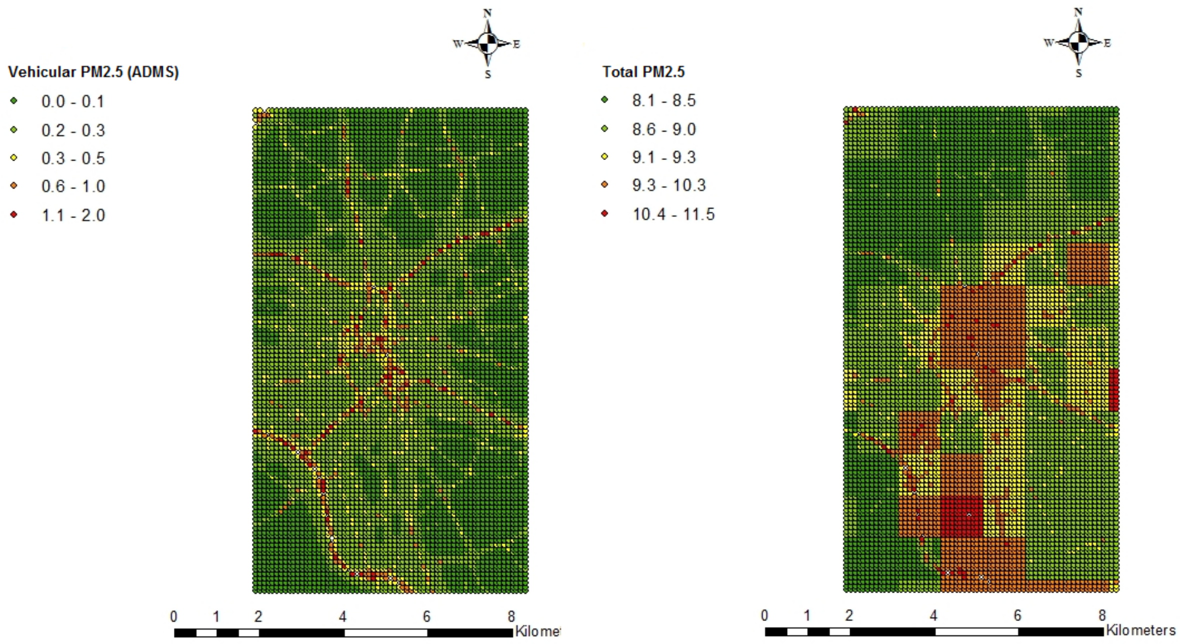
Modelled emissions of annual mean PM_{10} estimated by ADMS-Urban are shown in Figure 5-18 (left) and represent vehicular emissions, whilst the final annual mean modelled emissions including both vehicular and background emissions are shown in Figure 5-18 (right). Note the red square at the bottom, which is the site of an industrial area situated in Team Valley. It is perceived that the majority of PM_{10} is emitted by industrial sources.



All values are in μg/m^3

Figure 5-18: Vehicular PM_{10} concentrations (left), total PM_{10} concentrations (right)

Similarly, Figure 5-19 (left) indicates the modelled emissions of annual mean $PM_{2.5}$ concentrations estimated by the ADMS-Urban model due to vehicular emissions, whilst the final annual mean modelled emissions with vehicular plus background emissions are shown in Figure 5-19 (right). Again, it can be noticed that there is a red square at the bottom, which is the site of the industrial area situated in Team Valley. It is perceived that the majority of $PM_{2.5}$ is being emitted by industrial sources.



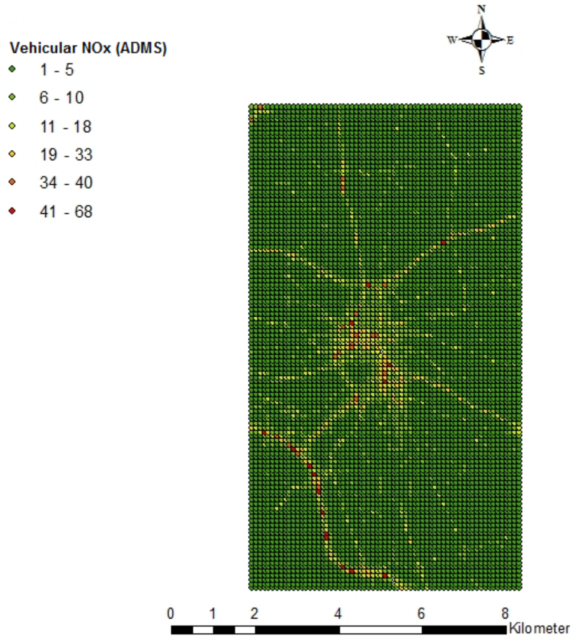
All values are in µg/m^3

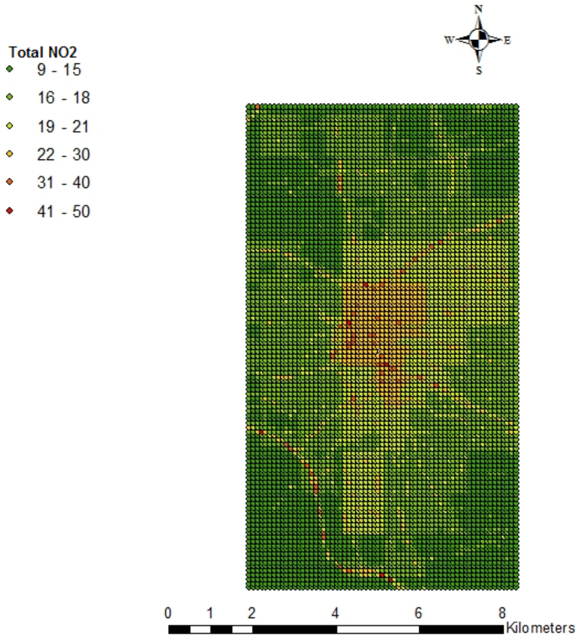
Figure 5-19: Vehicular $PM_{2.5}$ concentrations (left), total $PM_{2.5}$ concentrations (right)

In relation to annual mean modelled concentrations of NO_x and NO_2, rates of NO_x emissions released from vehicles were calculated and their dispersion subsequently modelled using ADMS-Urban. The distribution of NO_x concentrations attributed to vehicular emissions are shown in Figure 5-20 (left).

These NO_x emissions were converted to NO_2 levels utilising the ‘NO_x to NO_2 Calculator’ to give NO_2 levels due to vehicular emissions, whilst the contribution of vehicles in sourcing NO_2 emissions was removed from the map of NO_2 background emissions.

Subsequently, by adding NO_2 levels (converted from NO_x) to the revised values of NO_2 background concentrations, the final annual mean modelled NO_2 concentrations was produced as shown in Figure 5-20 (right).





All values are in µg/m^3

Figure 5-20: Vehicular NO_x concentrations (left), total NO_2 concentrations (right)

Four sites are estimated in 2014 to witness concentrations above the threshold air quality objective of 40 µg/m^3 in Newcastle City Centre and at the Tyne Bridge where a charging scheme is proposed to be installed (Tompkins, 2018).

5.9 Summary

The available transport model for 2010 was updated to the 2014 Baseline profile utilising guidance published by DEFRA. Criteria published by the Design Manual for Roads and Bridges were used for the calibration and validation of the traffic flow model by comparing modelled and observed traffic flows using TADU. The results for correlation coefficients (R) are very close to the target values set by DMRB.

Emissions rates from the Baseline were modelled by the PITHEM, to produce a series of files that comprise road characteristics such as length, sites of street canyons, and emission rates related to each road. Similarly, emissions rates were calculated by the latest version of the Emission Factors Toolkit (EFT v7) that supports the year 2014. Emission rates produced by EFT v7 were replacing those calculated by the PITHEM, given that the latter is based on EFT v5.

The dispersion of emissions was modelled utilising PITHEM outputs in the air quality model (ADMS-Urban) to estimate pollutant concentrations at pollution monitoring stations. By taking the background emissions into consideration, the comparison between the modelled and monitored concentrations indicated good validation, complying with TG16 limits.

The Baseline model is able to estimate pollutant concentrations at any selected receptor related to 2014. This facility is used in chapter 7 to predict pollution concentrations at the General Practitioner (GP) sites in 2014 are then be compared to the pollutant concentrations in 2030 resulting from 2030 scenarios developed in chapter 6.

CHAPTER 6

6. Development of Scenarios for 2030

6.1 Introduction

In chapter 5, the TPM in relation to 2010 was updated, calibrated and validated to provide the 2014 Baseline profile. In addition, the ability of the Baseline to model pollution concentrations were validated against real-world pollution levels recorded at monitoring stations. This chapter updates the Baseline model to 2030 to include a profile of the business-as-usual (BAU) scenario and a series of scenarios with a range of EV penetration levels. These 2030 scenarios include:

1. 'CCC': Committee on Climate Change proposal for 30% of cars and 38% of vans are electric;
2. 'E-Bus': Electrification of all buses;
3. 'E-Car': Electrification of all cars;
4. 'E-Car_E-Bus': Electrification of all cars and buses;
5. 'E-Car_E-LGV': Electrification of all cars and LGVs;
6. 'All-EV': Electrification of all vehicles.

The outputs of the 2030 scenarios beside the BAU scenario are compared with output from the Baseline in chapter 7.

6.2 Assumptions used in the RTF, TEMPro and NAEI to develop the BAU and CCC scenarios

In this section a review of data and assumptions used in the RTF calculator, TEMPro and NAEI to generate traffic growth factors used in developing BAU and CCC scenarios is presented.

6.2.1 Trip End Model Presentation Program (TEMPro)

The Trip End Model Presentation Program (TEMPro) is a program developed by the Department for Transport to provide projections of traffic growth used in transport models

(Clark, 2016). TEMPro produces output relating to the data from the National Trip End Model (NTEM).

The NTEM takes exogenous projections of population, employment and housing supply, and combines these with projections of car ownership and trip rates to forecast future numbers of trips by a person at a detailed spatial level and for different segments of the population. The structure of the NTEM is shown in Figure 6-1. The NTEM dataset and suite of models provides an initial forecast of travel demand (DfT, 2018b, p. 16) based on:

- The 2011 Census;
- ONS 2014-based population projections;
- Dwellings projections updated using local authority plans and annual monitoring reports;
- Employment projections updated using UK Commission for Employment and Skill (UKCES) 2012-based employment projections from the Working Futures project;
- The distribution of employment and workers by region in the base year 2011 (and hence in all years) updated using Workforce jobs statistics and the Labour Force Survey;
- A comprehensive updating and re-estimation of the National Car Ownership Model; and
- Re-estimated trip rates based on the National Travel Survey (NTS).

6.2.2 The National Transport Model (NTM)

The National Transport Model (NTM) is a modelling framework for England and Wales. The NTM has been updated to incorporate the latest available National Transport Survey (NTS) data and represents a 2015 base year. The NTM consists of a number of sub-models. Central to the modelling framework is the transport demand model (formally known as PASS1) and two separate supply models: the Fitting on Regional Growth Effects (FORGE) and the National Trip End Model Presentation Programme (TEMPro). Inputs to PASS1 are the outputs of FORGE and NTEM, which predict trip rates according to different population types (such as employed male in a household of and one car). Figure 6-1 shows the structure of the NTM.

The FORGE takes as input the car traffic growth measure obtained from the PASS1 as well as growth in freight traffic from the Multi-modal Freight Model (MFM). The effect of growth in car and freight traffic is predicted by considering historical traffic data, income (Gross Domestic Product (GDP)), fleet composition, fuel prices and efficiency, and average fuel cost per unit of time. If congestion grows and speeds on busy roads decrease, a series of elasticity-based rules are used to re-distribute traffic between links on different roads and periods of time. As traffic is shifted, the model framework recalculates speeds and provides new estimates of car journey costs. Iterations take place between the PASS1 and FORGE to exchange congestion impact and traffic growth estimates to allow recalculation of mode share. Outputs of FORGE are forecasts of levels of traffic, congestion and emissions. Furthermore, in order to represent the effects of road policies, adjustments to link volume/capacity ratios and speed/flow relationships also take place.

According to the Road Traffic Forecasts 2018 (DfT, 2018b, p. 30), the main inputs and assumptions for the NTM are:

1. NTEM 7.2, which includes:
    a. Projections from ONS for population.
    b. Licence holding and projections of car ownership.
    c. Declining trip rates from 2011 to 2016 which then remain constant from 2016 to 2050.
    d. Projections of employment.
    e. Projections of households and dwellings.
2. Forecasts from the Central Office for Budget Responsibility (OBR) for GDP, updated recently using the OBR's long-term economic determinants.
3. Central forecasts from the Department for Business, Energy and Industrial Strategy (BEIS) for fuel prices as stated in the WebTAG.
4. BEIS Manufacturing Index (Energy Demand Model).
5. WebTAG Values of Time.
6. Assumptions around the electric vehicle mileage split in line with WebTAG.

7. Vehicle fuel efficiency forecasts.

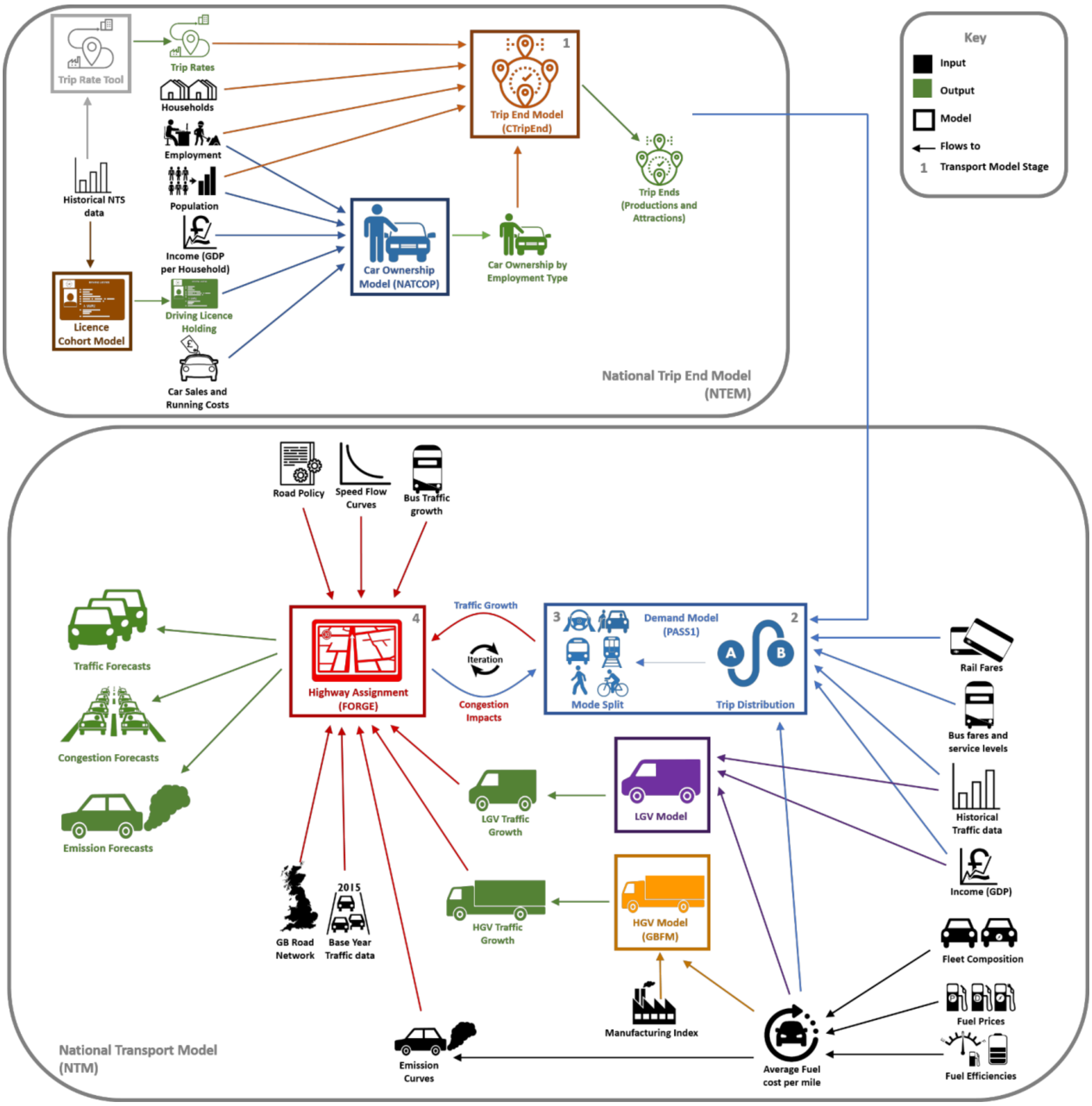
8. The modelled road network updated to include all fully committed schemes being implemented as part of the first Roads Investment Strategy (RIS1).

9. With regards to Ultra-Low Emission Vehicles (ULEVs):

    a. These forecasts include implemented and adopted policies only. These do not include future policies or government ambitions that have not been legislated; for example, possible future car and van CO_2 emissions regulations.

    b. The proportion of zero emission mileage is modelled as if these were electric vehicles. It captures distances driven by both battery electric and plug-in hybrid electric cars and LGVs.

    c. Existing taxation policies are assumed to be maintained. Fuel costs for ULEVs are significantly less expensive than for petrol and diesel cars.



Source: DfT (2018b, p. 13)

Figure 6-1: Diagram of the National Transport Model & National Trip End Model

6.2.3 National Atmospheric Emissions Inventory (NAEI)

The recent fleet composition projections data provided by the National Atmospheric Emissions Inventory were used in this research (NAEI, 2017). Also, they are the default fleet composition data incorporated in the Emissions Factor Toolkit (EFT). The latest vehicle fleet composition projections, referred to as Base 2018, are provided in two formats; for example, in the basic fleet split worksheet. According to the NAEI (2017), this worksheet provides the typical fleet mix according to vehicle type on different types of urban and rural roads and motorway in each of the Devolved Administrations. For the years up to 2016, data are based on the actual road traffic statistics published by the DfT, while for future years data are based on the DfT's Road Traffic Forecast for the UK outside of London. These are consistent with the DfT traffic projections published in the Road Traffic Forecasts 2015, but based on actual vehicle kilometres travelled in 2016. These data include a fuel split for cars and LGVs based on trends observed on different road types using Automatic Number Plate Recognition data from the DfT's Roadside Survey and projected turnover in the fleet based on revised new car sales projections provided by the DfT in February 2018, but using unchanged assumptions concerning the petrol/diesel sales mix and uptake rates of hybrid and electric cars according to data provided by the DfT (January 2017).

6.2.4 The Committee on Climate Change (CCC)

The Committee on Climate Change (CCC) has identified electric powertrains as the key technology to mitigate emissions of light-duty vehicles in support of the UK's target of an 80% reduction in greenhouse gases from 1990 levels by 2050. The CCC has commissioned Element Energy (2013) to carry out analysis of how to achieve the target of the 4th carbon budget by means of a shift to ultra-low emission vehicles, in particular electric vehicles (EVs), in order to decarbonise the transport sector through the 2020s. The Committee's analysis proposed that EVs should constitute 60% of new car sales by 2030 for transport decarbonisation reasons.

From an economic perspective, the CCC expected a reduction in technology costs leading to capital cost premiums for EVs to be offset by significantly lower running costs, which means that EVs are expected to compete with conventional cars on an economic resource cost basis.

The CCC has introduced a plan for the EV uptake pathway targets for 2020 and 2030 and presents the action targets that would be necessary in order to increase the market penetration of EVs. In addition, the Committee assumed the supply projection to be in the region of 30% annual growth in production capacity post-2015; the European EV production capacity is estimated at 2.36 million in 2020. The plan's assumptions made by the CCC for EV uptake pathway for cars and vans are as follows:

- Indicative 9% market share of EVs by 2020.
- EVs constitute 60% market share by 2030, with a technology split of 35% PHEVs and 25% BEVs.
- 100% market share of EVs by 2040.

EVs captured around 1% of UK sales in 2012, more than a decade after their introduction. In contrast, diesel cars had a 5% market share in the UK in 1990, 20 years after initially being introduced. The market share of diesel cars was around 15% by 2000, after which their adoption accelerated to almost 50% in 2010. This indicates the above-mentioned assumptions for future EV uptake could be achievable compared to diesel car uptake in the UK market if incentives could replicate.

In relation to charging infrastructure up to 2020, approximately 70% of UK drivers are expected to have access to overnight charging. The CCC expected that 80% of new houses with car owners built between 2015 and 2020 will be provided with a charging point and 95% will be post-2020. The rapid charging infrastructure coverage provides for at least 20% of UK drivers by 2020, 60% by 2025 and 100% by 2030.

6.3 Trend of EV Take-up in the UK

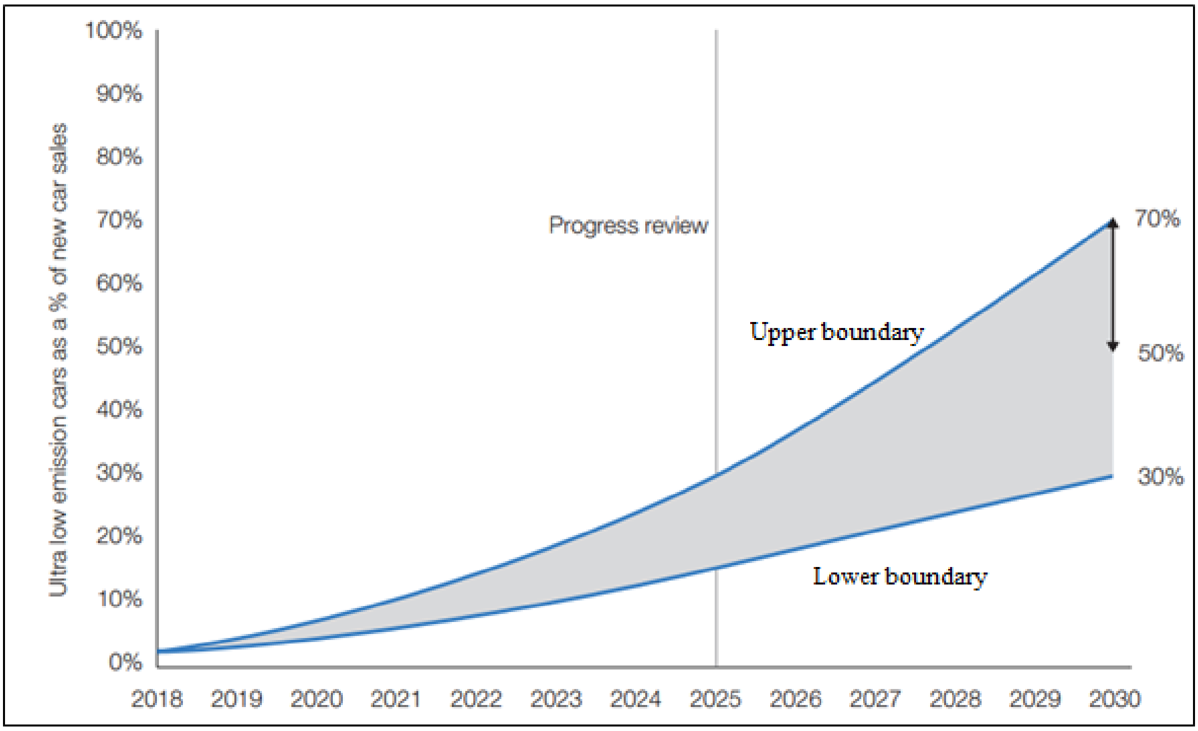
The UK government has set out plans and strategies to achieve cleaner road transport within the next decade. Top of the list of these strategies is the design and manufacture of ultra-low emission vehicles with nearly £1.5 billion of investment in this venture (DfT, 2018a). The UK aims to be a leader in the design and manufacturing of ultra-low emission vehicles, and for all new cars and vans to be 100% ultra-low emission by 2040. The UK government has made known in its desire to meet nitrogen dioxide (NO_2) limits plan by 2040 to stop the sale of new conventional petrol and diesel cars and vans, replacing them with 100% ultra-low emission vehicles by 2050 focusing on EVs (DfT, 2018a). This plan is not without its challenges. Consumers have exercised concern about the technology, high upfront costs of EVs, and the availability of infrastructure (e.g. charging points) specifically in the countryside and remote areas, not only within cities. These issues are currently a huge barrier to the uptake of EV according to DfT (2018a). Some of the major challenges of EV uptake in the UK will be discussed in the subsections to follows.

6.3.1 Adequate Vehicle Supply

The question of supply is very important and mainly concerning capacity and the performance of EVs compared to that of fossil fuel vehicles. There are hundreds of fossil fuelled car and van options on the market and only 38 cars which are eligible for a plug-in car grant. Furthermore, the supply to EVs for commercial use is particularly challenging because, currently, technologies for ultra-low emission HGVs are less developed than for cars and vans. Only nine out of the multitude of fossil fuel HGV manufacturers are eligible for the plug-in grant, all of the vehicles concerned are of limited capacity of 3.5 tonnes or less. More innovation and investment is required to bridge the gaps in technology, capacity, and performance if the UK is to complete the transition to ultra-low-emission or low emission EV cars, vans, and HGVs by 2040 (DfT, 2018a). Otherwise, if EV vehicles are rolled out solely on the basis of environmental concern above its practicality, sustainability, and affordability, it will prove unsustainable and quite costly to the economy and the public.

6.3.2 A Strong Consumer Base and the Appropriate Market Conditions

Currently this is a big challenge. While nearly 40% of consumers considering the purchase of a new car say they would like an EV. However, only 2% actually purchase one (DfT, 2018a, p. 9). Figure 6-2 illustrates the anticipated upper and lower boundary for the future uptake trajectory of ultra-low emission cars as a percentage of new car sales. The chart shows that by 2030, the uptake of EVs could be up to at least 70%. It can be argued that, if EVs are left to chart their own course on the free market, the UK may never reach its target of 100% ultra-low emission vehicles on the road by 2050. The DfT (2018a) have stated that "*the right incentives and policy framework will be vital to increasing the numbers of consumers who want to buy ultra-low emission vehicles*".



Source: Road to Zero Strategy (DfT, 2018a, p. 10)
Figure 6-2: Projections boundaries of new sales of ultra-low emissions cars as a proportion of all car sales

6.3.3 Lack of Knowledge about EV Technology

The novelty of EV technology creates a challenge to the uptake of EVs in the UK. Lyndhurst (2015) suggested that automotive consumers seem to have inaccurate preconceptions about performance, acceleration, top speed, purchase price, driving range, recharging time and operating costs of EVs, leading to low uptake. Hence, greater awareness and campaigns to educate the public about the benefits and performance of EVs almost on a par with internal combustion engine vehicles is imperative. The government has demonstrated its commitment to an ultra-low emission future for the UK by starting to collaborate with international partnership, industry, businesses, environmental groups, academia, devolved administrations, local government and consumers to create awareness of the benefits of ultra-low emission vehicles (ULEVs) (DfT, 2018a). Furthermore, government policies such as the plug-in car grant and the plug-in van grant have proven pivotal in the purchase decisions of 85% of ULEV customers (OLEV, 2013a). Other incentives provided by the government to enhance the attractiveness of ULEVs include: exemptions from or low rates for fuel duty for electric vehicles and hybrids respectively, exemptions from vehicle excise duty (VED) for alternative fuel vehicles that release less than 100g/km of CO_2; exemption from the London congestion charge, 3% lower company car tax (CCT) than for fossil fuel vehicles until 2020, and discounted parking and taxi licencing fees (OLEV, 2013a).

6.3.4 High Up-front Purchase Price and Limited Range

New technology is seldom inexpensive. By implication, consumers may not be willing to put forward the money to replace old technology. However, new technology usually outperforms older technology in almost every respect. In a free market, desire among consumers for technology with better performance drives them to buy new technology. Then, the increase in demand leads to mass production and economies of scale and, hence, brings a drop in price. Consumers are provided with innovative technology that outperforms the older technology at affordable prices and manufacturers still make a profit. Currently, however, EVs have not proven to outperform fossil fuel vehicles in any category except for emissions. For example, the limited range puts EVs at a disadvantage in terms of building a strong consumer base (OLEV, 2013a; Lyndhurst, 2015; DfT, 2018a). Range anxiety is at the top of the list of barriers, according to Lyndhurst (2015). Automotive consumers desire to be able to travel long distances when needed, as they associate vehicle ownership with a sense of freedom. The same consumers require that EVs are able to drive for long distances too. According to (Anable *et al.*, 2014), until the EVs improves to at least 150 miles, they will remain unattractive to consumers for use as a main car.

Most consumers purchase items and new technology for the value they can derive from it personally. This can be termed as the personal benefit. However, there is a societal benefit of this EV technology, but people seldom do things for the sake of society. According to the Office for Low Emission Vehicles (OLEV), consumers must be made aware of the societal benefits of EVs (such as improved air quality, and new jobs) which directly affects their lives and living (OLEV, 2013a). Advocates of intervention to launch technology such as OLEV into the free market state that the government provides all the necessary incentives; grants, subsidies, and policies to drive consumers adoption of EVs.

6.3.5 A Fit-for-Purpose Infrastructure Network

Infrastructure provision is a major concern for the mass adoption of ultra-low emission vehicles (OLEV, 2013a; Lyndhurst, 2015; DfT, 2018a). Consumers need assurance that with EVs they will be able to make the journeys they want to make (OLEV, 2013a). Bunce et al. (2014) suggested that up to about 60% of potential buyers of EVs feel that the public charging infrastructure is not adequate for them to consider buying an EV. Furthermore, 70% of respondents in Bunce et al. (2014) study, stated that the amount of time required to recharge an EV is a barrier to buying EVs (Lyndhurst, 2015). The duration of recharging time coupled with a lack of range amounts to levels of inflexibility and inconvenience that consumers are not ready to tolerate. Manufacturers, on the other hand, need there to be a sufficient number of EVs on the roads to justify investment in infrastructure. For the consumer experience of EVs to be improved, more infrastructure must be built so that EV drivers can easily locate and access charging stations that are inexpensive, efficient and reliable (DfT, 2018a). More infrastructure is needed, in terms of not only charging points (both at home and on-street) but also electricity capacity. As the uptake of EVs increases, it is imperative that the electric power network is capable of accommodating additional demand if EVs are to compete with, and eventually completely phase-out fossil fuel vehicles (OLEV, 2013a). This is an area where government intervention is needed.

6.3.6 Government Policy

A study of 349 drivers of EVs indicated that drivers adapt almost immediately to the vehicles and found them easy to drive and user-friendly (OLEV, 2013a). Although the UK is one of the leading pioneers in the uptake of ULEVs, it is still behind other major players in the market such as Germany, Norway, France, and the Netherlands. This, however, has provided the UK with a unique position of learning from the mistakes of those countries that are ahead in ULEV uptake. For example, although these countries have adopted measures which have enabled the hurdles to ULEV uptake to be overcome much more rapidly, this has come at significant cost to taxpayers and may have involved policy developments which would be challenging to replicate in the UK. OLEV (2013a) asserted that "*the policy challenge for Government and for industry here is to ensure that the objective of promoting ULEV uptake is balanced appropriately with other considerations, not least affordability*."

The achievement of the 2040 target for all new UK vehicle sales to be ULEVs is difficult to predict with any certainty. The uptake of EVs is determined by various factors, many of which are outside the control of government. According to OLEV (2013a), these factors include the speed and cost at which manufacturers can roll out ULEVs to the market; battery costs, fuel prices, residual values, purchase models (for example, battery leasing), and accessible home and on-street rapid charging facilities.

6.4 Business-as-Usual (BAU) Scenario

By 2030, the dominant standard class of conventional vehicle would be the Euro 6/VI (or a higher standard) which entered service in 2014 or later (DEFRA, 2017b; NAEI, 2017), when the allowable emission rate thresholds were notably narrowed and tightened up (EC, 2012). For instance, NO_x emissions from all diesel vehicles in Euro 6/VI legislation have been reduced significantly to approximately half in comparison with the previous Euro 5/V classes (EC, 2012). Nevertheless, the sixth Euro diesel models, from some original equipment manufacturers were discovered to have had their in-service emissions of NO_x in the UK fraudulently manipulated. An independent committee commissioned by the Department for Transport established that NO_x service emissions of light duty Euro 6 vehicles are causing emissions up to 6 times greater than the 80 mg/km official legislative NEDC laboratory (Department for Transport, 2016, pp. 22, 23). This might increase doubts regarding the merits of conventional vehicles in addressing environmental issues, leading to a fresh consideration of increasing EV uptake in future vehicle fleets as the key solution to environmental issues, as reflected the government's 'Road to Zero' strategy. The mitigating of global warming and lowering of NO_2 concentrations were the principal reasons for the announcement of the UK government that all new car sales in 2040 would be exclusively ultra-low emission cars, in a proactive step toward the aim of ultra-low emission cars forming the entire national car fleet by 2050 (DfT, 2018a, p. 7).

Projections of future traffic flows based on existing traffic flows are possible by the generation of traffic growth factors by means of the 'RTF Calculator' and TEMPro tools. Factors for traffic growth for 2030 based on 2014 in Newcastle upon Tyne and Gateshead have been extracted using these tools and combined as proposed in DEFRA guidance (DEFRA, 2010). For example, combinations of growth factors for traffic in Newcastle from 2014 to 2030 are illustrated in Table 6-1. These factors indicate an increase in the traffic flow of all vehicle types except for buses. The flow of LGV traffic is expected to exhibit the greatest increase 39% by 2030 compared to traffic growth associated with other vehicles.

It can be argued that the growth rates anticipated via the RTF and TEMPro tools are deliverable, given that they were presented based on real comprehensive data in the NTM and NTEM models. The NTM is a multi-modal model for Great Britain that has evolved from the 1997 national road traffic forecasts where it was used to forecast future traffic growth such as in a study conducted by Chatterjee and Gordon (2006) who explored alternative future scenarios for Great Britain in the year 2030. Both models were developed by professional teams who can access detailed information and analyse and interpret this information using valuable resources provided by the government.

The growth in traffic flow anticipated by the NTM and NTEM models would be accommodated by the expansion projects conducted on networks in Newcastle and Gateshead as a joint commitment between both councils to overcome transport problems in the local area to increase network capacity using several methods, such as changes in signal control and junction improvements to provide more capacity and reduce the duration of recurrent congestion. The councils are committed to investing in complementary small, local schemes as well as improving traffic management (DfT, 2014a, p. 9). Meanwhile, the government made a commitment to develop proposals for the improvement of major roads as presented in the Road Investment Strategy with a budget of £15.2 billion between 2015 and 2021, such as for the part of the A1 in the study area (DfT, 2014a, p. 9). These schemes would help the networks to accommodate the forecast in traffic increased flows.

Worthy of note is that the impact of interventions involving new technology, such as the increasing penetration of electric vehicles into the fleet, could be investigated if it applied to the baseline year 2014 or to the baseline year of the intervention year. In the former case the impact of the intervention would be as if the interventions could be immediately implemented. The advantage of this is the comparison of the interventions is with the base case year 2014 which is of known (or at least validated) traffic flows and fleet characteristics. However, the study of the increasing penetration of electric vehicle proportion in the base year with zero growth in the traffic flow was performed previously by Soret *et al.* (2014), who analysed three electrification scenarios for 13%, 26% and 40% of vehicles in Barcelona and Madrid for 2011. In this research the traffic growth and changes in fleet characteristics were considered for each intervention year. In this way a more realistic outcome was achieved. However, it must be recognised that the traffic flows for the base case are forecasted flows and therefore less reliable.

Other scenarios such as car sharing schemes which would not increase number of cars but potentially increased VKT travelled. However, this is beyond the scope of this thesis.

Table 6-1: Combined growth factors for traffic in Newcastle from 2014 to 2030

Period	Car	LGV	HGV	Bus
AM	1.17	1.39	1.11	0.89
IP	1.17	1.39	1.11	0.89
PM	1.23	1.39	1.11	0.89

These traffic growth factors were applied to the Baseline traffic model in order to update it to create the profile for BAU scenarios.

6.4.1 Projection of Electric Car Proportions in 2030

In the UK, the use of electric cars has grown rapidly over the past few years, and the cumulative registration of electric cars increased from 1,300 units in 2010 to more than 100 times that at the end of 2017, reaching 131,000 units (DfT, 2018g). Therefore, extra care should be taken in assigning electric cars to the BAU scenario for 2030. Available insights concerning the growth of electric cars in 2030 can be found in the following sources:

1) Fleet compositions projection published by the National Atmospheric Emissions Inventory (NAEI, 2017);

2) Extrapolation of the percentage of licenced electric cars in the fleet (Department for Transport, 2018; DfT, 2018g); and

3) Projections published by the Office for Low Emissions Vehicles and Road to Zero Strategy (OLEV, 2013a; DfT, 2018a).

6.4.1.1 NAEI Projections for the Proportion of Electric Cars by 2030

In the National Atmospheric Emissions Inventory (NAEI) published in December 2017, the basic fleet projections for proportions of vehicle-miles travelled (VMT) are broken down by vehicle type. These proportions are the default fleet composition data incorporated in the Emission Factor Toolkit (DEFRA, 2017b, p. 30; NAEI, 2017). In 2030, the NAEI estimates electric cars will form 2% of VMT, whilst petrol and diesel cars account for 47.5% and 33% respectively in urban areas in England excluding London, as detailed in Table 6-2 (NAEI, 2017). It worth noting that electric LGV, account for 0.4%; double the contribution of petrol LGVs in the VMT.

Table 6-2: VMT projections by vehicle type in urban areas in England, excluding London, by 2030

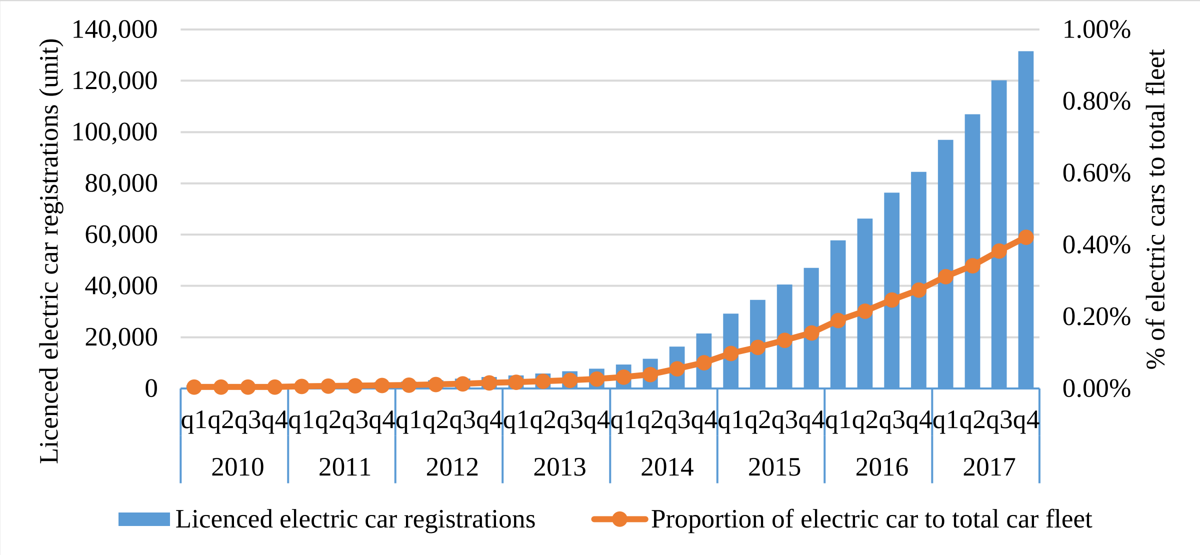
Vehicle type	Percentage to VMT
Electric car	2.0%
Petrol car	47.5%
Diesel car	33.0%
Electric LGV	0.4%
Petrol LGV	0.2%
Diesel LGV	13.8%
Rigid	1.0%
Artic	0.3%
PSV	0.9%
Motorcycle	0.9%
Total	**100%**

Source: NAEI (2017)

Proportions of different vehicle types in the VMT published by the NAEI assumed the default contributions of vehicle types in traffic flow (Goodman *et al.*, 2016; DEFRA, 2017b, p. 30). In the BAU scenario, the flows of cars, LGVs, HGVs and buses were provided separately for each road link. Thus, for car flow, the NAEI proportions of electric, petrol and diesel cars need to be scaled up to 2.4%, 57.6% and 40%, so that these classes' proportions add up to 100% to form the flow for cars. Similarly, electric LGVs, petrol LGVs and diesel LGVs account for 2.84%, 1.42% and 95.74% respectively of the flow representing LGVs. Rigid HGVs account for 76.3% and articulate HGVs 23.7% of HGV flow, whilst 99.8% of bus flow driven by conventional buses and 0.20% electric buses.

6.4.1.2 Extrapolation of Electric Car Growth to 2030 (DfT figures)

Licenced vehicle registrations published quarterly by the DfT reveal that the cumulative number of electric cars exceeded 131,000 at the end of 2017, which represents 0.42% of approximately 32 million registered cars in the UK, as documented in DfT statistics (Department for Transport, 2018; DfT, 2018g). The historical growth of electric cars compared to all car registrations since 2011, as illustrated in Figure 6-3, shows the steady increase in electric cars in terms of proportions of registrations and percentages of the car fleet.



Source: Department for Transport (2018) and DfT (2018g)

Figure 6-3: Growth of proportion of electric cars among cumulative registered cars

If the proportions of electric cars continue to increase following the trend in the UK to date, it is most likely that electric cars will reach nearly 4.5% of the UK car fleet by 2030 (Forecast using actual figures to date with a 2nd order polynomial trend line $R^2 = 0.99$ extrapolated to 2030), as illustrated in Figure 6-4. The linear and power trend lines were excluded because their R^2 values are the smallest. It is assumed in the BAU scenario that there will be no policy incentives to adopt EVs over this period.

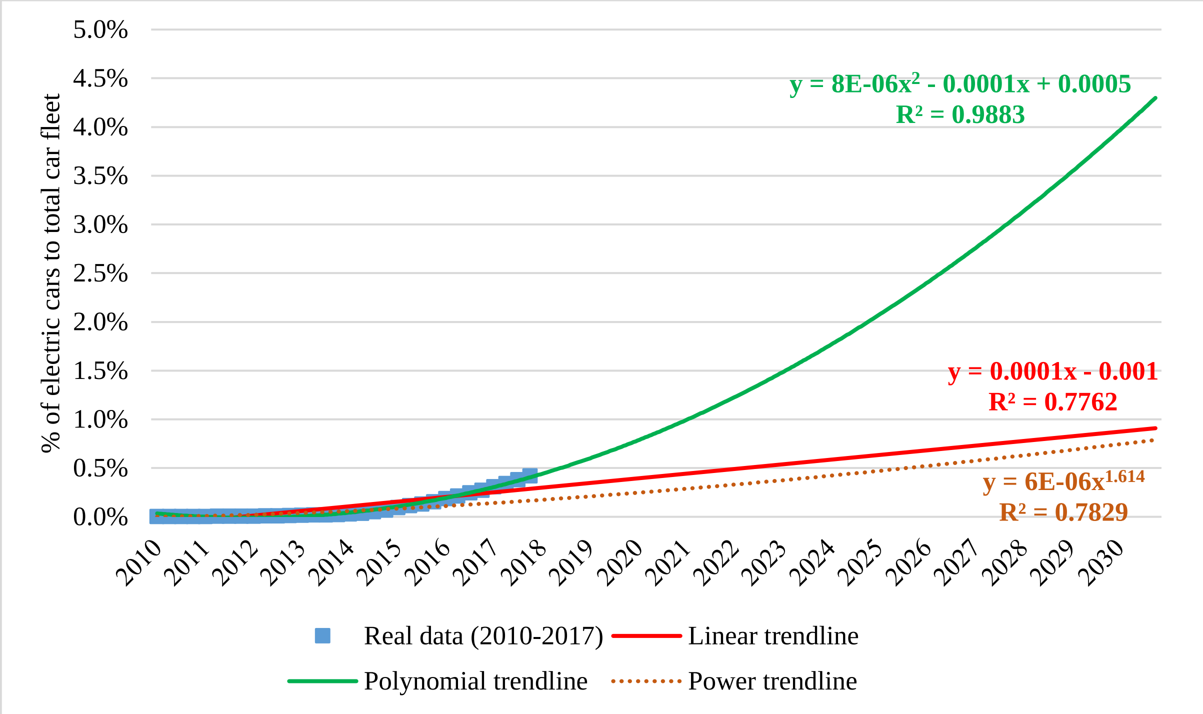


Figure 6-4: Extrapolation of percentage of electric cars of all cars

6.4.1.3 Projections of OLEV and Road to Zero Strategy

The Office for Low Emission Vehicles (OLEV) is a joint body representing the Departments for Transport, Business Innovation & Skills and Energy & Climate Change established to encourage and support measures to increase the penetration of ultra-low emissions vehicles (ULEVs). OLEV is funding the ULEV industry with over £500 million in grant during 2015 to 2020 including incentives for the purchase of ULEVs and installing home charging units and supporting public charging unit infrastructure (OLEV, 2013a, p. 12).

Based on 2013 circumstances in relation to ULEV growth, OLEV has outlined projections for ultra-low emissions car sales from 2011 to 2020. OLEV estimates that the sale of new electric cars is expected to range between 1% and 2.4% as a percentage of total car sales in 2017, as Figure 6-5 demonstrates. This projection agrees with reality, as the official registration of new electric car sales in 2017 exceeded 51,000 electric cars representing 2% of the 2.5 million total new car sales (DfT, 2018h). Likewise, OLEV estimated that in 2020 the share of electric car sales might range between 3% to 7% of new car sales. OLEV projections do not cover 2030, which is the year of the BAU scenario.

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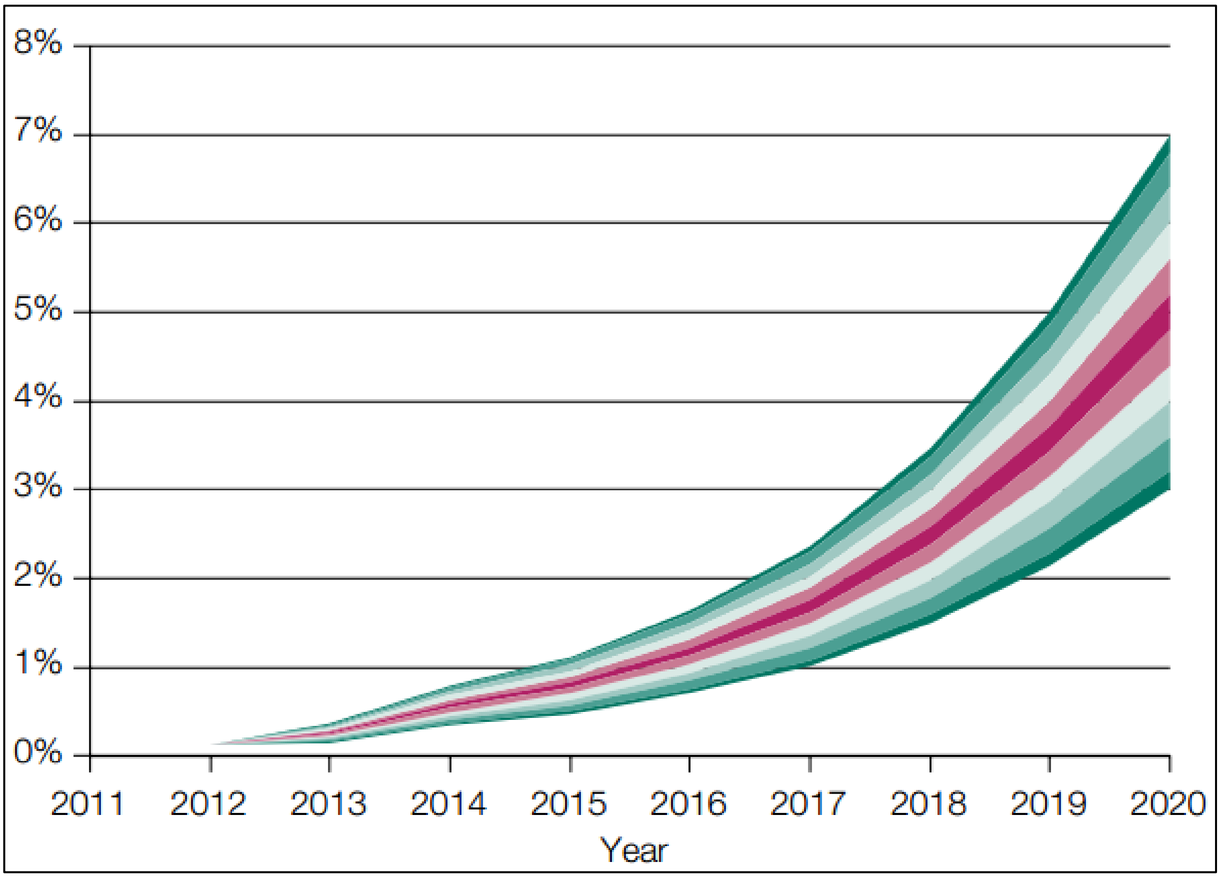


Note: Colour boundaries indicate various projections
Source: OLEV (2013a, p. 99)

Figure 6-5: Projected ULEV car sales as a proportion of all new car sales 2011-2020

The Department for Transport (DfT) published projections of new sales of electric cars in relation to total car sales in its most recent publication, the 'Road to Zero Strategy'. Based on current circumstances, this outlines the banning of conventional vehicle sales in the country in 2040 and huge investments in cleaner transport. The strategy confirms the expectations of OLEV concerning to the years 2018 to 2020 and extends the projections to 2030, when the sale of new electric cars is expected to reach 70% at the upper boundary and 30% at the lower boundary of the projections, as was displayed in Figure 6-2.

