Modelling of Vehicular Emissions and their Potential Environmental, Health, and Economic Impacts

A dissertation submitted to the University of Dublin, Trinity College in the partial fulfilment of the requirements for the degree Doctor of Philosophy

by

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Declaration

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Abstract

Road traffic is one of the greatest contributors to greenhouse gas and reducing it has become one of the main targets for sustainable transport policies. Analysis of the main factors influencing greenhouse gas emissions is essential for designing environmentally efficient strategies for road transport. On the other hand, air pollution has been associated with a series of health effects such as stroke, lung cancer, chronic and acute respiratory diseases, including asthma. Diesel vehicles are one of the major sources of two harmful air pollutants, $PM_{2.5}$ and NOx. Reducing NOx emission levels and diesel use has become a prime concern since it was discovered that most of the diesel Euro 5 and Euro 6 passenger cars and light commercial vehicles are violating the NOx emission standards. The electrification of the transportation sector has become necessary considering the growth in transport demand, resulting greenhouse gas, urban air pollution and fossil fuel depletion. A considerable percentage of EU urban population is exposed to harmful levels of air pollution concentration. Ireland has a target to reduce the GHG emission levels by 30% in 2030 relative to 2005 levels, however, GHG levels are expected to increase. Moreover, the NOx emissions in Ireland have failed to meet the national emission ceiling target leading to serious health concerns. However, high NOx emission levels may or may not lead to high NOx concentration at the population exposure level.

This thesis comprehensively evaluates the potential environmental, financial and health impacts associated with the Irish road transport fleet and further, proposes a solution through a set of hypothetical policy introduction which can potentially mitigate the impacts and increase EV uptake and investigates the possible reduction in impacts from alternative traffic scenarios as a result of these policy interventions. Emission levels from road transport were estimated using COPERT 5 for the current and several future scenarios with the continuation of Business as Usual situation and from alternative scenarios. An uncertainty modelling of COPERT 5 was carried out through sensitivity analysis of COPERT input parameters. NOx and $PM_{2.5}$ pollution at the population exposure level attributable to road transport emissions were modelled using OSPM for the current and future scenarios. Additionally, health impacts in terms of premature mortality incidences and Disability Adjusted Life Years attributable to long term exposure to NOx and $PM_{2.5}$ were estimated using WHO's burden of disease approach. The findings of this study indicate that emission levels from the future fleets are expected to increase under BaU scenarios due to the increase in car ownership levels and low uptake of alternative fuel and technology options. However, the emission levels can potentially be reduced with alternative fleet options as a result of additional policy measures. The results also revealed that Ireland's air quality is good with respect to NOx and $PM_{2.5}$ pollution when compared to WHO and EU recommended safe limits. Discrepancies between the concentrations recorded at the monitoring locations and the modelled values were observed. Although the overall air quality was found to be fair, the spatial

variation of the pollution showed that some areas in Dublin city are exposed to significantly high levels of NOx and $PM_{2.5}$ concentrations. The pollution levels and resulting health outcomes and financial damages are expected to increase, but, can be improved with the proposed alternative scenarios. The findings of this research not only provide policy makers with a timely and useful evaluation of the potential impacts of road transport fleet in Ireland but also sets an example of how controlling air pollution can be prioritised, compared to other policy interventions aimed at improving air quality and public health.

List of Publications

Journal Papers:

1. Dey, S., Caulfield, B., & Ghosh, B. 2018. The potential health, financial and environmental impacts of dieselgate in Ireland. Transportation Planning and Technology, 41:1, 17-36, DOI: 10.1080/03081060.2018.1402743
2. Dey, S., Caulfield, B., & Ghosh, B. 2018. Potential health and economic benefits of banning diesel traffic in Dublin, Ireland. Journal of Transport & Health, 10, 156-166, DOI: 10.1016/j.jth.2018.04.006
3. Dey, S., Caulfield, B., & Ghosh, B. 2018. Modelling Uncertainty of Vehicular Emissions Inventory: A Case Study of Ireland. Journal of Cleaner Production, 213, 1115-1126. DOI: 10.1016/j.jclepro.2018.12.125

Conferences:

1. Dey, S., Caulfield, B., & Ghosh, B. 2017. The Potential Environmental Impacts of *dieselgate* in Ireland. 49th Universities' Transport Study Group (UTSG) conference, Dublin, Ireland, January 4-6.
2. Dey, S., Caulfield, B., & Ghosh, B. 2017. Examining alternative fuel options and potential emission reductions from changes in public transport bus fleet in Ireland. 8th Irish Transportation Research Network (ITRN) conference, Dublin, Ireland, August 28-29.
3. Dey, S., Caulfield, B., & Ghosh, B. 2018. Quantification of Diesel Burden and the Potential Impact of Diesel-Ban in Dublin, Ireland. 97th Transportation Research Board (TRB) conference, Washington DC, January 7-11.
4. Dey, S., Caulfield, B., & Ghosh, B. 2018. Modelling NOx concentrations and emissions from road transport at street level in Dublin City. Civil Engineering Research in Ireland (CERI) conference, Dublin, Ireland, August 29-30.

List of Acronyms

AADT	Annual Average Daily Traffic
AAM	Annual Average Mileage
BaU	Business as Usual
BEV	Battery Electric Vehicle
BOD	Burden of Disease
C_0	Counterfactual concentration
CH_4	Methane
CNG	Compressed Natural Gas
CO	Carbon monoxide
CO_2	Carbon dioxide
COPERT	COmputer Programme to calculate Emissions from Road Transport
CSO	Central Statistics Office
DALY	Disability Adjusted Life Years
DTTaS	Department of Transport, Tourism and Sports
ED	Electoral District
EEA	European Environment Agency
EEV	Enhanced Environmentally friendly Vehicle
EF	Emission Factors
EPA	Environmental Protection Agency
ERMES	European Research for Mobile Emission Sources
EU	European Union
EV	Electric Vehicles
GDA	Greater Dublin Area
GHG	Greenhouse Gas
GIS	Geographic Information System
HDV	Heavy Duty Vehicles
HEV	Hybrid Electric Vehicle
ICEV	Internal Combustion Engine Vehicle
Km	Kilometre
Kmph	Kilometre per hour
kWh	Kilowatt
K-S	Kolmogorov-Smirnov
LDV	Light Duty Vehicles
LPG	Liquefied Petroleum Gas
MS	Mileage Share

N_2O	Nitrous oxide
NEI	National Emissions Inventory
NMVOC	Non-Methane Volatile Organic Compounds
NO	Nitric oxide
NO_2	Nitrogen dioxide
non-GHG	non-Greenhouse Gas
NOx	Oxides of nitrogen
O_3	Ozone
OSPM	Operational Street Pollution Model
PC	Passenger Car
PCU	Passenger Car Unit
PDF	Probability Density Functions
PEMS	Portable Emission Measurement Systems
PHEV	Plug-in Hybrid Vehicle
PM	Particulate Matter
PM_{10}	Particulate Matter with diameter of less than 10 micrometres
$PM_{2.5}$	Particulate Matter with diameter of less than 2.5 micrometres
RH	Relative Humidity
RR	Relative Risk
SEAI	Sustainable Energy Authority of Ireland
SO_2	Sulphur Dioxide
SOx	Oxides of Sulphur
TJ	Terajoule
TL	Trip Length
TTW	Tank-to-Wheel
US EPA	United States Environmental protection Agency
VOC	Volatile Organic Compounds
VRT	Vehicle Registration Tax
VSL	Value of Statistical Life
VW	Volkswagen
WHO	World Health Organisation
WTT	Well-to-Tank
WTW	Well-to-Wheel
YLD	Years of healthy Life lost due to Disability
YLL	Years of Life Lost

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Chapter 1: Introduction

1.1. Background

1.1.1. Air pollution and road transport sector as a source of air pollution

Air pollution is both a social and an environmental problem, as it leads to a huge number of adverse effects on the climate, ecosystem, the built environment and human health (European Environment Agency (EEA), 2016a). Air pollution is the single largest environmental health risk in Europe at this moment (EEA, 2016a) and reducing pollution levels is one of the prime concerns. Greenhouse gases (GHGs) such as carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) trap heat in the atmosphere. As part of the Earth's carbon cycle, CO_2 is present in the atmosphere, however, due to human activities, the carbon cycle is being altered. CO_2 is the primary GHG emitted through human activities (United States Environmental protection Agency (US EPA), 2018a). Other air pollutants, often referred to as non-greenhouse gas (non-GHG), include carbon monoxide (CO), oxides of nitrogen (NOx), Particulate Matter with diameter of less than 2.5 micrometres ($PM_{2.5}$), Particulate Matter with diameter of less than 10 micrometres (PM_{10}), Volatile Organic Compounds (VOC), and Non-Methane Volatile Organic Compounds (NMVOC). These pollutants are also naturally present in the atmosphere in small amount but may cause harm to human health and eco-system if the levels are higher than specified safe limits for a certain period. There are six key air pollutants known as "criteria air pollutants" that cause harm to human health and the environment, they are ground-level ozone (O_3), Particulate Matter (PM), NOx, CO, Sulphur Dioxide (SO_2) and lead (US EPA, 2018b).

Transportation is one of the largest sources of both GHG and non-GHG emissions. In Europe, GHG emissions share from the transport sector is 19.5% of total GHG emissions in 2014 as shown in Figure 1.1 (EEA, 2018a). Road transport constitutes the major portion of the emissions from the overall transportation sector. Moreover, due to the booming transport demand, reducing emissions from road transport has attained more attention over the past few years. Several targets are now being set and policies are being implemented aiming to decrease fossil fuel use and increase the uptake of low or zero emissions transport modes, and thereby, reduce emission levels.

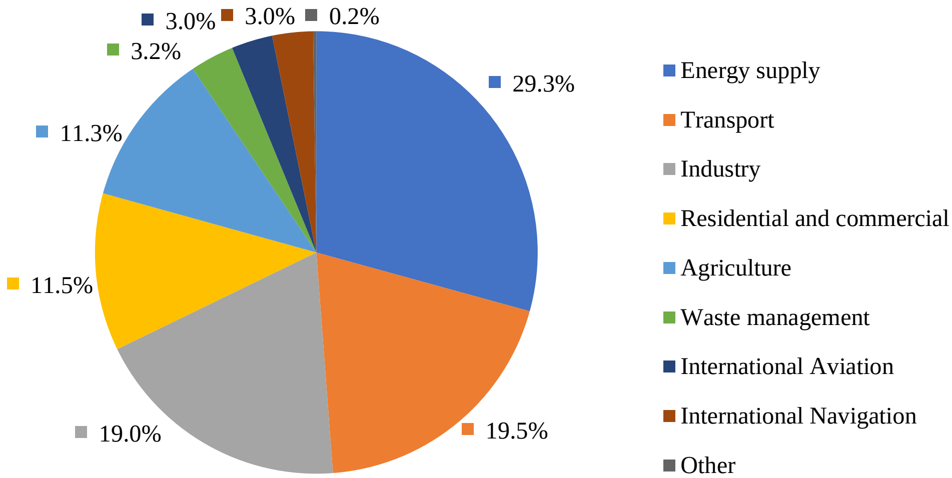

Figure 1.1. Sectoral shares of GHG emissions in Europe (EEA, 2018a)

The main pollutants discharged from road transport are CO_2, CO, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC (EEA, 2017). Figure 1.2 illustrates the sectoral shares of CO, NMVOC, NOx, PM_{10}, $PM_{2.5}$ and SOx (Oxides of Sulphur) in Europe in 2015 (EEA, 2017). While GHG, like CO_2, is responsible for global warming and climate change, non-GHGs like CO, NOx, VOC, NMVOC, PM directly or indirectly affect human health.

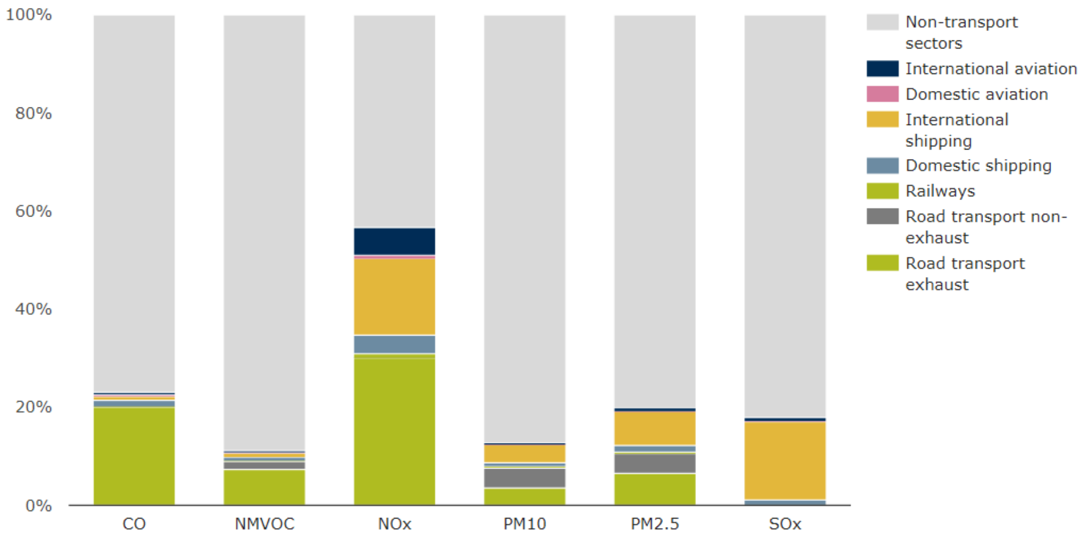

Figure 1.2. Sectoral contribution to emissions of the main air pollutants (EEA, 2017)

Air pollution has been associated with several health effects such as stroke, lung cancer, chronic and acute respiratory diseases, including asthma (World Health Organisation (WHO), 2016a). The health issues resulting from exposure to air pollution also have a high cost to society and business, health services and the people who suffer from related illnesses and premature deaths (RCP/RCPCH, 2016). Emissions not only possess health consequences but also cause damage to materials, crops, biodiversity (Korzhenevych et al., 2014) which have associated damage costs.

NOx which are comprised of NO_2 (nitrogen dioxide) and NO (nitric oxide), are key air pollutants which contributes to atmospheric levels of NOx, acid rain, fine particulate matter, and ground-level O_3 (US EPA, 2016a). 85-91% of European Union (EU) urban population is exposed to harmful levels of air pollution concentration (WHO, 2013a). NO_2 has severe health effects including, lung function decrements, inflammation, and permeability, susceptibility to infection, cardiac effects etc. (US EPA, 2016b). Moreover, in *dieselgate* it was found that certain diesel Euro 5 PC (Passenger Car) models are cheating the emission test with the help of a defeat device fitted with a software which turns the full emissions controls for NOx on only during the test and at other times the 2.0 litre and 3.0 litre engines emit NOx up to 40 times and up to nine times the standard, respectively (US EPA, 2016c). The Euro 5 and Euro 6 engines were originally planned to reduce the ambient NO_X concentration due to better vehicular technologies producing less NO_X emission than their predecessors which may not be the reality. Also, it has been suspected that a total of 21.4 million Euro 5 PCs and 2.2 million Euro 5 Light Commercial Vehicles (LCVs) across Europe are faulty (Transport and Environment, 2016). Diesel vehicles are also a source of another toxic air pollutant, $PM_{2.5}$. Reducing NOx and $PM_{2.5}$ levels and resulting health effects have become the prime concern in Europe at this moment (WHO, 2016b).

1.1.2. Road transport emissions in Ireland and emissions targets

In Ireland, 19.8% of overall GHG emissions result from the transportation sector, out of which road transport contributed to 18.9% in 2015 (Environmental Protection Agency (EPA), 2017), as shown in Figure 1.3. The GHG emissions from the transport sector have increased by 130.3% and from road transport alone by 136.7%, compared to 1990 emission levels (EPA, 2017a).

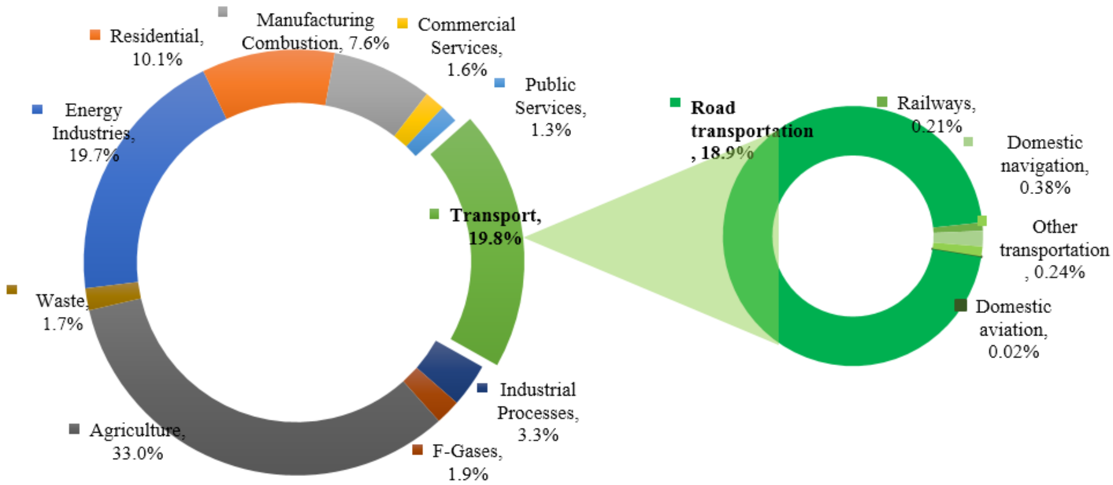

Figure 1.3. Sectoral GHG emissions shares in Ireland in 2015

In 2008, it was announced that of 10% of the private car fleet (approximately 230,000 vehicles) in Ireland to be electric by 2020 (Sustainable Energy Authority of Ireland (SEAI), 2014). In addition to having 10% Electric Vehicles (EVs) in the PC fleet in 2020, Ireland's target is to derive 16% of the final energy use and 10% of the transport energy use from renewable sources (SEAI, 2016). Ireland has an overall target to reduce the GHG by 30% by 2030 relative to 2005 levels (European Commission, 2018). However, transport emissions are projected to show a strong growth over the period 2015-2020 resulting in 10% to 12% increase in GHG emissions compared to 2015 levels (EPA, 2017b). This reflects the strong economic growth forecasted over the upcoming period (EPA, 2017b). The 12% increase is estimated based on the measures which are already in place (i.e. with existing measures), such as, Vehicle Registration Tax (VRT) and motor tax (introduced in 2008), carbon tax imposed on fuels since 2010, improvements to the fuel economy of the private cars etc. With the additional measures, which includes 8% of the transport energy demand to be drawn from renewables by 2020 and 10,000 EV deployments by 2020, emissions from the transportation sector are expected to increase by 10% to 13 megatonne eCO_2. Figure 1.4 shows the trend of GHG emissions from road transport over 1990 and the current year along with the projected levels until 2035 (EPA, 2017b).

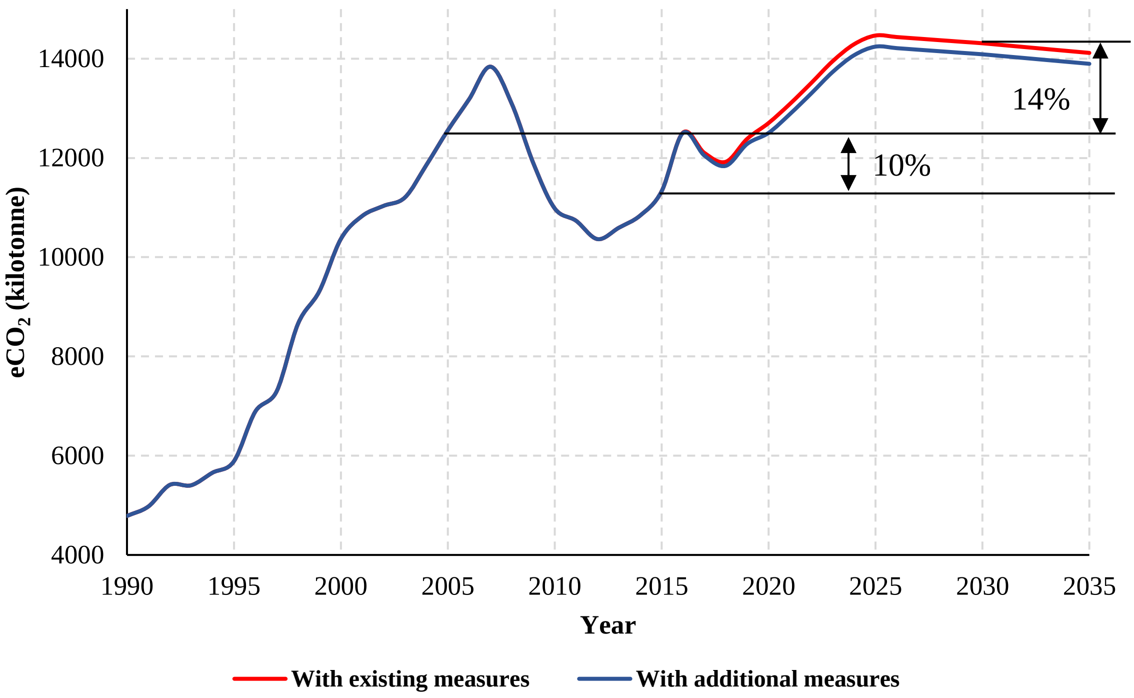

Figure 1.4. GHG emissions trend from road transport over 1990-2035

GHG emissions from road transport alone are expected to increase by 14% in 2030 relative to 2005 levels with the existing measures (Figure 1.4). It can also be observed in Figure 1.4 that GHG emission levels in 2015 have reduced by 10% with respect to GHG emissions in 2005. The graph also indicates an increasing trend till 2025 and then decreases at a slower rate. Despite there being several incentives such as the grant of up to €5,000 for every Battery Electric Vehicles

(BEVs) or Plug-In Hybrid Vehicles (PHEVs) purchase, VRT reliefs of €5,000 for BEV and €2,500 for PHEV, annual motor tax for EV is €120 (SEAI, 2017a), EV uptake is significantly low in Ireland. Therefore, the existing policies should be examined, and new policies should be implemented to mitigate the Internal Combustion Engine Vehicle (ICEV) use and increase the EV uptake and thereby, reduce both the GHG and toxic non-GHG emissions. Also, it is essential to determine how different fleet compositions can lead to changes in emission levels in 2030 and how the 2030 emissions target can potentially be met. This will help in designing the policies in a more effective way.

Ireland has relatively good air quality compared to some other EU member countries and meets the EU specified guidelines (EPA, 2016). However, while ambient NOx levels are within the limit in Ireland at the monitoring locations, the NOx emission levels have failed to meet the National Emission Ceilings in 2010 and continue to be above the National Emission Ceilings target (EPA, 2016) as shown in Figure 1.5. Diesel engines tend to emit a higher percentage of NOx (EPA, 2015) and it is expected that the NOx emission levels in Ireland will increase due to a significant increase in the number of diesel vehicles in the fleet since 2008 and due to the *dieselgate*. This will have huge health and financial impact, especially in urban areas.

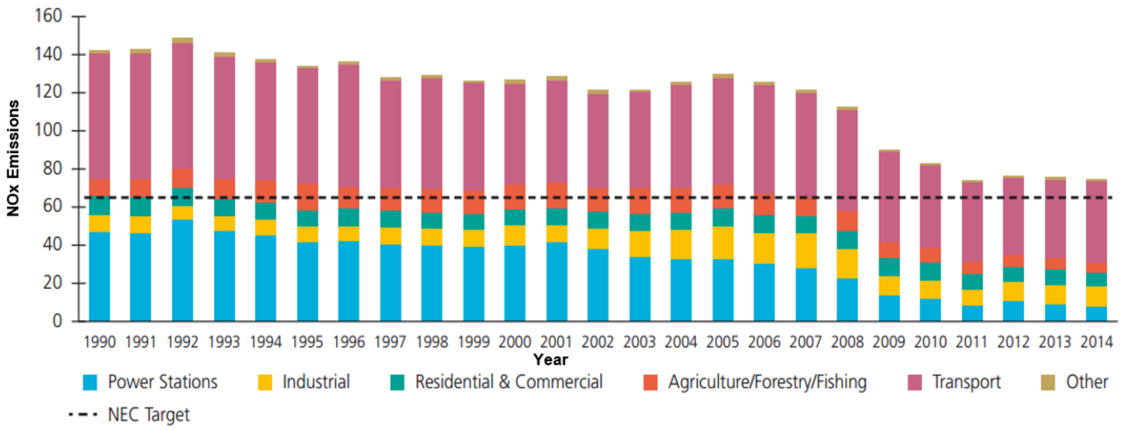

Figure 1.5. NOx emission sources and trends 1990-2014 (EPA, 2015)

Ireland has the highest share of newly registered diesel cars in Europe with the share of 70.06% new diesel cars in the fleet in 2016 (ACEA, 2018). The share in the overall fleet is expected to increase further with Business as Usual (BaU) scenario. The rise in the number of diesel passenger vehicles can be linked to the Irish Carbon-based vehicle tax system whereby vehicles purchased from 2008 onwards would be taxed based on their CO_2 emissions intensity rather than their engine capacity (the previous approach) (Giblin and McNabola, 2009). This resulted in a shift in new vehicle purchasing patterns from petrol to diesel. Figure 1.6 shows the new car registration pattern in Ireland over the period 2007-2015 (SIMI, 2017).

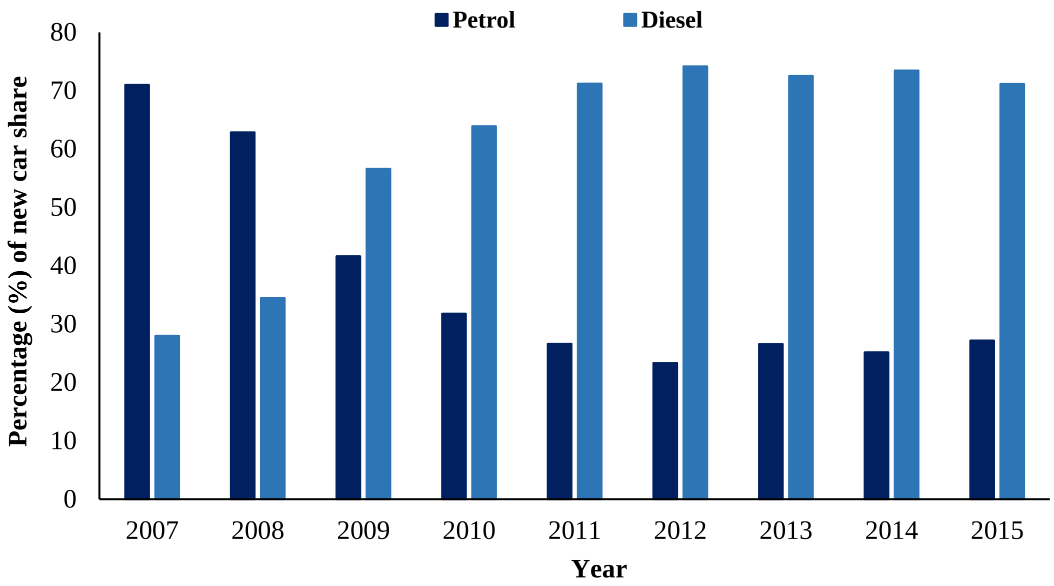

Figure 1.6. New car registration pattern over 2007-2015

The relationship between emission and concentration is still obscure as pollutant concentration is correlated with many other factors related to wind profile and building geometry. Also, it is not NOx in general, rather it is the NO_2 which is harmful to human health. It is essential to investigate if the high levels of NOx emissions lead to high levels of NO_2 concentrations beyond the WHO and EU specified safe limits. Also, the information on area specific population exposure level to NOx or NO_2 and resulting health impacts are not available in Ireland.

1.2. Research Problem Statement

Air pollution is a key concern at this moment and road transport is a major contributor to air pollution both globally and in Ireland. With increasing transport demand, increased diesel usage and lower uptake of low emission vehicles, these emission levels and the subsequent environmental, health and economic impacts are expected to increase in the future. Therefore, the aim of this research is to examine the present state of emission levels from road transport in Ireland and, then examine the causes of high emissions and evaluate a set of policies to mitigate the emission levels. Further, this research aims to investigate the urban air quality, and thereby, examine resulting environmental, health and cost impacts of two of the key air pollutants NOx and $PM_{2.5}$ generated from road transport and quantify the potential improvements in these impacts with new policy interventions.

1.3. Research Objectives

The specific objectives of this work are to,

- Examine the current state of emission levels from road transport and its economic impacts in Ireland, and possible uncertainty associated with the estimates.
- Evaluate expected emission levels from the future fleets and potential emissions reductions from changes in road transport fleets with increased uptake of alternative low emissions fuel and vehicle technology options and increased renewable energy use.
- Examine potential emissions reductions from changes in composition of road transport fleets with the alternative tax implementation and policy measures to feasibly mitigate road traffic emissions in Ireland.
- Assess environmental impacts of emissions from road transport in the present and future scenarios through dispersion modelling and understand how the high levels of NOx emissions lead to changes in NOx and NO_2 concentrations at population exposure level.
- Assess health impacts due to long-term exposure to $PM_{2.5}$ and NO_2 pollution attributable to road transport emissions.
- Evaluate financial impacts due to health outcomes from long-term exposure to $PM_{2.5}$ and NO_2 pollution attributable to road transport emissions.

1.4. Organisation of the thesis

The organisation of the thesis is detailed in this section. The thesis consists of nine chapters as shown in Figure 1.7.

Chapter 1 discusses the importance and motivation behind this research, describes the research gap, and introduces the objectives and key methodologies used to assess the objectives.

Chapter 2 provides a detailed review of literature including road transport emission estimation and dispersion modelling methods and presents a review of existing knowledge on emissions related to road transport, ways and policies to mitigate those emission levels, related health consequences and financial impacts.

Chapter 3 presents the methodology followed in this research to assess vehicular emissions and resulting environmental impacts in terms of pollutant concentrations at population exposure level, health impacts in terms of premature death incidences and DALYs, and economic impacts due to premature deaths and damage caused by pollutants. This chapter also details the scenarios designed to evaluate emissions and their impacts.

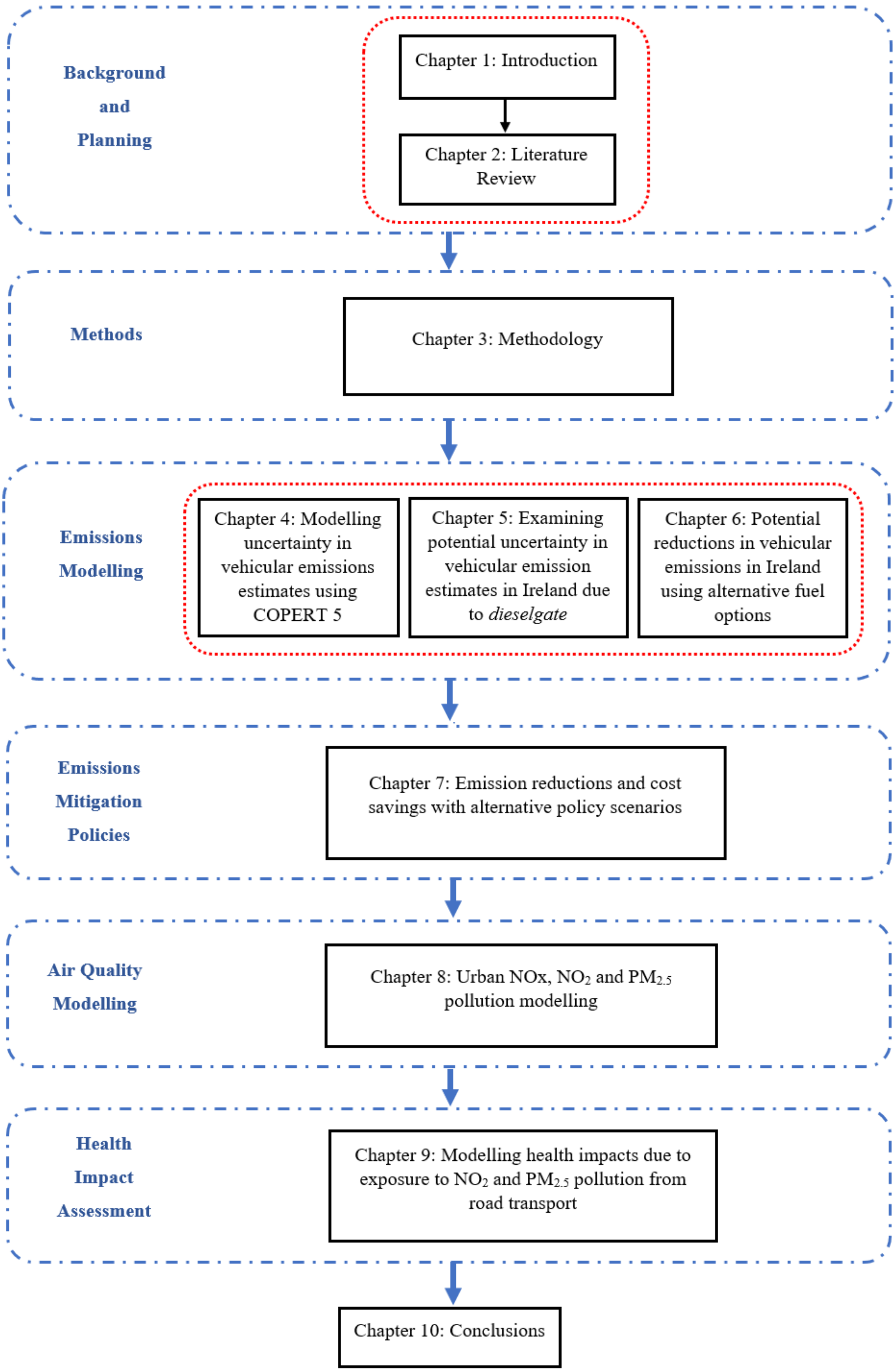

Figure 1.7. Thesis Outline

In addition, the data utilised to model the emissions and subsequent impacts are described in Chapter 3.

Chapter 4 presents the emission levels in 2015 which is considered as the baseline scenario for this research and possible uncertainty in emission estimates due to variations in input parameters of COPERT v.5.1.1, which was used in this research to model emissions from road transport.

Chapter 5 presents the potential uncertainty associated with emission estimations in Ireland due to certain diesel vehicles not obeying the emissions standards and its consequential influence on economic and health impacts.

Chapter 6 provides the expected emission levels from the predicted future Irish PC fleet and PT bus fleet under BaU. This chapter also provides potential emissions reductions from alternative fleet compositions with increased use of alternative fuel/technology options. Additionally, the fleet compositions that can possibly help Ireland in meeting its 2030 GHG emissions goal are also estimated by backcasting the emissions target of 2030 which is to reduce 30% emissions relative to 2005.

Chapter 7 presents the expected diesel emission levels over future years and how it can potentially be mitigated by introducing new policy measures aimed at reducing diesel use and increase low emission vehicles.

Chapter 8 provides street level emissions and concentration modelling results for NOx, NO_2 and $PM_{2.5}$ in 2015. This chapter also includes the increase in pollution levels of the mentioned pollutants in 2030 with BaU, i.e. continuation of the current situation with no additional policy implementation and possible reduction in pollution levels in 2030 with new policy measures.

Chapter 9 presents the health and cost impacts due to long-term exposure to modelled NO_2 and $PM_{2.5}$ levels. Potential increase in these impacts in 2030 under BaU and possible savings with additional policy measures are also presented in this chapter.

Chapter 10 concludes the thesis by summarising the main contributions of the thesis. This is followed by a discussion on the limitations of this research and, then suggestions on some directions for future research.

Next chapter discusses a detailed review of the existing studies in the field related to the objectives of this thesis.

Chapter 2: Literature Review

The research documented in this thesis lies at the intersection of several areas of research and, builds upon previous works in various fields, such as, examining potential emissions reductions from alternative fuel and technology use, quantifying potential uncertainty in vehicular emissions estimates, air quality modelling and assessment of health and cost impacts of air pollution. The most relevant research to date in each of these areas is described in this chapter in order to provide the context for the original work to follow. Based on the objectives of this research, the literature review was conducted under three broad categories, namely, emissions modelling, dispersion modelling and impact assessment of air pollution, as presented in Figure 2.1.

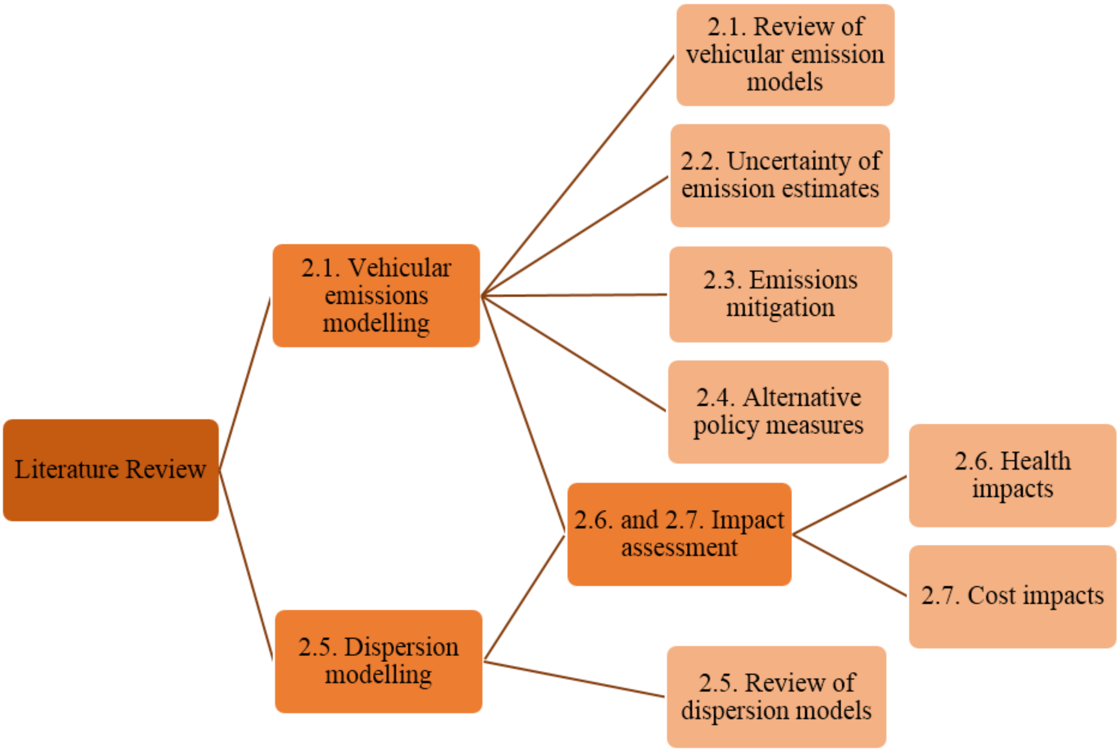

Figure 2.1. Structure of the literature review

2.1. Vehicular emissions modelling

This section reviews some of the popular road transport emissions models used in different countries and based on the review identifies the most suitable model for this research. Also, a summary of relevant studies carried out on vehicular emissions modelling is presented in this section.

2.1.1. Classification of emission models

There are various models used to calculate emissions from road transport which can broadly be classified into static models (also known as top-down or macro-scale emission models) and dynamic models (also known as bottom-up or micro-scale models) (Elkafoury et al., 2014). The static models can further be classified into average speed models and aggregated emission factor models whereas dynamic models can be categorised into traffic situation model and instantaneous model. These models are discussed in the following sections. Table 2.1 shows the input data required by these types of models to define the vehicle operation, specific characteristics and their major areas of application.

2.1.1.1. Average speed model

Average speed models calculate the emissions based on the average speed of the vehicle during a trip and also traffic volume. Average speed emission models are the most commonly used type of approach to model emission levels from road transport. These models do not allow to calculate emission on a fine spatial resolution, however, this limitation is not so relevant for vehicular emission calculation in national level (Elkafoury et al., 2014). A few examples of average speed models are COPERT, Vehicle Emissions Prediction Model (VEPM) etc.

2.1.1.2. Aggregated emission factor model

Aggregated emission factor models operate at the simplest level, with a single Emission Factor (EF) being used for a broad category of vehicles and a general driving condition such as urban roads, rural roads etc. (Wang and McGlinchy, 2009). These types of models calculate vehicular emission on the basis of fuel consumption and kilometres (Km) travelled by the vehicle (Elkafoury et al., 2014). Examples of this type of model are the Mobile Source Emission Factor Model (MOBILE), National Atmospheric Emission Inventory (NAEI).

2.1.1.3. Traffic situation model

In this type of modelling approach, driving dynamics are also taken into account along with average speed. Traffic situations are defined by traffic conditions (e.g. congested, free flow, stop and go etc.) on a specific type of road such as urban, along with the speed limit value on that particular road (Wang and McGlinchy, 2009). One issue with this type of model is that it requires detailed statistics about vehicle speed and traffic situations associated with the trips (Elkafoury et al., 2014). Examples of traffic situation models are HandBook Emission FActors for road transport (HBEFA), Assessment of Road Transport EMISsion model (ARTEMIS) and Vehicle Fleet Emission Model (VFEM) etc.

Table 2.1. Emission models (New Zealand Transport Agency, 2013)

Type	Input data required to define vehicle operation	Characteristics	Application
Average speed model	Average trip speed	Speed and vehicle type/ technology specific	National and regional inventories
Aggregated emission factor model	Area or road type	Simplest level, no speed or vehicle specific dependency	National inventories
Traffic situation Model	Type of road, speed limits, congestion level	Driving pattern (speed, acceleration etc.) and vehicle type/technology specific	Environmental impact assessment, Urban Traffic Management assessment
Instantaneous Model	Driving profile, vehicle specific data (power, speed etc.)	Micro-scale modelling, individual vehicle specific	Urban Traffic Management assessment

2.1.1.4. Instantaneous model

These are the models which operate at the highest level of complexity. Models of this type assign some emission rates to each combination of instantaneous speed and acceleration rate (Wang and McGlinchy, 2009). This type of model requires detailed data about the vehicle, engine characteristics, road geometry and ambient temperature (Elkafoury et al., 2014). Example of the Instantaneous or modal model is Passenger car and Heavy duty vehicle Emission Model (PHEM).

2.1.2. Existing emission modelling tools

This section presents a summary of various models developed and popularly used in different countries and continents such as Europe, USA, New Zealand to calculate emission levels from road transport. Table 2.2 presents the advantages and disadvantages of the popularly used transportation emissions models in different parts of the world.

Table 2.2. Summary of the reviewed transportation emissions models

Name [*]	Suitability	Type	Advantages	Shortcomings
HBEFA [2.5.2.1]	German, Switzerland, Austria	Traffic situation model	Uses traffic situation approach.	Applicable in limited places.
ARTEMIS [2.5.2.2]	Europe	Traffic situation model	Use one of the largest emission databases. Uses traffic situation approach and improved a lot upon HBEFA.	Few sub-models are very complex.
MOBILE [2.5.2.3]	US	Aggregated Emission factor model	Models freeway ramps separately, Calculates the effects of gross emitters.	No engine size breakdown for light vehicles BERs based on Federal Test Procedure rather than “real-world” driving cycles.
MOVES [2.5.2.4]	US	Aggregated Emission factor model	Applicable to motorcycles, cars and buses and non-road emissions are also planned to be included in future. Can estimate future year emissions or energy consumption more precisely as it accounts for future changes in the vehicle fleet and its activity.	Cannot be applied to Ireland.
NAEI [2.5.2.5]	Europe	Aggregated Emission factor model	Suitable for light vehicles. Calculates pollution for a variety of pollutants.	EFs for the heavy vehicles are not good as number of vehicles tested is very small.
PHEM [2.5.2.6]	Europe	Instantaneous Model	Gives consistent results for most of the vehicle categories. Calculates EFs for different road gradients, different vehicle loadings and different gear shift behaviours of drivers in a consistent way Can set up engine emission maps from all sources of measurements as long as high-quality instantaneous test results are available.	Cannot be applied to Ireland.
VEPM [2.5.2.8]	New Zealand	Average Speed Model	European database (COPERT III for heavy vehicle and NAEI for light vehicles) was used as the primary source of model development.	Based mostly on European data which sometimes causes an issue for emission calculation for New Zealand.

			For non-European like Japanese and NZ, JCAP1 and NZ2 were used respectively.	
VFEM [2.5.2.9]	New Zealand	Traffic situation model	Vehicle fleet and traffic sub-models have been revised many times.	Only 12 traffic situations have been defined whereas for ARTEMIS there are more than 200 traffic situations, EFs have not been revised.
VERSIT+ [2.5.2.10]	Europe	Traffic situation model	Takes into account complex emission behaviour for the modern LDVs (Light Duty Vehicles)	Suitable for LDVs only, Good for local use.
COPERT 5 [2.5.2.11]	Europe	Average Speed Model	Suitable for emission calculation for both light and heavy-duty vehicle. Has different sets of emission models for urban, rural and highways. Developed based on tier 3 methodology. Calculate emissions for new technology vehicles such as hybrid electric vehicles. Possible to calculate emissions for peak and off-peak hours separately. Calculate non-exhaust[1] PM emission.	

*Refers to section numbers for the detailed description of the models.

2.1.2.1. Handbook Emission Factors for Road Transport (HBEFA)

HBEFA is a traffic situation model. The first version of HBEFA was published in 1995 and the most recent version of HBEFA (v.3.2) was produced in 2014. It is developed on behalf of several European countries (e.g. Germany, France, Sweden, Switzerland, Austria) (HBEFA, 2014). Colberg et al. (2005) used HBEFA to compare the modelled and measured emission levels in Switzerland and reported that HBEFA demonstrated good agreement with the measured values. It takes into account all important vehicle classes, including passenger cars, light commercial vehicles, heavy duty trucks, buses, motorcycles, mopeds etc. differentiated by fuel, engine capacity and weight classes for a variety of traffic situations. The input factors required to

[1] Non-exhaust particulate emissions (moving vehicles): "Particulate (PM) emissions due to wear of tires, brakes, roads and re-suspended road dust." (Smit and Ntziachristos, 2013)

calculate emission are (Schmied, 2014), vehicle category (PC, motorcycle, LCV, urban bus, single truck, coaches, truck trailer), vehicle size, fuel type (diesel, gasoline, Compressed Natural Gas (CNG), Liquefied Petroleum Gas (LPG), Flexible Fuel Vehicle), technology class i.e. emission standards (Pre Euro 1, Euro 1, Euro 2, Euro 3, Euro 4, Euro 5, Euro 6), load factor (for trucks) and reduction technologies (Particle filter, Silicon Controlled Rectifier, Exhaust Gas Recirculation), road gradient, traffic situation/driving cycles. HBEFA calculates emissions of GHG and most of the air pollutants from road transport. HBEFA provides EFs (hot exhaust emission[2], cold start emission[3], evaporative emission[4]) for all regulated and important non-regulated air pollutants. HBEFA can be applicable to city/local levels or regional levels. However, HBEFA is developed based on a database with all country-specific vehicle fleet data necessary for running the model. It is not possible for the user to apply the model for a different country than those already included in the database (Wang and McGlinchy, 2009; Schmied, 2014; NZ Transport Agency, 2013). Thus, HBEFA cannot be applied for calculating emission from road transportation in the Irish scenario.

2.1.2.2. Assessment of Road Transport Emission Model (ARTEMIS)

As mentioned earlier, ARTEMIS is a traffic situation model (André et al., 2004). This is one of the most comprehensive transportation emission models and it can operate at both macro and micro level (Wang and McGlinchy, 2009). The model contains five sub-models, one traffic situation model, one average speed model, two instantaneous models, and one kinematic regression model. The instantaneous models and kinematic regression models are used in calculating emission from light vehicle, but the models are very complex. The average speed model is the same as COPERT and traffic situation model is better than HBEFA as it contains more than 200 traffic situations (Wang and McGlinchy, 2009). In terms of required input data, ARTEMIS requires detailed data regarding, vehicle activity, fleet composition, and driving condition. Also, a detailed classification of the vehicles (e.g. size, technology etc.) are required in order to achieve a measurable acceptable level of accuracy in emission calculation. The vehicles are to be classified as, cars, light-duty vehicles, motorcycles, Heavy Duty Vehicles (HDV), buses, and coaches and sub-categories such as rigid, articulated etc. can also to be provided.

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[2] Hot exhaust emissions: “Exhaust emissions that occur under ‘hot stabilized’ conditions, which means that the engine and the emission control system (e.g. catalytic converter) have reached their typical operating temperatures” (Smit and Ntziachristos, 2013)

[3] Cold start emissions: “Exhaust emissions that occur in addition to hot running emissions because engines and catalysts are not (fully) warmed up and operate in a non-optimal manner” (Smit and Ntziachristos, 2013)

[4] Evaporative emissions (parked vehicles): “Non-exhaust hydrocarbon losses through the vehicle's fuel system” (Smit and Ntziachristos, 2013)

The model can estimate most of the pollutants, including, regulated pollutants such as CO, Hydrocarbon, NOx, PM, SO_2, lead, and non-regulated pollutants such as CO_2, benzene, toluene, xylene, polycyclic aromatic hydrocarbons, methane, ammonia etc. (André et al., 2004). ARTEMIS can calculate emission for the road, rail, air and ship transport modes and provides consistent emission estimates at both national and regional level. The ARTEMIS tools were designed for three main applications, i.e. emission inventories, scenario calculation for assessing the impacts of alternative measures and inputs for air quality models in order to assess spatial and temporal impacts on the environment (UNECE Transport Division report, 2012). The model has many similarities with COPERT and HBEFA models, especially in terms of input vehicle classes, and outputs of GHG and other air pollutants. For light duty vehicles, ARTEMIS model has been improved compared to the approaches used in COPERT 4 and HBEFA (Wang and McGlinchy, 2009). André (2004), Roujol and Joumard (2009) used ARTEMIS in vehicular emissions related researches. However, as per the UNECE Transport Division report (2012), ARTEMIS has only been fully implemented for compiling national air emission inventories in four countries, i.e. Germany, Austria, Switzerland and Sweden. Application of the model to other countries is not possible without the involvement of the ARTEMIS modelling team to make it compatible for use in other countries.

2.1.2.3. Mobile Source Emission Factor Model (MOBILE)

MOBILE is also an aggregated emission factor model. MOBILE is a US EPA model which calculates emissions from almost all types of vehicle (28 types of vehicles were tested). It was first developed as MOBILE1 in 1978 and has been upgraded periodically since then. MOBILE6 is the latest approved US EPA motor vehicle emission factor model (US EPA, 2016d). The basic parameters considered for aggregation are vehicle model year, vehicle type and engine technology. Emission rates depend on many factors which can be broadly classified into six categories i.e. travel-related, weather-related, vehicle-related, roadway related, traffic-related, and driver related (Ahn et al., 2002). The input parameters for MOBILE6 are calendar year, month, weekend/weekday, hourly temperature, altitude, fuel characteristics (reid vapour pressure, sulphur content, oxygenate content etc.), humidity and solar load, registration (age) distribution by vehicle class, annual mileage accumulation by vehicle class, diesel sales fractions by vehicle class and model year, average speed distribution by hour and roadway, trip end distribution by hour, distribution of vehicle miles travelled by roadway type, engine starts per day by vehicle class and distribution by hour, engine start soak time distribution by hour, average trip length distribution, distribution of vehicle miles travelled by vehicle class, natural gas vehicle fractions, HC species output, Particle size cut off, EFs for PM and Hazardous Air Pollutants, output format specifications and selections etc. (US EPA, 2003). One notable feature about this model is that it calculates emission from motorway and motorway ramps separately.

It was found that vehicles emit significantly more pollutants on motorway ramps than on motorways. This observation indicates that the models which are not considering ramps separately are underestimating the emission (Wang and McGlinchy, 2009). One drawback of this model is that it considers all passenger cars in the same category irrespective of the engine size. Given the differences between the vehicle fleets in Ireland and US, as well as between driving conditions in these two countries, it is not recommended to use this model for emission calculation in EU member countries including Ireland.

2.1.2.4. Motor Vehicle Emission Simulator (MOVES)

MOVES is also a US EPA model. In 2010, EPA approved the MOVES model and the latest version of MOVES is MOVES2014a (US EPA, 2016). It can estimate emissions from all on-road vehicles including cars, trucks, motorcycles, and buses. MOVES is capable of estimating emissions from mobile sources at the national level for criteria air pollutants (i.e. particulate matter, photochemical oxidants and ground-level O_3, carbon monoxide, sulphur oxides, nitrogen oxides, and lead), and greenhouse gases. It derives its emission estimates from second-by-second vehicle performance characteristics for various driving modes (e.g. cruise, acceleration etc.). It incorporates large amounts of data from a wide variety of sources, such as data from vehicle inspection and maintenance programs, remote sensing device testing, certification testing, Portable Emission Measurement Systems (PEMS) (US EPA, 2012). MOVES can also calculate emissions due to combustion, evaporation, and other processes (brake wear, tyre wear, well-to-pump, vehicle manufacture and disposal). Several researchers in the US has used MOVES in traffic emission studies (Wallace, 2012; Zhao, 2013).
MOVES can also estimate future year emissions or energy consumption by taking into account the future changes in the vehicle fleet and its activity. MOVES is designed to take into account fleet turnover and changes in vehicle activity. MOVES can estimate future changes in emission levels as it calculates emissions based on vehicle age, vehicle type, vehicle activity, fuel types and road types. MOVES gives better estimate for not only CO_2 but also other GHGs, compared to other transportation emissions models in US that calculates emissions based on fuel sales as other GHGs are more dependent on fuel controls than fuel consumption. However, as mentioned earlier, MOVES was conceived for the United States, for which it includes a default database of meteorology, vehicle fleet, vehicle activity, fuel, and emission control program data (US EPA, 2012). Thus, MOVES is clearly not suitable for computing emissions from Irish road transport fleet.

2.1.2.5. National Atmospheric Emission Inventory (NAEI)

NAEI is an aggregated emission factor model. This model was mainly developed for the UK in 1970 (NAEI, 2014). It provides historic and projected emission estimates of a range of air quality pollutants and GHGs across a wide range of sectors including transport, waste, energy, industry etc. NAEI calculates emission from hot exhaust, cold start, brake and tyre wear. The database of NAEI comprises of emission data from about 2800 vehicles and over 25000 tests but a very small number of heavy vehicles were tested. Though NAEI is referred to as aggregated emission factor model the aggregated EFs have been derived from average speeds of vehicle fleet composition and traffic volume in the UK (Wang and McGlinchy, 2009). Therefore, has to be checked and calibrated if it is to be used for other countries.

2.1.2.6. Passenger Car and Heavy Vehicle Emission Models (PHEM)

PHEM is an Instantaneous model. PHEM is a vehicle simulation tool capable of simulating vehicle hot and cold emissions for different driving cycles, gear shift strategies, vehicle loadings, road gradients, vehicle characteristics (mass, size, air resistance, etc.) (European Research for Mobile Emission Sources (ERMES), 2016). PHEM has been validated by emission measurements both from LDVs and HDVs in the laboratories under different test conditions and on the road with PEMS. If fed with a detailed list of vehicle specifications, PHEM is capable of modelling emission levels on a wide variety of conditions not covered by the included measurements. Average EF for each vehicle category is then produced taking into consideration the fleet population and technology options (TU Graz, 2009).

Advantages of PEMS are that it is capable of simulating influences of different driving cycles, different road gradients, different gear shift strategies, different vehicle characteristics (mass, size, air resistance etc.), different vehicle loadings in a consistent way based on engine emission maps and most importantly, the model is validated for most of the pollutants. It has good accuracy for estimating fuel consumption and CO_2, NOx, PM and Particle Number. However, Lejri et al. (2018) reported that NOx emission estimates from PHEM have 9.2% absolute global relative error if simulated traffic data is used. The disadvantages with this model are, it needs instantaneous emission data for vehicle speeds, detailed vehicle related data from each measured vehicle and it has uncertainties in the accuracy of estimated CO and HC emissions from modern cars.

2.1.2.7. Transport Emission Model (TREMOD)

TREMOD (TRansport Emission MODel) was developed in 1993 in the framework of the research and development project carried out on behalf of the German Federal Environmental Agency (TREMOD, 2015). The scope of the project was the analysis of motorized transport (its mileage, energy use and emissions) in Germany. In road transport sector, TREMOD is harmonized with the HBEFA. TREMOD estimates emissions for all types of passenger transportation (cars, two-wheelers, buses, trains, aircraft) and all methods of freight transportation (lorries, LCVs and trailers, trains, navigation vessels, aircraft) for Germany. Since the model uses the HBEFA for calculating road transport emissions, all necessary input data on the vehicle fleet in operation are included in the database of the model. This includes the number of vehicles, travelling speeds for urban, rural and highway conditions, annual mileage values and their shares, vehicle divisions as per fuel and technology level (UNECE Transport Division, 2012). The application of this model to other countries is limited by its country specific, in this case Germany, traffic database.

2.1.2.8. Vehicle Emissions Prediction Model (VEPM)

VEPM is an average speed model. It was developed by the Transport Agency and Auckland Council to predict emissions from vehicles in the New Zealand fleet (NZ Transport Agency, 2013). VEPM is generally appropriate for assessments of air quality effects where average emissions are required over 1 hour or 24 hours assessment periods. VEPM cannot accurately represent, extreme driver behaviour, emissions from a particular vehicle, micro events, e.g. emissions over a short time period at a particular location. The model takes three engine capacity categories for passenger cars, <1.4L, 1.4-2.0L and >2.0L. Whereas it was reported that vehicles with engine capacities above 3.0L account for around 20% of the passenger cars. This means that VEPM may underestimate fuel consumption and CO_2 emissions (NZ Transport Agency, 2013). The key parameters included in this model are cold start emissions, the rate of removal of the catalytic converter, fuel properties and emission performance degradation due to vehicles cumulative distance travelled (Wang and McGlinchy, 2009). This model had specifically been developed for NZ transportation emission calculation and cannot be used for emission calculation for Irish road transportation.

2.1.2.9. Vehicle Fleet emission model (VFEM)

VFEM is a traffic situation model and it is based on the New Zealand fleet data. Four types of road situations have been incorporated into this model e.g. Urban, Sub-urban, Rural and Motorways. Three level of services has been included such as, free flow, congested flow and interrupted the flow. VFEM has three submodels, a fleet model, a traffic activity model and an emission factor model. Initially, vehicles were divided into three categories depending on engine size but now they are divided into five categories to represent the New Zealand fleet better. Engine size is a very important factor as CO_2 emission rate is closely related to engine size. The main drawback of this model is its poor documentation of emission and fuel consumption factors.

Moreover, many emission data used for model development are based on assumption rather than the actual observation (Wang and McGlinchy, 2009).

2.1.2.10. VERSIT+

VERSIT+ is also a traffic situation model. It is developed by TNO (Toegepast Natuurwetenschappelijk Onderzoek), in Norway (ERMES, 2016). VERSIT+ has two different models for Light duty (LD) and Heavy duty (HD) vehicles. The aim of VERSIT+ LD is to predict traffic stream emissions for light-duty vehicles in any particular traffic situation. It is based on a database of 12,000 emission tests, mimicking all aspects of real-world driving behaviour. Whereas VERRSIT+ HD is largely based on PHEM software (Lange, 2008). EFs are differentiated for various vehicle types and traffic situations and take into account real-world driving conditions. The pollutants that VERSIT+ can calculate are, regulated, CO_2, NO_2, $PM_{2.5}$, EC, polycyclic aromatic hydrocarbon, PM wear (tyre, Brake etc.) (ERMES, 2013).

VERSIT+ takes account of the complex emission behaviour of modern light-duty vehicles with advanced exhaust systems that makes it suitable for supporting local air quality improvement and traffic management policies. VERSIT+ can be used to design for local air quality improvement, investigating national greenhouse gas reduction strategies in a consistent manner by projecting emissions for road traffic (trucks, buses, passenger cars and motorcycles) into the future (Smit et al., 2007). However, this model has been developed typically for the Dutch situation, thus this might not give reliable results for emissions typical of Irish traffic behaviour.