An estimation of the proportion of electric cars as a percentage of the total car stock in 2030 is required in order to predict the contribution of electric cars in the total car population. To estimate electric car stock, the following assumptions were made. Yearly percentages of electric car sales estimated in Figure 6-4 at the lower boundary were used to determine the likely cumulative electric car stock in 2030. For simplicity, the lifetime of electric cars was assumed to be 10 years, since the life span of electric car batteries is estimated to be more than 8 years of service or 100,000 miles (DfT, 2018a, p. 96). In addition, at least 2.5 million new car sales will be assumed annually from 2020 to 2030, because the average annual number of new car registrations from 2001 to 2017 was 2.3 million in the UK, and 2.5 million new cars were registered for the first time in 2017 (DfT, 2018h). Following these assumptions, cumulative electric cars would be 4.2 million, as displayed in Table 6-3, by following the lower boundary of the projections of the Road to Zero Strategy. Other projections of electric car numbers for the UK by 2030 range between 3 million to 20.6 million (Grau *et al.*, 2009).

Table 6-3: Projection of electric car stock to 2030

Year	% of new car sales	New car sales	New electric car sales
2021	4%	2,500,000	100,000
2022	7%	2,500,000	175,000
2023	10%	2,500,000	250,000
2024	12%	2,500,000	300,000
2025	15%	2,500,000	375,000
2026	18%	2,500,000	450,000
2027	20%	2,500,000	500,000
2028	25%	2,500,000	625,000
2029	28%	2,500,000	700,000
2030	30%	2,500,000	750,000
			Stock in 2030: 4,225,000

The percentage of electric cars in the car fleet was estimated in the future by assuming the total car registrations in 2030 can be forecast by extrapolation of the past registrations of cars from 1994 to 2017 as presented in Figure 6-6 (Department for Transport, 2018). In the figure, linear and quadratic trend lines were fitted to extrapolate the current trend for growth in the UK up to 2030. Although the quadratic trend line achieved a higher R^2, it was excluded because its function starts with a negative value suggesting that there is negative growth in the uptake of electric vehicles which is clearly unrealistic. Hence, as an alternative, the linear trend line was considered. The linear trend line (R^2=0.96) suggested that the car fleet will comprise approximately 37 million cars. This estimate is lower than the projection made by Grau *et al.* (2009) of 42.4 million cars. In 2030, the expected number of electric cars may possibly be 4.2 million, while the expected car stock is estimated to be 37 million. Thus, the proportion of 11% represents the number of electric cars in the car stock.

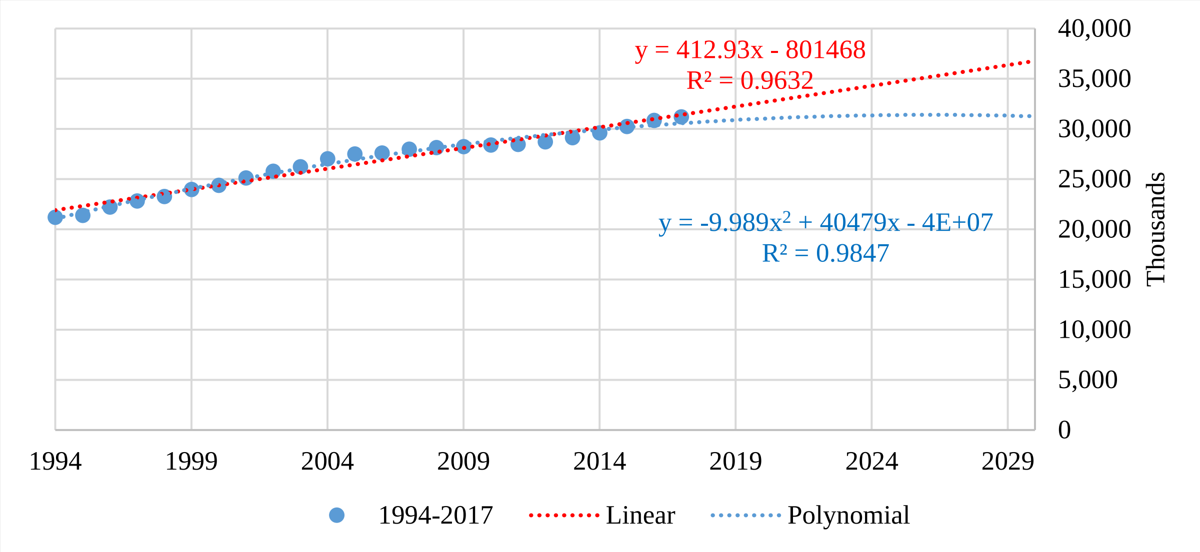


Figure 6-6: Extrapolation of annual cumulative car registrations in the UK (1994-2017)

6.4.1.4 Summary of Insight Projections Related to Proportion of Electric Cars by 2030

The NAEI (2017) has published figures on the contribution of electric cars to VMT driven by cars in 2030. Assuming this contribution in VMT represents the proportion of electric cars in the car flow, this means that electric cars will account for 2.4 % of car fleet flow. Whereas extrapolating the trend of growth of the share of electric cars in the car fleet indicates that 4.5% of the car stock would be electric. Additionally, extrapolating the figures in relation to new electric sales provided by OLEV and the Road to Zero strategy suggests that electric cars shall form 11% of the car fleet. Table 6-4 summarises these projections.

Table 6-4: Projection of electric car proportions in 2030

NAEI	DfT	OLEV & Road to Zero
2.4% of car VMT	4.5% of number of cars	11% of total car stock

Three insights were considered in forecasting the proportion of electric cars among car flows and stock by 2030. The projections derived from the DfT, OLEV and the Road to Zero strategy were based on extrapolations of the current trends in electric car growth rates. These projections presented numbers or percentages of electric cars among the total car stock. They expected that the electric cars would form 4.5% and 11% respectively of the car fleet by 2030.

In contrast, the NAEI figures provided the contribution of electric cars as a percentage of car flows rather than vehicle numbers, and this thesis is interested in vehicular emissions which also associated with vehicle flow. Moreover, the NAEI proportions represent the ‘default basic spilt’ incorporated in the EFT (DEFRA, 2017b, p. 30; NAEI, 2017). Nonetheless, an important key element which should be taken into account is the UK government's plans to restrict new car sales in 2040 to only ultra-low emission cars, in order to pave the way for ultra-low emission cars forming the whole national car fleet by 2050 (DEFRA and DfT, 2017b, p. 4). This plan might increase the penetration of electric cars on the roads. It thus appears to be more appropriate to adopt the NAEI projections in this research, although other insights should be taken into account. Therefore, the presumed contribution of electric cars was increased to 5% instead of 2.4% as regards car flow. To cope with these proportions of electric car, the share of petrol and diesel cars in the car flow were adjusted to 56% and 39% respectively in 2030. Other vehicle types, such as LGV and HGV, were made electric according to the NAEI projections, as Table 6-5 shows (NAEI, 2017).

Table 6-5: Proportion (%) of vehicle types in BAU scenario

Vehicle type	Propulsion	Proportion (%)
Car	Electric	5
	Petrol	56
	Diesel	39
	Total	**100**
LGV	Electric	2.8
	Petrol	1.4
	Diesel	95.7
	Total	**100**
HGV	Rigid diesel	76.3
	Articulated diesel	23.7
	Total	**100**
Bus	Conventional diesel	99.8
	Electric bus	0.2
	Total	**100**

Difficulties in forecasting electric vehicle growth could be attributed to their current comparatively high cost and lengthy time to market regarding manufacturing rates (OLEV, 2013a, p. 99). Likewise, other countries have targets to capture large proportions of global electric car production. In addition, fuel prices depend mostly on the global price of crude oil and infrastructure relating to electric cars are also important (OLEV, 2013a, p. 99). Moreover, the cost of batteries is expected to become significantly lower in the coming years (Bloomberg, 2017; IEA, 2018, p. 11). For example, the cost of a battery per kWh has reduced over the past ten years from $1,000 per kWh in 2010 to $273 in 2016 (Curry, 2017). This will certainly contribute to lowering the capital cost of electric cars. Furthermore, it can be argued that, without the car scrapping scheme that will facilitate the replacement of conventional vehicles by electric as well as gradual increases in fuel prices in the UK to help bring together the running and upfront costs of conventional vehicles, to be equivalent to the total cost of using electric cars, the growth of the take-up of electric cars may be only moderate.

6.5 Scenarios with a Range of EV Uptake Levels

EV investment should target vehicles that are driven predominantly in urban cores such as buses as well as vehicles with high utilisation rates in order to reduce human exposure to pollution, thus addressing the health-related issues of local air pollution. Chapter 5 shows that passenger cars and buses are the dominant sources of vehicular emissions at monitoring station sites in urban cores. The pollution emitted by these two types of vehicles was considered in developing 2030 scenarios.

In this chapter, six plausible scenarios of EV adoption in the vehicle fleet were developed for 2030 traffic flows in order to investigate their impact on air quality and health. These scenarios were built on the proportions of cars, LGVs, HGVs and buses related to BAU scenarios. The scenarios enable us to test and investigate different levels of adoption of electric vehicles for the range of vehicle modes being considered. The scenarios are as follows:

1) Committee on Climate Change (CCC) scenario;
2) Electrification of all buses (E-Bus) scenario;
3) Electrification of all cars (E-Car) scenario;
4) Electrification of all cars and buses (E-Car_E-Bus) scenario;
5) Electrification of all cars and LGVs (E-Car_E-LGV) scenario; and
6) Electrification of all vehicles (All-EV) scenario.

Investigating these scenarios allows estimations to be made of the emission of local pollutants and their impact on air quality and associated disease burden in terms of reduction in mortality and hospital admissions.

6.5.1 Committee on Climate Change (CCC) Scenario

In 2008, the Climate Change Act was legislated in the UK, making it the first country to introduce legally binding national targets for the lowering of GHG emissions by 2050 by 80%

of 1990 levels in order to alleviate climate change effects (Element Energy, 2013), given that the road-transport sector is the main contributor of GHG emissions (Chatterton, 2011; Hitchcock *et al.*, 2014, p. 8; Hooftman *et al.*, 2018). In addition, EVs can be considered to be a key technology in decarbonising this sector to meet GHG reduction targets by 2050 (Element Energy, 2013). Therefore, the Committee on Climate Change (CCC) provided valuable insights into projected EV sales until 2030 in aiming to achieve the 2050 targets for lowering GHG emissions.

In order to achieve CO_2 mitigation in the transport sector and subsequently to achieve the target of reducing GHG emissions in 2050 by 80% of 1990 levels, the CCC identified a possible pathway in the UK to achieve the high adoption of electric cars and vans. The CCC proposed that electric car and van would account for 60% of market sales by 2030, which would constitute 37% of the car and van stocks as shown in Table 6-6 (Element Energy, 2013, p. 171). The CCC recommended that 18% and 12% of cars would be PHEVs and BEVs respectively, as would 14% and 24% of vans.

Table 6-6: CCC recommendations for sales and stock of EV uptake in 2030

		Sale, 10^3	**Sale (%)**	**Stocks, millions**	**Stocks (%)**
EVs (cars, vans)	**All EVs**	2,100	60%	13.6	37%
	PHEVs	1,243	35%	7.68	21%
	BEVs	892	25%	5.91	16%
E-cars	**All EVs**	1,834	60%	11.42	30%
	PHEVs	1,141	37%	6.9	18%
	BEVs	693	23%	4.52	12%
E-vans	**All EVs**	301	60%	2.18	38%
	PHEVs	102	20%	0.78	14%
	BEVs	199	40%	1.4	24%

Source: Element Energy (2013, p. 171)

To calculate emission rates related to the CCC scenario, the proportions of PHEVs and BEVs for cars should be expressed in percentages of total car flow. According to cumulative vehicle registration data broken down by fuel type published in its Table VEH0203, DfT (2018i) highlighted that petrol cars had accounted for an average of 61.5% over the previous five years of car registrations, whilst their average contribution to VMT was 58% (NAEI, 2017). Meanwhile, the same figures indicated that diesel cars accounted for an average of 37.5% of car registrations (DfT, 2018i), whilst their contribution to VMT was 42%. The VMT for petrol cars was lower than their proportion in the car stock by 3.5% and that of diesel cars was greater than their proportion by 4.5%. To create a balance between these variations in petrol and diesel car stocks and VMT, the contribution of EVs to the car VMT were assumed to be equal to their proportions in the car stock in CCC scenario. Therefore, in this scenario, 18% and 12% of car flows would represent PHEVs and BEVs respectively, as would 14% and 24% of van flows, as Table 6-7 demonstrates.

Table 6-7: **Proportion (%) of vehicle types in CCC scenario**

Vehicle type	Propulsion		Proportion (%)
Car	Electric car	PHEV	18
		BEV	12
	Petrol		41.3
	Diesel		28.7
	Total		**100**
LGV	Electric LGV	PHEV	14
		BEV	24
	Petrol		0.9
	Diesel		61.1
	Total		**100**

6.5.2 Electrification of Buses (E-Bus) Scenario

Buses are considered to be the most-used public transport mode, given that they have access to almost all urban parts of the nation and due to their massive carriage capacity relating to passenger numbers. In England, the number of local bus passenger journeys was 2.2 billion, excluding London, in the twelve months between 2016 and 2017 (DfT, 2017c). Excluding London, Tyne and Wear registered high passenger journeys during this period ranking in fourth place among other counties with 114 million journeys (DfT, 2017b). According to the TWITA (2011, p. 34), 78% of public transport journeys were made by buses that covered 90% of the distance travelled in Tyne and Wear and in Newcastle upon Tyne, for instance, approximately 140,000 people commute daily to the urban core (TWRI, 2008, p. 1).

The high demand for bus journeys among passengers means that they are susceptible to the risk of exposure to air pollution. Motorised traffic is principally responsible for more than 60% of average NO_x concentration at roadsides, whilst 16% has been attributed to emissions from bus flows in the UK in 2015 (DEFRA and DfT, 2017a, p. 7). The emissions associated with buses raises serious public health questions due to the exposure to air pollutants. For instance $PM_{2.5}$ levels are greatest for bus users rather than car passengers (McNabola *et al.*, 2008). This exposure occurs not only during waiting times at bus stops but also while travelling, given that approximately 25% of inside-cabin pollutants are emitted by the buses themselves (Behrentz *et al.*, 2004; Adar *et al.*, 2008).

A strict evaluation of the relationship between reduced bus emissions and morbidity has been conducted in New York City in the United States by merging bus traffic-associated particulate matter and NO_x spatial concentrations at residential centroids with hospitalisation data for those residents who live close to those centroids. It was found that stricter transport bus emissions standards are associated with reduced emergency department visits for respiratory illnesses such as asthma and bronchitis (Ngo, 2015). Moreover, the extensive use of electric buses would improve both the quality of air by mitigating traffic-related emissions and, subsequently, health. In a recent study performed in Macao, He *et al.* (2018) indicated that emissions concentrations significantly decreased by 60% for NO_x, 10% for CO_2 and 40% for $PM_{2.5}$ in a comparison of electric and diesel buses under conditions of in-service emissions. Furthermore, it was proven that doses of inhaled polluted air for passengers using electric buses is lower compared to those using diesel buses, petrol and diesel cars (Zuurbier *et al.*, 2010). In 2017, China has witnessed successful bus electrification in Shenzhen (Song *et al.*, 2018). The success of the adoption of fully electric buses is a result of cooperation between local government and private investors, leading to a decrease in fiscal loads on governmental expenses and the encouragement of private investment involvement in the green sector of the automobile industry (Li *et al.*, 2016b). The electrification of buses is highly likely to reduce passenger exposure to pollution.

Furthermore, the OLEV has allocated £500 million to increase the uptake of ULEV buses from 2015 to 2020 (OLEV, 2014), and the Low Emission Bus Scheme (LEBS1) was announced in 2016 with a budget of £30.4 million to introduce more than 300 brand new low-emission buses to the roads of England and Wales (OLEV, 2018b, p. 4). Furthermore, £100 million was injected into the Scheme, of which £40 million was dedicated to the Clean Bus Technology Fund to subsidise local authorities to retrofit their existing bus fleets. The remaining £60 million was dedicated to allow local councils and bus operators to purchase new clean buses. Gateshead Council received £1.5 million to retrofit 79 buses working on 9 routes (Gateshead Council, 2018), whilst Newcastle Council obtained nearly £700,000 from the Clean Bus Technology Fund from 2017 to 2019 to be used to treat 43 buses (DfT & DEFRA, 2018). It should be noted that £12 million of the £60 million was spent on buying 150 low emission buses, whilst the remaining £48 million was devoted to purchasing ultra-low emissions buses (OLEV, 2018b). This scheme builds on the successful Green Bus Fund scheme of £87 million that delivered 1,250 ultra-low emission buses from 2009 to 2013 (OLEV, 2013a, p. 53).

In Tyne and Wear, the electrification of buses appears to be feasible. Nearly 90% of bus journeys are operated by two main providers: the Go Ahead Group and Stagecoach (DfT, 2017a). It is known that electrically powered buses need to be stored in their own depots where they can be recharged overnight utilising comparatively low electricity tariffs. In addition, solar panels can be installed on depot roofs to produce renewable energy and free power, leading to better health benefits resulting from improved air quality (Erickson and Jennings, 2017).

More locally, electrification of the bus fleet in Newcastle upon Tyne and Gateshead seems possible given that most of the services are run by only two operators. The availability of depots and flexibility in schedules allows for buses to undergo overnight recharging, which utilises low electricity tariffs. This is the most crucial factor is funding which is expected to be available from programmes financed by the OLEV. The E-Bus scenario was based on the BAU scenario where electric, petrol and diesel cars represent 5%, 56% and 39% of car flows respectively; but 100% of the bus flow represents electric buses. Table 6-8 demonstrates traffic flow by vehicle type with regard to the E-Bus scenario.

Table 6-8: **Proportion (%) of vehicle types in E-Bus scenario**

Vehicle type	Propulsion	Proportion (%)
Car	Electric	5
	Petrol	56
	Diesel	39
	Total	**100**
LGV	Electric	2.8
	Petrol	1.4
	Diesel	95.7
	Total	**100**
HGV	Rigid diesel	76.3
	Articulated diesel	23.7
	Total	**100**
Bus	Conventional diesel	-
	Electric bus	100
	Total	**100**

6.5.3 Electrification of All Cars (E-Car) Scenario

Cars have accounted for 78% of vehicle flows in the past ten years (DfT, 2018e) and comprise 84% of vehicle registrations since 1994 (Department for Transport, 2018). This large contribution of cars in traffic activity causes more pollution compared to other vehicle types.

For example, passenger cars caused nearly half of NO_x emissions attributed to road transport, where diesel cars alone caused 42% of these emissions in 2016 (DEFRA, 2018; DfT, 2018a, p. 26). Passenger cars not only caused an increase in NO_x emissions but also contributed 15% of total UK GHG emissions in 2016 (DfT, 2018a, p. 28)

Furthermore, the UK government supports the purchasing of ultra-low emission vehicles with a package of incentives. The OLEV offers a 35% discount up to £3,500 for purchasing any of 19 models of ULEVs under 'category 1', which are cars with CO_2 emission rates of less than 50 g/km for a minimum range of 70 miles. The grant increases to £7,500 if these models are licensed to be used as taxis (OLEV, 2018a). It is worth mentioning that the OLEV's 'category 1' should take into account other emissions beside CO_2 such as NO_x. Next Green Car Ltd has developed the NGC rating index, which expresses the impact of vehicles on the environment during the production phase; for instance, in the production of materials, manufacture of vehicles and vehicle transport. Moreover, phases are also taken into account to include fuel production, refining and distribution; and an end-of-life phase (Next Green Car Limited, 2016). The NGC index ranges between zero for the greenest vehicles to 100+ for the most polluting vehicles. The index considers the impact of real driving emissions of GHGs such as CO_2, CH_4 and N_2O, and likewise air quality pollutants including NO_x, HCs, CO, PM_{10} and SO_x. Indeed, the NGC index takes into account most aspects of the effect of the relevant harmful emissions with the full life cycle of vehicles on the environment.

The motivation for this scenario is the government plan to set ultra-low emission cars, focusing on electric vehicles, as the only option for purchasing new cars in 2040, targeting ultra-low emission cars to constitute the entire car stock in 2050 (DfT, 2018a). In the same context, the scrapping scheme that replaces conventional vehicles with ultra-low emission vehicles is a key element in speeding up the transition toward ultra-low emission vehicles by 2050. A further key element is to increase the conventional fuel price to a level where the total cost of conventional vehicles equals the cost of ultra-low emissions vehicles. However, aspects of this solution might trigger objections and lead to negative consequences such as the recent large-scale demonstrations in France.

The E-Car scenario assumes that electric cars will account for 100% of car flows. Other vehicle proportions were assumed in the BAU scenario. Table 6-9 shows the percentages of vehicle types in vehicle flows in the E-Car scenario.

Table 6-9: Proportion (%) of vehicle types in E-Car scenario

Vehicle type	Propulsion	Proportion (%)
Car	Electric	100
	Petrol	-
	Diesel	-
	Total	**100**
LGV	Electric	2.84
	Petrol	1.42
	Diesel	95.74
	Total	**100**
HGV	Rigid diesel	76.30
	Articulated diesel	23.70
	Total	**100**
Bus	Conventional diesel	99.8
	Electric bus	0.2
	Total	**100**

6.5.4 Electrification of All Cars and Buses (E-Car_E-Bus) Scenario

Cars and buses are responsible for the most polluted traffic flows in urban cores in Newcastle and Gateshead. As mentioned in the previous chapter, 95% of NO_x emissions are attributed to bus and car flows in the study area. Switching these vehicle types to electric vehicles might deliver an improvement in air quality and impact positively on health.

In the E-Car_E-Bus scenario, the proportions of electric cars and electric buses were set to 100% of car and bus flows, as Table 6-10 illustrates.

Table 6-10: Proportion (%) of vehicle types in E-Car_E-Bus scenario

Vehicle type	Propulsion	Proportion (%)
Car	Electric	100
	Petrol	-
	Diesel	-
	Total	**100**
LGV	Electric	2.84
	Petrol	1.42
	Diesel	95.74
	Total	**100**
HGV	Rigid diesel	76.30
	Articulated diesel	23.70
	Total	**100**
Bus	Conventional diesel	-
	Electric bus	100
	Total	**100**

6.5.5 Electrification of All Cars and LGVs (E-Car_E-LGV) Scenario

The registration of LGVs in the North East has witnessed a vast increase, and official statistics published by the DfT demonstrate that licenced LGVs in the region rose by 38% from approximately 108,000 in 2010 to 149,000 in 2011 (DfT, 2018j). Additionally, the proportion of LGVs in the vehicle flow increased from 13.5% in 2010 to 15.5% in 2017 in the UK (DfT, 2018e). The increase in the penetration of LGV electrification could be of benefit to air quality and health.

The-Car_E-LGV scenario assumes that 100% of cars and LGVs are electric, as revealed in Table 6-11.

Table 6-11: **Proportion (%) of vehicle types in E-Car_E-LGV scenario**

Vehicle type	Propulsion	Proportion (%)
Car	Electric	100
	Petrol	-
	Diesel	-
	Total	**100**
LGV	Electric	100
	Petrol	-
	Diesel	-
	Total	**100**
HGV	Rigid diesel	76.30
	Articulated diesel	23.70
	Total	**100**
Bus	Conventional diesel	99.8
	Electric bus	0.2
	Total	**100**

6.5.6 Electrification of All Vehicles (All-EV) Scenario

Currently, considerable effort is being made to establish Clean Air Zones (CAZs), where all vehicles entering each CAZ should be cleaner. According to previous plans to minimise air pollution levels, the DEFRA forecasts a reduction in NO_2 concentration levels in most cities in England by 2020, except for cities such as London, Birmingham, Leeds, Nottingham, Derby and Southampton. As most NO_2 levels originate from vehicular emissions, the introduction of the CAZ plan is designed to reduce breaches of limits relating to NO_2 source emissions from road transport in the above-mentioned cities (DEFRA, 2015, p. 15).

The primary idea of the CAZ plan is to reduce the number of old diesel and petrol vehicles entering the urban cores as DEFRA (2015, pp. 21, 27) defined the minimum emission standards proposed for the most common conventional vehicles as Euro 6 diesel vehicles, Euro 4 petrol vehicles and Euro VI HGVs, buses and coaches. This suggests diesel vehicles belonging to the sixth Euro standard and higher class are permitted to enter CAZs. Moreover, vehicles which do not meet these standards will be charged to enter these zones. It is important to mention that there is substantiated evidence that those diesel cars of Euro 5 and 6 are responsible for increasing NO_2 concentrations (Department for Transport, 2016, pp. 22, 23).

The CAZ rules emphasise that all diesel vehicles must comply with the Sixth European Emissions standards, whilst petrol cars and vans must comply with the Fourth European

Emissions standards to allow them to enter certain areas that witness high NO_2 concentrations. Therefore, CAZs are categorised to discourage the following non-compliant vehicle types:

Class A: Buses, coaches and taxis.

Class B: Buses, coaches, taxis and heavy goods vehicles (HGVs).

Class C: Buses, coaches, taxis, HGVs and light goods vehicles (LGVs).

Class D: Buses, coaches, taxis, HGVs, LGVs and cars.

It is proposed that Class B, which is a group of vehicles including buses, coaches, taxis and, HGVs that should achieve Euro VI emissions standards, will be implemented so as to establish clean air zones in Nottingham, Derby and Southampton, whilst clean air zones in Birmingham and Leeds will utilise Class C which is a group of vehicles including vehicles in the Class B group and LGVs. At current levels of NO_2, it is anticipated that air quality objective will be met without excluding cars in the above-mentioned cities.

Furthermore, a study that links CAZs (discouraging the most polluting vehicles from entering a certain area) and health was conducted in Germany, specifically on infants born between 2012 and 2015. After legislating the Clean Air Zone Act, Gehrsitz (2017) performed a regression model in to correlate monitored pollution levels and infant health *before* and *after* establishing low emission zones to investigate the impact of establishing low emission zones or clean air zones on air quality and birth outcomes. Gehrsitz (2017) found a modest improvement in birth weight and a moderate reduction in PM_{10} levels. The study also demonstrated that the number of days when average daily PM_{10} exceeded 50 μg/m^3, were reduced by three per year. suggesting that the evaluation of other health outcomes such as for example respiratory diseases, is eminently possible.

Therefore, the novelty in this work is that the vehicular emissions in two boroughs were estimated and the dispersion of these emissions, considering measured meteorological data, was modelled. In addition, the air quality was spatially predicted assuming new vehicle technology namely, electric vehicle was introduced into the vehicle fleet to explore potential changes in pollution concentrations in future years. Additionally, the reduction in mortality numbers and hospital admissions due to respiratory diseases were quantified.

The variable that impact the health is the methodology of developing relative risks. For example, exposure to $PM_{2.5}$ has the most impact on health. Although emissions attributed to bus flows is small as presented in section 7.2.1, 7.2.2 and 7.2.3, bus emission has significant impact on health because buses have access to urban cores.

The purpose of the CAZ plan is to encourage cleaner vehicles to drive in certain zones to improve air quality. This idea was adopted in the hypothesis of the (All-EV) scenarios by extending these zones to cover the entire study area. This scenario evaluates the most optimistic circumstances of cleaner traffic flows associated with the electrification of all vehicles.

6.6 Feasibility of 2030 Scenarios

In addition to the business-as-usual (BAU) scenario, six scenarios with a range of EV adoption levels were created in this chapter. These 2030 scenarios include:

1. 'CCC': Committee on Climate Change proposal for 30% of cars and 38% of vans being electric;
2. 'E-Bus': Electrification of all buses;
3. 'E-Car': Electrification of all cars;
4. 'E-Car_E-Bus': Electrification of all cars and buses;
5. 'E-Car_E-LGV': Electrification of all cars and LGVs; and
6. 'All-EV': Electrification of all vehicles.

These scenarios whilst extreme, provide boundaries for what is achievable. The question is the timescale for the implementation of these scenarios. The only exception is for the CCC scenario which is considered a realistic goal for the UK government to achieve.

The CCC scenario is related to the vision proposed by the Committee on Climate Change in order to achieve the mitigation of GHG emissions by the 2030 target. This vision is primarily focused on the increasing adoption of both hybrid and pure electric vehicles on the roads. Nevertheless, current incentives for the purchase of electric vehicles are dedicated to totally electric cars only. Therefore, it is difficult to believe that this scenario is feasible given that the grants for purchasing hybrid electric cars are no longer available.

The feasibility of the E-Bus where buses are propelled by electric motors is extremely promising, because the implementation of this scenario is related to direct government funding via projects dedicated to increasing the penetration of ultra-low emissions buses, similar to the Green Bus Fund scheme of £87 million that delivered 1,250 ultra-low emission buses from 2009 to 2013 (OLEV, 2013a, p. 53). Moreover, only a few operators run buses in Newcastle and Gateshead, which would ease the transition to electric buses. Given that bus operators have their own depots where charging could take place. Additionally, charging buses at night when the electricity tariff is at a minimum could be an option for electric bus batteries. A further point worth noting is that solar panels could be set on bus depot roofs, so that sustainable clean energy could be used. Likewise, commuting by bus appears to be the most preferred mode of transport not only in the study area but also in London. This will please taxpayers given that a portion of their contribution will go to something they use, and it also means that taxpayers would be exposed to less air pollution because buses can access the urban core where most exposure to air pollution occurs. In relation to timescales, if decision-makers adopted the bus electrification scenario on a wide scale, it could be achieved before 2030.

The feasibility of the E-Car scenario is extremely high due to the ban on new conventional car sales that will take place by 2040; given that the government announced recently the ban will be by 2035 (HIRST, 2020). This will accelerate the rate of electric car penetration which in turn increases the feasibility of the E-Car scenario and therefore suggests that the electrification of most new cars by 2030 is possible. In comparison to conventional cars, electric cars frequently have lower running costs, whilst the total cost of ownership, which combines the purchase price and operating expenses, declines annually from the year of introduction (Palmer *et al.*, 2018). The government would be able to implement the E-car scenario in Newcastle and Gateshead 14 years after the ban on conventional cars was activated, because conventional vehicles on roads in the UK have an average lifespan of approximately 14 years (DfT, 2018a, p. 13). Moreover, it is essential to increase public awareness regarding the low running costs associated with electric cars and the considerable potential for air quality improvements which will have an impact on public health. The scrappage scheme for the replacement conventional cars with electric cars and increasing tax and therefore fossil fuel prices are other methods that could be implemented regarding the earlier adoption of the E-Car scenario.

Following on from above the likelihood of the adoption of the E-Car_E-Bus scenario depends on the high feasibility of the electrification of both cars and buses.

The electrification of all cars and LGVs is represented in the E-Car_E-LGV scenario. It appears that the increasing penetration of electric LGVs is unlikely to cover the entire LGV fleet given the rate of annual growth of cumulative numbers of LGVs in the UK being the highest among the vehicle types over the past 5 years. From a hypothetical point of view, it is important for policymakers to understand the implications of such scenarios to establish policy that is beneficial for society. Hence, this scenario was also investigated.

The All-EV scenario involves the electrification of all vehicles, including HGVs. The feasibility of this scenario is low, because in the UK most HGVs are propelled by engines that make use of diesel fuel (DfT, 2018a, p. 117) and also because diesel engines are more efficient than petrol engines in HGVs (DfT, 2018a, p. 125). Likewise, retrofitting new technologies to existing HGVs could reduce emissions at lower cost than purchasing a new vehicle, at least in the short term (DfT, 2018a, p. 13). Additionally, an electric HGV is more expensive than its conventional counterparts (DfT, 2018a, p. 55). Moreover, there has been a huge reduction in the amount of emissions released from HGVs from the fifth to the sixth Euro HGV. For these reasons, the All-EV scenario might not be feasible. Therefore, this scenario can be considered a hypothetical scenario, in order to investigate the maximum penetration of electric vehicles. Such a hypothetical scenario will give the upper bound of benefits by means of testing the impact of the electrification of all vehicles.

To sum up, the scenario relating to the electrification of all buses (E-Bus) has the greatest feasibility among the specified scenarios because it relies on government policy and deals with a few bus operators. Although technology has lowered emission quantities released from diesel engine powered buses, the concentration of those emissions on bus routes in urban cores are high and are harmful to the population. The government should take this crucial information into consideration when preparing the relevant policies.

6.7 Summary

The procedure for updating the traffic model from 2014 to 2030 following DEFRA guidelines has been presented in developing the BAU scenario, whilst the proportions of electric cars were adopted from VMT projections by the NAEI.

Besides the BAU for 2030, six other scenarios were developed. The CCC scenario was developed according to the vehicle fleet mix proposed by the Committee on Climate Change for 2030, in order to reduce carbon dioxide emissions in 2050 by 80% of 1990 levels.

The bus electrification scenario was developed from the understanding that significant parts of bus journeys are operated by two operators in the study area. Also, buses can reach the urban core and cause levels of human exposure to pollution to increase.

The scenario for electrifying all cars (E-Car) was developed because they cause nearly 50% of NO_x emissions released by road transport and 15% of national CO_2 emissions. In the E-Car scenario, all cars would be electric. Similarly, in the E-Bus scenario all buses would be electric. The E-Car_E-Bus scenario is a combination of these two scenarios. All cars and LGVs are assumed to be electric in the E-Car_E-LGV scenario which was derived from the E-Car scenario. Finally, the all-vehicles electrification (All-EV) scenario was derived in the research from the plan by DEFRA to implement Clean Air Zones in some cities by preventing old cars from entering certain zones via city centre roads. These zones were expanded here to cover the entire study area.

The BAU and 2030 scenarios developed in this chapter and the Baseline which was developed in chapter 5 are evaluated in chapter 7. This includes the calculation of emissions rates related to traffic flows in the network, estimations of pollution concentrations and quantifications of the disease burden.

CHAPTER 7

7. Estimation of Emissions, Air Quality and Health Impacts of the Scenarios

7.1 Introduction

The Baseline scenario related to 2014 was developed in chapter 5 and the BAU and six other scenarios for 2030 were developed in chapter 6. The 2030 scenarios include the following:

1. ‘CCC’ scenario: Where the pathway proposes that in 2030, plug-in hybrid vehicles (PHEVs) form 18% and battery electric vehicles (BEVs) form 12% of the car fleet. Van fleet would comprise PHEVs 14% and BEVs 24% (Element Energy, 2013, p. 171);

2. ‘E-Bus’ scenario: Based on vehicles proportions in the BAU scenario except for electrifying all buses;

3. ‘E-Car’: Increasing the percentage of electric cars up to 100% in the BAU proportions;

4. ‘E-Car_E-Bus’: All cars and buses are assumed fully electric in the BAU scenario;

5. ‘E-Car_E-LGV’: Setting both electric cars and electric LGVs percentages in the BAU scenario to 100%; and

6. ‘All-EV’ scenario: Based on the hypothesis of electrifying all vehicle fleets by 2030.

In this chapter, the rates of vehicular emissions released in these scenarios is calculated utilising the Emissions Factors Toolkit (EFT v8). The modelling of the dispersion of those emissions is accomplished by running ADMS-Urban several times taking each GP practice site in the study area as a receptor, bearing in mind that patients have a tendency to register with the GP practice closest to their homes (Santos *et al.*, 2017; Beghelli, 2018, p. 84).

In order to model dispersion of emissions released by traffic flow on the entire road network in Newcastle and Gateshead, the ADMS-Urban was run six times for each scenario to estimate pollution concentrations at each GP practice site. Six runs were needed because the available licence for ADMS-Urban limits the size of the network to less than 3,000 road links per run; one run to model the dispersion of emissions released by cars, LGVs and HGVs and five runs to model the dispersion of bus emissions due to the substantial number of road links in the bus network. Given that seven scenarios need to be modelled, this meant that 42 runs of the ADMS-Urban model were made altogether.

In order to quantify the health impact resulting from improvements in air quality, dose-response coefficients are implemented that describe the association between exposure to an increase in concentration (e.g. 10 μg/m^3) of a pollutant and the consequent probability of premature death and admission to hospital. It was decided to adopt coefficients that are most appropriate for the UK, in particular those produced by governmental organisations or affiliates. These coefficients do not have thresholds, and so even though an ambient pollutant concentrations may be within legally established thresholds, it could still harm a person's health (Katsouyanni, 2003; WHO, 2003; Olmo *et al.*, 2011; WHO, 2013b). Hence, the probability of dying prematurely and/or being admitted to hospital due to exposure to those pollutants begins from a zero concentration. Implementing dose-response relationships assists in quantifying the expected disease burden (or health gain) in terms of reducing premature deaths and hospitalisations resulting from the improvement of air quality. The products of the number of patients, differences in NO_2, PM_{10} and $PM_{2.5}$ levels at each GP practice and corresponding dose-response coefficients are used to quantify the health gain, as shown in the following formula:

$$\textbf{Health gain} = \textbf{Number of deaths or hospital admissions} \times \frac{\mathbf{1}}{\mathbf{10}}\,\textbf{Change in pollutant concentration} \times \textbf{Dose-response coefficient}$$

This chapter investigates the expected emissions concentrations and disease burden in terms of mortality numbers and hospital admissions due to respiratory disease by 2030 for several scenarios.

7.2 Vehicular Emissions Changes by 2030 Scenarios

In general, quantities of vehicular emissions depend on vehicle type and speed, driving style and the type of propulsion technology involved (Ericsson, 2001; Muslim *et al.*, 2018). A propulsion technology such as in battery-driven electric vehicles (BEVs) has the benefit of not causing any exhaust emissions although, like any conventional vehicle, they release non-exhaust emissions including $PM_{2.5}$ and PM_{10} due to friction between the brakes and brake disks as well as between the tyres and road surfaces (Jeong *et al.*, 2019). It appears that EVs would not make a significant difference in lowering particulate matter emissions due to their weight compared to conventional vehicles, although driving style does play a significant role in releasing those emissions (Timmers and Achten, 2016; Timmers and Achten, 2018). The proportion of $PM_{2.5}$ emissions represents the dominant contribution to the mass of particulate matter released from exhaust pipes (Jeong *et al.*, 2019). $PM_{2.5}$ and PM_{10} are created in similar quantities from brakes and clutch wear emissions, whilst PM_{10} is the dominant contributor in road abrasion, tyre wear and road dust re-suspension (Ketzel *et al.*, 2007).

In relation to NO_x emissions, most vehicles in 2030 were assigned to the Euro 6/VI emission standards, and a significant drop was assigned to NO_x emissions thresholds in comparison to the previous Euro standards regulating thresholds regarding rates of vehicle emissions which become stricter in every successive directive (DEFRA, 2016c; DEFRA, 2017c). Conventional vehicles may conform to these standards under test conditions, but pollutant emissions released during real-world driving conditions breach Euro 1/I – 6/VI emission standards, particularly for diesel vehicles (Department for Transport, 2016; Moody and Tate, 2017). Additionally, NO_x emissions of two European vehicle manufacturers were discovered to be higher than the specified limit (Dey *et al.*, 2018b) despite fraudulently certified compliance, and more than one million such vehicles were sold in the UK by one manufacturer causing increased NO_x emissions to be released illegally into the atmosphere (Oldenkamp *et al.*, 2016). Therefore, any improvement in air quality in the BAU scenario would occur only if Euro 6/VI vehicles are committed to the legislated strict emission limits.

The vehicular emissions associated with the developed BAU and 2030 scenarios were calculated. Rates of $PM_{2.5}$, PM_{10} and NO_x emissions were modelled using EFT v8 for the 2030 scenarios, whilst EFT v7 was used for the 2014 Baseline scenario given that EFT v8 does not support data in the year 2014. The following sections show the results for the vehicle emissions inventory of $PM_{2.5}$, PM_{10} and NO_x accompanying the traffic flows of the Baseline and 2030 scenarios.

7.2.1 Total Fine Particulate Matter ($PM_{2.5}$) Emissions

Calculations in vehicle emissions rates modelling indicate that the total $PM_{2.5}$ emissions released by traffic networks in Newcastle and Gateshead during 2014 was 95 tonnes i.e. the Baseline scenario. This amount is expected to be reduced by 17% to reach 78.5 tonnes in the BAU scenario, as presented in Figure 7-1. The largest reduction compared to the Baseline scenario is 23% in the All-EV scenario followed by the E-Car_E-LGV scenario with a 22% reduction. Further scenarios can achieve reductions in $PM_{2.5}$ emissions from 21% to 18%, achieving a similar reduction in $PM_{2.5}$ emissions to that of the All-EV scenario compared to the 2014 Baseline scenario. This is because exhaust emissions represent a relatively low amount of vehicular $PM_{2.5}$ emissions. For instance, in the 2030 BAU scenario, exhaust emissions accounted for 7% (5.5 tonnes) of $PM_{2.5}$ emissions. In addition, 26% of $PM_{2.5}$ emissions (20 tonnes) are from road surface abrasion, 31% (24 tonnes) from brake friction and 37% (29 tonnes) from tyre wear, as calculated by EFT v8 for $PM_{2.5}$ emissions associated with the traffic flow in the 2030 BAU scenario. Although particulate matter is generated from sources other than just vehicular emissions, it the amounts will still be significant if the number of vehicles on the roads remains large.

Among vehicle traffic flows, passenger cars are responsible for more than 50% of vehicular $PM_{2.5}$ emissions in all scenarios. For example, in the 2030 BAU scenario, car flows account for 58% of total vehicular $PM_{2.5}$ emissions of which 26% was from road abrasion, 29% from brake wear, 6% from exhaust emissions and 39% from tyre wear.

For cars and LGVs, tyre wear was the dominant part of the $PM_{2.5}$ emissions, whilst brake wear was the dominant component in $PM_{2.5}$ emissions released from buses and HGVs.

The contribution of buses to $PM_{2.5}$ emissions is less than that of other vehicle types. Moreover, it was noted that buses accounted for the lowest amounts of $PM_{2.5}$ emissions in all scenarios ranging from 3.9 tonnes to 4.3 tonnes in the 2030 scenarios given that it was 9.4 tonnes in the 2014 Baseline scenario. In the 2030 BAU scenario, $PM_{2.5}$ emissions released by buses can be broken down into 1.4 tonnes (33%) from abrasion wear, 1.5 tonnes (34%) from brake wear, 0.4 tonnes (9%) from exhaust and 1 tonne (24%) from tyre wear. This means exhaust $PM_{2.5}$ represents a small amount of the total $PM_{2.5}$ bus emissions.

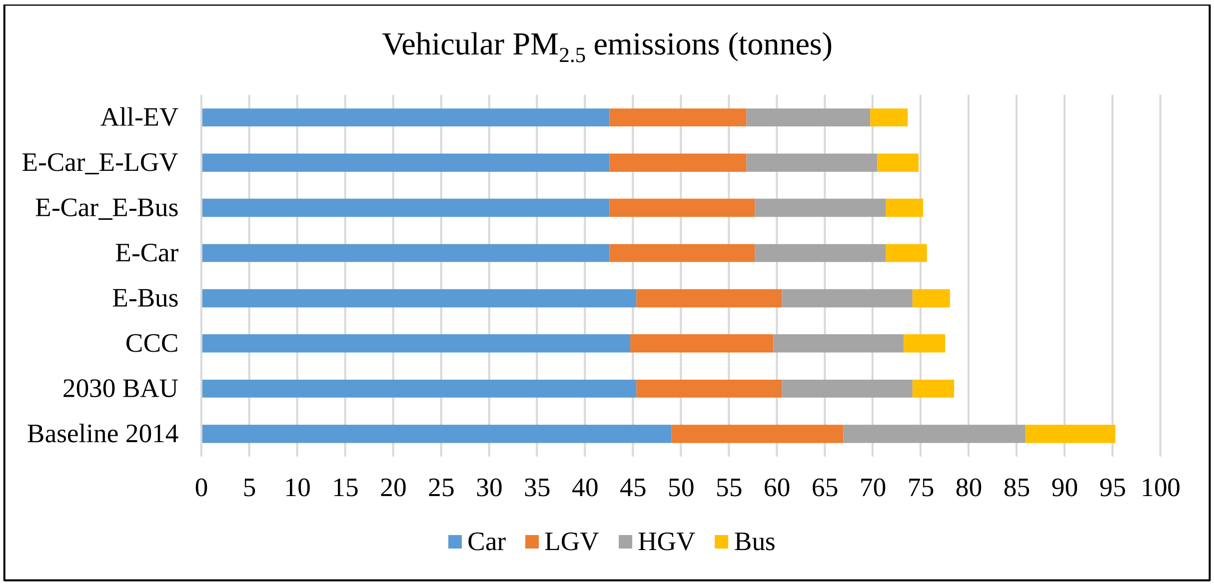


Figure 7-1: Vehicular $PM_{2.5}$ emissions by vehicle type for all scenarios

7.2.2 Total Coarse Particulate Matter (PM_{10}) Emissions

Modelling the entire traffic network for Newcastle and Gateshead in the 2014 Baseline scenario led to a figure of 152.5 tonnes for PM_{10} emissions. It is estimated that those PM_{10} emissions will be reduced to 144.8 tonnes in Newcastle and Gateshead by means of the 2030 BAU scenario, primarily due to the reduction in PM_{10} emissions related to the flow of HGVs and buses. Compared to the 2014 Baseline scenario, the reduction in PM_{10} emissions is expected to range from 5.3% in the E-Bus to 8% in the All-EV scenario, whilst the E-Car_E-LGV scenario would achieve a similar reduction to the All-EV scenario outcomes, of 7.6% reduction in PM_{10} emissions compared to the 2014 Baseline scenario, as Figure 7-2 shows.

The contribution of car flows to PM_{10} emissions is the largest in all scenarios. Even with the full electrification of cars, PM_{10} emissions related to car flow only slightly decreases to 80 tonnes in the E-Car, E-Car_E-Bus, E-Car_E-LGV and All-EV scenarios. This is because exhaust emissions represent a small amount of the total emissions of PM_{10}. For example, in the 2030 BAU scenario, PM_{10} emissions related to exhaust emissions is only 4% for car flow, 5% for bus flow and 3% for each of LGV and HGV flows, as calculated by EFT v8.

On the other hand, in the smaller city network focusing on the AQMA at Newcastle city centre, vehicular PM_{10} emission released from cars was estimated to slightly increase, to reach 4.5 tonne in 2014 Baseline scenario compared to 4.36 tonne in 2011 as modelled by Goodman *et al.* (2014, p. 97), this represent an increase of 3%.

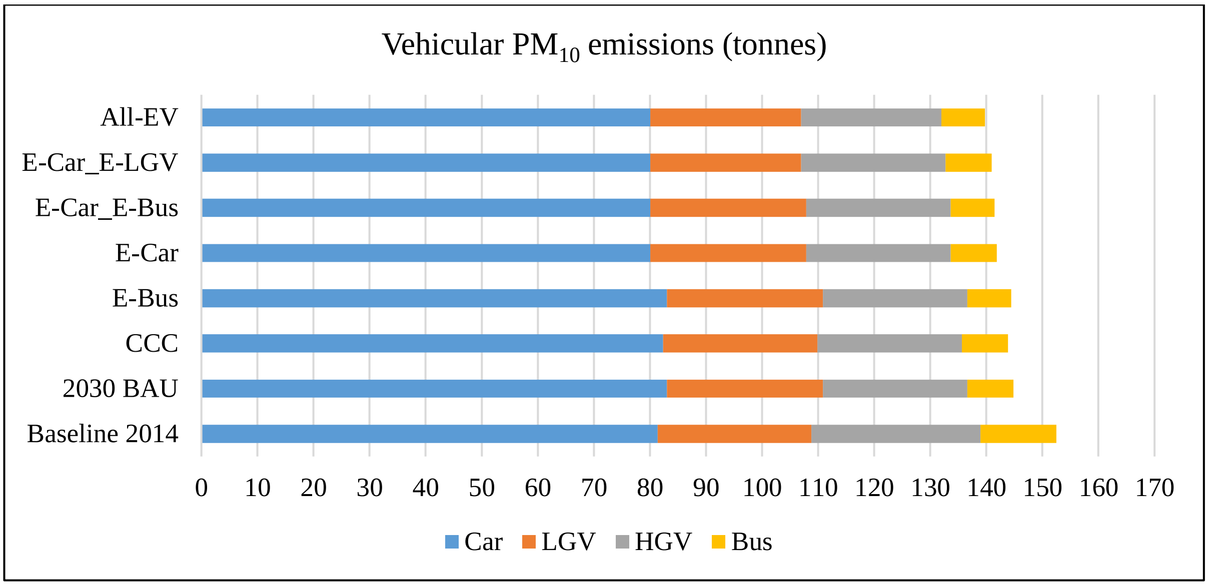


Figure 7-2: Vehicular PM_{10} emissions by vehicle type for all scenarios

7.2.3 Total Nitrogen Oxides (NO_x) Emissions

Total and source distributions in the modelling of NO_x emissions for Newcastle and Gateshead are presented in Figure 7-3. Emissions of NO_x in the Baseline were estimated to have reached 2,100 tonnes in 2014 Baseline scenario and would be reduced by 70% in the 2030 BAU scenario to 640 tonnes. All vehicular NO_x emissions will disappear in the All-EV scenario, as they are originated from exhausts and as all vehicles are assumed to be ultra-low emissions vehicles. In the E-Car_E-LGV scenario, a huge reduction of 95% in NO_x emissions compared to the Baseline scenario can be achieved, whilst substituting only cars with electric vehicles (i.e. E-Car scenario) will generate a reduction in NO_x emissions by 85% compared to the Baseline scenario. Adopting both electric cars and LGVs will lower the NO_x emission ceiling to a level that is similar to the All-EV scenario when all vehicles are electric. Undoubtedly, investing in the substitution of only one vehicle type, such as only cars or buses, is less expensive than investing in the shifting of all vehicles to being electric.

Emissions of NO_x related to HGVs were estimated to be 556 tonnes in 2014 Baseline, whilst in the 2030 BAU scenario emissions are predicted to be 51 tonnes only. This indicates a reduction of more than 90% in NO_x emissions released by HGVs. The Euro V standards for HGVs states that NO_x emissions should not exceed a threshold of 2.0 g/kWh, whilst the Euro VI standards specify lowering the NO_x emissions threshold by 80% to only 0.4 g/kWh. According to the Emissions Factors Toolkit version 7, for year the 2014 (EFT v8 does not support 2014), the proportion of Euro VI in HGVs is only 19% for rigid HGVs and 31% for articulate HGVs. These percentages in the BAU scenario will jump to 100% in 2030. This can be justified as follows. The Road Haulage Association (2018, p. 6) pointed out that the total HGVs fleet comprised 18% Euro VI HGVs in 2015 and 36% in 2017. In addition, it is estimated that a proportion of Euro VI HGV class will reach 63% by 2021 and 81% by 2025 of total HGVs. This suggests that the assumptions made by the Emissions Factors Toolkit that by 2030 the Euro VI would form the entire HGVs fleet can be considered realistic. In addition, from February 2019, Euro VI HGVs were eligible for a 10% reduction in the amount they pay, whereas HGVs that do not meet the latest Euro VI emissions standards will pay 20% more (DfT, 2018a, p. 64; DEFRA, 2019a, p. 46). This may possibly explain why NO_x emissions related to HGVs are expected be reduced significantly from 556 tonnes in 2014 to 51 tonnes by 2030.

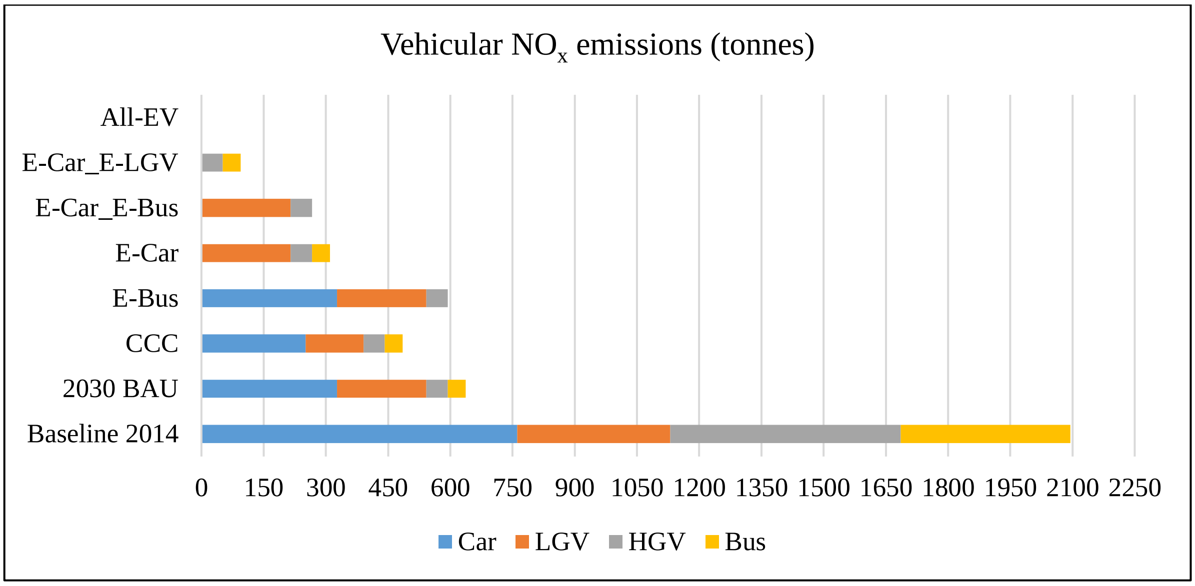


Figure 7-3: Vehicular NO_x emissions by vehicle type for all scenarios

These reductions in total vehicular emissions are likely to be compatible with the NECD, including 2001/81/EC and Directive 2016/2284/EU, in relation to targets for lowering NO_x and $PM_{2.5}$ total emissions for the period 2002-2030 compared to the 2005 total emissions inventory.

Moreover, it was found that NO_x emissions released by cars in the AQMA in Newcastle city centre in 2014, as estimated in the Baseline scenario, were slightly higher than in 2010. In this work, cars' NO_x emissions reached 47.5 tonnes, whilst Goodman et al. (2014, p. 97) found NO_x emissions equivalent to 43.9 tonnes. However, in the same area, the NO_x emissions attributed to HGVs in 2010 was 36.7 tonnes (Goodman *et al.*, 2014, p. 97), which is higher than the estimation of 27.6 tonnes found in research reported in this thesis. This reduction in NO_x emissions released by HGVs between 2010 and 2014 and corresponding improvement in air quality was attributed to HGV engine technology, as old and new HGVs were equipped with engines that are compatible with Euro VI standards. This reflects the urgency of logistics operators to register new vehicles prior to the enforcement of the sixth environmental standards in January 2014.

7.3 Air Quality for the BAU Scenario

Currently, several local authorities in the UK, including Newcastle and Gateshead, have failed to comply with the Ambient Air Quality Directive (AQD), where pollutant concentrations should not exceed certain thresholds. The UK at this point has surpassed the NO_2 limit in 37 out of the 43 zones (DEFRA, 2017a, p. 43). In addition, compliance with the AQD might be challenging in the UK , since the DEFRA (2015, p. 15) has forecast that, by 2020, the concentrations of NO_2 would breach the AQD objectives in London, Birmingham, Leeds, Nottingham, Derby and Southampton.

To investigate air quality levels in Newcastle and Gateshead by 2030 in the BAU scenario, the dispersion of emissions was modelled to estimate annual hourly mean concentrations using a resolution to 100 metre grids. The boundaries of the study area were reduced, as mentioned previously, to accelerate the modelling of emissions dispersion since it would take a prohibitive amount of computational time to model the entire area of Newcastle and Gateshead at such a resolution (Goodman *et al.*, 2014, p. 65). This is due to the limited resources dedicated to this research, given that ADMS-Urban software runs on a standard Windows machine. Hence, the boundaries of the study area were reduced to the lower-left Ordnance Survey (OS) coordinates of (422200, 558200) and upper-right OS coordinates of (428600, 569600) to include the most polluted areas the Air Quality Management Areas (AQMAs) in the centres of Newcastle and Gateshead.

7.3.1 Additional Inputs to the Modelling of Air Quality

To run the modelling of air quality by using ADMS-Urban, several inputs are required; for instance, emission rates associated with each road and their time varying emission factors which are necessary to create a diurnal emissions profile. These emissions were calculated by using EFT v8, and subsequently the PITHEM outputs were amended accordingly. Details of street canyons in the study area were assumed to be the same as in 2014 when they were obtained from the ‘Newcastle/Gateshead Low-Emission Zone Feasibility Study’ (Goodman *et al.*, 2014, p. 146) and were incorporated into the PITHEM outputs. Subsequently, all of the PITHEM outputs were entered into ADMS-Urban to run the air quality model.

A vital input was the meteorological records for 8,760 hours in 2030. It was difficult to predict hourly meteorological parameters, such as wind speed, wind direction, temperature, cloud cover and precipitation for 2030. It was beyond the scope of this research to forecast the weather for 2030 on an hourly basis, given that the aim of this research was to investigate the impact of increasing EV penetration on emissions, air quality and disease burden; although it is acknowledged that cold years typically escalate NO_x concentrations (Goodman *et al.*, 2014, p. 30). In addition, Woodcock *et al.* (2009) used previous meteorological parameters for 2006 to model future air quality in London by 2030.

Therefore, meteorological data to 2014 was used for 2030 in running the ADMS-Urban to assess changes in emission concentrations between 2014 and 2030.

Moreover, background emission concentrations in ADMS-Urban were set to zero. After excluding the contribution of roads, background emissions concentrations were added to the outputs of ADMS-Urban based on DEFRA figures.

7.3.2 Air Quality Impact of the 2030 BAU Scenario

In 2030, air quality would improve due to improvements in vehicle technology, such as nearly all of the vehicles would meet Euro 6/VI or higher standards. This is in addition to the presence of more EVs which would undoubtedly lead to reduction in exhaust emissions. In addition, a significant drop in background emissions between 2014 and 2030 is projected to take place. For example, background concentrations of NO_2 are expected to be typically reduced by 42% in Newcastle and Gateshead by 2030 compared to 2014 levels (DEFRA, 2016a).

Moreover, the average $PM_{2.5}$ concentration was estimated to be 6.1 µg/m^3 in the BAU scenario, which is lower than the finding of 8.2 µg/m^3 for average $PM_{2.5}$ concentration modelled by Woodcock *et al.* (2009) in their air quality modelling study relating to a 2030 business-as-usual scenario in London

The dispersion of emissions released by traffic activities in the 2030 BAU scenario was modelled. The result indicated that the highest concentrations of PM_{10} of 13.7 µg/m^3 and $PM_{2.5}$ of 8.3 µg/m^3 are located at Team Valley Trading Estate in Gateshead. At the same site, background emissions map produced by the DEFRA estimated that total concentrations (including road contributions) of PM_{10} and $PM_{2.5}$ would be 11.9 µg/m^3 and 7.4 µg/m^3 in 2030 (DEFRA, 2016a).

Given that in 2014, the background emissions map estimated total concentrations of 14.9 μg/m^3 and 10.4 μg/m^3 of PM_{10} and $PM_{2.5}$, which are lower than the modelled results of 16.3 μg/m^3 and 11.2 μg/m^3 for PM_{10} and $PM_{2.5}$ concentrations in the Baseline scenario, those values are within the thresholds set by the UK air quality objectives (DEFRA, 2016d), except for the modelled and background emissions values of $PM_{2.5}$ which are higher than the 10 μg/m^3 threshold set by the WHO (2018a).

In contrast, a significant improvement in NO_2 levels is estimated by the 2030 BAU scenario compared to the Baseline scenario. Hence, the highest level of modelled NO_2 concentrations in the BAU scenario is expected to be 22.8 μg/m^3 at the Tyne Bridge, which is 46% less than the breaching value of 49.1 μg/m^3 modelled for the 2014 Baseline scenario at the Bridge. At the same site, the background emissions map (including roads contribution) estimated 16.5 μg/m^3 in 2030 and 26.6 μg/m^3 in 2014 in terms of NO_2 concentrations (DEFRA, 2016a). It should be noted that in relation to the Tyne Bridge, Newcastle and Gateshead Councils may possibly apply a charging scheme in an attempt to reduce pollution and traffic congestion (Tompkins, 2018).

Table 7-1 compares pollutant concentrations as modelled in this thesis for 2014 Baseline and 2030 BAU scenarios for the highest NO_2, PM_{10} and $PM_{2.5}$ concentrations; and estimations made by DEFRA background emissions.

Table 7-1: Highest pollutant concentrations in the Baseline 2014 and BAU 2030; and DEFRA estimations.

Pollutant	Year	ADMS (modelled) + DEFRA (no road) (μg/m^3)	DEFRA Background map (all sources) (μg/m^3)
NO_2	2014	49.1	26.6
PM_{10}		16.3	14.9
$PM_{2.5}$		11.2	10.4
NO_2	2030	22.8	16.5
PM_{10}		13.7	11.9
$PM_{2.5}$		8.3	7.4

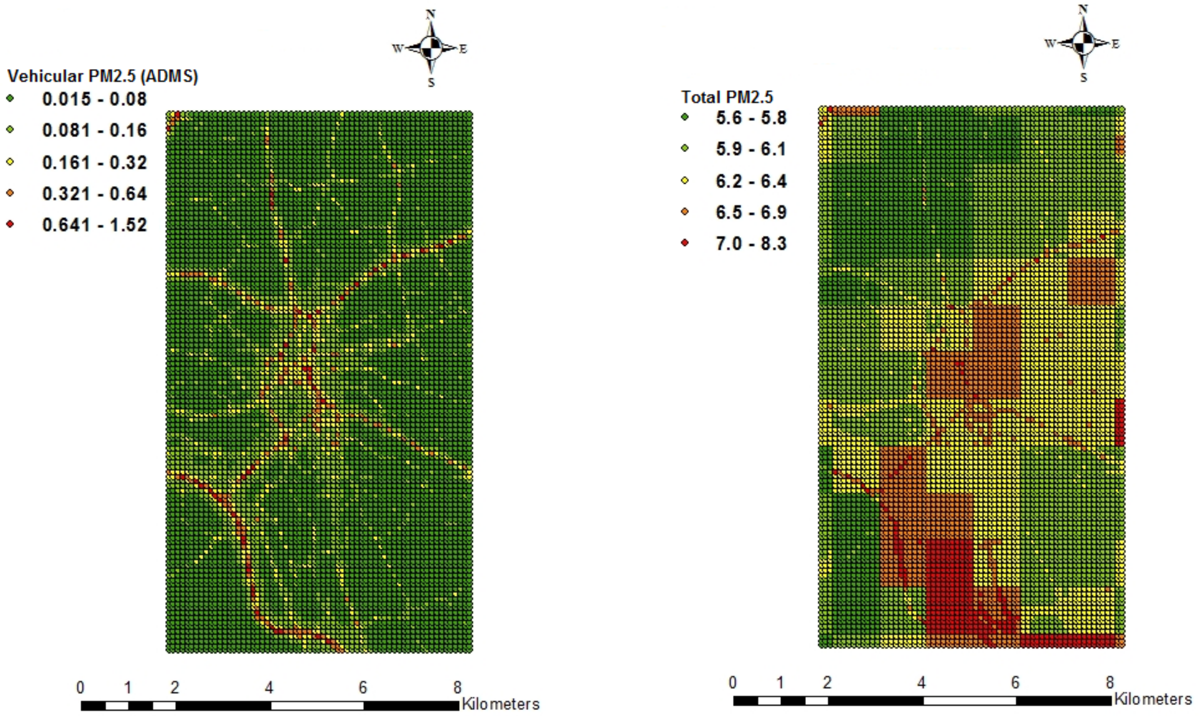
Figures 7-4, 7-5 and 7-6 reveal distributions of $PM_{2.5}$, PM_{10} and NO_2 levels with a resolution of 100 metres representing a component related to the AQMAs in the study area. Modelled vehicular emission concentrations are shown in the Figures on the left, whilst the Figures on the right represent the modelled vehicular and background map emissions concentrations adjusted for the contributions of roads.

The Figures show that of the $PM_{2.5}$ and PM_{10} concentrations, vehicular concentrations represent approximately 20% of the total. Whilst, vehicular emissions represent 50% of total NO_2 concentrations.

Plotting of the distribution of the differences in pollution concentrations would be more beneficial to identify 100m×100m grid square locations where the air pollution breaches the national targets and is still defined as an Air Quality Management Area (AQMA). Nevertheless, by 2030, it is expected that the air quality inside the AQMAs in Newcastle and Gateshead would have improved and the reasons for the AQMA declaration would have disappeared. The sixth Euro standard vehicles would form the majority of the vehicle fleet by 2030 as forecast by the Emissions Factors Toolkit (EFT). If vehicles equipped with engines compatible with the sixth Euro standard are achieved the promised reductions in real-world driving cycles, the level of pollution would be significantly lowered, seeing as the sixth Euro class, compared to the previous Euro standards, is extremely strict in relation to emissions allowances. For example, released NO_x emissions from a diesel car was 180 mg/km in the Euro 5 standards, whilst in the 6 Euro standards, the allowance for released NO_x emissions is reduced by 56% to 80 mg/km. This will undeniably remove all contribution from road-transport in the study area and AQMA in particular.

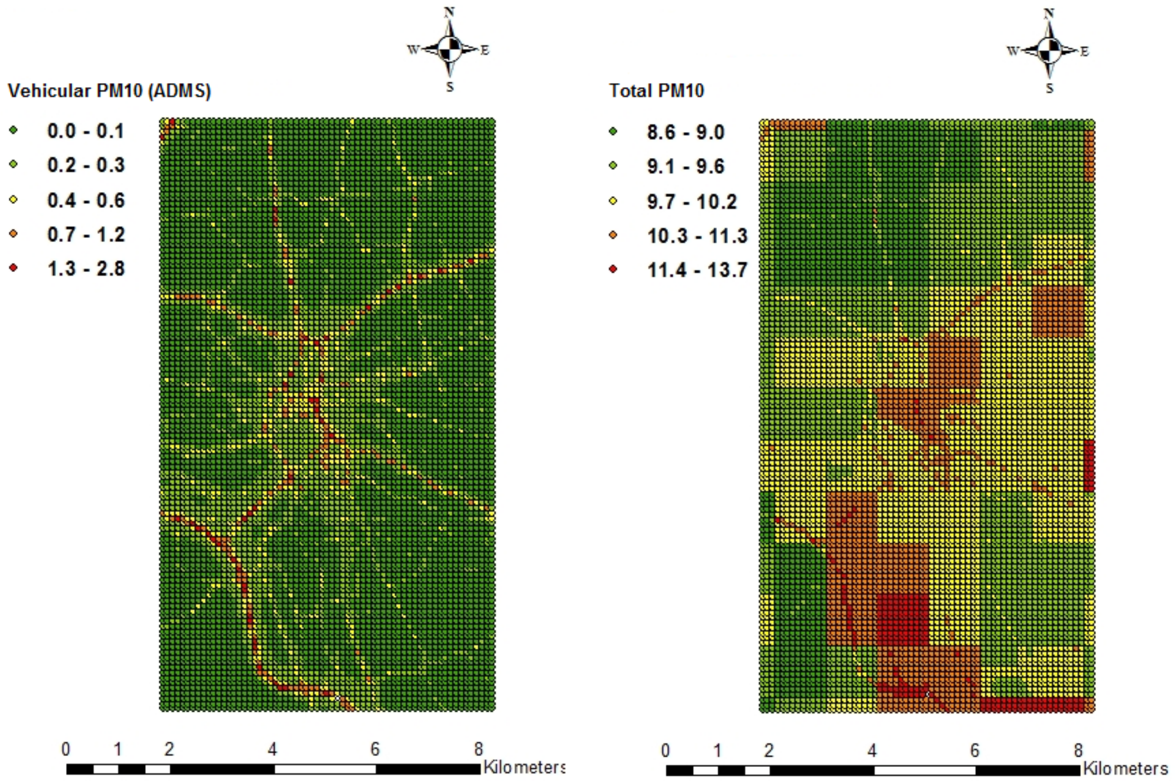
Furthermore, a section of roads bounded by lower-left Ordnance Survey (OS) coordinates of (422200, 428600) and upper-right coordinates of (558200, 569600), as presented in Figures 7-4, 7-5 and 7-6, was selected to run a high resolution air quality model of 100m×100m because this section covers all AQMAs in the study area. Air quality was modelled in this section considering all vehicular emissions in Newcastle and Gateshead using a 100m output grid resolution. These figures showed that all pollution concentrations would be within the thresholds as presented by the national targets.

It should be noted that using output with a resolution of 100m to model the entire study area gave lengthy estimated run times that may take months to be completed. To reduce the prolonged runtime, the number of link sections was limited to Newcastle/Gateshead urban core areas bounded by the abovementioned coordinates. These boundaries were previously used in the work conducted by Goodman et al. (2014, p. 65), who pointed out that the area confined between these boundaries is a good representation of the AQMAs in Newcastle and Gateshead.



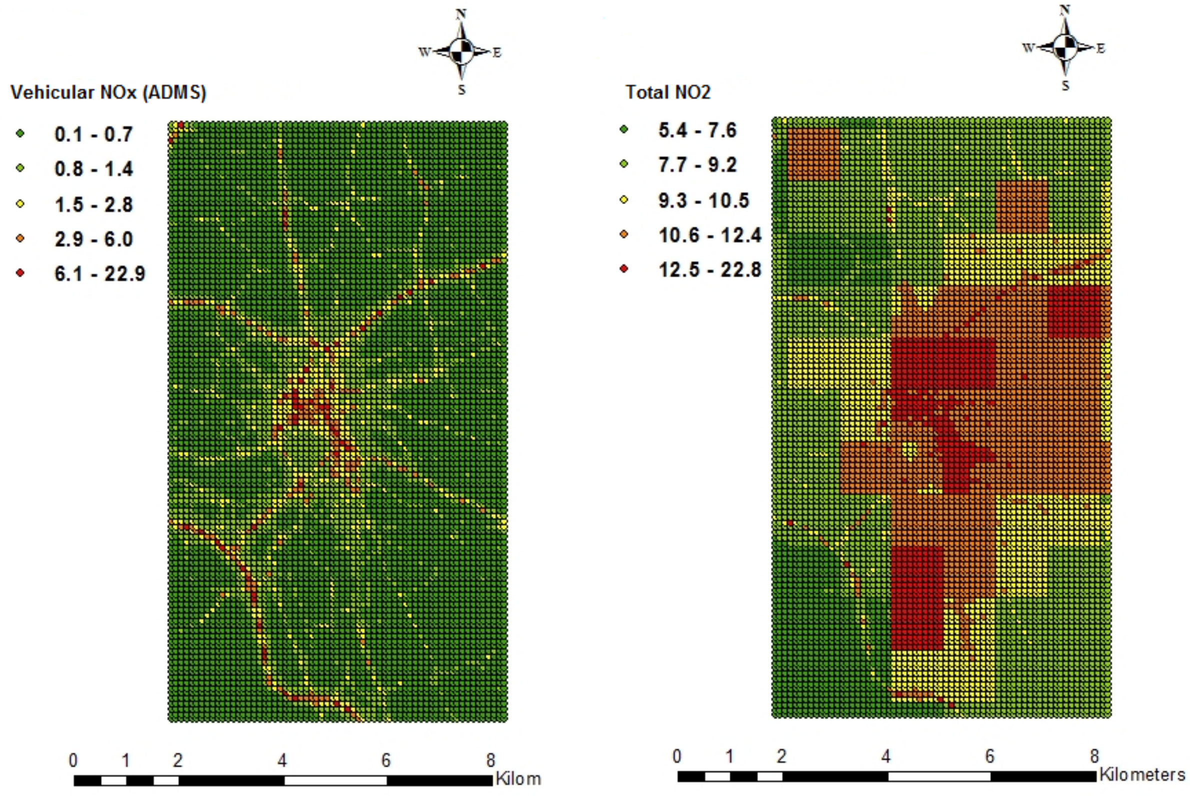
All values are annual hourly means in µg/m^3

Figure 7-4: Vehicular $PM_{2.5}$ concentrations (left), total $PM_{2.5}$ concentrations (right)



All values are annual hourly means in µg/m^3

Figure 7-5: Vehicular PM_{10} concentrations (left), total PM_{10} concentrations (right)



All values are annual hourly means in µg/m^3

Figure 7-6: Vehicular NO_x concentrations (left), total NO_2 concentrations (right)

7.4 Air Quality for 2030 the Scenarios at GP Sites

General practitioner (GP) sites were set as receptors during the modelling of air quality using ADMS-Urban for the Baseline and all 2030 scenarios, assuming that the population are exposed to the same pollution levels as those which occur at their GP sites. This assumption is based on people's tendency to join a neighbourhood GP (Santos *et al.*, 2017; Beghelli, 2018). Some patients' details such as their addresses are considered confidential and obtaining them required legal cover, and so the GP site was used to represent the exposure at each patient's home address. It is accepted that there may be exceptions to this, and these will be highlighted in this section.

7.4.1 Changes in $PM_{2.5}$ Concentrations

The pollution concentrations in all 2030 scenarios were compared to the 2014 Baseline scenario. The concentrations of $PM_{2.5}$ were estimated at all the GP sites. In general, vehicular emissions make low contributions to these concentrations. The highest involvement of vehicular emissions to $PM_{2.5}$ concentration is expected to take place at GP_44, the Ponteland Road Health Centre in Newcastle, where vehicular $PM_{2.5}$ of 0.9 µg/m^3 in the 2014 Baseline scenario and 0.4 µg/m^3 by 2030 in the BAU scenario. Total $PM_{2.5}$ concentrations was 9.0

μg/m^3 in the Baseline scenario which is expected to be reduced by 3.0 μg/m^3 to 6.0 μg/m^3 in the BAU scenario at this GP site. Reductions in other 2030 scenarios were the same for 3.0 μg/m^3 compared to the Baseline scenario at the same GP site.

The highest reduction of 3.4 μg/m^3 would take place at the site of GP_38, the Newcastle Medical Centre located in Eldon Square in the AQMAs, when the $PM_{2.5}$ concentration was modelled at 9.5 μg/m^3 in 2014 and 6.1 μg/m^3 in all 2030 scenarios. The vehicular $PM_{2.5}$ is expected to be reduced from 0.3 μg/m^3 to 0.1 μg/m^3. At another GP site located in the AQMA in Gateshead, a small contribution of vehicular emissions in $PM_{2.5}$ concentrations of 0.3 μg/m^3. Figure 7-7 shows $PM_{2.5}$ concentrations at selected GP sites located in AQMAs in Newcastle and Gateshead for the 2030 BAU and 2030 scenarios rather than from local vehicle flows.

7.4.2 Changes in PM_{10} Concentrations

Similar to $PM_{2.5}$, traffic emissions contributed an extremely small amount to PM_{10} concentrations at GP locations. The highest contribution of vehicular emissions of PM_{10} at 1.5 μg/m^3 was modelled at GP_44 in 2014 and this would be 0.7 μg/m^3 in the 2030 scenarios, out of total PM_{10} concentrations of 12.7 μg/m^3 in 2014, and from 9.4 μg/m^3 to 9.4 μg/m^3 in the 2030 scenarios.

At GP_09, the highest PM_{10} concentration in 2014 was modelled at 15.2 μg/m^3. In all 2030 scenarios, the expected PM_{10} level would be reduced to 12 μg/m^3 primarily due to the reduction in background source emissions other than traffic emissions.

The highest change of 3.4 μg/m^3 is expected to occur at the site of GP_35, the Metro Interchange Surgery in Gateshead, where the PM_{10} concentration was estimated to be 14 μg/m^3 in the 2014 Baseline scenario and 10.5 μg/m^3 by 2030 in all scenarios. Among the PM_{10} components released by industry, domestic, rail, residual salt and point sources, most PM_{10} components consist of residual salt which currently contributes more than 52% of the annual daily mean PM_{10} concentrations at this site. The results for PM_{10} concentrations at some GP sites inside the AQMAs in Newcastle and Gateshead are presented in Figure 7-8.

Although the exhaust emissions of particulate matter has been reduced by 2030 in all scenarios and eliminated completely in the All-EV scenarios, non-exhaust emissions can persist as a consequence of wear and tear regarding brakes and tyres and road dust resuspension (Woodcock *et al.*, 2009).

Emissions of $PM_{2.5}$ and PM_{10} comprise several components including vehicular, industrial, domestic, rail, residual salt and point sources. Given that the mean of $PM_{2.5}$ and PM_{10} concentrations estimated in the GP sites at Eldon Square Medical Centre in Newcastle and the Metro Interchange in Gateshead were calculated for only one hour, for example from 17:00 to 18:00 hour, the mean $PM_{2.5}$ and PM_{10} released from bus flow be higher than those from other particulate matters sources. This is because the buses do not operate 24 hours a day and yet the average $PM_{2.5}$ and PM_{10} are calculated based on 24 hours. Whilst other components of particulate matters such as residual salt, secondary particular matters are released over 24 hours and their $PM_{2.5}$ and PM_{10} is calculated based on 24 hours, although they fluctuated over the entire day. Likewise, concentrations of $PM_{2.5}$ and PM_{10} components are provided by DEFRA in annual hourly mean values as published in the background emissions map. Hence, the comparison of the annual hourly mean of $PM_{2.5}$ and PM_{10} between bus source and other sources, is biased toward residual salt. Furthermore, the UK national air quality standards emphasise that the threshold of a 1hour concentration of NO_2 is less than 200 µg/m^3 and not to be exceeded more than 18 times. Therefore, a similar 1 hour threshold should be legalised in relation to $PM_{2.5}$ and PM_{10} concentrations in the UK.

In addition, it is true the GP sites at Eldon Square Medical Centre in Newcastle and the Metro Interchange in Gateshead are located in commercial areas away from where most people reside. Conversely, at these sites, significant human exposure to air pollution takes place during the day because it is their place of work, shopping and leisure. It is important to investigate the causes of change in air pollution occurring at these sites, because it adds a degree of uncertainty when the impact on health is investigated. Ideally models should estimate the magnitude of the time individuals are exposed to different concentrations whilst at home, travelling, at work, at leisure whilst shopping etc., and sum the product of the duration and concentration for the entire population. The data for this type of analysis is unavailable; therefore, is out of the scope of this research.

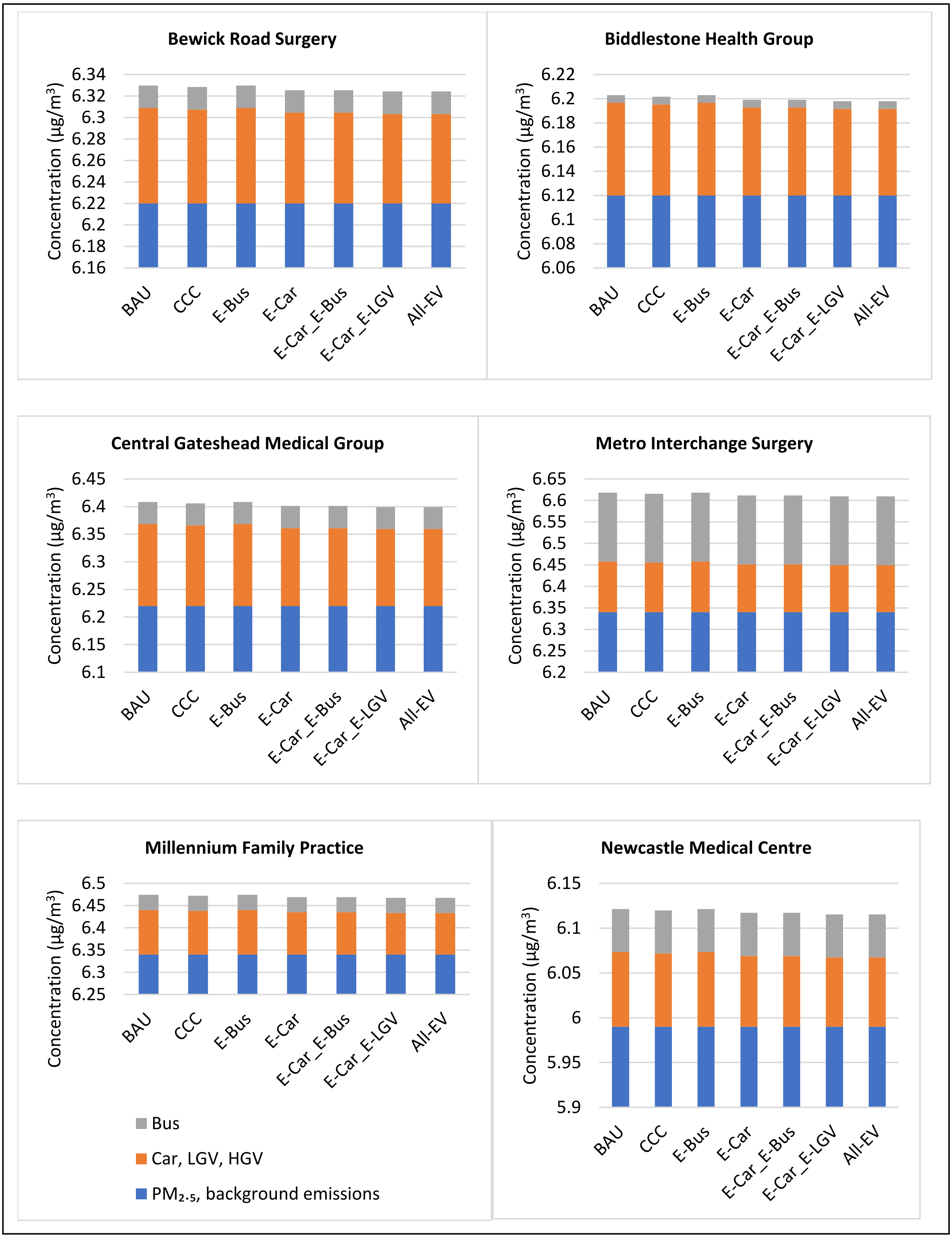


Figure 7-7: PM$_{2.5}$ concentrations at several GP sites between the BAU scenario and 2030 scenarios

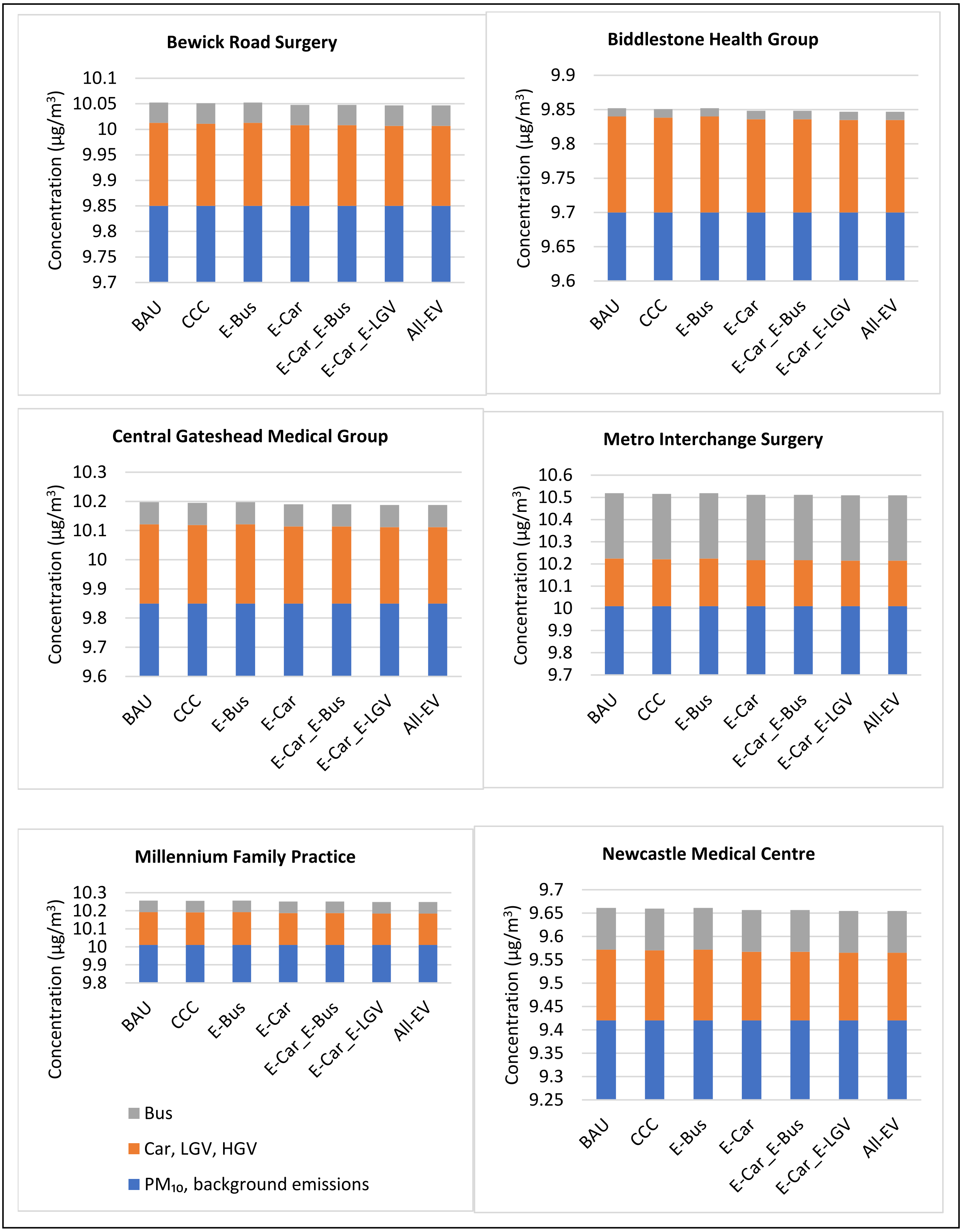


Figure 7-8: PM_{10} concentrations at several GP sites between the BAU scenario and 2030 scenarios

7.4.3 Change in NO_2 Concentrations

In relation to changes in NO_2 concentrations between 2014 and 2030, it was noticed that the greatest contribution of vehicular emission was at GP_35, the Metro Interchange Surgery representing 44% of 33.8 µg/m^3 of NO_2 concentrations in 2014. By 2030, this value would be reduced by 19.1 µg/m^3 to 14.6 µg/m^3 in the BAU scenario. In the other scenarios, the NO_2 concentration would be 14.5 µg/m^3 in CCC, 12.3 µg/m^3 in E-Bus, 14.3 µg/m^3 in E-Car, 12.0 µg/m^3 in E-Car_E-Bus, 14.0 µg/m^3 in E-Car_E-LGV and 11.7 µg/m^3 in All-EV. Thus, substituting existing buses with electric ones would lead to reductions in NO_2 at sites which are located close to bus stations, such as the Metro Interchange Surgery and Newcastle Medical Centre.

The concentrations of NO_2 at GP_36 and GP_56, which are in the vicinity of GP_35, the Metro Interchange Surgery, would be reduced compared to the Baseline scenario by 10.7 µg/m^3 and 13.1 µg/m^3 in the BAU scenario. The variations in reductions between these GP sites might be attributed to GP_35, which is extremely close to Jackson Street and subjected to diurnal low-speed traffic that varies between 7.5 to 11.5 mph, seeing as vehicles at such speed tend to release higher amounts of NO_2 emissions.

Detailed results for the reduction in NO_2 levels between the 2030 BAU and 2030 scenarios for some GP sites located in the AQMAs in Newcastle and Gateshead are presented in Figure 7-9.

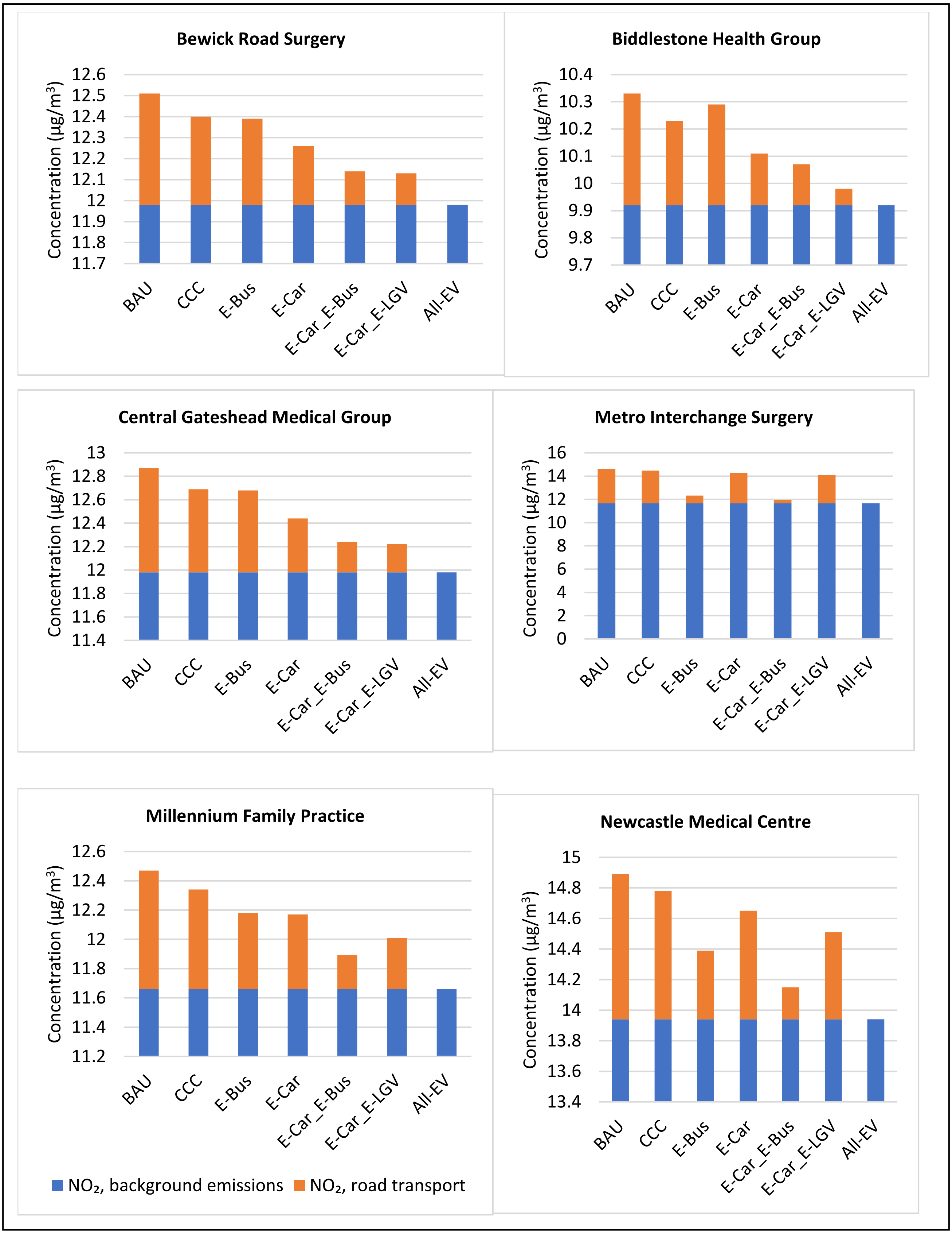


Figure 7-9: NO_2 concentrations at GP sites between the 2014 Baseline and 2030 scenario

In order to explore further the impact of the intervention scenarios the data was plotted to show the differences between the BAU and each scenario as shown in Figure 7-10.

From the graph in Figure 7-10, the effect of the different vehicle electrification scenarios shows significant reduction at all sites. However, where current air quality issues are experienced at GP practices located in proximity of bus stations such as GP_35 ‘Metro Interchange Surgery’; or next to roads with heavy traffic such as GP_44 ‘Poteland Road Health Centre’ reductions in NO_2 concentrations are higher. This is because of the reductions in released NO_2 from diesel cars of the sixth Euro compared to fifth standard and buses. Worthy of note is that by 2030 most of the vehicles would of the sixth Euro standard or higher. However, the performance of the diesel cars of the sixth Euro standards cannot be trusted. Given previous track records of the automotive industry and evidence of manipulation of the test procedure. In addition, although vehicle manufacturer are committed with lab and Real Driving Emissions (RDEs), Hooftman *et al.* (2018) reported that '*EU regulators have decided to allow diesel cars an exceedance of 2.1 times the 80 mg/km limit until 2020*'. Therefore, introduction of electric vehicles to mitigate air pollution is more reliable than improvement in the performance of ICE vehicles.

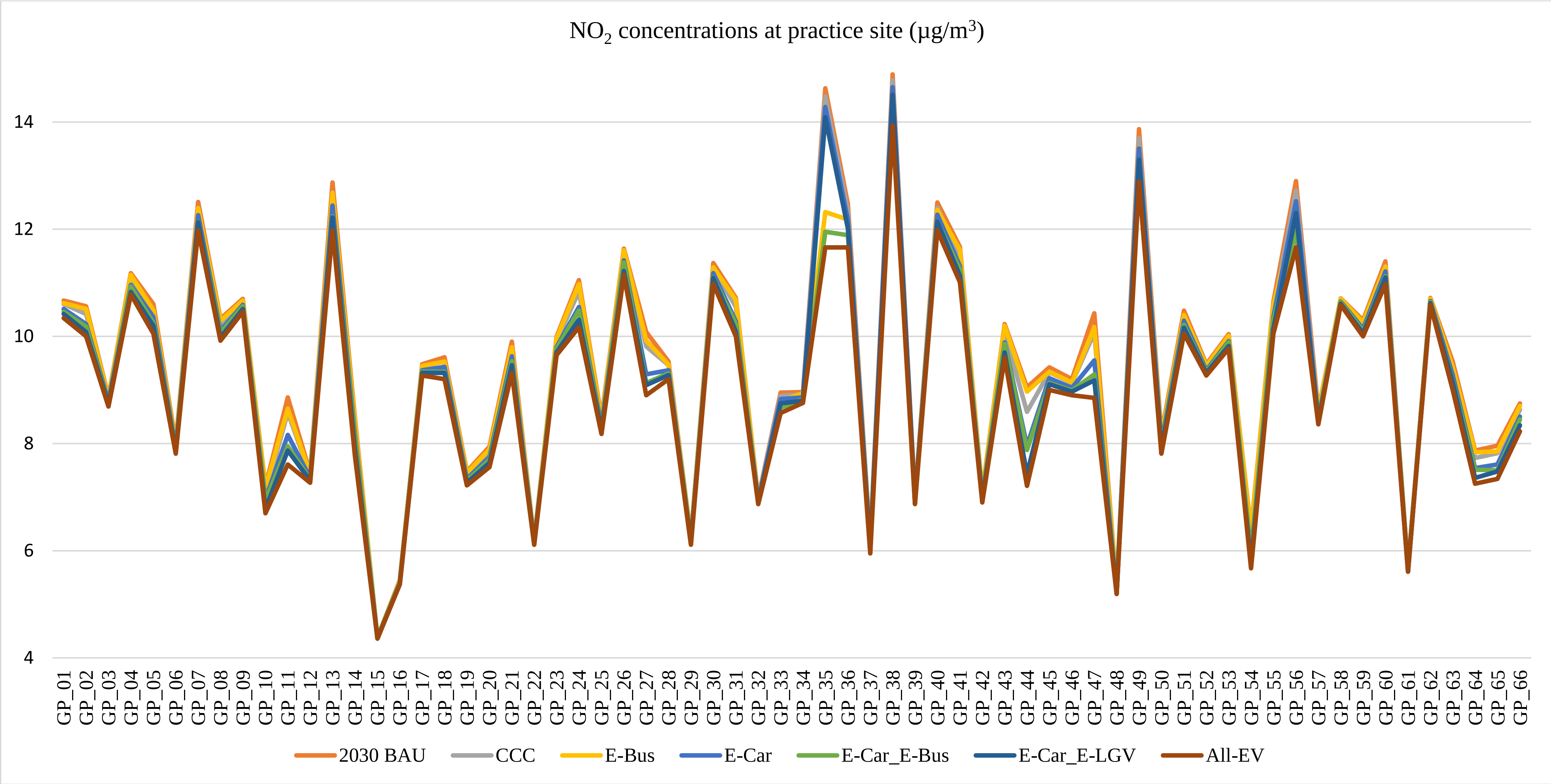
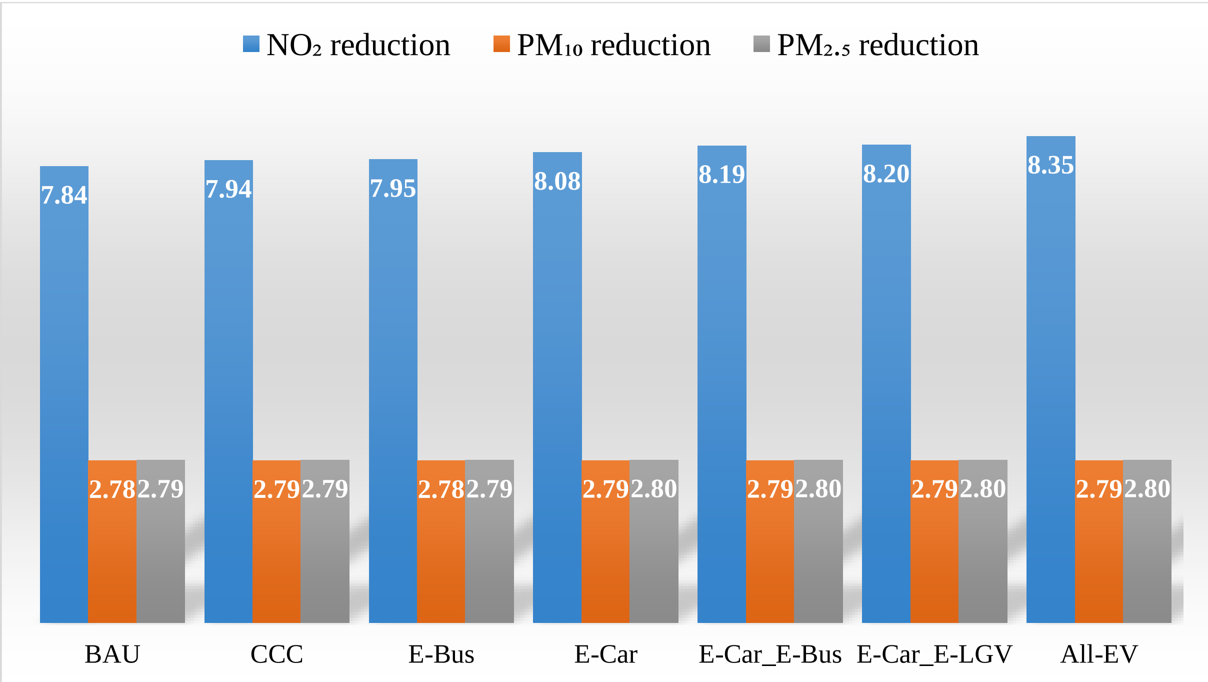


Figure 7-10: NO_2 differences in concentrations between the BAU and the practice sites for each scenario

7.4.4 Comments on Changes in Pollution Concentrations

Reductions in pollutants concentrations in geographic areas covered by 66 GP practices between the Baseline and the 2030 scenarios were averaged and presented in Figures 7-11. All scenarios indicate similar average reductions in annual mean levels of $PM_{2.5}$ and PM_{10} of around 2.8 μg/m^3. In relation to the average reductions in annual mean regarding NO_2 levels, there is a slight difference between all 2030 scenarios ranging from 7.8 to 8.3 μg/m^3, where the difference between BAU and the extreme vehicle electrification scenario All-EV, is only 0.50 μg/m^3.

Moreover, the implication of either electrification of both cars and buses or electrifying both cars and LGVs, would achieve similar outcomes for All-EV scenarios. Thus, investing in the transition to electric cars and electric buses and may cost less than investing in electrifying all vehicles types including HGVs.