2.1.2.11. Computer Programme to Calculate Emissions from Road Transport (COPERT)

COPERT is an average speed model. COPERT is developed for official road transport emission inventory preparation for EEA member countries (Austria, Belgium, Bulgaria, Croatia, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Macedonia, Germany, Greece, Hungary, Iceland, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Netherland, Norway, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, Switzerland, Turkey, United Kingdom) (EEA, 2016b). COPERT 5 is the most modified and latest available version of COPERT at this moment (EMISIA, 2018). COPERT was developed taking elements from few other popular emission models like MEET (Methodologies for Estimating air pollutant Emissions from Transport), ARTEMIS, COST and also based on findings of the project PARTICULATES. Its initial version was COPERT 85 (1989), followed by COPERT 90 (1993), COPERT II (1997) and COPERT III (1999). COPERT 4 (2006). COPERT II was the first one with a graphical user interface, built on MS Access 2. COPERT II provided EFs up to Euro 1. COPERT III has new features such as, hot EFs for Euro 1 passenger cars, reduction factors over Euro 1 according to AutoOil, impact on emissions from 2000, 2005 fuel qualities, cold-start methodology for post

Euro 1 PCs, emission degradation due to mileage, alternative evaporation methodology, updated hot EFs for non regulated pollutants, detailed NMVOC speciation, such as, polycyclic aromatic hydrocarbon, persistent organic pollutants, dioxins and furans.

COPERT 5 is suitable for calculating emission from a wide range and variety of vehicles e.g. CNG buses, LPG passenger cars, conventional HDVs in addition to the conventional petrol and diesel vehicles. Three types of roadway situations can be considered in COPERT i.e. Urban, Rural and Highways. The shortcoming with regards to heavy vehicle (>13T) emission estimates in NAEI is improved in COPERT and it gives more reliable results for emissions from the heavy vehicle as it is tested on a larger number of vehicles. COPERT 5 includes many important EFs such as cold over hot ratio, ambient temperature, vehicle use, mileage, fuel characteristics etc. For heavy vehicles, loading and gradients also are taken into account.

The uncertainty in estimating non-exhaust PM emission also is associated with COPERT like any other emission model estimates. However, it is an EU recommended model, regularly updated and used in relevant research, scientific and academic areas. The input data are consistent with the Eurostat classification. As a result, the model is well suited for EU Member States reporting detailed statistical information (UNECE Transport Division, 2012). COPERT is used in many areas (Figure 2.2) such as air quality and impact assessments, projections (energy, CO_2, other air pollutants), inventory preparation, new road (road section) construction, optimization of loading capacity of HDV, airport ground traffic (Kouridis et al., 2014).

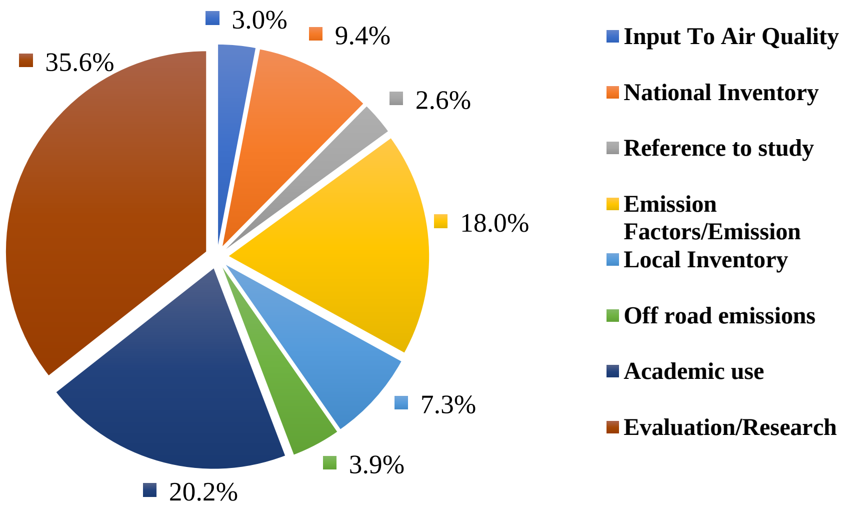


Figure 2.2. COPERT application in different areas (Kouridis et al., 2014)

2.1.3. Discussion

It can be seen from the above literature review that the vehicle emission models vary in their modelling approaches, and in the levels of detail required in their input data. They are suitable for different applications and situations regarding spatial and temporal scales and depending on whether the models are being used to test relative changes from different scenarios or to predict absolute levels of emissions at a given time or place. COPERT and MOVES are two of the most extensively used modelling methods to calculate emission from road transport. As mentioned earlier, MOVES is a US EPA model and developed on the basis of the vehicle fleet and traffic behaviour observed in the US. In order to use them for Irish conditions (or any other European country), the models have to be calibrated and validated with the country specific data. The other emission models (Table 2.2) are also limited by their country specific application. There are other emission models which were developed for specific European countries. HBEFA is one of such models, which have been derived based on the driving characteristics in Germany and Switzerland (Wang and McGlinchy, 2009). In order to check its applicability and accuracy for any other European country, Borge et al. (2012) have studied NOx emissions from road transportation, mainly NO_2 emission in Madrid (Spain) and compared the two models, i.e. HBEFA and COPERT. It was found that the emission value obtained from HBEFA was more than that of COPERT. The authors explained that COPERT underestimated the pollution probably because it considers the average speed whereas in HBEFA different driving dynamics can be considered and different speeds can be fed. After validating both the models, it was observed that the actual emission value was below the model estimated values which indicate that COPERT predicts more realistic emission values. HBEFA is a traffic situation model (combination of road type, speed limit and service level) and takes into account many traffic situations but it has been developed on traffic data and driving scenarios seen in other countries (Germany, Switzerland, Austria, Norway and Sweden), thus in order to use or compare the emission from road transport, it is essential to examine the traffic situation and traffic behaviour in the country under study.

COPERT is suitable for road transport emission estimation in a European country such as Ireland. COPERT is an EU standard software tool that has been designed to develop a national or state level motor vehicle emission inventories, and it uses EFs as a function of vehicle speed. COPERT development was funded by EEA and was developed for use by national experts to calculate road transport emissions to be included in official annual national inventories (EEA, 2018b). COPERT is a widely used emission model in Europe for its ease of use, broad scope and reliable results. Figure 2.3 illustrates the road transport emissions models used in different countries in Europe. Figure 2.4 shows the continent wide share of COPERT use.

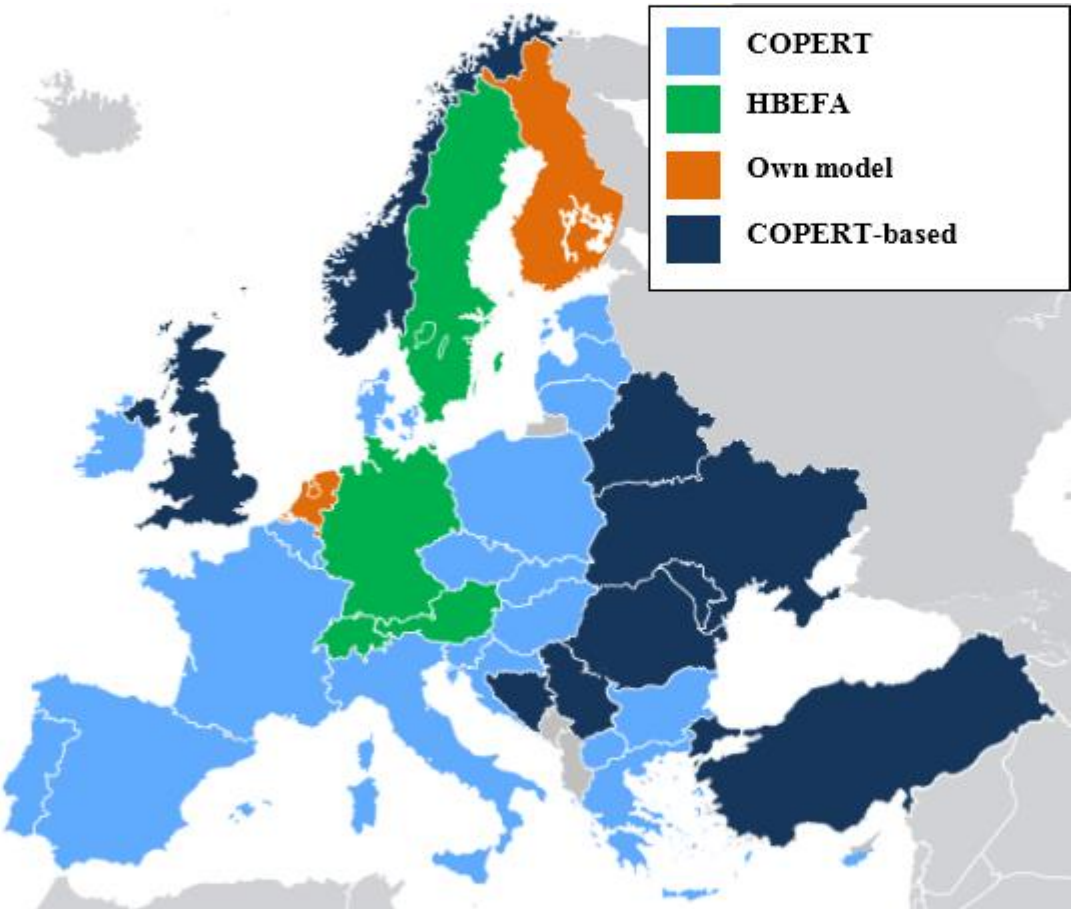


Figure 2.3. Distribution of different model usage for transportation-emission calculation (ERMES, 2016)

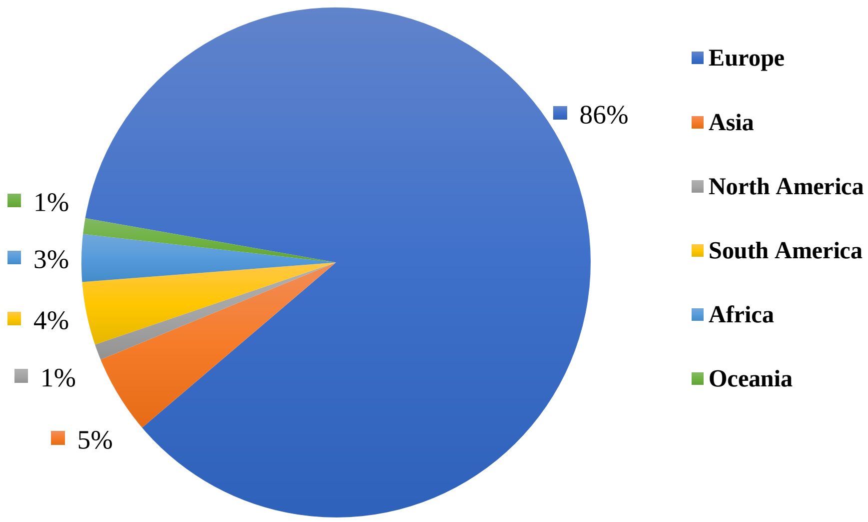


Figure 2.4. Continent wise distribution of COPERT application (Kouridis et al., 2014)

Many researchers in Europe have used COPERT in their studies (Fameli and Assimakopoulos, 2016; Ong et al., 2011; Pouliot et al., 2012). Pouliot et al. (2012) reported that use of the COPERT software tool to calculate road transport emissions allows for a comparable, and consistent data collecting and emissions reporting procedure, complying with the requirements of international conventions and protocols and EU legislation. In Ireland, COPERT has been used for a significant number of years in research (Caulfield, 2009; Farrell et al., 2010; Brady and O'Mahony, 2011; Doorley et al., 2015; Alam et al., 2015) related to estimation of emission levels or reduction in emission levels resulting from Irish road transport fleet. COPERT also is used in national vehicular emissions inventory preparation in Ireland. In summary, it can be concluded that COPERT 5 was found to be suitable for this research for the following reasons,

- It calculates emission of all (important) pollutants from road transport.
- Internationally recognised, used by many European countries for reporting legislative emissions data (EMISIA, 2018).
- It covers all (major) vehicle classes and suitable for emission calculation for both light and heavy vehicles.
- It is developed based on the European countries specific database (EMISIA, 2014b) and is used to develop national emission inventory preparation and its subsequent applications in 22 European countries (Kioutsioukis et al., 2010).
- Diesel NOx EFs are revised in COPERT 5 after it was found that diesel PCs and LCVs are emitting much higher emissions than anticipated (EMISIA, 2018).
- It provides a user-friendly graphical interface to introduce, view and export data (Kouridis et al., 2014).
- It has a different set of emission models for urban, rural and highways.
- Allows specification of energy consumption.
- Improved NOx emissions factors compared with COPERT 4.
- Allows the user to specify separate speeds for urban peak and off-peak hours.
- Capable of calculating non-exhaust PM emissions.

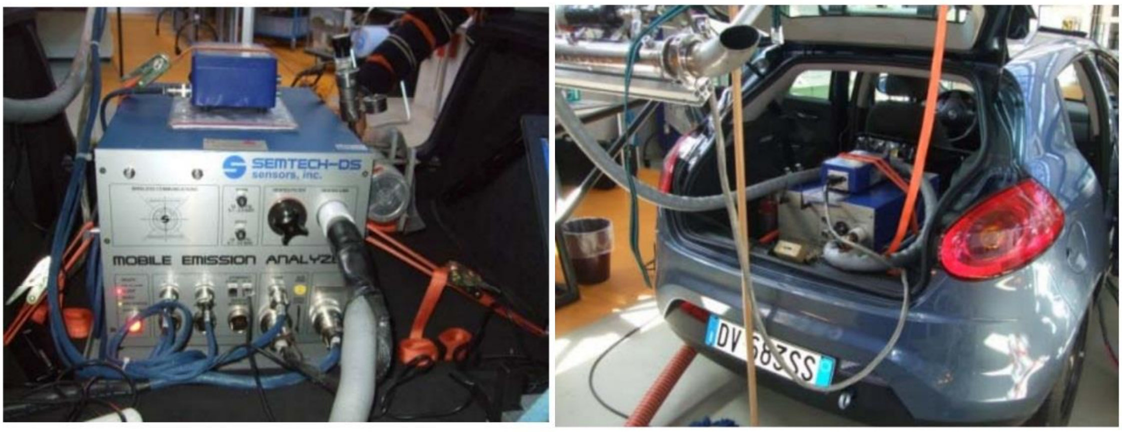
2.1.4. Validation of COPERT

There are many methods for validating environmental models, such as in Laboratory, Onboard, in wind Tunnel, Remote Sensing, Ambient Concentration Measurements and Ambient Mass Balance Method (Smit et al., 2010). Every method has its pros and cons. However, a few studies have been carried out so far to validate COPERT. As mentioned previously, Borge et al. (2012) have used COPERT and HBEFA to calculate NOx emission and compared values with monitored values. The authors reported that the emission values from COPERT give more realistic results than HBEFA. According to Funk et al. (2001) and Mellios et al. (2006), a method to validate road traffic urban emission inventories is to perform a comparison between emission estimates and ambient air quality data by using CO/NOx ratio. Broderick et al. (2007) compared emissions results from COPERT III and concluded that COPERT III shows good agreement with the composite EFs but has a tendency to underestimate the evaporative emissions. An evaluation of COPERT III was also carried out by Ekstrom et al. (2004) with the data collected using remote sensing method. The dataset contained emission measurements of CO, NO, HC for gasoline PCs and diesel PCs and HDVs. For gasoline PCs, the model showed good agreement for NOx emission but overpredicted the CO emission and HC estimate was reasonably good. For HDVs NOx EFs were found to be lower than those from remote sensing measurements.

López et al. (2011) conducted tests on buses using Horiba OBS 2200 on board emission measurement system. The buses were then driven through two predefined roads in Madrid. It was observed that emission values calculated from COPERT 4 were lower than those from experimental tests, especially for CO, HC and NOx. Achour et al. (2011) used a portable gas analyser to calculate real emission and the measured data were compared with the emission obtained from COPERT 4. It was found that COPERT 4 overestimated the CO emissions and underestimated the NO emissions. Berkowicz et al. (2006) compared COPERT emission results in Danish condition with emission levels estimated by OSPM. It was reported that there is underestimation by COPERT for NOx and CO emissions.

Kousoulidou et al. (2010) have validated COPERT with real-world data with the help of PEMS. PEMS is a very useful and popular system for validating EFs. In order to validate COPERT, a Euro 5 diesel passenger car (equipped with a particulate filter) was fitted with a PEMS. The vehicle was then driven over three predefined routes designed to incorporate a variety of driving conditions. The concentrations of the tailpipe emission were measured on a second-by-second basis. Engine speed, torque and other data readings of the test vehicle were also recorded from the Engine Control Unit. GPS data and vehicle-mounted weather station data were also recorded. The instrumental setup used by Kousoulidou et al. (2010) for validating COPERT is shown in Figure 2.5. The results were compared to the EFs provided by the COPERT emission model for validation purpose. Emissions of CO_2, HC and CO correlated well with COPERT values, but NOx emission levels were consistently higher than the COPERT EFs.

Even though various researchers have reported that COPERT 4 underestimates NO or NOx, there is not enough information given on the input information provided to calculate emissions from COPERT in those evaluation studies. It is very important to use the input parameters measured as accurately as possible in order to have promising results. Also, these studies were carried out with an older version of COPERT and NOx EFs are revised in COPERT 5.



(a) **(b)**

Figure 2.5. Instrumental set up (a) Semtech-DS mobile emission analyser; (b) Stock Fiat Bravo 1.6 JTD (DPF Diesel) 88 kW 300 N·m @ 1500 rpm, 300 N·m @ 1500 rpm (Kousoulidou et al., 2010)

2.1.5. COPERT algorithm

Depending on the extent of data availability, three different approaches, namely, Tier 1, Tier 2 and Tier 3 can be used to calculate emissions (EEA, 2016c). COPERT 5 follows the Tier 3 approach which uses a combination of firm technical data, such as, EFs and detailed activity corresponding to each technology class. COPERT 5 uses the improved methodology in terms of updated NO_X EFs for diesel passenger cars and light commercial vehicles.

The following set of equations are used to calculate the total emissions in COPERT (Ntziachristos and Zissis, 2014),

$$E_{total} = E_{hot} + E_{cold} \tag{2.1}$$

where, E_{total} is the total emissions of a pollutant; E_{hot} is the emissions during stabilised engine operation (hot exhaust emissions) and E_{cold} is the cold start emissions discharged during transient thermal engine operation. The hot exhaust emission is calculated using the following equation,

$$E_{hot;\,p,t,r} = N_t * M_{t,r} * EF_{hot;\,p,t,r} \tag{2.2}$$

where, $E_{hot;p,t,r}$ is the hot exhaust emissions of the pollutant *p*, produced in the period concerned by vehicles of technology *t* driven on roads of type *r*; N_t is the number of vehicles of technology *t* in the period concerned; $M_{t,r}$ is the mileage per vehicle driven on roads of type *r* by vehicles of technology *t*; $EF_{hot;p,t,r}$ is the EF for pollutant *p*, relevant for the vehicle technology *t*, operated on roads of type *r*. Cold-start emissions are introduced into the calculation as additional emissions per km using the following formula,

$$E_{cold;\,p,t} = \beta_{p,t} * N_t * M_t * EF_{hot;\,p,t} * \left(e^{cold}/e^{hot}{}_{|p,t} - 1\right) \tag{2.3}$$

where, $E_{cold;p,t}$ is the cold-start emissions of pollutant p (for the reference year), produced by vehicle technology t; $\beta_{p,t}$ is the fraction of mileage driven with a cold engine or the catalyst operated below the light-off temperature for pollutant p and vehicle technology t; M_t is total mileage per vehicle in vehicle technology t; $e^{cold} / e^{hot}|_{p,t}$ is cold/hot emission quotient for pollutant p and vehicles of technology class t.

2.2. Uncertainty associated with emission estimates

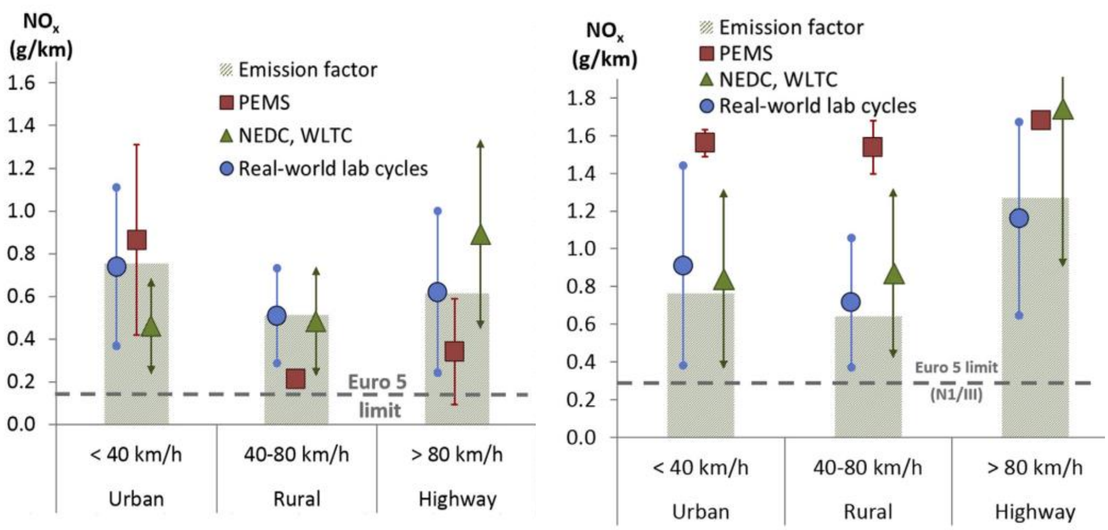
NEI is the main component of air quality management and used in air pollution control programme, emission projections, emission prevention and control measures, quantification of actual emissions, development of policies to prevent and control emissions and environmental impact assessment (US EPA, 2017; Yan et al., 2014; ACAP, 2007; Souza et al., 2013). Therefore, it is very important to calculate air pollutant emission levels with a known and acceptable level of accuracy. The quality of the inventory can significantly be improved by using detailed input data and reducing any uncertainty (Kouridis et al., 2010). Emission levels are affected by various factors associated with the road (road surface condition, gradient, pavement type etc.), vehicle (engine size, fuel type, technology class etc.), environment (relative humidity, temperature etc.) and traffic (speed, acceleration etc.) (Demir et al., 2011). Most of the vehicular emissions models utilize these factors in some form. But models are imperfect abstractions of reality and due to lack of availability of precise input data, all outputs are subject to imprecision and uncertainty (Loucks et al., 2005). Uncertainty estimation is very important to provide information about the sources of uncertainty and their potential reduction (Borrego et al., 2008). One component of uncertainty related to model estimates is the uncertainty related to input data. Therefore, it is recommended that information on uncertainty must be included while reporting the NEI (European Union, 2016; Eggleston et al., 2006). Uncertainty in emissions estimates has been assessed by several researchers (Kioutsioukis et al., 2004; Saikawa et al., 2017; Notte et al., 2018).

Sensitivity analysis, on the other hand, helps to build confidence in the model by studying the uncertainties that are often associated with parameters in models (Yao et al., 2014). Sensitivity analysis is "the study of how the uncertainty in the output of a model (numerical or otherwise) can be apportioned to different sources of uncertainty in the model input" (Saltelli, 2002). A sensitivity analysis combined with uncertainty analysis can help to understand if the current state of knowledge about the input data and related uncertainties is sufficient to take an informed reliable decision (Kioutsioukis et al., 2004). Thereby, it helps to identify which data or parameters need resource allocation to achieve the desired level of confidence on the results (Kioutsioukis et al., 2004). Several researchers have carried out a sensitivity analysis of emissions modelling parameters (Choi et al., 2010; Yao et al., 2014; Singh et al., 2012; Garcia et al., 2013; Bell et al., 2013).

Fameli and Assimakopoulos (2015) explored the effects of various important input parameters such as temperature, speed, mileage share and trip length, that are required in calculating emissions, on emission levels for Greece and Attica (Greater Athens Area) using COPERT 4. The results show that 30% of NOx and 40% of CO_2, CO, VOC, NMVOC emissions come from Attica. Pallavidino et al. (2014) studied the advantages and key factors of a bottom up approach in determining road transport emission inventory using Trefic software which implements COPERT 4 EFs for most of the pollutants and concluded that the bottom-up methodology should be preferred as it allows a more transparent and straightforward choice of input parameters. Vanhulsel et al. (2014) examined several parameters that influence the uncertainty associated with the outcomes from a road transport emissions model. The authors used the E-Motion Road model which is based on the COPERT 4 methodology. The uncertainty was assessed by varying model input parameters such as trip lengths, speed profiles, hot exhaust EFs and cold start EFs and thereby, evaluating their effect on the emission levels. The authors observed that the effect is noteworthy for CO_2, and even more pronounced for other air pollutants such as $PM_{2.5}$, NOx and VOC.

Apart from the uncertainty related to input parameters to be provided by the user, there could be uncertainties associated with the default EFs used in the model. In Volkswagen (VW) scandal, VW and Audi PCs have been found to violate Euro 5 emission standards resulting in higher NOx emission levels from these vehicles than from vehicles with properly operating emission control systems. Even though emission standards became stringent with every progressive emission standard directive, in reality, in-service emissions from diesel vehicles have not reduced at all through the Euro 1-5 emission standards (Moody and Tate, 2017). Transport and Environment (2016) has reported that VW Euro 5 LCVs produce cleanest vans when empty but exceeds the limit by 225% when full. Also, in tests conducted by Kadijk et al. (2016) on a wide range of Euro 5 vehicle makes and models, it was observed that the NOx values obtained in real-world conditions in a lab or on the road are significantly higher than that obtained from the type approval test or Euro 5 limit. Also, it has been pointed out that, earlier, the model EFs could partially be linked to the real-world emissions considering the difference in them with respect to driving behaviour under real-world condition. But now the difference in emissions are much higher even when the vehicles are driven under such conditions that are comparable to the type approval test condition (Kadijk et al., 2016). Weiss et al. (2011) found the NOx values for diesel Euro 5 cars to be in the range 0.4-1 g/km which is 3-5 times more than the European driving cycle values. Ntziachristos et al. (2016) present mean NOx EFs for PC and LCV measured under different conditions (Figure 2.6). These EFs are obtained from real-world lab tests, measurements from PEMS, and other driving cycles like New European Driving Cycle, Worldwide harmonized Light vehicles Test Cycle results.

It has been suspected that a total of 21.4 million Euro 5 PCs and 2.2 million Euro 5 LCVs across Europe are faulty (Transport and Environment, 2016). It is important to investigate whether the discrepancies between the real-world emission levels and the lab tested levels are adjusted in assessing the environmental impacts of NOx. Moody and Tate (2017) investigated the NOx emissions behaviour of in service Euro 6 vehicles under real driving and reported that Euro 6 diesel cars show a significant improvement over Euro 5 diesel cars. Ireland had the highest number of newly registered Euro 5 diesel vehicles in Europe in 2014 with an overall share of 43.57% (Howley et al., 2015) in the overall fleet which is expected to increase to 73.9% by 2025. Whereas, the entire LCV and bus fleet in Ireland are diesel powered. NOx is a classic air pollutant and is responsible for a wide variety of health effects and financial burden. Therefore, the potential impacts of *dieselgate* in Ireland should be examined and alternative policy measures should be implemented accordingly to reduce the emissions and its subsequent impacts.



(a) Passenger Cars **(b) Light Commercial Vehicles.**

Figure 2.6. Simplified current model NOx EFs and measured emission levels for diesel Euro 5. Uncertainty ranges correspond to ±standard deviation (Ntziachristos et al., 2016)

2.3. Emissions mitigation through alternative fuel options

As mentioned in Chapter 1, road transport is responsible for the emissions of various regulated and unregulated pollutants (Gkatzoflias et al., 2012). Also, the transport sector is the largest energy-consuming sector in Ireland, with a share of 42% of overall energy demand in 2015 (Howley et al., 2015). The emissions from road transport account for 95.8% of the overall transport emissions in Ireland (EPA, 2017a). The number of cars in Ireland has increased by 46% (Department of the Environment and Local Government, 2001; DTTaS, 2016b) over the past 15 years. Major parts of CO_2 emissions result from passenger cars (Fameli and Assimakopoulos, 2015). The electrification of the transportation sector has become necessary considering the increase in transport demand, resulting in greenhouse gas, urban air pollution and fossil fuel depletion (Weldon et al., 2016). EVs offer zero tailpipe emission which are more important in the case of urban areas where both the population and vehicle densities are high. Electric mobility is the most promising alternative to ICEVs towards a cleaner transportation sector (Casals et al., 2016). However, the net emissions savings from EV use have always been a concern as it largely depends on the fuel from which the electricity is produced. Thus, while evaluating the environmental benefits of EVs over conventional vehicles, one must consider the emissions resulted from the energy production. The Well-to-Wheel (WTW) methodology is commonly used to estimate fuel efficiency of a vehicle in its use phase, which can be considered as a combination of the Well-to-Tank (WTT) and the Tank-to-Wheel (TTW) (Campanari et al., 2009; Hawkins et al., 2012). The WTT phase comprises the emissions resulted during fuel extraction, refining, and distribution activities needed to fill the vehicle tank. The TTW emissions include the emissions produced by fuel combustion to generate traction power. The environmental impact of ICEV mostly depends on the TTW phase. Whereas, in the case of EVs, the TTW emissions are zero. Thus, in the carbon footprint assessment of EVs, WTT i.e., the emissions resulting from the electricity generation is analysed (Nicolay, 2000). It is very difficult to determine the electricity consumed by EVs as it depends on many factors, such as driving behaviour, use of auxiliaries, weather conditions (Badin et al., 2013; De Vroey et al., 2013). Therefore, a wide range of electricity consumption, varying from 0.10 Kilowatt/Kilometre (kWh/km) to 0.24 kWh/km, has been reported by the researchers (Campanari et al., 2009; De Vroey et al., 2013; Hawkins et al., 2012; Helms et al., 2010; Strecker et al., 2014). In addition to the consumption variability, the emissions from the electricity production and distribution were also assessed by the researchers (Helms et al., 2010). Thus, unlike ICEVs, the environmental impacts of EVs are largely variable depending on the source of electricity and electricity consumption while in use (Campanari et al., 2009). However, production of electric vehicles, their use and demolition are comparable to other conventional vehicles (Vreugdenhil, 2017). Table 2.3 provides a list of NOx and PM EFs, WTW CO_2 emissions, TTW fuel consumptions and TTW electricity requirements of the available fuel/technology options for PCs (European Commission, 2007).

The advantages of electric cars depend on the primary source of electric supply and the efficiency of the power stations (Engerer and Horn, 2010). In electricity production, higher the use of coal lower is the advantage in CO_2 reduction. There would be no advantage if the electricity is produced entirely from coal (Engerer and Horn, 2010). If BEVs (Battery Electric Vehicles) are fuelled with electricity produced from coal power then the per kilometre CO_2 emissions are higher than that of ICEVs (Wolfram and Lutsey, 2016). For Plug-in Hybrid Vehicles (PHEVs), the emission behaviour is less clear but if 50-50 distribution is assumed, i.e. 50% of the time fuel is used as power and 50% of the time electricity is used (Stewart et al., 2015) then the GHG emission behaviour is similar as BEV. If wind energy is used as the source of energy, then there is a huge scope of GHG emissions savings with WTW emissions being 6g CO_2 per km.

Table 2.3. Passenger car energy consumption and EFs for different fuel types

Technology	**Weight**	**NOx (g/km)**	**PM (g/km)**	**WTW CO_2 (g/km)**	**TTW FC (MJ/km)**	**TTW Electricity (MJ/km)**
Conventional gasoline – Euro VI	1.4-2.0 t	0.029	0.001	193,490	2,240	0
Conventional diesel – Euro VI	<2.0 t	0.207	0.001	159,792	1,830	0
Hybrid	Mean	0.120	0.002	139,655	1,168	0.292
Electric	Mean	0	0	154,800	0	1200
CNG stoich. 2017	Mean	0.009	0	120,920	1,880	0
Biofuel 20	Mean	0.411	0.016	140,028	1,770	0
Ethanol 15	Mean	0.399	0.011	158,733	1,900	0
Biofuel 100	Mean	0.447	0.011	87,491	1,880	0
Ethanol 100	Mean	0.399	0.009	108,810	1,900	0

Davies and Kurani (2013) explored the actual implications for energy and emissions impacts PHEV based on observation rather than assumption. Unlike EVs, PHEVs allow the flexibility to be run by gasoline when there is not a chance to charge the vehicle. But with this flexibility comes the uncertainty in estimating the shares of electricity and fuel consumption and resulting emissions, as it largely depends on many factors associated with charging and driving behaviour. Lorf et al. (2013) studied energy consumption and CO_2 emissions from HEV (Hybrid Electric Vehicle), PHEV, EV and ICEV presented results from 40 vehicles and the results show that depending on the source of primary energy WTW CO_2 emissions could be lower in PHEV and HEV than BEV.

Smith (2010) examined the potential benefits of PHEV in reducing primary energy use and CO_2 emissions in Ireland. It was concluded that PHEVs have the potential to reduce primary energy requirement and CO_2 by more than 50% per kilometre. The author also reported that PHEV operating in EV mode will account for approximately 10% passenger car kilometre in 2020 and 50% in 2030. PHEV mitigate the tailpipe emission which contributes to the local air pollution and climate change (Ralston and Nigro, 2011; Sharma et al., 2013). Wu et al. (2015) pointed out that significant reduction in CO_2 and NOx in HEV is possible compared to the conventional gasoline and diesel vehicles. It was also reported in the study that even though the purchase cost is higher for HEV, but with time the total costs of vehicle purchase and fuel consumption of diesel and petrol cars exceed that of HEV in less than 3 years. Ke et al. (2017) report that by 2030 the WTW CO_2 by BEV should approach 100g/km due to a decrease in fossil electricity. Though the NOx emissions from BEV are higher than conventional gasoline vehicles by 66% it is expected that by 2020 there would be a reduction benefit of 41% due to a decrease in fossil usage in electricity production. This study concludes that BEV and PHEV could significantly reduce the WTW CO_2 by 32% and 46% respectively compared to multiport fuel injection. This is due to the rapid switch from coal to natural gas in electricity production.

Hydrogen Fuel Cell Electric Vehicles are also free from tailpipe emissions except for water vapour. Use of Hydrogen Fuel Cell Electric Vehicles can lead to the lowest GHG emissions if wind-powered electrolysis is used to produce the hydrogen, but this currently is very expensive to be considered as a viable option. Currently, the hydrogen is produced by reforming the natural gas where the WTW GHGE is 177g eCO_2 per km. whereas use of natural gas in producing electricity can result in 90g eCO_2 per km. With average European electricity sources, BEVs provide an about 40%–50% GHG benefit compared with average vehicles (Wolfram and Lutsey, 2016). Brady and O'Mahony (2011) studied the potential emission reduction with high (25%), medium (15%) and low (10%) market penetrations of EVs to be used in the work trips in Dublin to meet Ireland's EV target for 2020 which was to have 10% EVs in the fleet in 2020.

Alternative fuel options such as LPG, CNG, bio-CNG are available for buses and they have significant potential in reducing emissions. Public Transport (PT) bus fleet allows the introduction of alternative fuel options and reduction of emissions on a large scale. The most effective measure to reduce emission levels is the fleet renewal with the potential to reduce emissions on an average by 16.04% for all major pollutants, CO, CO_2, NOx, SO_2, $PM_{2.5}$, PM_{10}, CH_4 (Lumbreras et al., 2008). Several researchers have studied the potential of alternative fuels such as natural gas, fuel cell in reducing emission levels (Cohen, 2005; Karlstrom, 2005, Gonçalves et al., 2009). Bio-CNG, electric buses are successfully being used by other European countries. Since 2002, there have been 9000 tonnes of CO_2 reduction per year from buses after implementing biogas use produced from organic waste in Sweden (IEA, 2013). It has been estimated that, in 2020, 18% of the total EU buses will be CNG, LPG powered and 30% will be a hybrid, electric and fuel cell powered (Mahmoud et al., 2016). CNG has 113% fuel cost savings over gasoline and 57% over diesel buses (Khan et al., 2015). Biomethane is one of the most indigenous nonresidue European transport biofuels and has the potential to reduce emissions by 75% (Korres, 2010). Ryan and Caulfield (2010) examined optimal fuel type for urban bus operations in Dublin and concluded that only renewing the fleet with better technology will not be enough measure to reduce the emission levels significantly, therefore, incorporation of alternative fuel options is necessary. Biomethane is renewable and offers a reduction of around 80% CO_2 compared to diesel if produced from municipal waste (Baldwin, 2008). Similar to electric cars, electric buses have zero TTW emissions but the WTT emissions need to be considered. If the electricity is produced from renewable sources then the WTT emission for battery electric buses is as low as 20 eCO_2/km (Mahmoud et al., 2016).

2.4. Alternative policy measures

EV purchase rate in Ireland is very low even after providing fiscal incentives to increase the uptake rate. Figure 2.7 shows the new car registration pattern with respect to engine type over past 10 years (SIMI, 2017). SEAI offers a grant of up to €5,000 for every BEV or PHEV purchase. In addition to the grant, there are VRT reliefs of €5,000 for BEV and €2,500 for PHEV (SEAI, 2017a). The annual motor tax for EV is €120. Even after all these measures, the EV uptake is significantly low in Ireland. Hao et al. (2006) suggested that the financial incentives that would reduce the emissions are, financial subsidy on low emission vehicles, subsidies on retrofit programmes when feasible, financial bonus (cash for replacement) to encourage early scrappage, increase local parking fees or increase in congestion pricing.

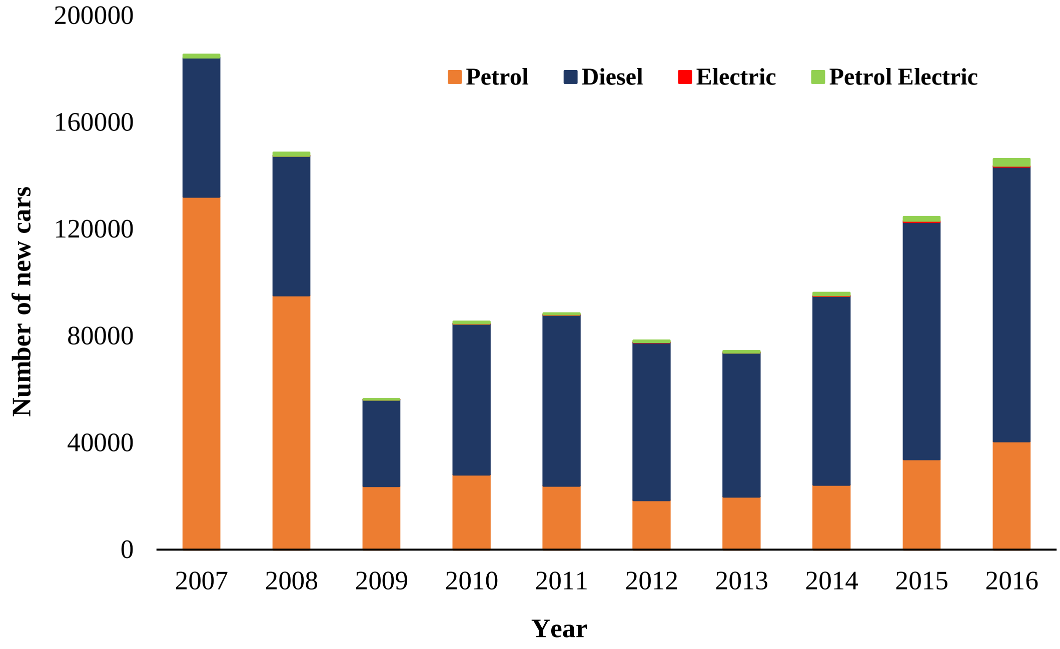


Figure 2.7. New car registration pattern over 2007-2016

Norway is known to be the electric vehicle capital of Europe. In a survey conducted in Norway to find the reason for the people to prefer EVs over ICEVs, 54% have reported the reason as less expensive followed by 25% as EVs are environmentally smart (Aasness and Odeck, 2015). The incentives (Aasness and Odeck, 2015) which resulted in a rapid increase in EV uptake in Norway are,

- Temporary Exemption from on-off registration tax
- Exemption from annual vehicle tax
- Exemption from road tolls
- Temporary use of transit lanes
- Exemption from value added tax
- Further reduction in company car tax
- Exemption from parking fees on municipal owned parking facilities
- Reduced company car tax
- Permanent use of transit lanes
- Exemption from paying car ferry fees

Aasness and Odeck (2015) suggested that these policies were beneficial for Norway as the electricity production was hydropower based. The authors also mentioned that "the Norwegian EV policy should be terminated as soon as possible and that this policy should not be implemented by other countries." as these incentives have led to huge increase in the use of private cars instead of public transportation and cycling. Valeri and Danielis (2015) studied the market penetration of cars with alternative fuel powertrain technologies in Italy. The authors considered 5 scenarios, such as, introducing a state subsidy on purchasing low emission cars, 20% increase in fossil fuel price, a €5,000 price decrease in BEVs, a threefold range increase for BEVs and the last one is all those previous scenarios together. It was found that only when all the incentives were implemented together, i.e. in the fifth scenario the BEV market penetration can be increased significantly, about 15%. Nanaki and Koroneos (2016) designed different alternative fuel penetration scenarios and found that maximum emission saving is possible with 10% natural gas, 10% biodiesel, 10% electricity resulting in 49%-78% savings in CO, NOx, OM, HC and 21% in CO_2. Main components of electric mobility are vehicles, batteries, charging points, and services (Nanaki and Koroneos, 2016).

Due to increasing health concern from this very high level of NOx emissions from road transport, several countries and cities have announced to ban diesel use and/or new diesel vehicles sales from certain future years. European cities like Paris, Athens, Madrid have announced to ban diesel cars and vans from 2025 (McGrath, 2016), Brussels to ban all diesel vehicles from 2030 (Manthey, 2018), and Rome to ban diesel cars from 2024 (Dow, 2018). In some parts of Hamburg, Germany a ban on old diesel vehicles has already started (Byrne, 2018). France planned to become carbon neutral by 2050 (BBC, 2017). Norway plans that by 2025 all new cars sold should be zero emissions (Knudsen and Doyle, 2018). The UK is set to ban both the diesel and gasoline vehicle sales from 2040 (Asthana and Taylor, 2017).

2.5. Dispersion modelling

Atmospheric dispersion modelling is a simulation of the physical and chemical phenomenon governing the transport, dispersion and transformation of pollutants into the atmosphere. Air pollutant concentrations are estimated by dispersion models based on the pollutant emissions and the nature of the atmosphere. The most common application of dispersion modelling is to assess the potential environmental and health impacts of emissions from a source of pollution. Some of the other uses of dispersion models are, assessing compliance with air quality guidelines, managing existing emissions, planning new facilities, designing ambient air monitoring networks, forecasting pollution episodes, evaluating policy and mitigation strategies, identifying the main contributors to existing air pollution problems, saving cost and time over monitoring.

2.5.1. Types of dispersion models

The dispersion models can broadly be classified as Box models, Gaussian models, Lagrangian models and Computational Fluid Dynamic models (Holmes and Morawska, 2006). The following are the brief descriptions of the mentioned types of models (Holmes and Morawska, 2006):

2.5.1.1. Box Models

The working principle of box models is the conversion of mass. The road or the study area is considered as a box into which the pollutants are discharged and undergo physical and chemical transformations. These types of models assume that the air inside the box is well mixed leading to uniform concentrations throughout. The advantage of box model is the simplicity of its meteorology which allows inclusion of detailed chemical reaction mechanisms and detailed treatment of the aerosol dynamics, enabling the physical and chemical phenomenon of the particles within the atmosphere to be represented much better. However, a box model generally does not take into account the local concentrations of the pollutants as an input parameter while defining the initial conditions. Therefore, they are not suitable for modelling the concentrations of particles where they are likely to be highly influenced by the local changes to the emissions and wind field.

2.5.1.2. Gaussian models

Gaussian or Gaussian plume models are widely used in atmospheric dispersion modelling. This type of model calculates concentrations in a straight line from the source to the receptor within the modelling domain. The width of the plume is determined by factors defined by travel time from the source or stability classes. One limitation of Gaussian plume models is that they do not consider the time required for the pollutant to travel to the receptor as they use steady state approximations. Further, Gaussian plume equation assumes that there is no interaction between plumes. However, this can be overcome by dividing the emission source into a series of puffs over time which will allow the wind speed to be varied.

2.5.1.3. Lagrangian models

Lagrangian models also consider the region of the air as a box which follows a trajectory as it moves downwind. This type of model incorporates the change in concentrations due to the turbulence of wind components, mean fluid velocity and molecular diffusion. The pollutant concentration is a product of source related term and a Probability Density Function (PDF) as the pollutants travel from one point to another. Lagrangian models take into account the changes in concentration due to mean fluid velocity, molecular diffusion, and turbulence of the wind components. It is possible to model the non-linear chemistry in Lagrangian models using "either the superimposition of a concentration grid on the domain, followed by calculation of the concentration in each grid or the particle can be treated as an expanded box and the photochemical module of the model applied to each box". Based on the meteorological data, the variance of the wind velocity changes and Lagrangian autocorrelation function are calculated. The diffusion of the pollutants in Lagrangian models are characterised by the generation of semi-random numbers and hence, not confined by stability classes like in the Gaussian models.

2.5.1.4. Computational Fluid Dynamic models

This type of model is based on the principle of mass and momentum conversion and provide a complex analysis of fluid flow. Computational Fluid Dynamics models use finite difference and finite volume methods in three dimensions to resolve Navier-Stokes equations to analyse the fluid flow. Turbulence is classically incorporated using k-ε closure methods to estimate the viscosity parameters in both the pollution transport and momentum equations. It is assumed that the pollutants become diluted equally in all the directions. Although this treatment performs well on a flat boundary layer, the closure method needs to be modified when a stratified boundary layer exists to include the Coriolis force[5] and reduced wind shear in the upper atmosphere, which results in an overestimation of the eddy viscosity.

2.5.2. Dispersion modelling tools

Several dispersion modelling software tools are available. The most popularly used models are reviewed to examine their suitability in the case of Ireland. Selected line source dispersion models are described in the following section.

2.5.2.1. California Line source model (CALINE)

CALINE is a line source air quality model developed by the California Department of Transportation (Hatano et al., 1989). It is a Gaussian type of model and employs a mixing zone concept to characterise pollutant dispersion over the roadway. The purpose of the model is to assess air quality impacts near transportation facilities. For a given source, meteorology and site geometry, CALINE can predict pollutant concentrations for receptors located within 500 meters of the roadway. In addition to predicting concentrations of non-reactive pollutants such as CO, the model can predict concentrations of NO_2 and PM. CALINE can not only model air quality in street canyons but also pollution in special situation, such as near an intersection, and parking facilities (Vardoulakis, 2003).

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[5] Coriolis force - It is an inertia force that acts on objects that are in motion within a frame of reference that rotates with respect to an inertial frame.

2.5.2.2. Calculation of Air pollution from Road traffic (CAR)

CAR or CAR International was developed based on wind tunnel experiments, measurements and theoretical considerations for determining air quality in city streets at the kerbside (Boeft et al. 1996). In CAR the kerbside is considered as the receptor as it is the closest place to the traffic and normally occupied by pedestrians. It can estimate average concentrations of relatively inert pollutants and NO_2. Pollutant concentrations are calculated at distances between 5 to 30 metres (m) at a height of 1.5 m from the pavement surface. A set of different building geometries is defined to capture the complex and strong influence the buildings in the immediate vicinity on emission dispersion (Vardoulakis, 2003).

2.5.2.3. STREET-SRI (Stanford Research Institute)

STREET is developed based on box type model. The model assumes that the pollutant concentrations occurring at the road side due to road traffic consist of two components, one is due to the existing urban background concentration and other is due to the direct contribution from traffic emissions on that street. STREET simulates pollutant concentrations on both sides of the streets at the defined distance and height of the receptor. The concentration on the windward side is modelled such that it decreases vertically due to fresh air through the top of the canyon. While on the leeward side, the concentrations are assumed to be inversely proportional to the distance between the pollution source and the receptor location (Johnson et al., 1973; Dabberdt et al., 1973). STREET is parameterised using data from a regular street canyon and therefore, needs to be recalibrated before being applied to any other canyon with other street geometry (Vardoulakis, 2003).

2.5.2.4. Canyon Plume Box Model (CPBM)

CPBM consists of two components of modelling methodology (Vardoulakis, 2003). It uses a Gaussian model to derive the direct contribution from the source and a box model to calculate the effect of turbulence on the concentrations. The plume generated inside the street canyon is divided into three zones, which are assumed to disperse as per Gaussian plume model following straight line trajectories. The total concentration on the leeward side is calculated by adding the direct contribution from the source to the recirculated fraction. Whereas in the case of the windward side, the only contribution is due to the turbulence due to the recirculating air and here, the dilution of the concentrations due to the fresh air ventilation is also taken into account. A simpler plume model is used to incorporate the effect of wind parallel to the street axis or for very low wind speeds.

2.5.2.5. AEOLIUS

AEOLIUS model is developed by the UK meteorological office (Buckland, 1998). It is based on similar concepts and techniques previously used for the development of the OSPM. Nevertheless, there are some discrepancies in concentrations estimated by the two models, which could be due to different methods of coding, parameterisation, data pre-processing techniques (Vardoulakis, 2003). There are two screening versions of AEOLIUS, namely AEOLIUS Screen and AEOLIUSQ Emission. Like OSPM, in AEOLIUS too, the mechanical turbulence in the street is empirically derived due to the wind and vehicle traffic.

2.5.2.6. AERMOD

AERMOD is developed by the US EPA (Cimorelli et al., 2004). AERMOD is a steady state Gaussian plume type of model. AERMOD can, not only model concentrations as a result of the line source of emissions but also from point, area and volume sources. However, AERMOD is more suitable for short range dispersion. In the stable boundary layer, the pollutant distribution is assumed to be Gaussian in both the horizontal and vertical directions. In case of the convective boundary layer, whilst the nature of horizontal distribution is considered the same, the vertical distribution is modelled using a bi-Gaussian PDF developed by Willis and Deardorff (1981). Therefore, AERMOD is capable of modelling "buoyant plumes and incorporates a treatment of lofting, whereby the plume remains near the top of the boundary layer before mixing with the convective boundary layer" (Holmes and Morawska, 2006). Also, generally, Gaussian models are limited to treatment of flows over a simple terrain, however, AERMOD incorporates a simple method to approximate air flows over complex terrain (Snyder et al., 1985). In AERMOD, the atmosphere is described by relationships using only a single measurement of surface wind speed, direction and temperature to predict vertical profiles of wind speed, wind direction, turbulence, temperature and temperature gradient.

2.5.2.7. Operational Street Pollution Model (OSPM)

OSPM is developed by the Department of Environmental Science at Aarhus University (Aarhus University, 2018). OSPM is an evolved version of CPBM (Hertel and Berkowicz, 1989). Therefore, like CPBM, OSPM also combines one Gaussian plume model and a box model. However, in addition, OSPM has another model to take into account the effect of the background concentration of pollutants. If the wind is not parallel to the street axis, the vortex is assumed to form inside the canyon. The length of the vortex along the direction of the synoptic wind is considered to be twice the up-wind building height. For winds with speed below 2 meters per second, vortex length is considered to decrease with the wind speed (Berkowicz, 2000). The maximum possible width of the recirculation zone is the width of the canyon. The relationship between roof level and street level winds in the canyon is calculated by a logarithmic relationship that takes into account the synoptic wind direction, the surface roughness length, and the height of initial dispersion of vehicle exhausts. Pollutant concentrations on the leeward side of the street are calculated by adding the concentrations due to direct emission contribution from the vehicles and the recirculation contributions. While the concentrations on the windward side are calculated by only taking into account the direct contribution of emissions generated outside the recirculation zone. However, if the recirculation zone extends throughout the whole canyon, only the recirculation component is considered on the windward side. In case of near parallel wind, emissions from outside the recirculation zone may contribute to the leeward pollutant concentrations. The concentrations on both sides of the canyon become equal when the wind flow is parallel to the street axis or wind speed is near zero (Vardoulakis, 2003).

2.5.3. Choice of the dispersion model

Following are some of the characteristics of an ideal air pollution model (Stern, 1984),

- Physically realistic and accurate
- Universal i.e. suitable for various emission sources, temporal and spatial scales, meteorological and topographical conditions, air pollutants.
- Easy to understand and use
- Supported by readily available input data
- Fully verified on real data
- Well documented
- Interactive with users
- Computationally fast
- Easy to adapt for running on a simple computer
- Can be integrated with other software-based modelling systems

Based on all these characteristics and available literature OSPM were chosen to model air pollution from road transport in this research. Researchers have used OSPM to model traffic pollution concentration in Ireland (Pilla and Broderick, 2015; Ganguly and Broderick, 2009) and in other countries (Lazic et al., 2016; Kumar et al., 2016; Ottosen et al., 2015; Assael et al., 2008; Ghenu et al., 2008; Berkowicz et al., 2008; Berkowicz et al., 2006; Kukkonen et al., 2001).

2.5.4. OSPM Mathematical Model

OSPM computes the total concentration of a pollutant in the street as the sum of the concentration related to direct emissions from vehicles (C_d), the recirculated component (C_r) and the urban background concentration (C_b) of pollutant outside the canyon. Therefore, the total concentration can be expressed as follows,

$$C_S = C_d + C_r + C_b \tag{2.4}$$

The direct contribution is calculated using a plume model and the recirculation contribution is calculated by a box model (Berkowicz et al., 1997). It is assumed that the traffic and the resulting emissions are uniformly distributed across the street. The emission field is treated as a number of infinitesimal line sources of thickness 'dx' aligned perpendicular to the wind direction at the street level. The emission density (dQ) for such a line source is calculated as,

$$dQ = \frac{Q}{W} dx \tag{2.5}$$

Where Q is the emissions in the street (g m^{-1} s^{-1}) and W is the width of the street.

The concentration at a point located at a distance x from the line source is given by,

$$dC_d = \sqrt{\frac{2}{\pi}} \frac{dQ}{u_b \sigma_z(x)} \tag{2.6}$$

Where u_b is the wind speed at the street level and $\sigma_z(x)$ is the vertical dispersion parameter at a downwind distance x.

Equation 2.6 is integrated along the wind path at the street level. The integration path depends on wind direction, extension of the recirculation zone and the street length. If the angle between the roof level wind direction and street axis is ϕ, then the street level wind too forms an angle ϕ with the street axis in the recirculation zone. But outside the recirculation zone, the wind direction is the same as that at the roof level.

$$C_d = \sqrt{\frac{2}{\pi}} \frac{Q}{W\sigma_w} ln \frac{h_o + (\sigma_w/u_b)W}{h_o} \tag{2.7}$$

h_0 is the effective release height of car exhaust due to initial dispersion. σ_w is the vertical velocity fluctuation due to mechanical turbulence generated by wind and vehicle traffic in the street. The concentration in the recirculation zone is calculated assuming that the inflow rate of pollutants into the recirculation zone is equal to the outflow rate and that the pollutants are well mixed inside the zone. When the recirculation zone extends through the whole canyon, the direct contribution at the windward side is zero and the total concentration equals the concentration due to the recirculation component. However, the concentration at the leeward side is always the sum of the direct contribution and recirculated component. σ_w is calculated by the following relationship:

$$\sigma_w = \sqrt{(\alpha u)^2 + \sigma_{w0}^2} \tag{2.8}$$

Where, u is wind speed at the street level, α is a proportionality constant (given empirically the value of 0.1), σ_{w0} is the traffic induced turbulence defined as below,

$$\sigma_{w0} = b\left(\frac{N.V.S^2}{W}\right)^{1/2} \tag{2.9}$$

b is an aerodynamic drag coefficient with an empirical value of 0.3, N is the number of vehicles on the street per unit time, V is the average speed of the vehicle, and S^2 is the road surface occupied by one single vehicle, and W is the width of the canyon.

The inflow of pollutant is equivalent to QL_r/L while the outflow can be separated in diffusion though the upper edge and advection through the side edge. By equating the inflow and the outflow, the mathematical definition of the recirculation component of pollutants can be derived and the expression reads as,

$$C_r = \frac{Q}{W}\frac{L_r}{\sigma_{wt}L_t + U_t L_{S1} + uL_{S2}} \tag{2.10}$$

In Equation 2.10, L_r, L_t, L_{s1} and L_{s2} are dimensions of the trapezoidal recirculation area, and σ_{wt} is the ventilation velocity expressed by the following equation:

$$\sigma_{wt} = \sqrt{(\lambda U)^2 + F_{roof}\sigma_{w0}^2} \tag{2.11}$$

U is the wind speed at the roof level, λ and F_{roof} are proportionality constants whose values are given as 0.1 and 0.4 respectively. The width of the recirculation zone is calculated as:

$$L_r = F_{vortex}H_b r \sin\phi \tag{2.12}$$

Where F_{vortex} is a proportionality constant (set equal to 2); H_b is the height of the canyon; r is a wind speed dependant factor reflecting the strength of the vortex, and ϕ is the angle between the roof-level wind direction and the street axis. The value of r is governed by the conditions below:

$$r = \begin{cases} 1 & if\ U > U_{critical} \\ \frac{U}{U_{critical}} & Otherwise \end{cases} \tag{2.13}$$

$U_{critical}$ is critical wind velocity for the formation of a vortex in a street and is empirically defined as 2m/s (Vardoulakis et al., 2002). The relationship between the roof level (U) and street level wind (u) for a regular canyon is given by:

$$u = U\frac{ln(^{h_0}/_{z_0})}{ln(^{H}/_{z_0})}(1 - F_{wind}\sin\phi) \tag{2.14}$$

Where, z_0 is the surface roughness length of the area under consideration, and F_{wind} an empirical constant is given a value of 0.2. The wind speed at roof level (U) is calculated from the input wind (U_a) which corresponds to a meteorological mast of generally different height, using the simple relationship,

$$U = F_{mast}.U_a \tag{2.15}$$

The empirical parameter F_{mast} is given the value of 0.82, which is derived from a logarithmic law similar to expression (7.11). The geometry and wind flow in the recirculation zone and formation of the recirculation zone are shown in Appendix D.

2.6. Health Impact Assessment

Air pollution is associated with 7 million premature deaths globally every year (WHO, 2014) and is linked to 491,000 deaths in Europe annually (EEA, 2016a). Air quality standards are set by WHO and by the EU to protect air quality and human health. However, 85-91% of the EU urban population is exposed to harmful levels of air pollutant concentrations (WHO, 2013a). The health impacts of air pollution are generally quantified in terms of premature mortality incidences and morbidity. Mortality reflects a reduction in life expectancy due to premature deaths as a result of air pollution exposure. Premature mortality is deaths that occur before a person reaches an expected age. This expected age is typically the age of standard life expectancy for a country and gender (EEA, 2017). Premature deaths are considered to be preventable if their cause can be eliminated. Whereas, morbidity relates to the occurrence of illness and years lived with disability or disease. Morbidity includes not only chronic conditions but also less severe effects because air pollution affects the population on a daily basis and might lead to a worse health condition. Morbidity outcomes include hospitalisation and emergency room visits, asthma attacks, bronchitis, respiratory symptoms, and lost work and school days (WHO, 2004). These impacts are referred as Burden of Disease (BOD) of air pollution and usually measured in terms of Disability Adjusted Life Years (DALY) which is the sum of Years of Life Lost (YLLs) and Years of healthy Life lost due to Disability or Years Lived with Disability (YLDs) (WHO, 2003). YLLs are defined as the years of potential life lost due to premature death. It is an estimate of "the average number of years that a person would have lived if he or she had not died prematurely" (EEA, 2017). YLL is lower for deaths at an older age and greater for deaths at a younger age as it considers the age at which deaths occur. Therefore, it gives more precise information than the number of premature deaths alone. Whereas, YLD reflects the years lived with the disease.