All values are in μg/m^3

Figure 7-11: Average reductions in pollutant levels at all GP practices

Generally, the outcomes relating to the BAU are similar to other 2030 scenarios because EFT v8 assigns almost all vehicles the Euro 6/VI in the BAU scenario, whereas most of emission rates are greatly diminished compared to Euro 5/V. For example, NO_x emission thresholds have been lowered significantly to approximately 50% of previous Euro standards limits, as these standards which regulate limits regarding vehicle emissions rates are becoming stricter with each successive directive. Conventional vehicles could comply with these thresholds of

Euro standards under test conditions. Nonetheless, emissions released during real-world driving conditions have not decreased in line with the Euro 1/I –6/VI emission standards, particularly for diesel vehicles (Yang *et al.*, 2015; Department for Transport, 2016; EU, 2016; Moody and Tate, 2017; Cha *et al.*, 2019). Owing to the discrepancy between the amount of emissions during laboratory and road tests, the European Union has proposed the replacement of the European type-approval cycle (NEDC) with the worldwide harmonised light vehicle test procedure which adopts the Portable Emissions Measurement System (PEMS) (EU, 2016) from 2017 onwards (Yang *et al.*, 2015). In the PEMS, car manufacturers are permitted to exceed emission limits in the Euro standards emissions thresholds by 210% in 2017 and 150% by 2020 (Hooftman *et al.*, 2018). This implies that manufacturers of diesel cars have to comply with lab testing and furthermore, that they are strongly encouraged to reduce discrepancies between lab and PEMS testing in relation to over-released NO_x emissions until the 2020s. Given that all vehicles must be committed to emissions limits during the dynamometer type-approval testing (i.e. laboratory tests). However, discrepancies in the measured emissions between lab and Real Driving Emissions (RDEs) were identified particularly in diesel car emissions. Therefore, for the automotive manufacturers to be allowed sufficient lead-time to adapt strategies were put into place to give time to address and eliminate these discrepancies. A two-phase approach was implemented to reduce the RDE NO_x emissions instead of imposing direct compliance. Two thresholds of 80 mg/km limit, conformity factors (CF) – or multipliers were set as 2.1 in 2019 and 1.5 in 2020. Hooftman *et al.* (2018) reported that '*EU regulators have decided to allow diesel cars an exceedance of 2,1 times the 80 mg/km limit until 2020*'. Hence, car manufacturers have to comply with lab testing and have been encouraged to reduce discrepancies between lab and RDE testing.

Vehicle emissions such as NO_x, PM_{10} and $PM_{2.5}$ were calculated using the Emissions Factors Toolkit (EFT) developed by the DEFRA. Recent evidence indicates that real-world quantities of the emissions of NO_x, PM_{10} and $PM_{2.5}$ are greater than the quantities calculated using the EFT. Therefore, the DEFRA has issued updated emissions coefficients at different vehicle speeds in EFT v7 and v8 (DEFRA, 2016c, p. 4; DEFRA, 2017b, p. 7). However, the updated values may be unsatisfactory due to suspicions relating to the performance of the emissions control of Euro 6/VI vehicles, which are the dominant Euro standards in the BAU scenario, in compliance with emission thresholds. For example, vehicles tend to release greater emissions on roads with steeper gradients, through tunnels, or when vehicles have not been adequately maintained. Furthermore, as vehicles age the performance of their emissions control declines (Carslaw *et al.*, 2011a, p. 63; Hooftman *et al.*, 2018). Emission rates for $PM_{2.5}$, PM_{10} and NO_2

calculated by EFT v7 and v8 could possibly underestimate the reality (Hood *et al.*, 2018; Rushton *et al.*, 2018), which may explain the similarity in the BAU and other 2030 scenarios, in particular in releasing $PM_{2.5}$ and PM_{10} emissions.

The Emissions Factors Toolkit (EFT) (version 9), incorporates emissions based upon variable road gradients on each link and the inclusion of emissions based upon variable HGV loads on each link. Unfortunately, these features such as the impact of variable road gradients and HGV loading were not available on previous versions of the EFT i.e. version 8. Because version 9 was published on May 2019 after the submission of this thesis. The EFT does not estimate emissions, such as congestion occurrence (in case of conventional vehicles), also driver behaviour effects such as harsh acceleration, gear changing and braking also need to be addressed. This thesis recommends that investigation of these issues should be considered in future research. With regard to substitute of internal combustion vehicles with electric vehicles, the emission benefits would increase given electric vehicles do not emit exhaust emissions at point of use.

It can be argued that electric vehicles emission is at a minimum level during periods of congestion. This is for the reason that electric vehicles do not emit exhaust emissions. The only source would be the non-exhaust emissions which may be lower for electric vehicles because of range anxiety, drivers may drive more Eco-friendly reducing non-exhaust emissions as a consequence.

7.4.5 Determination of the Effect on Exposure Level Due to a Small Change in the Location of the Receptor Site

The results for the scenarios have prompted various questions regarding their reliability. Therefore, receptor modelling was performed at a 100 metre grid resolution (see Figure 7-12) in order to investigate the sensitivity to small changes in the location of the receptor sites.

Of particular interest is the GP number 38, the Newcastle Medical Centre, which was selected to perform the sensitivity analysis because it is located in the city centre where shoppers, workers, commuters and pedestrians are exposed to substantial amounts of air pollution. Additionally, this site is located close to a bus station where high levels of emissions are attributed to bus movements. Furthermore, the catchment area NE1, NE2, NE3 and NE4 accommodates the students of Newcastle University.

Figure 7-12 shows the distribution of the annual hourly mean of NO_2 concentrations around the GP site. The NO_2 concentration modelled at this site was 27 µg/m^3. In a circle with a diameter of 100 metres, the concentrations change from 3% to 11%. East of the GP site, the NO_2 concentration was found to change from 0 to 4% over a distance of 500 metres up approximately to the A167 road, where the NO_2 concentration jumps to 40 µg/m^3. Furthermore, West of the GP site, concentrations change from 3% to 9%.

However, southwest of the GP surgery, at a distance of 150 metres, the concentration increases to 60 µg/m^3. This noticeable increase in NO_2 levels is a result of the location of the main bus stations where a considerable number of buses are releasing primary NO_2. Bus services from the two bus stations (Eldon Square and Haymarket) link Newcastle city centre with Gateshead, the Metro Centre, Durham, Sunderland, Alnwick and various other places in the North East0. Moreover, 130 metres away from the GP site, in the Northwest direction, NO_2 levels increase by 60% due to the existence of a junction - traffic lights which link three roads opposite the main Haymarket metro station in the city centre.

In general, and consistent with the literature (section 4.5.3.4) the NO_2 concentrations appear to change slightly over distance up to hundreds metre, unless there is heavy traffic flow or congestion at a junction and/or the existence of a bus station or bus stations.

The public more around in these city centre and urban environments but spend much longer periods indoors at their place of work/education or at home. Also, given that exposures are averaged over longer periods variations over small spatial scales because smoothed levels of particulate matters in a similar way to NO_2 (given that particulate matter emissions is from the tailpipe of diesel vehicles and from the total fleet due to brake and tyre abrasion and wear and tears) also are sensitive to industrial sources. Industrial emissions are controlled well in the UK but there some residential areas affecting by the levels of particulate matters and relatively small changes in levels are observed at receptor sites.

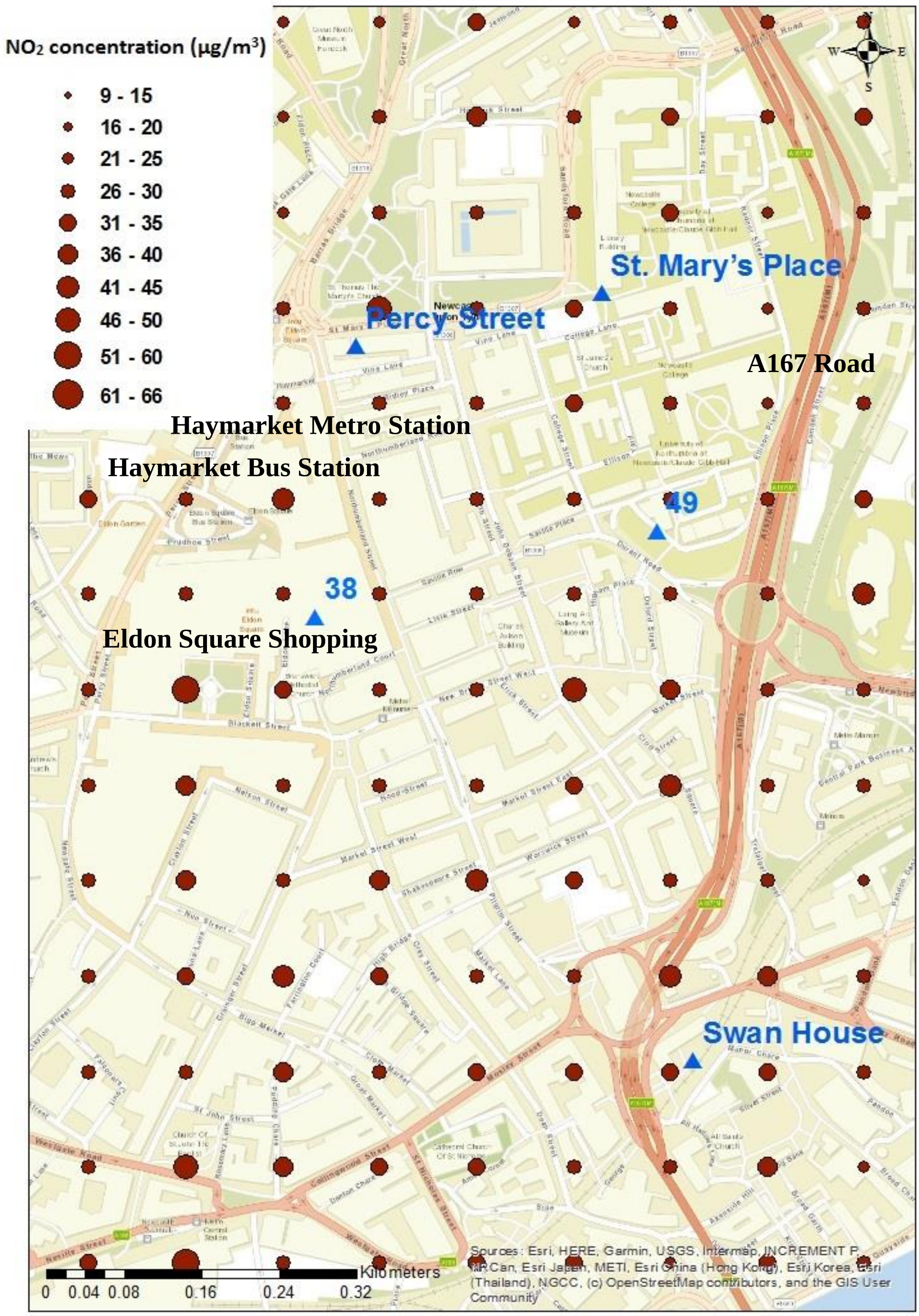


Figure 7-12: Distribution of NO_2 concentrations over 100 metre spacing

7.5 Disease Burden Due to Long-Term Exposure to Air Pollution

Disease burdens due to changes in long-term exposure to NO_2, PM_{10} and $PM_{2.5}$ of all 2030 scenarios was investigated by assuming that populations are exposed to the concentrations that took place at the GP sites close to their residential houses. Resulting disease burdens in terms of reducing premature deaths and admissions to hospital due to respiratory diseases were quantified by applying coefficients of the dose-response relationship concerning changes in the annual mean concentrations of pollutants between the 2014 Baseline and the 2030 traffic scenarios. The dose-response relationship coefficients describe the association between long-term exposure to 10 µg/m^3 increments of NO_2, PM_{10} and $PM_{2.5}$ with the probability of early mortality and being admitted to hospital, and the results are presented in Table 7-2. These coefficients were developed by the UK governmental affiliates and have been selected for used in this study.

Table 7-2: Dose-response relationships for long-term exposure of health outcomes

Pollutant	Mortality (per 10µg/m^3)	Reference	Hospitalisation (per 10µg/m^3)	Reference
NO_2	2.50%	COMEAP (2015a)	17.0%	Lee and Sarran (2015)
PM_{10}	7.0%	Carey *et al.* (2013)	8.0%	Lee and Sarran (2015)
$PM_{2.5}$	6.0%	COMEAP (2009)	32.0%	Lee and Sarran (2015)

7.5.1 Reductions in Hospitalisations

The changes in annual means of NO_2, PM_{10} and $PM_{2.5}$ concentrations between the Baseline and all 2030 scenarios were calculated for each of the 66 GP practices in Section 7.4. Subsequently, the appropriate dose-response coefficient was multiplied by each concentration change in order to quantify likely reduction the occurrence of hospital admissions for patients distributed based on their GP registration, as described in the following formula:

$$\textbf{Health gain} = \sum_{i}^{66}[(\textbf{Number of hospitalisations}) \times \frac{\mathbf{1}}{\mathbf{10}} (\textbf{Concentration change}) \times (\textbf{Response coefficient})]$$

where:

- Health gain: aggregated reductions in hospital admissions for patients registered with the 66 GPs;
- Number of hospitalisations: number of patients admitted to hospital in 2014 registered with a certain GP;
- Concentration change: annual mean concentration change between the Baseline and a 2030 scenario in µg/m^3 at a GP site;

- Dose-response coefficient: probability of being admitted to hospital associated with exposure per 10 μg/m^3 increments of a pollutant.

Health gains in terms of reductions in hospital admissions resulting from the mitigation of pollution levels have been quantified. Lee and Sarran (2015) suggested that the dose-response to each 10 μg/m^3 increase of NO_2, PM_{10} and $PM_{2.5}$ would be most likely to cause an increase in hospitalisations in England due to respiratory diseases of 17%, 8% and 32% respectively.

Mitigating NO_2 concentrations by 2030 in all scenarios would result in a greater reduction in hospitalisations compared to the health gains attributed to mitigating PM_{10} and $PM_{2.5}$ concentrations at most of the GP sites. For example, reductions in NO_2, PM_{10} and $PM_{2.5}$ concentrations would be 6.4 μg/m^3, 2.7 μg/m^3 and 2.8 μg/m^3 at the GP_03 site in the BAU scenario. The corresponding reductions in hospital admissions are estimated to be 12, 2 and 9 due to reductions in NO_2, PM_{10} and $PM_{2.5}$ concentrations respectively, given that 106 hospital admissions were officially recorded in 2014 for patients registered with GP_03.

By summing-up all reductions in hospital admissions which would be expected for patients registered with all 66 GPs, the total number of reductions in hospitalisation would be 1,286 due to reduction in NO_2 concentrations; 217 due to reductions in PM_{10} concentrations and 874 due to reductions in $PM_{2.5}$ concentrations in the 2030 BAU scenario compared to the 9,693 hospital admissions registered at all GPs over Newcastle and Gateshead in 2014.

Furthermore, mitigating $PM_{2.5}$ levels would produce more reductions in hospitalisations than NO_2 mitigation at the practices GP_09, GP_15 and GP_48. Thus, the number of hospitalisations avoided in the BAU scenario would reach 1,297 cases by 2030 due to lowering both the NO_2 and $PM_{2.5}$ levels. Figure 7-13 shows the expected reductions in respiratory hospital admissions in the 2030 BAU scenario compared to the 2014 Baseline due to the response to mitigations in NO_2, PM_{10} and $PM_{2.5}$ concentrations for patients in Newcastle and Gateshead broken down by GP practices.

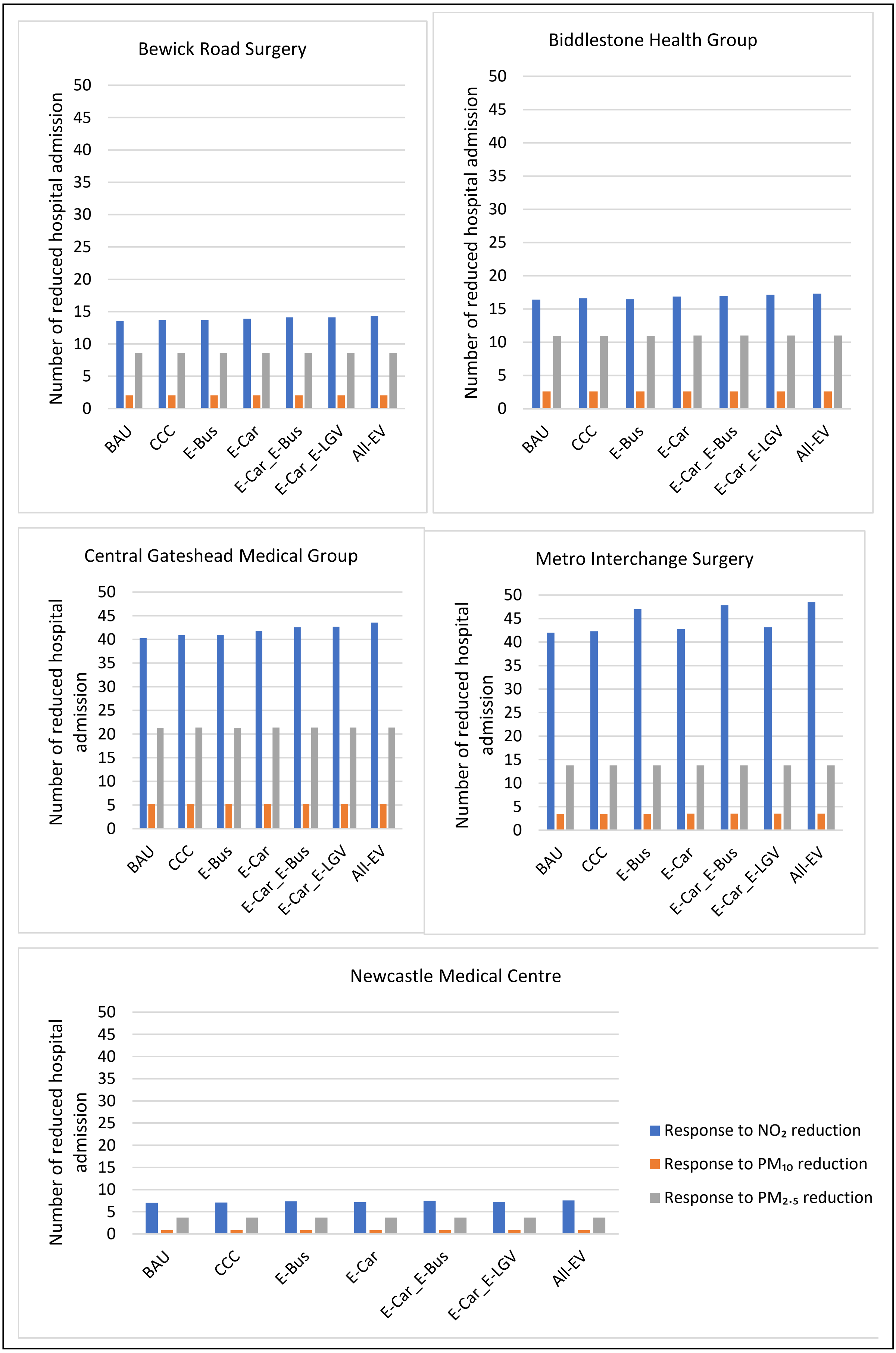


Figure 7-13: Reduced hospital admissions in response to reductions in $PM_{2.5}$, PM_{10} and NO_2 levels, 2030 BAU vs 2030 scenarios at some GP sites located in the AQMAs

In relation to the other 2030 scenarios, reductions in hospitalisations in response to lowering concentrations of NO_2, PM_{10} and $PM_{2.5}$ were aggregated from all GPs for all scenarios and would equal 1,313 for CCC, 1,315 for E-Bus, 1,352 for E-Car_E-Bus, 1,353 for E-Car_E-LGV and 1,377 for the All-EV scenarios compared to the 2014 Baseline, as Figure 7-14 demonstrates. An additional column is added to the graph for each scenario to represent the combined health gain resulting from lowering either NO_2 or $PM_{2.5}$ concentrations, whichever is greater. In the All-EV scenario, hospital admissions would be expected to be reduced to a greater extent than in the BAU scenario by 80 cases. Reductions in hospital admissions achieved in the E-Car_E-Bus scenario would be nearly similar to the reductions achieved in the E-Car_E-LGV scenario. Detailed hospitalisation reductions for all 2030 scenarios at all GP sites are presented in Appendix B.

In general, the 2030 scenarios could reduce the 9,693 hospital admissions recorded in 2014 by 14%. In China, the improvement in air quality that occurred between 2001 and 2010 were associated with reduced hospital admissions in 2010 by 31,810, representing 69% of the 46,738 hospitalisations officially registered in 2001. This vast improvement occurred in the city of Taiyuan because the local government shutdown 12 industrial factories in 2005 as well as another 121 factories by 2012. Moreover, it demolished 1,235 coal-fired boilers and modified nearly 400 boilers using clean fuel (Tang *et al.*, 2014).

In a study conducted in England between 2007 and 2011 to investigate the relationship between respiratory hospital admissions and long-term exposure to NO_2, O_3, $PM_{2.5}$, PM_{10} and SO_2 by 2050; Pannullo *et al.* (2017) ascertained that NO_2 has the greatest association with the rate of respiratory hospitalisations compared to the other pollutants. Similarly, the hospitalisation rate attributed to NO_2 would be lower than the present rate by 2050. This reflects the fact that reductions in NO_2 levels will lead to the largest health gains in term of reducing hospital admissions for respiratory problems in all of the 2030 scenarios in Newcastle and Gateshead compared to reducing PM_{10} and $PM_{2.5}$ levels. Indeed, improvements in air quality have a positive effect in reducing hospital admissions.
It is worth mentioning that, if the growth in population between 2014 and 2030 is taken into account in calculating the health gain from improving air quality in terms of respiratory hospitalisations, the reductions may be significantly higher.

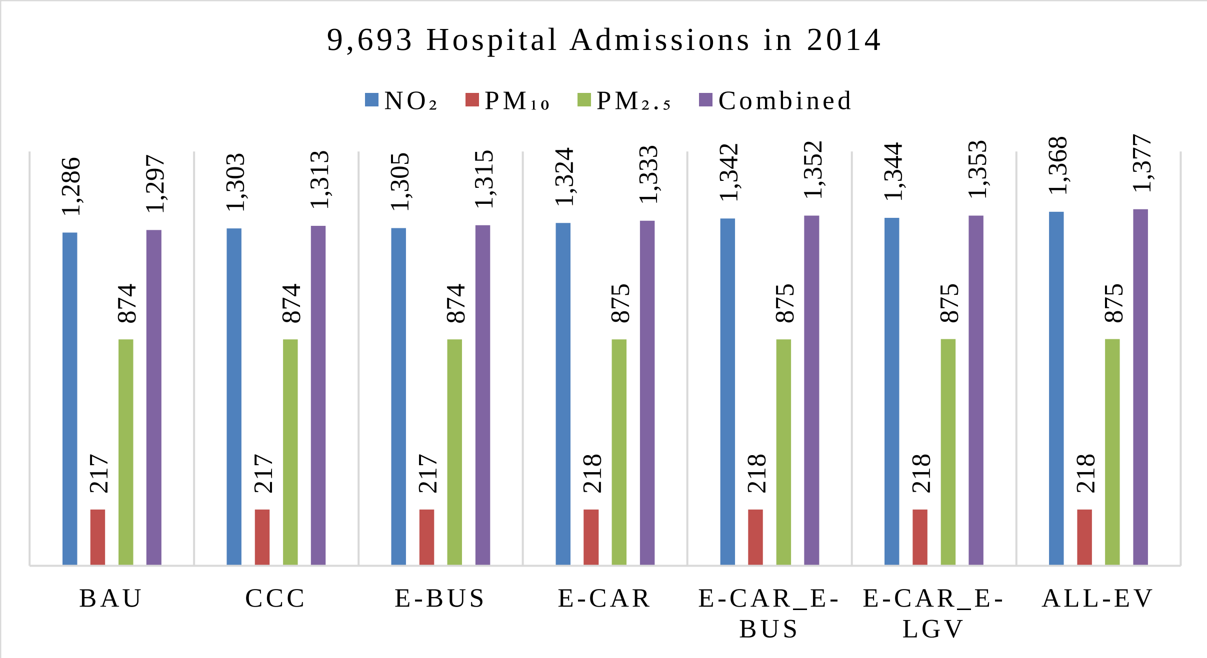


Figure 7-14: Reduced hospitalisations in response to pollution mitigation, 2014 vs 2030 scenarios

7.5.2 Reductions in Premature Deaths

The scale of likely reductions in premature deaths by 2030 was quantified based on the use of dose-response coefficients that describe the association between changes in pollutant concentrations between the 2014 Baseline 2014 and 2030 and the probability of early mortality. COMEAP (2015a) has shown that the dose-response to each 10 µg/m^3 increase in NO_2 concentration most probably causes a 2.5% increase in all-cause mortality, whilst a 7.0% increase in all-cause mortality is associated with a 10 µg/m^3 increase in PM_{10} levels (Carey *et al.*, 2013). A study conducted by COMEAP (2009) reported that a 10 µg/m^3 increase in $PM_{2.5}$, led to a 6.0% increase in all-cause mortality. All of these coefficients were developed by UK government affiliates and were determined to be appropriate to be applied in Newcastle and Gateshead in this present study.

Improving air quality should reduce exposure to air pollution and subsequently lead to a significant reduction in premature mortality (Tang *et al.*, 2014). In the study area, the reduction in premature deaths by 2030 as a result of improving air quality for each group of patients who had registered with the same GP was quantified using the following formula:

$$\textbf{Health gain} = \sum_{i}^{66}\left[(\textbf{Number of premature deaths}) \times \frac{1}{10}(\textbf{Concentration change}) \times (\textbf{Response coefficient})\right]$$

where:

- Health gain: aggregated reductions in premature deaths for patients who registered with 66 GPs;
- Number of premature deaths: number of mortalities occurring into each patients group in 2014 broken down by GP registrations;
- Concentration change: annual mean concentration change between the Baseline and a 2030 scenario in $\mu g/m^3$ at a GP site; and
- Dose-response coefficient: probability of premature death associated with exposure per 10 $\mu g/m^3$ increments of a pollutant.

Dose-response functions were applied to changes in NO_2, PM_{10} and $PM_{2.5}$ levels in terms of the numbers of deaths due to respiratory diseases among patients registered at each GP site. In 2014, 10 deaths due to respiratory burden were officially recorded for the patient group registered with GP_03. The change in PM_{10} concentration between the Baseline and BAU scenarios at this site would be 2.7 $\mu g/m^3$, which would lead to a health gain of a reduction of 0.19 deaths out of 10 cases. By summing up the reductions in early deaths for patients registered with all GPs in the study area, the expected reductions due to the mitigation of PM_{10} concentrations in the BAU scenario would be 14 incidents of premature death by 2030. Moreover, the reductions in NO_2 and $PM_{2.5}$ concentrations would produce reductions of 14 and 12 incidents of premature death. It should be noted that GPs numbered 1, 2, 4, 10, 11, 12, 15, 21, 27, 29, 32, 38, 42, 44, 46, 50, 51, 54 and 59 did not have any deaths attributed to respiratory diseases. However, reduction in PM_{10} concentrations would lead to the greatest reductions in early deaths at GP numbers 3, 6, 9, 16, 17, 18, 19, 20, 22, 23, 25, 26, 28, 33, 34, 37, 39, 41, 48, 52, 53, 57, 58, 61 and 62. The remaining GPs would witness the greatest reductions in early mortality attributed to a mitigation in NO_2 levels. Thus, the expected health gain from the BAU scenario would be a reduction of 15 with regard to early deaths. Figure 7-15 displays the expected reductions in early deaths for patients registered with 66 GP sites in Newcastle and Gateshead due to changes in NO_2, PM_{10} and $PM_{2.5}$ concentrations in the BAU scenario.

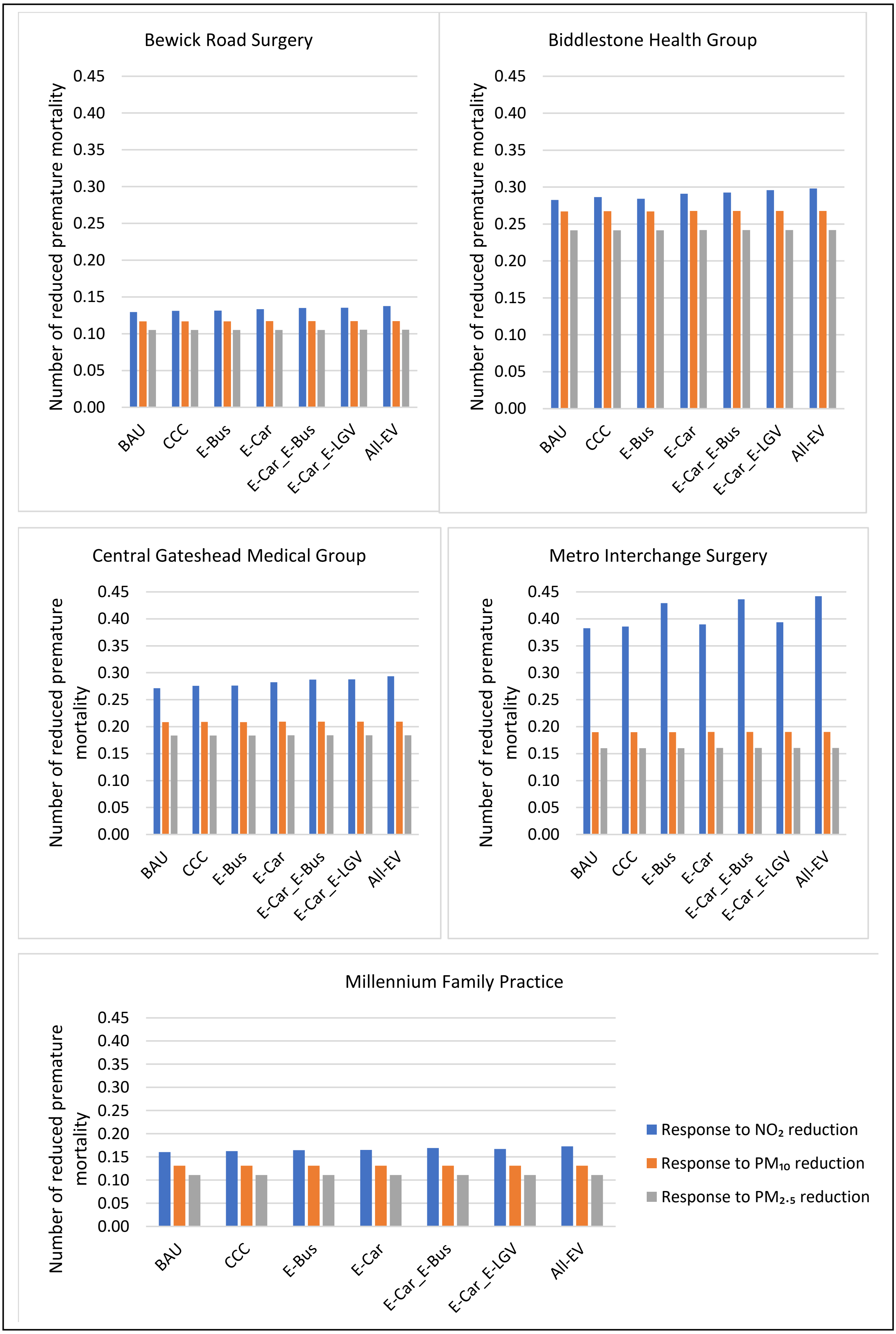


Figure 7-15: Reduced mortality in response to reductions in $PM_{2.5}$, PM_{10} and NO_2 levels, 2030 BAU vs 2030 scenarios at some GP sites located in the AQMAs

With regard to the other 2030 scenarios, in the All-EV scenarios, the expected health gains would include 16 fewer deaths, whereas for other scenarios there would be a reduction of 15 premature deaths. Figure 7-16 illustrates the reduction in mortality in response to reduced concentrations of NO_2, PM_{10} and $PM_{2.5}$ for all scenarios. The 'combined' column refers to the aggregation of the health gains from each GP practice resulting from the reduction of either NO_2 or PM_{10}, whichever is greater. Detailed health gains in terms of reductions in early mortality for all 2030 scenarios at all GP sites are presented in Appendix C.

Given that the population of Newcastle and Gateshead is approximately 400,000, the difference between All-EV and the BAU is only 1 fewer as regards early deaths. This means that the expected reduction in early deaths by 2030 would be approximately 2 per million. This finding is to some extent similar to findings from a study conducted in London and Delhi, India, on the impact of the introduction of low-carbon vehicles by 2030 compared to 2010. Woodcock *et al.* (2009) found that increasing the penetration of low-carbon vehicles would reduce premature deaths to 17 per million of the population in London (note population was estimated to be 8 million in 2011) due to a reduction in exposure to $PM_{2.5}$ concentrations by 2030 compared to the BAU. Similarly, the predicted level of annual mean $PM_{2.5}$ would decrease from 10 $\mu g/m^3$ in 2010 to 8 $\mu g/m^3$ by 2030 in London (Woodcock *et al.*, 2009). In Delhi, the expected reduction in premature mortality would be 74 per million of the population by 2030. This is for the reason that the pollution level is high compared to the UK, the average $PM_{2.5}$ concentration in Delhi was estimated to be 88.7 $\mu g/m^3$ in 2010 and 90.4 $\mu g/m^3$ by 2030 regarding business-as-usual (Woodcock *et al.*, 2009).

It should be noted that Woodcock *et al.* (2009) investigated the health impact of intervention on vehicle fleet composition by increasing the proportion of EVs, by means of replacing vehicle kilometres travelled (VKT) made by conventional vehicles and estimating the corresponding amount of vehicular emissions. Furthermore, dispersion was not simulated, and details of how the replacement of EV proportion carried out was not clearly presented, unlike in this thesis where spatial analysis of air quality was undertaken for each systematic change in EV fleets as described in earlier chapters.

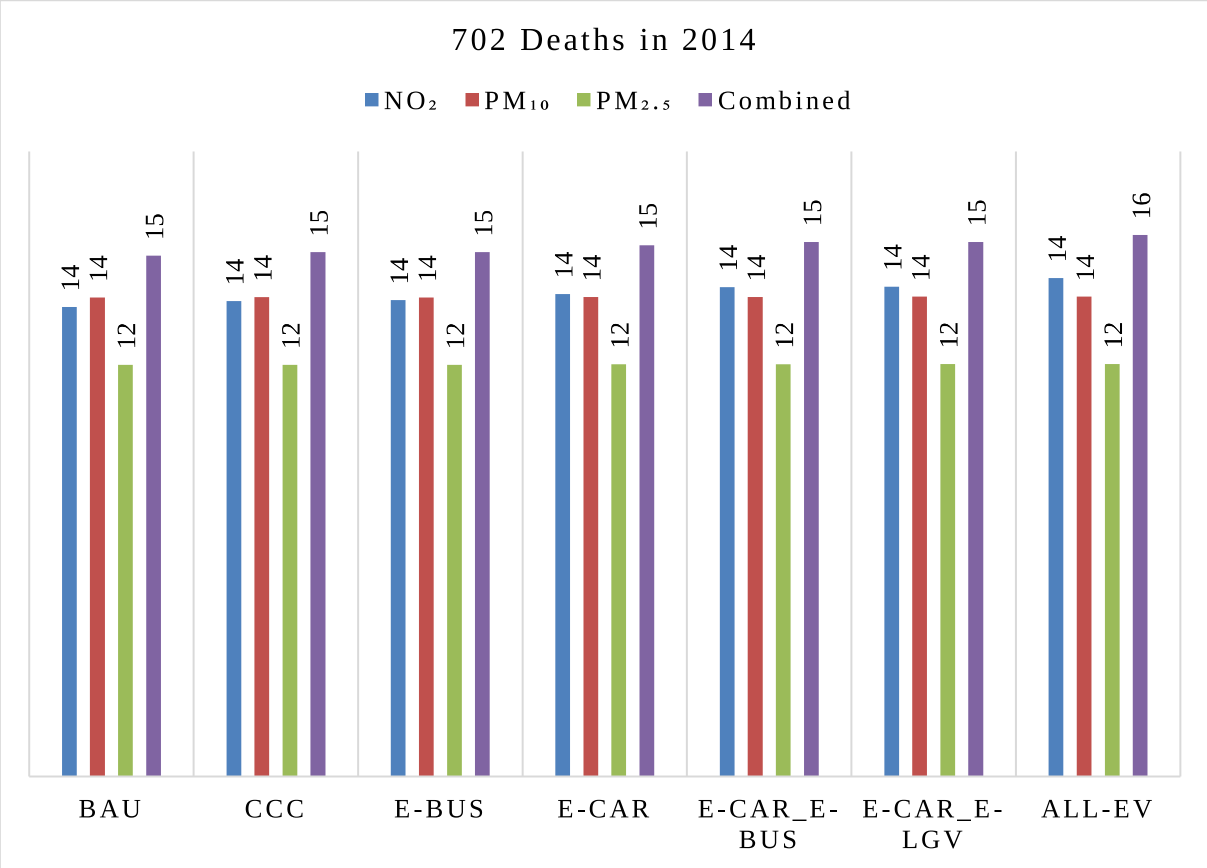


Figure 7-16: Reduced early mortality due to pollution reductions in several scenarios

7.6 Disease Burden Due to Short-Term Exposure to Air Pollution

Coefficients of dose-responses to short-term exposure to increases in NO_2, PM_{10} and $PM_{2.5}$ concentrations were used to quantify the health gains resulting from reduction in pollution levels. The estimated effect is expressed as excess risk per a 10 µg/m^3 daily increase in NO_2 exposure is 0.71% (Mills *et al.*, 2015), for PM_{10} it is 0.75% (COMEAP, 1998, p. 56) and for $PM_{2.5}$ 1.23% (WHO, 2013a) for all-cause mortality. Furthermore, the study conducted by the COMEAP (1998, p. 56) reported that a 10 µg/m^3 increase in daily NO_2 led to 0.50% excess risk and for daily PM_{10} concentrations led to a 0.80% increase in respiratory hospital admissions. Additionally, the same dose increase in $PM_{2.5}$ led to an extra risk of respiratory hospitalisation of 1.90% (WHO, 2013a). These coefficients are presented in Table 7-3.

Table 7-3: Excess risk of short-term exposure for mortality and hospitalisations

Pollutant	Mortality (per 10 µg/m^3)	References	Hospital admission (per 10 µg/m^3)	Reference
NO_2	0.71%	Mills *et al.* (2015)	0.50%	COMEAP (1998)
PM_{10}	0.75%	COMEAP (1998)	0.80%	COMEAP (1998)
$PM_{2.5}$	1.23%	WHO (2013a)	1.90%	WHO (2013a)

For each GP site, these coefficients were applied to the expected daily changes in NO_2, PM_{10} and $PM_{2.5}$ concentrations and the number of deaths and hospitalisations in order to quantify the daily health gain in terms of reducing hospitalisations and early deaths, as given in the following formula:

$$\textbf{Health gain} = \textbf{(Disease burden)} \times \frac{1}{10}\,\textbf{(Concentration change)} \times \textbf{(Dose-response coefficient)}$$

where:

- Health gain: aggregated reductions in hospitalisations or premature deaths for each patient group registered with a certain GP;
- Disease burden: mortality or hospitalisations occurring among patients in 2014 broken down by GP registration;
- Concentration change: reduction in daily dose of NO_2, PM_{10} and $PM_{2.5}$ concentrations between 2014 and a future scenario in $\mu g/m^3$ at a GP site;
- Dose-response coefficient: probability of premature death or hospitalisation associated with exposure to an increase in pollutant concentration.

7.6.1 Reduction in Hospitalisations

Improving air quality should lead to reductions in cases of hospital admissions (Tang *et al.*, 2014). The determination of likely health gains attributed to pollution alleviation in 2030 compared to the Baseline scenarios revealed that the expected reductions in $PM_{2.5}$ levels would result in more reductions in respiratory hospital admissions compared to other pollutants. For example, in the BAU scenario, the expected hospitalisation reductions for patients who registered with GP_03 due to mitigating $PM_{2.5}$ concentration is the greatest, followed by levels of NO_2 and PM_{10} being lowered, as can be observed in Figure 7-17. By averaging health gains in the same Figure for the 365 days of the year, the average reduction in hospital admissions for patients who registered with the GP_03 would be 0.56 incidents due to lowering $PM_{2.5}$ concentrations. Undertaking the same method for all 66 GPs as can be seen in Figure 7-18 and summing-up the average health gains from each GP, generates a health benefit of 52 reductions in hospitalisations from all GPs in the BAU scenario. Detailed hospitalisation reductions for all 2030 scenarios at all GP sites are presented in Appendix D.

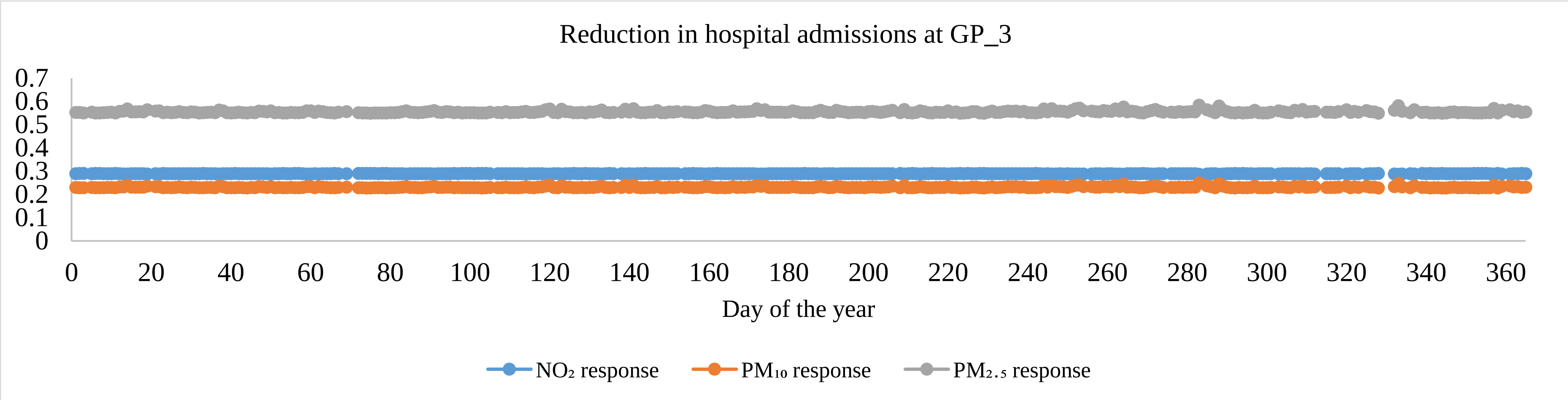


Figure 7-17: Expected daily reductions in hospital admissions at GP_3 due to pollution mitigation; BAU vs Baseline

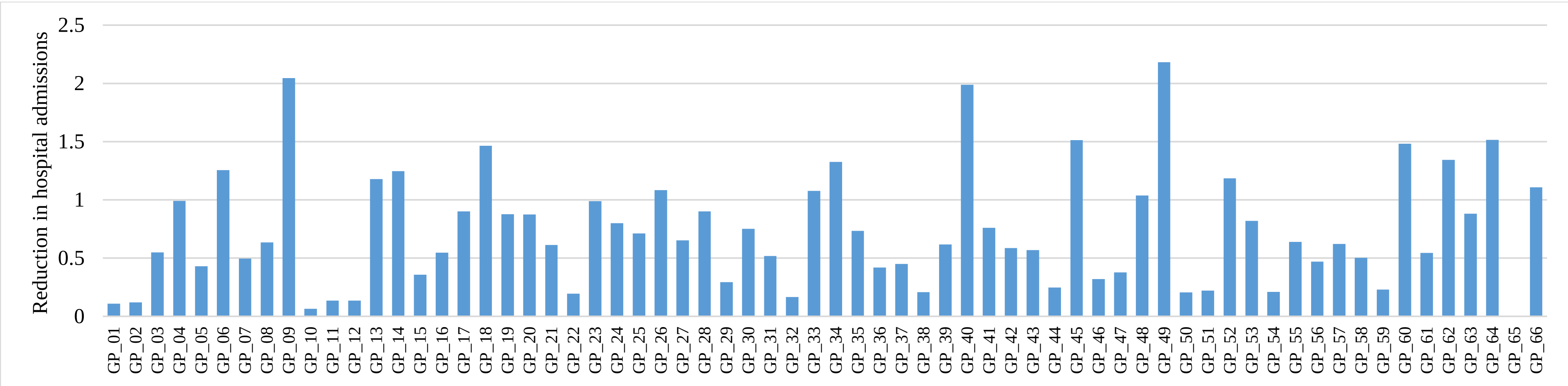


Figure 7-18: Average reduction in daily hospital admissions due to $PM_{2.5}$ reductions for 66 GP sites; BAU vs Baseline

Similarly, the lowering of $PM_{2.5}$ concentrations in the other 2030 scenarios indicated that the greatest health gains would be a reduction of 52 incidents of hospitalisations with all scenarios implemented. Vehicle electrifications can be seen to reduce admissions to hospital due to changes in the levels of NO_2 with reductions in hospitalisation ranging from 27 incidents in the BAU scenario, to 39 incidents in the CCC and E-Bus scenarios, to 40 incidents in the E-Car_E-Bus and E-Car_E-LGV scenarios and 41 incidents in All-EV. This is followed by a reduction of 22 incidents expected from lowering PM_{10} concentrations in all scenarios, as can be observed in Figure 7-19, which shows the expected reductions in hospital admissions on an annual basis for all 2030 scenarios. Given that the health gain is estimated based on daily changes in pollutant concentrations, then the health gain for all days of the year is averaged to form the annual health gain at each GP site.

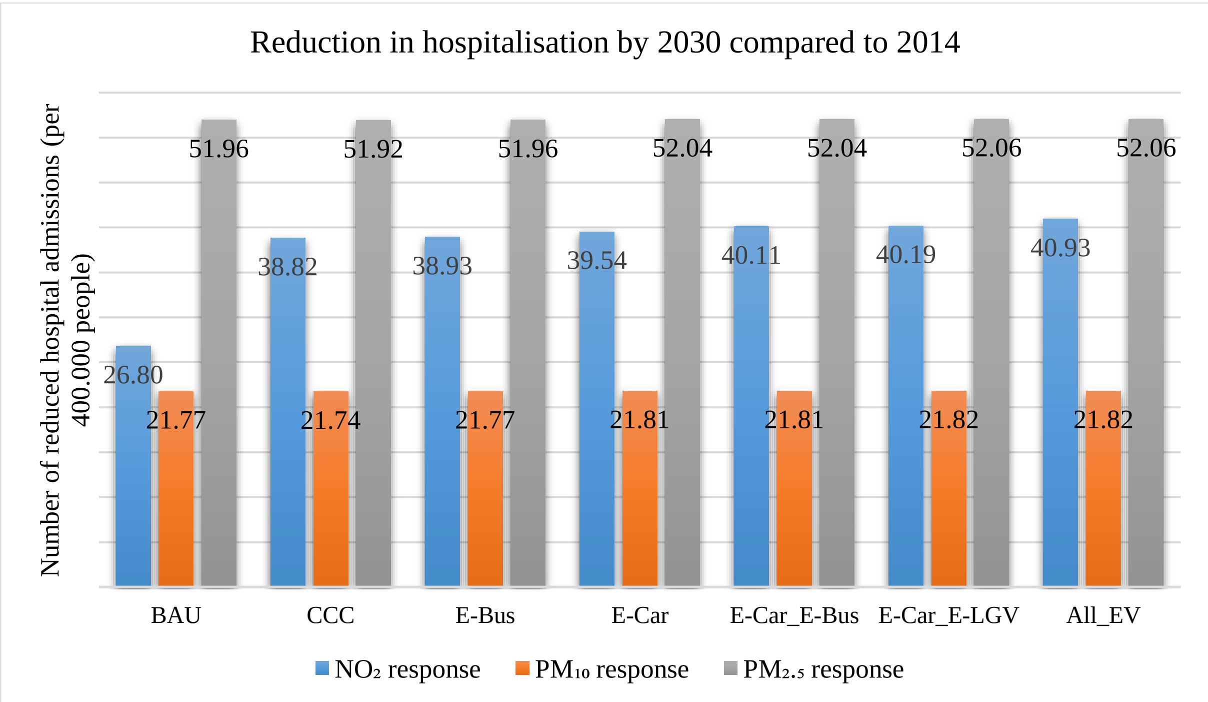


Figure 7-19: Reductions in hospitalisation over 66 surgery catchment areas attributed to the lowering of short-term exposure to all pollutants after the implementation of all 2030 scenarios

7.6.2 Reduction in Premature Deaths

Levels of NO_2, PM_{10} and $PM_{2.5}$ are projected to be lowered by 2030, which would improve the quality of air. According to Tang *et al.* (2014), as a result of air quality improvement, premature deaths are expected to be lowered. The mitigation of NO_2 levels would create more health gains than that of $PM_{2.5}$ or PM_{10} in terms of the numbers of annual premature deaths at most GP practices. For instance, the average reductions in numbers of premature deaths are expected to be 0.04, 0.02 and 0.03 in premature deaths due to the lowering of NO_2, PM_{10}

$PM_{2.5}$ concentrations respectively at GP_03, as revealed in Figure 7-20 for the BAU 2030 scenario. However, regarding the practices at GP_9, GP_26, GP_41, GP_48 and GP_61 the health gains attributed to lowering levels of $PM_{2.5}$ slightly outweigh those of reducing NO_2 levels. Therefore, health gains due to the mitigation of $PM_{2.5}$ levels were considered at these sites, as illustrated in Figure 7-21 for the BAU scenario. By summing the health gains at all GPs, a reduction in early deaths of 3 would be expected to occur by 2030 in the BAU scenario. Meanwhile the total number of deaths in 2014 was 702 and GP practices GP_1, GP_2, GP_4, GP_10, GP_11, GP_12, GP_15, GP_21, GP_27, GP_29, GP_32, GP_38 GP_42, GP_44, GP_46 GP_50, GP_51, GP_54 and GP_59 reported no deaths due to respiratory diseases. This may well explain the slight improvement in the prevention of premature deaths compared to reduced hospital admissions, which is indicated at all these sites.

The meteorological conditions data were obtained from the Met Office based on an hourly basis (i.e. 8760 lines of data to cover the whole year). Some of the data were missing because of maintenance, calibrations etc., at the time of the observations. If the line of data for any hour is missing, the ADMS-Urban would ignore this hour and provide a blank value for a pollutant concentration. Hence, the average pollutant concentrations for those days relating to missing met data will be calculated based on the available hourly concentrations and will not reflect genuine average 24 hours pollutant concentrations.

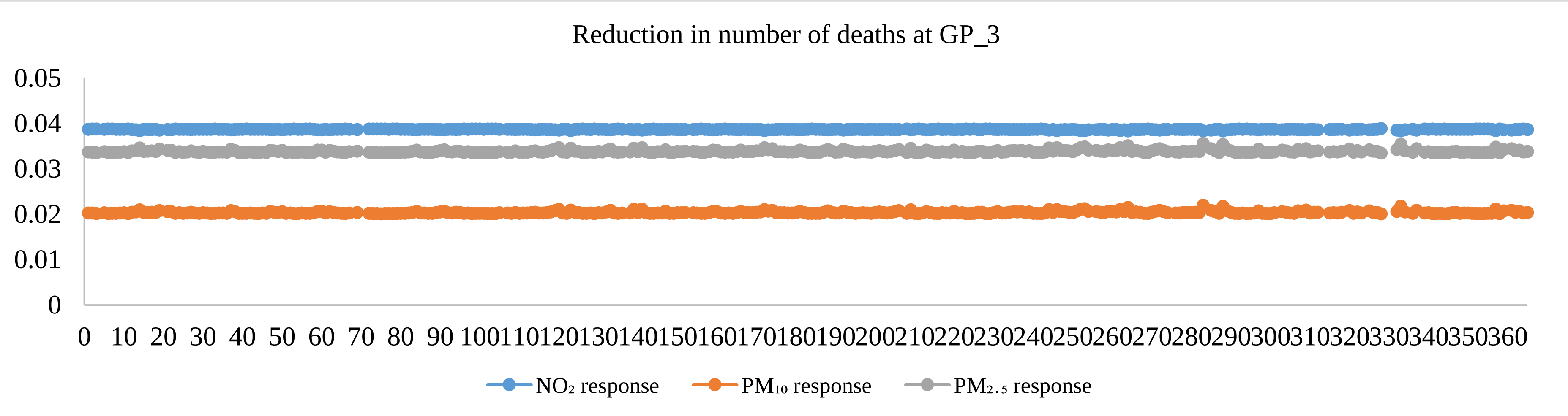


Figure 7-20: Expected daily reductions in premature deaths at GP_3 due to pollution mitigation; BAU 2030 vs Baseline

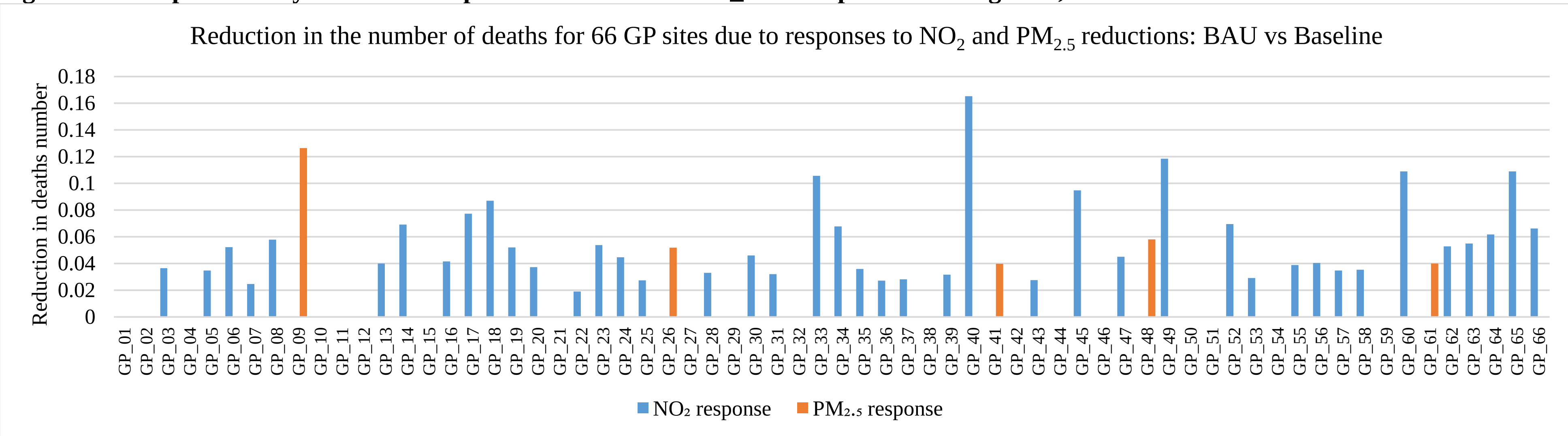


Figure 7-21: Average reduction in daily premature deaths due to $PM_{2.5}$ reductions for 66 GP sites; BAU 2030 vs Baseline

In the other 2030 scenarios, as the change in NO_2 would be greater compared to the BAU 2030 scenario, the expected number of reductions in premature deaths would be 4 incidents, as illustrated in Figure 7-22.

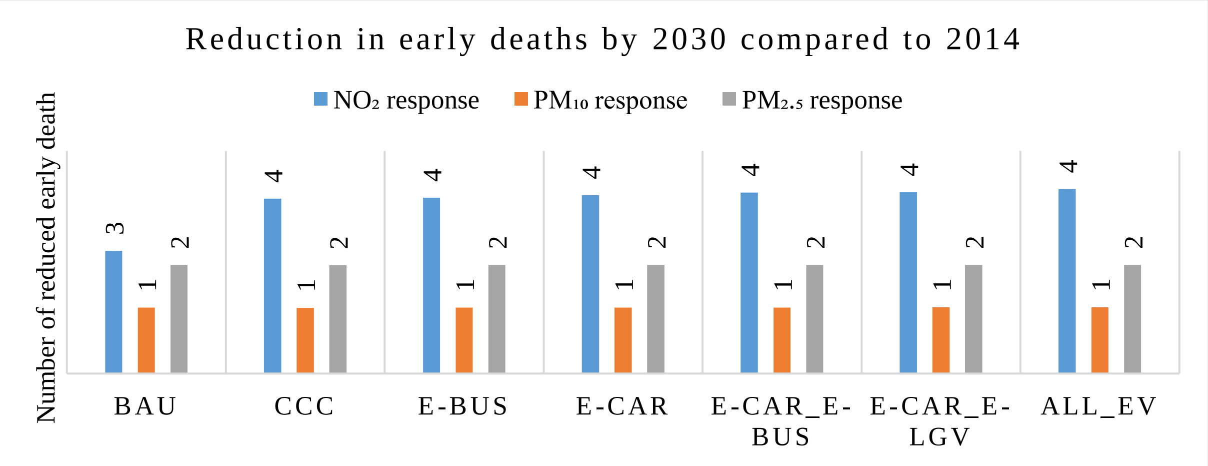


Figure 7-22: Reductions in premature deaths attributed to lowering short-term exposure to pollutants

The daily health gains of all of the 2030 scenarios are presented in Appendix E.

7.7 Discussion of Potential for Reduction in Mortality Overestimations

There are several uncertainties involved in the estimation of the mortality rates associated with exposure to outdoor air pollution. Over time, some of these could be reduced as new research is conducted. Nevertheless, several areas of uncertainty could be associated with any estimate. The main uncertainties are discussed briefly in this section.

Firstly, there is uncertainty in selecting a relative risk ratio identified in a specific study and used to assess the mortality in another region of the world. Studies of mortality estimates related to short-term exposure have been applied in several cities worldwide adding credibility to the procedure. Nevertheless, there is some ambiguity concerning the actual magnitude of the effect and the appropriate confidence interval, because long-term exposure is sensitive to the underlying assumptions. Secondly, uncertainty is associated with the general shape of the distribution of relative risk ratios and if there is a threshold concentration, for which there is no evidence in the literature and some studies have established health effects at very low concentrations of particulate matters (Ostro, 2004). Furthermore, most studies showed that there is a linear relationship between relative risk and ambient concentration within the range of exposures examined (Ostro, 2004). Thirdly, a major uncertainty involves co-pollutants. Specifically, it is possible that some of the estimated health effects of a pollutant is combined with other correlated pollutants for the reason that many of pollutants are from a common source such as the combustion of fuel. A fourth uncertainty concerns the baseline rates of the considered mortality in the studied population, given that it is assumed that there is a baseline occurrence level for the city or country of interest. However, the baseline occurrence could change over time as health habits, income and other factors change. A fifth uncertainty concerns the exposure assessment itself, which is based on the existing monitoring network and model-based estimates of NO_2, PM_{10} and $PM_{2.5}$ which are assumed to be representative of the general population.

In this thesis, most of the relative risk ratios employed were developed by research undertaken in the UK, such as the COMEAP, to reflect local occurrence. These ratios were used to quantify mortality numbers, such as exposure to an increase of 10 $\mu g/m^3$ in NO_2 concentration could probably cause a 2.5% increase in mortality (COMEAP, 2015a).

Additionally, in this thesis, the numbers of deaths by 2030 as a result of exposure to NO_2, PM_{10} and $PM_{2.5}$ are estimated to decline to between 12 to 14 cases in all 2030 scenarios, including the BAU scenario, compared to 2014 figures. Hence, there is no significant difference between the effects of each pollutant separately. This is the case for a 95% confidence interval regarding the probability of mortality as a result of exposure to NO_2, PM_{10} and $PM_{2.5}$.

In contrast, upper and lower estimates could be obtained by applying the upper and lower coefficients of the confidence intervals for estimating the relative risks. For NO_2, the reduction in mortality ranged between 6 to 23, and for PM_{10} the reduction in mortality ranged between 0 to 32, while for $PM_{2.5}$ the reduction in mortality ranged between 4 to 22. These ranges embrace the statistical uncertainty related to the mortality estimates.

7.8 Comparison with Other Studies

This thesis has investigated the impacts on the environment and health of several different interventions in vehicle fleet proportions introduced between 2014 and 2030. The investigation was carried out by applying models to simulate emissions and air quality in response to a range of vehicle mix scenarios. The assessment of the disease burden was considered by implementing dose-response functions which estimate the probability of death and hospital admission occurrences as a result to exposure to certain change in the amounts of air pollutants.

In Spain, Soret *et al.* (2014) conducted similar work in modelling the amounts and dispersion of vehicular emissions in response to three scenarios of conventional vehicle replacement with 13%, 26% and 40% of electric vehicles. However, modelling did not consider impacts for future growth in traffic along with proportions to forecast future interventions nor was the health impact examined. Potential reductions in pollutant emissions in Barcelona and Madrid of 5% and 4% in PM_{10} discharges and 27% and 25% in NO_x emanations were estimated. These improvements were based on the replacement of 40% of conventional vehicles with EVs in 2011. These comparisons were made *before* and *after* introduction of electric vehicles in Spanish fleet for the same year in 2011.

In this thesis, comparisons were made between the 2014 Baseline scenario and the 2030 scenarios. The reductions in vehicular emissions for the 2030 scenarios were compared to the 2014 Baseline scenario. The $PM_{2.5}$ emissions were found in 2030 scenarios to be lowered by 18% to 23%; whilst the PM_{10} emissions were found in 2030 scenarios to be lowered by 5% to 8% in comparison to the Baseline scenario. It should be noted that the quantity of $PM_{2.5}$ emissions was 60% of the quantity of PM_{10} emissions in 2014. In all the 2030 scenarios, quantities of $PM_{2.5}$ emission were 53% of PM_{10} emissions. This might explain higher reduction in $PM_{2.5}$ emissions than PM_{10} emissions. The reduction in PM_{10} emissions reductions are greater than those modelled in the research conducted by Soret *et al.* (2014), because in Barcelona and Madrid the comparisons were made for the same year 2011, whilst in this thesis the comparisons were conducted between 2014 and 2030 when all vehicles will be expected to be Euro 6 standard or higher.

Studies conducted by Buekers *et al.* (2014), Woodcock *et al.* (2009) and Dey *et al.* (2018a) have investigated the health impact of interventions in vehicle fleet share by increasing the proportion of EVs. However, it should be mentioned that those studies relied on replacing VKT by conventional vehicles and the amount of vehicular emissions and the dispersion methods were not simulated in the same way as this thesis where a spatial analysis of air quality was undertaken.

Gehrsitz (2017) performed a regression model to correlate monitored pollution levels in Germany and infant health before and after the establishment of low emission zones in an attempt to investigate the influence of such zones on air quality and birth outcomes. The study showed that the introduction of the most restrictive type of low emission zone reduces average concentrations of particulate matter by 4% up to 8% at a city's highest-polluting monitor. However, although the health outcomes resulting from reducing vehicular emissions were investigated, that study focused on birth health and not respiratory diseases as in this thesis. Table 7-4 illustrates the findings of some of the key studies in relation to interventions in vehicle fleet share and the impact on the environment and health. The main difficulty in making comparisons of this work with the literature is their modelling approaches and basic assumption are different.

Table 7-4: Environmental and health outcomes in various studies from interventions in vehicle fleet composition

Study	Location	Year		Method	Environmental and health outcomes
		Base	Target		
Buekers *et al.* (2014)	EU-27	2010	2030	Replacing 5% VKT by ICEVs with EVs.	In the UK, expected annual monetary benefit of €30.3 million and €46.6 million by avoiding external costs in 2011 and 2030.
Soret *et al.* (2014)	Barcelona	2011	2011	Modelling 40% of EV penetration scenario, using a chain of models to simulate emissions and dispersions.	Reduction in $PM_{2.5}$ emissions by 5%. Reduction in PM_{10} emissions by 4%. Reduction in NO_x emissions by 11%.
	Madrid				Reduction in $PM_{2.5}$ emissions by 22%. Reduction in PM_{10} emissions by 3%. Reduction in NO_x emissions by 17%.
Woodcock *et al.* (2009)	London	2030	2030	2030 scenarios assuming 35% (BAU) and 60% reduction in transport CO_2 emissions from 1990 levels. Replacing ICEV VKT with EV VKT. No mention of EV proportion.	Concentration of $PM_{2.5}$ reduced by 7.4 to 7.8 μg/m^3. Deaths reduced by 33.
Gehrsitz (2017)	Germany	2005	2012	Regression model drawn to correlate pollution levels and infant health	Concentration of PM_{10} reduced by 4 to 8%.
Dey *et al.* (2018a)	Dublin	2015	2030	EVs penetration: 50% of market penetration External cost of $PM_{2.5}$: 700 DALYs/kton External cost of NO_2: 90 DALYs/kton	Reduction in $PM_{2.5}$ emissions by 52%. Reduction in PM_{10} emissions by 39%. Reduction in NO_x emissions by 47%. Savings of 300 DALYs and £43.8 million.

7.9 Summary

The estimation of emissions rates and dispersion released from traffic activities for the BAU scenario and six scenarios for 2030 have been modelled and their impact on air quality and health investigated.

Emission rates modelling indicates that total $PM_{2.5}$ emissions are estimated to be reduced from 95.3 tonnes in 2014 by 17.5% to 78.5 tonnes in the 2030 BAU scenario. Full electrification of the bus stock only (the E-Bus scenario), would result in the lowest reduction of 17.9% in $PM_{2.5}$ emissions compared to the other 2030 scenarios. The All-EV scenario would result in a reduction of 22% followed by a 21.5% reduction regarding the E-Car_E-LGV scenario.

The PM_{10} inventory of emissions inventory released from road transport in 2014 is estimated to total 152.5 tonnes. In the BAU scenario, this would be lowered by 5.2% to 144.8 tonnes. The All-EV scenario would reduce it by 8.5% and there would be reductions of 7.8% with respect to the E-Car_E-Bus and E-Car_E-LGV scenarios.

Total emissions of NO_x emissions should witness a massive reduction in 2030 compared to the 2014 Baseline including a reduction from 2,100 tonnes of NO_x emissions to 637 tonnes in the BAU scenario to 593 tonnes in the E-Bus scenario, 485.5 in the CCC scenario, 310 in the E-Car scenario, 266.5 in the E-Car_E-Bus scenario, 95 tonnes in the E-Car_E-LGV scenario and zero NO_x emissions in the All-EV scenario. The E-Car_E-LGV scenario involving the electrification of the entire car and LGV fleet, would achieve outcomes at lower cost similar to the All-EV scenario outcomes in relation to the total $PM_{2.5}$, PM_{10} and NO_x emissions inventory.

All pollutant concentrations in the air quality modelling were compiled with the threshold set by the Ambient Air Quality Directive (AQD), which is that no pollutant concentration should be found in the 2030 which breaches AQD targets. Reductions in emissions concentrations modelled at GP sites varied between the 2030 scenarios compared to the 2014 Baseline. For example, the maximum reduction in NO_2 concentrations compared to the Baseline would occur at GP_35 of 19.1 $\mu g/m^3$ in the BAU scenario, 19.3 $\mu g/m^3$ in the CCC scenario, 19.5 $\mu g/m^3$ in the E-Car scenario, 19.7 $\mu g/m^3$ in the E-Car_E-LGV scenario, 21.6 $\mu g/m^3$ in the E-Bus scenario, 21.8 $\mu g/m^3$ in the E-Car_E-Bus scenario, in addition to 22.1 $\mu g/m^3$ concerning the All-EV scenario. Although these reductions in NO_2 concentrations are similar, the E-Bus scenario performs better than E-Car at this GP site in reducing NO_2 concentrations, given that car flows are responsible for NO_x emissions of 36% (762 tonnes) in the Baseline scenario and 49% (327 tonnes) of NO_x is released from car exhaust.

The health gains for all 2030 scenarios were estimated. The difference in health outcomes in reducing hospitalisation between the All-EV scenario and BAU scenario was 80 cases, which indicates a significant change due to the massive intervention in All-EV scenario, given that the suspicious performance of the Euro 6/VI standards vehicle which are the dominant standards in BAU scenario, in commitment with legally binding emissions thresholds.

Given that overall, buses emit much lower levels of emissions compared to other vehicles across the network, the 'E-Car_E-Bus' or 'E-Car_E-LGV' scenarios would achieve similar outcomes to the 'All-EV' scenario. However, bus services bring passengers to city centres therefore radials and areas in the vicinity of bus and train stations, buses become high proportion of the fleet. Depending on the orientation of the road with respect to the prevailing wind and whether or not the street is a canyon the emissions become trapped creating an AQMA. This was situation at the GP_35 'Metro Interchange Surgery' (section 7.4.3). Regarding the economic implications, the adoption of the E-Car_E-Bus scenario would improve the quality of air for a lower cost compared to the All-EV scenario for those locations where buses are the primary emission source.

It was estimated that more health gains would be attributed to reduction in the long-term rather than short-term exposure to pollutants. The outcomes depend on the dose-response coefficients which are associated with either long- or short-term exposure to air pollution.

CHAPTER 8

8. Summary, Discussion, Conclusions and Future Work

A summary of the present research provided in this chapter. A number of conclusions are drawn from the findings and are presented along with suggestions for future work to extend and enhance this area of research. The contribution made to academic research knowledge is also explained.

8.1 Summary

The aim of this research was to investigate the impact of the increasing the uptake of electric vehicles on air quality and health given that the coming years will witness widespread adoption of vehicles equipped with alternative drive technologies. This would result in reductions in tailpipe emissions and improvements in air quality as well as subsequent reductions in adverse effects on health. To date, the UK government has made commitments to invest more than £3.5 billion in providing better air quality and cleaner transport (DEFRA, 2019, p. 83). For example, the DEFRA and DfT (2017b, p. 4) have reported that, in 2011, the UK was the first country to announce its intention to limit car sales in 2040 to ultra-low emission cars only, paving the way for ultra-low emission cars to form the entire car stock in the UK by 2050. Most recently, the DfT (2018a) published 'The Road to Zero' strategy and DEFRA (2019) published 'Clean Air Strategy 2019' which confirm this intention, and it is proposed that the percentages of ultra-low emission vehicles should reach 70% and 40% of new car and van sales respectively in 2030 and intends replacing all government fleets will be replaced with ULEVs by 2030. The latter action will start by limiting the purchase of new cars to ULEVs aiming for them to represent a quarter of the government's vehicle stock in 2022. Undoubtedly, these actions will have a positive influence on the quality of air and subsequently on health, as 40,000 premature deaths can be attributed to air pollution in the UK annually, meaning that there is a cost to society of approximately £20 billion per annum due to the health-related consequences of people suffering the relevant diseases and early deaths (Royal College of Physicians, 2016). DEFRA recently estimated that, without the policies presented in the Clean Air Strategy 2019 to tackle air pollution, the annual cost of the impact on the population's health of air pollution could reach £1.7 billion and £5.3 billion by 2020 and 2030 respectively (DEFRA, 2019a, p. 98).

The impact of cleaner road transport on air quality and health in terms of reducing premature deaths and hospitalisations resulting from the widespread adoption of new vehicle technology have not been properly investigated in the literature. For example, the potential impact of increasing the penetration of electrics vehicle on air quality was investigated in Taiwan by Li *et al.* (2016a), but did not evaluate the health impact used an air quality model which assumed a replacement of the 2010 light-duty vehicle fleet under a number of electricity generation scenarios. They found that emissions of CO, VOCs, NO_x and $PM_{2.5}$ were expected to decline and that great benefits could be achieved from investing in renewable energy sources.

Similarly, Soret *et al.* (2014) analysed three electrification scenarios for 13%, 26% and 40% penetration of vehicles in Barcelona and Madrid. From their modelling, they found possibilities for emissions mitigation, particularly in relation to NO_x and CO. In Dublin, Ireland, the health effects of the increasing penetration of electric vehicles were quantified in terms of the sum of potential life years lost due to early mortality and years of productive life lost caused by disability in terms of disability adjusted life years (DALYs) (Dey *et al.*, 2018a). The authors highlighted that DALYs would be reduced by 300 by 2030 compared to 2015 as a result of the prohibition of diesel cars by 2025 and the gradual annual increase in electric car sales from 2010 up to 50% of the new car fleet. The methodology applied in Dey *et al.* (2018a) was based on replacing 50% of Vehicle Kilometres Travelled (VKT) by conventional vehicles with electric vehicles. This was followed by applying an external cost factor of 700 DALYs per kilo tonne of $PM_{2.5}$ and 90 DALYs per kilo tonne of NO_2 of the expected pollution reductions based on the VKT by electric vehicles. The amount of vehicular emissions and their dispersion were not simulated, unlike in this thesis where spatial analysis of air quality was performed using a dispersion model. Thus, the potential impact of reductions in vehicle emissions, improvements in air quality and reductions in premature deaths and hospitalisations could be evaluated the results of which are presented in this thesis. Modelling each step traffic, emissions, dispersion as well as health impacts is the novelty in the work presented in this thesis.

To test the impact of the increasing uptake of electric vehicles, Newcastle and Gateshead were selected as a case study because of the availability of network traffic and traffic data for both locations. For each road link, the hourly flows for cars, LGVs, HGVs and buses in 2010 was estimated in both boroughs (Goodman *et al.*, 2014).

The Baseline traffic model for 2014 was created according to DEFRA guidelines. The outputs from the 2010 transport model were updated to meet the Baseline profile which was validated against criteria developed by the Design Manual for Roads and Bridges. Although, the results of the correlation did not quite match the targets of the criteria, they are close to them.

The emissions released in the Baseline traffic model were calculated using PITHEM. The traffic network and data were entered into PITHEM to create a set of files consisting of traffic data on parameters such as the road network, road geometry, street canyons and hourly emissions rates associated with each road link. After processing, the output files can be read in terms of the air quality model using ADMS-Urban to estimate emissions concentrations. Although PITHEM is capable of calculating the emissions rates resulting from various vehicle classes, PITHEM incorporates the fifth Emissions Factors Toolkit (EFT v5), which lacks the ability to calculate emission rates released by vehicles equipped with alternative technology, and moreover speed-related emissions were not well represented in EFT v5. Therefore, the output files for PITHEM that consist of emission rates were amended using EFT v7 for more appropriate representations of the emissions rates released by the traffic flow in the Baseline scenario. This is because the most recent version EFT v8 published by the Department for Transport does not support the Baseline year of 2014.

To model the dispersion of vehicular emissions due to meteorological factors, ADMS-Urban was utilised. Hourly meteorological records, particularly for wind direction and speed, for 2014 were required. They have been obtained from the Met Office. In addition, background concentrations in ADMS-Urban were set to zero, because those concentrations vary in magnitude across the study area. The outputs of ADMS-Urban represent concentrations of pollutants originating from road transport only at selected receptors. Thus, contaminant concentrations from other sources should be taken into account to finalise the pollution levels and provide the bigger picture. According to the Environmental Act 1995, DEFRA is committed to publishing maps of air pollution background concentrations in order to support local authorities in carrying out the required reviews and assessments. Those concentrations are published on grids of 1 km by 1 km spacing and categorised by their sources, such as industry, minor roads and major roads. The contribution from major roads was removed and the remaining concentrations were topped up to their spatially corresponding pollutant concentrations as modelled by ADMS-Urban in order to finalise the pollution concentrations. Comparison of the final modelled and monitored concentrations indicated that the model is valid and in compliance with TG16 thresholds, indicating an acceptable goodness of fit.

The traffic scenario for the 2030 BAU scenario was developed based on the Baseline traffic model. This was accomplished by following the same DEFRA guidelines that were used to update the 2010 situation to the 2014 Baseline, whilst the vehicle class proportions were adopted from the NAEI (2017) projections. The CCC scenario was developed according to the report by Element Energy, Ecolane Consultancy and the University of Aberdeen who were commissioned by the CCC to propose a vehicle fleet mix by 2030 aiming to reduce carbon dioxide emissions in 2050 by 80% compared to 1990 levels (Element Energy, 2013). The electrification of buses (E-Bus) scenario was developed since significant proportions of bus journeys in the study area are managed by two bus operators. The scenario of the electrification of all vehicle fleets (All-EV) is built on the DEFRA plan to implement Clean Air Zones in Birmingham, Leeds, Southampton, Nottingham, Derby and London which intended to prevent relatively old vehicles from using roads in the city urban cores so as to improve air quality by 2020 (DEFRA, 2015, p. 15), but here extending all zones to cover the entire area of Newcastle and Gateshead in this scenario. Scenarios of all cars being electric (E-Car), all cars and all buses electric (E-Car_E-Bus), and all cars and LGVs electric (E-Car_E-LGV) were derived from the abovementioned scenarios and investigated in this thesis.

The PITHEM was run for each of the 2030 scenarios to calculate emission rates. The ADMS was run at 66 GP sites across Newcastle and Gateshead as receptors to simulate pollution concentrations in modelling each of all the 2030 scenarios. It is worth mentioning that the 2014 Baseline situation was assumed to be the same for 2030 scenarios except in relation to expected traffic growth. For instance, meteorological records for the 2014 Baseline were assumed to be the same for 2030 due to the difficulty of predicting future meteorological trends, especially on an hourly basis. Additionally, expansion in housing projects and the potential creation of street canyons were not considered in the work as it mainly aimed to investigate the impact of replacing conventional vehicles with electric vehicles on the environment and heath.

Quantities of emissions released by vehicles were investigated. In Newcastle and Gateshead, total emissions released by traffic flows were calculated to total 2,100 tonnes of NO_x, 152.5 tonnes of PM_{10} and 95.3 tonnes of $PM_{2.5}$ in the 2014 Baseline. These emissions are expected to be reduced in the 2030 BAU by 70% in NO_x, 5% in PM_{10} and 17.7% in $PM_{2.5}$ levels. It was found that car flows are the principal source in relation to these emissions, whilst bus flow causes the smallest contributions of PM_{10}, and $PM_{2.5}$.

The health impact of changes in the vehicle mix for each scenario was evaluated in this research. Respiratory diseases categorised from J00 to J99 on the ICD-10 scheme where codes from J00 to J06 indicate acute upper respiratory infections, were selected to determine the health status in the study area in terms of numbers of hospitalisations and premature deaths resulting from these diseases as a strong association between them and exposure to air pollution. The data on the number of in-hospital deaths and hospital admissions attributed to these diseases were purchased from NHS Digital for patients living in the pilot area. It is important to identify sites where those patients are exposed most to air pollution, such as their homes, in order to link the severity of pollution with areas where exposure to air pollution might occur. However, obtaining health data for details of residences was not possible because of legal issues in relation to the protection of patient privacy. In addition, Section 251, which permits access to sensitive patient details without their consent requires a medical purpose to be obviously declared, which was not possible in this research. For this reason, residents in Newcastle and Gateshead were assumed to be exposed to a level of pollution similar to at modelled pollution at the GP practice they were registered at. This is justifiable, given that inhabitants tend to register with the GP practice closest to their homes (Santos *et al.*, 2017; Beghelli, 2018).

Prior to 2015 the majority of the population tended to register with the closest GP practice for health care and GP practices only offered medical services to residents who lived within the practice boundaries in England. However, residents are allowed to receive health treatment by registering with any GP practice regardless of the practice boundary. Notwithstanding, GP practitioners can reject or cancel any registration for those who live outside the practice boundary if the practice capacity is limited or refuse to provide home visits due to the longer distance to the patient's home. Fortunately, the health data used in this thesis was based on 2014 statistics, i.e. prior to this permission being granted.

In addition, the elderly are the cohort most vulnerable to illness because of exposure to air pollution as reported by Spix *et al.* (1998). It can be argued that elderly people have a greater tendency to register with the closest GP practices for their convenience, as they are often retired. Similarly, it can be argued that parents prefer to register their children with the closest practice for same reasons.

Furthermore, measures that affect an individual's choice of GP practice in the UK have been evaluated by Santos *et al.* (2017), who examined the determinants of the choice of 3.4 million adults aged 25 and over from amongst nearly 1,000 family doctor practices. The researchers collected data which comprise a rich set of information concerning practice quality of care, distance from patients' residences to potential practices, in addition to other characteristics such as the age and gender of doctors in the practice and the country from which the doctor gained their qualification. In regard to distance from patient residence to potential practice, Santos *et al.* (2017) calculated straight-line distance between the centroid of 2,875 lower super-output areas (LSOAs) and all GP practices in the East Midlands Strategic Health

Authority. The mean distance of the LSOA centroid (in which patients resided) to the chosen practice was 1.9 km in the East Midlands (Santos *et al.*, 2017). This indicates that a circle with a diameter of 3.8 km surrounding a GP surgery site is a valid assumption and capture people's exposure to air pollution. Moreover, this finding is consistent with a PhD study at King's College, London, was based on the assumption of GP practices offering health services to people living nearby (Beghelli, 2018).

As mentioned previously a circle with a radius of 1.9 km around a GP practice site could well represent people's exposure to air pollution. For example, spatial variations in PM_{10} levels were studied within the San Joaquin Valley in California. PM_{10} concentrations varied by one-fifth over distances from 4 to 14 km from the core sites (Blanchard *et al.*, 1999). Burton *et al.* (1996) measured PM_{10} and $PM_{2.5}$ levels at eight sites ranging from 0.6 to 28.8 km from the centre of Philadelphia in intra-urban areas. Generally, significant correlations were found for $PM_{2.5}$ and PM_{10} concentrations measured at the eight sites. Pearson correlation coefficients between the sites were stronger for $PM_{2.5}$, ranging from 0.70 to 0.96, and slightly lower for PM_{10} in the range of 0.62 to 0.96. The study implied that concentrations at a central monitoring site can be used to characterise exposure concentrations over the city. Bari *et al.* (2003) correlated $PM_{2.5}$ hourly and longer-term averages at two monitoring sites 11 km apart in Manhattan and the Bronx, New York. Correlations between the daily average concentrations of $PM_{2.5}$ at the sites were high (R^2=0.92 with a slope=0.95). Annual absolute concentration levels were 15.2 $\mu g/m^3$ at the Bronx and 15.5 $\mu g/m^3$ at Manhattan. At six mobile sites and one fixed site within 3.3 km of each other in Basel, Switzerland Röösli *et al.* (2000) discovered a relatively homogeneous annual mean of PM_{10} mass concentration ranging from 27.6 to 32.0 $\mu g/m^3$. This notable spatial homogeneity in long-term mean PM_{10} levels clearly reduces the error of assigning data from one fixed monitoring site to all study subjects living in Basel. In China, Ye *et al.* (2003) conducted an analysis over one year of weekly $PM_{2.5}$ concentrations at two sites in Shanghai that were 4 km apart. A linear regression between the $PM_{2.5}$ mass concentrations at the two sites had a high correlation (R^2=0.94 and a slope=0.95). Annual mean concentrations at the sites were 57.9 and 61.4 $\mu g/m^3$ respectively, suggesting regionally sources were fairly homogeneous. He *et al.* (2001) monitored variations in average long-term concentrations of $PM_{2.5}$ at two sites located 10 km apart in Beijing. Annual mean concentrations of $PM_{2.5}$ at the sites were 115 and 127 $\mu g/m^3$, respectively.

The assumption that GP sites can be considered as receptors to estimate exposure to air pollution may possibly contain limitations, given that not every individual registers with the closest GP practice. Moreover, it should be mentioned that levels of air pollution are not expected to have significant variations over this distance, unless there is a heavy flow of vehicles and traffic at street canyons either at the GP practice or at the place of residence. This is particularly relevant to NO_2 concentrations. For example, Roorda-Knape *et al.* (1998) measured pollution levels near motorways in Holland. It was found that black smoke and NO_2 concentrations decreased with distance from the roadside, although no concentration gradients were observed for PM_{10} and $PM_{2.5}$. A study of 10 urban environments in the Emilia-Romagna region of Italy established Pearson correlation coefficients between sites for PM_{10} with a mean of 0.89 and for NO_2 with a mean of 0.77. The results highlighted that a single fixed-site monitoring station could not accurately characterise the spatial nature of air pollution at deep street canyons, high traffic densities and very low ventilation. However, their study unearthed evidence that the intra-urban spatial variability of particulate levels was low (Sajani *et al.*, 2004). Whilst having limitations, on balance, the assumption that the locations of GP surgery at which the patients are registered can be considered to be the receptor for the estimation of pollution level to assess public exposure.

Pollution concentrations were predicted to have decreased by 2030. The model shows that pollution concentrations would have declined compared to the 2014 Baseline levels. The average hourly annual reduction at the 66 GP sites would be by 8 $\mu g/m^3$ for NO_2 levels and 3 $\mu g/m^3$ each for PM_{10} and $PM_{2.5}$ levels. The largest reductions by 19.1 $\mu g/m^3$ (56.7%) in the NO_2 level and 3.4 $\mu g/m^3$ (24.4%) in the PM_{10} level would take place at GP_35 in the BAU scenario. From the electrification of 40% of the vehicle fleet in 2011, the hourly maximum value of NO_2 is expected to be reduced by 16% to 30 $\mu g/m^3$ and 35 $\mu g/m^3$ in Barcelona and Madrid respectively (Soret *et al.*, 2014). This is because traffic flows in Barcelona and Madrid are much heavier than in Newcastle and Gateshead given that Madrid is the capital city of Spain with a population of 6 million and Barcelona has a population of 5 million, whereas Newcastle and Gateshead have a combined population of not more than 400,000. Also, there are substantially more motorcycles and scooters in the Spanish fleets.

Dose-response coefficients which describe the association between exposure to a certain increment of pollutant level and the probability of being admitted to hospital or early death were applied to these expected reductions in pollutant concentrations at each GP site to quantify the health gain in terms of reductions in numbers of early deaths or hospital admissions due to respiratory disease. The health gain was quantified by the following equation:

$$\textbf{Health gain} = \textbf{Number of mortality (or hospital admission)} \times \frac{1}{10} \textbf{ Concentration reduction} \times \textbf{Dose-response coefficient}$$

In the BAU, the application of the dose-response coefficients associated with long-term exposure to reductions in the annual means of pollutant concentrations indicates that reductions of 1,286, 217 and 874 hospital admissions would be achieved in 2030 as a result of reducing NO_2, PM_{10} and $PM_{2.5}$ levels respectively. It was noticed that the largest reduction in hospital admissions would occur due to alleviating NO_2 concentrations. However, at several GP sites, lowering PM_{10} would achieve more health gains than lowering the NO_2 level. Thus, at those GP sites, health gains resulting from lowering the PM_{10} level were considered the main health gains at those particular sites. Hence, the combined health gain would be a reduction of 1,297 cases of hospitalisation. In the All-EV scenario, hospital admissions are expected to be reduced by 1,377; which is greater than the BAU health gains by 80 cases. The E-Car_E-Bus and E-Car_E-LGV scenarios lead to very similar outcomes regarding the avoidance of hospital admissions. With regard to reduced premature mortality in 2030 compared to 2014, 14 to 16 early deaths might be avoided if pollution could be lowered. This reduction in early mortality represents 0.003% of the population in Newcastle and Gateshead in 2014, if this percentage is scaled up to national population of the UK, the numbers of early mortalities could be reduced by 2,000.

Conversely, only modest health gains would be associated with reduction in short-term exposure to pollutants, which were quantified by applying dose-response coefficients to reductions in the daily means of NO_2, PM_{10} and $PM_{2.5}$ levels at each of the GP sites. For each GP site, the health gains vary in magnitude for 365 days. Hence, the minimum health gain associated with each of the reductions in the three pollutants levels was selected to represent the health gain for the whole year at each GP site. Those minimum health gains from the all GP sites were summed to represent the health gains for the study area. It was observed that a reduction in $PM_{2.5}$ levels would have a significant impact on avoiding a greater number of hospital admissions up to 50 occurrences in all of 2030 scenarios. Additionally, deaths would be reduced by 3 cases, primarily by the lowering of NO_2 levels.

8.2 Discussion

In this section, the results of the analysis from the different scenarios are compared to each other to address whether or not there are any statistical differences between the amount of emissions from traffic scenarios. Likewise, the findings of previous studies are highlighted in this section.

The emissions of NO_x solely released by cars varied between 762 tonnes in the Baseline scenario to 327 tonnes in the BAU 2030 scenario, 251 tonnes in the CCC scenario and zero emissions when cars were assumed to have become 100% electric. Emissions of NO_x from LGVs were estimated to be 369, 215, 139 tonnes in the Baseline, BAU and CCC scenarios respectively. The results of an analysis of variance (ANOVA) suggest that the NO_x emissions from all scenarios are significantly different with P-value of 3×10^{-5}. However, if the Baseline scenario is excluded, the difference between emissions levels resulting from the other scenarios was no longer significant with P-value of 0.2. This might be due to the fact that Euro 6 diesel light duty vehicles will be the dominant car fleet by 2030, breaching the standards for NO_x emissions for their engine class due to the use of defeat devices. Those devices switch on full control of the emission system during the emissions test and release higher emission levels by switching off the emission control system during real-world driving conditions (Dey *et al.*, 2018a). Quantities of emissions of PM_{10} and $PM_{2.5}$ did not differ in the scenarios because most of these emissions originate from non-exhaust sources regardless of the vehicle engine type or technology.

Car emissions in Newcastle and Gateshead resulting from the 2014 Baseline and the 2030 scenarios which were investigated in Section 7.2 indicate reductions of 1.6% in PM_{10} and 13.2% in $PM_{2.5}$ emissions and no NO_x emissions released by cars by 2030 compared to the 2014 levels, when all cars are assumed to be electric in the 2030 scenarios. Nevertheless, in the BAU scenario, these emissions were estimated to have increased by 2% in PM_{10} and decreased by 8.7% in $PM_{2.5}$ emissions and by 57% in NO_x emissions. Moreover, electrifying all buses by 2030 would reduce emissions by 42% in PM_{10} emissions and 58% in $PM_{2.5}$ compared to 2014 levels for bus emissions; whilst reductions by 39.3% in PM_{10}, 53.9% in $PM_{2.5}$ emissions and 89.3% in NO_x emissions were estimated in the BAU scenario.

With respect to the performance of all 2030 scenarios in reducing pollutant concentrations compared to the Baseline scenario, the reductions for both PM_{10} and $PM_{2.5}$ concentrations appear to be the same among all scenarios, as previously illustrated in Figures 7-7 and 7-8. Similar performances of 2030 scenarios in reducing PM_{10} and $PM_{2.5}$ concentrations can be attributed to the significant contribution of non-exhaust emissions in releasing these two pollutants. This creates similar responses in all scenarios of reducing numbers of early mortalities and hospital admissions to the changes in PM_{10} and $PM_{2.5}$ concentrations.

Conversely, reductions of NO_2 concentrations among the 2030 scenarios vary, as demonstrated in Figure 7-9. For example, bus electrification would lead to a noticeable reduction in NO_2 concentrations at GP_35 site, while the performance of E-Bus and E-Car_E-Bus scenarios in reducing NO_2 concentrations are similar to that of the All-EV performance scenario. Hence, this reflects the variation in the reductions of the numbers of early deaths and hospitalisations, as Figure 8-1 illustrates.

Although bus flows produce relatively lower quantities of pollutant emissions, bus emissions caused the concentrations to intensify in urban areas due to their ability to access the urban core.

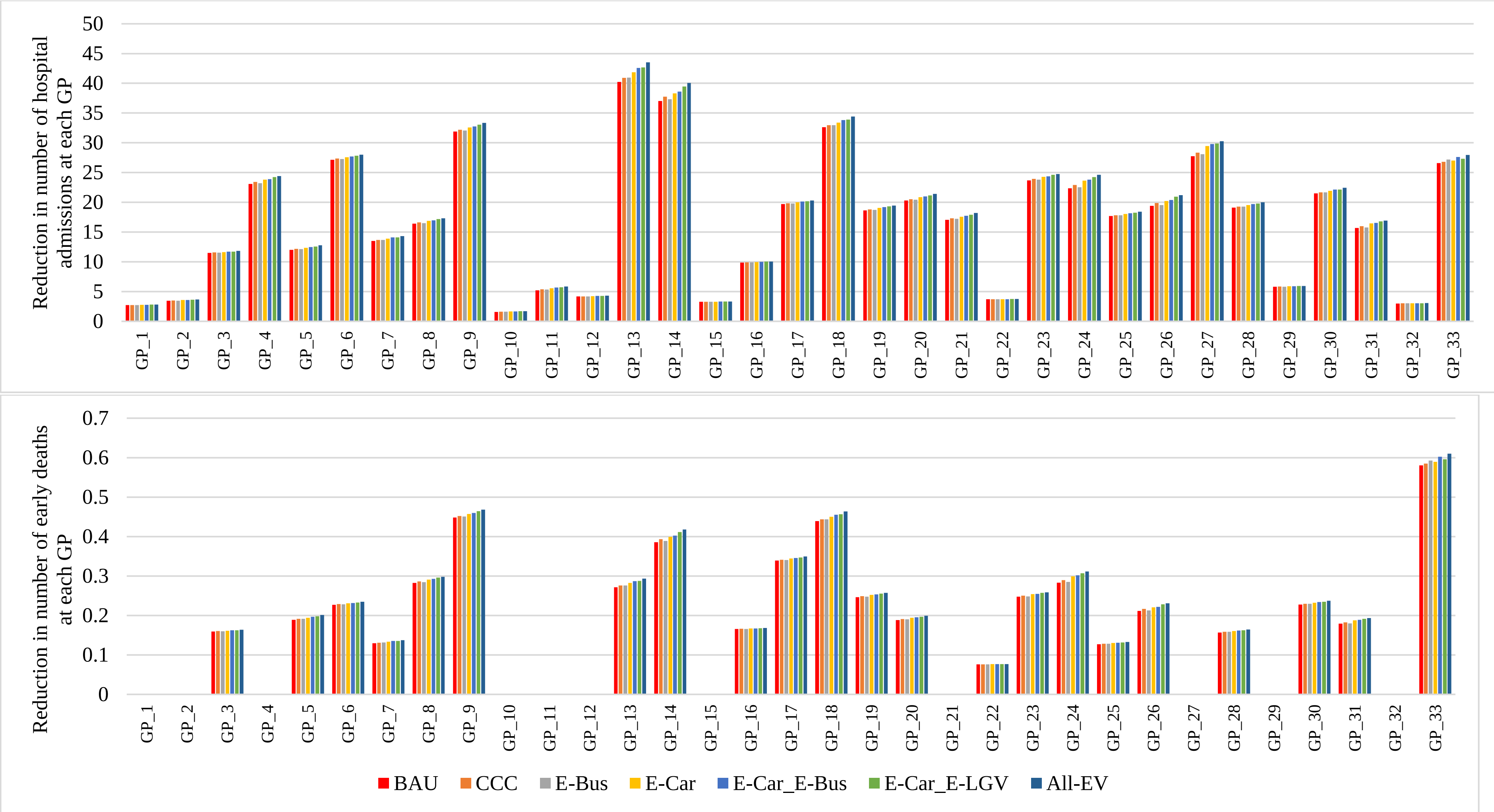


Figure 8-1: Reductions in hospital admission numbers (top) and early deaths (bottom) in response to reductions in NO_2 concentrations

In other studies, Dey *et al.* (2018a) estimated significant vehicular emission reductions in Dublin, Ireland, by 2030 compared to 2015 levels. They concluded that reductions in car emissions of 23% in NO_x, 31% in PM_{10} and 44% in $PM_{2.5}$ levels were possible due to a gradual increase in the annual sales of electric vehicles from 2010 to 2030. For example, their study assumed that electric vehicles would account for 25% of the total in 2025 increasing annually to 50% by 2030 for the sales of new cars in Dublin, given that the banning of diesel cars by 2025 in Ireland was taken into consideration in their study. In addition, new buses will be driven by bio-CNG (compressed natural gas) from 2025 leading to bus emissions being reduced by 54%, 58% and 72% in NO_x, PM_{10} and $PM_{2.5}$ emissions by 2030 compared to 2015 levels. The health effects due to these reductions in vehicular emissions of NO_x, PM_{10} and $PM_{2.5}$ by 2030 would be to potentially reduce DALYs by 300 and to save €43.8 million in the associated health and care cost in Dublin.

Electrifying all of the 2010 light duty vehicle fleet in Taiwan was expected to reduce NO_x and $PM_{2.5}$ emissions by 27% (Li *et al.*, 2016a). Furthermore, annual mean concentrations of NO_x and $PM_{2.5}$ at 46 sites in urban areas were expected to be reduced by 18% (4.5 ppb) and 6% (2.5 $\mu g/m^3$) respectively. Additionally, these reductions could be 21% (5.3 ppb) and 8% (3.1 $\mu g/m^3$), if the electricity used to power the vehicles was generated from clean sources. In this PhD thesis, the reductions from electrification all cars and LGVs the (E-Car_E-LGV) scenario by 2030 would reduce the annual mean concentrations of $PM_{2.5}$ by 2.8 $\mu g/m^3$.

Three scenarios for replacing of 13%, 26% and 40% of conventional vehicles with electric ones in relation to the 2011 fleet in Spain were investigated for Barcelona and Madrid (Soret *et al.*, 2014). Road transport emissions of PM_{10} and $PM_{2.5}$ were expected to be reduced by 1% and 2% respectively in both cities; while NO_x emissions were expected to be reduced by 4% in Barcelona and 6% in Madrid in the lower vehicle electrification scenario. Moreover, the 40% vehicle electrification scenario indicated reductions of 11% in NO_x emissions, 4% in PM_{10} and 5% in $PM_{2.5}$ in Barcelona and reductions of 17% in NO_x emissions, 3% in PM_{10} and 5% in $PM_{2.5}$ in Madrid.

Furthermore, Ferrero *et al.* (2016) calculated the influence of replacing conventional vehicles with electric vehicles from the 2011 fleet on air quality in Milan, Italy. They showed that a scenario replacing 50% of light vehicles with electric ones could reduce NO_2 concentrations by 5.5%. In Denver, USA, replacing all petrol cars with PHEVs was estimated to be able to reduce daily NO_x emissions by 27 tons but an increase of daily NO_x emissions of 3 tons at electricity sites in Denver would also occur (Brinkman *et al.*, 2010).

8.3 Conclusions

The main conclusions from the research undertaken in the present study are as follows:

1) Scenarios relating to the widespread adoption of electric vehicles by 2030 would impact positively on the environment, alleviating exposure to pollution originating from road transport, and it would have a significant impact on people's health in Newcastle and Gateshead.

2) The modelling of air quality for the 2030 scenarios indicated that NO_2 concentrations are expected to be significantly reduced by 2030 compared to 2014 levels. For example, at GP_44, NO_2 concentrations would be reduced from 67% to 59%. These percentages could be higher if background emissions were excluded from the comparison, although these emissions are expected to be reduced in the future. However, the reductions in the levels of PM_{10} and $PM_{2.5}$ would be similar among all 2030 scenarios. For example, at maximum reductions of 29.6% and 37.6 in PM_{10} and $PM_{2.5}$ respectively would take place at GP_15.

3) The evaluation of the health impact indicated that the number of hospital admissions would be reduced by 80 and 56 in the All-EV and E-Car_E-LGV scenarios respectively compared to BAU 2030. Although implementing the All-EV scenario would greatly increase the health gains, electrifying both cars and LGV would achieve nearly the same health outcomes at lower cost, given that LGV flows are expected to witness significantly greater growth than other vehicle types at 15% between 2014 and 2030.

4) The quantification of health gains resulting from the mitigation of levels of pollution in the 2030 scenarios depend primarily on the dose-response coefficients used, which describe the association between exposure to incremental in air pollution and the impact on health. This is because the health gain attributed to mitigating long-term exposure to air pollution is greater than that attributed to alleviating short-term exposure.

5) The investigation of the long-term impact of exposure to air pollution indicates that the reduction in NO_2 levels would achieve more health gains than reductions in other pollutant concentrations. Conversely, the investigation of the short-term impact of exposure to air pollution suggests that reductions in levels of $PM_{2.5}$ would typically achieve more health gains than other pollutants in terms of levels of resulting

hospitalisations, whilst lowering NO_2 levels would generally achieve more health gains than other pollutants in terms of numbers of premature deaths.