The popular approach to assess these health outcomes is based on RR coefficients. These coefficients reflect the health outcomes for a unit change in air pollutant concentrations. RR coefficients are developed based on meta-analysis in different regions. These are derived from concentration response functions or dose response functions which refer to “the relationship between exposure to pollution as a cause and specific outcomes as an effect” (OECD, 2005). The methods of health risk assessment due to air pollution is well established in Europe (Héroux et al., 2015). There are air quality guidelines designed by WHO to reduce health impacts of air pollution (WHO, 2005). These air quality guidelines are developed based on extensive scientific evidence on air pollution and its health consequences. However, as mentioned, though these guidelines offer reduced health impacts, the concentrations below these guideline values are found to possess health threats. Raaschou-Nielsen et al. (2012) studied traffic air pollution and its health impacts in Denmark and found that change in NO_2 concentration of 10 µg/m^3 showed a significant correlation with health outcomes. The impact of traffic related air pollution on human health in Austria, Switzerland and France was studied by Künzli et al. (2000) using exposure response functions obtained from epidemiological studies. Clancy et al. (2002) examined the effect of air pollution control measures on death rates in Dublin, Ireland and reported that “control of particulate air pollution could substantially diminish daily death”.

2.6.1. Health Impacts of $PM_{2.5}$

Particulate matters could be directly emitted to the atmosphere or formed from NOx, SO_2, NMVOCs and ammonia (EEA, 2014). In EU, total premature mortality incidences attributable to $PM_{2.5}$ exposure was estimated to be 332,000 in 2014 for the baseline concentration of 2.5 µg/m^3 (EEA, 2017). Namdeo and Bell (2005) studied health implications of fine and coarse particulates at different urban and rural sites in the UK. The authors found that fine and coarse particulate matters are strongly correlated in the urban sites but not in the rural sites. The authors also reported that if the modelled concentration levels were prevalent in London, the mortality incidences due to all causes will rise by 1.7% to 2.5%. Li et al. (2017) examined the health effects of $PM_{2.5}$ in Bejing and found that a considerable number of deaths are caused by $PM_{2.5}$ pollution. Walton et al. (2015) had also studied the number of deaths brought forward and YLL due to premature mortality that can be linked to $PM_{2.5}$ pollution in London. Walton et al. (2015) reported that, in 2010, the total number of deaths caused by short-term exposure to $PM_{2.5}$ in London is 818. Mortality burden of fine particulate matter in California was assessed by Ostro et al. (2007) and the authors found that $PM_{2.5}$ is associated with several mortality category, especially cardiovascular deaths. Pope et al. (2002) assessed the relationship between long-term exposure to $PM_{2.5}$ and all-cause, lung cancer, and cardiopulmonary mortality. Pope et al. (2002) reported that an increase in $PM_{2.5}$ concentrations by 10 µg/m^3 is associated with about a 6%, 8% and 4% higher risk of cardiopulmonary, lung cancer and all-cause mortality, respectively. Public health benefits due to reduced $PM_{2.5}$ in Tianjin, China were studied by Chen et al. (2017) using BenMAP. The results revealed that bringing daily $PM_{2.5}$ concentrations down to 75 µg/m^3 (which is the air quality standard in China for residential, industrial, commercial, and rural areas) can reduce cardiovascular and respiratory disease related deaths by 2000 and 280 per year respectively. In APHEIS (Air Pollution and Health: A European Information System) study (Boldo et al., 2006), health impact assessment of long-term exposure to $PM_{2.5}$ was carried out. It was found in that study that 16,926 premature deaths as a result of long-term exposure to $PM_{2.5}$ could be prevented every year if the $PM_{2.5}$ levels are reduced to 15 $\mu g/m^3$ in each of the 23 cities studied. BOD attributable to ambient fine particulate matter exposure was estimated by Lo et. al (2016) based on relative risks.

Brook et al. (2010) documented that exposure to $PM_{2.5}$ over a few hours to weeks can trigger nonfatal events and even, cardiovascular mortality and long-term exposure increases the risk for cardiovascular mortality to an even greater extent than exposures over a few days and reduces life expectancy. Also, reductions in PM levels were found to be associated with a decrease in cardiovascular disease related mortality within a time frame as short as a few years. Devos et al. (2015) found that a 10% reduction in $PM_{2.5}$ levels result in ischemic heart disease and heart rhythm disturbances related annual hospital admissions by 2.34% and 2.08% respectively in Belgium. Arranz et al. (2014) evaluated premature mortality incidences due to long-term exposure to $PM_{2.5}$ in Valladolid, Spain, according to WHO methodology. The authors found that 326 deaths can be associated to long-term exposure to $PM_{2.5}$ in Valladolid city with population of around 3,00,000. Tang et al. (2014) developed human health damage factors for $PM_{2.5}$ on the basis of global transport model. Different damage factors were reported for different parts of the world. An integrated risk function was developed by Burnett et al. (2014) for estimating global BOD attributable to ambient $PM_{2.5}$ exposure. RR functions were developed for all causes, namely, ischemic heart disease, cerebrovascular disease, chronic obstructive pulmonary disease, lung cancer and acute lower respiratory infection. The percent population attributable fraction derived from RR attributable to ambient fine particulate matter pollution exposure varied from 2 to 41 for ischemic heart disease, 1 to 43 for cerebrovascular disease, < 1 to 21 for chronic obstructive pulmonary disease, < 1 to 25 lung cancer and < 1 to 38 for acute lower respiratory infection. Faridi et al. (2018) studied the effect of long-term exposure to $PM_{2.5}$ in Tehran, Iran using AirQ plus software tool and reported that ambient $PM_{2.5}$ has contributed to 7.6% and 11.3% from all-cause annual mortality over 10 years' time period. Several other studies (Fann et al., 2018; Yin et al., 2017; Song et al., 2017; Martinez et al., 2018) were carried out to estimate health impacts of $PM_{2.5}$.

2.6.2. Health Impacts of NOx

As mentioned earlier, exposure to NOx has been linked with a range of serious health effects, including increased asthma attacks and other respiratory diseases (US EPA, 2016a). NOx is currently one of the air pollutants of most concern in Europe (WHO, 2016b). In EU, NO_2 has been found to be responsible for 229,000 premature deaths in 2014 for the baseline concentration of 10 µg/m^3 (EEA, 2017). In urban areas, a major contributor to NOx concentration is road traffic (Lewne et al., 2004). Walton et al. (2015) assessed the health impacts of NO_2 pollution in London. The authors used RR coefficients recommended by WHO Health risks of air pollution in Europe (HRAPIE) project (WHO, 2013b) to calculate mortality burden of NO_2. It was found that the total numbers of deaths brought forward, and respiratory hospital admissions were 461 and 419 respectively due to short-term exposure to 2010 levels of NO_2. César et al. (2015) studied the association between NOx exposure and deaths caused by respiratory disease. The authors reported that a reduction in $PM_{2.5}$ concentration by 3 µg/m^3 results in a decrease of 10-18% in risk of respiratory disease related deaths. Tao et al. (2012) reported that exposure to an increased level of NO_2 concentration by 10 µg/m^3 over 2 days results in associated mortality by 1.95% at 95% Confidence Interval. Long term consequences of NO_2 pollution in major urban areas were assessed by Nguyen et al. (2015). The number of deaths attributable to NO_2 pollution was also estimated by Skouloudis and Kassomenos (2014). It was reported that approximately 4000 deaths could be attributable to atmospheric pollution due to NO_2. Chaloulakou et al. (2008) studied NOx and NO_2 concentrations in Athens, Greece and their subsequent health implications using dose-response relation. The authors have reported that a person suffering from respiratory disease, are very sensitive to NO_2 at high concentrations. It was also mentioned that apart from reducing the direct negative impact of NO_2 on human health, effective implementation of the annual NO_2 standard can possibly indicate a reduction of concentrations of other harmful pollutants resulting from road transport. Namdeo and Bell (2005) also reported that a significant number of respiratory hospital admissions are caused by NO_2 pollution. In Belgium, 10% reduction in NO_2 concentrations lead to savings of annual hospital admissions due to ischemic heart disease and heart rhythm disturbances, respectively, by 3.93% and 3.46% (Devos et al., 2015). Keuken et al. (2012) studied the health implication of primary NO_2 emissions from road traffic in the Netherlands. Association of NO_2 pollution with the risk of mortality in California was examined by Jerrett et al. (2013).

Tang et al. (2015) estimated human health damage factors of NOx using a chemical transport model and reported that one kilotonne of NOx is responsible for 90 DALYs. Krewitt et al. (2001) developed damage factor to quantify human health impacts in terms of YLL attributable to per tonne of NOx emissions within the European population. Wang et al. (2016) calculated the public health impact in terms of premature deaths due to hidden NOx from diesel vehicles in California. A similar study was carried out by Holland et al. (2016) for the overall United States. Oldenkamp et al. (2016) assessed health damage caused by Volkswagen fraud utilising damage factors reported by Tang et al. (2015). The authors found that the health damage caused by excess NOx emissions above the legal limits from faulty Volkswagen LDVs was equal to around 680 DALYs in the US and 44,250 DALYs in EU over the years 2009-2015. Barrett et al. (2015) quantified the human health impacts of excess NOx emissions resulted due to Volkswagen emissions control defeat device. It was reported that 59 premature deaths in the US over the period 2008-2015 were caused by excess NOx emissions from Volkswagen vehicles fitted with emissions control defeat device.

2.7. Cost Impacts of pollution

As mentioned in Chapter 1, health issues resulting from exposure to air pollution not only have a high cost to society and business but also to the people who suffer from related illnesses and premature deaths (RCP/RCPCH, 2016). Emission from transport can result in local air quality problems and health impacts. High levels of pollutions also have the potential for secondary impacts on water quality, the built heritage and nature conservation resources (DTTaS, 2016a). Common Appraisal Framework for Transport Projects and Programmes (DTTaS, 2016a) for Ireland suggests marginal values of emissions to monetise the impacts of pollution and this includes damage costs of CO_2, $PM_{2.5}$, PM_{10}, NOx and VOC. It suggests that, in an urban environment, one tonne of $PM_{2.5}$ emission causes a damage worth €200,239.

Handbook on estimation of external costs in the transport sector (2008) reports air pollution costs derived based on impact pathway approach using resource cost and willingness to pay for human life (life years lost). All damages of environmental nuisances, such as health costs, material damages, crop losses, biosphere damages, long term risks are included in these costs. The cost function used to derive these damage costs is correlated to location, traffic volume and emission levels. Emissions and exposure data are needed to estimate these costs. The main cost drivers are population and settlement density, the sensitivity of area and level of emissions, depending on, type and condition of the vehicle, trip length (cold start emissions), and type of infrastructure, location, and speed characteristics. Figure 2.8 shows the method followed to quantify the damage costs of emissions.

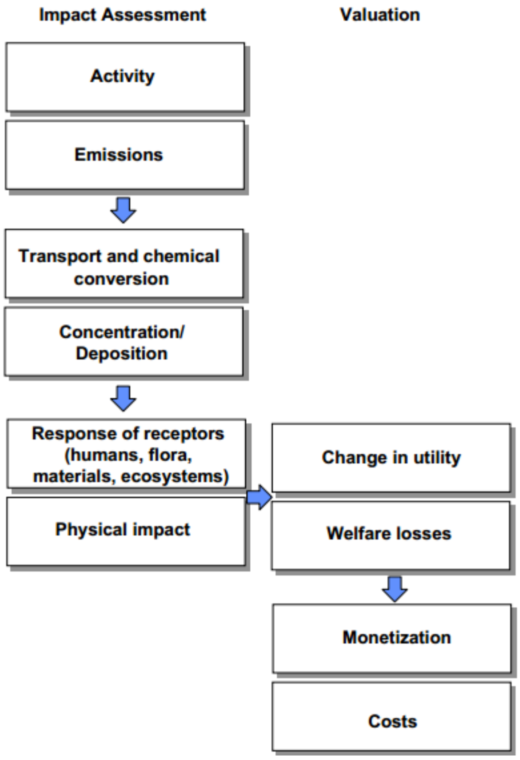


Figure 2.8. The Impact Pathway Approach for quantifying marginal external costs caused by air pollution (HEATCO, 2005)

In the UK, the annual cost of air pollution sums up to more than £20 billion (RCP/RCPCH, 2016). Yin et al. (2017) studied external costs of fine particulate pollution in Beijing, China and the authors reported that among all the health impacts, the financial loss due to premature deaths accounted for more than 80% of the overall external costs. Bai et al. (2018) conducted a comprehensive review of health costs of air pollution in China and also pointed out how the research on air pollution related health cost accounting has increased drastically in recent years. Pervin et al. (2008) reviewed the societal costs of health hazards due to air pollution from several research. In Belgium, the potential cost savings due to the reduction in air pollution in relation to cardiovascular disease were studied by Devos et al. (2015). The authors found that reduced risk of ischemic heart disease due to a 10% reduction in $PM_{2.5}$ and NO_2 can lead to cost savings of €5.0 million and €8.4 million, respectively. Whereas, for heart rhythm disturbances, 10% reductions in $PM_{2.5}$ and NO_2 levels result in a potential annual hospital cost savings of €3.6 million and €5.9 million respectively. Shen et al. (2017) evaluated damage costs due to respiratory diseases under severe air pollution and reported that the total medical cost of clinic visits for respiratory diseases derived from $PM_{2.5}$ pollution alone accounts for 0.5 to 1.6% of national total health expenditure in China.

Quah and Boon (2003) studied economic costs of health due to particulate air pollution. Based on the findings of the research, the authors stated that the environmental costs should not be ignored by the policy makers "in their pursuit for economic progress". Robinson and Hammitt (2015) examined VSL through research synthesis methods, including systematic review, meta-analysis, and expert elicitation. The authors reported VSL to lie between 7.6 million to 13.7 million US dollar (Robinson and Hammitt, 2015). Oldenkamp et al. (2016) evaluated the cost of human health damage from excess emissions due to VW fraud. They reported that over the period of 2009 to 2015, the faulty vehicles have caused damage worth 39 billion US dollars. Martinez et al. (2018) assessed economic costs of air pollution in the metropolitan area of Skopje and estimated a social cost of between €570 million and €1,470 million in 2012 due to premature mortality and hospital admissions for cardiovascular and respiratory diseases attributable to $PM_{2.5}$ pollution. Mauzerall et al. (2004) carried out an exploratory analysis to suggest "externality-correcting" prices to NOx emitters for health damages. The authors mentioned that "Charging emitters fees that are commensurate with the damage caused by their NO_x emissions would create an incentive for emitters to reduce emissions at times and in locations where they cause the largest damage."

2.8. Scope of the Research

Based on the literature review, a number of the research gaps were identified and this research aims to address these in the Irish context (a) by identifying potential sources of uncertainty in emission estimates from road transport; (b) by evaluating alternative policy measures with a focus to reduce diesel use and increase EV uptake and further, examining their potential in reducing the emission levels and meeting Ireland's GHG emission goals for 2030; (c) by assessing urban air quality as a result of road transport pollution; (d) by estimating health and cost impacts attributable to urban air pollution due to road transport emissions.

There have been several previous studies conducted to estimate the uncertainty associated with outcomes of models calculating emissions from road transportation. However, in Ireland, no attempt has been made so far to estimate the uncertainty associated with its reported NEIs. As mentioned in Section 2.2, considering the major applications of NEI it is very important to be aware of the sources of uncertainty and level of possible uncertainties. In this research, the NEI for PC fleet in Ireland (similar ones used in other countries) were calculated based on the available input data for COPERT in Ireland at this moment. Then the possible variations in the COPERT emission estimates with the possible variations in input parameters were examined and uncertainty in emission estimates was characterised by fitting PDFs. The detailed methodology and results regarding this work are presented in Section 3.3.1 and Chapter 4, respectively. As also mentioned in Section 2.2, many researchers especially in the US carried out research to quantify the impacts of dieselgate in terms of excess NOx emitted from vehicles fitted with emissions cheating device. It is very important for policymakers to know the potential impacts of dieselgate in Ireland for effective policy making. This information is not available for Ireland. Therefore, this thesis quantifies the potential environmental, health and economic impacts of dieselgate in Ireland and thereby, points out the importance of mitigating the issue. The details of the methodology and results are presented in Section 3.3.2 and Chapter 5, respectively.

As mentioned in Chapter 1 and earlier in this chapter, it has become absolutely necessary to shift towards low or zero emission transport modes due to increase in transport demand, fossil fuel depletion and increasing levels of emission levels. Many researches have been carried out to examine possible ways to reduce emission levels and what measures and policies can be implemented that will help increase the uptake of alternative travel options. No such studies have been carried out in Ireland that gives an overall evaluation of the expected levels of emissions in the future, potential reduction in emission levels with changes in fleet composition and how new policies can be introduced that will help to achieve this. This research gap will be addressed in Chapters 3, 6 and 7. Several studies have been conducted studying the environmental and health impacts of air pollutants. However, there is a gap in literature studying the relationship between emissions and concentrations and further examining their economic impacts. Especially due to dieselgate which resulted in very high levels of NOx, it is anticipated that they will result in huge health impacts including premature deaths. However, there has not been any study actually assessing if these high NOx emissions are causing high concentrations too beyond WHO/EU recommended safe limit. These research gaps are addressed in Chapters 3, 8 and 9.

2.9. Research questions and thesis overview

The focus of this thesis is built upon the research questions described below. Broadly, this thesis can be divided into four areas which are interconnected and findings from one chapter form the context or basis of the subsequent chapters. The four research areas are, vehicular emissions modelling, urban air quality modelling, health impact assessment and cost impact assessment. This research aims to first examine the current emission levels in Ireland from road transport fleet using the methodology and parameter values used in emissions inventory preparation in Ireland, based on which policies and strategies are made in Ireland. Then examine the possible uncertainty associated with the estimates which may have led to wrong emissions reporting and may continue to report wrong emissions and thereby, affect the policies and strategies aimed at reducing emissions and their impacts. Two possible sources of uncertainties were identified, one is input parameter related uncertainty, and another is NOx emissions related uncertainty due to dieselgate. The possible uncertainties in emission estimates were found to be high for some pollutants and the results provide useful information for better planning, modelling and policy making. Then based on Ireland's emissions goal for 2030, several scenarios were designed with an increased share of low emission fuel and technology options that will potentially help to meet the emissions target. The findings of this study indicated that due to the increase in car ownership level, huge electrification will be necessary to meet the 2030's emissions target. It was also found that the targets can only be met if both car ownership level reduces and the EV share increases. It was identified that emission levels can be reduced in 2030 compared to the expected levels under BaU situation if additional policy measures are implemented. The findings of this research provide policy makers with information on expected future emission levels, the potential of different shares of fuel and technology options in mitigating emission levels, and a realistic way to meet the GHG emissions target for 2030. As mentioned, the findings showed that the emissions can be significantly reduced if additional policies are implemented. Therefore, a set of policies was proposed such that they not only reduce diesel use in road transport but also increase EV uptake, and also generate funds for the government to invest in improving EV infrastructure. The next step of the research looks at the impacts of these emissions on air quality, public health and economy based on the finding on current emission levels, expected an increase in emissions in 2030 under BaU and possible reduction of those levels with new policy implications. Air pollution levels of NOx and $PM_{2.5}$ due to emissions in the current scenario, and in 2030 with BaU situation and with additional policy measures were investigated in all the major streets in Dublin city. It was found that even though overall concentrations are below the WHO and EU specified guideline values, there are some areas with noxious pollution levels. Premature mortality incidences and economic value associated with the total premature deaths attributable to long-term exposure to that NOx and $PM_{2.5}$ pollution levels were calculated. These health and economic impacts were also calculated based on the NOx and $PM_{2.5}$ concentrations recorded at the monitoring locations in Dublin city. The findings showed that concentrations recorded at air quality monitoring stations are lower than the modelled values, as a result, the overall impacts are underestimated. Therefore, air quality should be monitored at different locations at population exposure level. In addition, all these impacts were spatially distributed which is very important for better policy making and in prioritizing attention and investments in the more affected areas.

Emission modelling was performed using COPERT (COmputer Programme to calculate Emissions from Road Transport) which is a road transportation emissions model popularly used in research and in preparing National Emissions Inventory (NEI). Emissions at the street level were modelled using COPERT Street Level. Environmental impacts were assessed by modelling urban NOx, NO_2 and $PM_{2.5}$ concentrations at population exposure level on all the major streets in Dublin city using the dispersion model, OSPM (Operational Street Pollution Model). Health impacts in terms of the number of premature deaths due to long-term exposure to NO_2 and $PM_{2.5}$ were calculated using the WHO recommended methodology based on Relative Risk (RR) coefficients (WHO, 2004, 2013). Economic Impacts were evaluated in terms of damage costs of pollution based on Common Appraisal Framework for Transport Projects and Programmes in Ireland by the Department of Transport, Tourism and Sports (DTTaS, 2016a), Ireland and in terms of Value of Statistical Life (VSL) lost due to premature death attributable to long-term exposure to $PM_{2.5}$ and NO_2 pollution based on WHO (WHO, 2015) reported value for Ireland.

The research questions examined in this thesis can be as listed below,

- What is the level of uncertainty associated with national emissions inventory estimates with respect to variation in input parameters?
- What is the impact potential impact of *dieselgate* in Ireland in terms of excess NOx emissions and their subsequent health impacts?
- What is the expected future emission levels from road transport in Ireland if no additional measures are taken upon the existing ones?
- How the expected emission levels can be mitigated with additional policy measures?
- How Ireland's GHG emissions goal for 2030 can be met?
- How is the current urban air quality in Ireland in terms of NOx and $PM_{2.5}$ pollution attributable to road transport, and how will be the urban air quality in 2030 under BaU situation, and how the air quality can be improved with additional policy measured?
- Do high NOx emission levels above Euro standard legal limits lead to NO_2 concentrations exceed the WHO/EU recommended safe guideline value?
- Do the modelled NOx emissions have any relationship with NOx or NO_2 concentrations?
- What are the health impacts due to exposure to NOx and $PM_{2.5}$ concentrations in the present and what are the expected health impacts due to their expected levels in 2030 and the possible reduction in impacts as a result of new policy measures?
- What are the economic impacts due to damage caused by pollution from road transport, and what are the expected impacts from 2030 road traffic under BaU, and potential savings in costs in 2030 with additional policy implications?

Chapter 3: Methodology

3.1. Introduction: Modelling methodology

This chapter describes the methods used in this research to model the emission levels from road transport and their environmental, health and economic impacts. In addition to the description of the methods and software used in the modelling, the input data required, and outputs are also presented in this chapter. As mentioned earlier, emissions from the road traffic in Ireland were modelled using COPERT 5 at the network level. Therefore, the overall annual traffic was considered in modelling the network wide emission levels and their subsequent impacts. This provides overall vehicular emissions per year but does not give emissions at a finer spatial resolution such as on a particular area or street. These emission levels were calculated for the baseline scenario which refers to the emissions coming from 2015 fleet. At the network level, emissions were also modelled for several other scenarios in order to examine the potential increase in emission levels from the future road transport fleet with continuation of existing policies and potential reduction in emission levels due to alternative fleet composition with new policy measures. These emission levels were also modelled using COPERT 5. Then the potential impacts of current emission levels and of emissions from alternative scenarios were estimated following the methods identified from the literature review. Health impacts from network level emissions were evaluated based on the damage factor approach. These health impacts are expressed in terms of DALYs. Cost impacts were estimated using the damage cost values provided for each pollutant in Common Appraisal Framework for Transport Projects and Programmes for estimating the overall damage cost of emissions. The overall methodology used to calculate network wide emission levels and their economic and health impacts are shown schematically in Figure 3.1.

Now, in order to examine the environmental impacts of vehicular emissions, air pollution levels were modelled by calculating the pollutant concentrations at the population exposure level. The concentrations were modelled using OSPM at the street level. To understand how much emissions are causing how much change in the air quality at population exposure level, vehicular emissions were also modelled at the street level for the same locations where concentrations were modelled. The emission levels at the street level were estimated using COPERT Street Level software which uses the similar fundamental methodology as COPERT 5 except the fact that it takes into account the hourly traffic flow and speed in each road link. Therefore, it gives emission estimates at the finer level and areas with higher emission levels can be identified. Detailed description of the methodologies is discussed in the following section in this chapter.

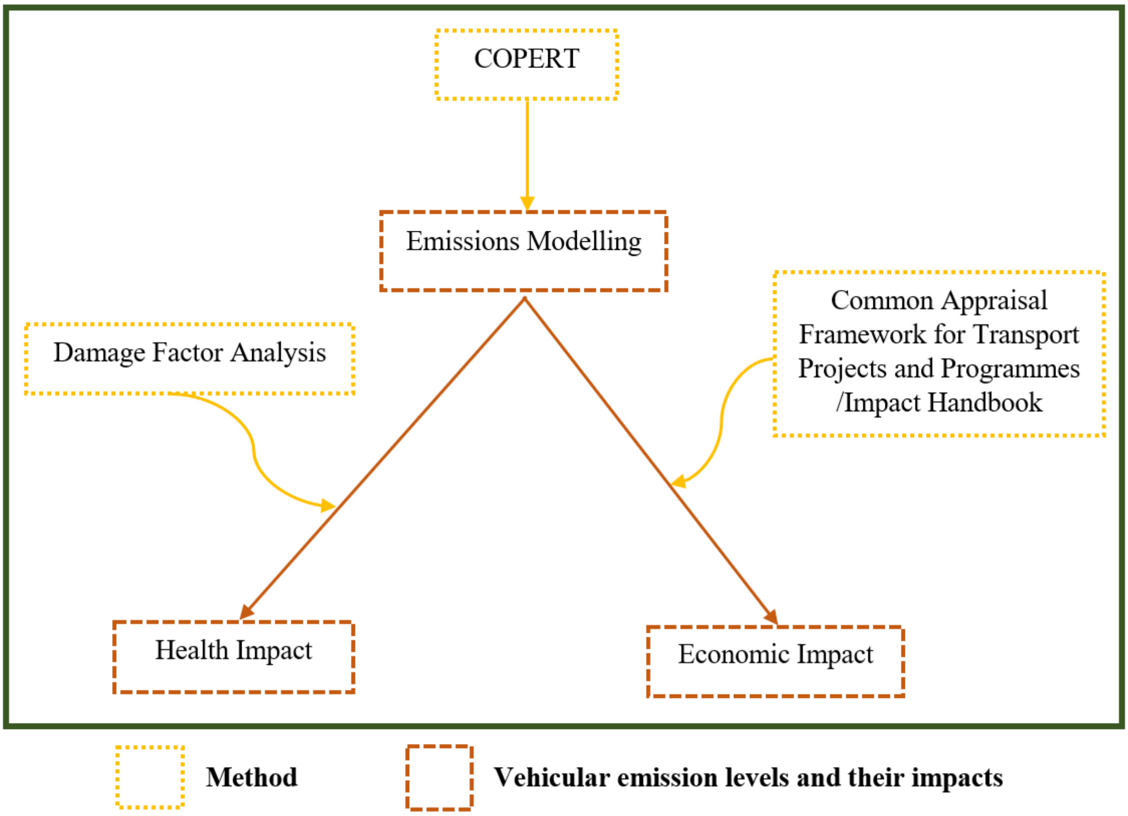


Figure 3.1. **Methodology used to calculate network level emissions and their health and economic impacts**

Thereafter, the concentrations were predicted at other areas using kriging interpolation tool in GIS, based on the modelled concentrations from OSPM. Then spatial variation maps of pollutant concentrations were produced which also helps in identifying the areas exposed to higher pollution levels than the WHO recommended safe guideline values. From these modelled concentration levels, the pollutant concentrations in the electoral districts were estimated to be able to calculate the health impacts of air pollution in each electoral district based on WHO's BOD method. The outcomes of health impact assessment following the WHO BOD method is premature mortality as a result of long term exposure to air pollution attributable to road transport. Then the economic impacts were calculated using the health impact estimates i.e. premature mortality incidences. WHO recommended VSL value for Ireland was used to calculate this economic impact.

This methodology provides an overall step by step process of calculating emissions and the potential concentration changes due to that compared to the safe permissible values at the exposure level. Then from concentration changes, it calculates the health impacts, and further, based on that cost impacts were calculated. Therefore, this methodology offers a new approach of calculating emissions from road transport and their potential impacts and is one of the major contributions of this thesis. Figure 3.2 shows the steps of the street level emissions modelling and impacts modelling methods.

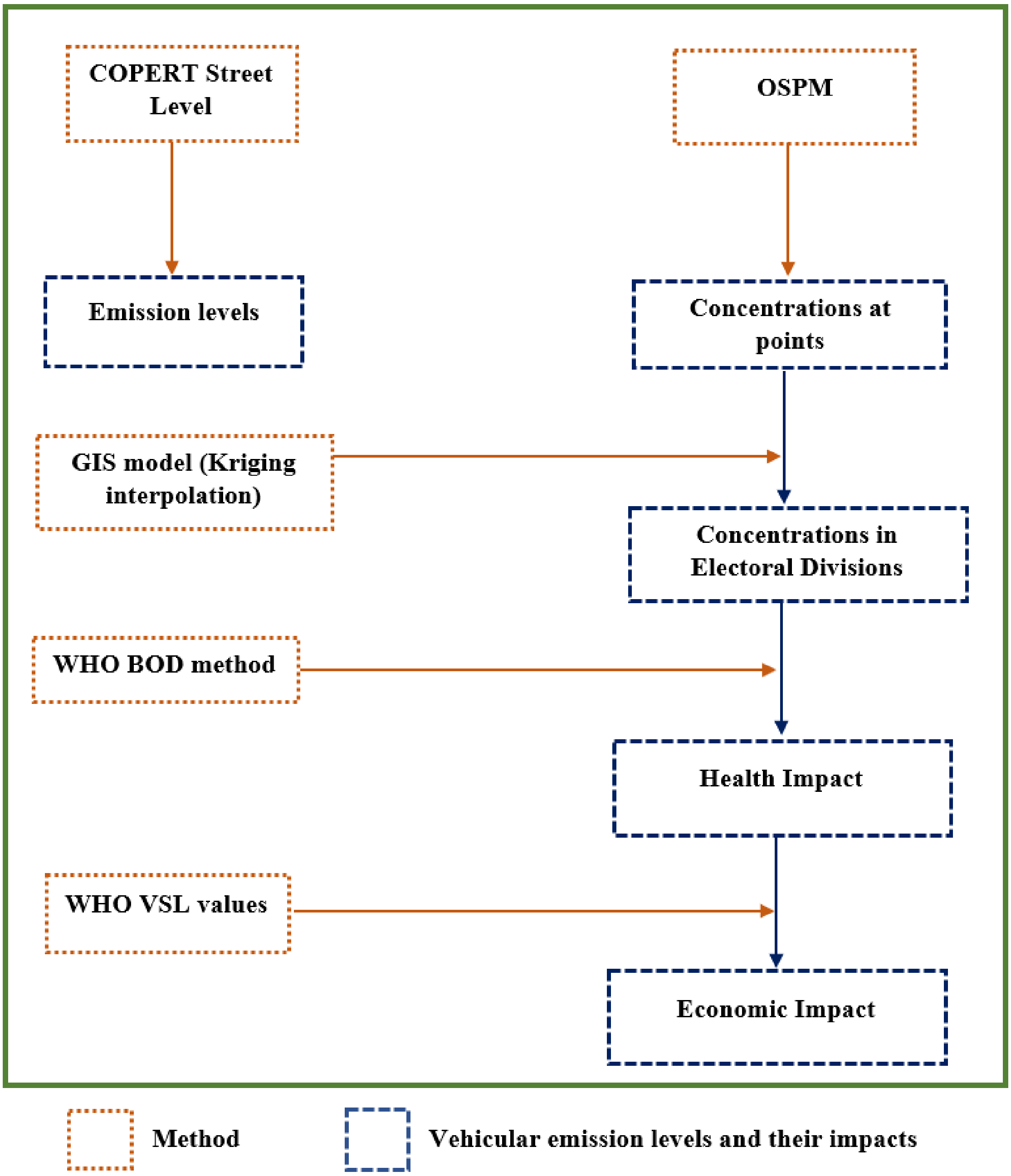


Figure 3.2. Methodology used to calculate street level emissions and their environmental, health and economic impacts

The street level emissions and potential impacts were not only calculated for the baseline scenario but also were calculated for 2030 under business as usual scenario to examine the potential increase in the emission levels and air pollution levels, health consequences and impacts on the economy due to that. In addition, potential reduction in emissions and impacts in 2030 as a result of alternative fleet composition were estimated following the same methodology. In the following sections, the detailed description of all the methodologies are presented along with their fundamentals, inputs required, and outputs of the software and methods used to model both network wide and street levels emissions and their impacts.

3.2. Modelling tools

This section describes in details the software tools and methods used to model emissions from road transport and to assess their impacts on environment, health and economy.

3.2.1. COPERT 5

As mentioned earlier, COPERT 5 was used to model emission levels from road transport in Ireland. COPERT algorithm and model details were discussed in Chapter 2. This section describes the input data required for COPERT to calculate emissions from road traffic and the model outputs. Figure 3.3 shows the input data required by COPERT 5 to calculate emissions.

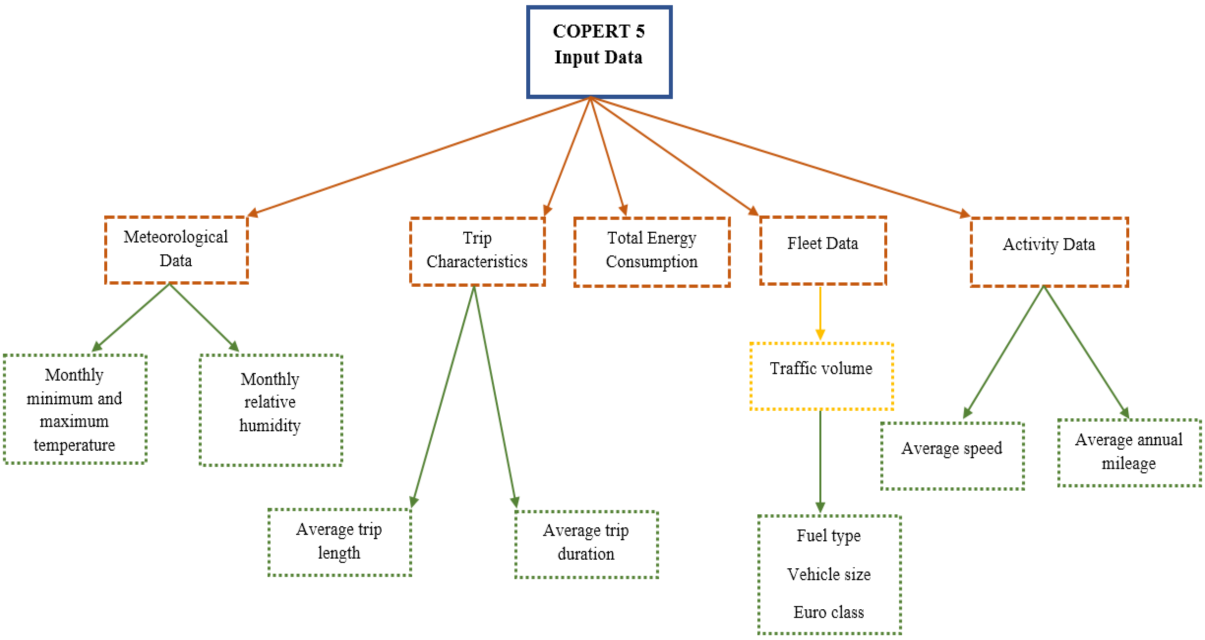


Figure 3.3. Input data required by COPERT 5

Some of the data that as shown in Figure 3.3 are mandatory and some are optional to be provided as input. The details of the main input parameters (Gkatzoflias et al., 2012) are described as follows,

- Country specific information – Name of the country, the year under study, monthly average minimum and maximum temperature (degree centigrade), monthly RH (%). In the environmental information form, the user is required to provide values for monthly average minimum and maximum temperatures and monthly RH. This is required to calculate the load of air-conditioning and fuel evaporation. Ambient temperature and RH affect air conditioning and therefore emission levels. The annual average trip length and average trip duration are to be provided also. These are the mandatory set of data.

- Energy consumption – Total annual energy consumption for each fuel type is required in Terajoule (TJ), however, this is an optional input depending on the accuracy required in the emission estimates.
- Fuel type – The fuel options available in COPERT are, leaded and unleaded petrol, diesel, LPG, CNG, biodiesel and bioethanol. Number of vehicles in each category need to be provided and this a mandatory information.
- Fleet configuration – Number of vehicles of each engine size and legislation standard class needs to be provided. Every class (sector) of vehicles, including passenger cars, LCVs, trucks, Buses, mopeds and motorcycles have sub-sectors on the basis of engine size, legislation standard and fuel type. The default combinations can be deleted, and a new vehicle sector can be added if all the sub-sector information is known. Hence the user's specific combination can be created depending on the data availability. Additionally, fleet data (annual mileage, mean fleet mileage), circulation data (average speed and driving share in each travel mode, i.e., urban, rural and highway) and evaporation data (Fuel tank size, canister size, the percentage of vehicles with fuel injection) need to be provided. Mean fleet mileage (km) is the average cumulative distance travelled by each type of vehicles since their introduction in the market (Gkatzoflias et al., 2012). This value helps to calculate an emission degradation factor depending on age. Tank size (litre), canister size (litre), the percentage of vehicles with fuel injection and evaporation control also are to be provided. COPERT has a default set of data for these evaporation related parameters and the default data can be used in the absence of country specific data.

The output emissions from COPERT 5 can be obtained in the form of cold start emission and hot exhaust emission i.e. source oriented and in the form of urban, rural and highways emissions i.e. driving mode oriented. COPERT 5 estimates emissions of all major (CO, NOx, VOC, PM, NH_3, SO_2, heavy metals) air pollutants as well as greenhouse gas emissions (CO_2, N_2O, CH_4) produced by different vehicle categories (PCs, LCVs, heavy duty trucks, buses, motorcycles, and mopeds). COPERT also provides speciation for NO/NO_2, elemental carbon and organic matter of PM and NMVOCs, including polycyclic aromatic hydrocarbon and persistent organic pollutants. The outputs i.e. emission quantities are obtained in tonnes. Table 3.1 summarizes the importance of input parameters used in COPERT in the correct estimation of emission levels and issues with their availability as reported by Ntziachristos et al. (2008). However, it does not include all the parameters, such as, environmental parameters which has influence particularly on cold start emission and trip characteristics.

Table 3.1. Importance and availability of statistics of different parameters (Ntziachristos et al., 2008)

Parameter	Importance	Notes/Particular Issues with availability
Number of vehicles per class	High	Question is the scooter and mopeds registration availability
Distinction of vehicle to fuel used	High	Question is the availability of records for vehicles retrofitted for alternative fuel use
Distribution of cars/motorcycles to engine classes	Medium	Not important for conventional pollutants, more important for CO_2 emission estimates
Distribution of heavy duty vehicles to weight classes	High	Vehicle size important both for conventional pollutant and CO_2 emissions
Distinction of vehicles to technology level	High	Imported, second-hand cars and scrappage rates are an issue
Annual mileage driven	High	Can be estimated from fuel consumption. The effect of mileage with age requires attention.
Rural, highway driving speeds	Less	Little effect on the EFs as their expected range of variation is given
Urban driving speed	Medium	Affects the EFs
Mileage share in different driving modes	Less	Little effect on emissions within their expected range of variation

3.2.2. COPERT Street level

Emissions at street level i.e. in road segments were calculated using COPERT Street Level. COPERT Street Level is an emission estimation software developed by EMISIA. This is a stand-alone software which can calculate emission at the hourly basis and even at a small road level and applicable for all EU countries. It can calculate emissions of regulated, non-regulated pollutants and greenhouse gases (CO, CO_2, PM, NOx and VOC). COPERT Street Level version

2.4 was used in this study to calculate emissions on the road segments. The methodology of COPERT Street Level is based on COPERT but it is structured to work alongside a traffic analysis tool. When creating a new project, the hourly input data assembled in excel files are loaded into the software. Basic input data include hourly traffic volume, the average speed of traffic on each road link, and length of each segment. Coordinates of each road also can be provided by the user to visualise the emissions on GIS maps. Traffic volume can further be disaggregated into fuel type, vehicle size and euro class for each vehicle type. Figure 3.4 shows the flowchart of required input data by COPERT Street Level and its outputs. COPERT Street Level calculates NOx emissions but does not give separate estimates for NO_2. Similarly, it provides an overall estimate for particulate matter and does not distinguish between $PM_{2.5}$ and PM_{10}. Therefore, the street level emissions of NO_2 and $PM_{2.5}$ were not calculated in this study.

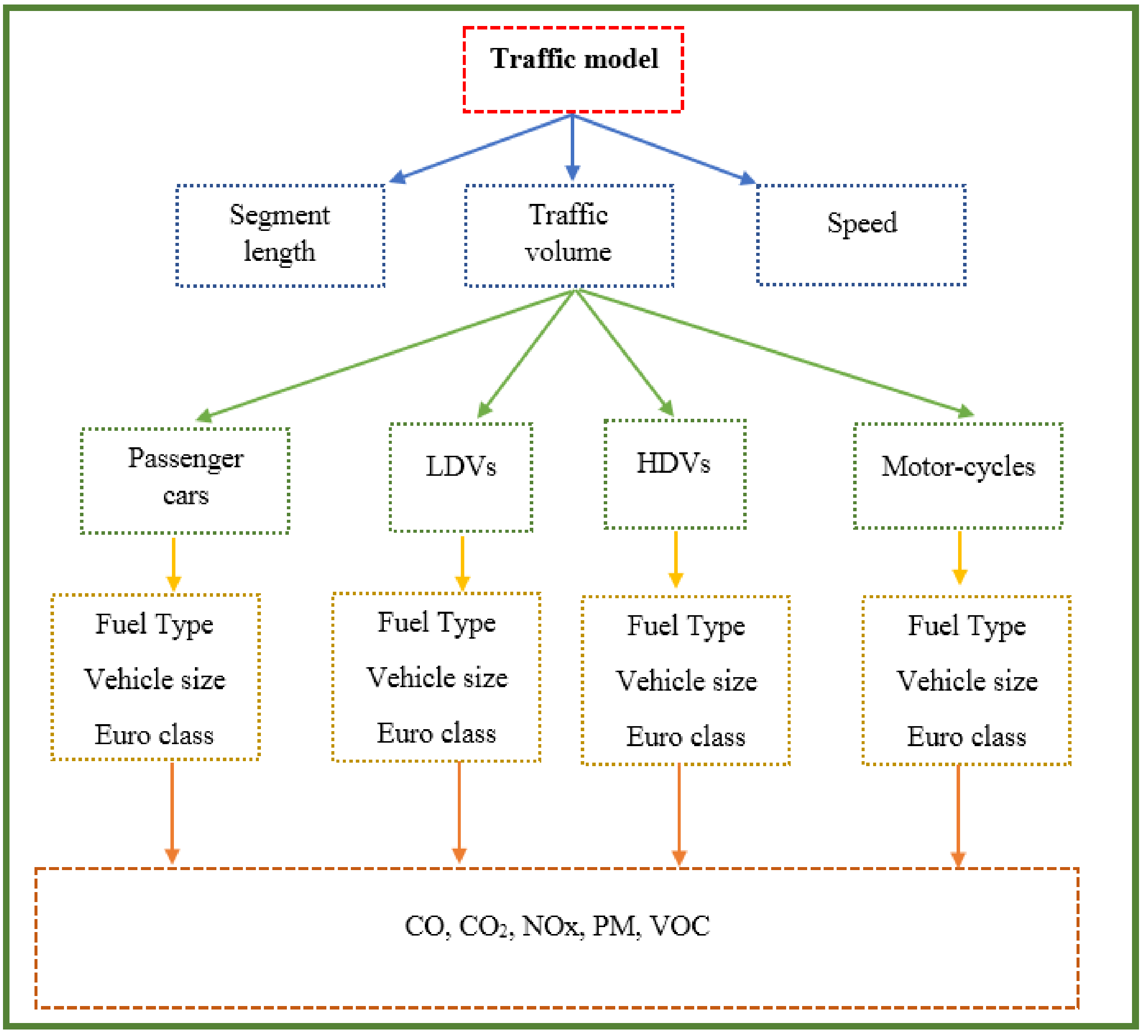


Figure 3.4. COPERT Street Level model flowchart

3.2.3. OSPM

As mentioned in Chapter 2, OSPM was used to assess the environmental impact of vehicular emissions by modelling pollutant concentrations. This section presents the methodology followed in OSPM and the necessary input data for OSPM. The parameterization of flow and dispersion conditions in OSPM was developed based on extensive analysis of experimental data and model tests (Berkowicz et al., 1997). OSPM calculates the concentration of exhaust gases based on a combination of a Gaussian plume model for direct contribution and box model to take into account the recirculation of the pollutants. The details of these models are discussed in Chapter 2. In street canyon, a wind vortex (as shown in Figure 3.5) is formed such that the direction of the wind at street level is opposite to the flow above roof level. Because of this characteristic, the emitted pollutants from the road traffic are transported towards the leeward side while the windward side is mainly exposed to the recirculated pollutants and background pollution in the street (Berkowicz, 2000). Windward side is defined as the side towards which the roof wind blows, and the leeward side is the side from which the roof wind blows (Buckland et al., 1998).

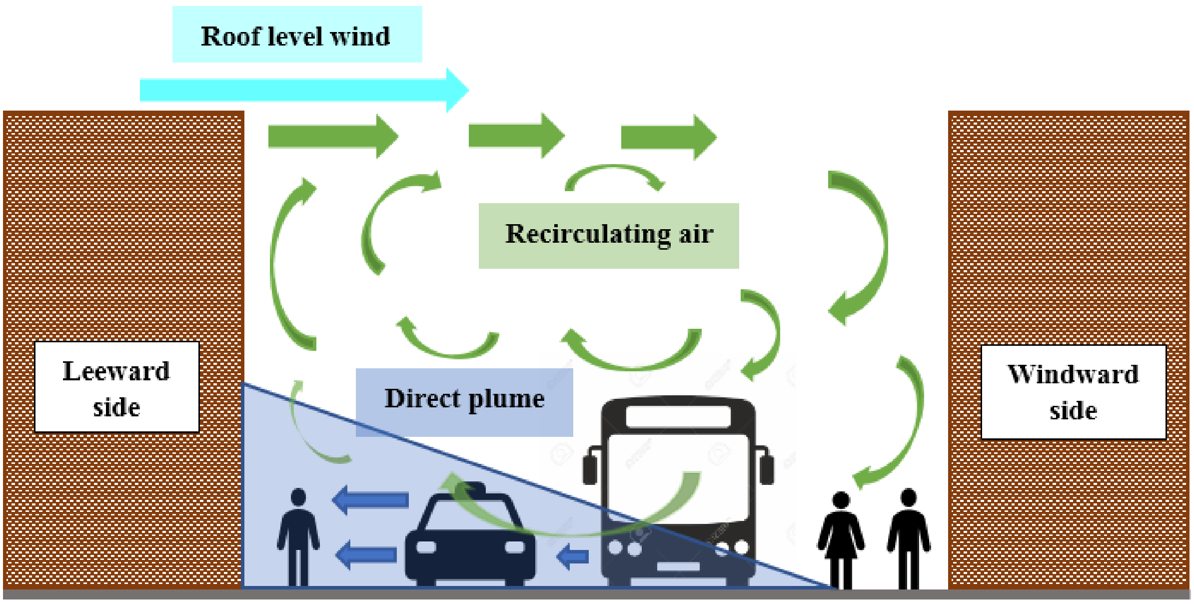


Figure 3.5. Schematic illustration of flow and dispersion conditions in street canyons

OSPM model development includes the following assumptions (Berkowicz, 2000):

- The vortex is formed in the street whenever a wind component perpendicular to the street axis exists. The length of the vortex is calculated along the wind direction and is two times the upwind building height. The length of the vortex decreases gradually with the wind speed when the roof level wind speeds are below 2 m/s.
- The upwind/leeward side receptor receives a contribution from the traffic emissions within the area occupied by the recirculated pollution and a portion of the emissions from outside of the vortex area. This portion is modelled by a factor, R, which is dependent on

wind speed and wind direction. This factor varies continuously from zero to one as the wind direction changes to 0 to 90 degrees with respect to the street axis.

- The downwind i.e. windward side receptor mainly receives contributions from the recirculated component. But if the whole street is occupied by the vortex, the traffic emissions from outside of the recirculation zone also contribute.
- When the wind is parallel to the street or the wind speed approaches to zero, concentrations on both sides of the street become equal.
- To calculate the direct contribution from within the vortex, a plume model which assumes linear dispersion of pollutants with the distance is used. It is assumed that traffic emissions homogeneously distributed across the street.
- The recirculation of pollutants is defined by a box model and concentrations are calculated assuming equality of the incoming and outgoing pollution flux. The incoming flux is the traffic emission while the outgoing flux is governed by the turbulence at the top of the street.

An important feature of OSPM is modelling of the turbulence in the street. The turbulence in the street is assumed to be composed of two parts, one that occurs due to traffic and the other is generated due to the wind speed. Traffic induced turbulence is more when the wind speed is low, as wind meandering affects the relation between wind direction and pollutant concentration in the street canyon. Therefore, in OSPM, the wind meandering is taken into account by averaging the calculated concentrations over a wind speed dependent wind direction range.

OSPM is a parameterised semi-empirical model (Berkowicz, 2000) and needs substantial numbers of input data which can be broadly classified as shown in Figure 3.6. Details of these input parameters are described in the following sections.

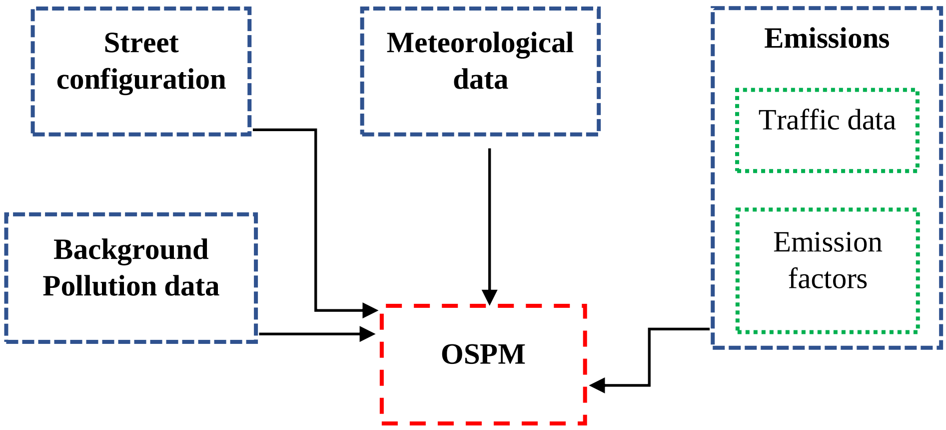


Figure 3.6. Required inputs by OSPM

Street configuration:

OSPM uses detailed street geometry and building data, which include road coordinates, height and width of the road segment, the distance between the centre of the road and the receptor, road orientation with respect to the north, the height of receptor, the height of the buildings on the sides of the roads. These data are to be provided for each road segment separately (See Appendix D).

Traffic Data:

OSPM has an inbuilt traffic editor programme called TrafEdit which allows the user to create and define traffic characteristics on each road link. Annual average daily traffic is provided for each road segment while creating the street collection file. Each street can be opened as a street configuration window where the street data can be edited. Hourly traffic composition is to be defined in terms of either number of vehicles or fraction of daily total or fraction of all vehicles. In this work, the predefined traffic file “Type_C” is used to disaggregate the hourly traffic volume on each road section as shown in Figure 3.7. There are five classes of vehicle type, i.e. Passenger car, Van, Bus, Truck_1 (<32t) and Truck_2 (>32t) that can be defined in the OSPM software as shown in Figure 3.7.

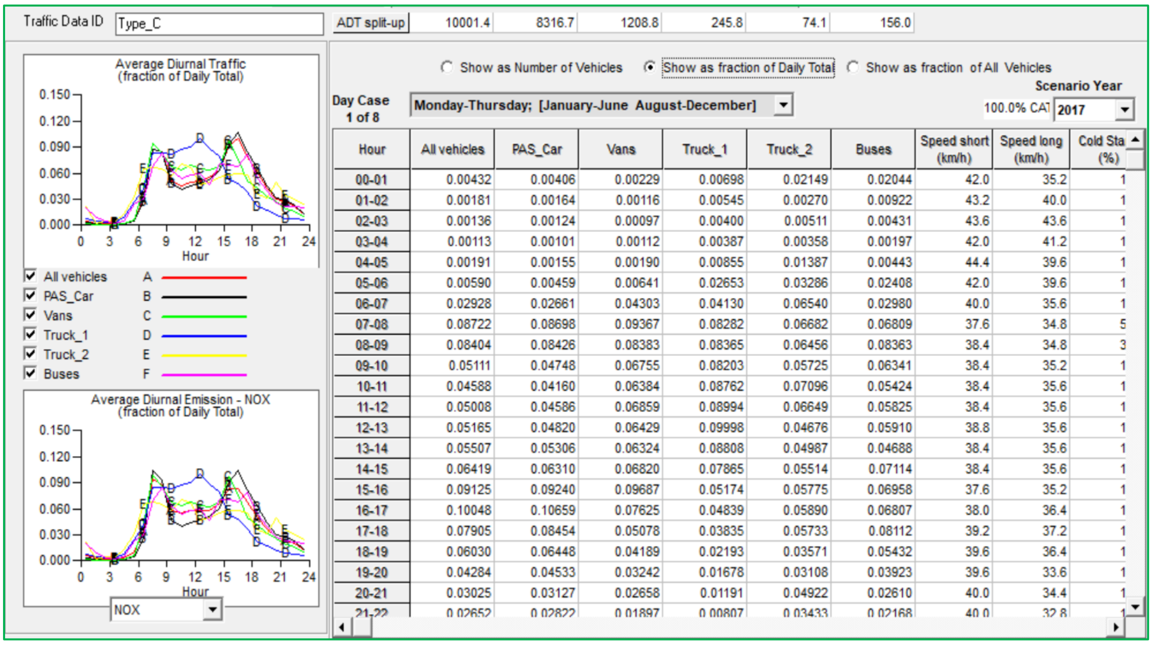


Figure 3.7. Type_C traffic composition

In TrafEdit, average hourly speed over short distances (Speed short) and long distances (Speed long) can be defined. Cold start driving percentages, i.e. the percentage when the engine is cold relative to its normal operating temperature after starting from a stationary position also can be defined. These traffic compositions can be defined separately for the following cases,

- Day Case 1: Monday-Thursday; [January-June August-December]
- Day Case 2: Friday; [January-June August-December]
- Day Case 3: Saturday; [January-June August-December]
- Day Case 4: Sunday; [January-June August-December]
- Day Case 5: Monday-Thursday; [July]
- Day Case 6: Friday; [July]
- Day Case 7: Saturday; [July]
- Day Case 8: Sunday; [July]

In Figure 3.7, the traffic data is shown for Day Case 1.

Traffic Emissions:

Emission factors for each vehicle class based as per vehicle type, fuel type, engine size and euro class can be defined in a program EmiFact which is integrated into OSPM. These emission factors are linked to user defined vehicle list and fuel list. In vehicle list file, a fraction of vehicles as per fuel type and engine size and engine technology class were provided. In the fuel list file, fuel compositions for petrol, diesel and LPG were defined with corrections, if any to be incorporated in calculating emission factor. The methodology to calculate emission factors in OSPM is based on COPERT. Effects due to mileage degradation and cold start correction also are considered (See Appendix D).

Meteorological and background data:

OPSM requires hourly meteorological and urban background data. Essential meteorological data are temperature, global radiation, wind speed, and wind direction. Wind speed and wind direction are assumed to represent the conditions above roof level in the study area. Temperature, global radiation and O_3 are used to take into account the chemical transformation between NO and O_3 and NO_2. The urban background data represents the general background pollution level in the city as measured above roof level.

The background concentration of O_3 is required to take into account the chemical transformation between O_3 and NO_2. The major portion of NOx emitted from motor vehicles consists of NO and only a small amount is in the form of NO_2. Transportation and dispersion are not the only processes determining the relation between emissions and ambient concentrations. Chemical transformation of pollutants plays a major role in the degradation of some pollutants and formation of some others. Now, in the case of street canyons, all these chemical changes are not important in determining the concentration in that street as the distance between the sources and receptors are short and therefore, only pollutants transformed from fastest chemical reactions can have a significant influence. CO and many hydrocarbons are categorized as inert on this scale (Berkowicz, 2000). However, this situation is different for NO_2, which is considered to have a severe impact on human health. The major part of NO_2 in the ambient air is mainly formed by chemical oxidation of NO by O_3 as shown in the equation below.

$$NO + O_3 = NO_2 + O_2 \tag{3.1}$$

The time requirement of this reactions is tens of seconds, therefore are of importance in assessing air quality in street canyon. Modelling of this process is incorporated in OSPM by utilising an analytical solution of the kinetics of reaction scheme (Berkowicz et al., 1997).

3.3 Estimation of Health Impacts

This section presents the methodology used in this study to calculate health impact of pollution. The health impacts of NOx and $PM_{2.5}$ emissions were calculated using the damage factors approach from total quantity of emissions and also using WHO BOD method from pollutant concentration as mentioned in section 3.1. In life cycle assessment, damage factors of air pollution are expressed as the amount of DALYs occurring due to emitted units (e.g. kg) of an air pollutant (Tang et al., 2015; Van Zelm et al., 2008; Hofstetter, 1998). DALYs represent the sum of years of potential life lost due to premature mortality and the years of productive life lost due to disability (WHO, 2017). Tang et al. (2015) estimated human health damage factors of $PM_{2.5}$ produced by black carbon and organic carbon, SO_2 and NOx using a global chemical transport model. These DFs expressed changes in DALYs due to the unit emission of NOx, SO_2, and BCOC. Oldenkamp et al. (2016) also have estimated health impacts due to an increased NOx emission caused by the fraud of VW utilising human health damage factors. In this study, an approach similar to Oldenkamp et al. (2016) was followed to calculate the human health damage factors associated with excess NOx and $PM_{2.5}$ emissions.

In order to calculate health impacts based on $PM_{2.5}$ and NO_2 concentrations WHO BOD method was followed. Health impacts in terms of the burden of disease due to long-term exposure to $PM_{2.5}$ and NO_2 were calculated. The burden of disease is generally measured in terms of the number of deaths and/or DALY. DALY is expressed as the sum of YLL and YLD. YLD can be calculated as below (WHO, 2003),

$$YLD = I * DW * L \tag{3.2}$$

Where *I* is the number incidence cases, *DW* is disability weight (a number on a scale of 0-1) and *L* is the average duration of disability in years. Due to lack of availability data, *YLD* were not calculated in this research. *YLL* is the potential number of years lost owing to premature death and calculated as follows (WHO, 2003),

$$YLL = D * L \tag{3.3}$$

Where D is a number of deaths, L is standard life expectancy at age of death, i.e. average remaining years a person would have lived in the absence of the disease.

WHO BOD (2004) method was used to calculate premature death incidences. BOD method for outdoor air pollution has four main components as shown in Figure 3.8.

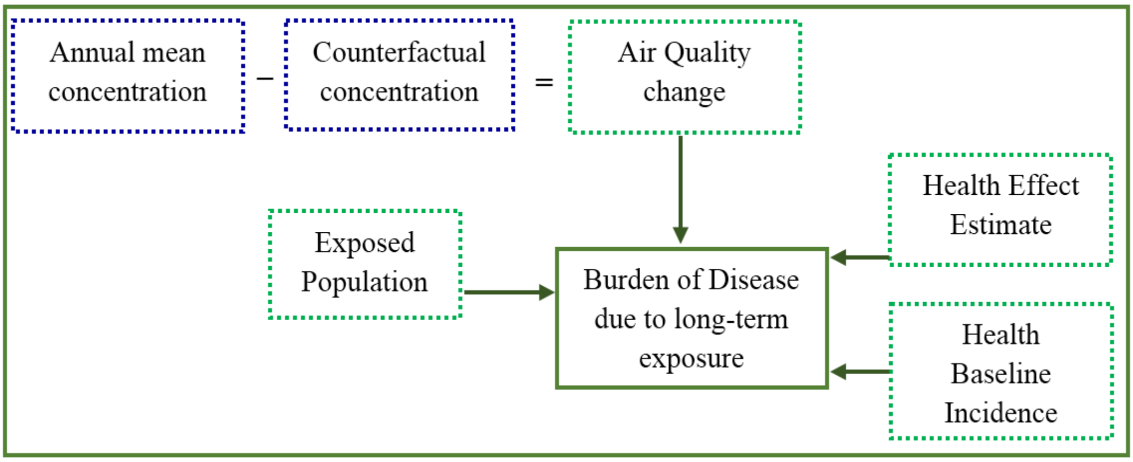


Figure 3.8. Required inputs for estimating BOD

- Air Quality change: This is taken into account by comparing the modelled concentration with the background concentration or baseline concentration. An assessment of exposure of the population to the pollutants based on modelled estimates.