6) Each of the 2030 scenarios are feasible although they differ in their implementation timescales. The exception is in relation to the CCC scenario. The E-Car scenario is highly feasible, but the timescale could be extended by 14 years (as mean vehicle lifespan in the UK is 14 years (DfT, 2018a, p. 13)) after the ban on the sale of new conventional cars has taken place. It is important for the Government to raise public awareness of the benefits of transition to electric cars in order to accelerate their penetration. The E-Bus scenario has the most feasible among the other scenarios. The propulsion of most buses is based on diesel engines which have elevated primary NO_2 due to the need for particle traps. Buses are run by a few different operators in Newcastle and Gateshead so through a quality bus partnership set up with the local authorities of Newcastle and Gateshead working together has made it more straightforward to apply for government grants to support operators financially to introduce ultra-low emission vehicles. These operators have suitable depots in which to store their fleet of buses, so that recharging of electric vehicles can take place. In addition, the buses do not operate overnight which gives the chance for low tariff electricity to be used. Moreover, solar panels could be installed on the roofs of depots providing the opportunity for sustainable clean energy to be generated. Also, it is believed that the timescale for implementing the E-Bus scenario could be short if the local governments are in favour of this particular scenario.

8.4 Limitations

This research has endeavoured to achieve a greater understanding of the impact of the increasing uptake of electric vehicles in lowering vehicular emissions, improving air quality and reducing numbers of hospital admissions and premature deaths. Several shortcomings were observed whilst conducting this research. For example, the first limitation in this study was in identifying where patients are exposed to air pollution. The most appropriate place to evaluate changes in patient's exposure to air pollution is where the patient lives and works. Nevertheless, details of patients’ residential addresses are considered to be private information and are difficult to obtain. Therefore, the addresses of the GP sites where patients are registered were assumed to represent patient exposure to air pollution and thus changes in pollution concentrations at GP sites were evaluated.

The second shortcoming is the difficulty related to predicting meteorological data for 2030 on an hourly basis. Therefore, it was assumed that the meteorological data for 2014 also applies in 2030, given that the aim of the study is simply to examine the impact of increasing the penetration of electric vehicles.

Traffic growth factors for 2030 were estimated based on traffic information for the base year. These growth factors have been applied uniformly to on the traffic network of the study area, although increased growth in housing construction in one area could make traffic more intense than these in other areas.

8.5 Suggestions for Future Research

The methodology of this research could be improved for future application as follows:

1. Investigations of the impact of the increasing penetration of electric vehicles on health can be carried out using a high-resolution analysis by geographically associative patient residences with pollution levels and using time series analysis to link episodes of hospitalisation with pollution levels. Additionally, the availability of age and gender records for patients would assist in categorising the patients most vulnerable to increases in exposure to air pollution. This data could be acquired from the NHS Digital Centre (previously known as HSCIC, the Health and Social Care Information Centre). This is where the NHS retains data concerning all hospital episodes. However, such detailed information for patients who had died or were admitted to hospital as a result of respiratory diseases are classified as identifiable and sensitive data by NHS Digital and their dissemination is only permitted under a legal basis such as Section 251 (for the control of patient information). To apply for this Section, an application must be approved by the Confidentiality Advisory Group. Unfortunately, a medical purpose, which was not available for this research, should be clearly stipulated in the application regardless of the strengths or importance of the research. Therefore, any forthcoming research should include a medical purpose in order to apply for legal justification to access sensitive patient data.

2. There is a lack of dose-response coefficients which describe the relationship between the probability of increases in hospitalisation and long-term exposure to increases in air pollutant concentrations. Future research could focus on the probability of hospitalisation due to long-term exposure to increases in air pollutant concentrations.

3. Non-exhaust emissions vary according to driving style, such as the use of harsh acceleration and deceleration which are generally linked with higher non-exhaust emissions. However, an aggressive driving style in electric vehicles will more rapidly deplete the battery charge, and so drivers may prefer to drive smoothly to save as much battery charge as possible and this is likely to reduce the amount of non-exhaust emissions released. Besides this, developments in battery technology will decrease the weight of electric vehicles compared to their conventional counterparts. Both driving style and vehicle weight have a direct influence in reducing the quantity of non-exhaust emissions.

4. In this research, fleet mix projections were defined based on national data published by NAEI (2017). This might not accurately represent vehicle fleet composition in Newcastle and Gateshead, although it can make the research output more generalisable on a national scale. Fleet composition projections on a local scale might better represent the reality in future research. Local authorities may generate this data, if resources are available to them. In addition, the research methodology can be applied to other cities, as it relies on predictions of the growth and mix of vehicles which are available from the NAEI website.

5. In this research, meteorological data for 2014 were used to model air quality in 2030. The projection of meteorological records for 2030 could lead to more accurate estimates of pollutant concentrations and, subsequently, better estimates of the disease burden since cold years typically escalate NO_x concentrations (Goodman *et al.*, 2014, p. 30). However, the projection of future meteorological data is not often available in the sufficiently fine resolution required for modelling in ADMS-Urban.

8.6 Contribution to Academic Research and Practice

This PhD thesis is the first piece of work that has investigated the impact of increasing proportions of electric vehicles within vehicle fleets, on air quality and subsequently the impact on reductions in hospitalisations and premature deaths. This thesis has used the traffic network of Newcastle and Gateshead as the study area to utilise real hour-by-hour meteorological records, as well as traffic flow data for cars, LGVs, HGVs and buses. Rates of vehicular emissions were calculated and their dispersion modelled. Real health data in relation to the respiratory burden for patients was linked to where their exposure to pollution might occur.

Previous research has depended predominantly on the assumption that a proportion of the vehicle distances travelled (VKT) by conventional vehicles would be travelled by electric vehicles instead. In this research, dose-response coefficients were spatially applied to the lowering of pollution levels in order to estimate health gains.

The work undertaken in this thesis was produced according to guidance proposed by UK government organisations, and the work introduces a framework of modelling processes to include updated traffic flows, the calculation of emissions rates, estimation of emissions concentrations and the quantification of expected reductions in hospital admissions and early mortality due to changes in pollutant concentrations. For example, the Baseline traffic model was updated and validated according to Department for Transport guidance. Similarly, the air quality modelling was validated by following DEFRA guidance. The proportions of vehicle types in the fleet were taken from NAEI to represent the national vehicle fleet mix, which means that this model can be applied in other study areas, both now and in the future in the UK to investigate the impact on air quality and health due to changes in the vehicle fleet mix, as long as local traffic parameters are available.

There are several policy implications which arise from this work. Firstly, this thesis provides robust evidence that reducing and if possible eliminating levels of vehicular emissions the resulting exposure can potentially considerably reduce the number of hospital admission cases occurring in Newcastle and Gateshead. The same could be said about other cities if levels of pollution were reduced, for example by means of increasing the proportion of electric vehicles.

Secondly, this work has demonstrated that notwithstanding the discrepancy between real-world driving cycles and proposed type approval testing, including road gradient, truck loading and thermal windows' effects not accounted for in those tests, the reliability of Euro 6 diesel vehicles complying with stricter standards compared to Euro 5, is the limiting factor in the amount of released emissions in particular NO_x emissions. Furthermore, pertaining to the merit of current testing and modelling systems, to reduce the discrepancy between real-world driving cycles and proposed type approval testing, the measurement of real-world vehicle emissions should be performed on actual roads to capture real-world driving conditions such as temperature, road gradients, air resistance, driver behaviour effect etc. These measurements should be carried out on a European scale, in other towns and cities to achieve European emission standards. Drivers behaviour is quite different across countries.

Thirdly, the framework of models used in the present thesis was established to model the updates in traffic flows for a certain year and locality, with the aim of calculating vehicular emissions, simulating their dispersion in an air quality model in great detail at the local authority level, considering background emissions and meteorological data effects consistent with local authority practice. Furthermore, the framework can be used to quantify the health burden in terms of reductions in mortality and hospital admissions attributed to respiratory diseases using GP sites as receptors over the study area. Thus, the implications of intervention in the vehicle fleet mix by adopting more EVs are quite clearly aligns with local authority requirements to make evidence based decisions. Other cities in the UK can utilise this framework and quantify the impact on mortality and hospitalisations due to any proposed interventions in fleet proportion as long as the relevant meteorological conditions, traffic flow and road network data are known. The outputs of the framework can inform policymakers of the subsequent impacts of current or future policies in relation to not only quantity of emissions but also concentrations and impact on health. In addition, this framework can be used to identify hotspots regarding the health burden and high pollutant concentrations over a study area.

On a European scale, where the factors relating to vehicular emissions are similar to those in the UK, given they were developed based on the European COPERT, the emissions estimates, air quality model and dose-response relationships can be implemented to estimate the air quality and health benefits of interventions such as market penetration of clean vehicles. However, factors for traffic growth in a particular may possibly be different in a European compared to a UK city.

Overall, the lesson learnt of global interest is that there is merit increasing the penetration of electric vehicles given their role in mitigating local air pollution resulting in improved air quality and better health outcomes.

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APPENDICES

Appendix A

Table A-1: Relative risk of mortality per 10μg/m^3 from long-term exposure to $PM_{2.5}$

Author	Period of analysis	Disease	Age	Relative risk
Pope *et al.* (2015)	1999-2008	All-cause	All	7% (6%, 9%)
Pope *et al.* (2015)	1999-2008	Cardiovascular	All	12% (10%, 15%)
Carey *et al.* (2013)	2003-2007	All-cause	>40	13% (0%, 27%)
Carey *et al.* (2013)	2003-2007	Respiratory	>40	30% (15%, 47%)
Puett *et al.* (2009)	1992-2002	All-cause	All	26% (2%, 54%)
Jerrett *et al.* (2013)	1982-2000	All-cause	All	3.2% (0.2%, 6.2%)
Jerrett *et al.* (2013)	1982-2000	Cardiovascular	All	6.4% (1.6%, 11.4%)
Jerrett *et al.* (2013)	1982-2000	Respiratory	All	4.6% (-4.7%, 14.8%)
Jerrett *et al.* (2009)	1999-2000	Any cause	All	0.1% (-0.4%, 0.7%)
Jerrett *et al.* (2009)	1999-2000	Respiratory	All	3% (-4%, 11%)
Jerrett *et al.* (2009)	1977-2000	Cardiovascular	All	1.4% (0.5%, 2.3%)
Laden *et al.* (2006)	1979-1988	Total mortality	All	16% (7%, 26%)
Laden *et al.* (2006)	1979-1988	Lung cancer	All	27% (-4%, 69%)
Laden *et al.* (2006)	1979-1988	Cardiovascular	All	28% (13%, 44%)
Laden *et al.* (2006)	1979-1988	Respiratory	All	8% (-21%, 49%)
Lipsett *et al.* (2011)	1999-2005	All-cause	All	1% (-5%, 9%)
Lipsett *et al.* (2011)	1999-2005	Cardiovascular	All	7% (-5%, 19%)

Author	Period of analysis	Disease	Age	Relative risk
Cesaroni *et al.* (2013)	2001-2010	All	All	4% (3%, 5%)
Cesaroni *et al.* (2013)	2001-2010	Cardiovascular	All	10% (6%, 13%)
Pope *et al.* (2002)	1979-1983 1999-2000	All-cause	all	6% (2%, 11%)
Dockery *et al.* (1993)	1976-1989	All-cause	>65	13% (4%, 23%)
Dockery *et al.* (1993)	1976-1989	Cardiovascular	>65	18% (6%, 32%)
Lepeule *et al.* (2012b)	1974-2009	All-cause	25-74	14% (7%, 22%)
Lepeule *et al.* (2012b)	1974-2009	Cardiovascular	25-74	26% (14%, 40%)
Lepeule *et al.* (2012b)	1974-2009	Lung-cancer	25-74	37% (7%, 75%)
Pope *et al.* (1995)	1982-1989	All	>30	26% (8%, 47%)
Jerrett *et al.* (2005)	1982-2000	All causes	All	17% (5%, 30%)
Jerrett *et al.* (2005)	1982-2000	Lung cancer	All	1.44% (0.98%, 2.11%)
Beelen *et al.* (2008)	1987-1996	Cardiovascular	55-69	4% (-10%, 21%)
Beelen *et al.* (2008)	1987-1996	Respiratory	55-69	7% (-25%, 52%)
Beelen *et al.* (2008)	1987-1996	Lung cancer	55-69	6% (-18%, 38%)
Beelen *et al.* (2008)	1987-1996	All	55-69	6% (3%, 16%)
Zeger *et al.* (2008)	2000-2005	All	All (Central)	6.8% (4.9%, 8.7%)
Zeger *et al.* (2008)	2000-2005	All	All (Eastern)	13.2% (9.5%, 16.9%)
Ostro *et al.* (2010)	2002-2007	All 6.1μg/m^3	Female >30	1.45% (1.36%, 1.55%)
Hart *et al.* (2011)	1985-2000	All	All	10% (3%, 18%)
Crouse *et al.* (2012)	1991-2001	All	All	10% (5%, 15%)
Kloog *et al.* (2013)	2000-2008	All	All	1.6% (1.5%, 1.8%)
Hoek *et al.* (2013)	Up to 2013	All	All	6.2% (4%, 8.3%)
Yin *et al.* (2017)	1990-2006	Non-accidental causes	>40	9% (8%, 9%)
Yin *et al.* (2017)	1990-2006	Cardiovascular	>40	9% (8%, 10%)

Author	Period of analysis	Disease	Age	Relative risk
Yin *et al*. (2017)	1990-2006	lung cancer	>40	12% (7%, 14%)
Yin *et al*. (2017)	1990-2006	COPD	>40	12% (10%, 13%)

Table A-2: Relative risk of mortality per 10μg/m³ from short-term exposure to $PM_{2.5}$

Author	Period of analysis	Disease	Age	Dose-response
Tsai *et al.* (2014)	2006-2008	All-cause	All	3.48% (1.74%, 5.8%)
Tsai *et al.* (2014)	2006-2008	Respiratory	All	2.32% (-4.64%, 9.86%)
Tsai *et al.* (2014)	2006-2008	Cardiovascular	All	4.06% (0.58%, 8.70%)
Braniš *et al.* (2010)	2006	Cardiovascular	All	5.6% (-10.20%, 24.10%)
Braniš *et al.* (2010)	2006	Respiratory	All	20.90% (-20.00%, 82.7%)
Braniš *et al.* (2010)	2006	All-cause	All	4.9% (-5.70%, 16.7%)
Garrett and Casimiro (2011)	2004-2006	All-cause	All	0.67% (0.19%, 1.16%)
Garrett and Casimiro (2011)	2004-2006	All-cause	>65 years	0.62% (0.10%, 1.15%)
Garrett and Casimiro (2011)	2004-2006	Cardiovascular	All	1.49% (0.32%, 2.67%)
Garrett and Casimiro (2011)	2004-2006	Cardiovascular	>65 years old	2.39% (1.29%, 3.5%)
Jiménez *et al.* (2011)	2003-2005	Respiratory	>65 years	2.5% (1.3%, 3.7%)
Lee *et al.* (2016)	2007-2011	All-cause	>65 years	1.56% (1.19%, 1.94%
Lee *et al.* (2016)	2007-2011	Cardiovascular	>65 years	2.32% (1.57%, 3.07%)
Qiu *et al.* (2015)	2001-2011	Respiratory	>65 years	0.8% (0.4%, 1.2%)
Qiu *et al.* (2015)	2001-2011	COPD	>65 years	1.2% (0.4%, 1.9%)
Li *et al.* (2015b)	2005-2009	Cardiovascular	All	0.59% (0.067%, 1.11%)
Li *et al.* (2015b)	2005-2009	Respiratory	All	0.36% (−0.59%, 1.30%)
Li *et al.* (2015b)	2005-2009	All-cause	All	0.36% (0.034%, 0.695)
Lee *et al.* (2015)	2005-2007	All-cause	All	0.38% (0.21%, 0.55%)
Lee *et al.* (2015)	2005-2007	Cardiovascular	All	0.96% (0.46%, 1.46%)
Lee *et al.* (2015)	2005-2007	Respiratory	All	1.00% (0.23%, 1.78%)
Lee *et al.* (2015)	2005-2007	All-cause	All	0.39% (−0.02%, 0.81%)
Lee *et al.* (2015)	2005-2007	Cardiovascular	All	0.69% (0.05%, 1.33%)

Author	Period of analysis	Disease	Age	Dose-response
Lee *et al.* (2015)	2005-2007	Respiratory	All	0.46% (−0.07%, 0.98%)
Valdés *et al.* (2012)	1998-2007	Cardiovascular	All	1.33% (0.87%, 1.78%)
Valdés *et al.* (2012)	1998-2007	Respiratory	All	1.75% (1.01%, 2.49%)
Valdés *et al.* (2012)	1998-2007	COPD	All	1.94% (0.63%, 3.27%)
Valdés *et al.* (2012)	1998-2007	Cardiovascular	>65	1.54% (1.05%, 2.04%)
Valdés *et al.* (2012)	1998-2007	Respiratory	>65	2.13% (1.34%, 2.93%)
Valdés *et al.* (2012)	1998-2007	COPD	>65	1.95% (0.54%, 3.38%)
Franklin *et al.* (2008)	2000-2005	All-cause	All	0.74% (0.41%, 1.07%)
Franklin *et al.* (2008)	2000-2005	Cardiovascular	All	0.47% (0.02%, 0.92%)
Franklin *et al.* (2008)	2000-2005	Respiratory	All	1.01% (−0.03%, 2.05%)
Yorifuji *et al.* (2016)	2002-2013	All-cause	>65 years	0.6% (0.3%, 0.9%)
Yorifuji *et al.* (2016)	2002-2013	Cardiovascular diseases	>65 years	0.8% (0.2%, 1.4%)
Yorifuji *et al.* (2016)	2002-2013	Respiratory	>65 years	1% (0.2%, 1.8%)
Dai *et al.* (2014)	2000-2006	All causes	All	1.18% (0.93%, 1.44%)
Dai *et al.* (2014)	2000-2006	Cardiovascular	All	1.03% (0.65%, 1.41%)
Dai *et al.* (2014)	2000-2006	Respiratory	All	1.71% (1.06%, 2.35%)
Neuberger *et al.* (2007)	2000-2004	All-cause	All age	2.6% (1.1%, 4.1%)
Neuberger *et al.* (2007)	2000-2004	Cardiovascular	All age	5.5% (3.0%, 8.1%)
Neuberger *et al.* (2007)	2000-2004	Respiratory	All age	6.4% (1.9%, 11.2%)
Neuberger *et al.* (2007)	2000-2004	COPD	All age	9% (1.9%, 16.6%)
Samoli *et al.* (2014)	2001-2010	Cardiac causes	All age	0.59% (-0.11%, 1.30%)
Samoli *et al.* (2014)	2001-2010	COPD	All age	1.02% (-0.79%, 2.87%)

Author	Period of analysis	Disease	Age	Dose-response
Slaughter *et al.* (2005)	1995-1999	Cardiovascular	All	1% (-3%, 4%)
Faustini *et al.* (2013)	2005-2009	COPD (11µg/m^3)	>35	11.6% (2%, 22.2%)
Kloog *et al.* (2013)	2000-2008	All	All	2.8% (2%, 3.5%)

Table A-3: Relative risk of hospitalisation per 10 µg/m^3 from short-term exposure to PM$_{2.5}$

Author	Period of analysis	Disease	Age	Dose-response
Rodopoulou *et al.* (2014)	2007-2011	Respiratory	All	5.2% (-0.5%, 11.3%)
Rodopoulou *et al.* (2014)	2007-2011	Cardiovascular	All	4.5% (-4.2%, 14.1%)
Braniš *et al.* (2010)	2006	Cardiovascular	All	16.4% (5.2%, 28.7%)
Braniš *et al.* (2010)	2006	Respiratory	All	33.4% (12.6%, 57.9%)
Liu *et al.* (2016)	2013-2014	Respiratory	All	1.4% (0.7%, 2.1%)
Kloog *et al.* (2014)	2000-2006	Respiratory	>65	2.23% (1.91%, 2.56%)
Kloog *et al.* (2014)	2000-2006	Cardiovascular	>65	0.78% (0.54%, 1.01%)
Kloog *et al.* (2014)	2000-2006	COPD	>65	1.83% (1.18%, 2.48%)
Xu *et al.* (2016)	2013-2013	Respiratory	All	0.23% (0.11%-0.34%)
Ignotti *et al.* (2010)	2005-2005	Respiratory	< 5 years	4.7% (0.6%, 9.1%)
Ignotti *et al.* (2010)	2005-2005	Respiratory	>65	4.3% (0.25%, 8.6%)
Zanobetti *et al.* (2009)	2000-2003	Cardiovascular	>65	1.89% (1.34%, 2.45%)
Zanobetti *et al.* (2009)	2000-2003	Respiratory	>65	2.07% (1.20%, 2.95%)
Peng *et al.* (2009)	2000-2006	Respiratory	>65	0.44% (–0.36%, 1.23%)
Peng *et al.* (2009)	2000-2006	Cardiovascular	>65	0.64% (0.12%, 1.15%)
Hansen *et al.* (2012)	2001-2007	Cardiovascular	All	4.48% (0.74%, 8.36%)
Nakhlé *et al.* (2015)	2012	Respiratory	All	1.6% (0, 3.2%)
Nakhlé *et al.* (2015)	2012	Respiratory	>65	3.6% (1.1%, 6%)
Nakhlé *et al.* (2015)	2012	Respiratory	<16	1.3% (-1.5%, 4.2%)
Guo *et al.* (2009)	2004-2006	Cardiovascular	All	0.5% (0.1%, 0.9%)
Barnett *et al.* (2006)	1998-2001	Cardiovascular (3.8 µg/m^3)	>65	1.3% (0.6%, 2.0%)
Fung *et al.* (2006)	1995-1999	Respiratory (4 µg/m^3)	>65	0.7% (-0.6%, 2%)
Halonen *et al.* (2008)	1998-2004	COPD (1.1 µg/m^3)	All	7.8% (3.5%, 12.3%)
Peel *et al.* (2005)	1993-2000	Respiratory	All	1.6% (-0.3%, 3.5%)

Author	Period of analysis	Disease	Age	Dose-response
Santus *et al.* (2012)	2007-2008	Respiratory	All	0.3% (-1.7%, 2.3%)
Chen *et al.* (2004)	2001-2005	COPD	All	8% (2%, 15%)
Belleudi *et al.* (2010)	1995-1999	COPD	>35	1.88% (-0.27%, 4.09%)
Dominici *et al.* (2006)	1999-2002	COPD	>65	0.91% (0.18%, 1.64%)
Dominici *et al.* (2006)	1999-2002	Respiratory	>65	0.92% (0.41%, 1.43%)
Ko *et al.* (2007)	2000-2004	Respiratory	>65	3.1% (2.6%, 3.6%)
Tsai *et al.* (2013)	2006-2010	Respiratory (17.6 $\mu g/m^3$)	All (Warm)	12% (8%, 16%)
Tsai *et al.* (2013)	2006-2010	Respiratory (17.6 $\mu g/m^3$)	All (Cool)	3% (0, 7%)
Santus *et al.* (2012)	2007-2008	asthma	All	0.3% (-1.7%, 2.3%)

Table A-4: Relative risk of mortality per 10 μg/m³ from short-term exposure to PM_{10}

Author	Period of analysis	Disease	Age	Dose-response
Romieu *et al.* (2012)	1997-2005	All-cause	All	0.77% (0.60%, 1.00%)
Romieu *et al.* (2012)	1997-2005	Cardiopulmonary	All	0.94% (0.84%, 1.05%)
Romieu *et al.* (2012)	1997-2005	Cardiopulmonary	>65	1.15% (0.93%, 1.37%)
Romieu *et al.* (2012)	1997-2005	Respiratory	All	1.39% (0.98%, 1.81%)
Romieu *et al.* (2012)	1997-2005	Respiratory	>65	0.72% (0.54%, 0.89%)
Romieu *et al.* (2012)	1997-2005	Cardiovascular	All	0.72% (0.54%, 0.89%)
Romieu *et al.* (2012)	1997-2005	Cardiovascular	>65	0.88% (0.59%, 1.18%)
Romieu *et al.* (2012)	1997-2005	COPD	All	2.44% (1.36%, 3.59%)
Romieu *et al.* (2012)	1997-2005	COPD	>65	1.98% (0.78%, 3.23%)
Chen *et al.* (2010b)	2004-2006	All-cause	All	0.24% (− 0.03%, 0.51%)
Chen *et al.* (2010b)	2004-2006	Cardiovascular	All	0.67% (0.29%, 1.04%)
Chen *et al.* (2010b)	2004-2006	Respiratory	All	0.21% (− 0.82%, 1.24%)
Guo *et al.* (2010)	2005-2007	Cardiovascular	All	0.48% (0.23%, 0.73%)
Carugno *et al.* (2016)	2003-2006	All-cause	All	0.30% (90%CrI: -0.21%, 0.70%)
Carugno *et al.* (2016)	2003-2006	Cardiovascular	All	0.30% (90%CrI: -0.21%, 0.82%)
Carugno *et al.* (2016)	2003-2006	Respiratory	All	1.64% (90%CrI: 0.56%, 2.72%)
Perez *et al.* (2015)	2001-2010	All-cause	All	0.2% (-0.1%, 0.6%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	All	0.3% (-0.2%, 0.8%)
Perez *et al.* (2015)	2001-2010	Respiratory	All	1.7% (0.2%, 3.3%)
Perez *et al.* (2015)	2001-2010	All-cause	>65	0.2% (-0.2%, 0.5%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	>65	0.1% (-0.4%, 0.6%)
Perez *et al.* (2015)	2001-2010	Respiratory	>65	1.7% (-0.1%, 3.5%)
Jiménez *et al.* (2011)	2003-2005	Respiratory	>65	3.2% (1.9%, 4.4%)
Qiu *et al.* (2015)	2001-2011	Respiratory	>65	0.7% (0.4%, 1.1%)
Qiu *et al.* (2015)	2001-2011	COPD	>65	1.1% (0.5%, 1.7%)
Chen *et al.* (2012a)	1996-2008	All-cause	All	0.35% (0.18%, 0.52%)
Chen *et al.* (2012a)	1996-2008	Cardiovascular	All	0.44% (0.23%, 0.64%)

Author	Period of analysis	Disease	Age	Dose-response
Chen *et al.* (2012a)	1996-2008	Respiratory	All	0.56% (0.31%, 0.81%)
Zhang *et al.* (2011a)	2003-2008	Respiratory	All	0.101% (0.057%, 0.145%)
Guo *et al.* (2014)	1999-2008	All-cause	All	0.40% (0.22%, 0.59%)
Guo *et al.* (2014)	1999-2008	Cardiovascular	All	0.13% (−0.16%, 0.41%)
Guo *et al.* (2014)	1999-2008	Respiratory	All	0.40% (0.07%, 0.73%)
Blanco-Becerra *et al.* (2014)	1998-2006	All-cause	All	0.76% (95%CI 0.27-1.26)
Blanco-Becerra *et al.* (2014)	1998-2006	Respiratory	All	1.30% (0.35%, 2.26%)
Yi *et al.* (2010)	2000-2006	All-cause	All	0.28% (0.12%, 0.44%)
Yi *et al.* (2010)	2000-2006	Cardiovascular	All	0.51% (0.19%, 0.83%)
Yi *et al.* (2010)	2000-2006	Respiratory	All	0.59% (–0.08%, 1.26%)
Wong *et al.* (2010)	2001-2004	All-cause	All	0.55% (0.26%, 0.85%)
Wong *et al.* (2010)	2001-2004	Cardiovascular	All	0.58% (0.22%, 0.93%)
Wong *et al.* (2010)	2001-2004	Respiratory	All	0.62% (0.22%, 1.02%)
Qian *et al.* (2007)	2001-2004	Respiratory	All	0.71% (0.20%, 1.23%)
Qian *et al.* (2007)	2001-2004	Cardiovascular	All	0.51% (0.28%, 0.75%)
Qian *et al.* (2007)	2001-2004	All-cause	All	0.36% (0.19%, 0.53%)
Neuberger *et al.* (2007)	2000-2004	All-cause	All age	1.2% (0.4%, 2.1%)
Neuberger *et al.* (2007)	2000-2004	Cardiovascular	All age	2.0% (0.9%, 3.1%)
Neuberger *et al.* (2007)	2000-2004	Respiratory	All age	3.0% (0.5%, 5.5%)
Neuberger *et al.* (2007)	2000-2004	COPD	All age	5.1% (1.3%, 9.1%)
Samoli *et al.* (2014)	2001-2010	Cardiac causes	All age	0.35% (-0.13%, 1.83%)
Faustini *et al.* (2013)	2005-2009	COPD (16µg/m^3)	>35	3.5% (-0.1%, 7.2%)
Kan and Chen (2003)	2000-2001	All-causes	>65	0.3% (0.1%, 0.5%)

Author	Period of analysis	Disease	Age	Dose-response
Meng *et al.* (2013)	2001-2008	COPD	All	0.78% (0.13%-1.42%)
Sunyer and Basagaña (2001)	1990-1995	All-causes (27µg/m^3)	>35	11.4% (0, 24%)
Wong *et al.* (2002)	1995-1998	Respiratory diseases	All	0.08% (0.1%, 1.4%)

Table A-5: Relative risk of hospitalisation per 10 μg/m^3 from short-term exposure to PM_{10}

Author	Period of analysis	Disease	Age	Dose-response
Zhang *et al.* (2015)	2009-2011	Respiratory	All	1.72% (1.0%, 3.1%)
Zhang *et al.* (2015)	2009-2011	Cardiovascular	>65	1.39% (0.3%, 7.6%)
Zhang *et al.* (2015)	2009-2011	Respiratory	>65	2.2% (1.5%, 3.4%)
Zhang *et al.* (2015)	2009-2011	Cardiovascular	All	3.4% (2.2%, 5.4%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	All	0.7% (0.1%, 1.2%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	≥65	1.1% (0.5%, 1.7%)
Tramuto *et al.* (2011)	2005-2007	Respiratory	All	2.2% (1.3%, 3.1%)
Rodopoulou *et al.* (2014)	2007-2011	Respiratory	All	3.2% (0.5%, 6.0%)
Rodopoulou *et al.* (2014)	2007-2011	Cardiovascular	All	3.6% (-0.4%, 7.7%)
Carugno *et al.* (2016)	2003-2006	Respiratory	All	0.77% (0.31%, 1.32%)
Phung *et al.* (2016)	2004-2007	Respiratory	All	0.7% (0.2%, 1.3%)
Perez *et al.* (2015)	2001-2010	All-cause	All	0.2% (0.01%, 0.3%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	All	0.4% (0.1%, 0.7%)
Perez *et al.* (2015)	2001-2010	Respiratory	All	0.2% (-0.4%, 0.9%)
Perez *et al.* (2015)	2001-2010	All-cause	>65	0.2% (-0.01%, 0.4%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	>65	0.4% (0.07%, 0.8%)
Perez *et al.* (2015)	2001-2010	Respiratory	>65	0.7% (-0.2%, 1.6%)
Andersen *et al.* (2007)	1999-2004	Cardiovascular	>65	1.9% (0.9%, 3%)
Andersen *et al.* (2007)	1999-2004	Respiratory	>65	2.6% (1%, 4.2%)
Andersen *et al.* (2007)	1999-2004	Asthma	5-18	5.5% (0.28%, 11%)
Middleton *et al.* (2008)	1995-2004	Cardiovascular	All	1.18% (-0.01%, 2.37%)
Middleton *et al.* (2008)	1995-2004	All-cause	All	0.85% (0.55%, 1.15%)
Middleton *et al.* (2008)	1995-2004	Respiratory	All	0.10% (-0.91%, 1.11%)
Yi *et al.* (2010)	2000-2006	Cardiovascular	All	0.77% (0.53%, 1.01%)
Yi *et al.* (2010)	2000-2006	Respiratory	All	1.19% (0.94%, 1.44%)
Hansen *et al.* (2012)	2001-2007	Cardiovascular	All	1.48% (0.24%, 2.74%)
Chen *et al.* (2007b)	1998-2001	Respiratory	All	4.0% (1.1%, 6.9%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	All	0.7% (0.1%, 1.2%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	>65	1.1% (0.5%, 1.7%)

Author	Period of analysis	Disease	Age	Dose-response
Prescott *et al.* (1998)	1992-1995	Cardiovascular	>65	2.3% (-1.9%, 6.9%)
Prescott *et al.* (1998)	1992-1995	Respiratory	>65	3.1% (-3.5%, 10.2%)
Gouveia *et al.* (2006)	1996-2000	Respiratory	>65	2.2% (1.4%, 3.1%)
Ebenstein *et al.* (2015)	2007-2009	Respiratory	All	0.80% (0.5%, 1.2%)
Nardocci *et al.* (2013)	2000-2008	Respiratory	All	4.25% (2.82%, 5.71%)
Nardocci *et al.* (2013)	2000-2008	Respiratory	< 5	5.74% (3.80%, 7.71%)
Nardocci *et al.* (2013)	2000-2008	Cardiovascular	>40	2.29% (0.86%, 3.73%)
Nakhlé *et al.* (2015)	2012	Respiratory	All	1.2% (0.4%, 2%)
Nakhlé *et al.* (2015)	2012	Respiratory	>64	1.9% (0.6%, 3.2%)
Nakhlé *et al.* (2015)	2012	Respiratory	<16	1.4% (0, 2.9%)
Freitas *et al.* (2016)	2001-2006	Respiratory	All	9.67% (7.54%, 11.84%)
Freitas *et al.* (2016)	2001-2006	Respiratory	< 5	6.60% (3.75%, 9.53%)
Nakhlé *et al.* (2015)	1996-2000	Respiratory	< 5	2.4% (1.7%, 3.1%)
Buadong *et al.* (2009)	2002-2006	Cardiovascular	>65	0.10% (0.03%, 0.19%)
Fung *et al.* (2006)	1995-1999	Respiratory (7.9 μg/m^3)	>65	1.4% (-0.2%, 2.9%)

Table A-6: Relative risk of all-cause mortality per 10 μg/m^3 from long-term exposure to PM_{10}

Author	Period of analysis	Disease	Age	Dose-response
Zhou *et al.* (2015)	2006-2010	Respiratory	>65	1.62% (0.22%, 3.46%)
Zhou *et al.* (2015)	2006-2010	Respiratory	All	1.05% (0.08%, 2.04%)
Carey *et al.* (2013)	2003-2007	All-cause	>40	7% (-1%, 16%)
Carey *et al.* (2013)	2003-2007	Respiratory	>40	30% (15%, 47%)
Hales *et al.* (2012)	1996-1999	All-cause	30-74	7% (3% to 10%)
Hales *et al.* (2012)	1996-1999	Respiratory	30-74	1.3% (0.5%, 2.1%)
Hales *et al.* (2012)	1996-1999	Cardiovascular	30-74	0.6% (0.1%, 1.1%)
Lipsett *et al.* (2011)	1996-2005	Cardiovascular	All	3% (-2%, 8%)
Heinrich *et al.* (2013)	1985-2008	All-cause	Female> 55	21% (5%, 38%)
Puett *et al.* (2008)	1992-2002	All-cause	Women 30-55	16 (5, 28)
Zhang *et al.* (2011b)	1998 - 2009	All-cause	>35	53 (50, 56)
Zhang *et al.* (2011b)	1998 - 2009	Cardiovascular	>35	55% (51%, 60%)

Table A-7: Relative risk of mortality per 10μg/m^3 from long-term exposure to NO_2

Author	Period of analysis	Disease	Age	City
Carey *et al.* (2013)	2002-2002	All-causes	>40	7% (4%, 11%)
Carey *et al.* (2013)	2002-2002	Respiratory	>40	17% (12%, 23%)
Puett *et al.* (2009)	1974-1976	All-causes	All	1.14% (1.03%, 1.25%)
Puett *et al.* (2009)	1974-1976	Lung cancer	All	1.48% (1.05%, 2.06%)
Puett *et al.* (2009)	1974-1976	Cardiopulmonary	All	1.27% (1.04%, 1.56%)
Jerrett *et al.* (2013)	1982-2000	All-causes	All	3.1% (0.8%, 5.6%)
Jerrett *et al.* (2013)	1982-2000	Cardiovascular	All	4.8% (1%, 8.7%)
Cesaroni *et al.* (2013)	2001-2010	All	All	3% (2%, 3%)
Cesaroni *et al.* (2013)	2001-2010	Cardiovascular	All	3% (2%, 4%)
Crouse *et al.* (2015)	1984-2006	All-causes (per 5 ppb)	All	5% (3%, 7%)
Crouse *et al.* (2015)	1984-2006	Cardiovascular (per 5 ppb)	All	4% (1%, 6%)
Crouse *et al.* (2015)	1984-2006	Respiratory (per 5 ppb)	All	4% (-1%, 8%)

Table A-8: Relative risk of mortality per 10μg/m^3 from short-term exposure to NO_2

Author	Period of analysis	Disease	Age	City
Chen *et al.* (2010b)	2004-2006	All-causes	All	1.30% (− 0.06%, 2.67%)
Chen *et al.* (2010b)	2004-2006	Cardiovascular	All	2.11% (0.22%, 4.00%)
Guo *et al.* (2010)	2005-2007	Cardiovascular	All	1.08% (0.13%, 2.04%)
Carugno *et al.* (2016)	2003-2006	All-causes	All	0.70% (90%: 0.20%, 1.18%)
Carugno *et al.* (2016)	2003-2006	Cardiovascular	All	1.12% (90%: 0.30%, 1.95%)
Carugno *et al.* (2016)	2003-2006	Respiratory	All	0.46% (90%CrI: -1.23%, 2.18%)
Perez *et al.* (2015)	2001-2010	All-causes	All	0.7% (0.1%, 1.3%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	All	0.4% (-0.1%, 0.8%)
Perez *et al.* (2015)	2001-2010	Respiratory	All	2.7% (0.5%, 5.0%)
Perez *et al.* (2015)	2001-2010	Any medical	>65	0.6% (0.0, 1.3%)
Perez *et al.* (2015)	2001-2010	Cardiovascular	>65	0.3% (-0.2%, 0.8%)
Perez *et al.* (2015)	2001-2010	Respiratory	>65	2.8% (0.7%, 4.9%)
Qiu *et al.* (2015)	2001-2011	Respiratory	>65	1.3% (0.9%, 1.8%)
Qiu *et al.* (2015)	2001-2011	COPD	>65	2.2% (1.4%, 3.0%)
Chen *et al.* (2012b)	1996-2008	All-causes	All	1.63% (1.09%, 2.17%)
Chen *et al.* (2012b)	1996-2008	Cardiovascular	All	1.80% (1.00%, 2.59%)
Chen *et al.* (2012b)	1996-2008	Respiratory	All	2.52% (1.44%, 3.59%)
Zhang *et al.* (2011a)	2003-2008	Respiratory	All	0.164% (0.144%, 0.184%)
Zhang *et al.* (2011a)	2003-2008	Cardiovascular	All	0.164% (0.144%, 0.184%)
Zhang *et al.* (2011a)	2003-2008	Cardiovascular	All	0.271% (0.086%, 0.457%)
Wong *et al.* (2010)	1996-2002	All-causes	All	1.23% (0.84%, 1.62%)
Wong *et al.* (2010)	1996-2002	Cardiovascular	All	1.36% (0.89%, 1.82%)
Wong *et al.* (2010)	2001-2004	Respiratory	All	1.48% (0.68%, 2.28%)
Neuberger *et al.* (2007)	2000-2004	All-causes	All age	2.9% (1.6%, 4.1%)
Neuberger *et al.* (2007)	2000-2004	Cardiovascular	All age	4.6% (2.9%, 6.3%)
Neuberger *et al.* (2007)	2000-2004	Respiratory	All age	6.7% (2.7%, 10.8%)
Neuberger *et al.* (2007)	2000-2004	COPD	All age	8.9% (3.0%, 10.8%)

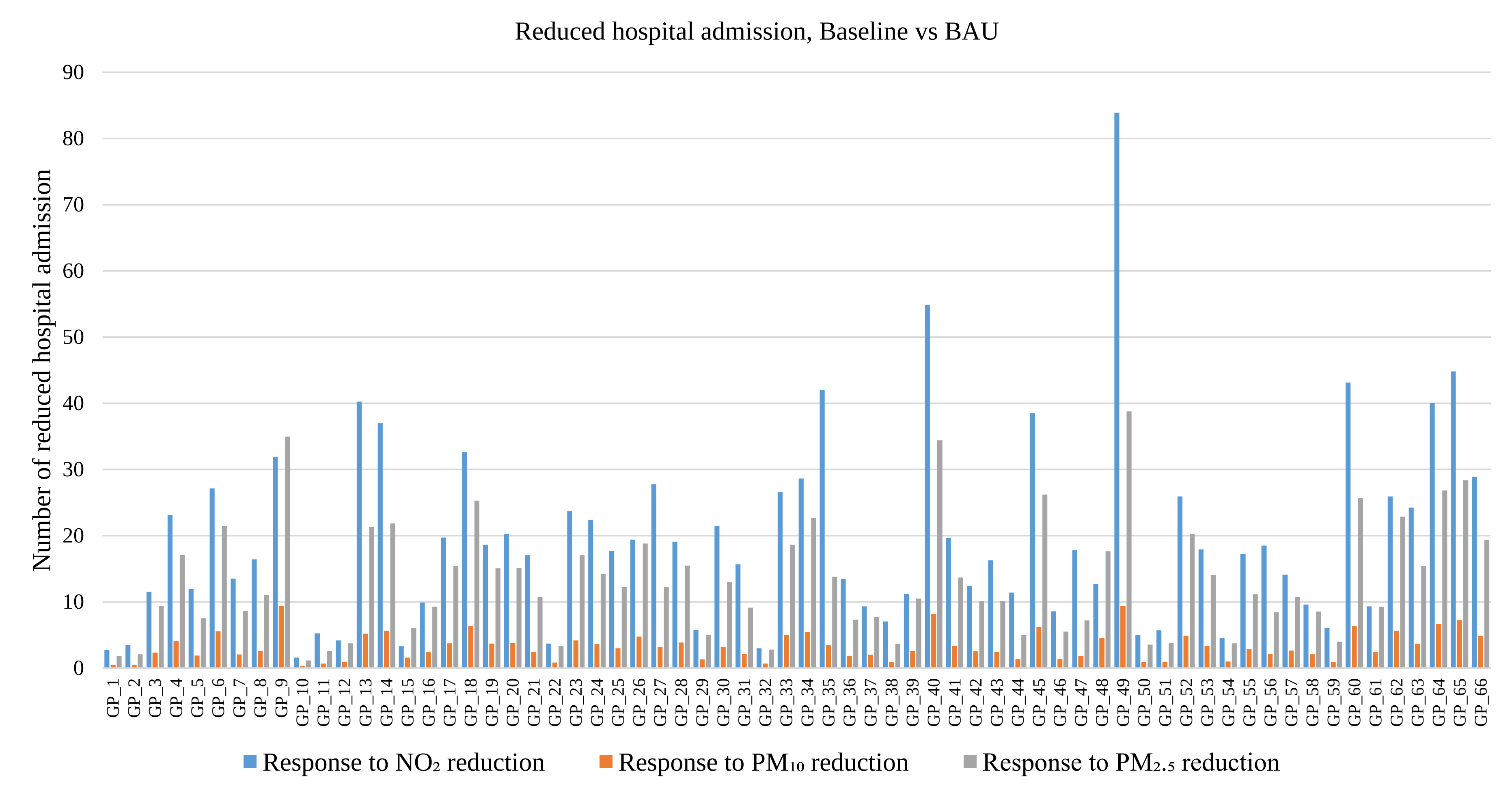
Author	Period of analysis	Disease	Age	City
Faustini *et al.* (2013)	2005-2009	COPD ($24\mu g/m^3$)	>35	19.6% (3.5%, 38.2%)
Kan and Chen (2003)	2000-2001	All-causes	>65	1.5% (0.8%, 2.2%)
Meng *et al.* (2013)	2001-2008	COPD	All	1.85% (1.40%, 2.29%)
Sunyer and Basagaña (2001)	1990-1995	COPD ($17\mu g/m^3$)	>35	3.4% (-1.7%, 7%)
Wong *et al.* (2002)	1995-1998	Respiratory	All	13% (0.4%, 2.2%)
Wong *et al.* (2002)	1995-1998	COPD	All	2.3% (0.6%, 4.1%)
Fischer *et al.* (2003)	1986-1994	COPD	45-64	5.3% (-8%, 20.5%)
Fischer *et al.* (2003)	1986-1994	COPD	65-74	17.3% (8.5%, 26.8%)
Fischer *et al.* (2003)	1986-1994	COPD	>75	4.6% (0, 9.4%)

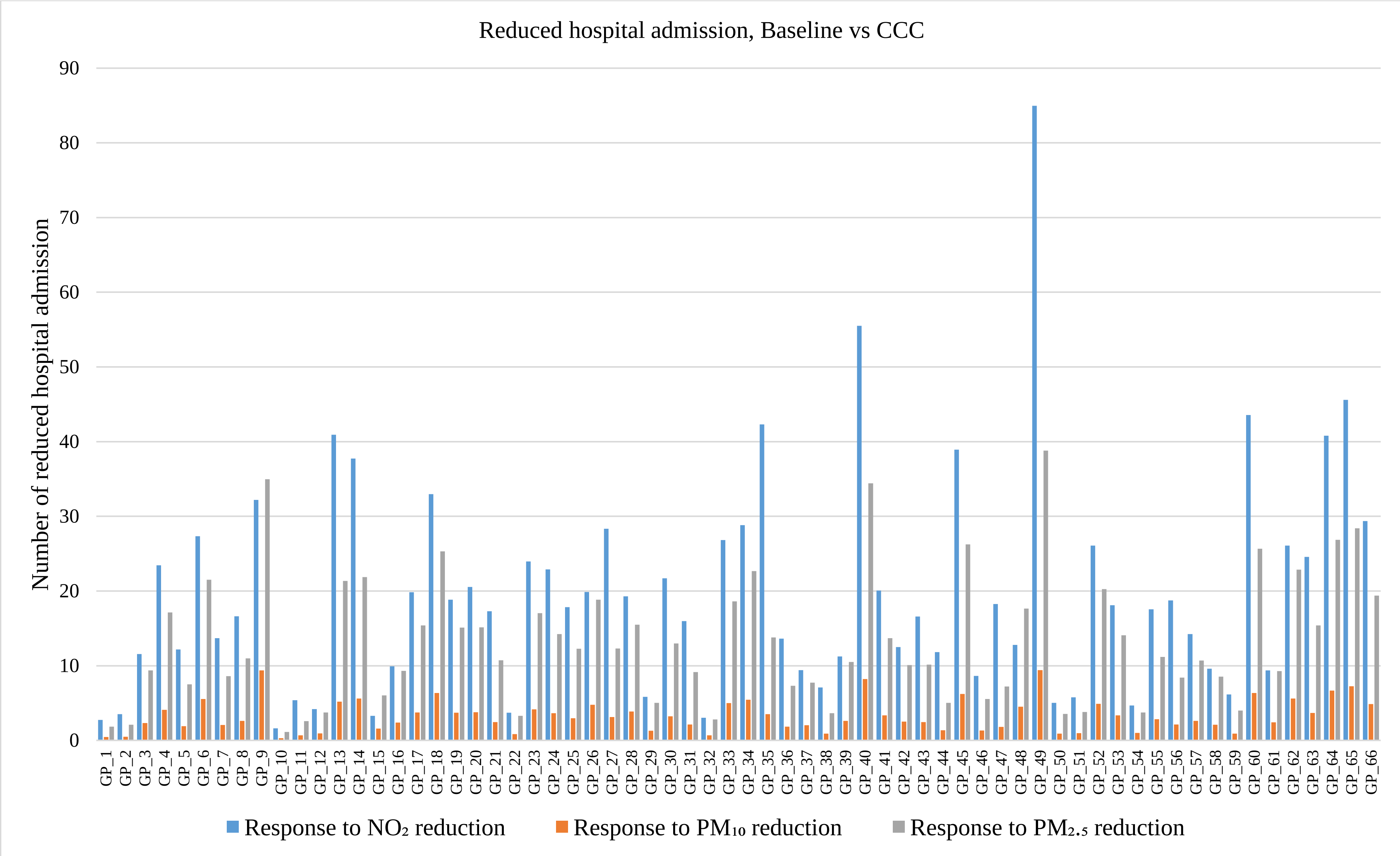
Table A-9: Relative risk of hospitalisation per 10 µg/m^3 from short-term exposure to NO_2

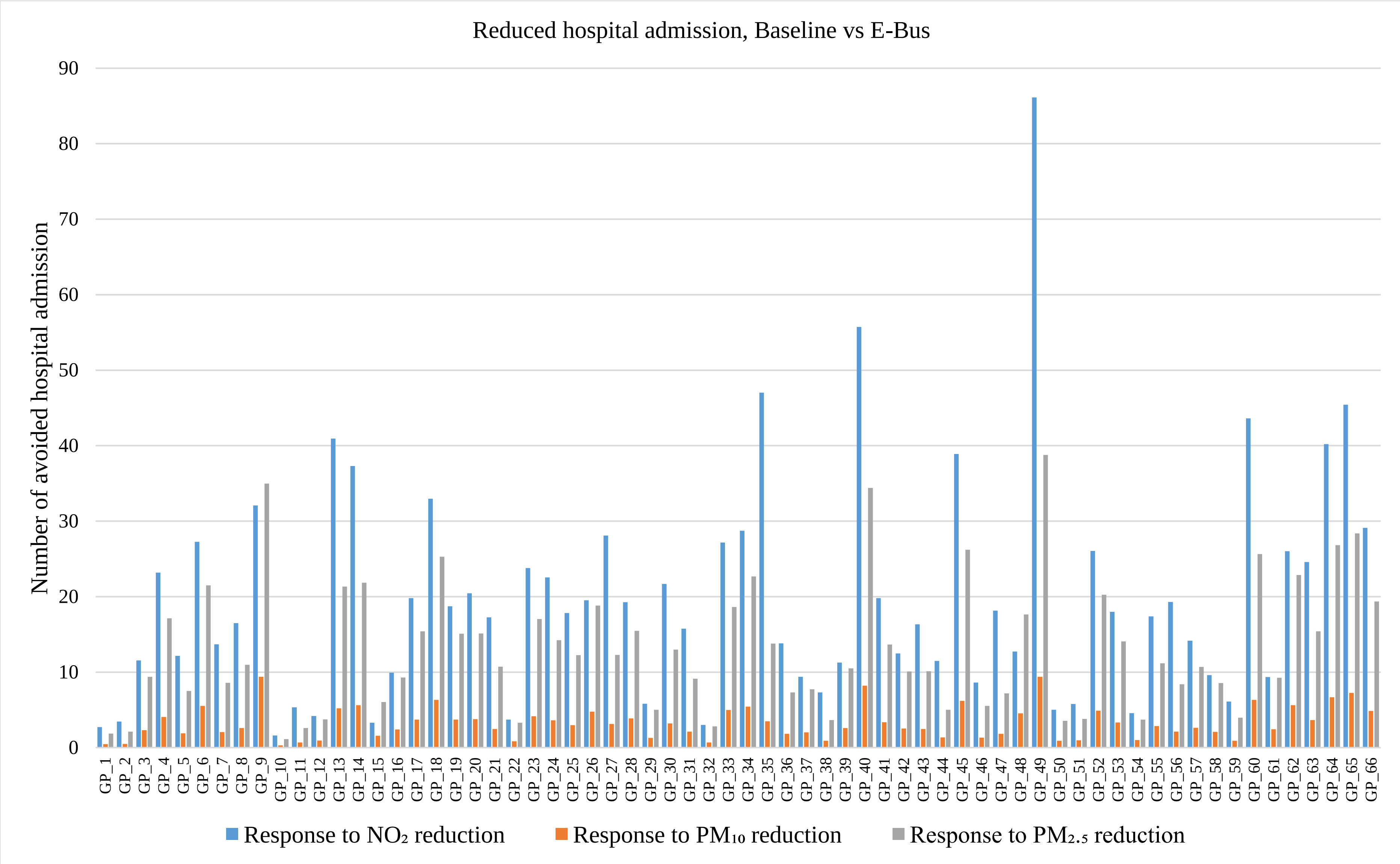
Author	Period of analysis	Disease	Age	City
Jevtić *et al.* (2014)	2007-2009	Cardiovascular (1 µg/m^3)	All	4.9% (0.9%, 9.1%) For NO_2 > 4 µg/m^3
Jevtić *et al.* (2014)	2007-2009	Cardiovascular (1 µg/m^3)	All	4.7% (0.7%, 8.9%) For NO_2 < 4 µg/m^3
Zhang *et al.* (2015)	2009-2011	Respiratory	All	2.57% (1.8%, 3.6%)
Zhang *et al.* (2015)	2009-2011	Cardiovascular	>65	1.18% (0.3%, 4.5%)
Zhang *et al.* (2015)	2009-2011	Respiratory	>65	1.6% (0.5%, 4.8%)
Zhang *et al.* (2015)	2009-2011	Cardiovascular	All	3.8% (1.9%, 7.4%)
Tramuto *et al.* (2011)	2005-2007	Respiratory	All	1.5% (0.4%, 2.6%)
Carugno *et al.* (2016)	2003-2006	Respiratory	All	1.20% (90%CrI: 0.53%, 1.81%)
Liu *et al.* (2016)	2013-2014	Respiratory	All	2.5% (0.8%, 4.2%)
Phung *et al.* (2016)	2004-2007	Respiratory	All	8% (6%, 11%)
Phung *et al.* (2016)	2004-2007	Cardiovascular	>65	0.8% (0.1%, 1%)
Andersen *et al.* (2007)	1999-2004	Cardiovascular	>65	1.8% (-1%, 4.7%)
Andersen *et al.* (2007)	1999-2004	Respiratory	>65	5.7% (1.2&, 10.2%)
Andersen *et al.* (2007)	1999-2004	Asthma	5-18	18% (4%, 33%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	All	0.5% (0.1%, 1.0%)
Larrieu *et al.* (2007)	1998-2003	Cardiovascular	>65	1.2% (0.7%, 1.7%)
Gouveia *et al.* (2006)	1996-2000	Respiratory	< 5 years	0.9% (0.5%, 1.3%)
Gouveia *et al.* (2006)	1996-2000	Respiratory	>65	1.2% (0.7%, 1.7%)
Guo *et al.* (2009)	2004-2006	Cardiovascular	All	1.6% (0.3%, 2.9%)
Barnett *et al.* (2006)	1998-2001	cardiovascular	>65	3.0% (2.1%, 3.9%)
Fung *et al.* (2006)	1995-1999	Respiratory	>65	1.2% (-0.3%, 2.7%)
Ko *et al.* (2007)	2000-2004	Respiratory	>65	2.6% (2.2%, 3.1%)
Santus *et al.* (2012)	2007-2008	asthma	All	1.3% (-0.1%, 2.7%)

Author	Period of analysis	Disease	Age	City
Sauerzapf *et al.* (2009)	2006-2007	COPD	>18	22% (9.2%, 36.2%)
Anderson *et al.* (1997)	1987-1992	COPD 50µg/m^3	All	2% (0.2%, 4.7%)
Faustini *et al.* (2013)	2001-2005	Respiratory	>35	1.19% (0.23%, 2.15%)
Faustini *et al.* (2013)	2001-2005	COPD	>35	0.82%% (-0.70%, 2.37%)
Fusco *et al.* (2001)	1995-1997	COPD 22.3µg/m^3	All age	2.5% (0.9%, 4.2%)
Fusco *et al.* (2001)	1995-1997	Respiratory 22.3µg/m^3	All age	4% (0.6%, 7.5%)
Yang *et al.* (2005)	1994-1998	COPD 5.5ppb	>65	11% (4%, 20%)

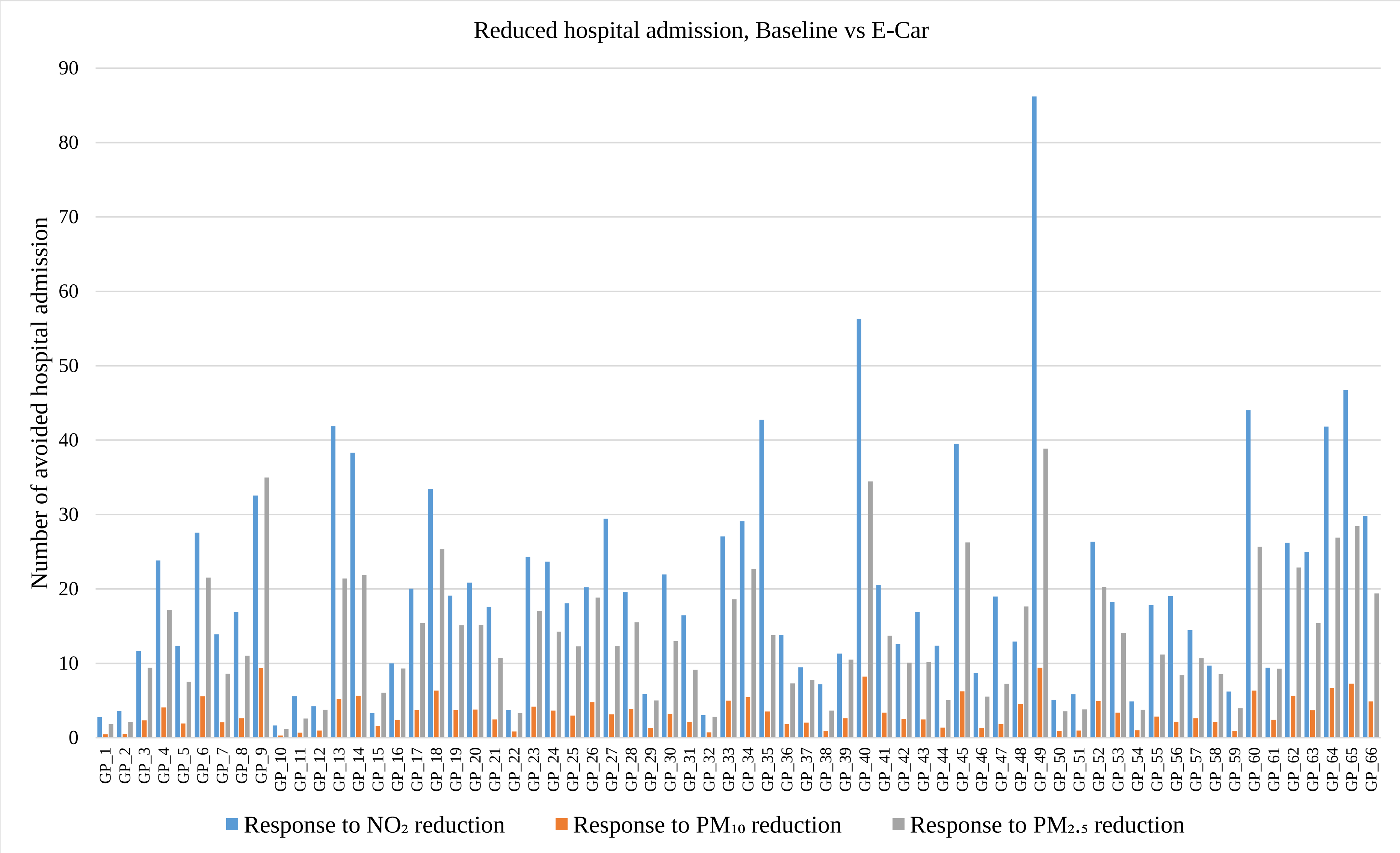
Appendix B: Reduced hospital admissions due to reduction in long-term exposure to NO_2, PM_{10} and $PM_{2.5}$ levels, Baseline vs all 2030 scenarios

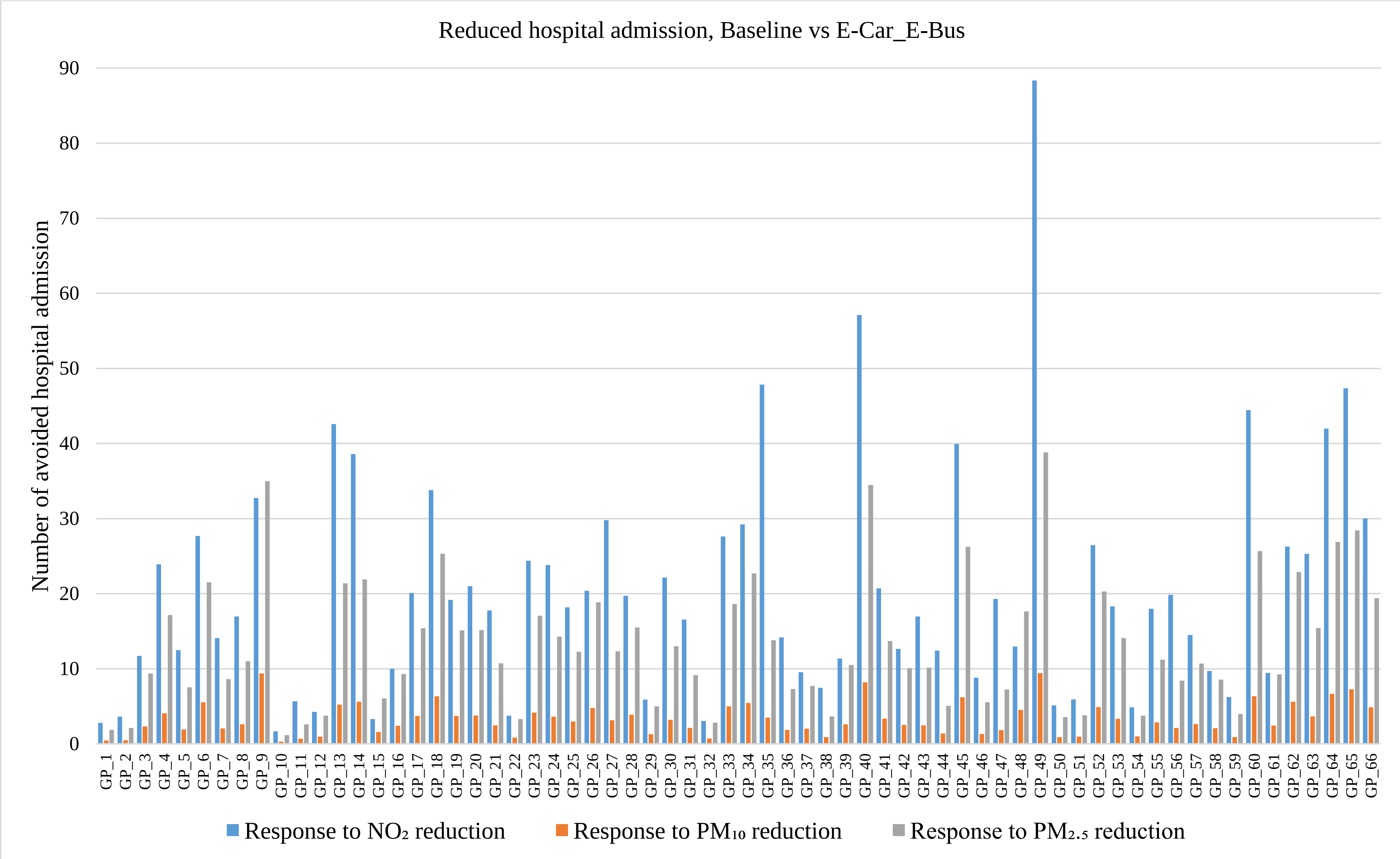


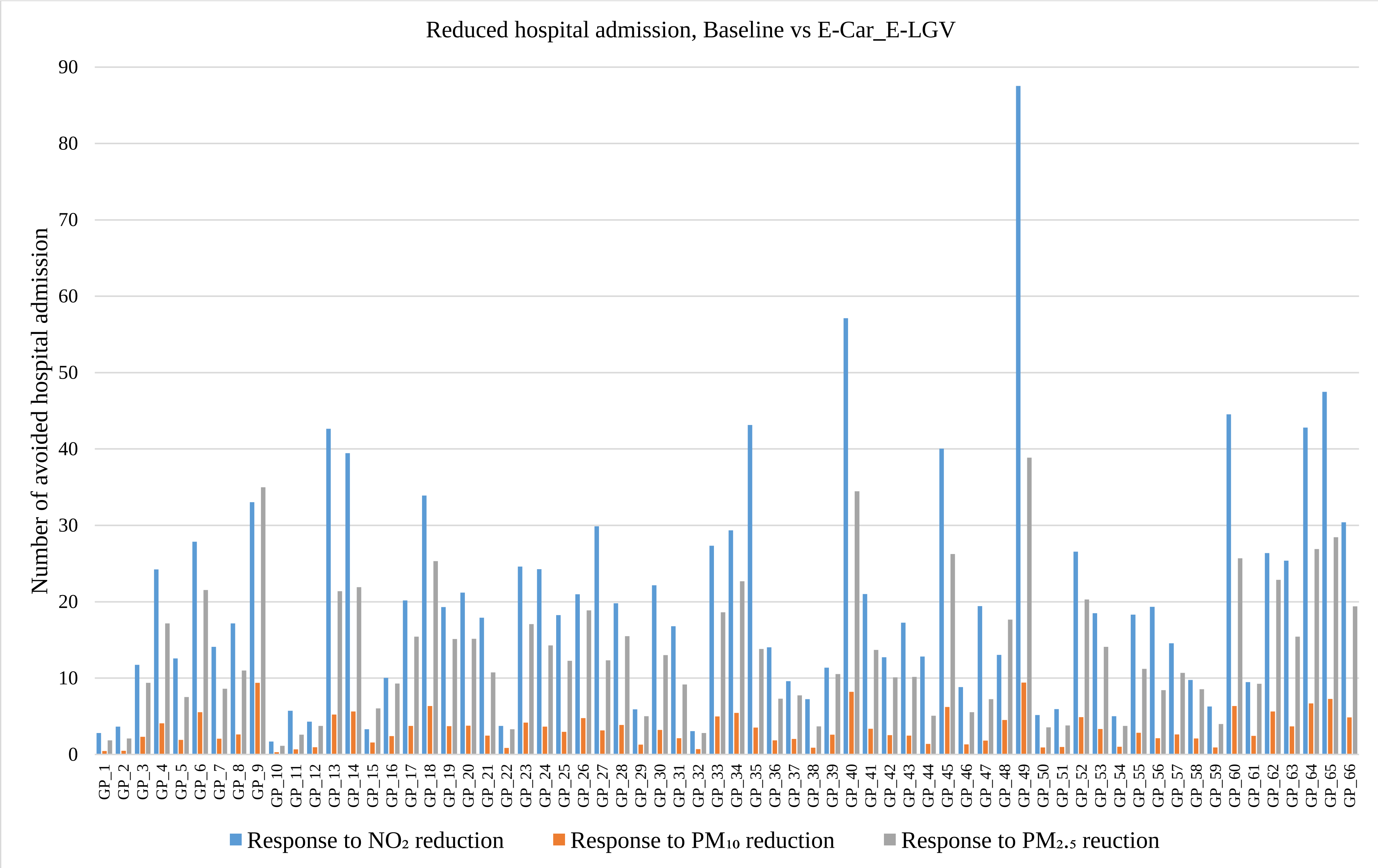

Reduced hospital admission, Baseline vs CCC
Number of reduced hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

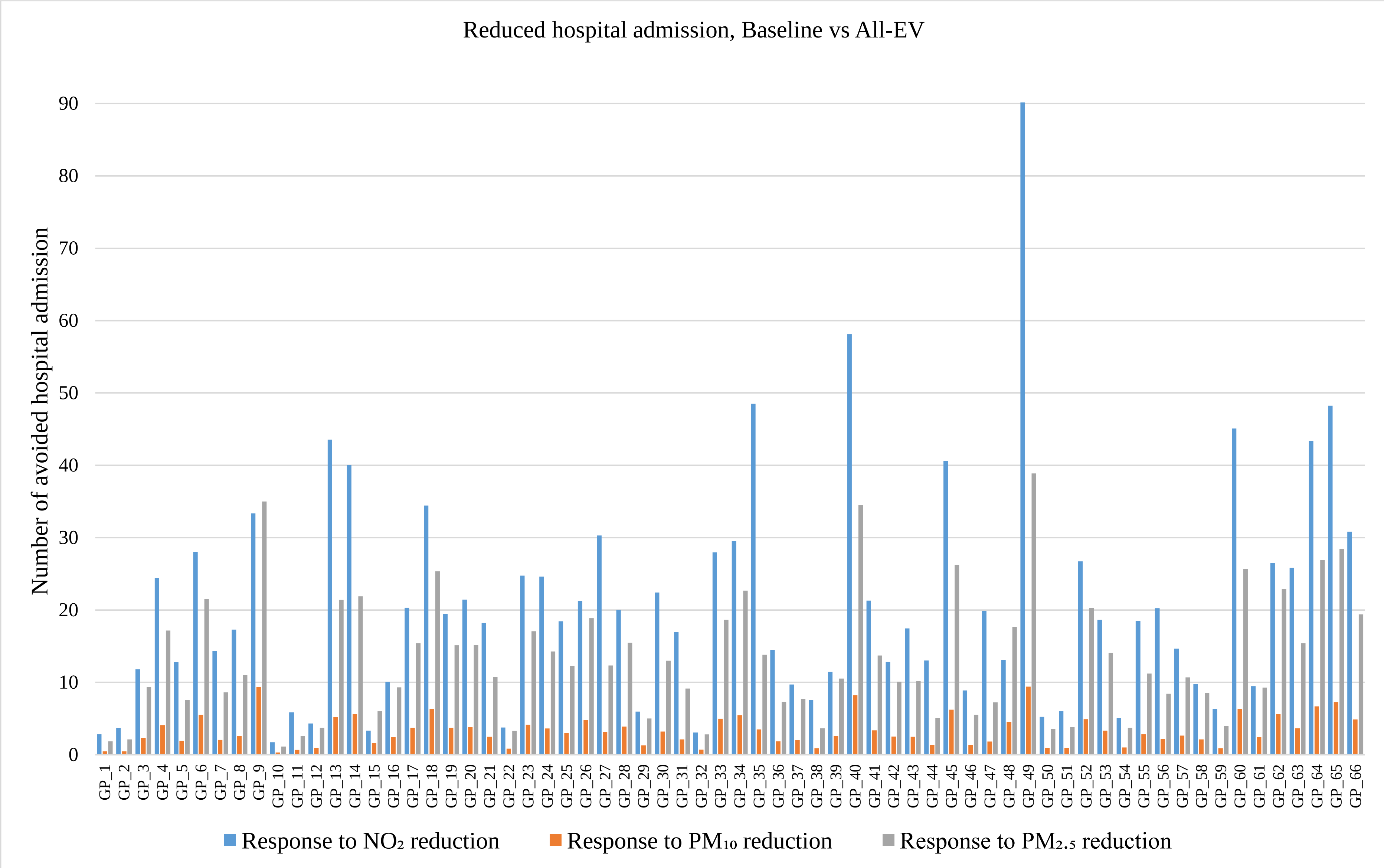


Reduced hospital admission, Baseline vs E-Bus
Number of avoided hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

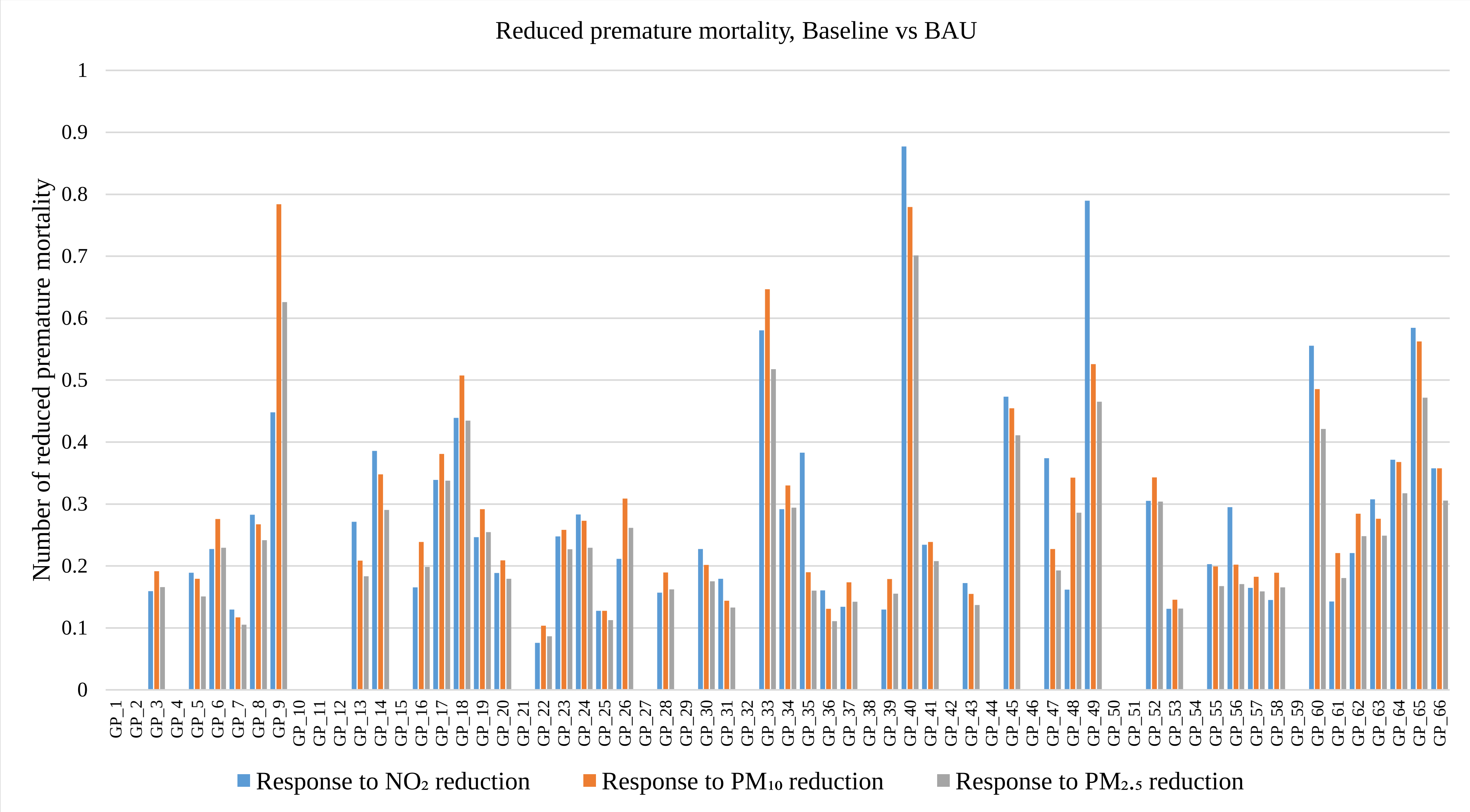

Reduced hospital admission, Baseline vs E-Car
Number of avoided hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1
GP_2
GP_3
GP_4
GP_5
GP_6
GP_7
GP_8
GP_9
GP_10
GP_11
GP_12
GP_13
GP_14
GP_15
GP_16
GP_17
GP_18
GP_19
GP_20
GP_21
GP_22
GP_23
GP_24
GP_25
GP_26
GP_27
GP_28
GP_29
GP_30
GP_31
GP_32
GP_33
GP_34
GP_35
GP_36
GP_37
GP_38
GP_39
GP_40
GP_41
GP_42
GP_43
GP_44
GP_45
GP_46
GP_47
GP_48
GP_49
GP_50
GP_51
GP_52
GP_53
GP_54
GP_55
GP_56
GP_57
GP_58
GP_59
GP_60
GP_61
GP_62
GP_63
GP_64
GP_65
GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

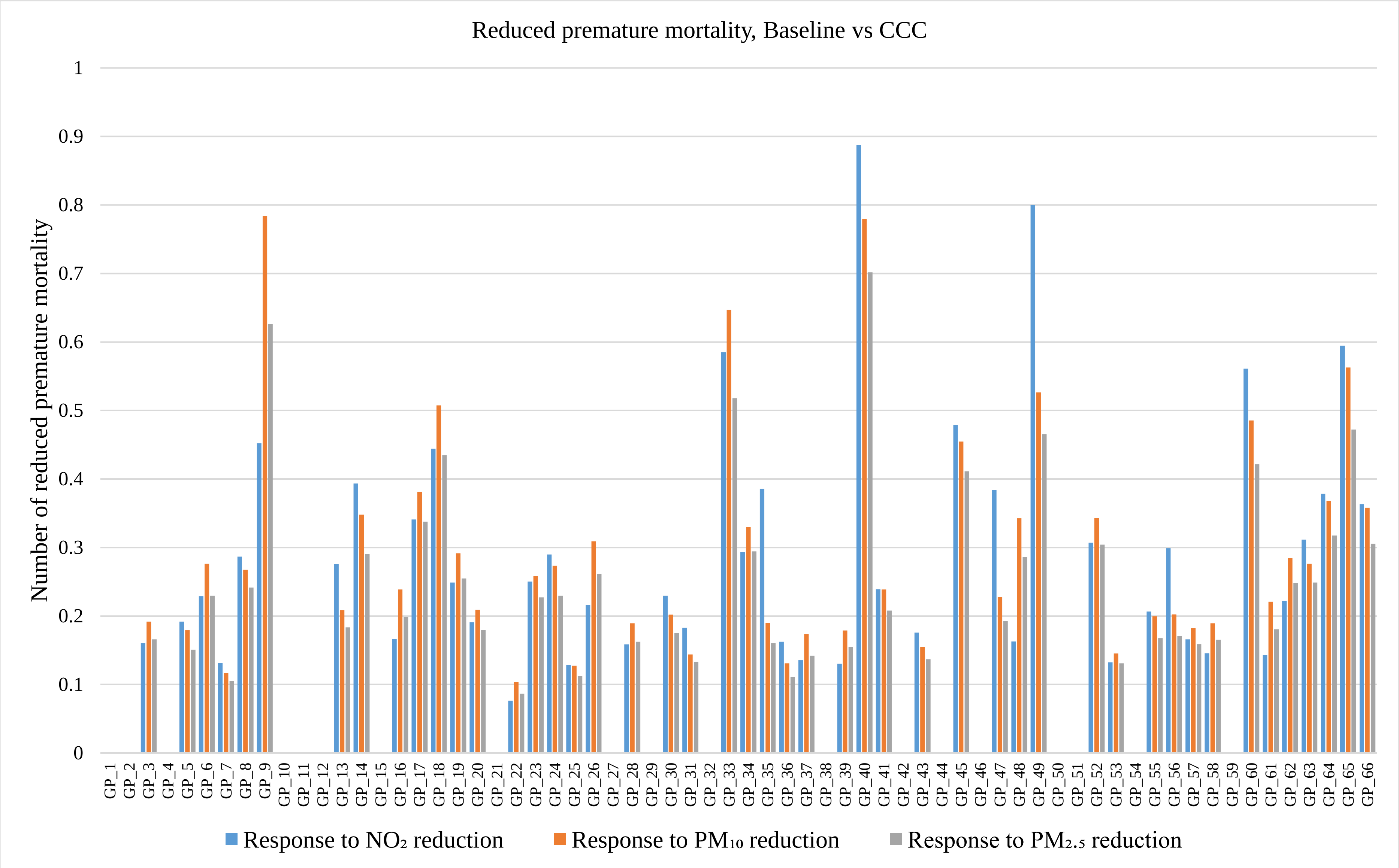

Reduced hospital admission, Baseline vs E-Car_E-Bus
Number of avoided hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

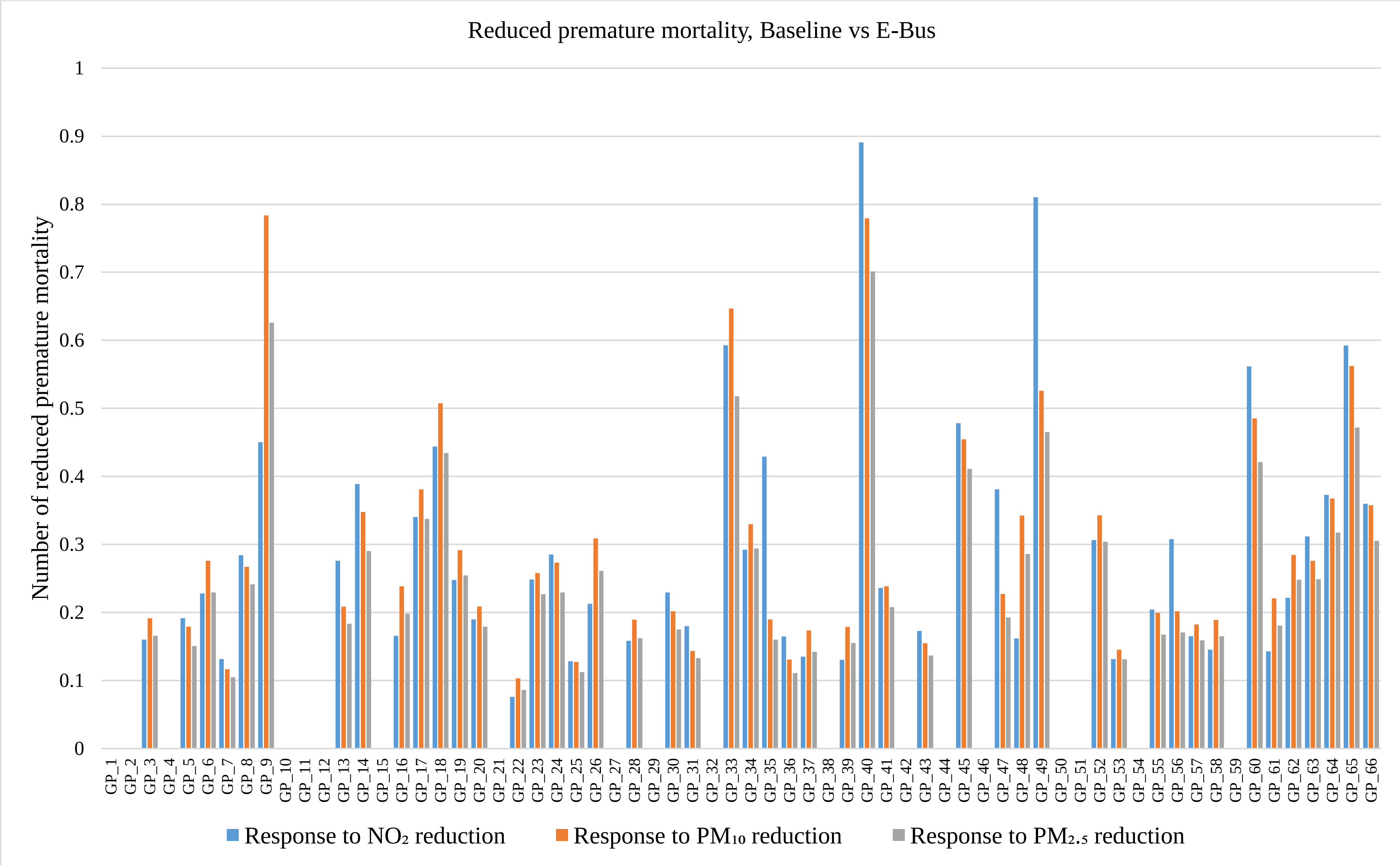

Reduced hospital admission, Baseline vs E-Car_E-LGV
Number of avoided hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1 … GP_66
Response to NO₂ reduction
Response to PM₁₀ reduction
Response to PM₂.₅ reuction

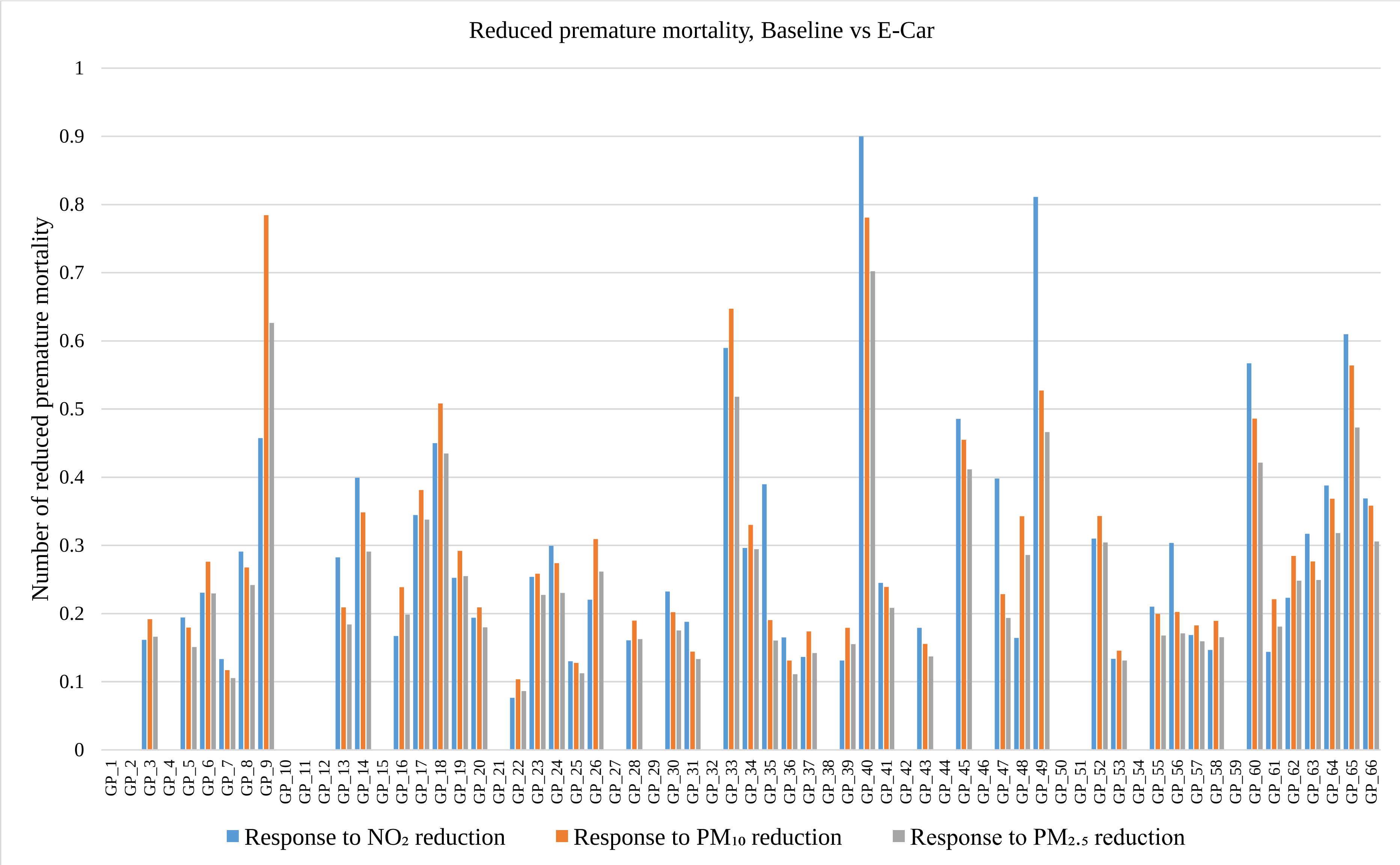

Reduced hospital admission, Baseline vs All-EV
Number of avoided hospital admission
0
10
20
30
40
50
60
70
80
90
GP_1 … GP_66
Response to NO₂ reduction
Response to PM₁₀ reduction
Response to PM₂.₅ reduction

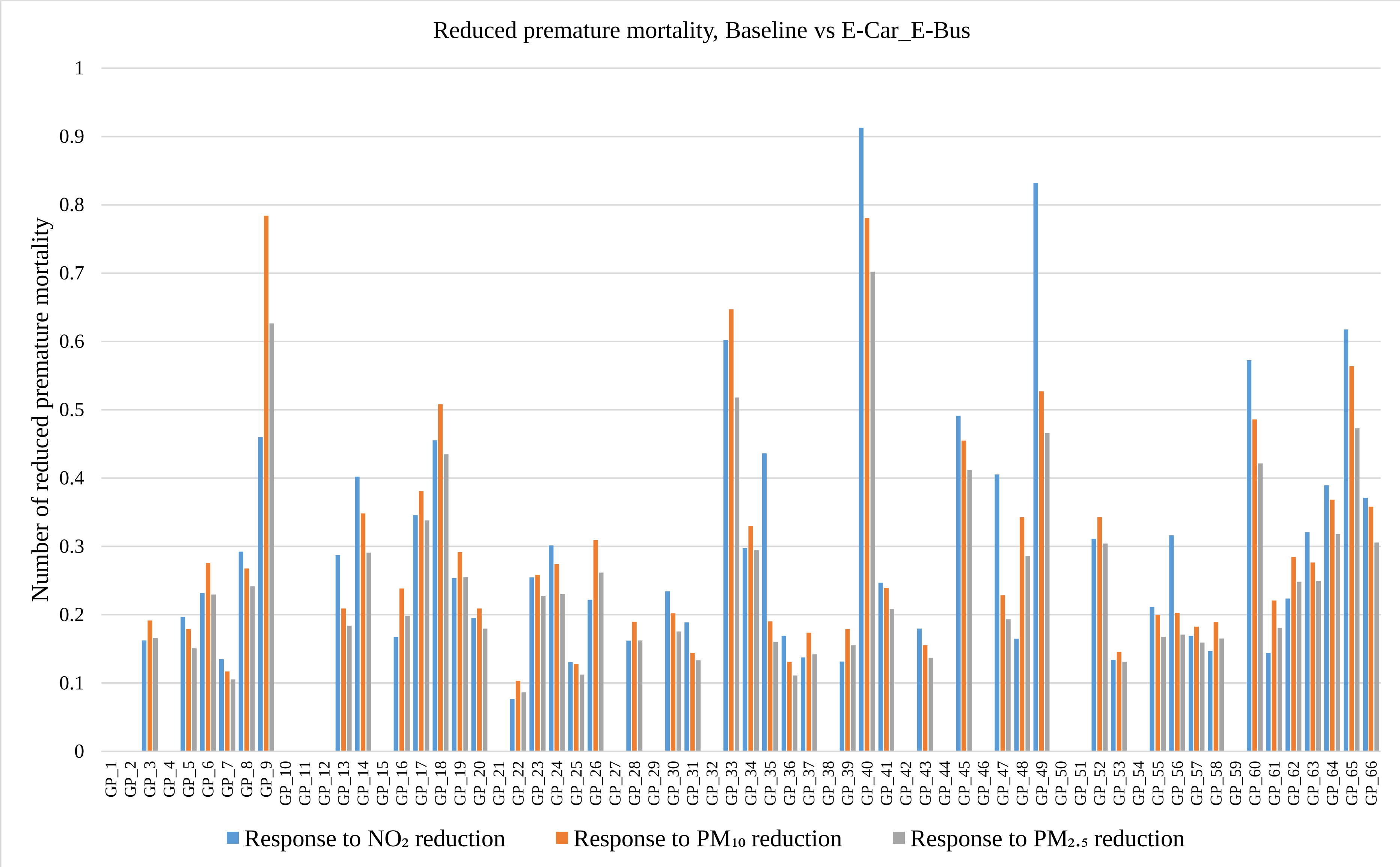
Appendix C: Reduced mortality due to reduction in long-term exposure to NO_2, PM_{10} and $PM_{2.5}$ levels, Baseline vs all 2030 scenarios

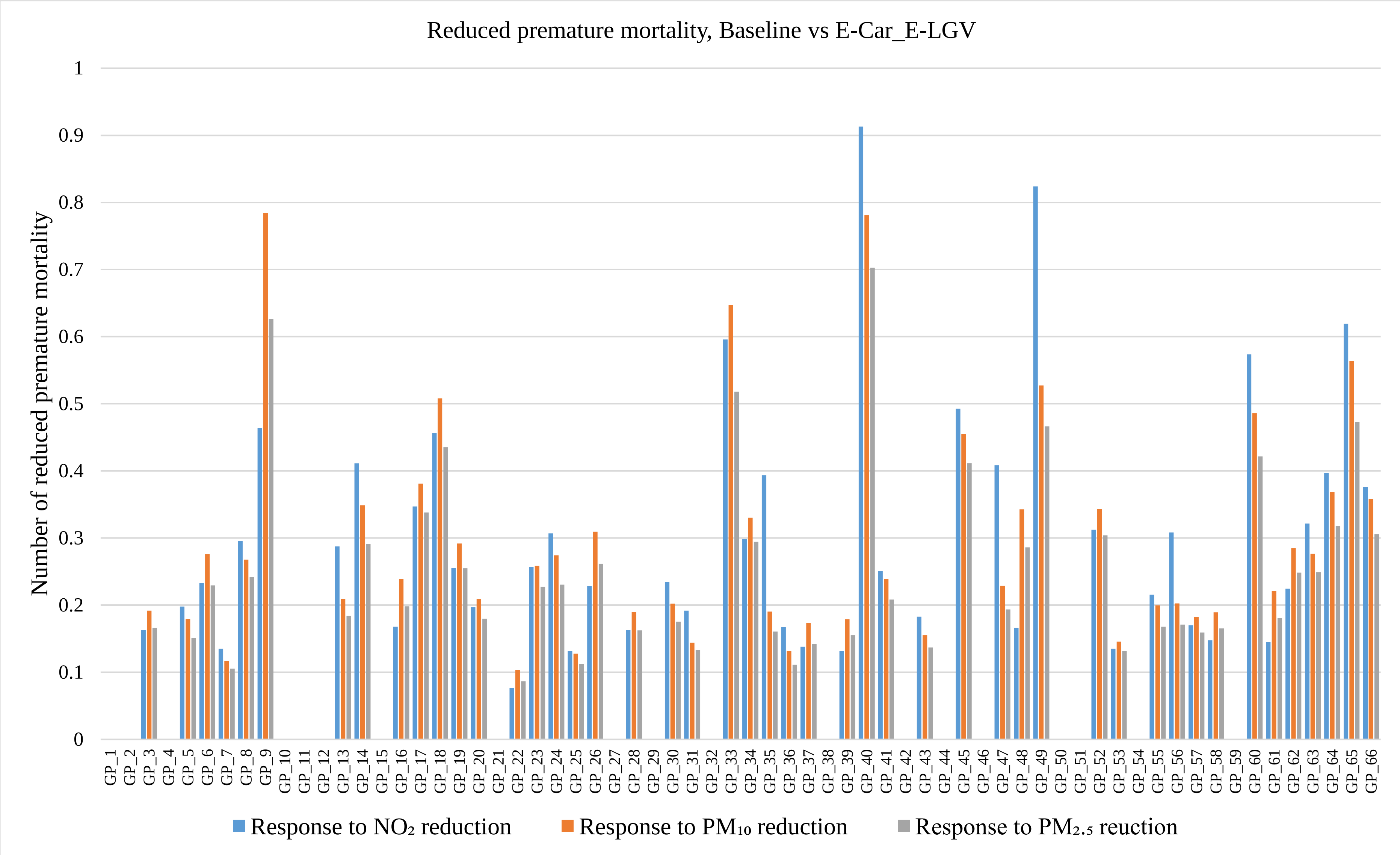



Reduced premature mortality, Baseline vs CCC
Number of reduced premature mortality
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

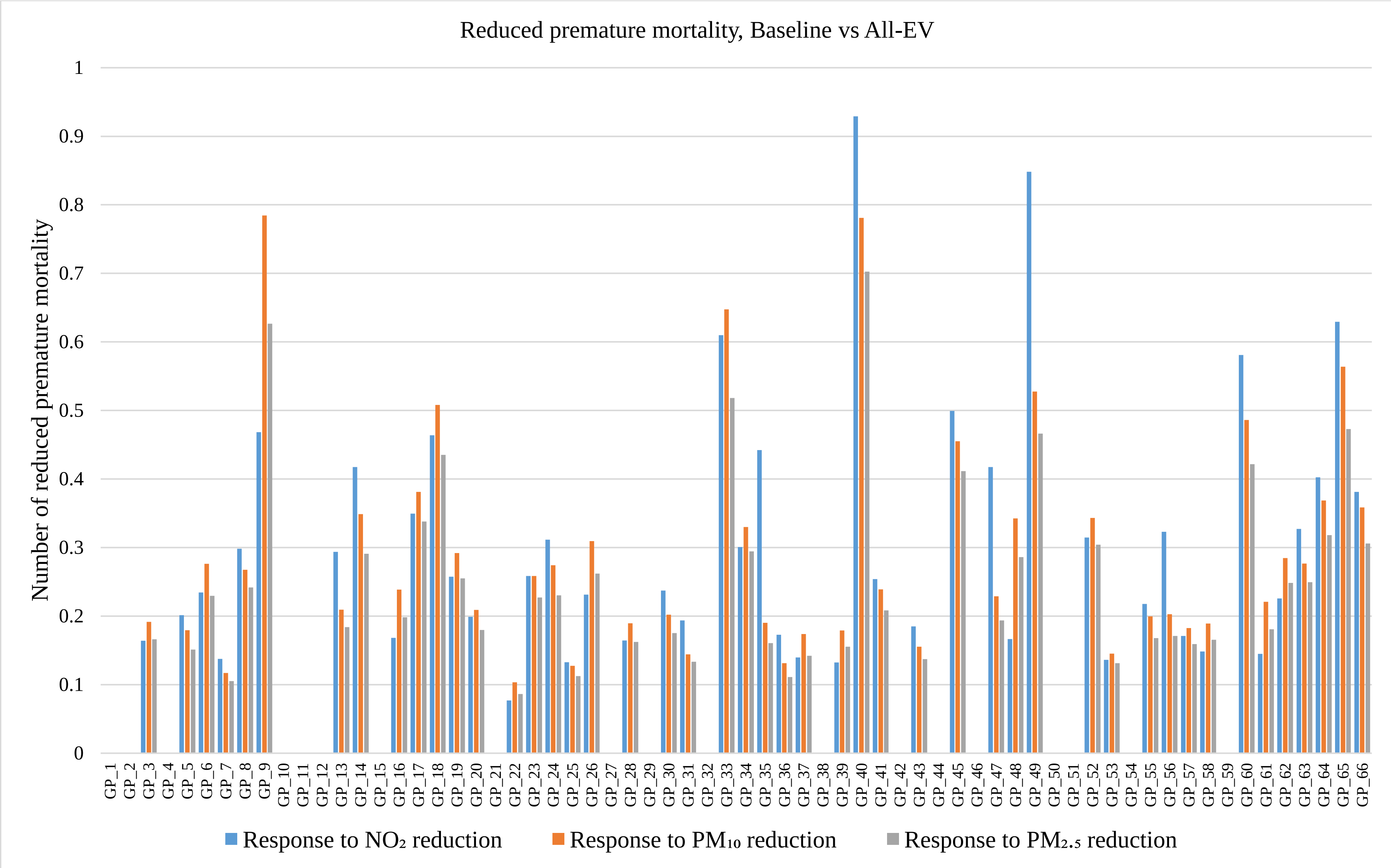

Reduced premature mortality, Baseline vs E-Bus
Number of reduced premature mortality
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction


Reduced premature mortality, Baseline vs E-Car
Number of reduced premature mortality
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction


Reduced premature mortality, Baseline vs E-Car_E-Bus
Number of reduced premature mortality
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