- Health Effect Estimate: It is the estimate of the risk of an adverse health effect or mortality due to a one-unit change in ambient air pollution. They are mainly derived from epidemiological studies.

- Exposed Population: It is the size of the population groups exposed to the pollution and the type of health effect of interest.

- Health Baseline Incidence: This is the incidence of health effect being estimated, e.g. the underlying mortality rate in the population. In other words, it is the average number of people who die or suffer from a particular health effect over a given period of time in a given population.

Taking these components into account, the expected number of deaths (E) due to outdoor air pollution is calculated as follows (WHO, 2004),

$$E = AF * B * P \tag{3.4}$$

Where *AF* is the Attributable Fraction of the health effects from air pollution for the exposed population, *B* is the mortality rate (Deaths/person/year), *P* is the relevant exposed population, AF is calculated as,

$$AF = \frac{RR-1}{RR} \tag{3.5}$$

RR is the relative risk at exposure, compared to the reference level and calculated using equation 3.6 (WHO, 2004; WHO, 2013b).

$$RR = exp[\beta(C - C_0)] \tag{3.6}$$

Where *C* is the current annual mean concentration of the pollutant (μg/m^3), C_0 is the baseline/counterfactual concentration of the pollutant (μg/m^3). The lowest concentration used to calculate the health impacts of a pollutant is referred as counterfactual concentration and represents the pollutant concentration that could be achieved without any man-made pollution by changes in the environment or the lowest concentration observed in epidemiological studies (WHO, 2004; WHO, 2016c). β is an effect estimate which indicates the risk of an adverse health effect due to a one-unit change in ambient air pollution.

3.4. Estimation of Economic Impacts

The economic impacts of pollution were calculated in terms of damage cost of pollution and VSL. Damage costs associated to per tonne of pollutant emission were reported by DTTaS (2016a), Ireland titled Common Appraisal Framework for Transport Projects and Programmes. These damage costs not only include the health effects but also the effects on crops, material (e.g. buildings) and biodiversity. The health damage costs constitute about 90% of the reported overall damage costs. Damage costs per tonne of pollutant were taken as €13.22, €5,851, €19,143, €1,438, €1,398, €200,239, €48,779, €16,985 for CO_2, NOx, PM_{10}, VOC, NMVOC, $PM_{2.5}$ (Urban), $PM_{2.5}$ (Suburban), $PM_{2.5}$ (Rural) respectively.

The economic cost of mortality impact is assessed by multiplying VSL by the number of premature deaths. VSL is derived by aggregating the willingness of an individual to pay to secure a marginal reduction in the risk of premature death (WHO Regional Office for Europe, 2015). The VSL value for Ireland was obtained from WHO Regional Office for Europe (2015).

Chapter 4: Uncertainty Modelling of National Vehicular Emissions Inventory using COPERT

4.1. Introduction

NEI provides comprehensive and detailed estimates of air pollutants from emissions sources and is used to determine the status of network wide emissions in reaching targets, to develop new strategies and policies, in impact assessment and future projections. European Commission in its National Emissions Ceilings Directive and by Intergovernmental Panel on Climate Change in its guidelines for National Greenhouse Gas Inventories recommend that NEI reports must include information on uncertainties (European Union, 2016; Eggleston et al., 2006). Considering the importance of accurate estimates of emissions and their implications, this work conducts a sensitivity analysis and investigates the potential uncertainty associated with the emission estimates using COPERT. As outlined in Chapter 2, the most up to date version of COPERT i.e. COPERT 5, released in late 2016 (EMISIA, 2018), was used in this research. COPERT 5 requires detailed meteorological, activity and fleet data. There are parameters such as, temperature, Relative Humidity (RH), speed, mileage share and trip length whose values are either not accurately measured or average values are considered in quantifying vehicular emission for the whole country. This study gives insight into the sensitivity of emission levels of major air pollutants to those specific parameters and thus identifying the potential of reducing emission by controlling some of those parameters.

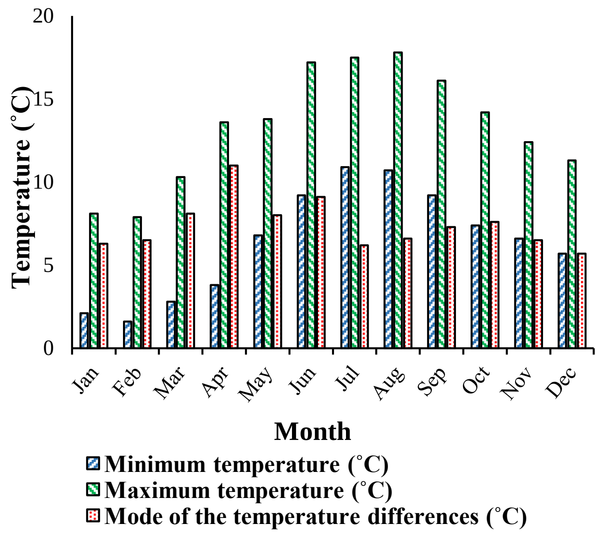
In this Chapter, the emission levels of CO_2, CO, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC, and N_2O have been reported by varying temperature, average speed, RH, driving mode share and average trip length. These parameters are varied one at a time, as well as two or more parameters, were varied simultaneously. Meteorological parameters, i.e. temperature-RH interaction and activity related parameters, i.e. speed, trip length, and mileage share interactions were examined. The individual effect of some of these parameters on the reduction of emission levels was studied (Fameli and Assimakopoulos, 2015; Vanhulsel et al., 2014; Andrias et al., 1993) in other countries. The present study extends their work by considering an exhaustive set of all parameters which lack precision, parameter interaction and a detailed sensitivity analysis. The level of variations in emission estimates with the variations in parameters shows the level of possible uncertainties in model outputs. Damage costs due to the air pollutants were computed for all the designed scenarios to understand potential underestimation or overestimation in cost impact assessments of damage caused by emissions. Finally, statistical analysis was carried out to understand the nature of uncertainty in COPERT 5 outputs and damage costs of the emission levels to input parameter variations. The emission levels, calculated using COPERT 5 due to parameter variations, were used to characterize the uncertainty associated with the passenger car emission inventory in terms of their probability distributions. In the next section 4.2, the data used in COPERT 5 in Ireland in preparing emissions inventory are described. This is followed by the descriptions of the designed scenarios, and the approach followed to develop the scenarios. The results obtained and discussion on the findings of this research are then presented in section 4.4, followed by a conclusion (section 4.5).

4.2. Data used for COPERT in Ireland

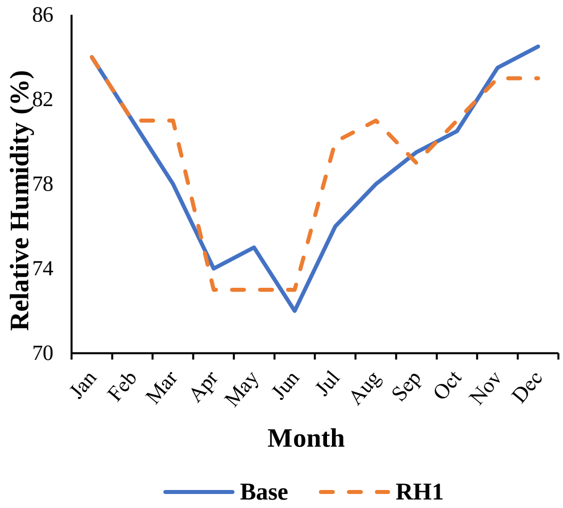
In this section, the data used in this research and their respective sources are described. The level of availability and the extent of their variability have also been described in this section. As previously mentioned, COPERT follows tier 3 methodology which calculates emissions based on detailed input data. Table 4.1 shows the necessary input parameters for COPERT 5 and their required level of disaggregation along with their level of availability and sources. Input data such as fuel/energy consumption and kilometres travelled can be derived from national car testing results. Whereas, the fleet configuration can be accurately obtained from the national vehicle registration database. Information on temperature, RH is recorded in monitoring stations across the country, but an average value is used in emission calculation for the entire country. Parameters, such as average trip length, average speed, and driving mode share are not measured at a detailed level and can significantly vary. The monthly average minimum and maximum temperatures are presented in Figure 4.1. The mode (the most frequently occurring data) of the daily minimum and maximum temperature gaps are also shown in Figure 4.1. The fleet data are mainly extracted from the Society of Irish Motor Industry (SIMI, 2017) and DTTaS (2015a). The detailed division of fleet data with respect to engine classes i.e. Small (<1.4 L), medium (1.4-2.0 L) and large (>2.0L), fuel type and technology classes (Euro 1, Euro 2 etc.) are shown in Figure 4.2.

Table 4.1. COPERT 5 input data, their sources and their level of availability in Ireland

Input data	Required level	Source	Measured/not measured
Fuel consumption (TJ)	Total for each fuel type	SEAI (SEAI, 2016)	Available
Fleet configuration	Disaggregated to each fuel-engine size-technology combination	Motorstats: The official statistics of the Irish Motor Industry (2016). DTTaS (2015a).	Available
Trip length (km)	Trip length for the vehicle type under calculation	National Travel Survey (CSO, 2014a)	Yearly average reported
Temperature (˚C)	Monthly minimum and maximum temperature	MET Éireann: The Irish Meteorological Service Online (2016)	Measured at monitoring stations
RH (%)	Monthly humidity	MET Éireann: The Irish Meteorological Service Online (2016) World Weather & Climate Information (2016)	Measured at monitoring stations
Driving share (%)	Disaggregated to Urban, Rural, Highway	Brady and O'Mahony (2011)	Not measured
Average speed (kmph)	Disaggregated to Urban, Rural, Highway	Road Safety Authority (2015)	Average free speed measured
Annual Average Mileage (AAM) (km)	Disaggregated to each fuel- engine size-technology combination	CSO (2014b) SEAI (2013)	Measured



(a)



(b)

Figure 4.1. Monthly average daily (a) minimum, maximum and mode of the temperature differences and (b) relative humidity (%), at the base scenario

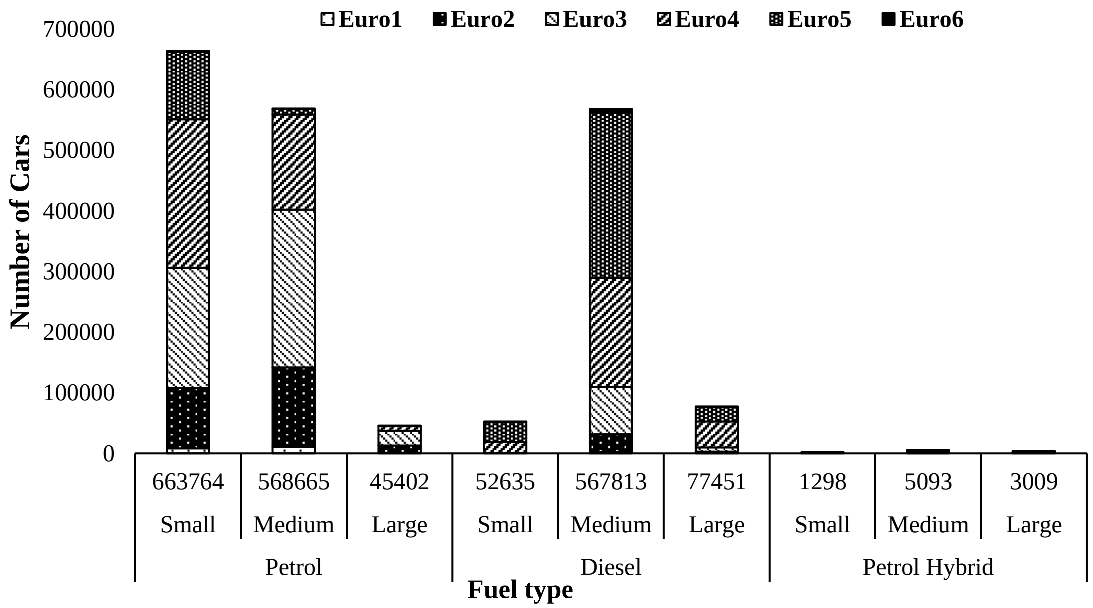


Figure 4.2. Passenger car fleet composition in 2015

The segregation of the PC fleet to technology classes was identified based on the year of commencement of the PCs (see appendix A). For diesel cars, the medium sized engines are significantly high than other engine sizes. Mileage shares for the base case were assumed to be 30%, 50%, 20% respectively for urban, rural and highway driving. These values were taken from the previous research carried out in Ireland (Brady and O'Mahony, 2011). The average urban driving speed for the base case was taken as 40 kmph, for rural as 60 kmph and highway as 100 kmph.

4.3. Scenario design

4.3.1. National Emission Inventory, Ireland (2015)

In order to calculate emissions inventory, passenger car fleet data were extracted from SIMI (2016). Three fuel categories were considered, petrol, diesel and petrol hybrid. Data were sorted into three engine classes, <1.4L, 1.4-2.0L, and >2.0L. A detailed description of the fleet composition is shown in section 3.2. AAM values for each engine size class varying from <900 cc to >3,000 cc (with 100 cc interval) for each year from 2000-2011 for diesel and petrol passenger cars were obtained from SEAI (2013) database provided by the National Car Testing services. These mileages were grouped into three engine categories, i.e. <1.4L, 1.4-2.0L and >2.0 listed in COPERT 5 for petrol and diesel cars. These AAMs for each class were then extrapolated using linear regression to get the AAM for 2015, as shown in Figure 4.3.

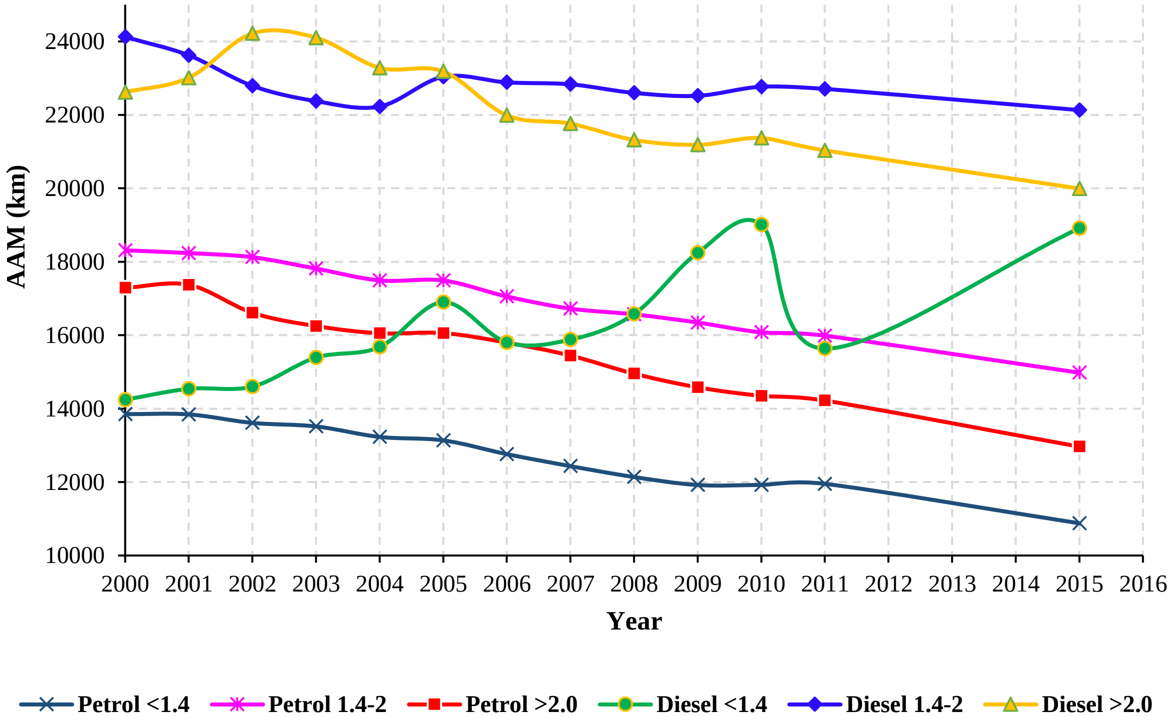


Figure 4.3. Disaggregated AAM (km/year) of passenger cars for 2000-2011 (SEAI, 2013) and 2015 (extrapolated)

The R^2 values obtained are shown in Table 4.2. R^2 or coefficient of determination is a measure of how well observed outcomes are replicated by the model. It can be observed from Figure 4.3 that AAM has a decreasing trend which is because of the increasing rate of car ownership in Ireland (SEAI, 2014).

Table 4.2. Coefficient of determination (R^2) values from the linear regression of AAMs

Engine size / Fuel type	Small	Medium	Large
Petrol	0.96	0.99	0.97
Diesel	0.57	0.29	0.66

The goodness of fit of mileage estimates of diesel vehicles is not as good as petrol vehicles. Alam et al. (2015) identified that diesel PCs do not show high co-relations with economic activity data which can be linked to more diesel powered vehicle use in recent years due to the introduction of CO_2 emissions based tax policy in Ireland. Annual mileage is an important parameter in determining emission levels. In order to examine the effect of inaccuracy in AAM consideration, COPERT was run also for the base scenario with AAM aggregated over all the engine size and fuel types. Aggregated AAM of all the cars in 2015 was obtained in the same way, i.e. by extrapolating 12-year's (2000-2011) aggregated AAM data. This showed a good fit with an R^2 value of 0.969. As mentioned earlier, emissions were calculated using COPERT 5 for eight pollutants (CO, CO_2, NO_X, $PM_{2.5}$, PM_{10}, VOC, NMVOC, and N_2O) in tonnes.

4.3.2. Sensitivity Analysis

Emission levels are affected by several parameters related to road, vehicle, environment, and traffic. Where the traffic data such as fleet composition can be obtained from the national database and sorted as per the requirement, meteorological and activity data are not accurately available or not possible to measure or spatially variable. In order to test the sensitivity of the model outputs to the input parameters and quantify the possible uncertainty associated with the model estimates, input parameters related to weather and activity were examined. Two approaches were taken to assess this. At first, the input parameters were varied 'one factor at a time', i.e. each parameter was varied individually while other input parameters were kept same. Then based on one factor at a time analysis, more sensitive parameters were identified, and two factors were varied simultaneously to examine the impact of factor (parameter) interaction on emission levels. The scenarios are described in the following sections.

4.3.2.1. One factor at a time

In this scenario, the effect of single factor variation on emission levels is assessed. The input parameters considered are temperature, RH, average speed, mileage share, and average trip length. This provides the knowledge and understanding of the sensitivity of the emission levels of the major air pollutants to those parameters. A summary of the designed scenarios is shown in

Table 4.3. The details of the base scenario, the designed scenarios and the approach followed to design the scenarios are described later in this section.

Table 4.3. Summary of the designed scenarios

Parameter	**Base scenario**	**Description of the tested scenario**
Temperature (T)	Monthly average minimum and maximum	-T1: Extreme minimum and extreme maximum
		-T2: Extreme minimum among recorded temperatures as monthly minimum and extreme minimum plus mode of the daily temperature variations in that month as monthly maximum
		-T3: Extreme maximum among recorded temperatures as monthly maximum and extreme maximum minus mode of the daily temperature variations in that month as monthly minimum
Relative Humidity (RH)	Monthly average	-RH1: Mode of the monthly RH
Speed (S)	Average speeds on Urban(40), Rural(60), Highway(100)	-S1: Speed limits on Urban(50), Rural(80) and Highway(110)
		-S2: Average lower speeds on Urban(25), Rural(50) and Highway(80)
Mileage Share (MS)	Urban:Rural:Highway as 30%:50%:20%	-MS1: Urban:Rural:Highway as 30%:40%:30% -MS2: Urban:Rural:Highway as 40%:40%:20% -MS3: Urban:Rural:Highway as 50%:40%:10%
Trip Length (TL)	15.1 km	-TL1: 6.0km
		-TL2: 9.1km
		-TL3: 12.1km
		-TL4: 18.1km
		-TL5: 21.1km
		-TL6: 24.2km

- *Temperature (T):*

COPERT requires monthly minimum and maximum temperatures for emission calculation. While calculating the emission levels for a whole country, the average of the recorded monthly maximum and minimum temperatures of all the stations are considered. But this may result in under- or over-estimation of emissions if some parts of a country experience significantly higher or lower temperature compared to other areas. To examine the level of variability in emission estimates as a result of temperature variations, three scenarios were designed in addition to the base scenario which presents the emission levels in 2015 taking temperatures as monthly averages of all the stations. The scenarios were designed such that the possible variation is captured. Therefore, in addition to considering the extreme minimum temperature and extreme maximum temperature, daily temperature differences between minimum and maximum were also taken into account while designing the scenarios. Further, the mode of the daily temperature differences was considered which reflects the variation of daily temperature which is mostly occurring. It may be more realistic to capture the gap between the minimum and maximum temperature by considering the mode of the temperature differences. The designed approach for temperature scenarios is described as follows,

Temperature scenario 1 (T1),

$$T_{m,min} = \{T_m\}_{min} \tag{4.1}$$

$$T_{m,max} = \{T_m\}_{max} \tag{4.2}$$

m = Month, i.e. Jan, Feb, Mar……..Nov, and Dec.

$T_{m,min}$ = Minimum temperature for emission calculation in month i

$T_{m,\,max}$= Maximum temperature for emission calculation in month i

$\{T_m\}_{min}$ = Minimum of all the recorded temperatures at the monitoring stations in month i

$\{T_m\}_{max}$= Maximum of all the recorded temperatures at the monitoring stations in month i

Temperature scenario 2 (T2),

$$T_{m,min} = \{T_m\}_{min} \tag{4.3}$$

$$T_{m,max} = \{T_m\}_{min} + d_{m,mode} \tag{4.4}$$

d_{mode} = Mode of the temperature gaps (shown in Figure 4.1) between maximum and minimum in month m

Temperature scenario 3 (T3),

$$T_{m,min} = \{T_m\}_{max} - d_{m,mode} \quad (4.5)$$

$$T_{m,max} = \{T_m\}_{max} \quad (4.6)$$

Table 4.4 presents the monthly minimum and maximum temperature values taken in the three temperature scenarios, T1, T2, and T3.

Table 4.4. Monthly minimum and maximum temperatures corresponding to the designed scenarios

	Temperature (˚C)						
Scenario →	**T1**			**T2**		**T3**	
	Minimum	Maximum	Mode of daily temperature differences	Minimum	Maximum	Minimum	Maximum
Jan	0.6	9.8	6.3	0.6	6.9	3.5	9.8
Feb	0.3	9.1	6.5	0.3	6.8	2.6	9.1
Mar	1.4	11.7	8.1	1.4	9.5	3.6	11.7
Apr	1.9	15.7	11.0	1.9	12.9	4.7	15.7
May	5.2	15.5	8.0	5.2	13.2	7.5	15.5
Jun	7.9	19.1	9.1	7.9	17.0	10.0	19.1
Jul	9.6	18.9	6.2	9.6	15.8	12.7	18.9
Aug	9.3	19.1	6.6	9.3	15.9	12.5	19.1
Sep	7.3	17.5	7.3	7.3	14.6	10.2	17.5
Oct	5.5	15.5	7.6	5.5	13.1	7.9	15.5
Nov	4.8	13.8	6.5	4.8	11.3	7.3	13.8
Dec	3.7	12.7	5.7	3.7	9.4	7.0	12.7

- *Speed (S):*

COPERT requires average speeds of vehicles in urban, rural and highway driving conditions. Speed information for these categories is not precisely found. Speed is one of the major parameters influencing vehicular emissions, therefore, it is very important to observe the effect of the possible variation in speed. In this study, the base scenario considers average speed for urban, rural, and highway as 40 kmph, 60 kmph, and 100 kmph respectively (Road Safety Authority, 2015; Alam et al., 2015). These speed values are reported by Road Safety Authority based on the free speed survey. To see the level of variability two extreme conditions were tested, one scenario (S1) considers the posted speed limits on urban, rural and highways and the other scenario (S2) considers the lowest recorded average speed under those driving conditions.

- *Mileage Share (MS):*

Mileage share is another very important factor in emissions calculation as the operating speed, road characteristics, traffic densities and thereby the exhausted emissions are different on regional roads, local roads, national roads etc. In COPERT, mileage share information is required for urban, rural and highways. The base mileage shares were taken as 30%, 50% and 20% for urban, rural, and highway respectively. Three scenarios were designed (see Table 4.3), to capture the variability, denoted as MS1, MS2, MS3. The scenarios are designed such that the sensitivity of emission to each driving mode can be studied by comparing the results which are presented separately for each driving mode.

- *Trip Length (TL):*

It is required to provide the average trip length (km) in COPERT. A single average trip length value is considered for a country average trip length. This is likely to vary and is important to assess the impact of trip length on emission levels. The trip length for the base case was taken as 15.1km (CSO, 2014a). Six scenarios (TL1, TL2, TL3, TL4, TL5, and TL6), as shown in Table 4.3, were considered by increasing and reducing the average base trip length by 20%. It is to be noted that the AAM values were considered to be the same in all the scenarios to understand the impact of average trip length on emissions. Also, this will help to identify those trips causing more emissions and thereby finding alternatives to replace those trips to reduce emission levels.

4.3.2.2. Factor interaction

In this case, the effects of multi-factor variation on emission levels were studied. Emissions were calculated by varying two or more factors simultaneously. The designed scenarios are described in the following subsections.

- Temperature-Relative Humidity: This scenario studies the impact of the variability of the weather parameters, i.e. temperature and RH that are considered in COPERT 5, on emission levels. The emission variations were studied for these two sets of RH (base and RH1) values against four temperature scenarios described in section 3.3.1. (i.e. base, T1, T2, and T3). Therefore, total eight emission estimates were obtained from a combination of four temperature scenarios and two RH scenarios.
- Urban Speed-Trip Length: It was found that emission levels are significantly sensitive to urban speed and trip length. Therefore, average urban speed and trip length were varied simultaneously to understand their interaction. In this scenario, a range of possible urban speeds and trip lengths were studied in terms of their impact on vehicular emissions. Based on the national travel survey data, a range of trip lengths varying from 5-19 km. and an urban speed range of 20-45kmph was examined.
- Urban Speed-Trip Length-Urban Driving Share: The results show that rural and highway emissions increase or decrease by the same percentage if the rural and highway driving shares are changed by certain percentages. However, it was observed that urban emission share is more sensitive to the urban mileage shares. Therefore, the speed and trip length combinations tested in the previous scenario were run for three additional driving shares, 20%, 40%, and 50%.

4.3.3. Uncertainty Analysis

Uncertainty associated with COPERT 5 outputs were modelled by identifying characteristics of probability distributions of the pollutant emissions. The emissions estimated from each of the scenarios were plotted as a histogram and fitted to the most suitable PDF. Therefore, the sources of uncertainties related to input parameters were taken into account. The goodness of fit was tested using Kolmogorov-Smirnov or K-S test at 5% significance level. K-S test is one of the most often used goodness of fit tests. The advantage of this test is that the nature of the K-S test is non-parametric (Hassani and Silva, 2015) and therefore, does not make any assumption about the distribution of data.

4.4. Results and discussion

In this section, the findings of the study of the sensitivity analysis are presented and observations from the results are discussed.

4.4.1. Effect of input parameters on emission

4.4.1.1. One Factor at a time

- *Temperature Scenario:*

Temperature is an important parameter in emission levels as it affects the cold start and evaporative EFs (Fameli and Assimakopoulos, 2015). The results obtained from the temperature scenario are presented in Figure 4.4. The results show that when extreme temperatures are considered, the difference in emission levels is not significant. The maximum difference was found for cold start PM emissions which are 0.8% though the difference in total emissions is 0.1%. The reason behind this can be the increase in cold start emission due to lower temperature is offset by the lower emissions when the maximum average temperature is higher than the base. To capture the emissions behaviour with lower temperature and higher temperature, T2 and T3 were designed. T2 and T3 represent more realistic situations as the mode of the daily temperature differences between lowest and highest temperatures are taken into consideration. From the emission values in Figure 4.4, it is observed that levels for cold and evaporative emissions increase (especially for CO, PM, and VOC) when the average monthly minimum and maximum temperatures are lower than that in the base scenario.

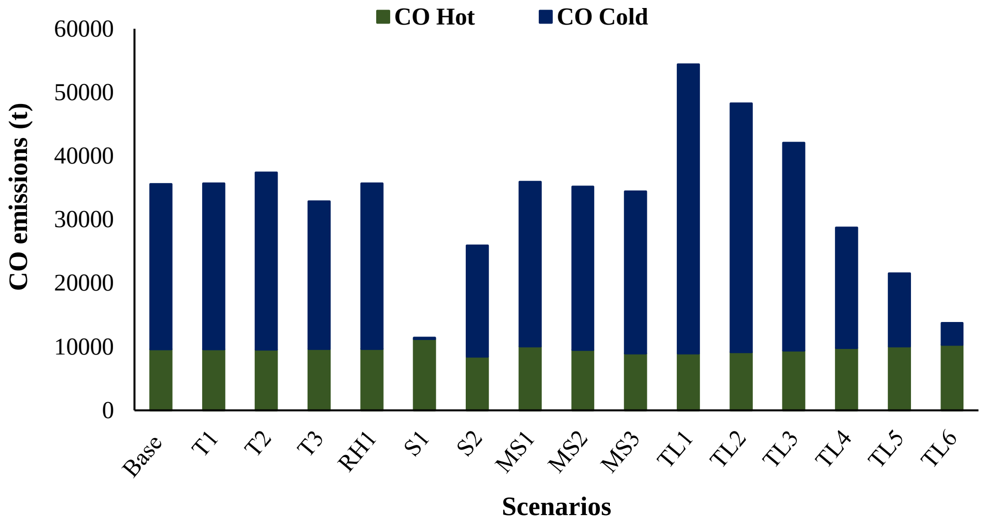
The results obtained for T3 (Figure 4.4), support the observation from T2 where it was shown that the emission levels decrease when both the minimum and maximum monthly average temperature were taken to be higher than the base case temperatures. The reduction in cold start emissions for CO, NOx, $PM_{2.5}$, VOC, and NMVOC was 6.6%, 8.2%, 14%, 6.2% and 6.8% respectively from the base scenario. There is no notable difference in emission levels when extreme monthly temperatures were used which is because the increase in cold start emission levels due to lower temperature was balanced by the reduction in cold-start emissions due to higher maximum temperature compared to the base scenario. However, when the minimum monthly temperatures were considered lower (T2) than the base temperature the cold start and evaporative emissions increase. Whereas, when the monthly average maximum temperatures were higher (T3) than the base, cold start and evaporative emission levels were lower. However, there were no considerable differences in hot exhaust emissions in any of the scenarios, thus, it can be concluded that temperature mainly affects cold-start emission levels.

- *Relative Humidity Scenario:*

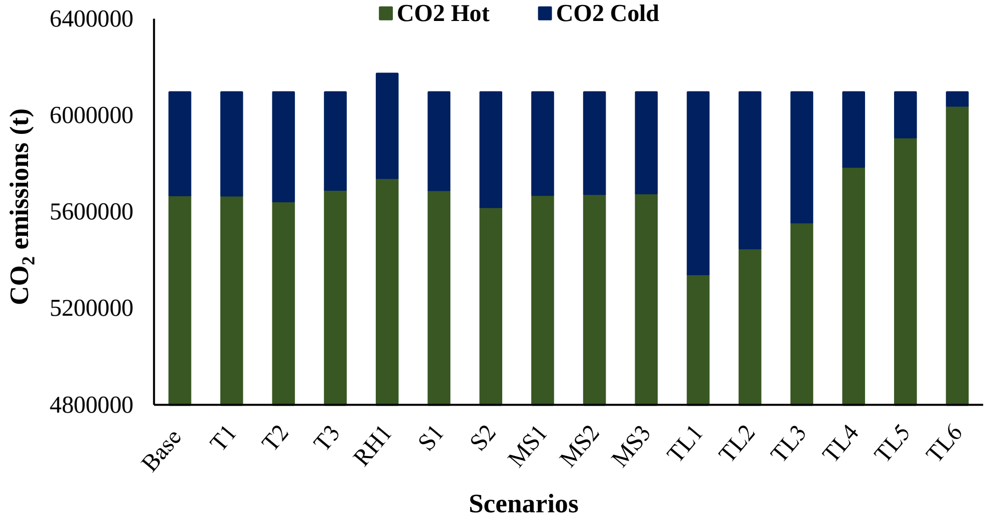
The results shown in Figure 4.4 indicates that there are no considerable changes in emission estimates when modes of RH values were taken instead of average values for the same monthly temperature values. RH is correlated to temperature; therefore, the impact of humidity can be better understood in the next section where the humidity and temperature are varied simultaneously.

- *Speed Scenario:*

It is observed that in S1 (Figure 4.4) when the speed limits are taken as the average operating speeds, there is a significant reduction in emission levels in VOC (70.7%) followed by NMVOC (69.9%), CO (67.9%) and PM_{10} (10.6%). Whereas in S2 (Figure 4.4), the differences in emission levels are not high except for CO (27.1%). It is to be noted that the average speeds in each driving mode is lower than the speed limits in the respective modes. When the average operating speed values were taken equal to the speed limits (S1) emission levels were lower which is expected as the fuel consumption is lower when the speed is higher. Lower average speed resulted in an increase in emissions from diesel cars but a decrease in petrol powered cars, thereby, decrease in overall emission levels as there are more petrol vehicles in the overall fleet. This can be linked to the presence of higher number of diesel vehicles and lower number of petrol vehicles with larger engine size in the fleet. Based on this observation, it may also be said that larger engine sized vehicles are more affected by average operating speeds than vehicles with a smaller engine. It was identified that CO, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC, N_2O emissions could be saved significantly if a higher average speed equal to the speed limit could be maintained.

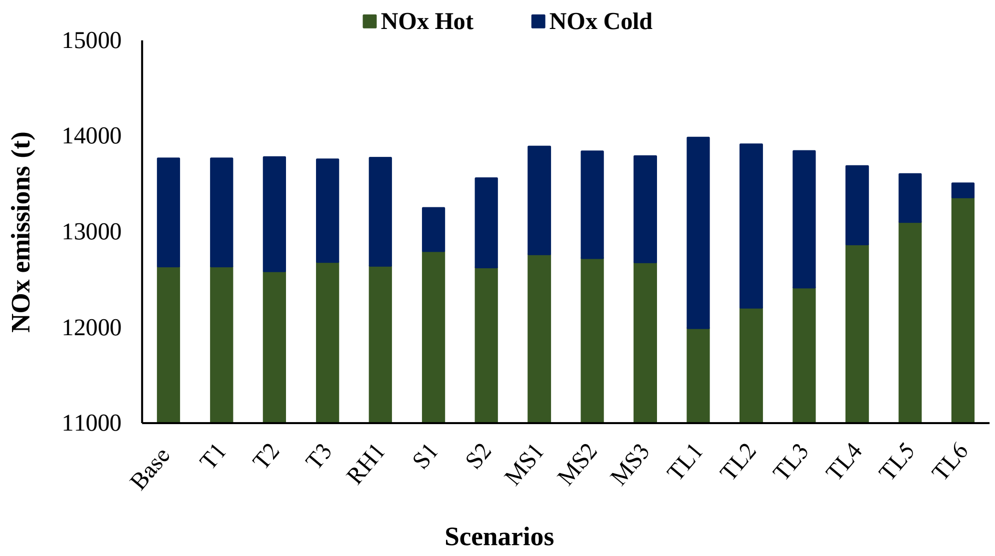


(a) Hot, cold, non-exhaust and evaporative emissions of CO

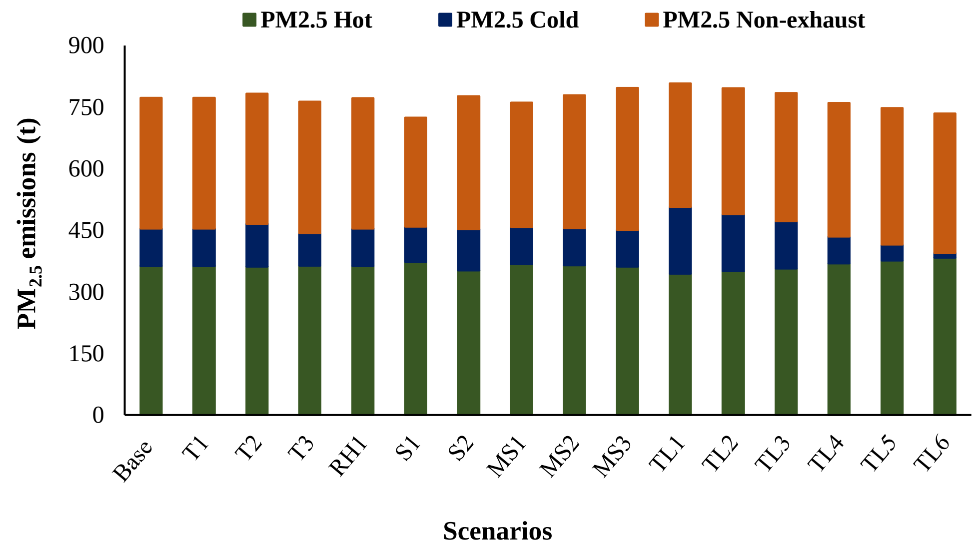


(b) Hot, cold, non-exhaust and evaporative emissions of CO_2

Figure 4.4. Hot, cold, non-exhaust and evaporative emissions from all the scenarios (cont.)

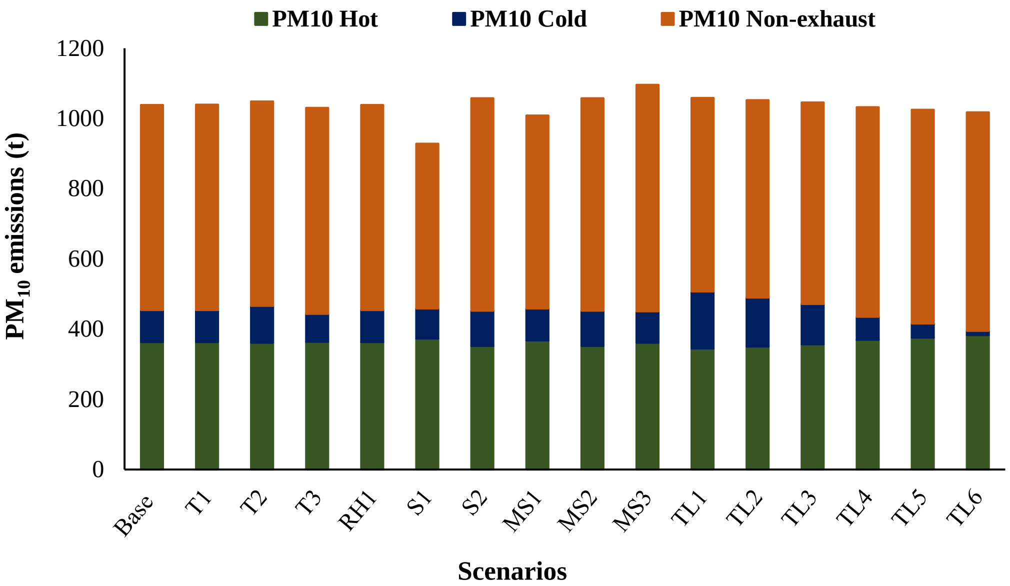


(c) Hot, cold, non-exhaust and evaporative emissions of NOx

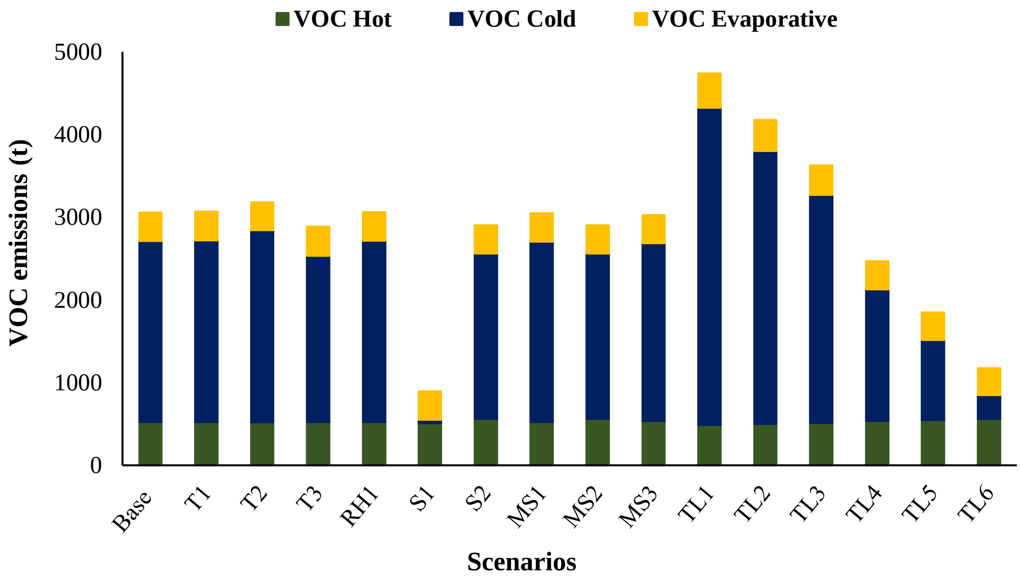


(d) Hot, cold, non-exhaust and evaporative emissions of $PM_{2.5}$

Figure 4.4. Hot, cold, non-exhaust and evaporative emissions from all the scenarios (cont.)

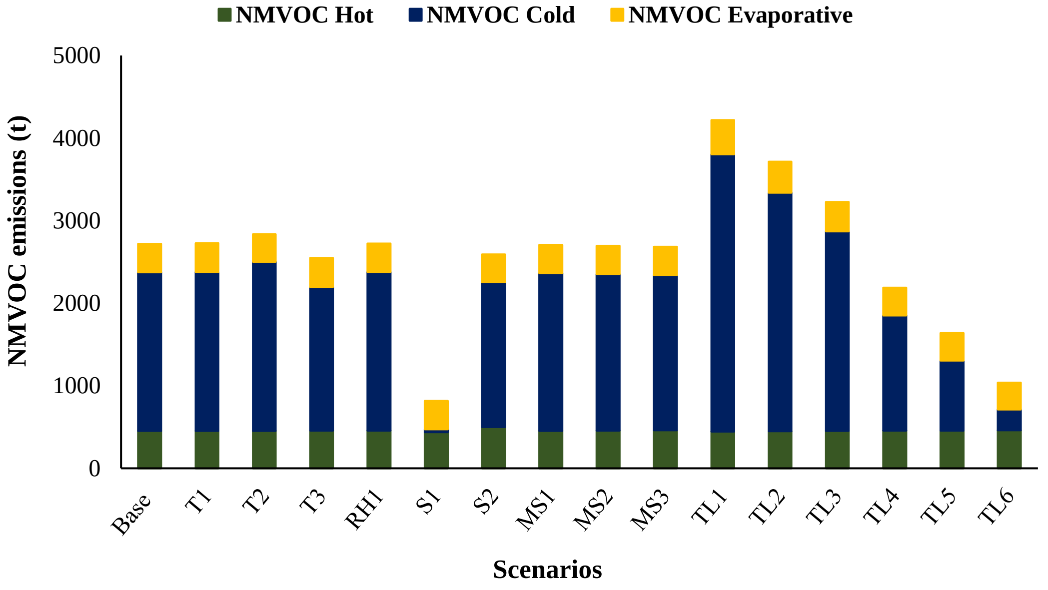


(e) Hot, cold, non-exhaust and evaporative emissions of PM_{10}

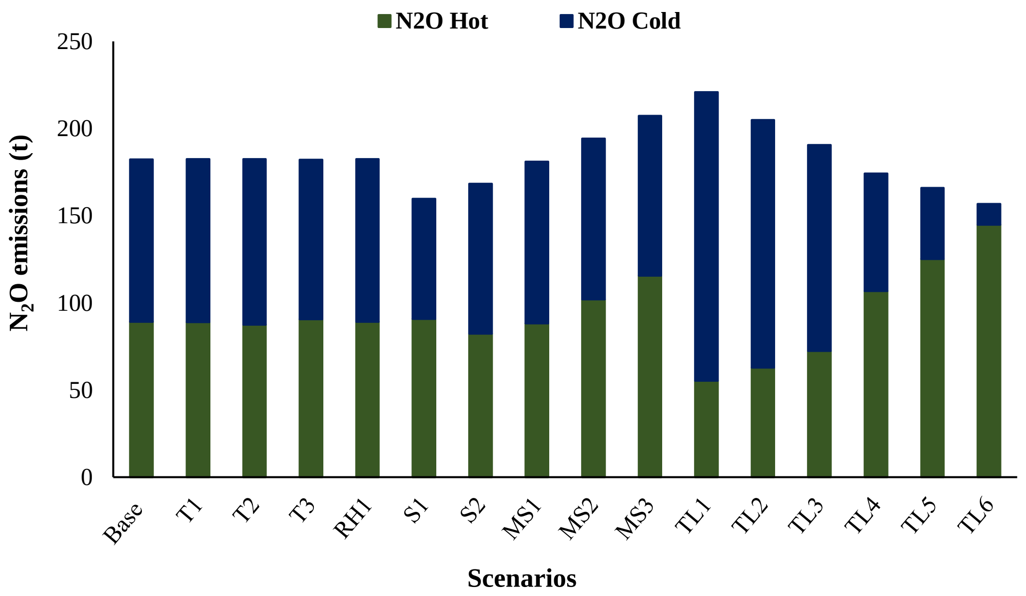


(f) Hot, cold, non-exhaust and evaporative emissions of VOC

Figure 4.4. Hot, cold, non-exhaust and evaporative emissions from all the scenarios (cont.)



(g) Hot, cold, non-exhaust and evaporative emissions of NMVOC



(h) Hot, cold, non-exhaust and evaporative emissions of N_2O

Figure 4.4. Hot, cold, non-exhaust and evaporative emissions from all the scenarios

- *Mileage Share Scenario:*

Three mileages share scenarios were designed to create the emissions inventory and to observe the effect of each type of driving condition on emission levels. In Table 4.5, separate emission levels resulted from urban, rural and highway driving are shown with the percentage differences with respect to the base scenario. The results indicate that CO_2, NO_X, and PM are more sensitive to urban driving share. It is observed that with 10% increase in urban driving share CO_2 emissions increase by 26% and NO_X and $PM_{2.5}$ increase by 25% and 23% respectively, whereas, with 10% reduction in rural share emissions reductions are around 20% for all the major air pollutants. Whereas 10% decrease in highway driving share results in about 50% lower emissions for CO, CO_2, PM, NO_X, and N_2O and 38% for VOC.

Table 4.5. Emission levels for mileage share scenarios

	Scenario, Driving mode share	**Driving mode**	**Pollutants**							
			CO	**CO_2**	**NOx**	**$PM_{2.5}$**	**PM_{10}**	**VOC**	**NMVOC**	**N_2O**
Emissions (tonnes)	Base, Urban:30 Rural:50 Highway: 20	Urban	28,587	$2.3*10^6$	5,257	331	443	2,649	2,367	112
		Rural	4,272	$2.7*10^6$	5,826	328	463	274	235	51
		Highway	2,699	$1.1*10^6$	2,682	114	133	135	116	19
Change in emissions (%) from base	MS1, Urban:30 Rural:40 Highway: 30	Urban	0	0	0	0	0	0	0	0
		Rural	-20	-20	-20	-20	-20	-18	-17	-20
		Highway	49	49	49	49	49	37	35	49
	MS2, Urban:40 Rural:40 Highway: 20	Urban	2	26	25	23	25	1	1	20
		Rural	-21	-21	-21	-21	-21	-18	-18	-21
		Highway	-1	-1	-1	-1	-1	-1	-31	-1
	MS3, Urban:50 Rural:40 Highway: 10	Urban	4	51	50	46	50	3	2	41
		Rural	-22	-21	-21	-21	-21	-19	-18	-21
		Highway	-51	-51	-51	-51	-51	-38	-36	-51

- *Trip length Scenario:*

In this study, six trip lengths, of which three were taken by reducing the base trip lengths by 20%,40% and 60% and other three by increasing the trip lengths by the same percentages were considered. The trip lengths examined were of 6.0km, 9.1km, 12.1km, 18.1km, 21.1km and 24.2km length. Table 4.6 presents total emissions from the base case with the average trip length of 15.1km and the percentage increase and decrease with the change in trip length. Figure 4.4 shows cold start, hot exhaust, evaporative and non-exhaust emissions separately for all trip length scenarios.

It can be observed from the results in Table 4.6 that the emissions for the shorter trips are significantly higher, especially for CO, VOC, and NMVOC, and with the increase in average trip length emission levels decrease. It was found that for lower average trip lengths emission levels for CO, VOC, NMVOC increase by 52-55%. For the average trip lengths varying from 18.1-24.2 km, the possible emissions savings range between 19-62% for CO, VOC, and NMVOC. This observation is in line with the findings of other researchers (Vanhulsel et al., 2014; Fameli and Assimakopoulos, 2015). There is no significant difference in CO_2 emissions was found, as it is mainly influenced by other factors such as, speed, fuel type, engine size etc. The fact that emission levels increase with the decrease in average trip length for the same annual mileage indicates the possibility of significant emissions savings by replacing the shorter trips with walking or cycling.

Table 4.6. Base case emission levels and percentage differences in emission levels for trip length scenarios

Pollutant	**Trip length(km)**						
	15.1	**6.0**	**9.1**	**12.1**	**18.1**	**21.1**	**24.2**
	Emission level (tonnes)	**Percentage difference from the base (%)**					
CO	35,558	53	36	18	-19	-39	-61
CO_2	6,095,743	0	0	0	0	0	0
NOx	13,765	2	1	0.6	-0.6	-1	-2
$PM_{2.5}$	772	5	3	2	-2	-3	-5
PM_{10}	1,039	2	1	0.6	-0.7	-1	-2
VOC	3,058	55	37	19	-19	-40	-62
NMVOC	2,717	55	37	19	-19	-40	-62
N_2O	182	21	12	4	-4	-9	-14

Total emission levels of all the pollutants from all the scenarios are presented in Figure 4.5 by the box-whisker plot to see the range of variations. The horizontal lines present the minimum and maximum values and the red line inside the box shows the median. The black horizontal lines above and below the box present the minimum and maximum values. Figure 4.5 shows that there are significant variations in emission levels due to possible variation in input parameters.

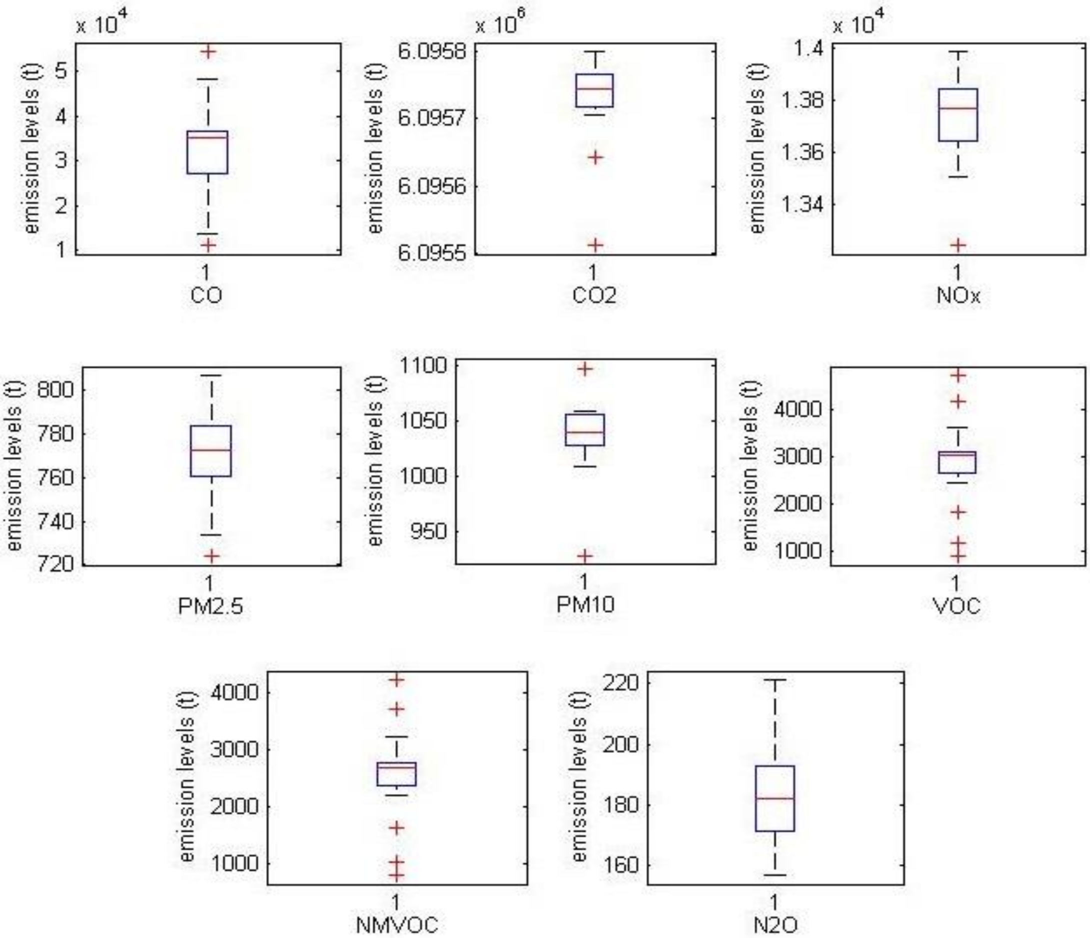


Figure 4.5. Minimum and maximum emission levels for major air pollutants

- *Aggregated and disaggregated mileage:*

This section presents the difference in emission levels of CO_2, CO, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC and N_2O estimated using aggregated AAM for the entire fleet. Aggregated AAM refers to a single averaged value of annual mileage considered for the entire fleet. As mentioned earlier, this study considered disaggregated i.e. separate mileage values for each of the fuel types and engine size ranges. Table 4.7. shows the emission levels when aggregated mileage is considered and when disaggregated mileage is considered in the base scenario, and the difference in emission levels when aggregated average mileages were taken instead of disaggregated mileage. The results from aggregated annual mileage and disaggregated annual mileage consideration do not show any significant difference in emission levels. The highest difference of 1.3% is seen in CO levels, whereas, the difference in CO_2 and $PM_{2.5}$ levels are insignificant (See Table 4.7).

Table 4.7. Emission levels from aggregated and disaggregated mileage consideration

Pollutants	Emission levels (tonnes)		Difference (%)
	Disaggregated mileage	Aggregated mileage	
CO	35,558	36,025.5	1.3
CO_2	6,095743	6,095741	0
NOx	13,766	13,715.3	-0.4
$PM_{2.5}$	772.4	772.1	-0.04
PM_{10}	1,038.7	1,039.4	0.1
VOC	3,058	3,034.3	-0.8
NMVOC	2,717	2,692.2	-0.9
N_2O	182.4	181.8	-0.3

4.4.1.2. Factor Interaction

- *Temperature-Relative Humidity:*

This section presents the emission variations against temperature and RH scenarios as shown in Figure 4.6. For CO_2, NOx, $PM_{2.5}$, PM_{10}, and N_2O, emissions are affected by T2 (Extreme minimum and maximum as minimum plus mode of the daily temperature differences in that month). Mode of temperature differences were used to take into account the most frequently occurring daily temperature differences. Therefore, it is observed that when both the lower and upper limit of average monthly temperature is lower than average, emission estimates of those pollutants are sensitive to RH. However, for T1 or T3, which considers relatively higher temperature ranges, the variation in emission levels are not significant. Also, there is no considerable difference observed for CO, VOC, and NMVOC in any of the scenarios compared to the base case. This indicates that these pollutants are not sensitive to RH in the temperature ranges explored in this study. Although the emissions of some pollutants are sensitive to temperature and RH, it depends largely on their interaction. More numbers of temperature and RH scenarios can be examined for other countries which experience different weather conditions than Ireland.

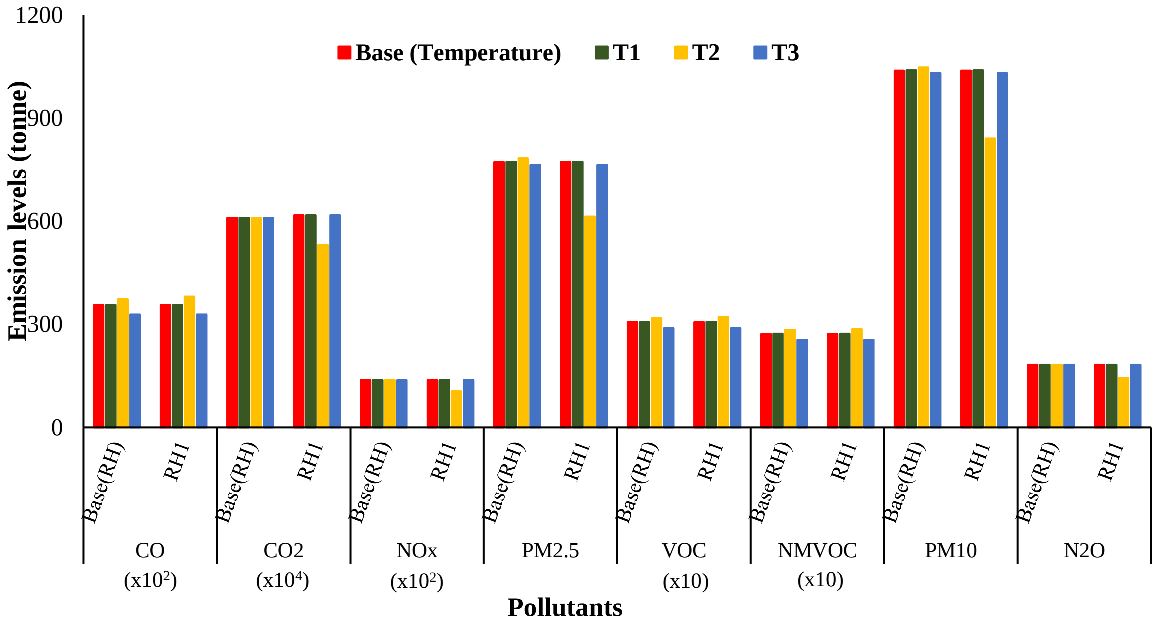


Figure 4.6. Annual emission levels against temperature and relative humidity

- *Urban speed-Trip Length:*

Figure 4.7 illustrates the 3D surface plots of CO, CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC and N_2O emissions against the average urban speed and trip length changes. All the pollutants follow a similar pattern as for individual parameter variations except NOx. For NOx, for lower speed in case of 20 and 25, emissions increase with increase with average trip length but in case of speed 30 or higher, NOx emission levels decrease with an increase in average trip length. For CO_2, the effect of speed is least with respect to the speed variation. For CO, VOC, and NMVOC, emission levels decrease with increase in speed till 30kmph and then start to increase. But as observed in S1, emissions start to decrease after that due to lower cold start emissions.

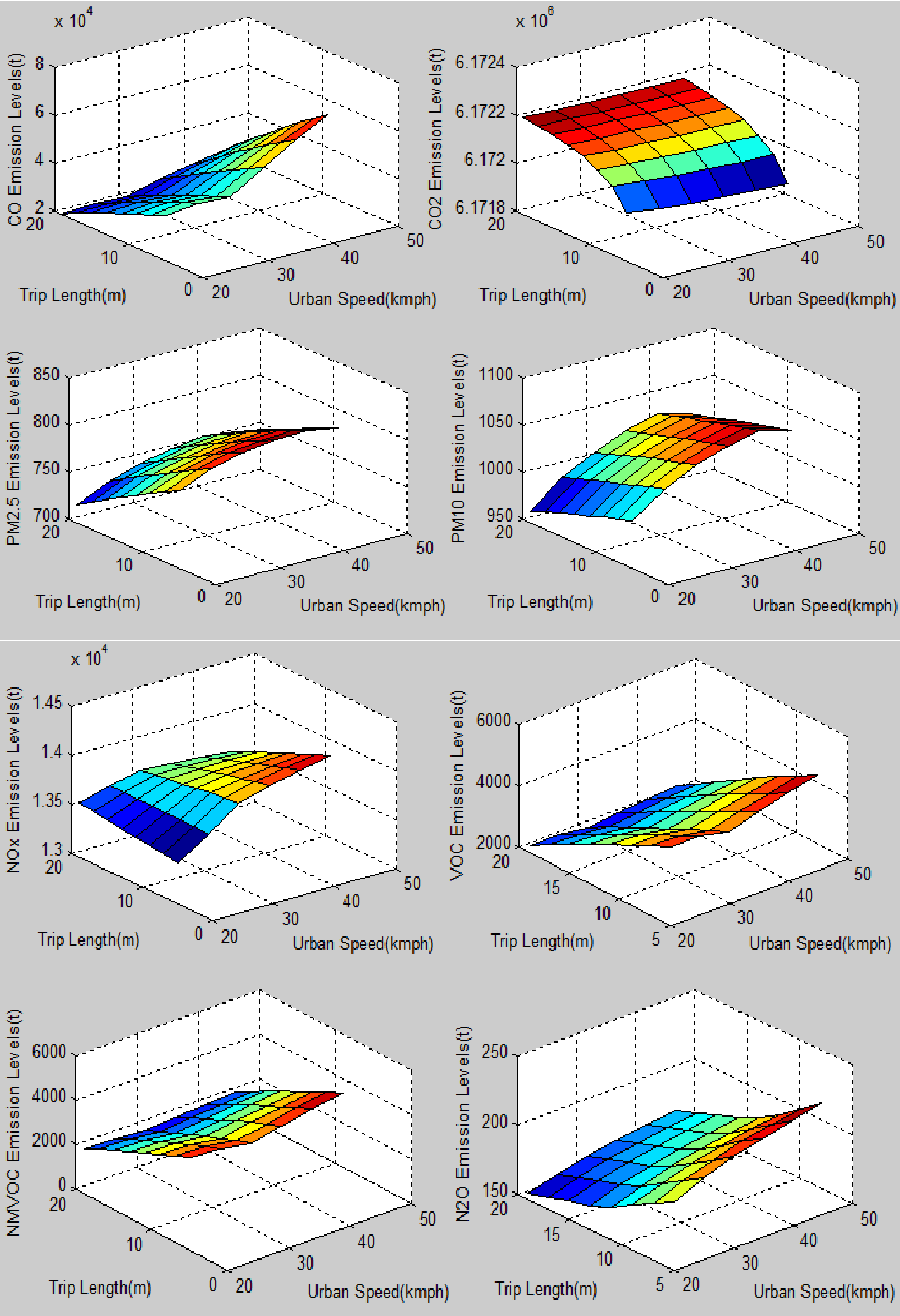


Figure 4.7. 3D surface plots of network wide annual emissions vs trip length and urban speed

4.4.2. Uncertainty Analysis

As mentioned in Section 3.4.3, COPERT 5 emissions estimates obtained from the different scenarios were characterised by fitting PDFs and the results are presented in this section. In addition to the scenario results presented in the previous sections, emission levels at other urban shares (20%, 40%, and 50%) were run for the same trip length and urban share combinations as shown in Figure 4.7. PDFs and the statistical parameters explaining the nature of the distributions are listed in Table 4.8. The coefficient of variation values presented in Table 4.8 indicate that CO_2, $PM_{2.5}$, NOx, and PM_{10} have a lower level of uncertainty in estimation with respect to input parameter variations in COPERT 5. However, CO, VOC, NMVOC, and N_2O have a higher level of uncertainty. It was statistically tested that the best-fitted PDFs of CO, CO_2, NOx, $PM_{2.5}$, and N_2O are lognormal distributions, VOC is gamma, NMVOC is log-logistic and PM_{10} is weibull at 5% significance level. Figure 4.8 illustrates the frequency histograms and PDFs for CO, $PM_{2.5}$, VOC, N_2O, PM_{10}, and NMVOC. The skewness values for frequency distribution of each pollutant are presented also in Table 4.8. Skewness is a measure of symmetry or lack of symmetry of the dataset. A skewness value between -0.5 and 0.5 indicates that the distribution is approximately symmetric.

Table 4.8. Statistical analysis results for each pollutant

Pollutant	Sample size	Mean	Standard deviation	Coefficient of variation	Skewness	PDF
CO	187	34,205	9,450	27.63	0.52	Lognormal(3P)[7]
CO_2	171	6,161,406	65,788	1.07	-1.06	Lognormal
NOx	186	13,747	300	2.18	-0.35	Lognormal
$PM_{2.5}$	187	771	27	3.44	-0.20	Lognormal
VOC	67	3,253	893	27.44	-0.34	Gamma
NMVOC	67	2,951	783	26.52	-0.17	Log-Logistic
PM_{10}	67	1,023	36	3.48	-0.52	Weibull
N_2O	67	186	19	10.19	0.35	Lognormal

[7]The 3-parameter lognormal distribution or Lognormal (3P) "is a general skew distribution in which the logarithm of any linear function of a given variable is normally distributed." Sangal and Biswas (1970). It is a more general form of the lognormal distribution that includes an additional location or shift parameter.

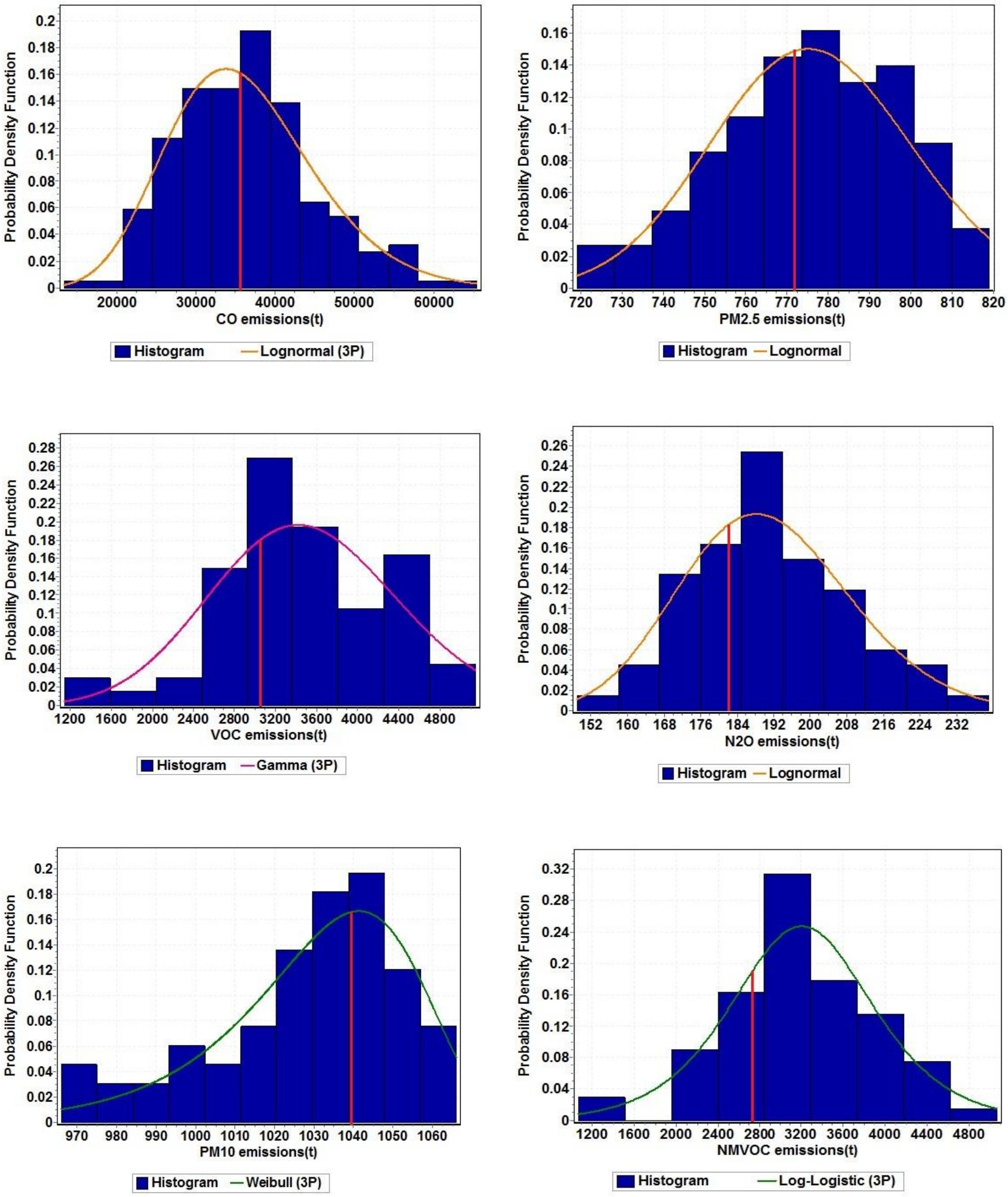


Figure 4.8. Histograms and PDFs of annual passenger car emission levels (Red line indicates the base case emission levels)

4.4.3. Cost implications of uncertainty

Greenhouse and non-greenhouse gases directly or indirectly cause damage to human health, crops, materials, plant and animal diversity. These damages caused by road transport emissions can be monetised using damage cost values per tonne of emissions (€ per tonne) (Table 4.9) reported by DTTaS (2016a) and in the handbook on external costs of transport (Korzhenevych et al., 2014). Based on the emission results from all the model runs, the damage costs in the base case, and minimum and maximum emissions cases have been reported in this section (Table 4.9).

This gives information about the possible range of variation in damage costs depending on potential variation in emission estimates.

Table 4.9. Emissions and associated potential damage costs of pollution in base, possible minimum and possible maximum cases

Pollutants	Emissions values (€/t)	Base		Minimum		Maximum	
		Emissions (tonne)	Damage costs (mil€)	Emissions (tonne)	Damage costs (mil€)	Emissions (tonne)	Damage costs (€)
CO_2	13.22	6,095,743	80.59	6,095,514	80.58	6,172,219	81.60
PM_{10}	19,143	1,039	19.88	841	16.09	1,060	20.29
NOx	5,851	13,765	80.54	10,507	61.48	14,140	82.73
VOC	1,438	3,058	4.40	896	1.29	5,124	7.37
NMVOC	1,398	2,717	3.80	818	1.14	4,573	6.39
$PM_{2.5}$ (urban)	200,239	331	66.26	267	53.41	536	107.24
$PM_{2.5}$ (suburban)	48,779	114	5.54	88	4.28	105	5.12
$PM_{2.5}$ (rural)	16,985	328	5.57	260	4.41	182	3.08
Total damage cost (mil€)		266.58		222.68		313.83	

As can be observed from Table 4.9, the damage cost of $PM_{2.5}$ in urban areas is very high compared to damage costs in suburban and rural areas. Total cost of damage due to pollution from PC fleet in Ireland was found to be €266.58 million in the base case with highest cost associated with CO_2, followed by NOx and $PM_{2.5}$. However, the costs calculated for possible minimum and maximum scenarios show that there is a possibility of overestimation by €43.9 million or underestimation by €47.25 million in the base scenario. It is also to be noted that even though the per tonne damage cost of NOx is lesser than $PM_{2.5}$, the total cost is greater for NOx than $PM_{2.5}$. This indicates higher levels of NOx pollution from the PC fleet.

4.4.4. COPERT 4 (v.11.3) vs COPERT 5 (v.1.1) estimates

In several studies (Achour et al. 2011; Berkowicz et al., 2006; Kousoulidou et al. 2010), it was reported that COPERT 4 underestimates NO_X. After it was found that diesel vehicles are emitting NOx at a higher level than expected, COPERT 5 was released with revised NOx EFs. In order to highlight the difference in outputs from COPERT 4 (v.11.3) and COPERT 5 (v.1.1), emission levels from the 2015 PC fleet were estimated using COPERT 4 (v.11.3). This information is important and useful to point out the necessary revision in emission level reporting in places where previous version of COPERT is still used to calculate and report emission inventories. Figure 4.9 presents the emission inventories for 2015 calculated using previous version of COPERT (v.4.11.3) and the revised version of COPERT (v.5.1.1) which has improved EFs for NO_X (EMISIA, 2018).

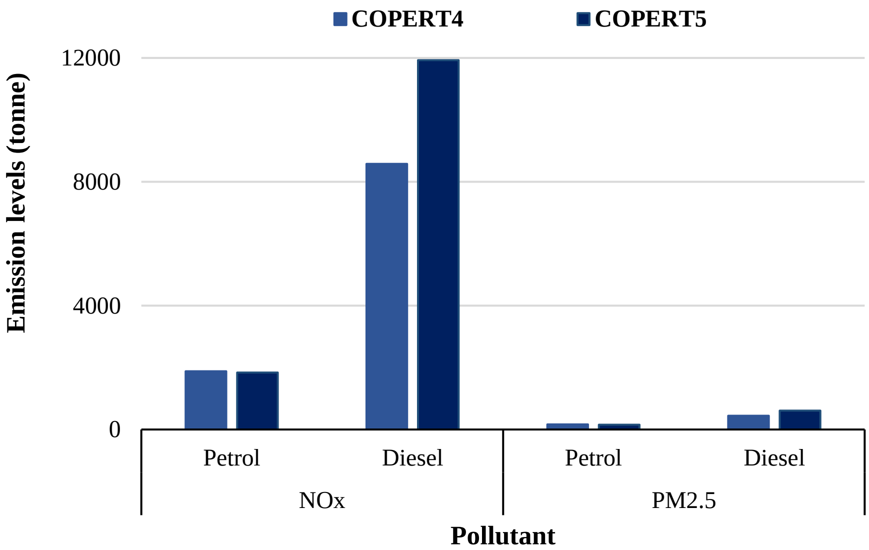


Figure 4.9. Comparison in emissions calculated using COPERT 4 (v.11.3) and COPERT 5 (v.1.1)

From Figure 4.9, it can be observed that diesel NO_X emissions from COPERT 5 are notably higher than COPERT 4 estimates. COPERT 5 emissions outputs are higher than COPERT 4 by 32% and 28%, respectively, for NO_X and $PM_{2.5}$. But there were no notable differences observed in CO_2 (0.03%) or CO (0.94%) outputs.

4.5. Conclusion

Considering the major applications of the national emissions inventory, it is important to calculate emission levels as accurately as possible. This research explores the uncertainties of CO, CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC and N_2O emissions from passenger cars due to input parameter variations in COPERT 5. The effect of the factors whose values are not accurately available or not always measured or averages are considered in preparing emissions inventory for the whole country, were identified and the sensitivity of emission levels to those input parameters was explored in this study. This provides useful information to the users of COPERT 5 in terms of the degree of precision that each of the input parameters requires. Hot exhaust, cold start, evaporative and non-exhaust emissions are presented for all the major air pollutants to have a better understanding of their emissions behaviour. Results show that emission levels are more sensitive to average operating speed and average trip length which are measured from sample sets. However, no significant variation in overall emission levels was observed with rural and highway mileage share variation which are not measured. The results revealed that overall emission levels of CO, VOC, NMVOC, N_2O are more sensitive than CO_2, NOx, PM to the parameters evaluated in this study. Uncertainty associated with CO_2 is less (-0.004 to 1.3%) with respect to those input parameter variation ranges. However, uncertainty in CO, VOC, NMVOC and N_2O estimates depends more upon average trip length, urban driving share and urban driving speed. The annual emission levels presented in this study indicate that, in reality, the emission levels might be much lower (up to 58% for CO, VOC, and NMVOC) or much higher (up to 79% for CO, VOC, and NMVOC) depending on the level of variation of the input parameters. Uncertainty in PM and NOx emission levels may deviate from the reported estimates by -24 to 3%. For N_2O, the under- and over-estimation may lie between 21 and 31% respectively. Depending on the extent of variations in input parameters, the damage costs of air pollution caused by passenger cars in Ireland can be lower by 16% or higher by 18% than the base case. The accurate consideration of the input parameters may result in an emission level considerably higher or lower than the current consideration leading to a better planning, modelling and policy making. It is always recommended that rather than reporting a single value of emission, a range of emission values accompanied by uncertainty for emission estimates should be reported to allow more credibility and transparency of the estimates. As mentioned earlier in Chapter 3, in addition to the input parameter variations, there could be other sources of uncertainty such as vehicles that are not obeying the emissions standards or cheating the emissions standards while undergoing the emissions test. Therefore, potential impacts of certain vehicles violating the emission standards, on emission levels in Ireland were explored in this research and the findings are presented in the next chapter.

Chapter 5: Examining Potential Uncertainty in Vehicular Emission Estimates in Ireland due to *dieselgate*

5.1. Introduction

In Chapter 4, possible uncertainty associated with NEI in Ireland due to variation in input parameters was examined. However, although the vehicle fleet related input parameters with known accuracy can be obtained, it is possible that there are uncertainties associated with EFs. As mentioned in Chapters 1 and 2, it was found that certain diesel vehicles do not follow emissions standards and emit much higher levels of NOx emissions than expected. For effective and efficient policy making and investments aiming to reduce emission levels from road transport, the knowledge of the actual state of emission levels is very important. The study presented in this chapter investigates the potential deviation in emission estimates in Ireland due to certain vehicles in the fleet not obeying the emissions standards. The findings of this work are mainly based upon the work published in Dey et al. (2018a). The transportation sector is the greatest contributor to air pollution and with the booming demand for transportation, reducing the pollution has become one of the main concerns (EEA, 2016a; WHO, 2016b). Emission standards for different vehicle type and size are designed to protect air quality and human health. Ideally, conventional diesel vehicles emit a higher percentage of pollutants such as NOx and $PM_{2.5}$ and a lower percentage of GHG such as CO_2 levels (EPA, 2015). However, as the *dieselgate* came to light, NOx emissions from diesel PCs and LCVs have become a matter of disquiet since it has been found that NOx emissions exceed the standard limits significantly. This issue first irrupted as Volkswagen scandal where it was identified that specific VW and Audi models (US EPA, 2016c) are not obeying the emission standards with the help of a defeat device with a cheating software. The software detects when a car is undergoing emissions testing and turns full emissions controls for NOx only during the test and at the other times, emissions controls are turned off, consequently 2.0 litre and 3.0 litre engines emit NOx by 10-40 times and up to 9 times than the standard limit, respectively. Transport and Environment (2016) has reported that not only specific VW and Audi PCs but many other LDVs are suspected to be cheating the emission standards.

In Ireland, the motor vehicle taxation system was changed from an engine capacity-based system in 2008 to a CO_2 emissions-based system with an aim to reduce GHG emission levels. This has led to a significant increase in diesel vehicle purchases. The percentages of new car registrations of diesel and petrol in Ireland were 28% and 71% respectively in 2007 which changed to 71% diesel and 27% petrol in 2015 (SIMI, 2017) due to the introduction of a new tax regime since July, 2008. Additionally, the entire LCV and bus fleet in Dublin is diesel powered. The percentage of cars with alternative fuel options, i.e. EV, HEV and PHEV was 1% in 2007 which has only increased by 1% over the period of 2007-2015. Due to *dieselgate* and this huge increase in diesel use, it is expected that there will be a huge amount of excess NOx discharged from the car and LCVs in Ireland in excess of the amount of NOx emissions expected from these fleets. This research reported in this section of the thesis intends to study the potential impacts up to 2015 in terms of this excess emission by Euro 5 LDVs i.e. PCs and LCVs in Ireland based on all the reported facts on *dieselgate*.

The first objective of this study is to examine the NOx emission levels from all the Euro 5 PCs and LCVs in Ireland. This situation is considered and studied based on researches where it has been found that the real-world EFs are a lot higher than the euro standard limit. But exactly which models or if all the models are faulty or fitted with defeat device are not known. Thus, the emission levels are tested considering all the Euro 5 PCs and LCVs to see the overall quantity of excess emission. Ntziachristos et al. (2016) presented mean NOx Emission Factors (EFs) for PCs and LCVs measured under different conditions, such as lab tests, on-road measurements and COPERT 4 (v. 11.3). In order to compute the emission levels based on lab test results and on-road measurements, in this study, NOx emission levels from Euro 5 PC and LCV in Ireland were calculated using COPERT 4 (v. 11.3). NOx emission levels from the fleet were also calculated using COPERT 5 (v.1.1), which has revised EFs for NOx, to show if COPERT 5 (v.1.1) underestimates the NOx emission levels or not. Additionally, the potential health and financial impacts of the excess NOx discharged from all the Euro 5 PCs and LCVs were evaluated. Additionally, NOx emission levels from the defective VW and Audi models as identified by US EPA were calculated to quantify the effect of the VW scandal on Ireland. In Ireland, the share of PCs and LCVs of VW and Audi brands is 16% (DTTaS, 2015a). Therefore, the potential impacts were quantified assuming all the Euro 5 cars and LCVs from VW and Audi are faulty. The next section 5.2 describes the methodology followed to design the scenarios and calculate NOx emission levels. Section 5.3 elaborates how the impacts of the excess NOx were evaluated. The data utilised to assess the hidden NOx is then presented in section 5.4. The results in terms of total and hidden NOx emissions discharged from potentially defective Euro 5 LDVs are presented and discussed in the following section. The conclusion of the chapter is presented in section 5.6.

5.2. Scenario design

The potential impacts of *dieselgate* were assessed by designing and assessing several scenarios. This section presents the designed scenarios and the methodology followed to determine the NOx emission levels from diesel Euro 5 PCs and LCVs based on COPERT 4 (v11.3), laboratory test results, PEMS measurements and also as per Euro standards. PEMS records the real-world emissions when the vehicle is driven on road and the measurements reflect the on-road emission levels. The summary of the different scenarios designed is presented in Table 5.1.

Table 5.1. Summary of the designed scenarios

Scenarios	**Description**
Scenario 0- Euro standard scenario	NOx emission levels as per euro standard specification for Euro 5 PCs and LCVs
Scenario 1A- COPERT base scenario	NOx emission levels from Euro 5 PCs and LCVs calculated using COPERT
Scenario 1B-PEMS base scenario	NOx emission levels based on the PEMS measurements and include all the Euro 5 PCs and LCVs in the fleet in Ireland
Scenario 1C- lab test base scenario	NOx emissions based on laboratory test measurements for all the Euro 5 PCs and LCVs present in the fleet in Ireland
Scenario 2A- VW COPERT base scenario	NOx emissions resulted from reportedly (US EPA, 2016e) faulty VW and Audi models present in the Irish PC fleet and calculated using COPERT
Scenario 2B- VW PEMS base scenario	NOx emissions from the reportedly faulty PC models based on PEMS measurements
Scenario 2C- VW lab test base scenario	NOx emission levels from the reportedly faulty PC models based on laboratory test results
Scenario 3A- VW PC hypothetical scenario	NOx emissions considering if all the VW and Audi PCs present in the Irish PC fleet are faulty
Scenario 3B- VW LCV hypothetical scenario	NOx emissions considering if all the VW and Audi LCVs present in the Irish LCV fleet are faulty

As mentioned earlier, real-world and laboratory emission levels were calculated based on findings reported by Ntziachristos et al. (2016). The researchers used COPERT 4, therefore COPERT 4 refers to the baseline scenario in this research to assess the potential deviations in NOx estimates in real-world and laboratory scenarios. The methodology followed to design the scenarios is described in detail later in this section.

- ***Scenario 0- Euro standard scenario***

This section describes the expected emission levels from Euro 5 PCs and LCVs following the Euro standard specifications. The standard emission levels for all the vehicles were calculated using the following equation,

$$E_{ijstd} = N_{ij} * M_{ij} * EF_{jeuro} * 10^{-6} \quad (5.1)$$

E_{ijstd} is the NOx emissions (tonne) in the year *i* (2011, 2012, 2013, 2014 and 2015) and vehicle type *j* (in this case, PC and LCV) as per the euro standard specifications; N_{ij} is the number vehicles of category *j* in year *i*; M_{ij} is the average annual mileage (km) in year *i* for vehicle type *j*; EF_{jeuro} is the Euro standard NOx EF (g/km) for Euro 5 vehicle type *j*. Euro standard NOx EFs were taken as 0.18 g/km for PCs (EEA, 2007) and 0.28 g/km for LCVs (Kadijk et al., 2015).

- ***Scenario 1A- COPERT base scenario***

In this scenario, the vehicular NOx emission levels resulted from the existing Euro 5 and LCV fleet in Ireland were calculated by COPERT 4 (v11.3) using the default EFs. For this study, overall Euro 5 fleet data were obtained from DTTaS by considering a total number of the newly registered vehicle since the introduction of Euro 5 vehicles (i.e. over the period of 2011-2015). Average annual mileages were calculated for each year based on past data (SEAI, 2013). The mileage shares for urban, rural and highway driving were taken as same in each year. The NOx emissions were then calculated separately for PCs as well as LCVs for each year from 2011 (i.e. when Euro 5 was introduced) to 2015 and added up to obtain the overall NOx emissions from entire Euro 5 PCs and LCVs until 2015.

- ***Scenario 1B- PEMS base scenario***

PEMS base scenario calculates the quantity of on-road NOx emissions from Euro 5 LDVs to show how much emissions the Euro 5 LDVs are possibly emitted while on-road compared to the Euro standard emission levels as calculated in Scenario 0. Also, this will show the quantity of overestimated (for Euro 5 PC) and ignored (for Euro 5 LCV) emissions by COPERT 4. In order to estimate the on-road emissions, urban, rural and highway emissions were calculated separately using COPERT 4 and then modified based on the PEMS EFs reported by Ntziachristos et al. (2016) to reflect the real-world NOx emissions, as shown in Figure 5.1. Ntziachristos et al. (2016) presented the EFs separately for urban, rural and highway driving, therefore, in order to estimate real-world emissions based on PEMS measurements urban, rural and highway emissions were needed to be calculated separately. In COPERT, an overall annual mileage value can be provided and then, mileage share percentages are to be provided to define the mileages driven in each mode share. Therefore, NOx emissions for the urban driving condition were calculated by keeping urban driving share as 100% and both rural and highway shares as zero.

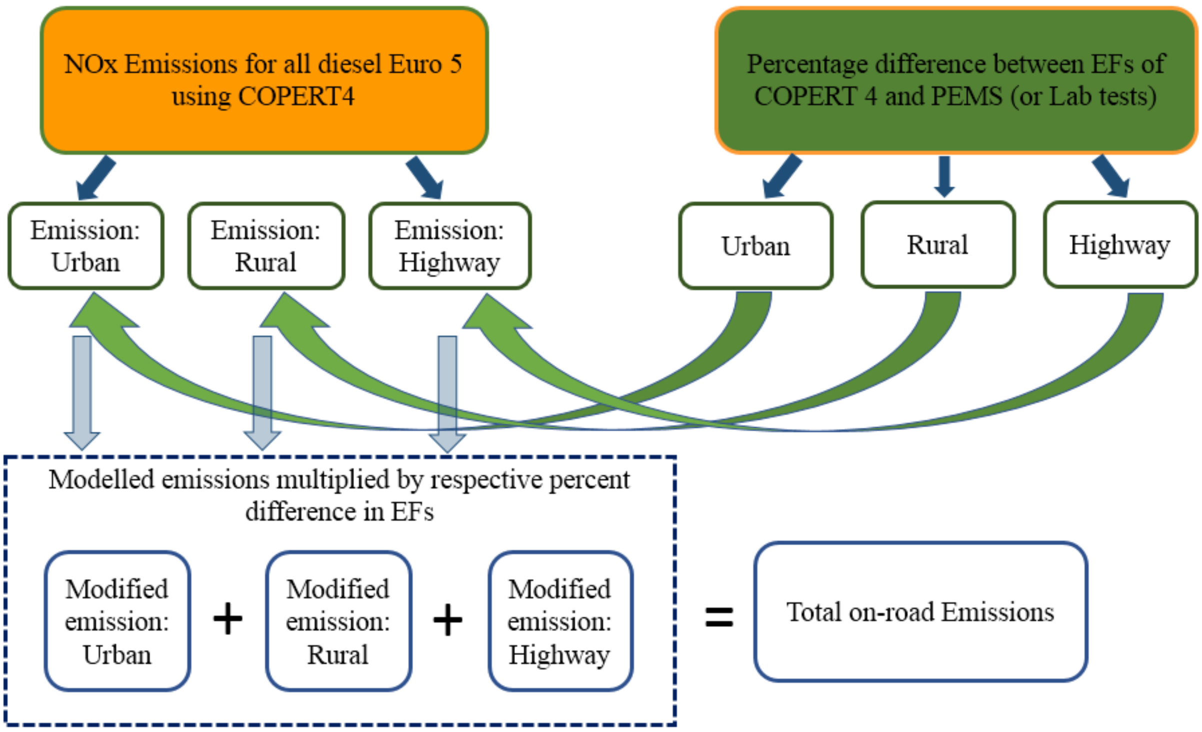


Figure 5.1. Schematic diagram of estimating on-road and lab test based real-world NOx emissions

The similar approaches were followed while calculating emissions for rural and highway driving conditions. Annual mileage values were disaggregated by their respective percentages of driving mode shares. Then to get the on-road emission values, NOx EFs (g/km) which are based on a significant number of on-road measurements (Ntziachristos et al., 2016) was taken as reference. The EFs in different conditions are presented in Table 5.2 and Table 5.3, respectively, for PCs and LCVs. These values were obtained from the figure (see Figure 2.6) presented by Ntziachristos et al., (2016). However, the differences between emission values obtained from COPERT 4 and on-road measurements are quite significant, and both are much higher than the Euro 5 NOx limit. The percentage differences between the COPERT and PEMS estimated EFs (g/km) were calculated based on findings of Ntziachristos et al., (2016). These percentage differences were used to estimate the on-road emission levels.

For LCVs, EF values from real-world laboratory test cycles were found to be higher than model estimated EFs for urban and rural roads, but EFs from on-road measurements were even higher in all the driving conditions and all the estimated values were significantly higher than the EU standard Euro 5 NOx EF limit.

Table 5.2. NOx EFs (g/km) from the graph (Figure 2.6) for urban, rural and highway for passenger cars

Driving mode	Euro 5 Standard EFs	COPERT/lab test EFs	PEMS EFs	Difference between COPERT and PEMS EFs (%)
Urban	0.18	0.76	0.88	+16
Rural	0.18	0.49	0.22	-55
Highway	0.18	0.61	0.35	-43

Table 5.3. NOx EFs (g/km) from the graph (Figure 2.6) for urban, rural and highway for light commercial vehicles

Driving mode	Euro 5 Standard EFs*	COPERT EFs	PEMS EFs	Difference between COPERT and PEMS EFs (%)	Lab test EFs	Difference between COPERT and laboratory test EFs (%)
Urban	0.28	0.78	1.55	+99	0.90	+15
Rural	0.28	0.64	1.52	+138	0.72	+13
Highway	0.28	1.25	1.64	+31	1.16	-7

*varies with respect to the weight, the highest among them is presented here.

Real-world emission levels were calculated by multiplying the separate emissions by a factor equal to the percentage differences (see Table 5.2 and Table 5.3) between COPERT EFs and PEMS EFs to estimate the real-world NOx emission levels using the methodological approach shown schematically in Figure 5.1. These individual emission levels calculated under urban, rural and highway driving conditions were then summed up to represent the total on-road NOx emission level from LCVs. The same methodology was followed to estimate emission for each year from 2011 to 2015.

- ***Scenario 1C- lab test base scenario***

Lab test base scenario presents the quantity of Euro 5 NOx emissions based on laboratory test results reported by Ntziachristos et al. (2016). A similar procedure, as was followed to obtain NOx emissions based on PEMS measurements, was used to achieve laboratory test emissions from Euro 5 PCs and LCVs (Figure 5.1). In this case, the percentage differences (see Tables 5.2

and 5.3) found between COPERT 4 EFs and lab test EFs were used to modify emission values calculated by COPERT 4. It can be observed in Table 5.2 that for Euro 5 PCs, real-world lab test EFs were found to be very close to COPERT 4 EFs. Thus, it is considered that NOx emission levels estimated by COPERT 4 are consistent with those resulted from lab tests. Therefore, PC NOx emissions have not been calculated separately and lab test base scenario only presents results for Euro 5 LCVs.

- ***Scenario 2A- VW COPERT base scenario***

VW base scenario conveys the NOx emitted by the specific vehicle fleet fitted with defeat devices as reported by US EPA. Table 5.4 presents the VW and Audi passenger car models that have been reportedly found to be cheating the NOx emissions.

Table 5.4. Affected VW and Audi passenger car models (US EPA, 2016e)

Affected 2.0 litre diesel models	**Affected 3.0 litre diesel models**
Jetta	Volkswagen Touareg
Jetta Sportswagen	Porsche Cayenne
Beetle	Audi A6 Quattro
Beetle Convertible	Audi A7 Quattro
Audi A3	Audi A8
Golf Sportwagen	Audi A8L
Golf	Audi Q5
Passat	Audi Q7

Specific models of VW and Audi passenger cars with 2.0 litre and 3.0 litre were extracted from the overall database of Irish Motor Industry (SIMI, 2017) and NOx emissions were then calculated using COPERT 4. Input parameters such as speed, mileage share, average annual mileage etc. were considered to be same as those in scenario 1A.

- ***Scenario 2B- VW PEMS base scenario***

This section presents the actual quantity of NOx discharged from US EPA reported VW and Audi models based on PEMS measurements. To calculate emissions measured by PEMS, a similar approach was followed as PEMS base scenario i.e. emissions were calculated separately for different driving modes using COPERT 4 and revised to calculate real-world emission levels as recorded using PEMS (Figure 5.1).

- ***Scenario 2C- VW lab test base scenario***

This section presents the NOx emission levels exhausted by faulty VW and Audi models as per lab test results (Ntziachristos et al., 2016). The similar procedure, as followed in case of lab test base scenario to estimate NOx emissions for overall Euro 5 fleet, was followed to obtain NOx emission levels from particular VW and Audi models in lab tests.

- ***Scenario 3A- VW PC hypothetical scenario***

Hypothetical Scenarios are designed to explore the effect of circumstances if all 59527 VW and Audi Euro 5 PCs (SIMI, 2017) and 12337 VW Euro 5 LCVs in Ireland are faulty. Both, Transport and Environment (2016) and Ntziachristos et al. (2016) indicates that it might not be only the reported models which are equipped with defeat devices. Based on this, it would be worthy to examine the emission impacts from not only the US EPA reported VW and Audi models but all the VW Euro 5 PCs. Hence, the following hypothetical situations were tested to determine the excess amount of NOx. VW PC hypothetical scenario presents NOx emission levels from all the aforementioned VW scenarios i.e. VW COPERT base scenario, VW PEMS base scenario and VW lab test base scenario, for overall VW and Audi Euro 5 car fleet in Ireland. The total number of all VW and Audi models in Ireland were extracted from the overall dataset of Irish Motor Industry (SIMI, 2017). NOx emissions for all VW and Audi models were estimated by the similar method as used to calculate emissions in the base scenarios (Figure 5.1).

- ***Scenario 3B- VW LCV hypothetical scenario***

In this section NOx emissions from all the VW Euro 5 LCVs are presented. The number of VW LCVs that are present in Ireland were extracted from the Irish Motor Industry (SIMI, 2017) database. The number of Audi LCVs are negligible, hence ignored in this study. Emissions were calculated by all three cases, i.e. using COPERT 4, and based on lab test results and PEMS measurements following the similar approach as mentioned in COPERT, PEMS and lab test base scenarios, respectively.

5.3. Impacts of NOx

The health and financial impacts of the hidden NOx caused due to *dieselgate* in Ireland have been calculated following the methods described in this section. BOD is a measure of the sum of YLLs and YLDs and is referred to as DALYs. This unit DALY value reported by Tang et al. (2015) for European countries has been multiplied by total extra NOx emissions in order to obtain the excess number of DALYs resulted due to the hidden NOx from Euro 5 LDVs.

To study the spatial variation of the impacts, total excess NOx emissions were distributed over all the counties in Ireland. County-wise population (CSO, 2011) and LDV count (SIMI, 2017) were extracted and plotted as shown in Figure 5.2. It can be observed from the graph that the R^2 value is good ($R^2 = 0.93$). Therefore, a linear relationship was assumed between population and vehicle count, and vehicle densities were used to distribute the health and cost impacts among the counties.

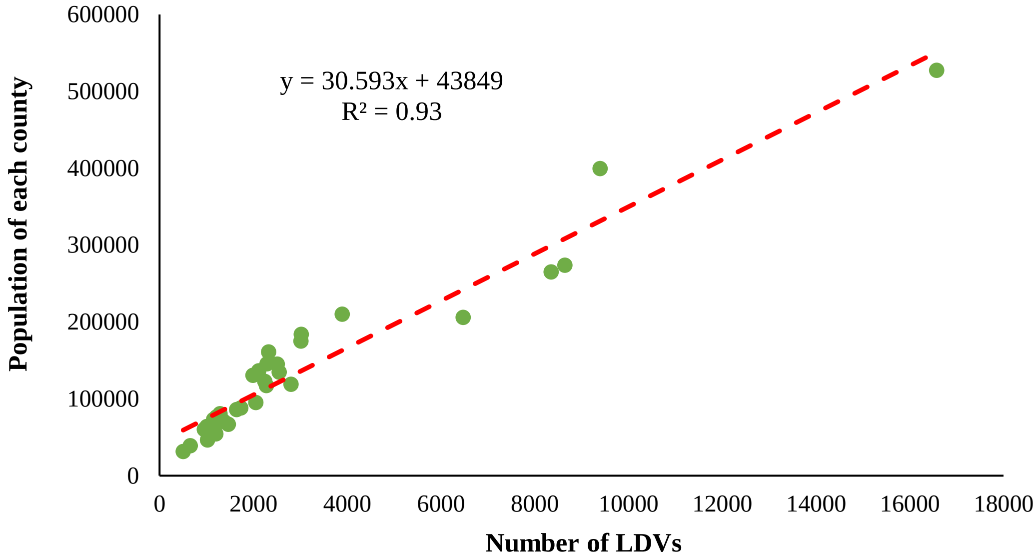


Figure 5.2. County specific total number of vehicles (PCs+LCVs) vs population

The county specific emission impacts were obtained by distributing the overall impacts by vehicle densities as shown in the following the equation,

$$\Delta EI_{ic} = \Delta EI_i * k_{ic} \tag{5.2}$$

Where, ΔEI_i is the emission impact of total excess NOx emissions in Ireland in the years i; ΔEI_{ic} is the emission impact of excess NOx in year i and county c; k_{ic} is spatial emissions distribution factor for county c in year i. This is derived by dividing the total number of vehicles in a county by the total number of vehicles in Ireland. Figure 5.3 shows the spatial distribution of LDV densities (i.e. the fraction of LDVs present in a specific county compared to the overall number of LDVs in Ireland) in the counties in Ireland. It can be observed from Figure 5.3 that the LDV density is highest in Dublin as it contains the highest share of LDVs in Ireland (DTTaS, 2015a).

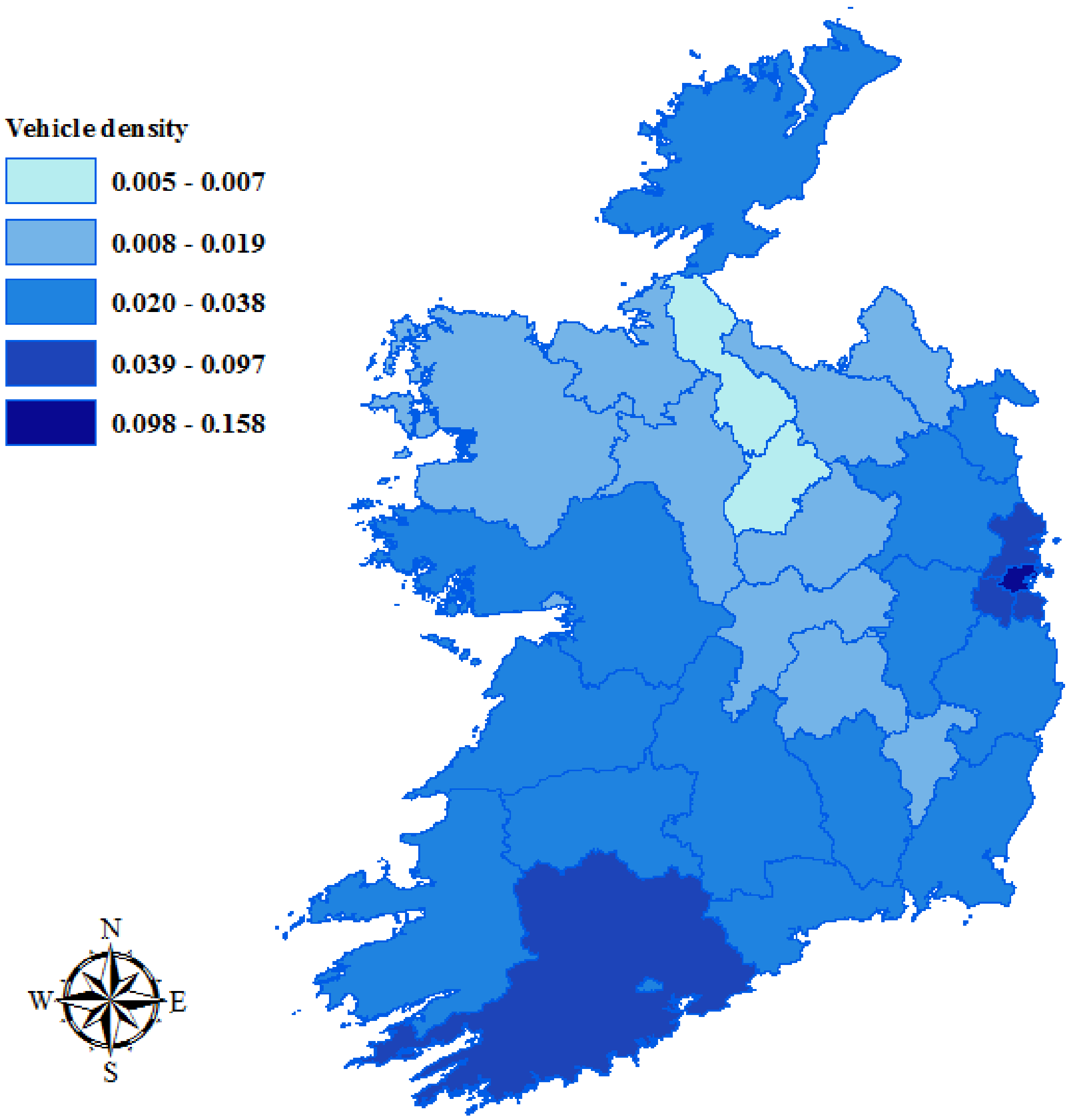


Figure 5.3. LDV densities in different counties in Ireland

5.4. Data description

Meteorological data, fleet data, speed data and mileage data were obtained from the same sources as described in Section 4.2. VW Euro 5 cars and fleet configuration were obtained from SIMI (The official statistics of the Irish Motor Industry, 2016). The total kilometre travelled by passenger cars were obtained from CSO (2014c) and it was divided by the total number of vehicles to obtain average annual mileage. Information on overall kilometres travelled were not available for 2015, thus mileage for 2015 was calculated by extrapolating previous years mileage data. Figure 5.4 shows the total number of diesel Euro 5 PCs and LCVs in Ireland and Figure 5.5 shows the AAMs for PCs and LCVs.

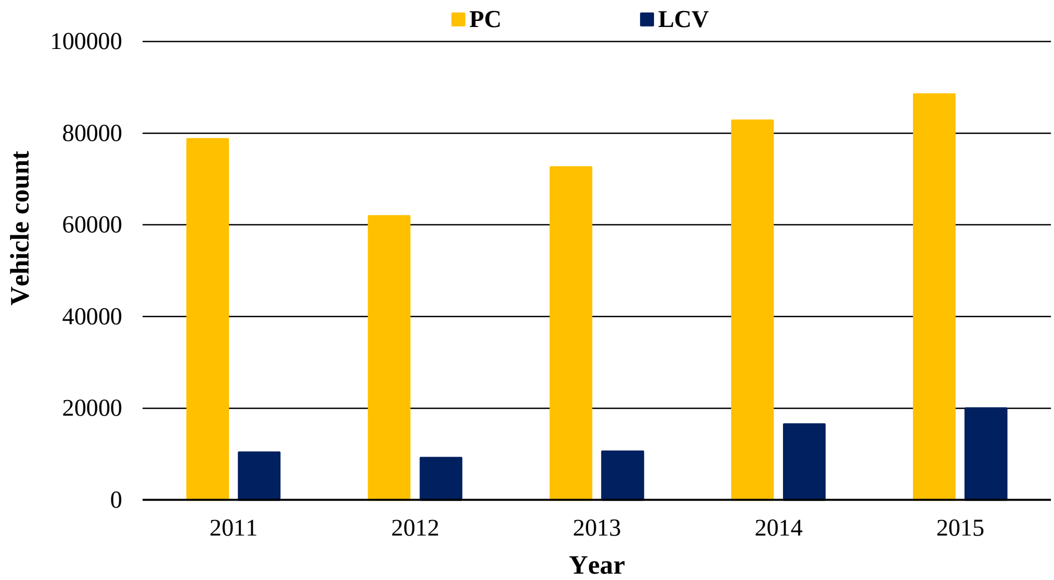


Figure 5.4. Total number of diesel Euro 5 LDVs

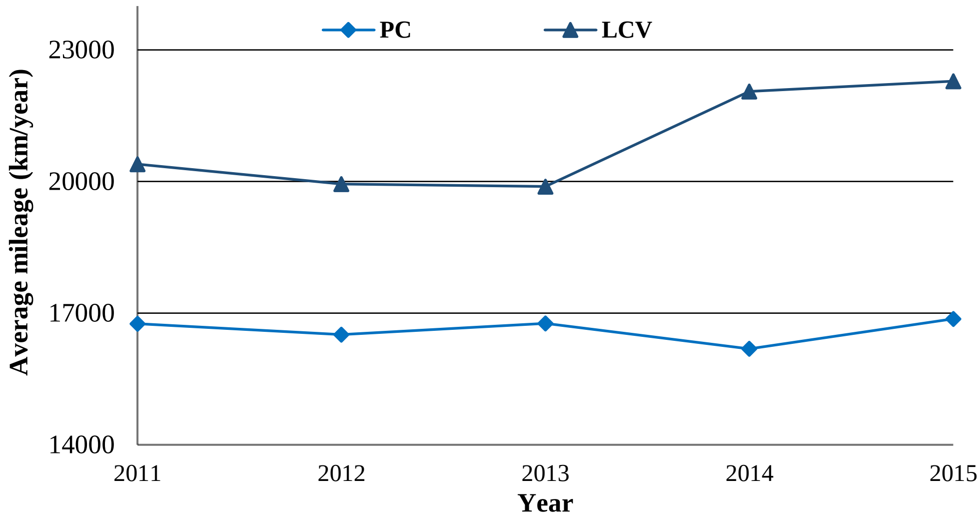


Figure 5.5. Annual average mileages (km/year) of PCs and LCVs

It is perceptible that the variation in average kilometre travelled per year is lesser compared to the variation in newly registered LDVs every year. The LCVs have increased at a higher rate with every year until 2014 while there is a sudden increase in average annual kilometre travelled after 2013. The number of affected diesel Euro 5 VW and Audi passenger cars introduced in Ireland per year are listed in Table 5.5. It can be observed from Table 5.5 that only the reported models by US EPA constitute a non-negligible share in the overall car fleet in Ireland.

Table 5.5. Number of affected VW and Audi vehicles

Year	VW		Audi		Total	The percentage in total fleet
	<=2.0 L	>2.0 L	<=2.0 L	>2.0 L		
2011	3,499	20	353	36	3,908	5
2012	3,543	21	351	38	3,953	6
2013	3,312	3	602	33	3,950	5
2014	4,088	2	1,939	247	6,276	8
2015	4,455	18	2,377	290	7,140	8

5.5. Results and Discussion

5.5.1. Existing NOx emission levels in Ireland

This section presents the emission levels for diesel Euro 5 LDVs in Ireland as calculated using COPERT 4 (v11.3) following the methodologies described in scenario 1A, 1B and 1C. Table 5.6 presents the overall quantity of NOx emissions (tonnes) estimated by COPERT and PEMS and the last column in Table 5.6 shows the Euro standard NOx values in the ideal case, i.e. NOx emissions if all the Euro 5 diesel passenger cars had followed the emission standard. A new version (5.1.1) of COPERT has been released with modified EFs for Euro 5 and Euro 6 PCs and LCVs. Improved hot EFs for NO_X were incorporated in COPERT 5 (v.1.1). Therefore, in this study, NOx emission levels were estimated using COPERT 5 as well and have been presented in Table 5.6 and 5.7. From Table 5.6, it can be observed that the differences between the model estimated values and PEMS estimates are significant (52%). The differences between these estimates might mislead the policymakers. However, COPERT 5 provides a more realistic estimate due to revision in EFs. There is a huge gap between the Euro standard and actual emissions. The emissions obtained via COPERT 4 and on-road measurements are 100-220% in excess than the euro standard NOx emission levels from diesel Euro 5 PC fleet. Table 5.7 summarises the overall model estimated and real-world emissions from Euro 5 LCVs along with the expected values, i.e. the total NOx emission from Euro 5 diesel LCVs if the standard emission specification was followed. The emissions of Euro 5 LCVs have been calculated with the new version of COPERT as well and shown in Table 5.7.

Table 5.6. Vehicle statistics and NOx emission levels for PCs

Year	Number of Vehicles	Emission levels (tonnes)			
		COPERT 4.11.3/laboratory test	COPERT 5.1.1	On-road (PEMS)	Euro Standard
2011	78,710	738	540	485 (-34%)	237 (-68%)
2012	61,910	572	395	376 (-34%)	184 (-68%)
2013	72,558	680	455	447 (-34%)	219 (-68%)
2014	82,777	746	511	491 (-34%)	241 (-68%)
2015	88,485	835	587	550 (-34%)	269 (-68%)
Total	384,440	3,571	2,488	2,349	1,150

Table 5.7. COPERT, on-road and lab test NOx emission levels for LCVs

Year	Vehicle count	Emission levels (tonnes)				
		COPERT 4 (v11.3)	COPERT 5 (v1.1)	On-road (PEMS)	Lab test	Euro standard
2011	10,355	194	344	252 (30%)	187 (-4%)	59 (-70%)
2012	9,159	168	297	218 (30%)	162 (-4%)	51 (-70%)
2013	10,536	192	340	249 (30%)	185 (-4%)	59 (-70%)
2014	16,457	332	588	431 (30%)	320 (-4%)	102 (-70%)
2015	19,942	408	723	529 (30%)	393 (-4%)	124 (-70%)
Total	66,449	1,293	2,293	1,678	1,247	395

From Table 5.7, it can be noticed that the real-world as calculated using PEMS measurements and model estimated emission quantities are about 200-500% higher than the anticipated values. Even though the model estimated urban and rural NOx EFs were observed to be more in lab test outcomes, combined emission amounts calculated by COPERT were estimated to be more than lab test results. On the other hand, the opposite pattern was observed when compared with on-road measurements. Emission values obtained from PEMS measurements were significantly higher than that estimated with COPERT. COPERT 4 (v11.3) underestimates and COPERT 5 (v1.1) overestimates the real-world emission by 23% and 37% respectively. However, in this study, all the driving shares were taken into account as realistically possible for Ireland. But these

PEMS measurements do not provide many details about the driving conditions as to where these PEMS EFs are applicable. COPERT estimates will vary widely depending on the driving mode share, average speed on each driving condition. Therefore, based on the results of this study it is concluded that COPERT 5 does not always overestimates the NOx emissions for LCVs. Moreover, COPERT 5 was found to estimate NOx emissions close to PEMS based on-road emissions in the case of PCs.

COPERT4 (v11.3) was used in Ireland for National Emission Inventory preparation as well as in scientific research. COPERT 5 became available in September 2016 with modified EFs following the research results (such as Ntziachristos et al., 2016) pointed out the necessary modification in NOx estimates by the previous version of COPERT 4. Thus, all the policy decisions in the country have been made based on the COPERT 4 emission factors as the real-world values. The discrepancy between the actual NOx discharge and model estimates should be accounted by the researchers and the policymakers, as it is likely to affect many areas given COPERT's extensive application in air quality and impact assessments, projections (energy, CO_2, pollutants), urban/regional inventories, new road (road section) construction etc. (Kouridis et al., 2014).

5.5.2. NOx emission levels from reported VW-Audi vehicles

In this section, the NOx emission levels calculated based on the methodology described in scenario 2A, 2B and 2C for the defective VW and Audi models in Ireland are provided. NOx emissions were calculated for the reportedly faulty PC models using COPERT 4. Emissions in scenario 2B and scenario 2C were estimated in the similar way as estimated for overall Euro 5 PC and LCV fleets. The expected NOx emissions as per Euro standard specification were calculated as well. Figure 5.6 shows the amount of NOx discharged from the US EPA reported faulty vehicles. It is observed from the results that the real-world emissions obtained from on-road measurements are almost double the Euro standard values. Reported models constitute about 6% of the total excess NOx exhausted by overall diesel Euro 5 vehicles. Thus, it is not only these vehicles but also several other models which are emitting more than the legal limit.

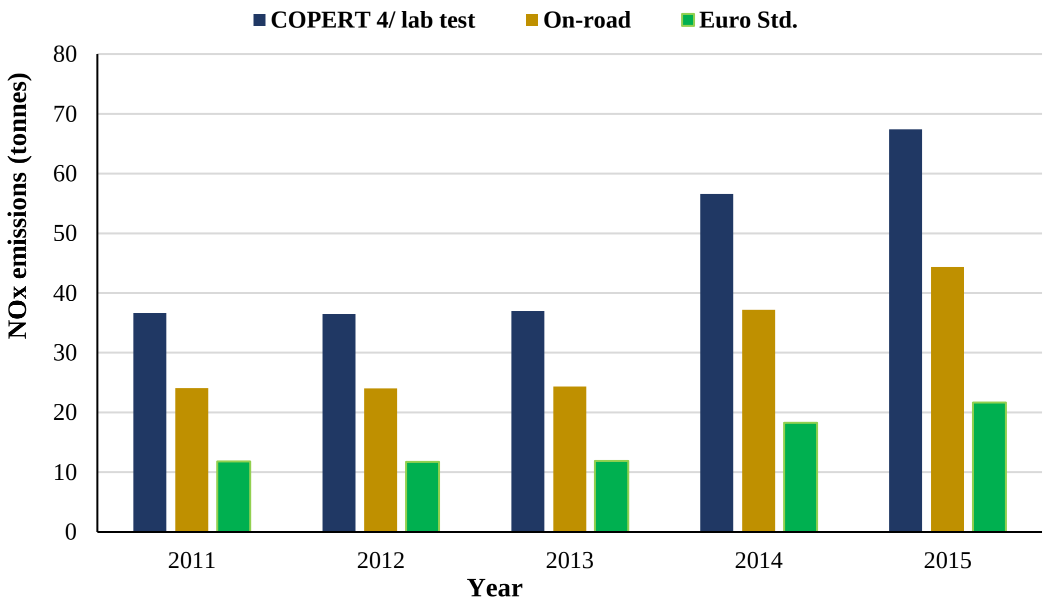


Figure 5.6. NOx emissions (tonnes) from affected VW passenger car models

5.5.3. Potential NOx emission levels from all VW-Audi vehicles

This section presents the NOx emissions resulting from all the VW and Audi Euro 5 LDVs from the assumption that all of them are cheating the emissions standards. Table 5.8 shows the total number of VW and Audi Euro 5 (2011-2015) diesel cars and LCVs that are present in Ireland.

Table 5.8. Possible faulty VW-Audi (PC+LCV) vehicles in Ireland

Year	Total PCs in the fleet			Percentage of total PC fleet	LCV	Percentage of total LCV fleet
	VW	Audi	Total		VW	
2011	8,299	3,037	11,336	14	1,858	18
2012	7,540	3,373	10,913	18	2,326	25
2013	6,792	3,432	10,224	14	2,084	20
2014	8,368	3,967	12,335	15	2,852	17
2015	10,284	4,435	14,719	17	3,217	16
Total	41,283	18,244	59,527	15	12,337	19

Figure 5.7. shows the NOx emission levels for all 59527 VW and Audi Euro 5 PCs considering if all Euro 5 VW-Audi PCs in Ireland are faulty. The PEMS estimated as well as modelled emission levels are significantly higher than those calculated by following the Euro standard EF for Euro 5 PC.

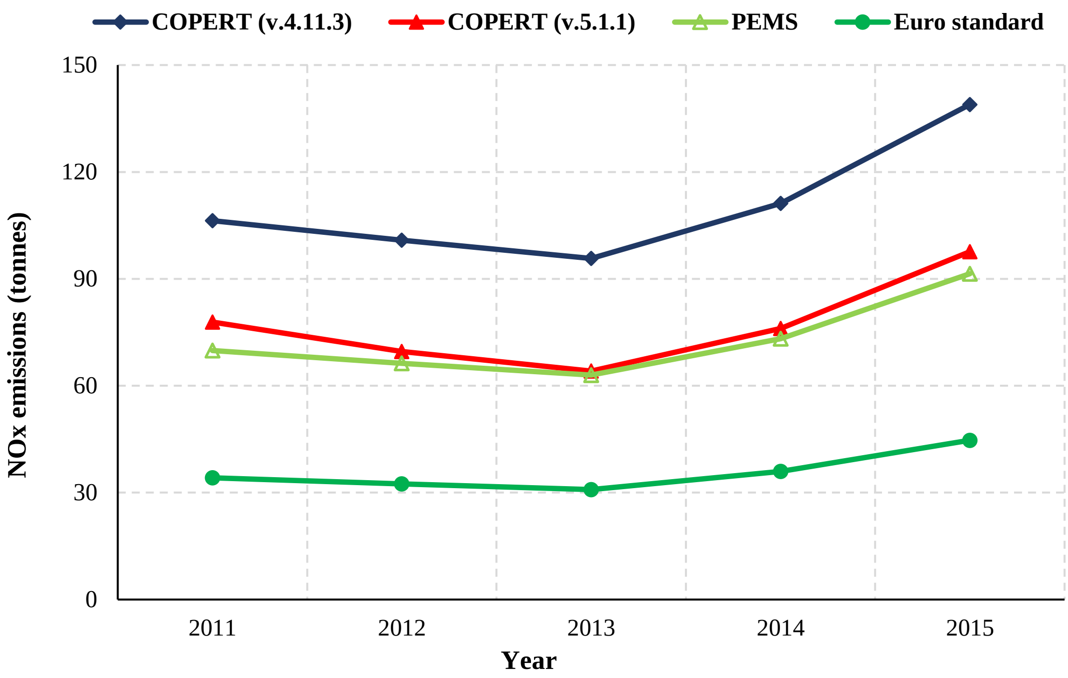


Figure 5.7. COPERT, PEMS and euro standard emission levels (tonnes) from VW and Audi PCs

PCs from VW and Audi alone were found to produce 15% of overall extra NOx emissions from entire diesel Euro 5 PCs. Figure 5.8 presents the NOx emission levels from 12337 VW Euro 5 LCVs in Ireland calculated using older and newer version of COPERT. The figure also shows the PEMS based on-road and laboratory test based real-world NOx emissions along with the Euro standard emissions. VW LCVs solely were found to contribute to a considerably large amount of NOx emissions which is 18% of the excess emissions produced by the overall LCV fleet in the country.