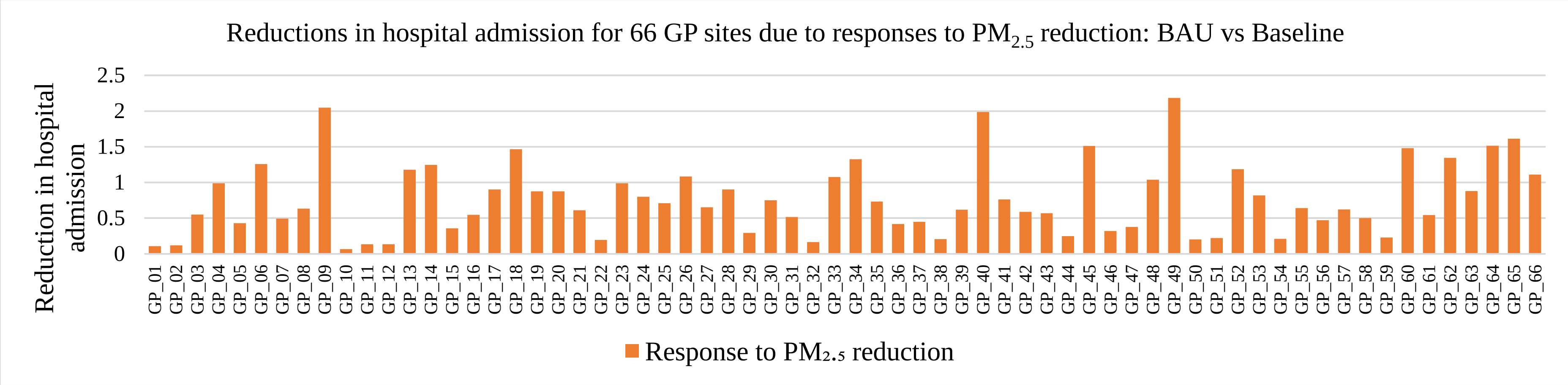


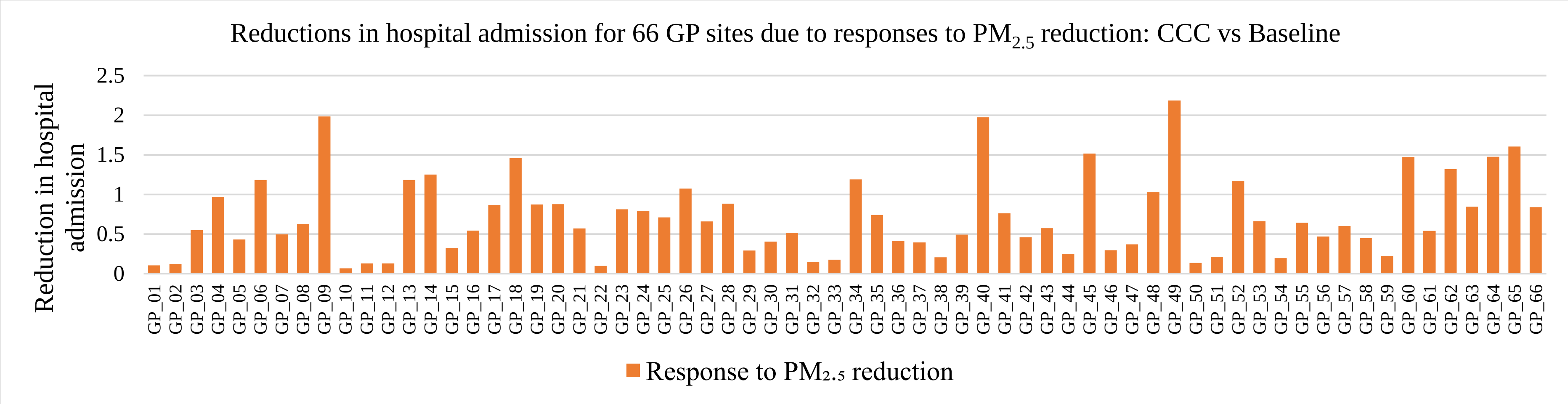
Reduced premature mortality, Baseline vs E-Car_E-LGV
Number of reduced premature mortality
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reuction

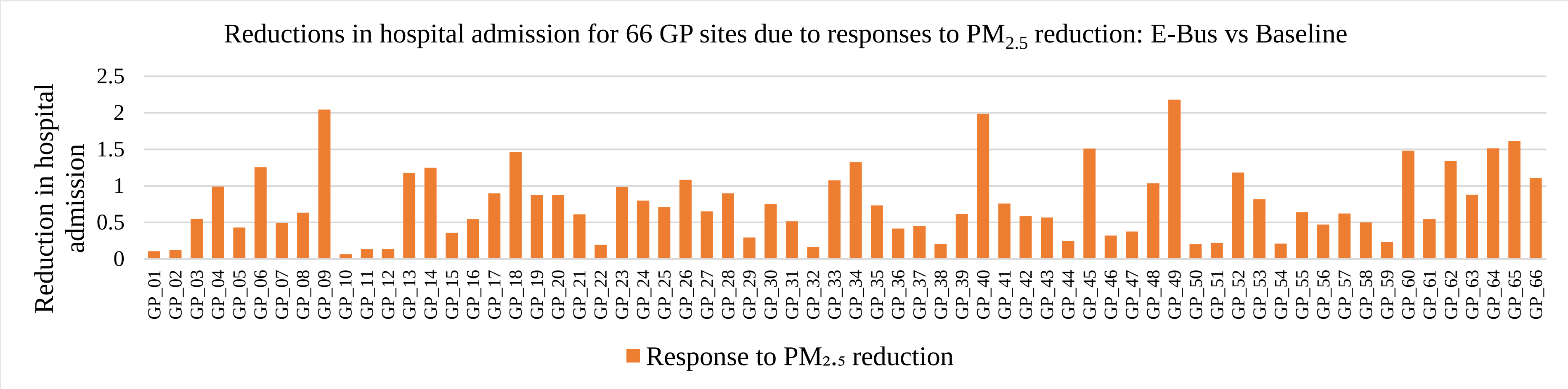


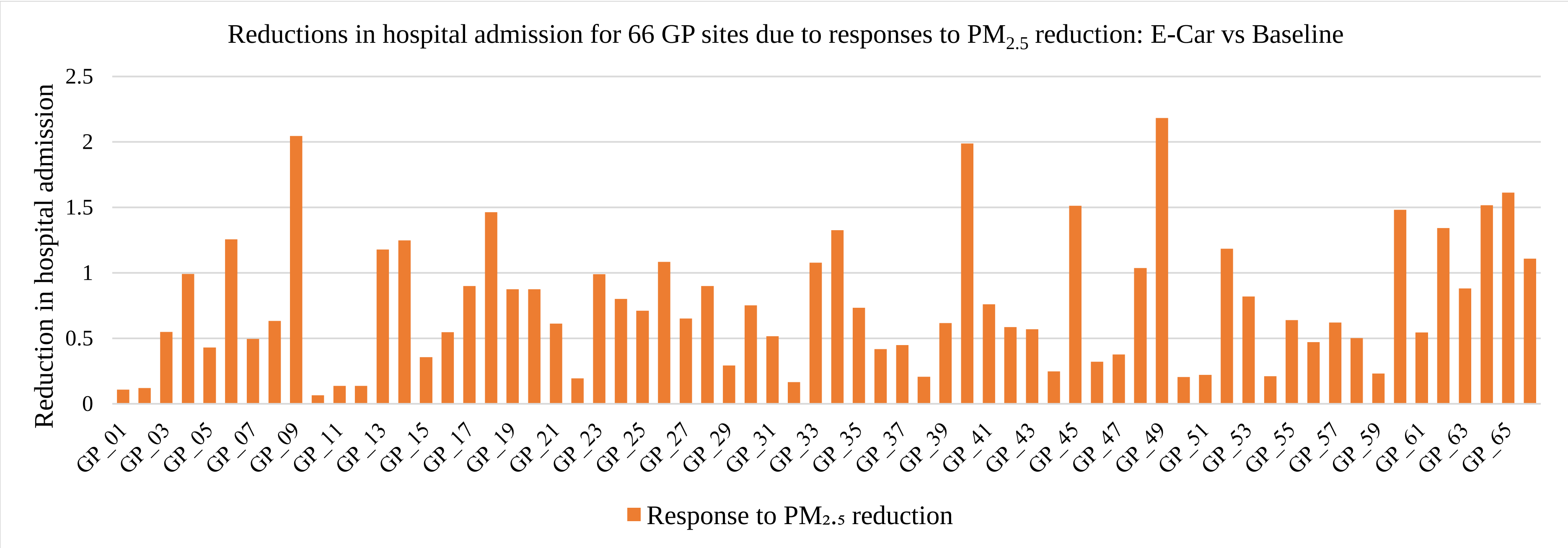
Reduced premature mortality, Baseline vs All-EV
Number of reduced premature mortality
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
GP_1 GP_2 GP_3 GP_4 GP_5 GP_6 GP_7 GP_8 GP_9 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reduction

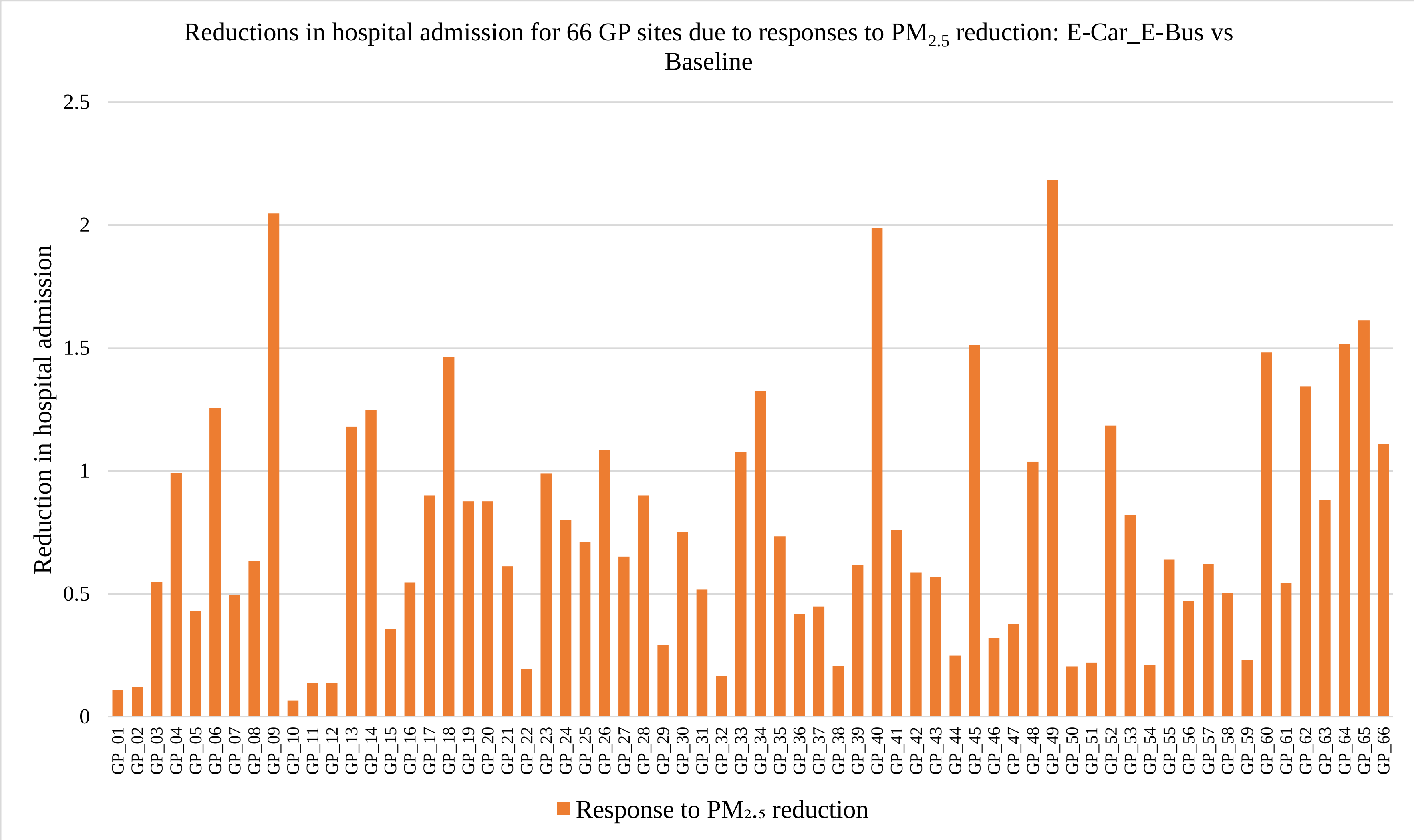
Appendix D: Reduced hospital admissions due to reduction in short-term exposure to NO_2, PM_{10} and $PM_{2.5}$ levels, Baseline vs all 2030 scenarios



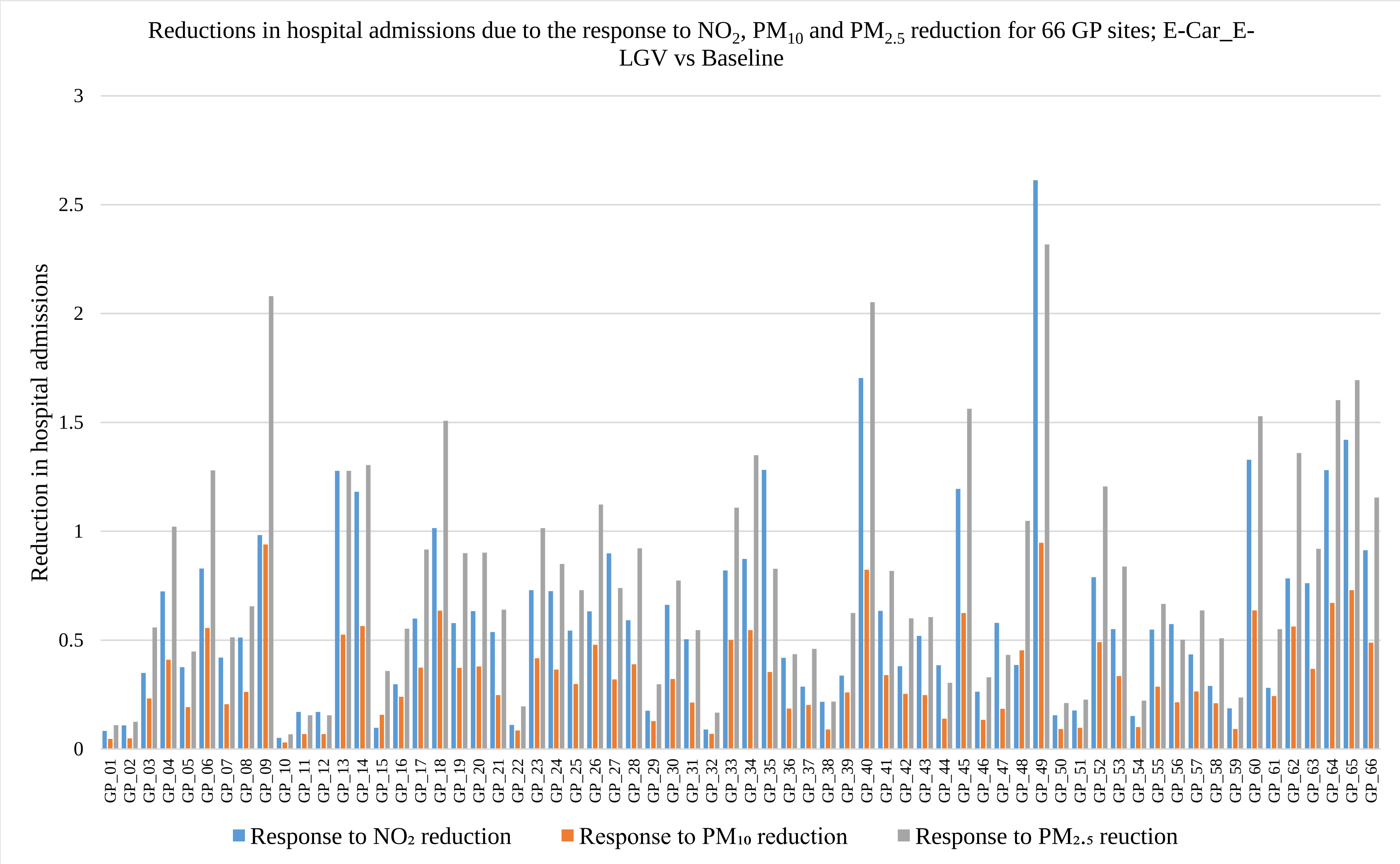



Reductions in hospital admission for 66 GP sites due to responses to PM2.5 reduction: E-Bus vs Baseline
Reduction in hospital admission
2.5
2
1.5
1
0.5
0
GP_01 … GP_66
Response to PM2.5 reduction

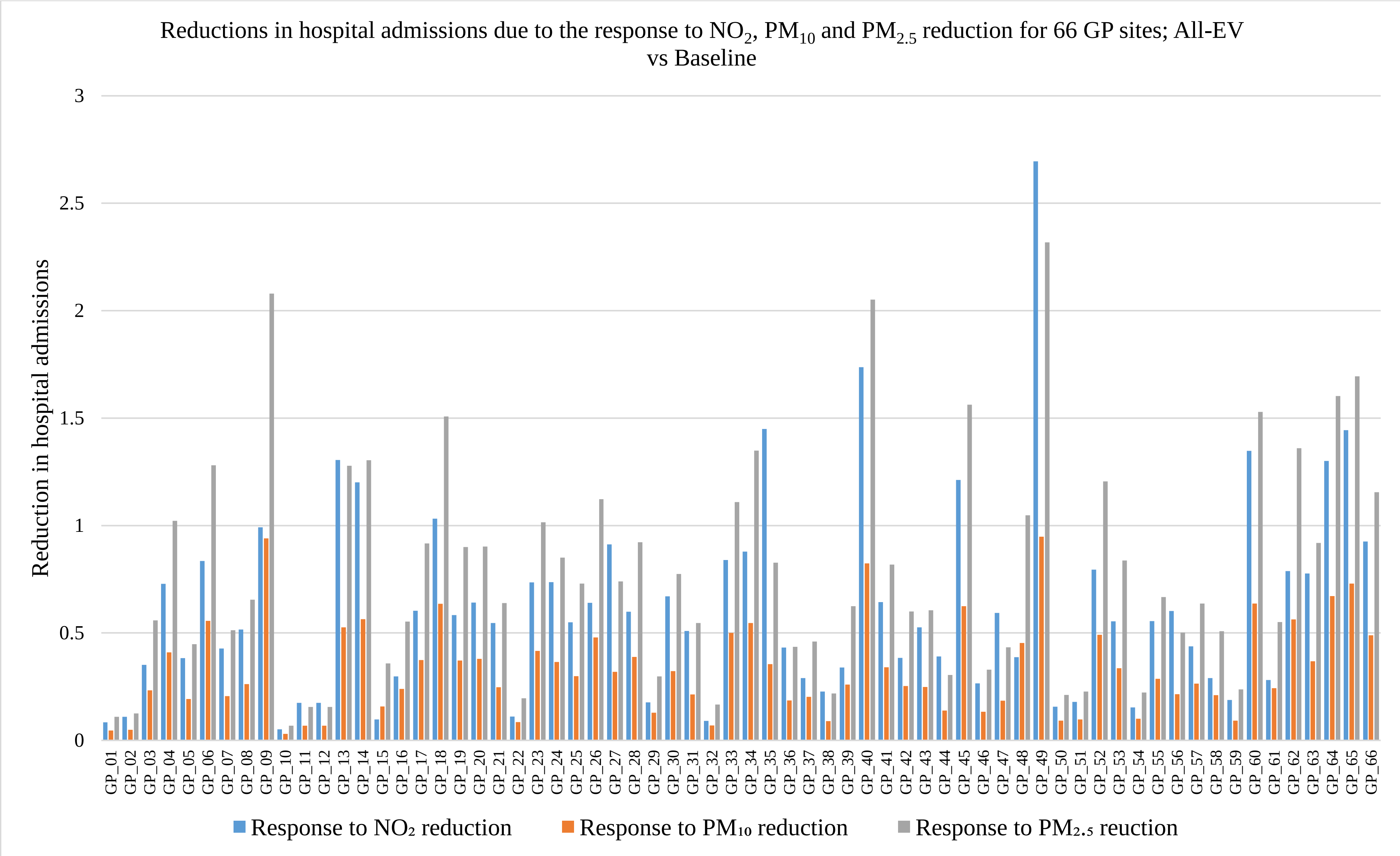

Reductions in hospital admission for 66 GP sites due to responses to PM2.5 reduction: E-Car vs Baseline
Reduction in hospital admission
2.5
2
1.5
1
0.5
0
GP_01 … GP_65
Response to PM2.5 reduction



Reductions in hospital admission for 66 GP sites due to responses to PM2.5 reduction: E-Car_E-Bus vs Baseline
Reduction in hospital admission
2.5
2
1.5
1
0.5
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to PM2.5 reduction

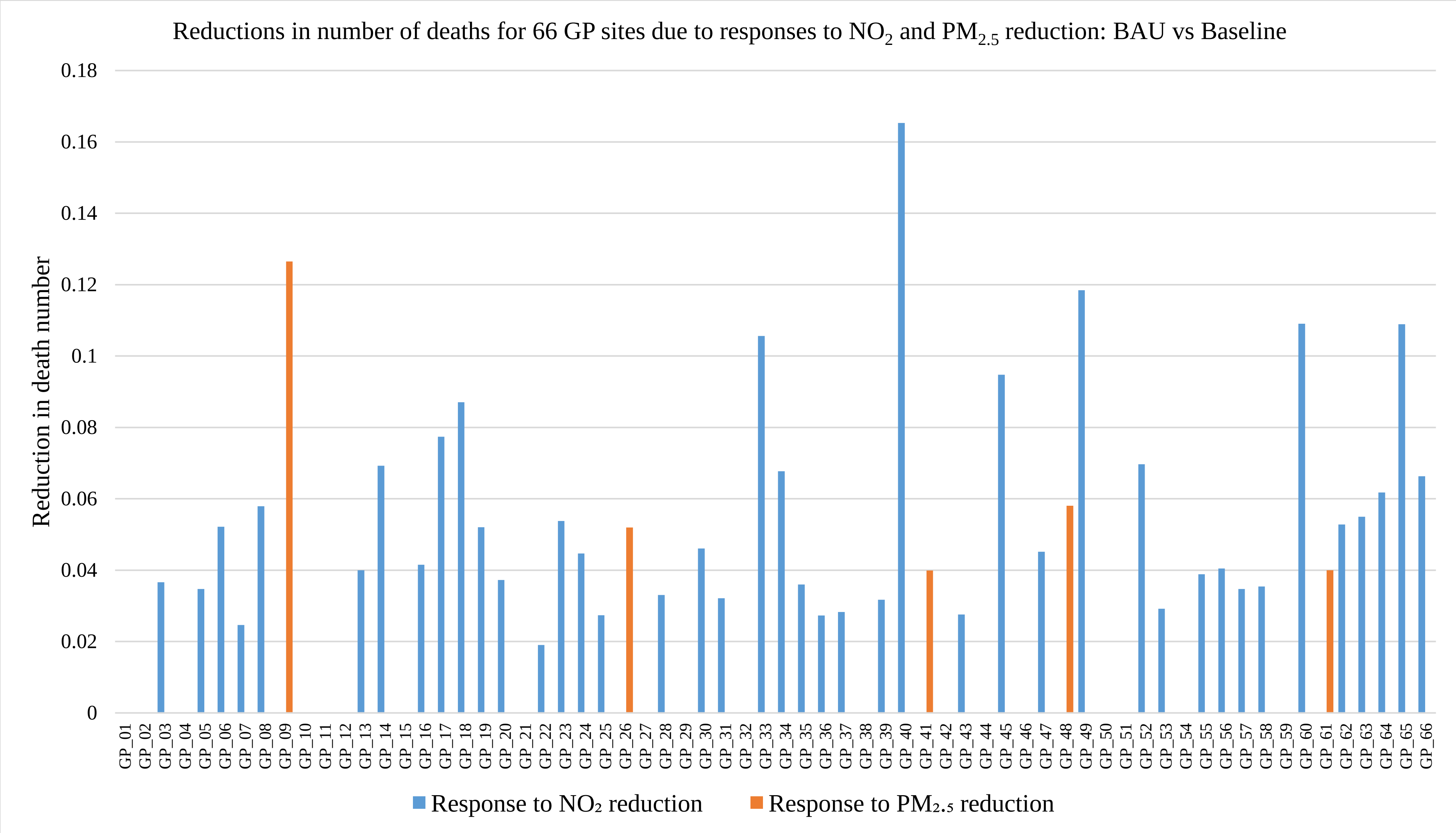


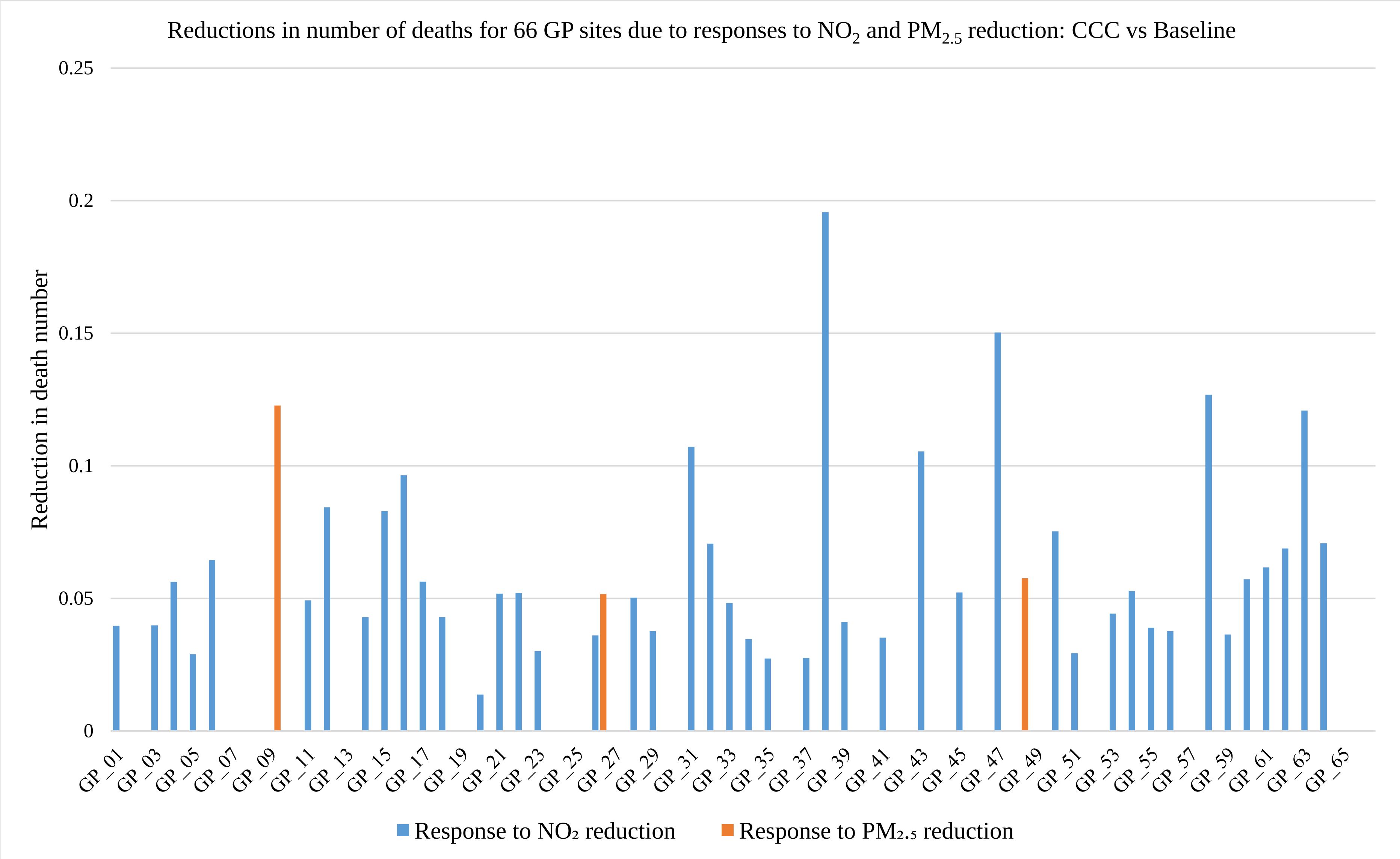
Reductions in hospital admissions due to the response to NO_2, PM_{10} and $PM_{2.5}$ reduction for 66 GP sites; E-Car_E-LGV vs Baseline
Reduction in hospital admissions
3
2.5
2
1.5
1
0.5
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reuction

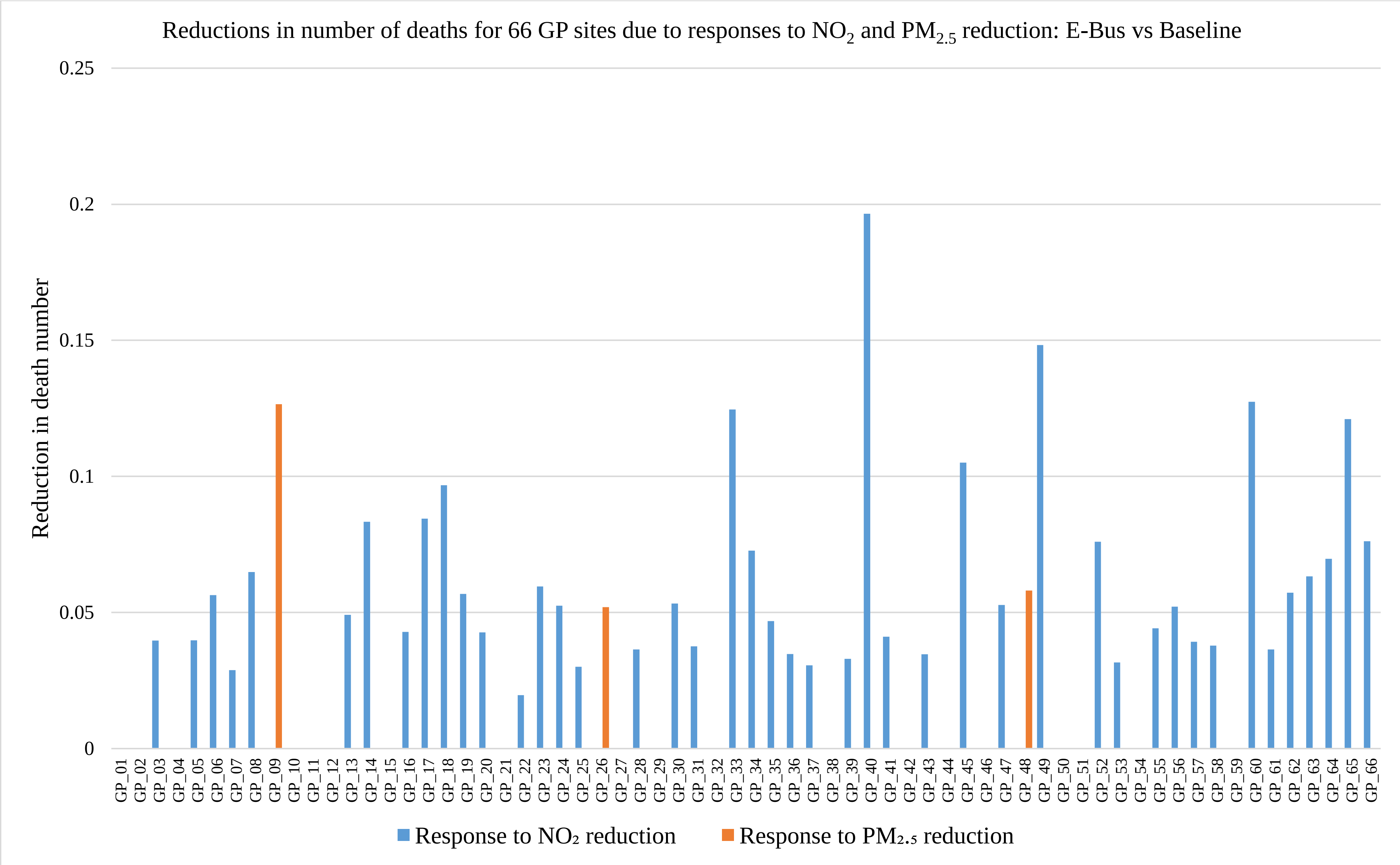


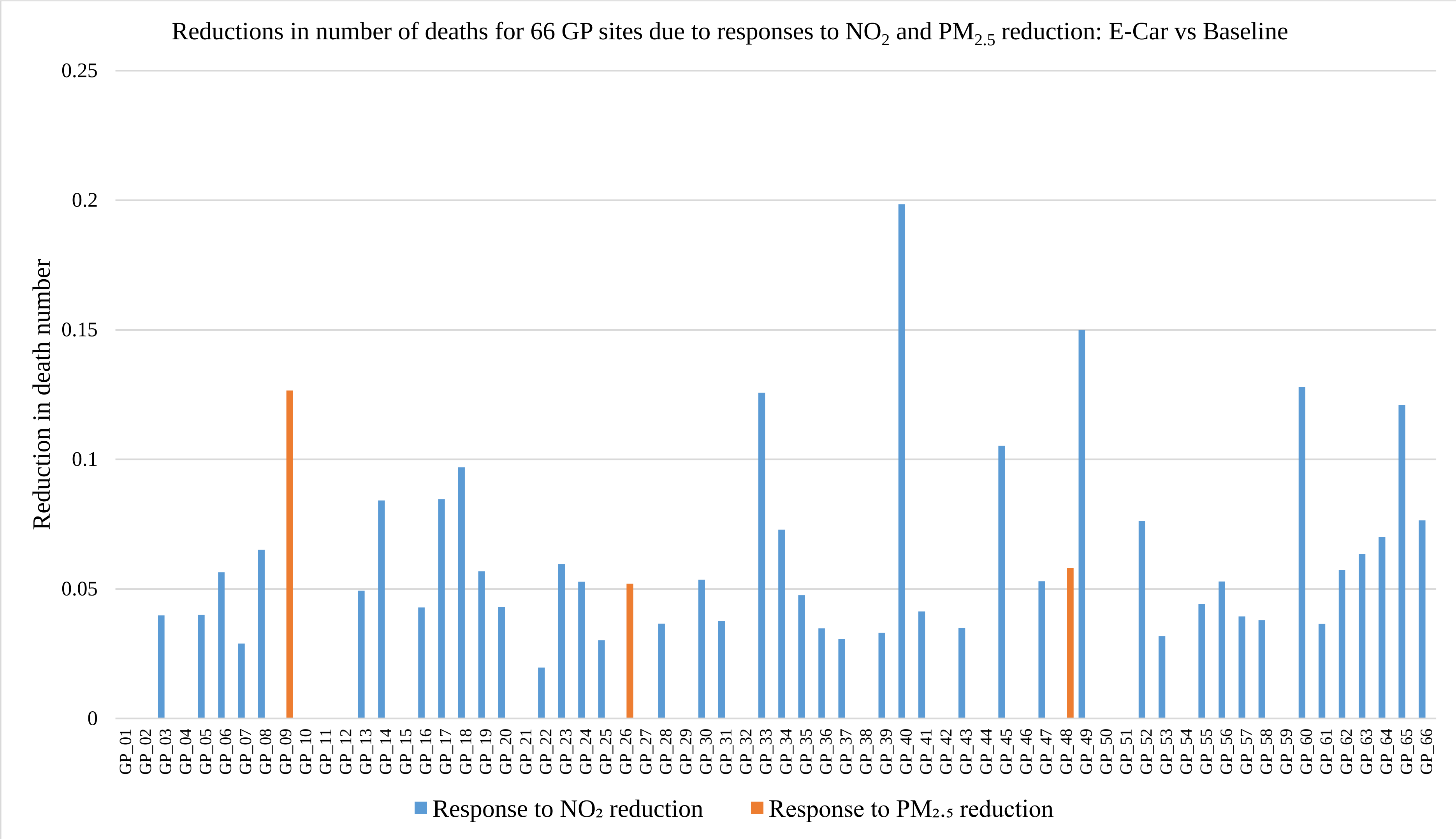
Reductions in hospital admissions due to the response to NO2, PM10 and PM2.5 reduction for 66 GP sites; All-EV vs Baseline
Reduction in hospital admissions
0
0.5
1
1.5
2
2.5
3
GP_01 … GP_66
Response to NO2 reduction
Response to PM10 reduction
Response to PM2.5 reuction

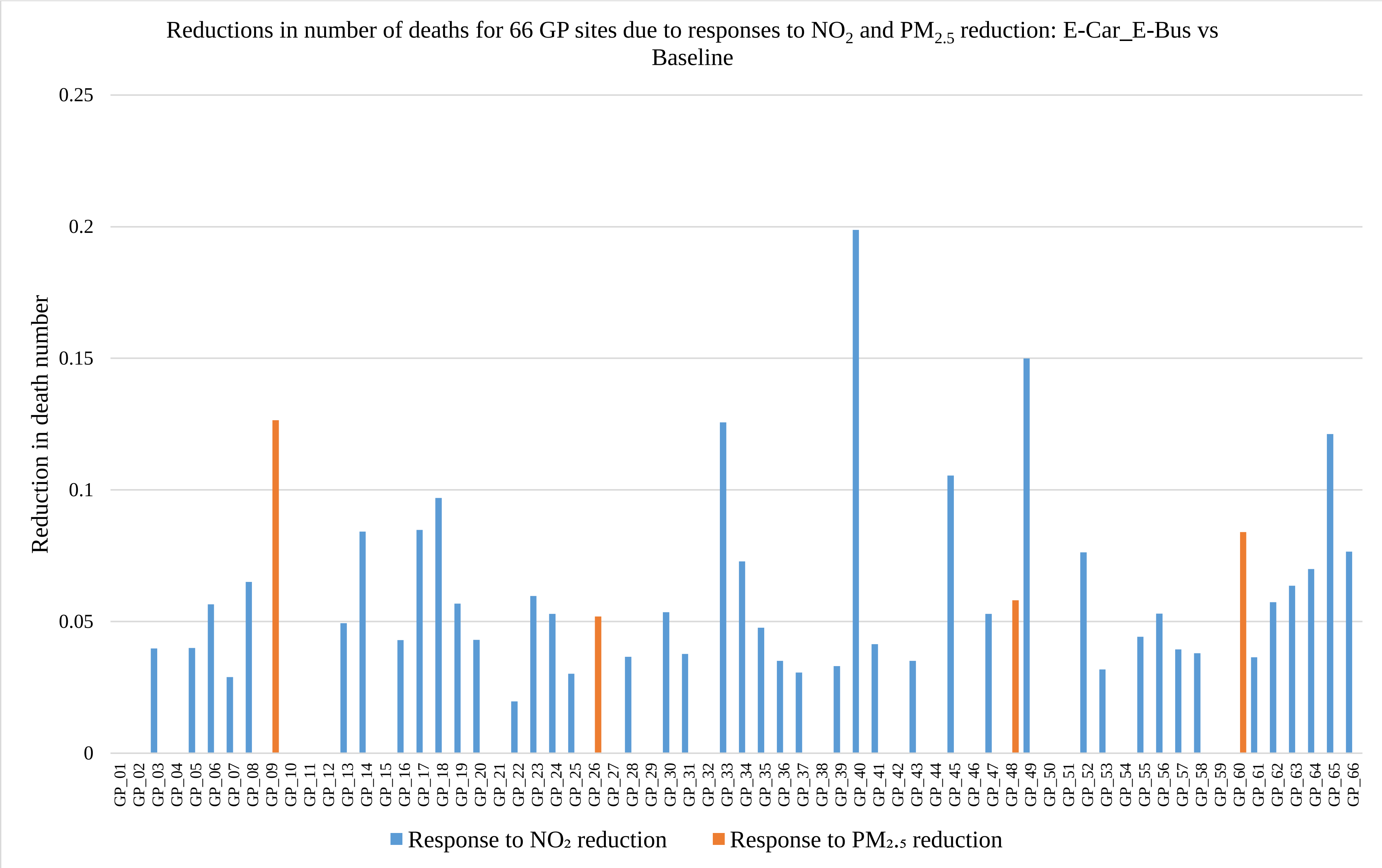
Appendix E: Reduced mortality due to reduction in short-term response to NO_2, PM_{10} and $PM_{2.5}$ levels, Baseline vs all 2030 scenarios

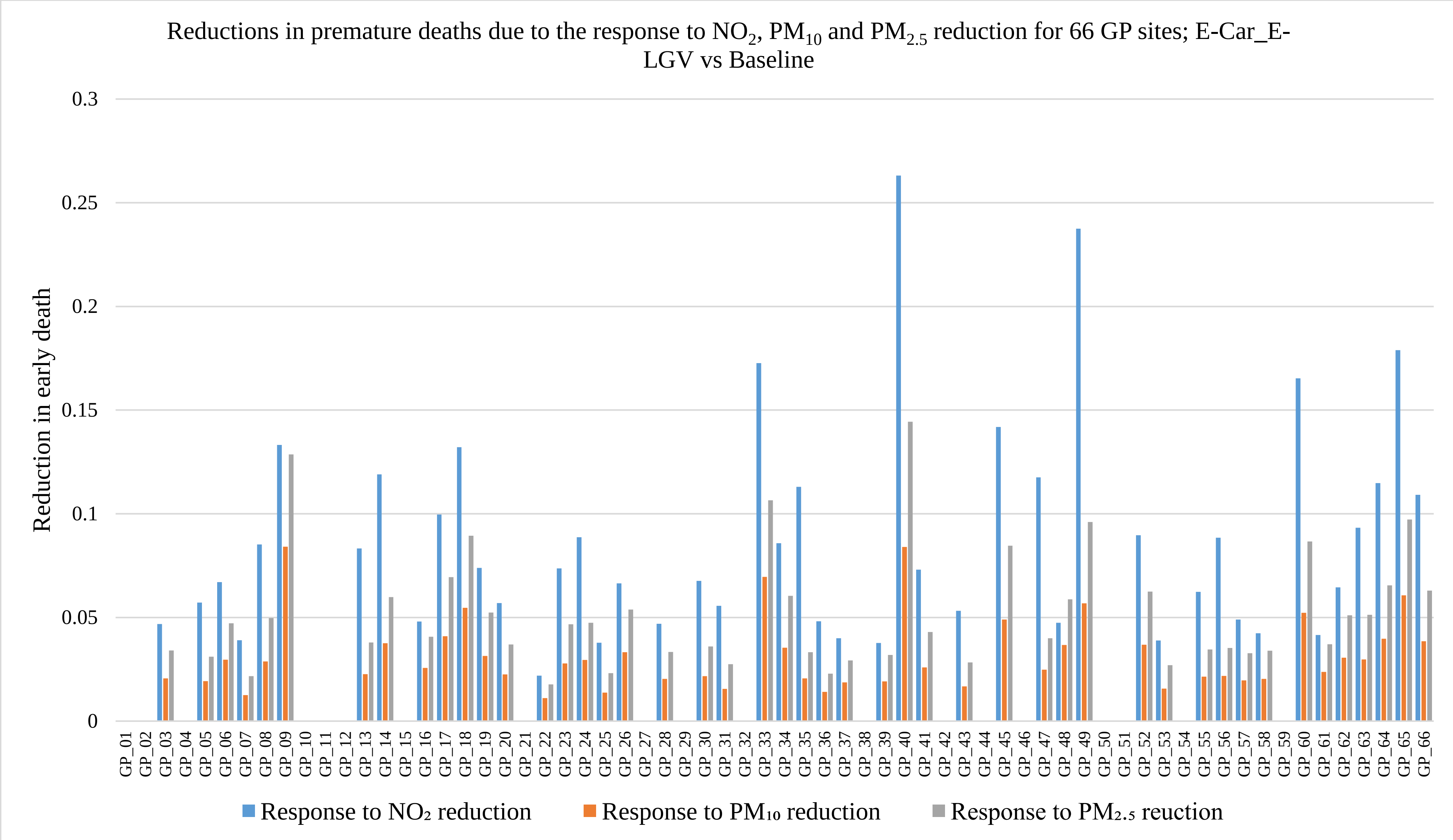


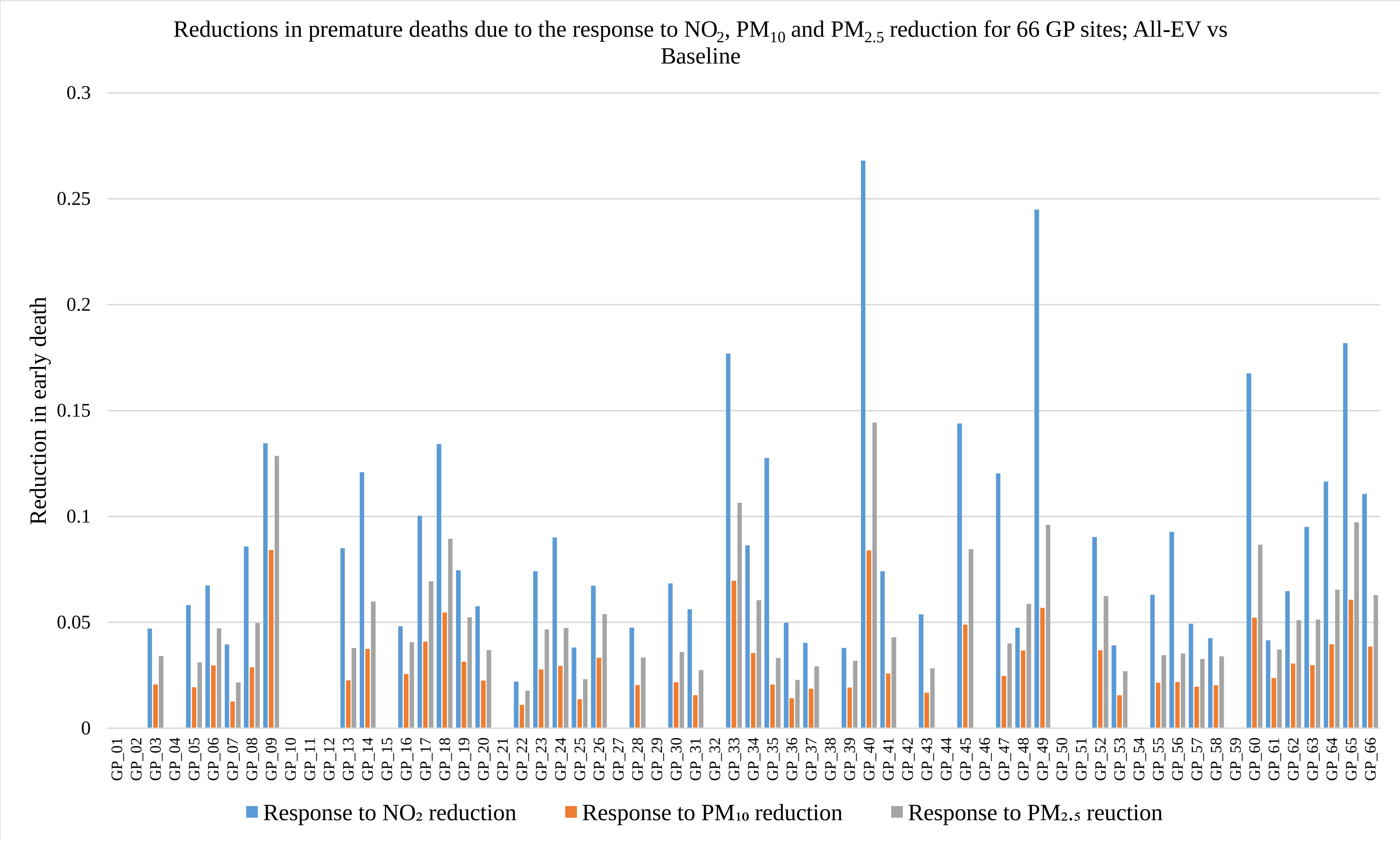

Reductions in number of deaths for 66 GP sites due to responses to NO2 and PM2.5 reduction: CCC vs Baseline
Reduction in death number
0.25
0.2
0.15
0.1
0.05
0
GP_01
GP_03
GP_05
GP_07
GP_09
GP_11
GP_13
GP_15
GP_17
GP_19
GP_21
GP_23
GP_25
GP_27
GP_29
GP_31
GP_33
GP_35
GP_37
GP_39
GP_41
GP_43
GP_45
GP_47
GP_49
GP_51
GP_53
GP_55
GP_57
GP_59
GP_61
GP_63
GP_65
Response to NO2 reduction
Response to PM2.5 reduction


Reductions in number of deaths for 66 GP sites due to responses to NO2 and PM2.5 reduction: E-Bus vs Baseline
Reduction in death number
0.25
0.2
0.15
0.1
0.05
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO2 reduction
Response to PM2.5 reduction


Reductions in number of deaths for 66 GP sites due to responses to NO_2 and $PM_{2.5}$ reduction: E-Car vs Baseline
Reduction in death number
0.25
0.2
0.15
0.1
0.05
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to $PM_{2.5}$ reduction


Reductions in number of deaths for 66 GP sites due to responses to NO_2 and $PM_{2.5}$ reduction: E-Car_E-Bus vs Baseline
Reduction in death number
0.25
0.2
0.15
0.1
0.05
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to $PM_{2.5}$ reduction


Reductions in premature deaths due to the response to NO_2, PM_{10} and $PM_{2.5}$ reduction for 66 GP sites; E-Car_E-LGV vs Baseline
Reduction in early death
0.3
0.25
0.2
0.15
0.1
0.05
0
GP_01 … GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reuction


Reductions in premature deaths due to the response to NO_2, PM_{10} and $PM_{2.5}$ reduction for 66 GP sites; All-EV vs Baseline
Reduction in early death
0.3
0.25
0.2
0.15
0.1
0.05
0
GP_01 GP_02 GP_03 GP_04 GP_05 GP_06 GP_07 GP_08 GP_09 GP_10 GP_11 GP_12 GP_13 GP_14 GP_15 GP_16 GP_17 GP_18 GP_19 GP_20 GP_21 GP_22 GP_23 GP_24 GP_25 GP_26 GP_27 GP_28 GP_29 GP_30 GP_31 GP_32 GP_33 GP_34 GP_35 GP_36 GP_37 GP_38 GP_39 GP_40 GP_41 GP_42 GP_43 GP_44 GP_45 GP_46 GP_47 GP_48 GP_49 GP_50 GP_51 GP_52 GP_53 GP_54 GP_55 GP_56 GP_57 GP_58 GP_59 GP_60 GP_61 GP_62 GP_63 GP_64 GP_65 GP_66
Response to NO_2 reduction
Response to PM_{10} reduction
Response to $PM_{2.5}$ reuction