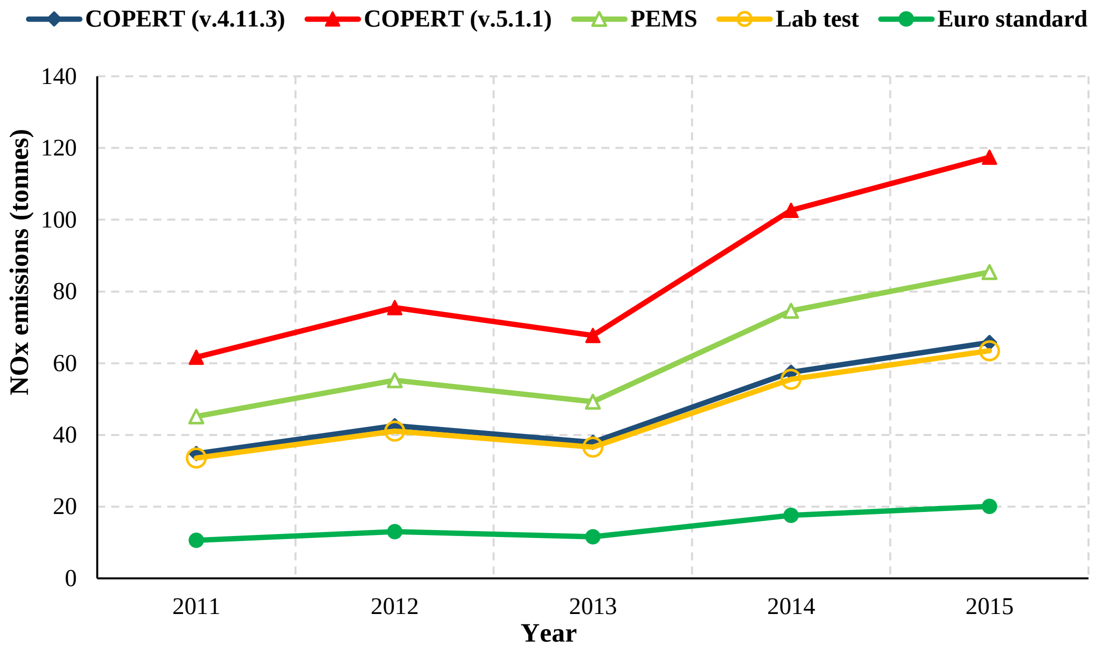


Figure 5.8. COPERT, PEMS, lab tests and euro standard emission levels (tonnes) from VW LCVs

5.5.4. Potential health and financial impact

This section presents the potential health and cost impacts of the hidden NOx from all the Euro 5 PCs and LCVs in Ireland. It has been assumed that all the vehicles that were newly registered over the period 2011-2015, remained in the market until 2015. Figure 5.9 shows the annual total excess NOx that was cumulatively exhausted by the Euro 5 PCs and LCVs relative to the expected Euro standard emission levels. The results presented in Figure 5.9 provides an overall picture of how much extra NOx emissions the potentially faulty vehicles have caused in Ireland since their introduction in the market until 2015. These excess NOx emission levels are calculated by subtracting Euro standard emission levels for PCs and LCVs from PEMS based on-road emission estimates presented in Tables 5.6 and 5.7.

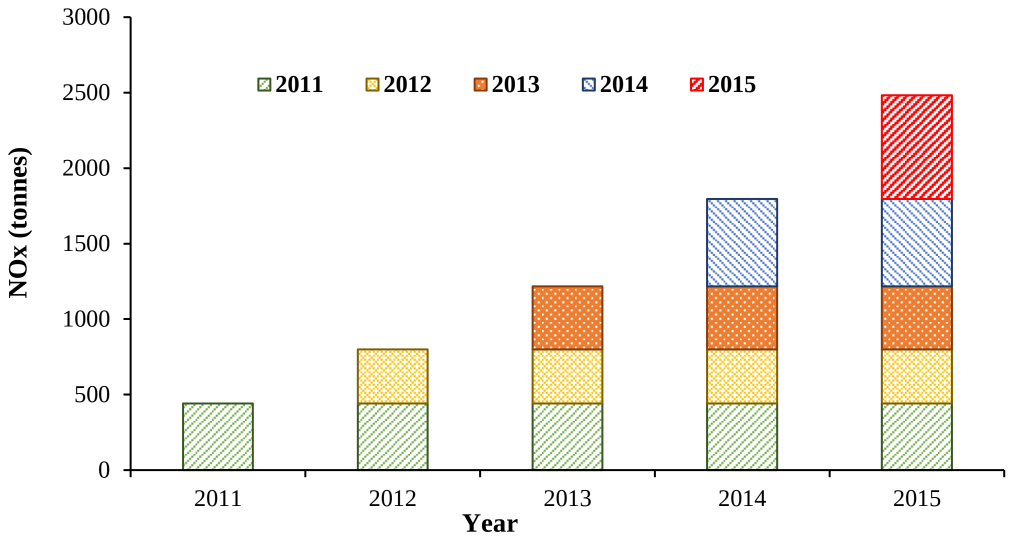


Figure 5.9. Total excess NOx emissions (tonnes) from all Euro 5 LDVs

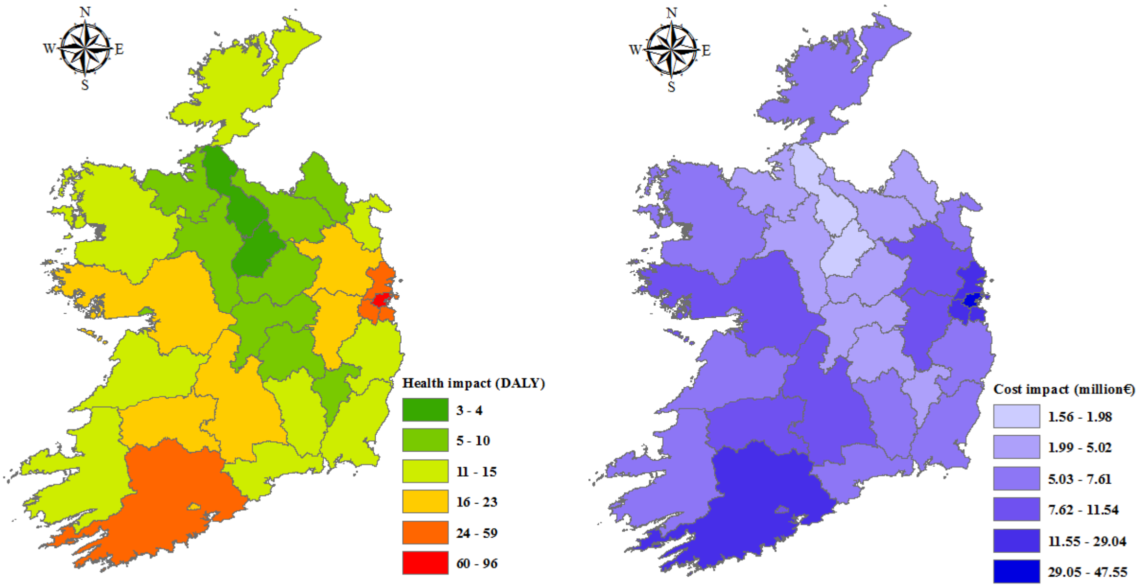
Figure 5.9 shows about 451,000 (section 5.4) of allegedly faulty vehicles have emitted 6,735.6 tonnes (Figure 5.9) of extra NOx to the atmosphere over the 5 years period. In Table 5.9 the annual extra NOx and their corresponding potential DALYs, mortality incidences and damage costs of the hidden NOx from Euro 5 LDVs are listed. The unit values used to access the BOD and mortality incidences are 90 DALYs/kt (Tang et al., 2014) and 10.23 incidences/kt (Oldenkamp et al., 2016) respectively. The VSL was taken as 3.75 million USD per premature death incident (WHO Regional Office for Europe, 2015) in Ireland and the damage cost was taken as €5,851 per tonne (DTTaS, 2016a) of extra NOx. It can be observed from the table (Table 5.9) that the potential faulty vehicles might have caused damage worth approximately €300 million.

Table 5.9. Potential health and cost damages due to extra NOx from faulty vehicles

Year	Extra NOx (kt)	Additional DALYs	Mortality incidences	Additional VSL (Million €)*	Additional Damage cost (Million €)
2011	0.44	39.64	5	18.75	2.57
2012	0.80	71.91	8	30.00	4.68
2013	1.22	109.59	12	45.00	7.14
2014	1.80	161.67	18	67.5	10.53
2015	2.48	223.40	25	93.75	14.51

*considering 1 US$ = €0.86 (27/09/2018)

Based on equation 5.2 the impacts were distributed among the counties in Ireland. Although this approach does not give the accurate information on spatial variation as the emission levels in different counties would depend more on total vehicle kilometres travelled rather than the total number of vehicles, however, due to lack of county specific mileage data, the emission levels were distributed based on the total number of vehicles in each county. This gives a rough idea about the county specific health and cost impacts. The spatial variations of health and cost impacts due to the excess NOx have been shown in Figure 5.10 (a) and Figure 5.10 (b) respectively.



(a) **(b)**

Figure 5.10. Spatial distribution of (a) health and (b) cost impacts due to excess NOx

It can be observed from the figures that the health and cost impacts are considerable, especially in the areas where vehicle densities are higher. If the faulty vehicles continue to be in use, the health impacts, especially in the urban areas will increase due to the higher impacts on those areas. Also, it is to be noted that the impacts are very high in Dublin, therefore, the air quality in Dublin must be examined at the population exposure level to assess if the levels are below the WHO and EU specified permissible limits or not. In Chapter 8 of this thesis, air quality in Dublin is examined, and based on that health and financial impacts are evaluated in Chapter 9.

5.6. Conclusion

This research work presented in this chapter aimed to quantify the disparity between the expected, modelled, on-road and simulated real-world NOx emission levels resulted from diesel Euro 5 PCs and LCVs that are present in Ireland. It can be observed from the NOx emission levels that even though emission standards have become tighter with every progressive Euro standard directive (e.g. NOx limit is 0.25g/km for Euro 4 and 0.18g/km for Euro 5), the on-road (PEMS based estimates) as well as real-world (laboratory test based simulation) emissions are more than the expected emission levels as per Euro standard specifications. This clearly indicates that Euro 5 vehicles are not obeying the respective standards. The reason/s behind this excessive discharge of NOx should be examined and measures should be taken to make sure that vehicles follow the Euro standard emission limits. Moreover, a significantly high deviation in NOx emission levels between COPERT 4 estimates and PEMS measurements based on-road estimates were observed. COPERT 4 (v.11.3) overestimated NOx emission levels from diesel Euro 5 PCs and underestimated the NOx emissions in case of LCVs.

Additionally, in this study, NOx emission estimates obtained from the revised version, i.e. from COPERT 5 (v.1.1), for diesel Euro 5 PCs and LCVs were compared to the COPERT 4, on-road and real-world estimates. It was found that for PCs, NOx outputs from COPERT 5 are close to on-road estimates recorded by PEMS. However, for LCVs, COPERT 5 estimates were higher. This does not essentially mean that COPERT 5 always overestimates the NOx emissions from Euro 5 diesel LCVs. Vehicular emissions depend on many factors, and as described in Chapter 3, it largely varies with average speed, average trip length and mileage shares. Higher number of COPERT runs with varying percentages of these input parameters should be run and compared to these on-road estimates to shed more light on the reason behind this discrepancy. Provided COPERT’s substantial applicability in many fields, this issue also should be examined further, and suitable measures should be implemented accordingly so that COPERT reflects the real emission as accurately as possible. On the other hand, it may be the case that COPERT 5 models the emission levels more accurately as it uses tier 3 methodology and it is an EU recommend model, compared to these PEMS based or lab test-based results as these experiments may have not been conducted on all the possible variations in vehicle make and model. Therefore, extensive testing on different Euro 5 and Euro 6 vehicles of various makes and models of vehicles are required to be carried out.

Even though few VW and Audi PC models have been proved to be cheating the NOx emission test cycle, the results show that the amount of PEMS estimated real-world NOx emissions for all diesel LDVs are huge. All potentially faulty diesel Euro 5 LDVs have emitted in excess of 6,740 tonnes of NOx in Ireland in the last 5 years (2011-2015 inclusive). The defective Euro 5 LDVs are probably responsible for 70 death incidences and approximately 606 DALYs. The average lifetime of a car in Europe is around 17 years (Transport and Environment, 2016) and this means that the Euro 5 LDVs are going to be excessively polluting for more than 10 years from now. Consequently, this will have huge financial and health impacts. Also, it is recommended that the tax incentives introduced to encourage the uptake of diesel vehicles should be revised accordingly. More incentives should rather be provided to buy electric vehicles, especially in the urban areas where the automobile density and resulting health impacts are higher. Also, policies to support phasing out of diesel vehicles as has been planned in other EU countries should be explored in the case of Ireland. For these reasons, Chapter 6 examines the expected emission levels from the predicted fleets under BaU and alternative scenarios supported through policies to reduce the emission levels with changes in fleet compositions. Then, in Chapter 7, policies which can potentially help in delivering those alternative scenarios and mitigate the emission levels are explored and potential changes in emission levels in different policy scenarios are evaluated.

Chapter 6: Potential Reductions in Vehicular Emissions in Ireland using Alternative Fuel Options

6.1. Introduction

In Chapters 4 and 5, the current state of vehicular emission levels in Ireland was evaluated and potential uncertainties associated with the estimated levels were quantified. The findings revealed that the actual emission levels are possibly very high than the anticipated levels. As mentioned in Chapters 1 and 2, the use of alternative fuels has become essential for environmental sustainability. In this chapter, potential emissions reductions from the increased use of alternative fuel and technology in road transport as a result of new policy measures have been examined. The policies are designed and discussed in Chapter 7.

In Ireland, the transportation sector has the largest share of primary energy demand and it is almost entirely (98%) dependent on oil (SEAI, 2014). Private cars have the highest share of energy use followed by road freight and aviation (SEAI, 2014). With the booming demand for transportation, especially car usage, incorporation of alternative fuel has become absolutely essential. In Ireland, approximately 70% of the GHG emissions from road transport come from PC fleet and PT bus fleets (Alam et al., 2015). This Chapter provides the existing emission levels from the PC and PT bus fleets in Ireland and potential emission reductions from changes in the PC and PT bus fleets with the increased uptake of alternative fuel and technology options. Additionally, this research will report a set of alternative PC and PT fleet compositions that will potentially help Ireland to meet its 2030 GHG emissions target which is to reduce GHG emission levels by 30% relative to its GHG levels in 2005. The GHG and other exhaust pollutants discharged from the passenger car fleet were calculated for the baseline scenario, 2015. Emission levels for future PC fleets in 2020, 2025, 2030, 2040 and 2050 were quantified under BaU situation and changes in emission levels compared to the base year 2015 were calculated. Different electric vehicle options, such as BEV, PHEV, HEV were examined in terms of their potential in reducing emission levels from PCs.

For PT bus fleet, several scenarios with different proportions of available alternative fuel and technology options such as electric, CNG and bio-CNG buses, were evaluated. PT bus fleet includes Dublin bus and bus Éireann which are the main public service bus operators in Ireland with a total of 1,441 buses in the 2015 fleet (National Transport Authority (NTA), 2016). In addition to examining alternative fuel options, newer technology options including Euro 6 and EEV (Enhanced Environmentally friendly Vehicle) buses were studied in terms of their potential in reducing emission levels. The scenarios are designed by considering the replacement of the present bus fleet with different percentages of available technology options (e.g. Euro 6, EEV) and fuel options such as CNG, bio-CNG and battery electric. The final energy consumptions in these scenarios were reported along with the total land area required to fulfil the annual bio-CNG energy demand to serve the PT bus fleet in Ireland.

Further to this, the required PC and PT fleet compositions were estimated to meet Ireland's GHG emissions target in 2030. This was achieved by calculating the target emission levels in 2030 from 2005 GHG levels and then backcasting the target emission levels to suggest the breakdowns of PC and PT fleet compositions necessary to meet the GHG emissions goal. In order to achieve this, two approaches were followed, first, fleet compositions for PC and PT were proposed such that it meets the emissions target separately by reducing 30% CO_2 emissions from each of these sectors and second, fleet compositions of PC and PT were determined so that 30% CO_2 emission levels are reduced from both the PC and PT fleets together. The GHG emitted from road transport mostly consists of CO_2 with a small percentage of N_2O (Nitrous Oxide), therefore, in this study, the CO_2 emissions were considered to represent the GHG emissions target from road transport. COPERT 5 was used to model the emission levels of the major air pollutants, namely, CO, CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC, N_2O, NMVOC emitted from the current fleet as well as from the alternative fleet scenarios. The damage cost of pollution for the current fleet and for the future PC and PT bus fleets were evaluated. The data and methods to estimate current and future emission levels are presented in the following section 6.2. Also, it includes the approach taken to design alternative scenarios to examine their potential in reducing future emission levels from PC and PT bus fleets. The findings in terms of emissions in alternative scenarios and discussions on the results are then presented in section 6.3. Alternative fuel/technology percentages required to be in the private car and bus fleets to meet 2030 GHG emissions goal are also presented in section 6.3. The conclusion of this study is then presented in section 6.4.

6.2. Methodology and data description

6.2.1. Passenger car scenarios

6.2.1.1. Baseline scenario

In this work, 2015 was taken as the baseline scenario and COPERT 5 was used to model the emission levels from the PC fleet in Ireland. As described in Chapter 2, COPERT 5 follows Tier

3 methodology which requires a detailed level of environmental information, fleet data, activity data in addition to requiring trip information and annual fuel consumption for different fuel types. The input data taken for the base scenario (2015) were same as described in detail in section 4.2. The breakdown of the PC fleet in Ireland in 2015, as per fuel type is shown in Table 6.1 (DTTaS, 2016). Current national CO_2 emissions were distributed with respect to the car densities in the counties. Greater Dublin Area (GDA) which is the region in Ireland comprising Dublin, Meath, Kildare and Wicklow, consists of about 40% of Ireland's population (CSO, 2011), and about 50% of the total PCs in Ireland (SIMI, 2017). Only three of the top 20 most densely populated areas in Ireland are located outside GDA (CSO, 2012). Therefore, the PC emissions share of GDA compared to the overall emission levels in Ireland was estimated in this study. This estimate is important for policy settings and emission reduction strategies as the GDA population will have pronounced effect of emissions due to having the largest share of population and automobile densities in Ireland. The fleet data for GDA were also extracted from the official statistics of the Irish Motor Industry (SIMI, 2017).

Table 6.1. 2015 PC fleet breakdown by fuel type

Fuel type	Petrol	Diesel	Hybrid petrol	Electric	Other
Percentage	55.38	43.57	0.51	0.05	0.49

6.2.1.2. Future BaU scenarios

The emission levels from the PC fleets in 2020, 2025, 2030, 2040 and 2050 following the BaU situation have been calculated using COPERT 5. BaU situation refers to the situation that is expected to exist without incorporation of any new policy measure other than the already existing ones. The BaU emission scenarios in 2020, 2025, 2030, 2040 and 2050 are referred as BaU_2020, BaU_2025, BaU_2030, BaU_2040 and BaU_2050 respectively. The population and car ownership levels in the future years were obtained from the Demographic and Economic Forecasting Report (NRA, 2014) for Ireland. Table 6.2 shows the forecasted population and car ownership in the future years based on which the emission levels in the future years were calculated. It can be observed that the percentage increase in population compared to 2015 level is 5.07% in 2030 but the percentage increase in car population is 28.7%.

Table 6.2. Predicted future population and car ownership levels

Year	Population (million)	Percentage change (%)	Car population (million)	Percentage change (%)
2015	4.677		1.985	
2016	4.773	2.1	2.023	1.9
2020	4.80	2.6	2.133	7.4
2025	4.91	5.0	2.312	16.5
2030	5.07	8.4	2.555	28.7
2040	5.30	13.3	2.906	46.4
2050	5.41	15.7	3.062	54.2

Whereas, in 2050 the predicted population increase in Ireland is 5.41% with the car ownership increase by 54.2%. Car compositions for future years under BaU were calculated using Systra's rolling_fleet_v7 model (DTTaS, 2015b). This model predicts future Irish fleet composition as per fuel type and technology class, and it was provided by DTTaS for use in this research. The predicted compositions in 2020, 2025, 2030, 2040, and 2050 are shown in Table 6.3.

Table 6.3. Passenger car fleet compositions with business as usual scenario in (a) 2020; (b) 2025; (c) 2030; (d) 2040; (e) 2050

Fuel type	**Year**				
	2020	2025	2030	2040	2050
Diesel	65.3	73.7	74.5	63.4	29
Petrol	33.1	24.8	24	20.2	9
Hybrid	1.5	1.4	1.3	1.2	1
Full Electric	0.1	0.1	0.2	15.2	61

The rolling fleet model predicts that the new car registration shares of EVs will be only 1.6% in 2025 and 2030, however, it will increase to 50.8% in 2040, and the new registration share of EVs will be 100% in 2050. The average annual future fleet mileages have been predicted by multiple regression model. Based on the past 12 years AAM data, AAM for the future fleets were predicted. As can be observed from Table 6.2, car population is expected to increase with the increase in population. Also, when AAMs were plotted against the number of cars, they were found to have a correlation. When the AAMs were modelled against both population and the total number of cars, a better fit was observed than when AAMs were modelled against the total number of cars alone. Therefore, the AAMs for the future car fleets were predicted based on past 12 years mileage data, and population and car ownership data from last 12 years as well as predicted future data (as shown in Table 6.2). Figure 6.1 shows the available AAM data and modelled mileages. It can be observed that the AAM has a decreasing trend which is due to the higher rate of increase in car ownership levels than the rate of increase in population. Table 6.4 shows the regression statistics for the modelled AAMs. The adjusted-R^2 calculates R^2 from only those variables whose inclusion in the model is significant. It is always recommended to calculate adjusted-R^2 in multiple regression. The R^2 and adjusted R^2 values of the mileage prediction model are significantly high which indicates the goodness of fit.

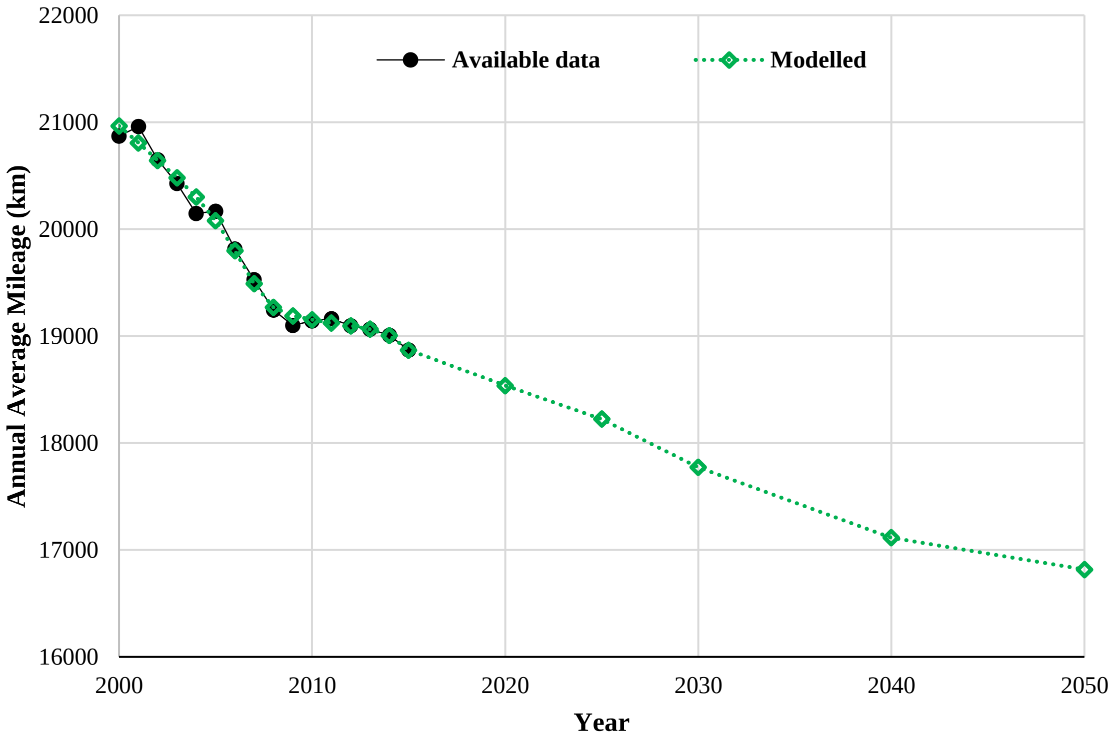


Figure 6.1. Annual average mileage of passenger cars in Ireland (km)

Table 6.4. Multiple regression Statistics

Multiple R	0.992753
R Square	0.985558
Adjusted R Square	0.982349
Standard Error	92.7079

6.2.1.3. Alternative scenarios

In order to examine the possible reduction in emission levels with the alternative fleet composition of PCs, three additional scenarios were designed. These scenarios were designed based on the assumption that new policy interventions will be implemented to encourage EV uptake and as a result, uptake of EVs will increase. Keeping Ireland's emissions goal for 2030 in mind, the emissions from the alternative PC fleet were computed for the year 2030. These three alternative scenarios were evaluated for low, medium and high EV purchase rates and referred to as 2030_low, 2030_medium and 2030_high, respectively. It has been assumed that the uptake will follow Logistic S-curve (Brady and O'Mahony, 2011; Smith, 2010). Therefore, the annual EV purchase rate is estimated by the following equations,

$$P_i = 1/\left(1+\exp(-\alpha*(Y_i - Y_0))\right) \quad (6.1)$$

Where,

$$\alpha = (\ln(1/P_1-1) - \ln(1/P_2-1))/(Y_2-Y_1) \quad (6.2)$$

and,

$$Y_0 = \ln(1/P_1)/\alpha + Y_1 \quad (6.3)$$

P_i is market penetration in year i
Y_i is the required year
Y_1 is the first year defined
Y_2 is the second year defined
P_1 is the expected penetration defined for Y_1
P_2 is the expected penetration defined for Y_2

EV market penetrations under 2030_low, 2030_medium and 2030_high scenarios were calculated from Equation 6.1 based on the following assumptions,

- 2030_low: 10% of total new PC registration in 2025 are EVs and 50% in 2030
- 2030_medium: 15% of total new PC registration in 2025 are EVs and 50% in 2030
- 2030_high: 25% of total new PC registration in 2025 are EVs and 50% in 2030

The S-curves showing the EV market penetration under the three hypothetical scenarios are presented in Figure 6.2. The number of total new car registration in each year has been taken as 110,000 based on past years data (SIMI, 2017). In 2030, with the low penetration scenario, the

EV will comprise about 10% of the overall car fleet and with medium and high penetration scenarios, the overall EV shares in the PC fleet will be 11.5% and, 16.4% respectively. Additionally, emissions were calculated for 2020 with the current Irish target which is to have 10,000 EVs in the PC fleet by 2020. The emissions were calculated under these scenarios in 2020 and 2030, and changes in emission levels were compared with 2015 levels.

The cost of health and other damages caused by the pollutants discharged by the PC and PT bus fleets were calculated by multiplying the total quantity of pollutants (tonnes) with unit damage cost per tonnes of the pollutant obtained from Handbook on External Costs of Transport (2014) and DTTaS (2016a).

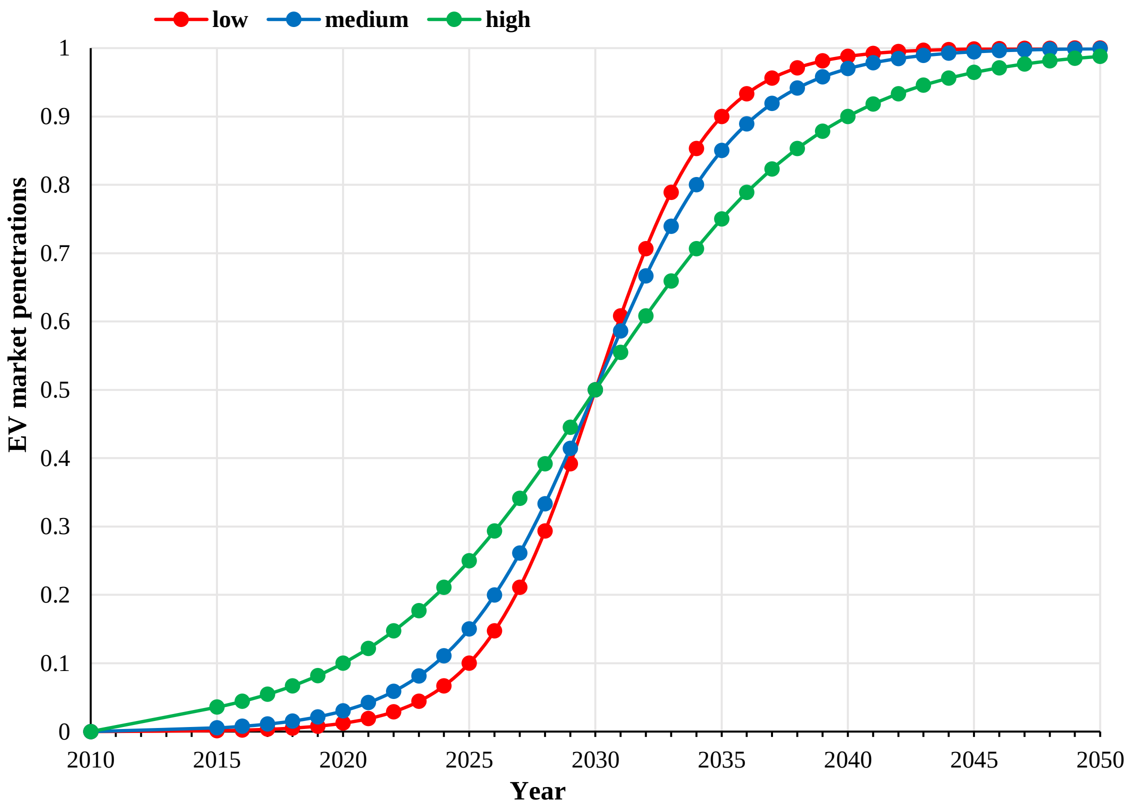


Figure 6.2. Market penetration pattern of electric vehicles

6.2.2. Public transport bus scenarios

The potential of PT bus fleet in reducing emissions has been assessed by designing four alternative scenarios in addition to examining the present emission levels from the current PT buses which use diesel and older engine technology classes for the entire fleet. The emission levels of CO_2, CO, NOx, $PM_{2.5}$, PM_{10}, N_2O, VOC, NMVOC were calculated in tonnes using COPERT 5. Emission calculation for buses using COPERT requires detailed input data in terms of fuel consumption, trip information (trip length, trip duration), activity (speed, mileage and mileage share), fleet configuration (number of buses of each fuel type and technology class) and environmental information (monthly average RH and monthly average minimum and maximum temperature). It is possible to calculate emissions from diesel, biodiesel and CNG buses in COPERT for the all the existing Euro technology classes. Table 6.5 presents a summary of the five scenarios that were examined in this study. These scenarios are described in detail in the following sections.

Table 6.5. Scenario descriptions for PT bus fleet

Scenario	**Technology class**	**Fuel type**	**Number of Buses**
1	Euro III	Diesel	666
	Euro IV	Diesel	218
	Euro V	Diesel	557
2	Euro VI	Diesel	1,441
3	Euro VI/EEV	CNG	1,441
4	Euro VI/EEV	Bio-CNG	1,441
5	Euro VI	Diesel	72
	Electric		1,369

- Scenario 1 (S1): In this scenario emissions were calculated for the base year, i.e. 2015 fleet. PT bus fleet in Ireland is diesel entirely powered and utilises engine of older euro technology classes which have higher emission factors. Dublin bus and bus Éireann being the dominant public service bus operators in Ireland have been considered in this study. The fleet data were obtained from Dublin Bus (2016) and the National Transport Authority (NTA, 2016). The present fleet composition corresponding to Euro class has been shown in Figure 6.3. For Dublin bus, the mileage share was taken as 100% urban, whereas, the mileage shares for bus Éireann were taken as 15% rural and 85% urban (NTA, 2016). COPERT 5 provides the scope of specifying the peak and off-peak driving percentages and corresponding average driving speeds. The urban share was further split into 50% for peak and 50% for off-peak hours with average peak hour speed taken as 13 kmph and average off-peak hour speed as 26.5 kmph (CSO, 2014a; Ryan and Caulfield, 2010; Alam et al., 2015; RSA, 2015). For Bus Éireann, average rural speed was assumed to be 40 kmph. AAMs were taken as 57,288km and 71,074km for Dublin bus and bus Éireann respectively (NTA, 2016).

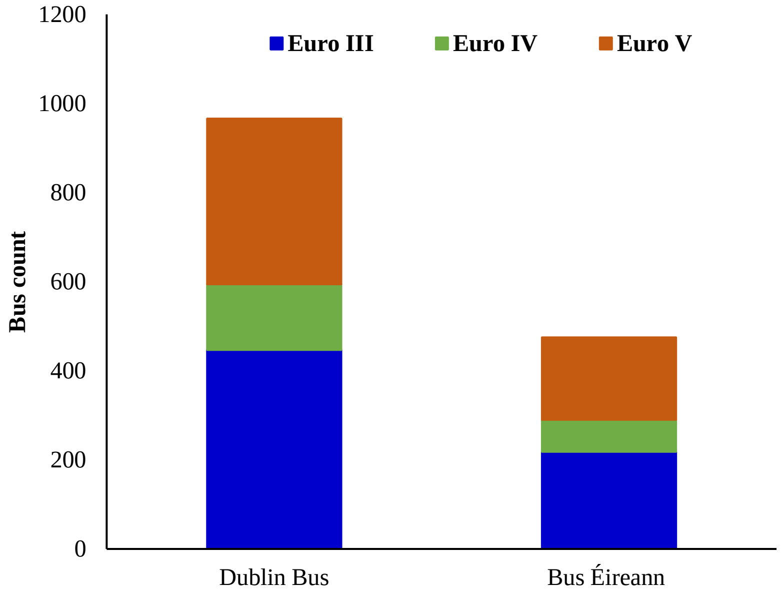


Figure 6.3. PT bus fleet composition as per technology class in 2015

- Scenario 2 (S2): This scenario presents the emission levels considering that all the buses are replaced by Euro VI diesel buses. Euro VI has improved technology, especially in terms of lower emission factors for NOx, PM, and VOCs. Emissions in this scenario were calculated using COPERT 5 with the entire fleet specified as Euro VI, whereas, the rest of the input parameters were taken as described in scenario 1.
- Scenario 3 (S3): This scenario tests the emissions reduction by replacing both fuel and vehicle technology. The emission levels resulted from the PT bus fleet if all the buses are replaced by EEV/Euro VI CNG were estimated. It was found that EEV and Euro VI have the same emission factors. Emission levels in this scenario were also calculated using COPERT 5 with other input parameters, except fuel and technology class, the same as for scenario 1.
- Scenario 4 (S4): In this scenario emissions were calculated based on the assumption that all PT buses are replaced by Bio-CNG Euro VI/EEV buses. Therefore, this scenario also assumes replacement of both fuel and engine as considered in S3. Grass silage was chosen to be the optimum feedstock to produce bio-CNG in Ireland and carbon neutrality of bio-CNG was taken as 60% (Ryan and Caulfield, 2010). The land area and feed stock required to meet the annual demand for bio-CNG were calculated.
- Scenario 5 (S5): This scenario evaluates the possible emission savings by replacing urban bus fleets with electric buses. Only urban PT bus fleet was considered due to the shorter driving range of electric buses varying from less than 100 km to 200 km depending on the duration of charging and efficiency of charging station (FCH-JU, 2012). In this scenario, the WTT emissions i.e. the emission due to electricity generation was calculated for two cases. The first

case assumes the energy source to be electricity generated from renewable sources which have WTW emissions of 20g eCO_2/km and the second case assumes that the required electricity comes from EU-mix with GHG emission rate 720g eCO_2/km (Mahmoud, 2016).

Additionally, the potential damage costs of pollution as a result of the emissions from PT bus fleet in Ireland and potential savings with the alternative fuel use were calculated.

6.2.3. PC and PT bus scenarios to meet the emissions target

As there is no separate target specific for road transport sector in Ireland, this study assumes the GHG emissions target for road transport is the same as the overall target for all sectors which is to reduce emission levels in 2030 by 30%, relative to 2005. In order to investigate the required PC and PT bus fleet compositions that will help Ireland to meet the 2030 emissions target, a set of scenarios were designed and evaluated. The target emission level in 2030 was determined by reducing the 2005 GHG emissions from road transport by 30%. The emissions in 2005 were obtained from EPA (2017). Based on the present CO_2 emissions shares (Alam et al., 2015) of cars and buses, target CO_2 emissions of the PC and PT bus sectors were estimated. The passenger car CO_2 emission share was taken as 67.5% of the total road transport emissions share and the same for Dublin bus and bus Éireann fleets was taken as 1% (Alam et al., 2015). The PC and PT bus fleet compositions that will result in the desired CO_2 emission levels in 2030 were then determined. Two approaches were taken to examine the fleet breakdowns, as described below,

- Approach 1:- Meeting target separately, i.e. reduce CO_2 emission levels from each PC and PT bus fleets separately by 30% in 2030, relative to 2005 emissions.
- Approach 2:- Meeting target combined, i.e. reduce total CO_2 emission levels from PC and PT by 30% in 2030, relative to 2005 emissions.

6.3. Results and discussion

6.3.1. Emissions from PC scenarios

6.3.1.1. Emissions from the current PC fleet (2015)

This section presents the emissions resulted from the existing PC fleet in Ireland as calculated using COPERT 5. Table 6.6 shows the total emissions from the passenger car fleet and disaggregated emission levels with respect to fuel type and engine class.

Table 6.6. Emission levels (tonnes) from 2015 passenger car fleet

Pollutant	Engine Size	Emissions (tonnes)			
		Petrol	Diesel	Hybrid Petrol	Total
CO_2	<1.4L	1,087,881	212,876	1,601	6,095,743
	1.4-2.0L	1,512,624	2,690,937	8,647	
	>2.0L	130,530	446,053	4,595	
CO	<1.4L	15,294	66	11	35,558
	1.4-2.0L	17,829	1,109	59	
	>2.0L	1,031	137	22	
NOx	<1.4L	713	752	0.5	13,766
	1.4-2.0L	1,049	9,986	2.6	
	>2.0L	72	1,189	1.3	
$PM_{2.5}$	<1.4L	70	31	0.1	772.1
	1.4-2.0L	83	510	0.6	
	>2.0L	6	73	0.3	
PM_{10}	<1.4L	120	41	0.2	1,039
	1.4-2.0L	142	635	1.2	
	>2.0L	10	88	0.6	
N_2O	<1.4L	16	9	0.02	182.18
	1.4-2.0L	23	118	0.11	
	>2.0L	2	15	0.05	
NMVOC	<1.4L	986	6	1	2,717
	1.4-2.0L	1,470	139	7	
	>2.0L	76	28	4	
VOC	<1.4L	1,123	6	1	3,058
	1.4-2.0L	1,649	150	7	
	>2.0L	89	29	4	

It can be observed that passenger cars alone contributed to about 6.1 megatonne of CO_2. Also, PC contribute to 13.8 kt of NOx and 772.1 tonnes of $PM_{2.5}$ both of which have a severe impact on human health, especially in urban areas. It can also be observed that most of this NOx and $PM_{2.5}$ emissions come from diesel vehicles. It is to be noted that although Ireland has carbon-based tax to incentivise users not to buy users of buying petrol vehicles, the CO_2 emission levels from diesel vehicles are more than petrol vehicles. This points out the need to revise the current vehicle taxation system and the introduction of new measures.

Total CO_2 emissions from PCs were distributed among 34 counties and plotted using GIS (Geographic Information System). Spatial distribution of CO_2 emissions, among the counties in

Ireland is shown in Figure 6.4. It can be seen from the figure that after Dublin, the CO_2 levels are highest in county Cork followed by Meath and Kildare. It was found that GDA shares 63%, 48%, 50%, 44%, 40%, 59%, 59%, 52% of overall Ireland's CO, CO_2, NO_X, $PM_{2.5}$, PM_{10}, VOC, NMVOC and N_2O emissions respectively. This indicates a higher impact of pollution on the GDA population than the rest of the country.

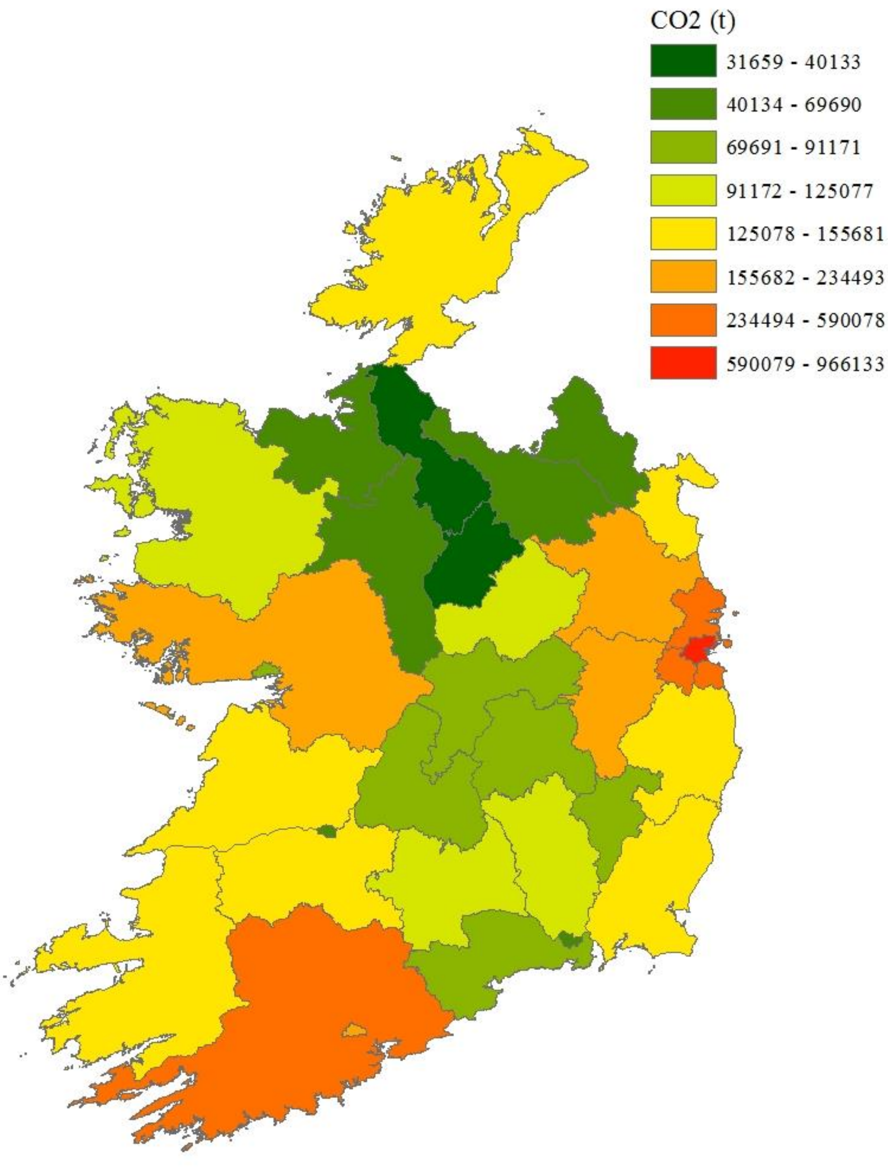


Figure 6.4. Spatial distribution of annual CO_2 emissions

6.3.1.2. Emission levels in 2020, 2025, 2030, 2040, 2050 from Passenger car fleet with BaU situation

This section presents expected emission levels from the PC fleet in 2020, 2025, 2030, 2040 and 2050 under BaU situation. The changes in emission levels of all the major air pollutants in the future years with respect to 2015 have been presented in Table 6.7. The CO_2 emission from electricity generation was taken as 582 gCO_2/kWh and electricity requirement per kilometre was taken as 0.15 kWh (SEAI, 2017b). Thus, WTT CO_2 emissions for BEVs was taken as 87.30g/km. Taking into account the combined measures of increasing the share of renewable generation and improvements in the overall efficiency of the electricity supply, it is assumed that the carbon intensity of the electricity supply will be 393g CO_2/kW h in 2020. In this case, the WTT CO_2 emissions will be 58.95g/km. WTT emissions for BEVs for both the cases were calculated and the differences in CO_2 emission levels have been reported (Table 6.7).

Table 6.7. Changes in emission levels in future years with business as usual scenario

Pollutant	2015 emissions (tonnes)	Percentage change from 2015				
		BaU_2020	BaU_2025	BaU_2030	BaU_2040	BaU_2050
TTW CO_2	6,095,743	22	28	39	28	-39
WTW CO_2		22	28	39	41	13
WTW CO_2 (with renewable electricity)		22	28	39	37	-4
CO	35,558	-49	-65	-63	-67	-84
NOx	13,766	36	49	55	44	-32
$PM_{2.5}$	772.1	4	-19	-22	-28	-66
PM_{10}	1,039	10	-5	-4	-11	-58
N_2O	182.18	28	36	41	30	-38
NMVOC	2,717	-50	-66	-65	-68	-84
VOC	3,058	-49	-64	-63	-66	-84

It can be observed from the results that in 2020, CO_2 emissions from private car fleet is expected to increase by 22% under BaU situation, whereas, it is expected to increase by 39% in 2030. Even with the renewable electricity production, emission levels are not expected to decrease due to the increase in car ownership and no significant increase in electric vehicle share. However, it can be noticed that there is 36% and 55% increase in NOx emission levels in 2020 and in 2030 respectively. The highest reduction in future fleets even with BaU can be seen in CO, VOC and NMVOC levels. This reduction can be attributable to the reduced share of petrol cars in the overall fleet. This is due to increase in diesel vehicles in the fleet. Overall, it can be seen that with the continuation of the BaU situation, CO_2, NOx and N_2O levels will increase until 2040 and a significant change can only be seen in 2050.

6.3.1.3. Emission levels in 2020 and 2030 from PC fleet with alternative scenarios

The potential changes in emission levels in 2030 from alternative scenarios, with low, medium and high EV market penetrations are presented in this section. Emission levels with the current target which is to have 10,000 EVs in the car fleet by 2020 also were calculated. Table 6.8 shows the changes in emission levels of CO_2, CO, NOx, $PM_{2.5}$, PM_{10}, N_2O, VOC and NMVOC from increased EV uptake, compared to 2015 emission levels. Table 6.8 also includes the CO_2 emissions from both the renewable and non-renewable source of electricity generation separately. The results show that in 2020, with 10,000 EVs in the fleet, the emission levels are not improved compared to the BaU_2020 scenario.

Table 6.8. Changes in emission levels in future years with incentivised alternative scenarios

Pollutant	2015	Percentage change from 2015			
		2020_10000 EVs	2030_low	2030_medium	2030_high
TTW CO_2	6,095,743	21	27	24	19
WTW CO_2		22	33	32	29
WTW CO_2 (with renewable electricity)		22	31	29	26
CO	35,558	-50	-72	-76	-81
NOx	13,766	36	53	52	36
$PM_{2.5}$	772.1	4	-28	-29	-32
PM_{10}	1,039	10	-12	-13	-17
N_2O	182.18	28	36	38	37
NMVOC	2,717	-50	-74	-78	-83
VOC	3,058	-49	-72	-77	-82

It can be observed from both the tables (Table 6.7 and Table 6.8) that with the BaU, the CO_2 emission levels are expected to increase by about 39% in 2030 with respect to 2015 but with the high market penetration, the increase in emission levels can be brought down by 20%. However, it is expected to increase by 19% from 2015 CO_2 levels. This also indicates that it is important for people to use active mode of transport and public transportation modes so that the total kilometres travelled by cars reduce. It can be noticed that due to technological improvements of

ICEV engines (i.e. more Euro 6 in the fleet) other pollutant levels have reduced, however, there is an increase in NOx levels because of *dieselgate* and increase in a number of diesel cars in the fleet.

6.3.2. Emissions from PT bus scenarios

This section presents the emission levels resulting from the existing PT bus fleet in Ireland and potential emissions savings from changing to alternative fuel and technology. Table 6.9 presents the emissions from the status quo and percentage change in the designed scenarios with respect to the base scenario.

Table 6.9. Emissions (tonnes) from the designed scenarios and their differences over base scenario

Pollutants	**Emissions (tonne)**	**The difference with base (%) in each scenario**			
	S1	**S2**	**S3**	**S4**	**S5**
CO	244.86	-88	-61	-61	-97
CO_2	99,185.35	-5	8	-57	-94* (-35**)
NOx	900.57	-94	-57	-57	-97
N_2O	1.47	137	-100	-100	-100
VOC	29.71	-85	216	216	-97
NMVOC	25.46	-84	-42	-42	-97
$PM_{2.5}$	16.07	-77	-74	-74	-99
PM_{10}	19.44	-64	-61	-61	-98

*Percentage decrease in CO_2 levels when the renewable source of electricity is used; ** Percentage decrease in CO_2 emissions when electricity from EU-mix is used as a source of electricity

The results indicate that if the fleet is renewed with Euro 6 technology classes, the NOx levels will improve by a huge percentage (97% reduction) compared to the present scenario. However, improvement in CO_2 levels is not significant. Also, the results reveal that bio-CNG provides a better scope of CO_2 reduction than electrification of the urban PT bus fleet if the electricity production is not renewable based. Table 6.10 presents the energy consumption in all the scenarios. Electricity requirements for buses in scenario 5 were calculated assuming the WTW energy consumptions as 18.66MJ/km and 10.33MJ/km for the energy sources being electricity from EU-mix and renewables respectively.

It can be seen that CNG buses have the highest energy requirement, whereas, a substantial reduction in energy consumption is possible in S5 for renewable based electricity. Scenario 4, which considers the alternative fuel option as bio-CNG and technology class as Euro 6/EEV for PT bus services, is a very suitable option for Ireland by utilizing the agricultural grass silage as feedstock in producing biomethane (Smyth et al., 2009).

Table 6.10. Energy consumption by PT bus fleet in 2015

Scenarios	**S1**	**S2**	**S3**	**S4**	**S5**	
Fuel consumption (TJ)	1,875	1,809	2,602	3,411	75	
Electricity consumption (TJ)	-	-	-	-	**EU-mix**	**Renewable**
					1,568	868

This study calculates the land area requirement if grass silage is considered to generate bio-gas to be used as a fuel option for PT buses. Table 6.11 presents the land area calculation based on diesel energy density and reports the final land area requirement for grass silage to satisfy the annual energy demand.

Table 6.11. Land area calculation using quantity and energy density of diesel

Parameter	**Unit**	**Value**
Diesel	litre	55,670,000
Diesel energy by volume	MJ/litre	33.7
Diesel	GJ	1,876,079
The energy density of methane	MJ/m^3	37.78
Methane	m^3	49,657,994
Biogas	55% of CH_4	90,287,261
Biogas yield	m^3/t of silage	123
Mass of silage	t	734,043
Silage yield	t/ha of land	60
land area	ha	12,234
Life Cycle Analysis land area	ha	21,091

The parasitic energy demand was taken as 42% of the total demand (Smyth et al., 2009) and based on this, final Life Cycle Analysis land area requirement was determined (Table 6.11). The energy value of methane was taken as 37.78MJ/m^3 (Smyth et al., 2009). Thus, 21,091ha land area is needed to fulfil the energy demand of bio-CNG for PT bus fleet in Ireland.

6.3.3. Cost implications

The cost impact of pollution from PC and PT bus fleets are presented in this section. Table 6.12 shows the damage cost of pollution from the existing PC fleet in 2015 and potential changes in the cost impact in future years under BaU scenarios.

Table 6.12. Damage costs of pollution from PC fleet in the base scenario and potential changes in BaU future scenarios

Pollutants	Damage cost (mil€)	Changes in damage costs relative to damage costs in 2015 (mil€)				
		BaU				
	2015	2020	2025	2030	2040	2050
TTW CO_2	80.59	17.73	22.56	31.43	22.56	-31.43
WTW CO_2		17.73	22.56	31.43	33.04	10.48
WTW $CO_{2renewable}$		17.73	22.56	31.43	29.82	-3.22
NOx	80.54	29.00	39.47	44.30	35.44	-25.77
$PM_{2.5}$	87.55	3.50	-16.63	-19.26	-24.51	-57.78
PM_{10}	19.89	1.99	-0.99	-0.80	-2.19	-11.54
VOC	4.40	-2.15	-2.81	-2.77	-2.90	-3.69
NMVOC	3.80	-1.90	-2.51	-2.47	-2.58	-3.19
Total (TTW CO_2)	276.77	48.16	39.08	50.43	25.82	-133.4
Total (WTW CO_2)		48.16	39.08	50.43	36.29	-91.5
Total (WTW $CO_{2renewable}$)		48.16	39.08	50.43	33.07	-105.2

It can be noticed that the existing PC fleet has caused damage of approximately €277 million in Ireland in 2015. Out of this total damage, $PM_{2.5}$ bears the maximum damage cost followed by CO_2 and NOx. However, in 2030, damage cost from $PM_{2.5}$ is expected to reduce by approximately 20% from 2015 impacts, whereas, NOx alone will lead to a rise in the annual cost impact by €44.3 million out of the total increase of €50.43 million. This indicates the necessity of particular attention to examine ways and policies to lower diesel use. Also, it can be seen that it is not until

2050 when these impacts can be improved significantly. Table 6.13 shows the potential changes in damage cost levels from PC fleets in 2030 with changes in PC fleet.

Table 6.13. Damage costs of pollution from PC fleet in the base scenario and potential changes in alternative scenarios

Pollutants	Damage cost (mil€)	Changes in damage costs relative to damage costs in 2015 (mil€)			
		2020	2030		
	2015	10000 EVs	low	medium	high
TTW CO_2	80.59	16.92	21.76	19.34	15.31
WTW CO_2		17.73	26.59	25.79	23.37
WTW $CO_{2renewable}$		17.73	24.98	23.37	20.95
NOx	80.54	29	42.69	41.88	29.00
$PM_{2.5}$	87.55	3.50	-24.51	-25.39	-28.02
PM_{10}	19.89	1.99	-2.39	-2.59	-3.38
VOC	4.40	-2.15	-3.17	-3.39	-3.61
NMVOC	3.80	-1.90	-2.81	-2.96	-3.15
Total (TTW CO_2)	276.77	47.36	31.57	26.90	6.15
Total (WTW CO_2)		48.16	36.40	33.35	14.21
Total (WTW $CO_{2renewable}$)		48.16	34.79	30.93	11.79

The results show that, in the 2030_high scenario, i.e. with 16% EVs in the fleet, the damage costs can be saved by €36.22 to €38.64 million (for an increase in total damage costs by €11.79 to €14.21 million from 2015, depending on the source of electricity production) compared to the BaU_2030 scenario. Even the 2030_low scenario offers a reduction of €14.03 to €15.64 million. The damage costs caused by the emissions have been shown in Table 6.14 along with the possible savings if alternative fuel and technology options are implemented.

Table 6.14. Damage costs of pollution from PT bus in the base scenario and potential changes in alternative scenarios

Pollutants	Cost of emissions (€)	Changes in damage costs relative to damage costs in S1 (€)			
	S1	S2	S3	S4	S5
CO_2	1.31	-0.06	0.11	-0.74	-1.26
NOx	5.27	-4.97	-3.02	-3.02	-5.11
VOC	0.04	-0.04	0.09	0.09	-0.04
NMVOC	0.04	-0.03	-0.02	-0.02	-0.03
$PM_{2.5}$	3.06	-2.35	-2.27	-2.27	-3.02
PM_{10}	0.37	-0.24	-0.23	-0.23	-0.37
Total (Mil€)	-10.09	-7.69	-5.34	-6.19	-9.83

It can be observed that the pollutants from the PT bus fleet alone have caused damage worth €10.09 million in 2015. Scenario 5 offers the highest annual saving in damage costs and scenario 3 offers the lowest possible savings.

6.3.4. Emission projections and compliance with EU's 2030 targets

This section presents the PC and PT fleet compositions that will potentially help in reducing 30% emissions from PCs and PT buses in 2030. The analysis was carried out on the basis of two approaches as described in section 5.2.3, and the results are presented in the following sections.

6.3.4.1. Approach 1- Meeting target combined

This section presents the PC and PT fleet compositions which jointly will help to meet the emission goal in 2030. This allows emissions from PC to be offset by PT and vice versa. It was found that the desired car fleet to meet the EU 2030 emissions target should be as shown in Figure 6.5.

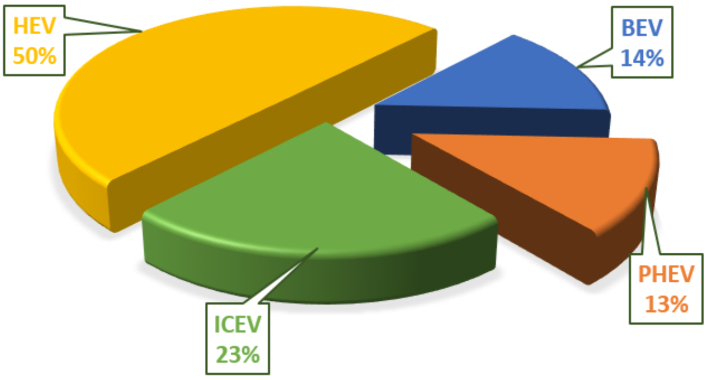


Figure 6.5. Required passenger car fleet composition to meet 2030 target

Based on the previous PT data, it was assumed that there will be a 15% increase in the number of PT buses in 2030. Assuming the life of a public service bus as 15 years the fleet composition for 2030 was estimated. Figure 6.6 shows projected 2030 PT bus fleet (Dublin Bus and Bus Éireann) compositions as per technology class.

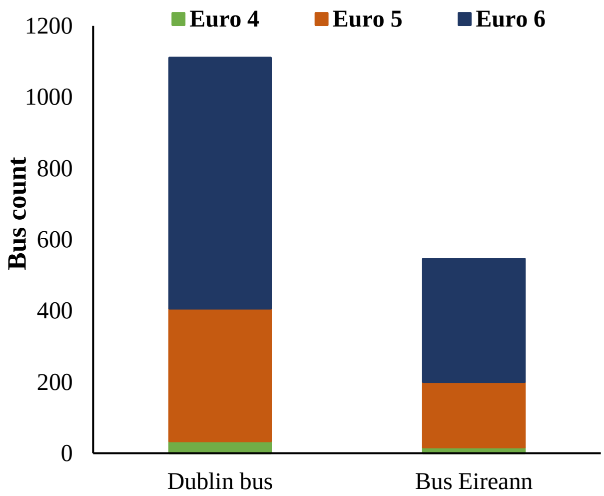


Figure 6.6. PT bus fleet composition in 2030 as per technology class

It was estimated that in order to reduce the combined emissions from PC and PT bus fleets by 30%, urban bus fleet should be replaced by electric buses. CO_2 emission levels in 2005, target emission levels in 2030 and potential reductions in CO_2 emissions from PC and PT bus fleets are presented in Table 6.15. It can be observed that a huge number of electric vehicles are to be deployed to reduce the GHG emission levels by 30% in 2030.

Table 6.15. Emission levels (tonnes) and potential reductions in 2030 relative to 2005 with the proposed fleet composition

Road Transport Mode	2005	Target year: 2030	
	CO_2 emissions (tonne)	CO_2 emissions (tonne)	% decrease relative to 2005 levels
Car	8,474,539	5,982,034	-29
Bus	118,330	7,913	-93
Total	8,592,869	5,989,947	-30

6.3.4.2. Approach 2- Meeting target separately

This section presents the required PC and PT fleet compositions that will potentially lead to reduce the CO_2 emissions individually from PC and PT sector by 30% in 2030, compared to the CO_2 emission levels in 2005.

Passenger car fleet:

The results presented in the previous section shows that with the increase in car ownership levels, the only way to achieve the emissions goals is a high level of electrification in the PC fleet. This section examines a more practical PC fleet breakdown with the reduced car ownership level that will result in fulfilling the targeted CO_2 emissions. To estimate the EV share, this scenario assumes high EV market penetration as estimated using S-curve (see section 5.2.1.3). Then the desired CO_2 emission levels were backcasted to determine the desired car ownership level and PC fleet composition. Table 6.16 shows the estimated fleet composition and car ownership.

Table 6.16. Desired car ownership and fleet composition to meet 2030 CO_2 emissions target

Engine size / Fuel type	<1.4L	1.4-2.0L	>2.0L
Petrol	191,312	66,328	3,917
Diesel	61,860	993,535	78,751
Petrol Hybrid	17,653	222,486	93,080
Electric vehicle	351,710		
Total	2,080,632		

The emission levels, as calculated using COPERT 5 from the fleet composition as shown in Table 6.16, are presented in Table 6.17. The CO_2 emissions in 2005 from PC fleet and target for 2030 are also presented in Table 6.17 along with the potential reduction in 2030, relative to 2005.

Table 6.17. Emission levels (tonnes) and reduction in 2030 relative to 2005 with the proposed PC fleet

Pollutants	Emissions in 2005 (tonne)	Emissions (tonne) in 2030				Reduction (%)
		From proposed fleet			Target	
		WTT	TTW	WTW		
CO_2	8,474,539	374,005	5,506,600	5,880,605	5,932,177	-31
CO		0	37,857	37,857		
NOx		0	13,065	13,065		
$PM_{2.5}$		0	399	399		
PM10		0	670	670		
N_2O		0	164	164		
NMVOC		0	924	924		
VOC		0	1,004	1,004		

The results reveal that, in 2030, with the proposed car ownership levels and fleet composition, CO_2 emissions can be lowered by 31% compared to 2005 levels. Also, it can be observed that emission levels of the toxic pollutants, i.e. NOx and $PM_{2.5}$ can also be decreased.

Public transport bus fleet:

The required PT bus fleet to reduce 30% CO_2 from its fleet and possible emissions reduction have been presented in this section. As mentioned in section 5.2.2, depending on the source and method of electricity production, TTW emissions can vary from 20gCO2/km to as high as 720gCO_2/km. Therefore, two cases were designed, Case 1 considers the determination of PT bus fleet composition if the produced electricity is renewable based (TTW CO_2 is 20gCO_2/km) and Case 2 considers the calculation of PT bus fleet breakdown if the source of electricity is EU-mix (TTW CO_2 is 720gCO_2/km). Table 6.18 presents the desired fleet composition in 2030 to meet the emissions target.

Table 6.18. PT bus fleet composition (%) to meet the 2030 target

	Case 1			Case 2		
Fuel type / Euro class	Diesel	Bio-CNG	Electric (Renewable)	Diesel	Bio-CNG	Electric (EU-mix)
Euro 4	2	-	-	1	-	-
Euro 5	22	-	-	15	-	-
Euro 6	41	25	10	29	30	25
Total	65	25	10	45	30	25

Based on the fleet estimated in Case 1 and Case 2, emissions were calculated using COPERT 5. Table 6.19 lists the separate emission levels from diesel, bio-CNG and electric buses for the proposed fleet (Table 6.18).

Table 6.19. Emission levels (tonnes) in 2030 with the proposed PT bus fleet

	Case 1			Case 2		
	Emission levels (tonne)					
Fuel type / Pollutants	Diesel	Bio-CNG	Electric (Renewable)	Diesel	Bio-CNG	Electric (EU-mix)
CO	79.21	27.74		54.70	33.22	
CO_2	70,038	12,343	205	48,454	14,782	18,477
NOx	230.22	110.83		158.99	132.73	
N2O	2.38	0.00		1.64	0.00	
VOC	3.90	27.04		2.70	32.38	
NMVOC	3.57	4.23		2.47	5.06	
$PM_{2.5}$	4.19	1.19		2.90	1.42	
PM_{10}	6.72	2.16		4.64	2.59	

The results presented in Table 6.19 show that there is a markable change in emission levels of CO_2 and NOx in both the Case 1 and Case 2. CO, PM and VOC levels are also significantly improved.

Table 6.20 presents the CO_2 emission levels from PT bus fleet in Ireland in 2005, CO_2 emission levels from the proposed fleets and the possible reductions under both the cases.

Table 6.20. Emissions reduction in 2030 relative to 2005 from PT bus fleet

CO_2 emissions (tonne) in 2005	CO_2 emissions (tonne) in 2030		Decrease (%) in 2030 relative to 2005 levels	
	Case 1	Case 2	Case 1	Case 2
118,330	82,587	81,713	30.21	30.94

It can be noticed in Table 6.20 that the emission levels can be reduced by 30% and 31% from PT bus fleet in Case 1 and Case 2 respectively. The feedstock and land area required to meet the bio-CNG energy demand in these two cases are presented in Table 6.21.

Table 6.21. Land area calculation in Case 1 and Case 2 using quantity and energy density of diesel

Parameter	Unit	Case 1	Case 2
Diesel	litre	16,003,244	19,203,893
Diesel energy by volume	MJ/litre	33.7	33.7
Diesel	GJ	539,309.3	647,171.2
The energy density of methane	MJ/m^3	37.78	37.78
Methane	m^3	14,274,987	17,129,989
Biogas	55% of CH_4	25,954,521	31,145,435
Biogas yield	m^3/t of silage	123	123
Mass of silage	t	211,012	253,215
Silage yield	t/ha of land	60	60
land area	ha	3,517	4,220
Life Cycle Analysis land area	ha	6,063	7,276

Final land areas required to fulfil the bio-CNG energy demand to serve PT bus fleet in Case 1 and Case 2 are 6,063 and 7,276 hectares. This not only offers a more feasible way to incorporate bio-CNG in PT bus fleet but also help in meeting the emissions target for 2030 from this sector.

6.4. Conclusion

This research has examined, through a detailed analysis, the potential emissions reduction from changes in PC fleet and PT bus fleet in Ireland. In 2015, the CO_2 emission levels from PC fleet alone was 6.1 megatonne. Whereas, emission levels of CO, NOx, $PM_{2.5}$, PM_{10}, N_2O, NMVOC and VOC were levels were 35,558; 13,766; 722; 1,039; 182; 2,717 and 3,058 tonnes respectively. The monetary value of the damage caused by these pollution levels was found to be €276.77 million in 2015 alone. To investigate the emission levels and its financial impacts, the future emission levels in 2020, 2025, 2030, 2040 and 2050 were estimated with the condition that no additional measure is taken, or no new policy is implemented upon the existing ones. The results showed that the emission levels of CO_2 from PC fleet are expected to increase by 18% and 31% respectively in 2020 and 2030 with respect to 2015 levels. This is due to the increase in car ownership levels by 7.4% in 2020 and by 28.7% in 2030 and a low percentage of zero emission share in the fleet. Moreover, in 2050 when it is expected to have 61% EVs in the fleet, although the tail pipe CO_2 emissions are expected to decrease by 39% the overall CO_2 emission levels are expected to increase by 13% if fossil fuel-based electricity is used. Emission levels of other pollutants, such as CO, VOC, NMVOC, $PM_{2.5}$ and PM_{10} are improved in 2025 and later years. The annual damage costs as a result of this emissions will increase by a sizable amount at least until 2040.

Looking at the high emission levels in the future fleet with BaU scenario, three hypothetical scenarios were designed to examine the potential reduction in emission levels if new policy measures are implemented and there is an increase in electric vehicle uptake. These additional scenarios assume that 10%, 15%, 25% new EV registrations in 2025 and finally, 50% new EV deployment in 2030. Based on these scenarios, the alternative fleet composition was determined for 2030 and their potential in reducing greenhouse gas and exhaust air pollutants were evaluated. It was found that the pollution levels are significantly improved in these scenarios. Even with the low EV penetration scenario, an increase in CO_2 levels can be lowered by at least 6% than with the BaU. Further, it was estimated that substantial electrification is necessary to fulfil the GHG target for 2030, which is to reduce GHG emission level by 30% relative to 2005 levels (European Commission, 2018). This does not offer a realistic solution to meet the target. Therefore, this study investigates rather a few more viable options that will lower emissions by 30% from PC and PT bus fleets in 2030 relative to 2005 CO_2 emissions from these sectors. It is expected that due to strong economic growth, the car ownership levels are expected to rise by 22.8% in 2030 compared to 2015. This study shows that if the increase in car ownership levels can be limited to 4.8%, the GHG emission goals can be met with high EV market penetration scenario. This indicates the necessity of policy making and investments towards an active mode of transports like walking and cycling infrastructures and facilities.

For PT buses, all the designed scenarios with options of fleet renewal with EEV, Euro 6, CNG, bio-CNG and electric buses offer significant savings in emission levels. Euro 6, being the cleaner technology shows a considerable reduction in CO, PM, NOx, VOCs emissions but does not significantly reduce the CO_2 emissions. From this view, CNG also is not a suitable option as the results show that the use of CNG as bus fuel will increase the CO_2 emissions by 8%. But when the emission levels resulting from the use of bio-CNG are compared, 57% CO_2 emissions reduction was observed with CO, NOx, $PM_{2.5}$ reductions being 61%, 57%, and 74%, respectively. Looking at the availability of grassland in Ireland, Bio-CNG offers a convenient option as an alternative fuel for PT buses. However, the land area required to serve the energy demand of the entire fleet is huge. Renewal of the fleet will not only reduce the emission levels but also will improve public health by reducing emission levels of toxic pollutants such as, NOx and $PM_{2.5}$ which have been linked to series of health effects such as stroke, lung cancer, chronic and acute respiratory diseases, including asthma (WHO, 2016a). It was found that if urban PT bus fleets in

Dublin, Cork, Galway, Limerick and Waterford (i.e. where Dublin bus and bus Éireann run) are replaced by electric buses the emission levels can be reduced by more than 90% compared to 2015 levels if renewable based energy is used for electricity production. Increased use of renewable energy and improved efficiency in electricity production play a major role in reducing emissions. With the electricity source being a renewable energy with high energy efficiency, the energy demand can be reduced by 49% relative to base scenario. Based on these findings, two more feasible PT fleet breakdowns comprising of diesel (Euro 4, 5, 6), bio-CNG (Euro 6/EEV) and electric buses have been suggested that will not only reduce the emissions of pollutants that are harmful to human health but also will reduce 30% CO_2 emissions from PT bus sector in Ireland in 2030, from 2005 levels.

The current and future BaU emission levels indicate that even though the existing VRT reliefs have resulted in reduced uptake of petrol cars with higher GHG emissions which was the initial intention of introducing this fiscal incentive, but has not helped in reducing overall emissions rather have increased it because of the shift from smaller petrol engine to medium or larger diesel engines which have similar or higher CO_2 EFs compared to smaller petrol engines. The tax incentives in favour of diesel cars were originally planned at reducing carbon emissions, and it was not known that diesel vehicles are cheating the emissions standards. Therefore, a high percentage of diesel cars, consequently, has caused to increase in NOx emissions which possess serious health threats. Also, despite the financial incentives provided by the Irish government towards EV purchase, the uptake is not significant. Therefore, it is necessary to look into the existing policies, consider revision and implementing new policies and measures. The policies that might lead to achieving the alternative scenarios examined in this Chapter are presented in the next Chapter.

Chapter 7: Modelling Vehicular Emission Reductions with Alternative Policy Interventions

7.1. Introduction

Based on the findings of Chapters 5 and 6, in this chapter, several policies were proposed and emissions reduction as a result of those additional policy measures were quantified. Air pollution has been linked to 491,000 deaths in Europe annually (EEA, 2016a) and diesel vehicles are one of the major sources of two deadly air pollutants, $PM_{2.5}$ and NOx. NOx is generated through typically any process of high-temperature combustion, most commonly from the burning of fossil fuels in the motor vehicles' combustion engine (EPA, 2016). It has been found that Euro 5 and Euro 6 diesel PCs and LCVs emit much higher NOx than Euro standard specifications (Transport and Environment, 2016). As mentioned earlier, Ireland has the highest number of newly registered diesel vehicles in Europe and the increase in diesel car purchase can be linked to change in VRT from engine size based to CO_2 emissions based. Also, this has led to a substantial increase in diesel vehicles of larger engine size (>1.4L), as shown in Figure 7.1. Also, the Irish fleet has a significantly high number of older vehicles which are more polluting than the newer technology classes. This in addition to *dieselgate* has resulted in significant health and financial consequences, especially in urban areas. Therefore, the introduction of new policies with an intention to reduce the use of ICEVs and increase the use of sustainable transport modes, has become necessary.

As mentioned in section 3.2.5, damage cost of pollution of one tonne of $PM_{2.5}$ in urban areas is €200,239 which is very high compared to suburban or rural areas where per tonne of $PM_{2.5}$ emission has associated damage costs of €48,779 and €16,985 respectively. Also, it was identified from the results presented in Chapters 5 and 6 that Dublin has the highest level of road transport emissions and its subsequent impacts. Therefore, the present study quantifies the environmental, financial and health burdens from PCs, LCVs and buses in Dublin under the existing conditions and estimates the potential impact of the increasing numbers of diesel cars, LCVs and buses over the next decade following the current trend projected. As a preventative measure, the present study examines the impact of banning diesel vehicles older than 20 years from 2018 through the initiation of a phase-out policy. Furthermore, the potential impacts of a policy to ban new diesel vehicle sales from the year 2025 is estimated also. Consequently, the results of this study make a strong case for policies and investments aimed at reducing the environmental impact caused by urban transportation and improving public health.

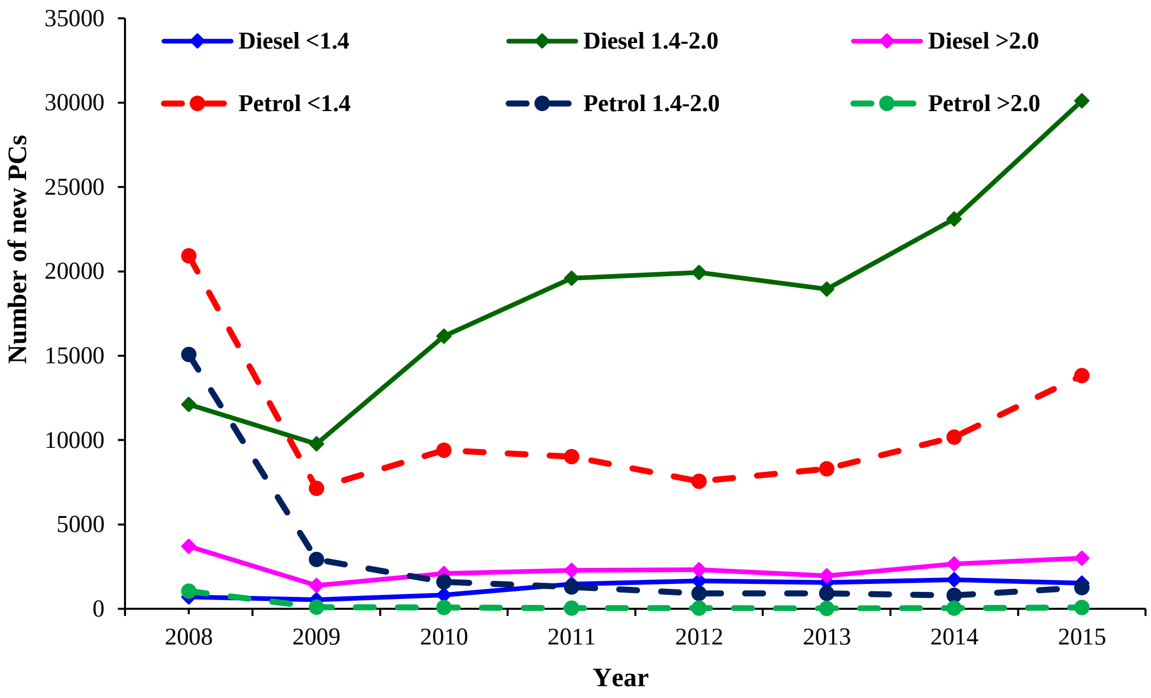


Figure 7.1. New car registration pattern by engine size and fuel type in Dublin

The potential environmental impact is estimated through simulated scenarios designed in COPERT 5 (EMISIA, 2018). The health burden is estimated using a damage factor based method (Oldenkamp et al. 2016; Tang et al., 2015; Hofstetter, 1998) as described in 3.2.4. The associated financial impact considering direct and secondary effects are estimated and presented in a number of hypothetical scenarios. In addition to the two key policies, a range of other policies to tackle the increase in diesel vehicle ownership such as, increasing diesel excise duty[6], thereby, making it equal to the petrol excise by 2021 and eliminating the CO_2 emissions based motor taxation was investigated. The revenue from these fiscal policy decisions will generate funds which can be used in improving the EV infrastructure and replacing the current bus fleet with bio-CNG compliant buses. The diesel ban and tax changes, on the other hand, will increase the EV uptake and help Ireland to meet its 2030 target in reducing GHG emissions.

The next section 7.2 describes the scenarios considered and the methodology followed design the scenarios and to assess the environmental, health and financial burden. This is followed by a detailed description of the data used (section 7.3). Potential emissions reduction resulting from those additional policy measures and from other scenarios were then quantified and presented in section 7.4. A discussion on the results and policy implications are presented in the following Section 7.5, and finally, the conclusion of this study is discussed in Section 7.6. The contents of this chapter are mainly based upon the work published in Dey et al. (2018b).

7.2. Policy and scenario design

This section presents the current and future scenarios designed in this study and the methodology developed to assess the environmental, health and economic impacts caused by diesel use in road transport in Dublin. A schematic diagram of the methodology showing the timeline of the proposed policies and resulting scenarios has been presented in Figure 7.2. Four scenarios were examined and evaluated in terms of assessing their impacts on the environment, public health, and economy. The first scenario presents the existing situation in 2015 (base scenario), then the current trend projected over the next 10 years is presented in the BaU scenario. The third is the diesel phase-out scenario where the possible outcomes of a set of proposed policy measures in minimising the impacts of diesel use in road transport are explored until the end of 2024. Finally, the impacts in the year 2030 were assessed to quantify the long-term effects of introducing different policy measures including banning the sale of new diesel vehicle in 2025. This scenario has been referred to as the diesel ban scenario. The detailed description of the proposed policies and resulting scenarios are given in the following sub-sections.

7.2.1. Base Scenario: emissions from the current diesel fleet in Dublin (2015)

In this scenario, emissions from the entire PC, LCV and bus fleets in Dublin were calculated using COPERT 5 (EMISIA, 2018) for the baseline scenario, 2015. The COPERT model uses detailed data in terms of meteorological information, fleet data and activity data as elaborated in detail in sections 3.2.1 and 4.2. After *dieselgate* came to light, COPERT has been updated to take into account the discrepancy in NOx EFs as described in Chapter 6. As mentioned earlier, the term *dieselgate* was used to explain the scandal where it was found that Euro 5 and Euro 6 diesel PCs and LCVs were violating the NOx emission standards for their engine classes with the help of a defeat device which turns on the full emission control system during the emissions test and emits higher levels of emission by turning off the emission control system during other times. Emission levels of all the major pollutants, such as CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC, NMVOC were estimated in tonnes. The annual emissions and associated impacts in terms of health and cost were calculated for the base scenario.

Impact Assessment

The damage costs due to these pollutants resulting from the PC, LCV and bus fleet in Dublin were calculated by multiplying the total emissions (tonne) by respective unit damage costs.

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[6] Excise duty is the tax on the sale of a particular good or service, collected by a producer or retailer and paid to the government.

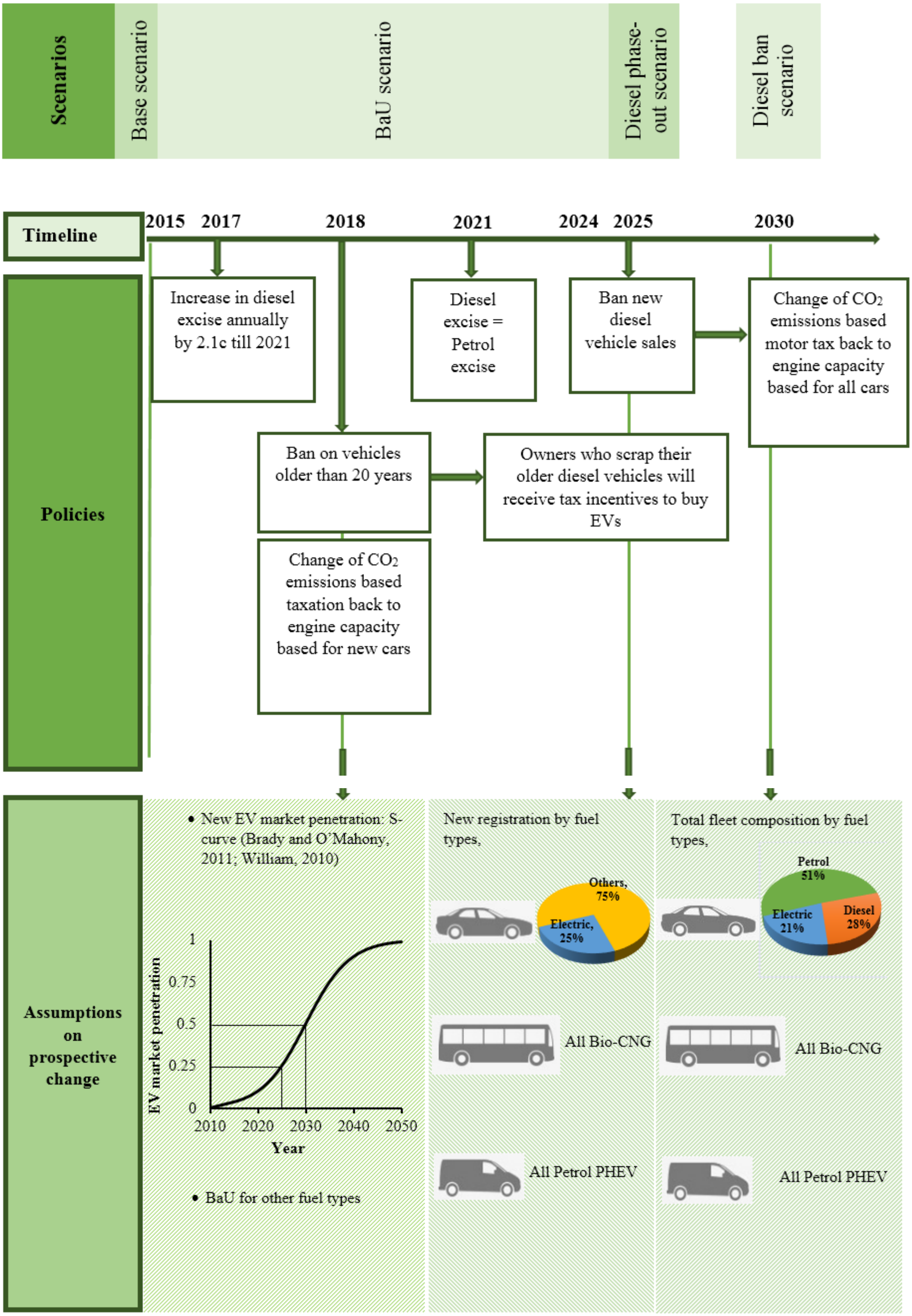


Figure 7.2. Schematic diagram of the roadmap of interventions to deliver targets by 2030

Unit damage costs (€/t) from emissions were obtained from the Handbook on external costs of transport (Korzhenevych et al., 2014) and DTTaS (2016a). Health effects due to NOx were calculated in terms of DALYs. In this study, the damage factor approach as described in section 3.2.4 to estimate health impacts of NOx has been followed to assess DALYs associated with NOx and $PM_{2.5}$ from diesel road transport fleet in Dublin. In Europe, NOx and $PM_{2.5}$ cause health damages at the rate of 90 DALYs/kilotonne (Tang et al., 2015) and 700 DALYs/kilotonne (Hofstetter, 1998), respectively.

7.2.2. BaU Scenario (2015-2024)

The BaU scenario presents the information on potential emissions that will be generated from diesel PCs, LCVs and buses in Dublin cumulatively over the years 2015-2024 if the current trend in fuel use continues. The associated cumulative damage costs and health impacts from NOx and $PM_{2.5}$ in Dublin are also estimated under this scenario.

Future fleet determination

The main challenge in estimating the emissions from the future fleet was to obtain the disaggregated future fleet compositions corresponding to each technology class. Total percentages of diesel passenger cars in the future fleets were calculated using Systra's rolling_fleet_model (v.7.0) (DTTaS, 2015b). The compositions of the future car fleets as per age were estimated based on the historical vehicle survival rate (Alam et al., 2015) and current fleet composition (DTTaS, 2015a). Here survival rate refers to the chances of a vehicle staying in the fleet depending on their age. For example, if a 15 years old vehicle has a survival rate of 0.3, that means a vehicle purchased today will have 30% chance that after 15 years, it will be running on the road. The available historical survival rate of cars in Ireland is based on 1972-1984 data (Alam et al., 2015). Thus, the survival rate is updated by multiplying the historical survival rates by the fraction of cars in the 2015 fleet as per age. This approach was followed as it not only helps to classify the vehicles into Euro classes but also identifies the vehicles older than 20 years. Figure 7.3 shows the historical survival rate (Alam et al., 2015) and the predicted percentage of diesel cars in the future fleet as per age. The future LCV fleet compositions were determined based on the past trend on survival rate (DTTaS, 2015a; DTTaS, 2016b). The same approach was followed in order to sort the future bus fleet into euro standard classes.

Impact assessment

The emission levels from the future diesel fleet of cars, LCVs and buses were calculated using COPERT 5 for each year separately and summed up to obtain the cumulative emissions over 2015-2024. The resulting damage costs for CO_2, PM_{10}, $PM_{2.5}$, NOx, VOC, NMVOC were also calculated. Similar procedure as used in the base scenario was followed in order to quantify the damage costs and health impact in terms of DALYs in the next 10 years under BaU situation.

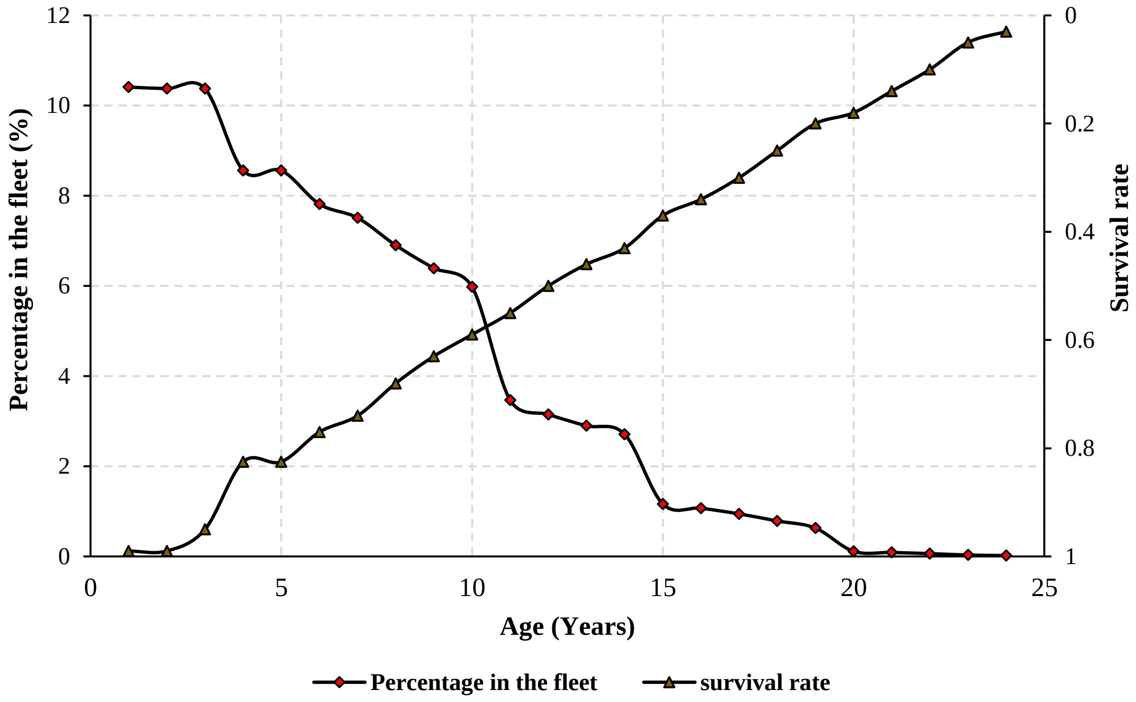


Figure 7.3. Percentage of diesel cars in the fleet by age and survival rate

7.2.3. Diesel Phase-out Scenario (2018-2024)

The diesel phase-out scenario considers a set of new policy measures as shown in Figure 7.2 but calculates the potential savings in cumulative emissions and its subsequent impacts up to 2024 i.e. until the time when older vehicle phase-out policy is in place, but diesel ban is yet to be implemented. Therefore, the policies considered in the diesel phase-out scenario are:

1. Vehicles older than 20 years will be banned from 2018, this will imply that all the Euro 1 (PC:1992-1996; LCV:1994-1997; bus:1994-1996), Euro 2 (passenger car:1997-2001; LCV:1998-2001; bus:1997-2001) and Euro 3 (PC, LCV and bus:2002-2005) vehicles (Dieselnet, 2016) which are more polluting than the later technology classes will be phased out by the end of 2024.
2. The diesel excise duty will be increased by 2.1c from 2017 until 2021 when it will be equal to the petrol excise duty, which was 23% higher than diesel at the moment, by 2021 (Department of Finance, 2016) to demotivate the diesel car uptake. Earlier the excise duty on diesel was 47.9c and for petrol 58.77c.
3. CO_2 emissions based motor tax will be changed back to engine capacity based from 2018 on newly registered cars.

4. Fiscal incentives will be given to the consumers for scrapping their diesel vehicles to buy EVs.

Some of these policies such as an increase in diesel excise duties are planned for Ireland (RTE, 2016; Public Policy, 2016), and some of these policies are to be introduced elsewhere as discussed in the literature review (Chapter 2) and are evaluated in the case of Ireland in this study. It is expected that there will be an increased market penetration of EV in Ireland as a result of these policy interventions. It is assumed that this market penetration will follow an S-curve (Figure 6.2) as described in section 6.2.1.3. Three EV penetration scenarios, i.e. low, medium and high, in the same way as section 6.2.1.3, were considered here. The fleet configurations shown in Figure 7.2 for 2030 is for the high EV purchase scenario. In the diesel phase-out scenario, the immediate impacts of the proposed policies till the end of 2024 in terms of the reductions in emissions and the resulting changes in health and fiscal damage is calculated. The cost savings from improved health damage risks will be given as an incentive to consumers who will scrap their older diesel vehicles and replace them with an EV.

Impact assessment

The changes in emission levels and the resulting health and economic impacts of this scenario were calculated. The extra revenue generated from increased diesel excise (see, policy 2) was calculated by multiplying the AAM (km), the total number of cars and fuel consumption per kilometre (litres/km) and increased cost (cents/litre), as shown in the following equation (7.1).

$$R_{di} = FC * N_{di} * M_{di} * FT * 10^{\wedge} - 8 \tag{7.1}$$

R_{di} = Revenue from diesel sales in year i (€ million)

FC = Fuel consumption per kilometre (litres/km)

N_{di} = number of diesel cars in year i (km)

M_{di} = AAM of diesel cars in year i

FT = Increase in fuel tax (cents/litre)

The extra revenue generated due to the change in the taxation system for diesel vehicles (see, policy 3) were calculated. In Ireland, for LCVs and buses, the motor tax is determined and imposed based on the weight class. The extra revenue generated from the changes in motor taxation system was calculated by subtracting revenue generated from CO_2 emissions based taxation system, which is the present approach, from the revenue generated from the older engine capacity based taxation system. There are twelve CO_2 based tax bands, namely, A0, A1, A2, A3, A4, B1, B2, C, D, E, F and G which correspond to CO_2 emissions ranges 0-1 g/km, 2-80 g/km, 81-100 g/km, 101-110 g/km, 111-120 g/km, 121-130 g/km, 131-140 g/km, 141-155 g/km, 156-

170 g/km, 171-190 g/km, 191-225 g/km and 225-999 g/km respectively. Whereas, for engine size based taxes, there are twenty one engine size classes with a 100cc interval, covering the range from 0 to 3000cc and another separate tax band for cars with an engine size greater than 3000cc. The CO_2 emissions based and engine capacity based annual tax values were obtained from DTTaS (DTTaS, 2017) for the CO_2 bands and engine size classes. The number of cars in each CO_2 emissions-based tax classes and each engine capacity-based tax classes were obtained from DTTaS (DTTaS, 2017) for the fleet in 2015, the same percentage shares in the future fleet were assumed in order to estimate the revenue generated from this policy change over 2018-2024. The revenues were calculated based on both the approaches and listed in Table 7.1 along with the taxation bands. Increased excise duty on diesel fuel and withdrawal of CO_2 emissions based tax incentive might discourage consumers from buying new diesel vehicles.

7.2.4. Diesel Ban Scenario (2030)

This scenario considers banning sales of new diesel vehicles starting from the year 2025 in addition to the continuation of all the policies mentioned in the diesel phase-out scenario and calculates annual emission levels from the overall road transport fleet in the year 2030. In this scenario, the changes in emission levels in 2030 compared to 2015 has been explored. It has been assumed that with increased market penetration of EVs, the share of EV in annual new car registrations will be 25% in 2025 and 50% in 2030 (see S-curve in Figure 6.2). The remaining new registrations will comprise of petrol and hybrid petrol vehicles. The electricity requirement of EVs was considered to be 150Wh/km and CO_2 emissions from electricity generation to be 582g/kWh (SEAI, 2017b). The car population in 2030 was estimated based on population, economic growth and economic stability and obtained from Demographic and Economic Forecasting Report (NRA, 2014), Ireland. From 2025, new buses will be powered by bio-CNG and new LCVs will be petrol-plug-in hybrids. The changes in emission levels and resulting impacts in 2030 have been calculated based on the entire PC fleet, Dublin bus fleet and the LCV fleet in the year 2030. While calculating the emissions from LCVs, it was assumed that 40% of the journey will be made in the electric mode. CO_2 emissions from bio-CNG buses were considered to be 40% of the CNG powered buses (Ryan and Caulfield, 2010). The bus fleet and the LCV fleet in 2030 was assumed to increase by 10% and 25% respectively compared to the 2015 fleet following the past trends in both the fleets.

Table 7.1. Revenues from the previous and current motor taxation methods

Current approach			**Previous approach**		
CO_2 emissions class	**Annual tax rates (€)**	**Revenue (€) 2018-2024**	**Engine capacity bands (cc)**	**Annual tax rates (€)**	**Revenue (€) 2018-2024**
A3	190	2,411,121	0 - 1000	199	1,058
A4	200	5,089,924	1101 - 1200	330	117,556
B1	270	4,347,230	1201 - 1300	358	186,156
B2	280	6,693,707	1301 - 1400	385	2,507,972
C	390	4,397,745	1401 - 1500	413	6,849,332
D	570	2,858,580	1501 - 1600	514	16,490,649
			1601 - 1700	544	5,089,414
			1701 - 1800	636	97,388
			1901 - 2000	710	15,704,621
			2101 - 2200	951	4,790,367
			2201 - 2300	994	407,999
			2301 - 2400	1,034	45,081
			2401 - 2500	1,080	5,742
			2701 - 2800	1,391	4,437
			2901 - 3000	1,494	1,853,986
			3001 - 15000	1,809	11,542

7.3. Data Description

This section describes the data used in this study to assess the emission levels resulting from the diesel fleet in Dublin using COPERT 5. The main input data required in COPERT are fuel consumption (TJ), fleet data disaggregated to each fuel type-euro standard-engine size class, AAM (km), mileage share (%), monthly average minimum and maximum temperature (˚C), monthly average RH (%), average speed (kmph) and average trip length (km) as described in detail in section 3.2.1. The sources of data were the same as described in section 4.2. The monthly minimum and maximum recorded temperatures in Dublin are presented in Figure 7.4.

The PC, LCV and bus fleet data (SIMI, 2017; Dublin Bus, 2016) in Dublin in 2015 and AAM have been presented in Table 7.2 and Table 7.3. In the table, car engine sizes, small, medium and large refer to the engine sizes <1.4L, 1.4-2.0L and 2.0L respectively. For LCVs, N1 I, N1 II and N1 III represent the weight classes <1305kg, 1305-1760kg, and 1760-3500kg respectively.

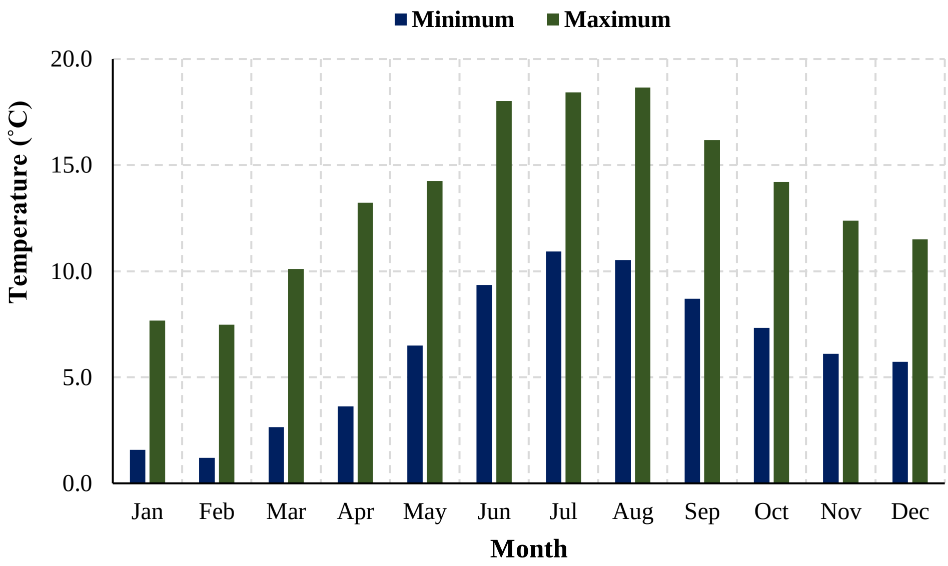


Figure 7.4. Monthly minimum and maximum recorded temperature in Dublin in 2015

Table 7.2. Disaggregated diesel car fleet in Dublin

Engine Size	Euro class	Number	AAM (km)
Small	4	5,987	18,368
	5	10,764	
	6	64	
Medium	1	637	22,862
	2	9,427	
	3	25,032	
	4	57,899	
	5	87,453	
	6	1,783	
Large	1	64	19,802
	2	892	
	3	2,293	
	4	13,694	
	5	7,771	
	6	191	

Table 7.3. Disaggregated LCV and bus fleet in Dublin

LCV				**Bus**		
Weight class	Euro class	Number	AAM (km)	Euro class	Number	AAM (km)
N1 I	2	346	22,344	3	447	57,288
	3	631				
	4	876				
	5	1,459				
	6	68				
N1 II	2	501		4	147	
	3	914				
	4	1,268				
	5	2,113				
	6	99				
N1 III	2	5,046		5	372	
	3	9,202				
	4	13,093				
	5	19,464				
	6	1,059				

7.4. Results

The results of this study in terms of environmental, health and economic impacts as obtained from analysing the different scenarios have been presented in this section.

7.4.1. Base scenario

The emission levels, health and cost impacts, calculated following the methods described in section 7.2.1, are presented here. The total number of PCs, LCVs and buses were 514,000; 56,141 and 966 respectively, which constitute about 91.31% of the entire road transport fleet (DTTaS, 2015a) in Dublin, while the rest of the road transport fleet constitutes of motorcycles, motor caravans, large public service vehicles, excavators etc. (DTTaS, 2015a). The results for the base scenario are shown in Table 7.4. It can be observed that the unit damage costs of NOx and $PM_{2.5}$ are significant. Hence, these pollutants have caused a total loss of €80 mil in 2015.

7.4.2. Diesel emissions in the BaU scenario during 2015-2024

The potential environmental, financial and health damage that will be caused by diesel cars, LCVs and buses in 10 years from the base year of 2015 are presented in this section. Table 7.5 shows the cumulative emissions and associated impacts caused by the diesel share of road traffic (cars, LCVs, and buses) in Dublin during 2015-2024.

Table 7.4. Annual emission levels from all cars, LCVs and buses with associated cost and health damage in Dublin in 2015 (base scenario)

Impacts	Pollutants					
	CO_2	NOx	$PM_{2.5}$	PM_{10}	VOC	NMVOC
Car: emissions (tonne)	1,458,903	2,987.46	185.26	268.39	776.14	679.54
LCV: emissions (tonne)	301,831	1332.69	61.67	81.44	60.39	58.39
Bus: emissions (tonne)	62,513	571.84	10.14	12.23	18.91	16.11
Unit damage cost (€/tonne)	13.22	5851	200,239	19143	1438	1398
Total damage cost (mil€)	24.10	28.62	51.48	6.93	1.23	1.05
Health damage (DALYs)	-	440	180	-	-	-

Table 7.5. Cumulative emission levels from diesel cars, LCVs and buses with resulting cost and DALYs in Dublin over the period 2015-2024 (BaU Scenario)

Impacts	Air pollutants					
	CO_2	NOx	$PM_{2.5}$	PM_{10}	VOC	NMVOC
Car: emissions (tonne)	11,236,074	37,854	1,271	1,984	234	167
LCV: emissions (tonne)	2,999,432	12,649	472	669	342	332
Bus: emissions (tonne)	639,119	4,582	80	102	133	114
Total emissions (tonne)	14,874,623	55,085	1,823	2,755	709	613
Damage cost (mil€)	148.54	221.48	254.57	37.97	0.34	0.23
Total DALYs	-	4,958	1,276	-	-	-

7.4.3. Diesel phase-out scenario

The possible reduction in emission levels and savings in damage cost up to the end of 2024 considering the policy changes proposed in section 7.2.3 are presented here. In addition to that, the revenues generated from increased excise duty on diesel and changing the motor vehicle tax back to engine capacity based system from CO_2 emissions based on new cars have been reported. Table 7.1 shows the CO_2 emissions and engine capacity bands with their respective tax amounts. Only the engine classes and CO_2 emissions bands to which the future diesel car fleet is expected to belong are shown in the table. Table 7.6 shows the reduction in pollution levels and damage costs by the end of 2024 along with the revenues from the changes in taxation policies. The revenue from diesel excise tax includes the revenue generated from cars, LCVs and buses due to the increased fuel price calculated over the period 2017-2024. Whereas, the revenue values in the columns (Table 7.6), namely, revenue from CO_2 based tax and revenue from engine based tax include only PCs. It can be observed that from the changes in taxation and implementation of new policies, a significant amount of the burden can be reduced and will keep on reducing as these policies continue to be in place.

Table 7.6. Potential changes in emissions, damage costs and revenue generated in diesel phase-out scenario (2018-2024)

Pollutants	**Emissions reduction (tonne)**	**Damage cost savings (€) (2018-2024)**	**Revenue from diesel excise tax (Mil€)**	**Revenue from CO_2 based tax (Mil€)**	**Revenue from engine based tax (Mil€)**	**Net revenue from CO_2 based tax withdrawal (Mil€)**
CO_2	371,657	4,913,304	490.74	25.80	54.16	28.36
NOx	1,614	9,443,043				
$PM_{2.5}$	114.90	23,008,314				
PM_{10}	137.14	2,625,196				
VOC	65.62	94,366.74				
NMVOC	57.38	80,216.25				

7.4.4. Diesel Ban Scenario

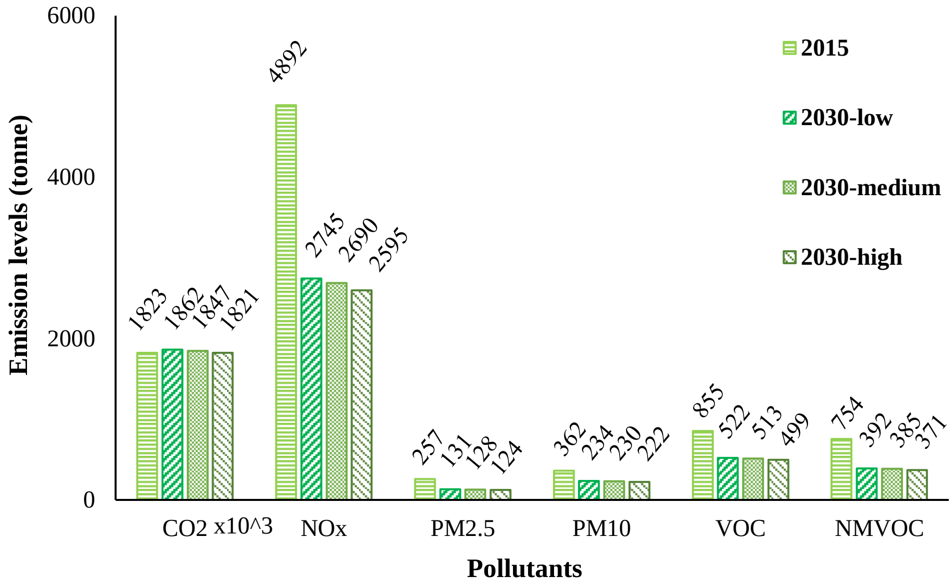
This section presents the emission levels in 2030 after the implementation of the proposed bunch of policies in section 7.2.3 and 7.2.4. In Table 7.7, total emissions of CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC and NMVOC from all cars, buses, and LCVs have been tabulated along with the change in emission levels, damage costs and DALYs compared to 2015 values. With the new policy measures, the potential reduction in NOx levels in 2030 is 2,297 tonnes, whereas, for $PM_{2.5}$ it is 133 tonnes, relative to 2015 levels. This could potentially lessen the health damage in 2030 by 300 DALYs as a result of NOx and $PM_{2.5}$ pollution from road transport.

Table 7.7. Potential changes in annual emissions, health damage and cost in 2030 compared to 2015 (diesel ban scenario)

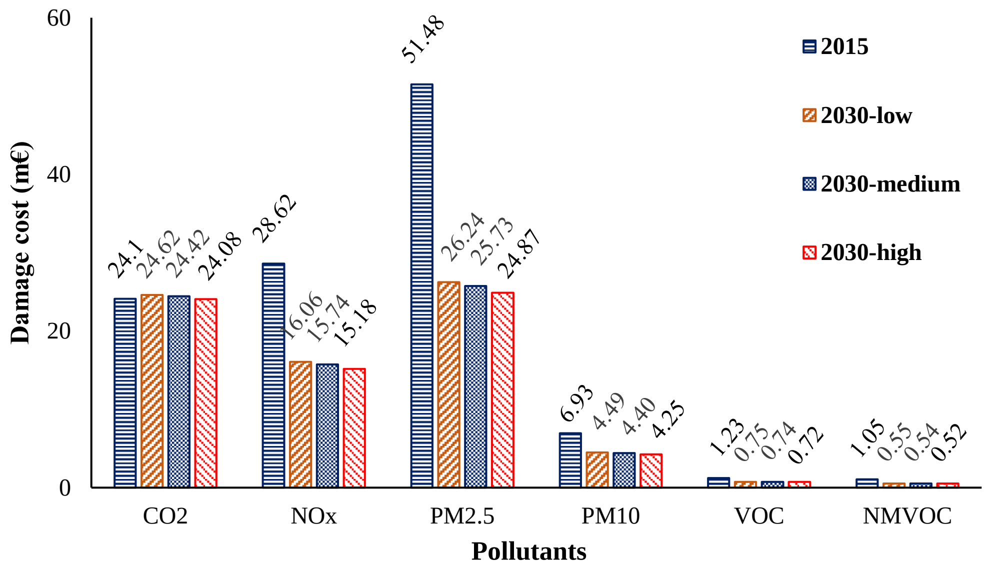
Fleet		CO_2	NOx	$PM_{2.5}$	PM_{10}	VOC	NMVOC
Car	Emissions (tonne)	1,544,846	2,297	104	185	362	300
	% change	+6	-23	-44	-31	-53	-56
	Cost savings (€)	-1,136,167	4,041,268	16,261,460	1,596,631	595,921	530,059
	DALY	-	62	57	-	-	-
Bus	Emissions (tonne)	29,549	266	3	5	65	10
	% change	-53	-54	-72	-58	243	-36
	Cost savings (€)	435,778	1,791,024	1,466,000	136,411	-66,124	8,048
	DALY	-	28	5	-	-	-
LCV	Emissions (tonne)	247,059	32	17	32	72	61
	% change	-18	-98	-72	-61	19	5
	Cost savings (€)	724,076	7,609,214	8,877,380	943,978	-16,280	-4,331
	DALY	-	117	31	-	-	-

7.4.5. Sensitivity analysis

A sensitivity analysis has also been conducted to take into account the variability and other possible outcomes of the assumptions made in this study. The assumption made in respect of EV uptake i.e. 25% and 50% of the new vehicles will be EVs in 2025 and 2030 respectively, was considered as the high market penetration scenario. Additionally, two scenarios with low and medium EV uptake rates were designed and emission reductions were calculated as mentioned in section 7.2.3. The additional scenarios assume that in low and medium cases the new EV market penetration will have 10% and 15% share of total new vehicle registrations in 2025 respectively, and 50% in 2030. Section 6.2.1.3 provided more details on the yearly market penetrations of EVs in the three possible scenarios. The overall environmental and economic impact of the two scenarios, i.e. the base scenario (2015) and the diesel ban scenario (2030) are shown in Figure 7.5 (a) and Figure 7.5 (b) respectively.



(a)



(b)

Figure 7.5. (a) Emissions (tonnes) and (b) Damage costs (mil€) of pollution in 2015 and 2030

The figures show that there are significant reductions in $PM_{2.5}$ and NOx emission levels and resulting damage costs in low, medium and high scenarios in 2030 compared to 2015. However, there are smaller variations in emissions and damage costs among low, medium and high scenarios. The reason behind this is even though these scenarios indicate different levels of market penetrations of EVs annually, the variations in the total share in the overall vehicle fleet in 2030 is not high. In 2030, under low market penetration scenario, the overall ICEV share in the overall fleet is 85%, whereas, under medium and high scenarios the ICEV shares are 83% and 80% respectively.

7.5. Discussions and Policy Implications

The main findings of this study in terms of reduction in environmental (emissions), health (DALYs) and financial (damage cost of pollutants on health, materials, crops, and biodiversity) burden, as a consequence of the policy changes, have been discussed in this section. PCs, LCVs and buses in Dublin have discharged 1,823kt, 4.89kt, 257t, 362t, 855t, 754t of CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC and NMVOC respectively in the base scenario. Cumulatively, this has caused a damage of €113.42 million.

It was estimated that the diesel fleet alone would cause a health and other monetary damage of €663.1 million in the next 10 years from the base year of which approximately 72% is borne by NOx and $PM_{2.5}$. It is possible that as a result of the cumulative measures, including phase-out of older vehicles from 2018 and banning new diesel vehicles from 2025, the annual emission levels of CO_2, NOx, $PM_{2.5}$, PM_{10}, VOC and NMVOC will reduce by 0.1%, 47%, 52%, 39%, 42% and 51% respectively in 2030 compared to 2015 levels even with increase in car ownership by 29%, in LCV numbers by 25% and bus numbers by 10%. In 2025, the CO_2 and NOx emissions are expected to rise by 8% and 17% ICEVs but $PM_{2.5}$ will reduce by 33%. It can be observed from Table 7.7 that the CO_2 emissions from cars will increase in 2030 by 6% compared to 2015 levels which are due to the increase in petrol-powered cars after the diesel ban. However, the CO_2 levels will start to decrease, and overall damage will further reduce with increased EV uptake. The results from the proposed scenarios show that the proposed policy measures generate an annual savings of €43.8 million and an annual reduction of 300 DALYs in Dublin in 2030. With the implementation of the older diesel vehicle ban policy, the damage cost saved could be €40.16 million by 2024. The money saved in terms of damage cost reduction can be provided as scrappage incentives to the consumers.

Dublin has the highest population density in Ireland, therefore, like any other urban centre, the health effects on the population are more pronounced. Emission standards are designed to protect air quality and human health. After *dieselgate*, where it was identified that Euro 5 and Euro 6 light-duty vehicles are not obeying the emission standards, reducing pollutant levels have become one of the major concerns of the researchers and policy makers. It has been estimated that NOx and $PM_{2.5}$ from road transport have resulted in 620 DALYs in 2015, whereas, with the proposed policies these numbers can be reduced by nearly 50% in 2030. Also, it has been calculated that NOx and $PM_{2.5}$ emissions from diesel fleet alone will cause about 6234 DALYs during 2015-

2024 with the present trend. Therefore, like other countries, Ireland also should consider implementing diesel ban policies in order to improve and protect air quality and health.

For cars, earlier the motor vehicle taxation was based on engine capacity which was changed to CO_2 emissions based in 2008 with the aim to reduce GHG levels. A diesel car having an engine size of 1500cc pays €413 as annual registration tax following engine capacity-based taxation system, whereas, with the CO_2 based taxation system, the same diesel car qualifies to pay an annual tax of €200 considering the average diesel tailpipe CO_2 emissions as 120g/km. This has led to the rapid increase in diesel car ownership. This study proposes a change of this tax system back to the engine capacity-based approach which on the one hand will demotivate new diesel vehicle uptakes and on the other hand, will generate revenue which could be invested in improving the EV infrastructure, such as installing more fast-charging stations. There would not be much difference in tax generated from petrol as most of the petrol cars are of smaller engine capacity with size less than 1400 cc which have annual taxes varying from 199-385€ and when taxed based on CO_2 emissions it varies from 270-390€ for the range of emissions 121-155gCO_2/km. Due to these new policies, there will be an increase in the number of petrol car registrations up to 2030 after which market penetrations of EVs are assumed to offset the share of petrol cars. Consequently, this, along with the diesel-ban policies also may solve the problem of Ireland regarding low EV uptake.

SEAI offers a grant of up to €5000 for every BEV or PHEV purchase. In addition to the grant, there are VRT reliefs of €5000 for BEV and €2500 for PHEV (SEAI, 2017a). The annual motor tax for EV is €120. Even after all these measures, the EV uptake is significantly low in Ireland. EVs have zero tailpipe emissions, therefore, electrification of the fleet is important to mitigate overall emission levels. There is a limited number of EV models available in Ireland and currently with only 85 fast charging stations. Due to the proposed fiscal policies, i.e. levelling the diesel excise to petrol excise duty by 2021 and changing of CO_2 emissions based tax back to the previous engine capacity-based tax system becoming effective from 2018 and applicable to all new cars, a revenue of €519.10 million will be generated by 2024. This money can be utilized in improving the EV infrastructure, changing the diesel bus fleet to bio-CNG compliant buses, building facilities to encourage the use of active travel options such as walking and cycling. Active travel will not only be beneficial for the environment but also will have major health benefits as a result of increased physical activity (RCP/RCPCH, 2016).

7.6. Conclusion

This study evaluates, through a comprehensive analysis, the potential environmental, financial and health burdens from the existing diesel fleet in Dublin and the policies that can potentially mitigate these impacts. A hypothetical scenario based emission estimation has been performed on different options using COPERT in this study. The greatest impacts of policies such as banning diesel vehicles will be improved public health and reduction in premature mortality. This study has found that NOx and $PM_{2.5}$ from diesel-powered cars, LCVs and buses in Dublin will possibly cause a loss of 6234 DALYs by 2024. Thus, it is very important that effective measures are taken as soon as possible. The results show that banning older cars, banning new diesel vehicle sales, increased diesel excise duty, changing motor tax to the previous engine capacity-based approach are effective measures to reduce the impacts significantly. This reduction in emissions and resulting health and cost damages will continue to decrease as the older diesel vehicles leave the market and EV market penetration increases. Ireland's target is to reduce GHG emissions by 30% in 2030, relative to its 2005 emission levels (European Commission, 2018) but it is expected to increase by 14% (EPA, 2017a) with BaU due to very low EV market penetration rate. In addition to reducing toxic and deadly air pollutants such as NOx and $PM_{2.5}$, the policy measures considered in this study will help in increasing EV uptake rate and move towards reaching the emissions target. Though the revenue from tax changes have been calculated up to 2024, more revenue will be generated which could be invested in greening the transport.

The findings of this study provide policy makers with a timely and useful evaluation of the potential impacts of the diesel fleet in Dublin. This research also suggests an effective way to phase out the more polluting older diesel vehicles from the fleet. The results also direct towards the scope of emission reduction and health improvement by introducing new policy measures, which ensures a reduction in environmental and health damage as well as generate revenue to invest towards making the fleet greener. Thus, this study recommends the Irish policy makers to consider diesel-ban and tax changes in Dublin, which will not only improve the urban air quality and public health but also will potentially increase EV uptake at a faster rate provided that the EV infrastructure is improved. The urban air quality due to emissions in the base case and change in urban air quality in 2030 due to emissions with BaU and with additional policy implications are investigated in the next Chapter. Although the results presented in Chapters 5, and 7 showed that actual NOx emission levels are very high compared to the Euro standard emission levels, the actual environmental impacts cannot be fully understood without modelling the air quality. Based on the spatial variations of emissions and its impacts presented in Chapters 5 and 6, it was identified that the emissions and the impacts are highest in Dublin which is expected due to the highest number of vehicles compared to other counties. The findings in Chapter 8 will reflect the environmental impacts attributable to road transport emissions and the expected impacts in 2030 and the potential improvement in air quality in 2030 with the policies proposed in this Chapter.

Chapter 8: Urban Air Pollution Modelling

8.1 Introduction

The findings from Chapters 5, 6 and 7 indicated that NOx emission levels are considerably high, and the emissions and its impacts are highest in Dublin. The results from Chapters 6 and 7 also revealed that the emission levels are going to increase in 2030 with no additional measure taken upon the existing ones, however, with additional policy measures as described in Chapter 7 there is potential for mitigation. Now, in order to understand how the emissions, influence the air quality at population exposure level, the air pollution was modelled at street level in the urban centre in Dublin. The environmental impacts as a result of emissions in 2015 and in 2030 under BaU and with new policies were assessed. Even though on-road NOx emissions were found to be higher, it may not lead to high concentration levels. The relationship between emissions and concentration is still obscure as it depends on several parameters such as wind speed, wind direction, street configuration, building geometry and other meteorological properties. As mentioned earlier, the transport sector has the highest share of NOx emissions in Ireland and total annual NOx emission levels exceed the limit of national emissions ceiling. However, the concentrations at the monitoring locations are below the permissible limits specified by EU and WHO. NOx, which comprises of NO and NO_2 contribute to acid rain and the formation of ground level O_3 which is very harmful to human health (US EPA, 2017; EPA, 2016). Whereas, exposure to NO_2 is associated with health effects such as reduced lung function, airway responsiveness, increased reactivity to natural allergens and increased risk of respiratory infections in children (EPA, 2018). Thus, it is very important to investigate whether the high levels of NOx emitted from the road transport compared to Euro standard emissions, as reported in Chapter 5, cause the NO_2 concentrations to exceed the permissible limit at the level at which people inhale the air. Therefore, this research aims to study the effect of NOx emissions from road transport on urban NO_2 pollution. Also, in order to investigate how the emission levels are related to the concentration, the emissions and concentrations attributable to the road transport were modelled on the major roads in Dublin city and compared to examine the possible linkage. In addition to modelling NOx and NO_2 pollution, concentrations of $PM_{2.5}$ on the same streets were also modelled as road transport also is a major source of $PM_{2.5}$ which possesses severe health impacts as well. These modelled pollution levels were compared to the safe guideline value for annual average daily concentrations provided by WHO and EU.

As mentioned earlier OSPM (v.5.1.90) was chosen to model air pollutant concentrations on the street segments in this research and the emissions at street level in Dublin city were calculated using COPERT Street Level software (EMISIA, 2017). Emission levels also were calculated based on the Euro standard emission factor specifications to examine if the modelled emission exceeds the specified emission levels in the street studied. The emissions and concentrations modelling were carried out over 1743 points in 155 streets within the canal cordon of Dublin city for the year 2015. It is expected that the number of diesel passenger cars will continue to increase and in 2030 the share of diesel passenger cars in the Irish road fleet will be 75% (DTTaS, 2016b) while other road transportation modes will still be diesel dependent. Based on this prediction, in this study, the emission levels and the concentrations were modelled with the expected road transport population and fleet compositions in 2030 fleet to see if the concentration values will exceed the permissible safe limits in the year 2030 under business as usual situation (referred as 2030BaU).

The policy interventions described in Chapter 7 were further investigated to examine their potential in reducing the concentration levels of NOx, NO_2 and $PM_{2.5}$. These policies include, increase in fuel tax, removing diesel vehicle tax incentives, banning vehicles older than 20 years and banning new diesel vehicle sales from 2025 (see section 7.2.3 for more details). It was found that these policy measures have the potential to reduce the NOx emissions approximately by 47% in 2030, compared to 2015 levels under high market penetration of EVs. Therefore, the emissions and concentrations were modelled for 2030 from the alternative fleet due to reduced diesel use and high EV market penetration as a result of the policy interventions evaluated in Chapter 7 and this scenario is referred as 2030Policy. “low” and “medium” EV scenarios were not examined for their potential in reducing environmental impacts of emissions. All three policy scenarios, i.e. low, medium and high were tested at a smaller scale (on 34 street segments) before going into the full area of study. But no significant differences were found among the three cases and hence, the research was not carried out for low and medium policy scenarios.

The study area of this work is described in detail in the following sections along with the input data descriptions as used in COPERT Street level (section 3.2.2) for street level emissions modelling and OSPM (section 3.2.3) for dispersion modelling. The results of emissions modelling and concentration modelling at street level are presented in section 8.4 and section 8.5 respectively. The following section 8.6 compares road by road emissions and concentrations and discusses the observations. The final section 8.7 concludes the chapter.

8.2. Street level emissions modelling using COPERT Street Level

The emission levels are modelled in all the major streets within the canal cordon in Dublin city as shown in Figure 8.1. Canal cordon represents 33 locations as shown in Figure 8.2. At these locations, detailed traffic data are collected, and these cordon locations were chosen to ensure that any person entering the city centre from outside will have to pass through one of these locations.

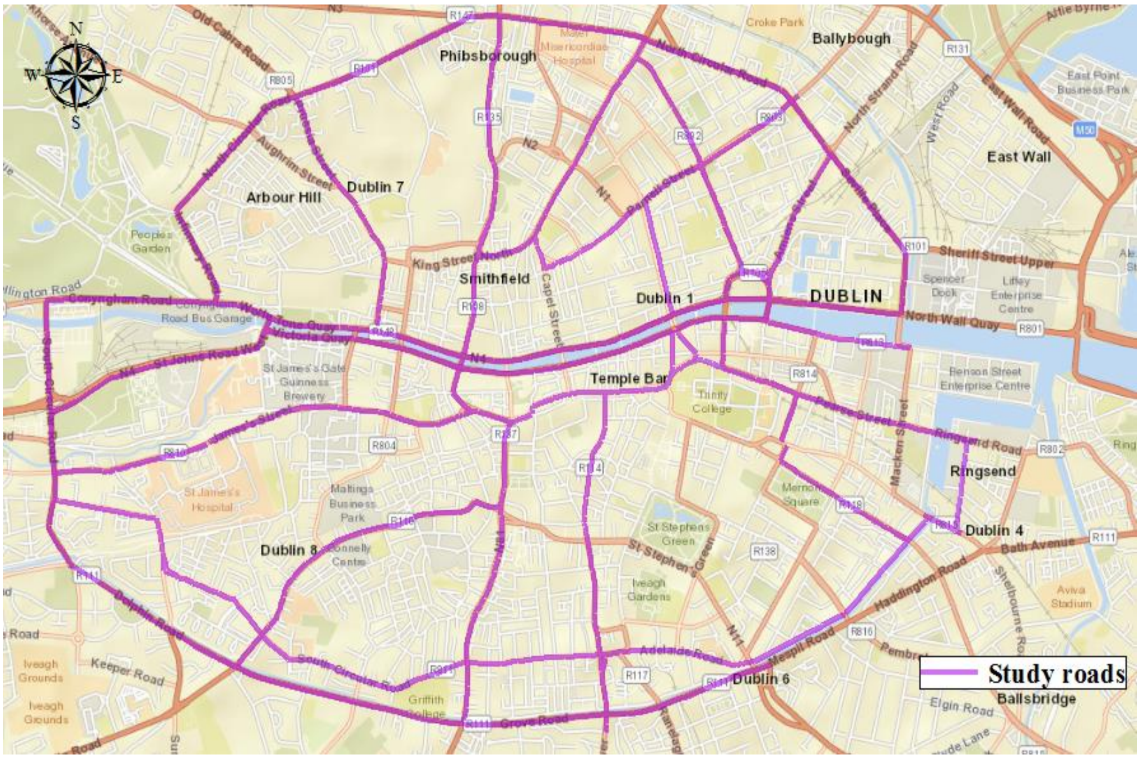


Figure 8.1. Study roads

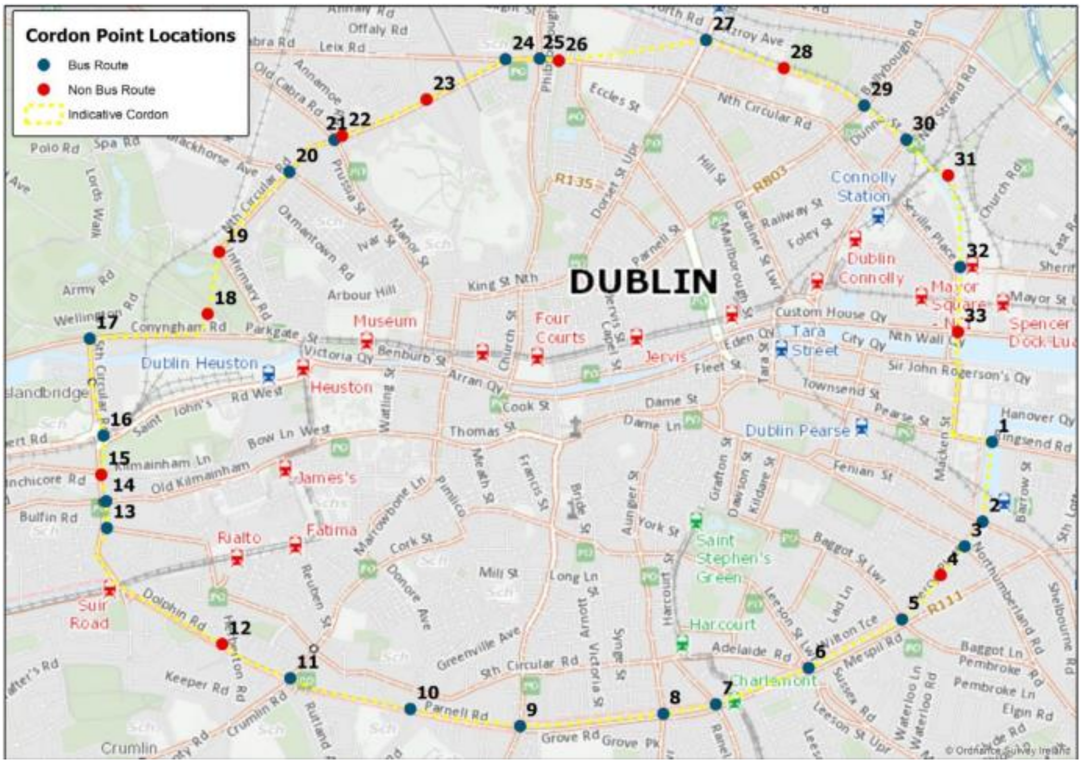


Figure 8.2. Canal cordon point locations (NTA, 2018)

Average speeds of vehicles on the study roads are shown in Figure 8.3. It can be observed that most of the roads have average speed in the range of 46-50 kilometres per hour (kmph). Annual Average Daily Traffic (AADT) on the modelled locations are shown in Figure 8.4 in Passenger Car Unit (PCU). The PCU factors are provided in Appendix B.

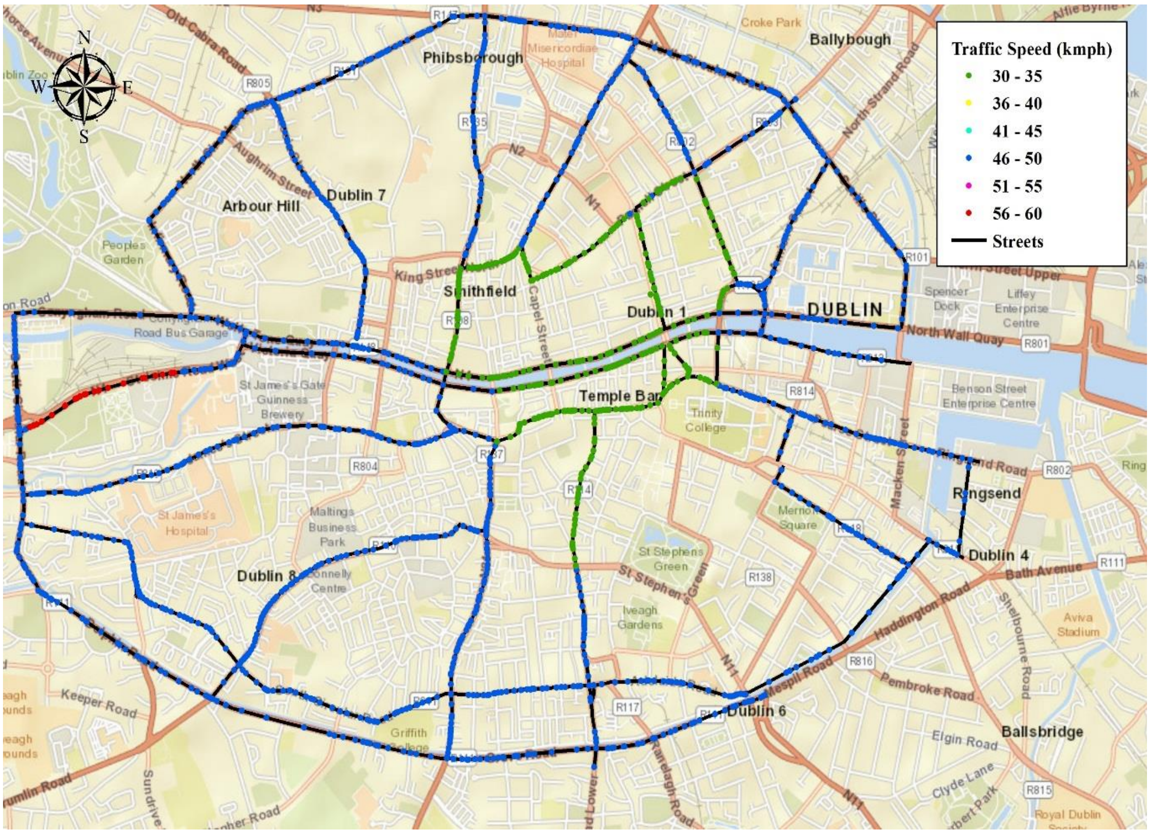


Figure 8.3. Traffic speeds on the study roads

Figure 8.4. AADT (PCU) on the study roads

Most of the roads contain traffic volume between 15000 and 25000 PCUs daily. Segment length, speed on a link and annual average hourly traffic data in terms of PCU on each road segment were supplied by Dublin City Council for 2015. In COPERT Street Level, PCU factors for each vehicle type and percentage share of each vehicle type in the overall fleet need to be defined. Based on those it disaggregates the overall traffic volume data into number of vehicles in each category of vehicle type.

It is possible to provide different compositions for each hour (see Appendix C) in COPERT Street Level. However, in this case, the composition is assumed to be the same in each hour to match the hourly traffic composition considered in OSPM. Figure 8.5 presents the hourly traffic configuration defined as input files. Total traffic volume data was divided into the individual volume of each vehicle type, such as PCs, LCVs, HDVs, buses and mopeds using the PCU factors used by Dublin City Council to aggregate the data. Then the traffic volume was refined to fuel types, technology classes and engine size (passenger cars)/weight class (for LCVs, HDVs and buses) based on the information obtained from Irish bulletin of vehicle and driver statistics DTTaS (2015a) and SIMI (2016). COPERT Street Level software window where these data were specified can be found in Appendix C. This was carried out to increase the accuracy of the estimates. The outputs are calculated as hourly emission levels on each road segment.

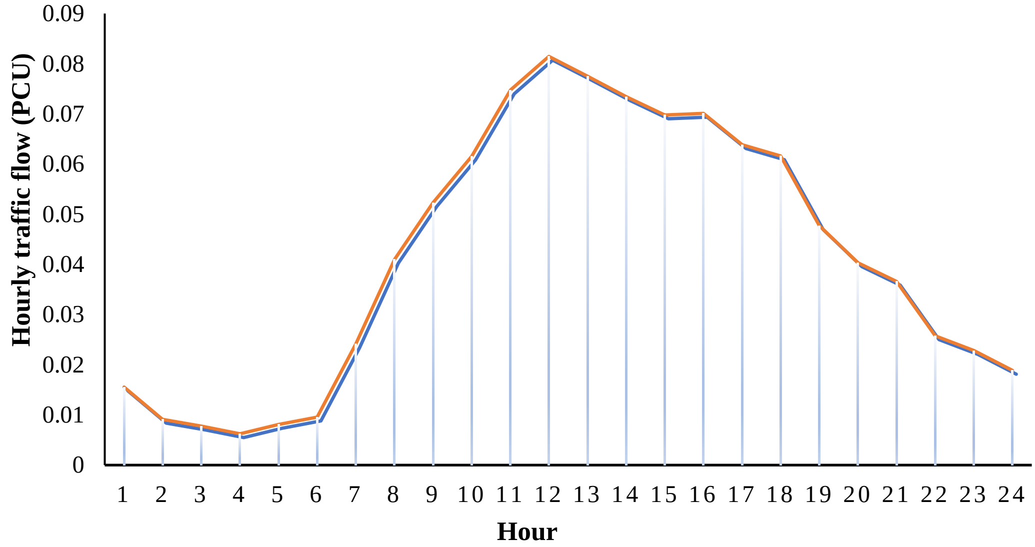


Figure 8.5. Hourly traffic composition

Traffic volume and the fleet composition for 2030 were estimated using National Transport Model (NRA, 2014) and using Systra's_rolling_fleet model (DTTaS, 2015b) respectively, as described in Chapters 5 and 6. Euro standard vehicular emissions were calculated by multiplying the number of vehicles in each category, the length of the road links and euro standard NOx emission factors corresponding to the respective vehicle category.

8.3. Dispersion Modelling

This section presents the data used in OSPM to estimate the pollutant concentrations on the study roads. Major roads within the canal cordon in Dublin city were chosen to be the study area as shown in Figure 8.6. Roads are divided into segments and data were entered for each segment. Road geometry and building data were provided by Dublin City Council in GIS format. The data is processed and added to OSPM and it forms the street network as shown in Figure 8.7. The average length of each street segment was taken as low as was practical and was approximately 100 meters to increase the accuracy of the modelling results. In this study, the fleet composition of passenger car, vans, bus, truck were obtained from DTTaS and CSO corresponding to each fuel and technology type. Updated NOx emission factors for Euro 5 PCs and LCVs are used as per updated COPERT emission factors based on *dieselgate* findings.

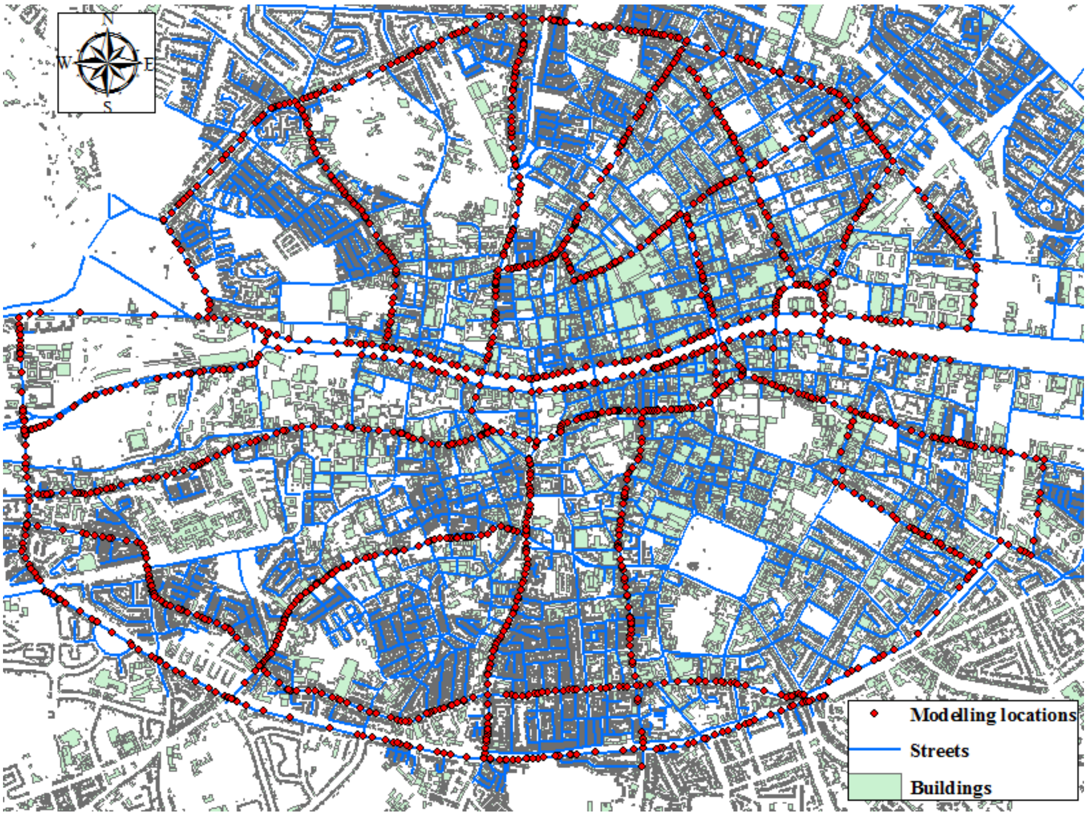


Figure 8.6. Study area and modelling locations

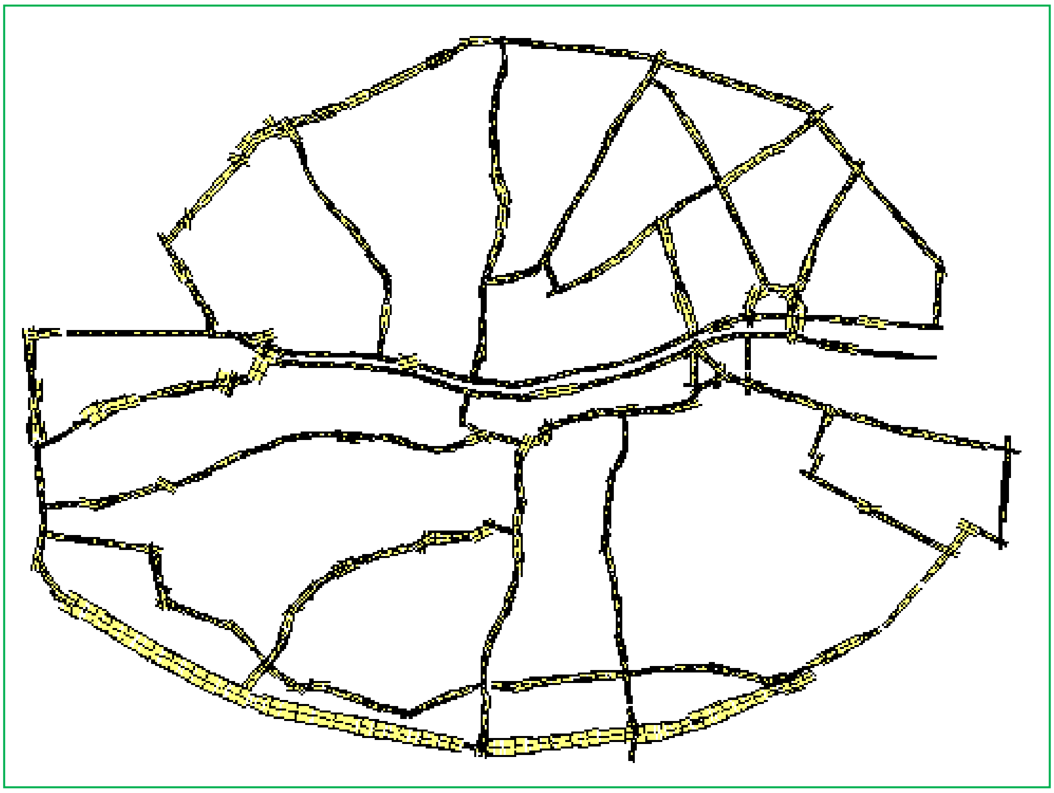


Figure 8.7. The layout of the study streets in OSPM

In this study, file specific to Dublin city was created and used. Meteorological data were obtained from MET Éireann (The Irish Meteorological Service, 2017). Urban background data for NOx, NO_2, $PM_{2.5}$ and O_3 were obtained from EPA, Ireland (EPA, 2017c, 2017d) monitoring locations. Figure 8.8 shows the hourly permissible EU/WHO limit (European Commission, 2017; WHO, 2005) and the NO_2 concentration data (EPA, 2017c, 2017d) recorded at monitoring locations in Dublin. Figure 8.9, 8.10, and 8.11 respectively present the hourly concentrations of NOx, $PM_{2.5}$ and O_3 recorded in monitoring stations in 2015 (EPA, 2017c, 2017d). These hourly concentration data were fed into OSPM for defining background concentrations of those pollutants in the study area.

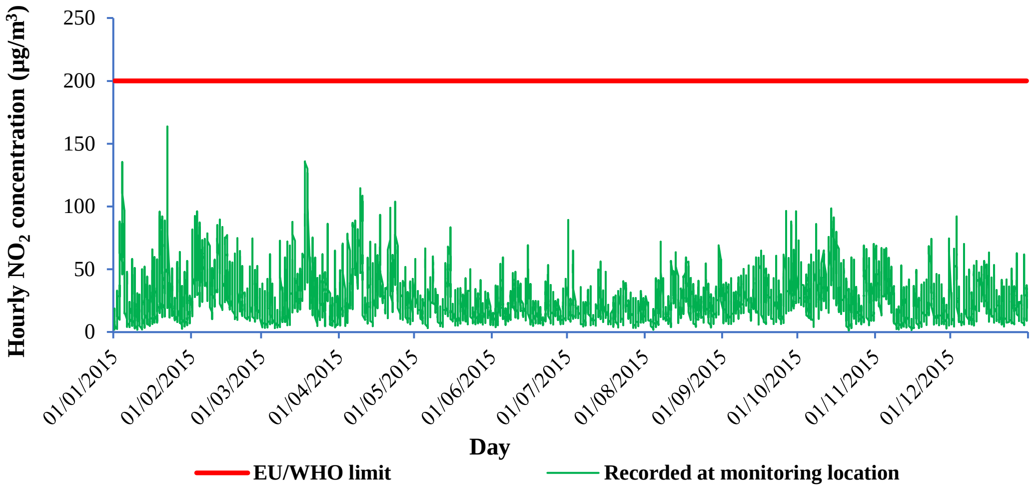


Figure 8.8. Permissible NO_2 limit and recorded concentration at monitoring locations

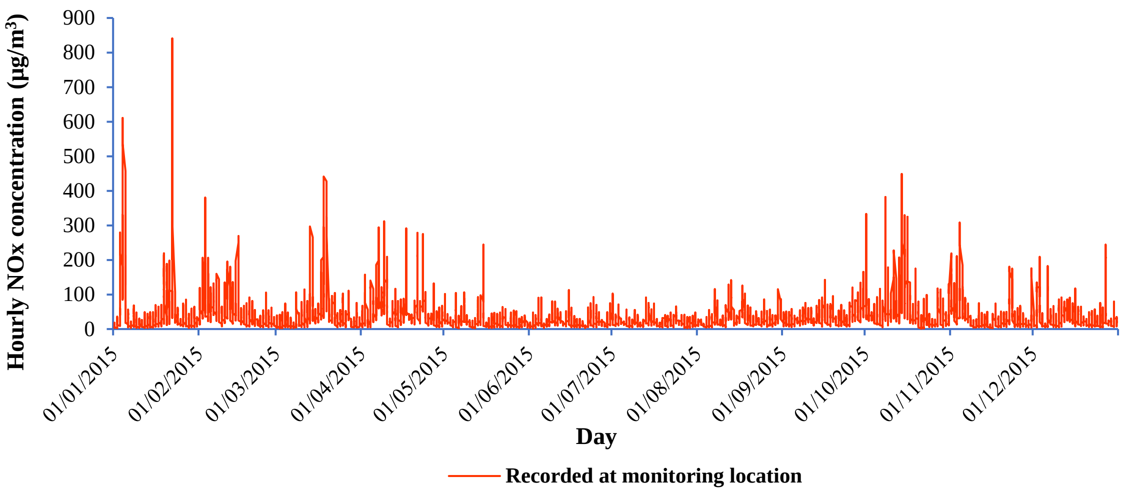


Figure 8.9. The recorded NOx concentration at monitoring locations

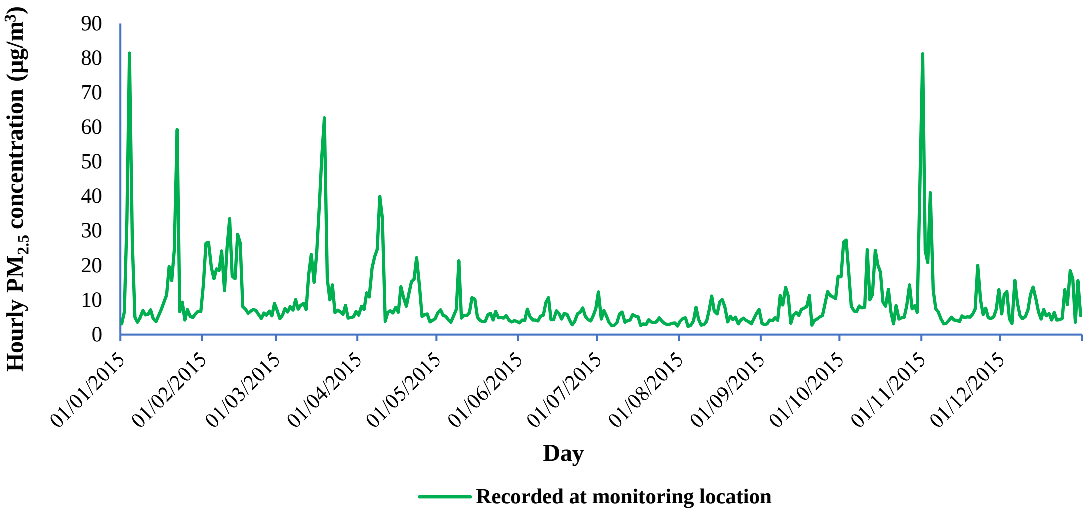


Figure 8.10. The recorded $PM_{2.5}$ concentration at monitoring locations

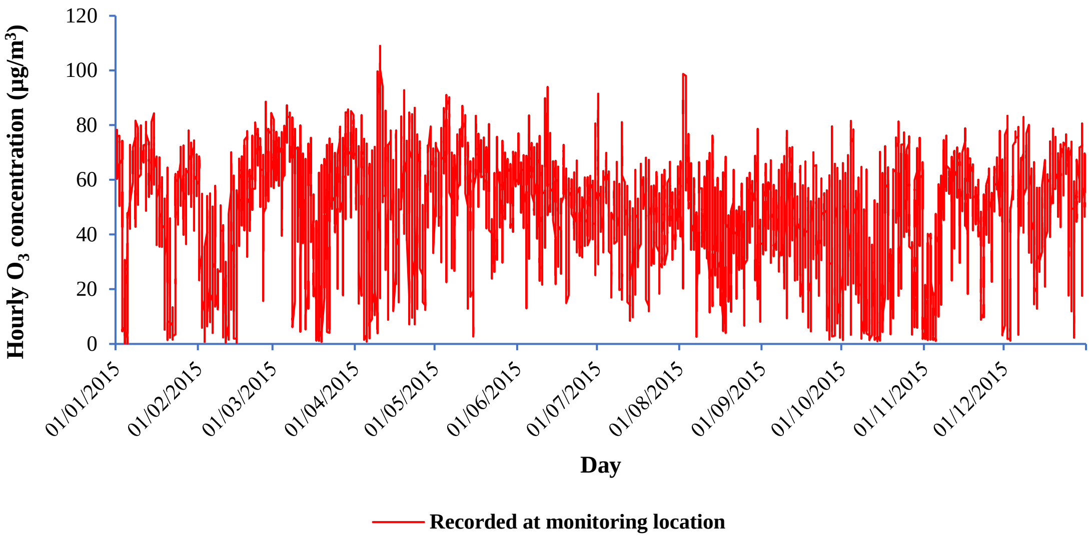
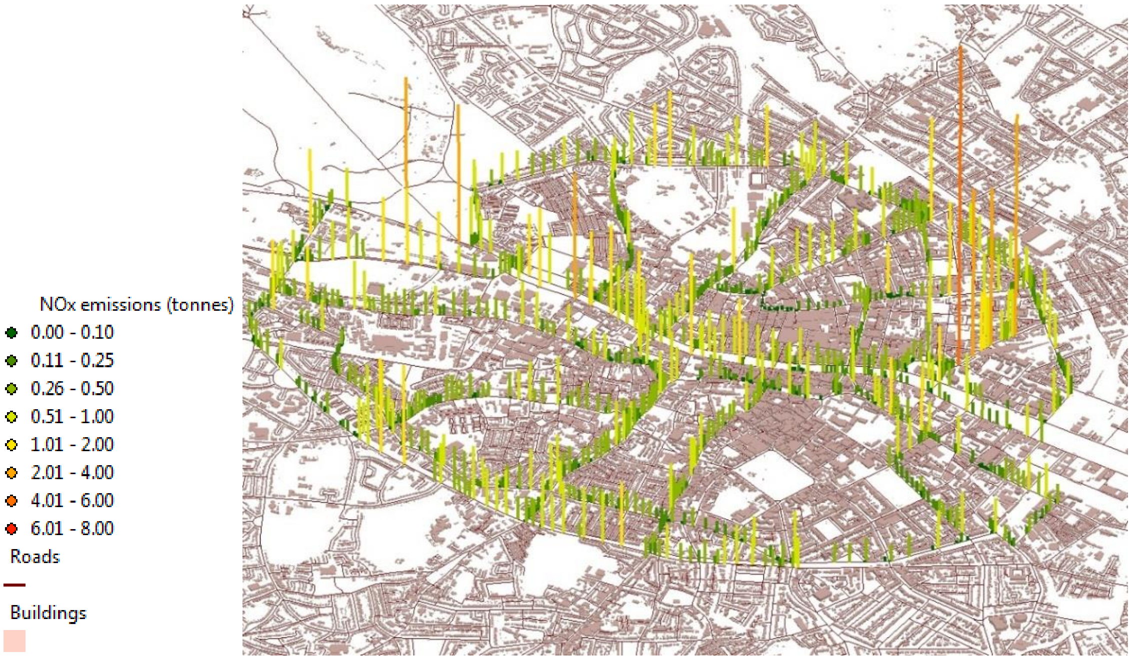


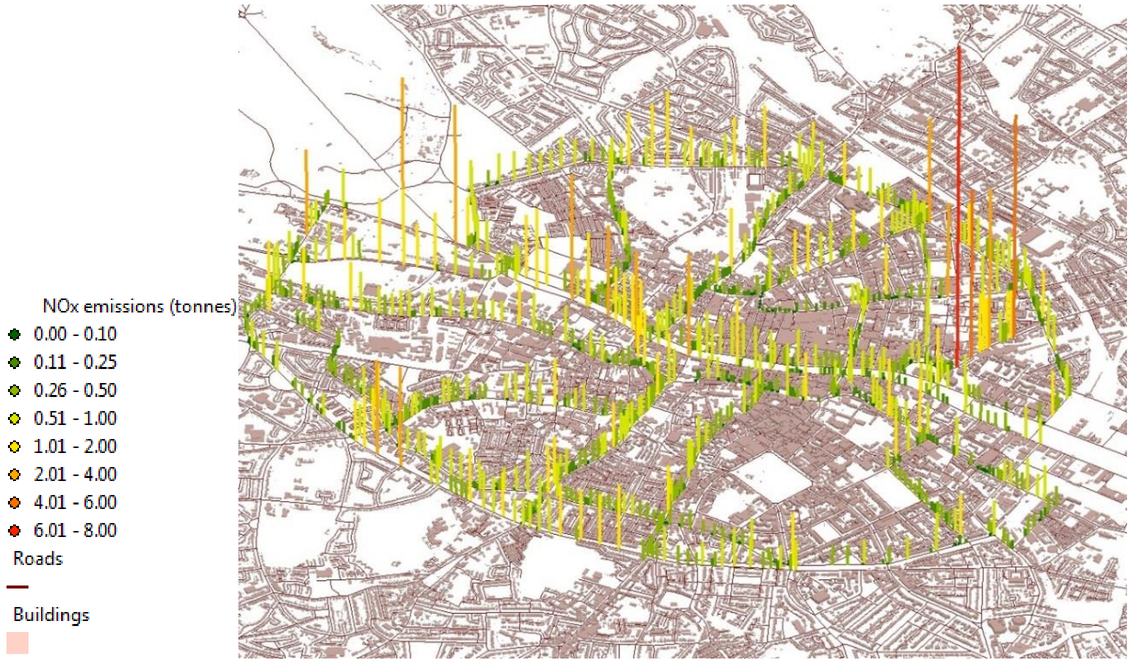
Figure 8.11. The recorded O_3 concentration at monitoring locations

8.4. Emissions on the streets

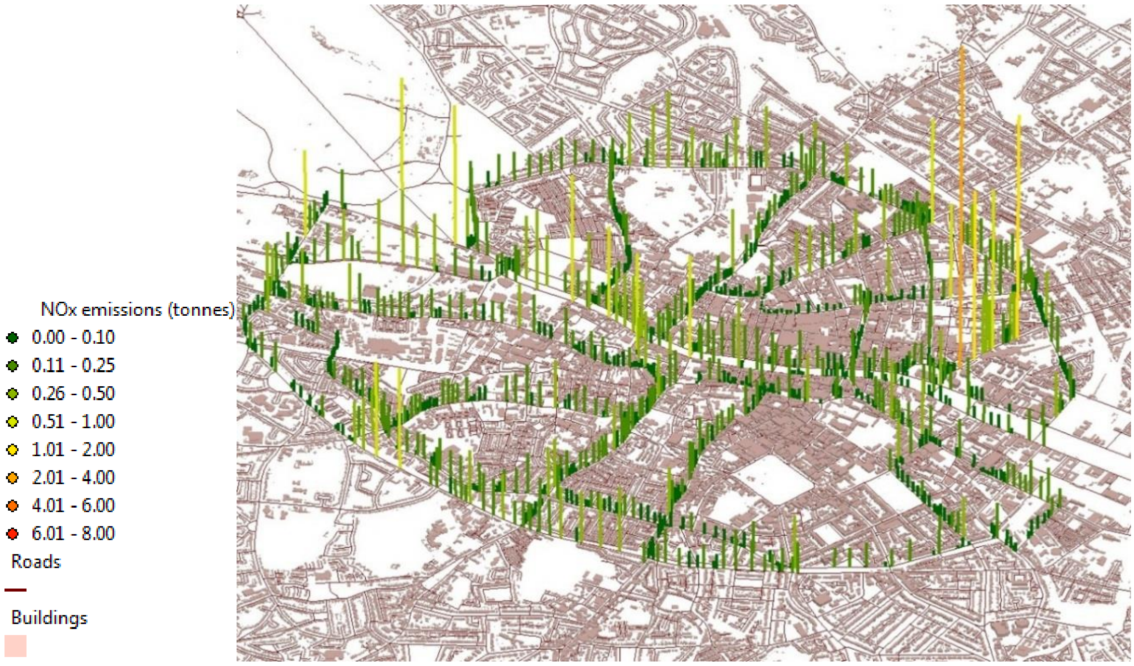
This section presents modelled annual average daily emissions of NOx at the street level resulted from the 2015 fleet and 2030 fleets under business as usual scenario and with new policy interventions. The COPERT Street Level modelling outcomes i.e. an annual average of total daily emission levels is presented in the form of GIS maps for better visualisation and understanding. Figure 8.12 (a), (b) and (c) presents the NOx emission levels (in tonnes) in 2015, 2030BaU and 2030Policy respectively.



(a) 2015



(b) 2030BaU



(c) 2030Policy

Figure 8.12. Modelled NOx emissions (tonnes) in (a) 2015; (b) 2030BaU; (c) 2030Policy

Table 8.1 presents the modelled emission levels of NOx obtained from COPERT Street Level. Expected NOx emission levels following Euro standard specifications were calculated for the study scenarios and they also are presented in Table 8.1. The results presented in Figure 8.12 revealed that in 2015, among the study roads, 41% of the overall road segments have daily emission levels between 0.26 and 2 tonnes. In 2030BaU, it is expected that there will be 52% street links in that emission range. However, in the 2030Policy scenario, this can be brought down to 7%. Also, it was found that with the alternative traffic fleet composition resulting from new policy applications, there will be 66% of total road segments containing NOx emission levels below 0.1 tonne and 27% in the range of 0.11-0.25 tonnes.

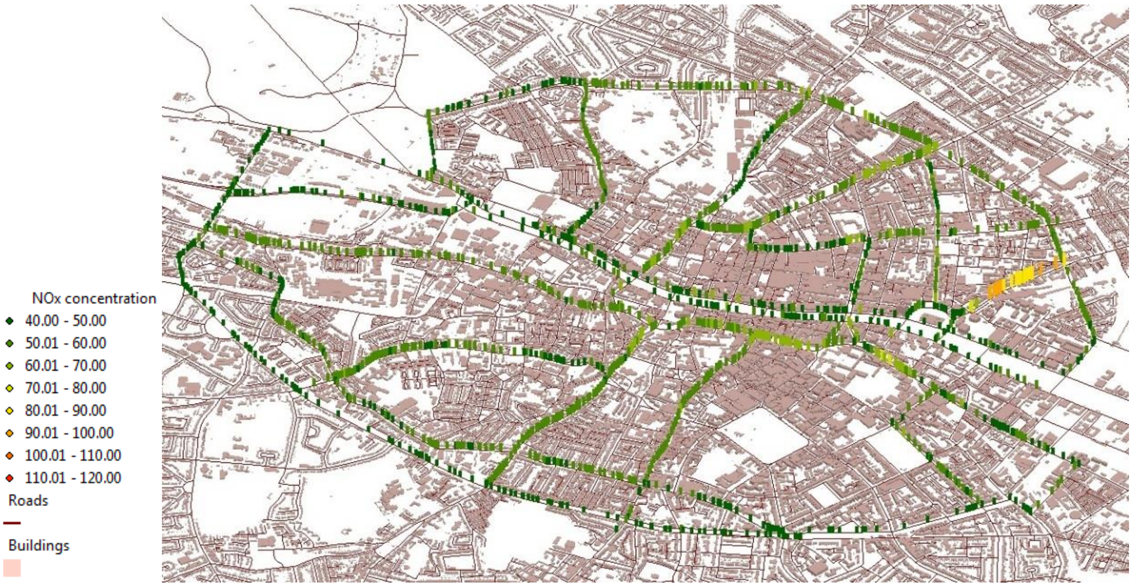
Table 8.1. Annual average daily NOx emissions and concentration

Study year	NOx Emission levels (tonnes)		Concentrations (μg/m^3)	
	Euro standard	Modelled	Measured	Modelled
2015	296	429	40.38	52.35
2030BaU	283	580	-	56.76
2030Policy	132	145	-	47.78

The results reveal that NOx emission levels are 45% higher than Euro standard emission levels in 2015. Whereas, under BaU, NOx levels from road transport fleet are projected to be 580 tonnes and the euro standard emissions are 283 tonnes. It can be observed that Euro standard emissions in 2030 are expected to be lower than that of 2015, this is because of a higher share of vehicles with stricter Euro standard emission factors (Euro 5 and Euro 6) in the fleet and retirement of the older vehicles with more polluting technology. Whereas, in reality, these NOx emission levels are about 105% higher in 2030 under BaU situation. However, upon implementing new policies (i.e. in 2030policy scenario) aiming at the reduction of diesel use, a reduction in NOx emissions by 75% from road transport fleet could be achieved in 2030, relative to the scenario in 2030 with no additional measure than the existing ones (2030 BaU scenario). Whereas, the difference between Euro standard emissions and modelled emissions can be reduced to 10% in the 2030Policy scenario.

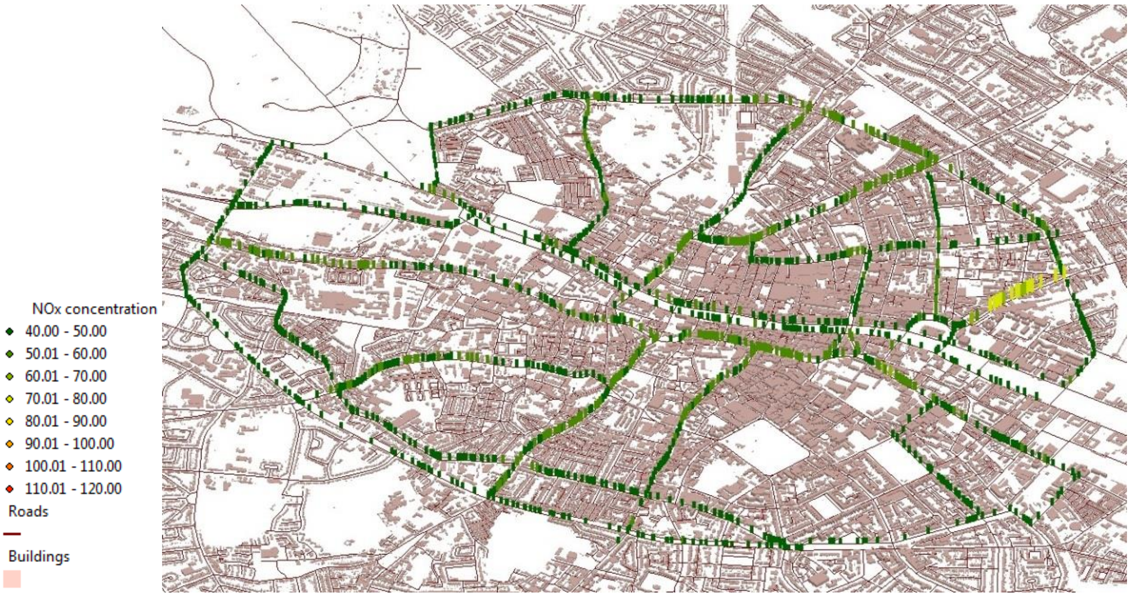
8.5. Concentration results

This section presents the OSPM modelling outcomes in terms of NOx, NO_2 and $PM_{2.5}$ concentrations on the same street canyons where the emissions were modelled. Figure 8.13 (a), (b) and (c) presents the street level NOx concentrations in 2015, 2030BaU and 2030Policy respectively. Therefore, on a particular street segment, NOx concentration shown in Figure 8.13 can be associated with the NOx emissions shown in Figure 8.12 on the same street.



(a) 2015

(b) 2030 BaU

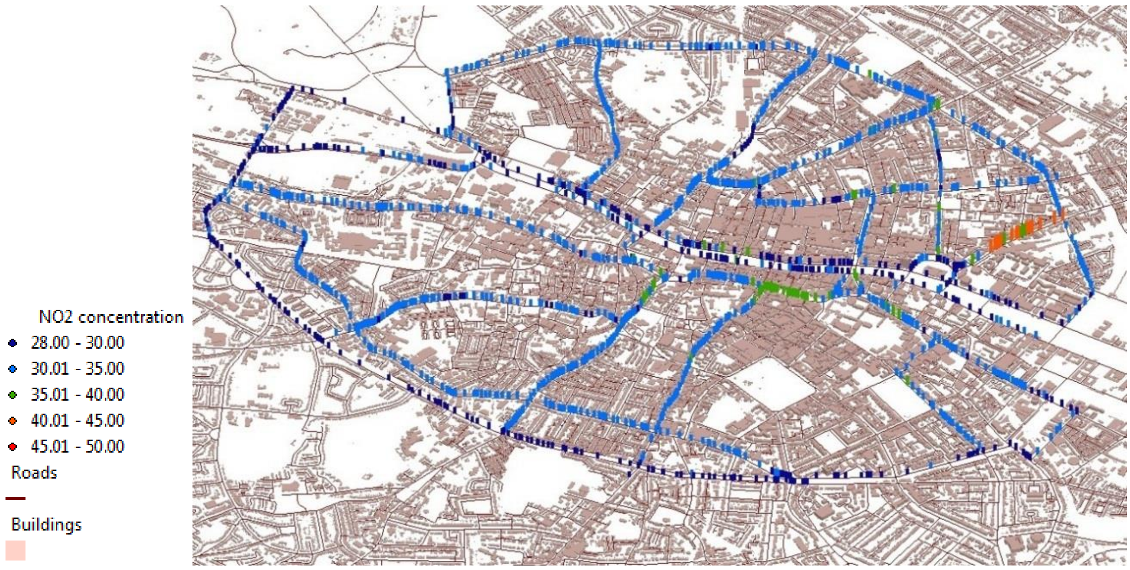


(c) 2030Policy

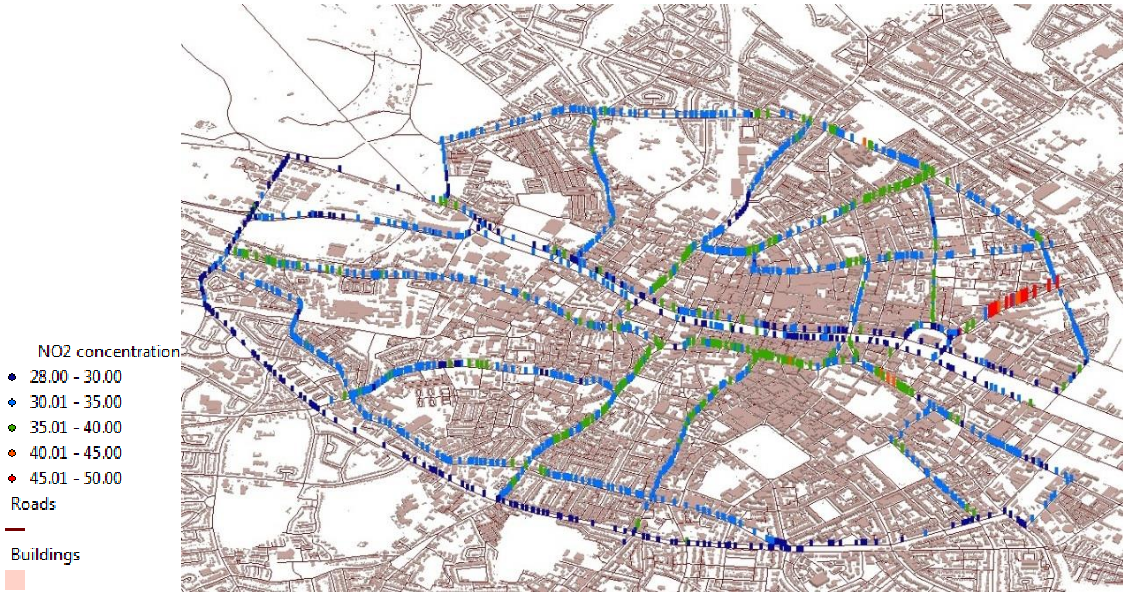
Figure 8.13. Modelled NOx concentrations (tonnes) in (a) 2015; (b) 2030BaU; (c) 2030Policy

Figure 8.13, (a), (b) and (c) show that the NOx exposure at the street level is expected to increase in 2030BaU but is significantly improved in the 2030Policy scenario. In 2015, 90% of the streets have concentrations between 40 to 60 μg/m^3 and in 2030BaU it is expected to decrease to 65%. However, in the 2030Policy scenario, 99% of the street canyons potentially will have NOx concentrations below 60 μg/m^3 out of which 71% with concentration levels below 50 μg/m^3.

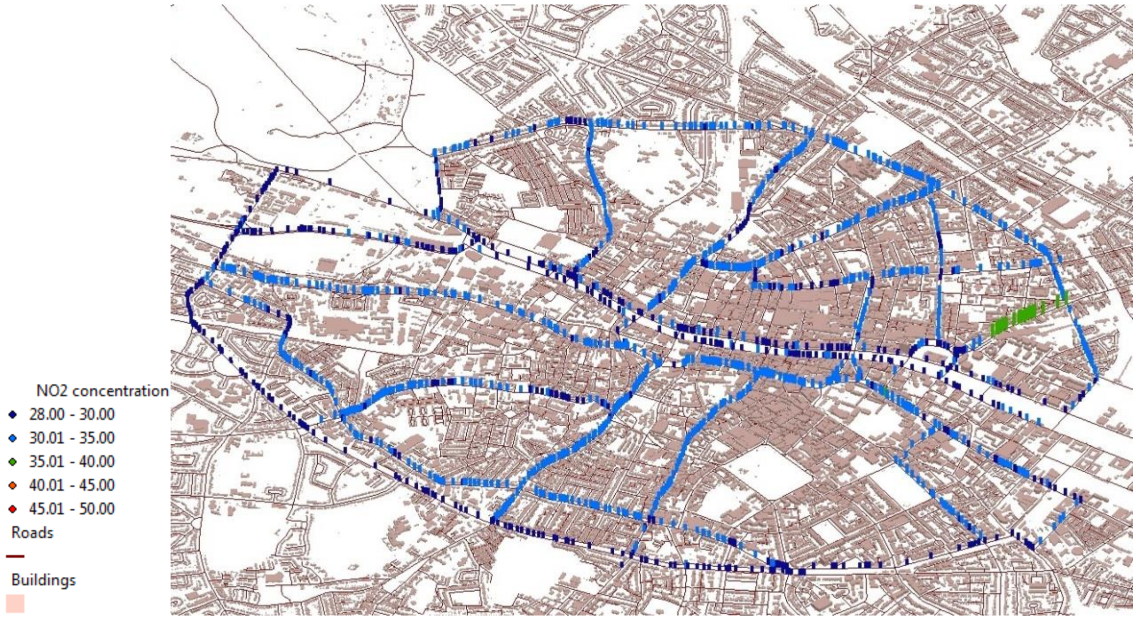
As mentioned earlier, though NO_2 were found to cause harm to human health, NO does not have a direct impact on human health. But NO eventually forms toxic NO_2 in the process of chemical transformation with O_3. NO_2 exposure levels due to both the direct NO_2 contribution from vehicle tailpipe and the NO_2 resulted from NO transformation are presented in Figure 8.14. This gives a better picture of the impact of NOx pollution on population health in different places. Figure 8.14 (a), (b) and (c) shows the NO_2 concentration levels on the study locations in 2015, 2030BaU and 2030Policy respectively. The results reveal that NO_2 concentration levels at the street levels are reasonably good when compared to the limit of 40 µg/m^3 recommended by WHO or EU. It was found that among 1,743 modelled street canyons, only 21 sections exceeded the guideline limit value, in 2015. Also, the results show that 94% of the total street segments have concentrations below 35 µg/m^3. In 2030BaU scenario, 2% of the modelled locations exceed the safe limit and 79% below 35 µg/m^3. NO_2 exposure at all the modelled street segments was found to be below 40 µg/m^3 in a 2030Policy scenario with 99% having NO_2 concentrations below 35 µg/m^3.



(a) 2015



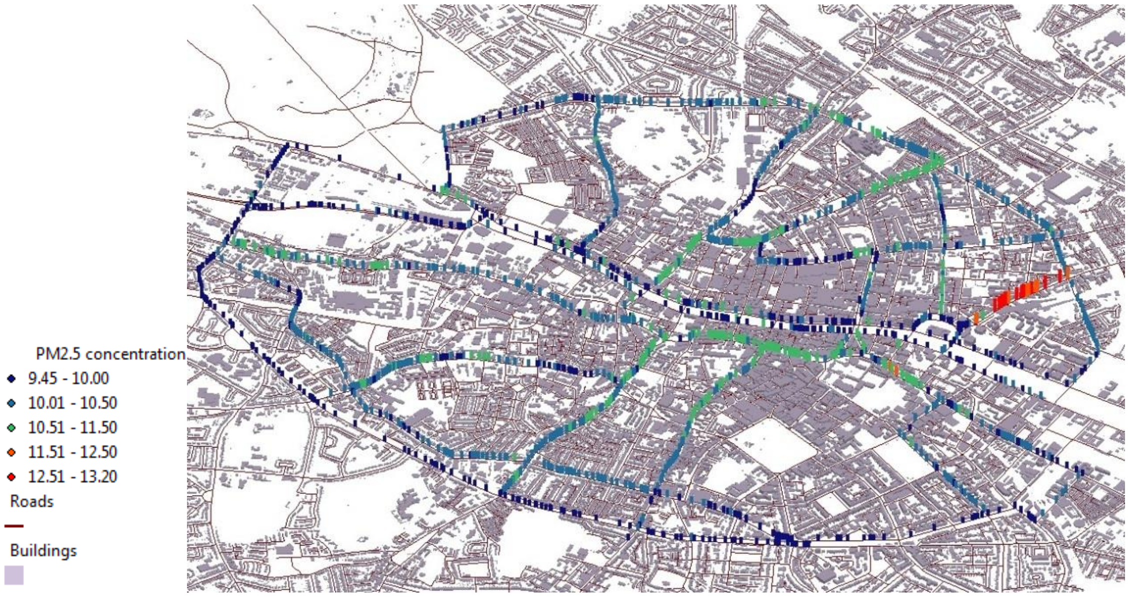
(b) 2030BaU



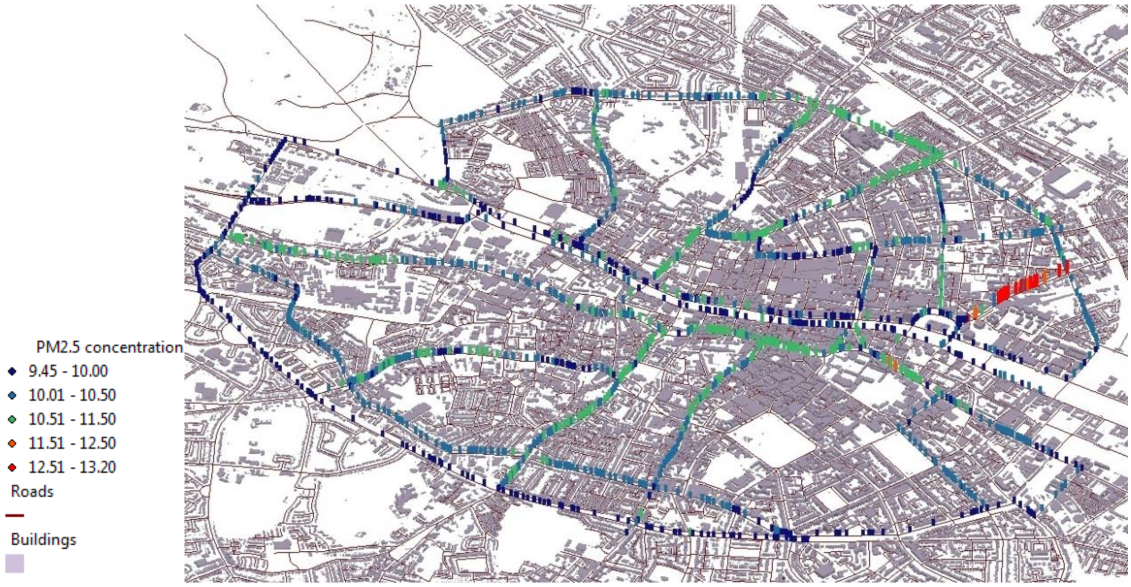
(b) 2030Policy

Figure 8.14. Modelled NO_2 concentrations (tonnes) in (a) 2015; (b) 2030BaU; (c) 2030Policy

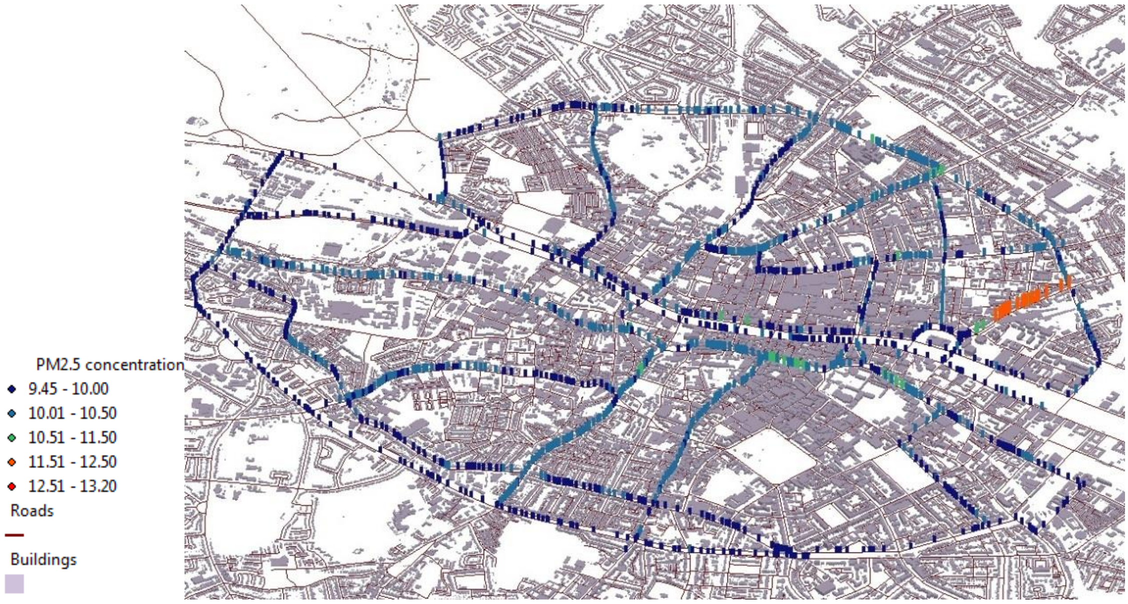
Annual average daily concentrations of $PM_{2.5}$ at the street level are presented in Figure 8.15 (a), (b) and (c) for 2015, 2030BaU and 2030Policy scenarios respectively. The results indicate that in 2015 68% of the modelled locations have $PM_{2.5}$ concentrations higher than WHO/EU safe recommended value of 10µg/m^3. In 2030, with the continuation of the BaU situation relative to 2015, this percentage will increase by 1%. This increase in $PM_{2.5}$ pollution level in 2030BaU is not as high as NOx or NO_2 levels compared to 2015. This is because 2030 fleet has a higher share of newer technology classes of vehicles which emit less $PM_{2.5}$ leading to a smaller increase in $PM_{2.5}$ concentration levels even though there is a significant increase in overall vehicle population and ICEV share. In the 2030Policy scenario, it is possible to reduce the number of locations exceeding the guideline value to 46% from 69% in 2030BaU. This is a considerable reduction given that there will be a significant number of petrol powered vehicles in the fleet.



(a) 2015



(b) 2030BaU



(c) 2030Policy

Figure 8.15. Modelled $PM_{2.5}$ concentrations (tonnes) in (a) 2015; (b) 2030BaU; (c) 2030Policy

Table 8.2 presents an average of modelled NO_2 and $PM_{2.5}$ concentrations at the study roads along with the recorded concentrations of those pollutants at air quality monitoring stations (EPA, 2017c, 2017d). Also, permissible limits specified by WHO and EU (WHO, 2005; European Commission, 2017) are listed in Table 8.2. It can be observed that both the modelled $PM_{2.5}$ and NO_2 concentrations are higher at the population exposure level than the levels recorded at the monitoring stations by 8% and 12% respectively. Also, it can be noted that although $PM_{2.5}$ concentrations exceed the guideline limit at some study points, the average of all the roads is below 10 µg/m^3 in the 2030Policy scenario.

Table 8.2. Annual average daily concentrations of $PM_{2.5}$ and NO_2

Study year/scenario	Annual mean concentrations (µg/m^3)					
	$PM_{2.5}$			NO_2		
	Recorded	WHO/EU	Modelled	Measured	WHO/EU	Modelled
2015	9.44	10	10.23	28.25	40	31.76
2030BaU	-	10	10.24	-	40	32.93
2030Policy	-	10	9.98	-	40	30.52

In Figures 8.16 and 8.17, concentrations of NO_2 and $PM_{2.5}$ at all the modelled street canyons are presented to obtain a better picture of their deviations with respect to the WHO/EU recommended safe guideline values. The figures include concentrations in all the study scenarios, i.e. 2015, 2030BaU and 2030Policy. Looking at Figure 8.16 it is fair to say that in the study area NO_2 pollution level is reasonably good when compared to the specified safe limit. For $PM_{2.5}$, the exposure levels are observed to be substantially high at several study points in 2015 and 2030BaU. Figure 8.17 shows that even though the situation can potentially be improved in the 2030Policy scenario, there are multiple locations with higher concentration levels than 10 μg/m^3.

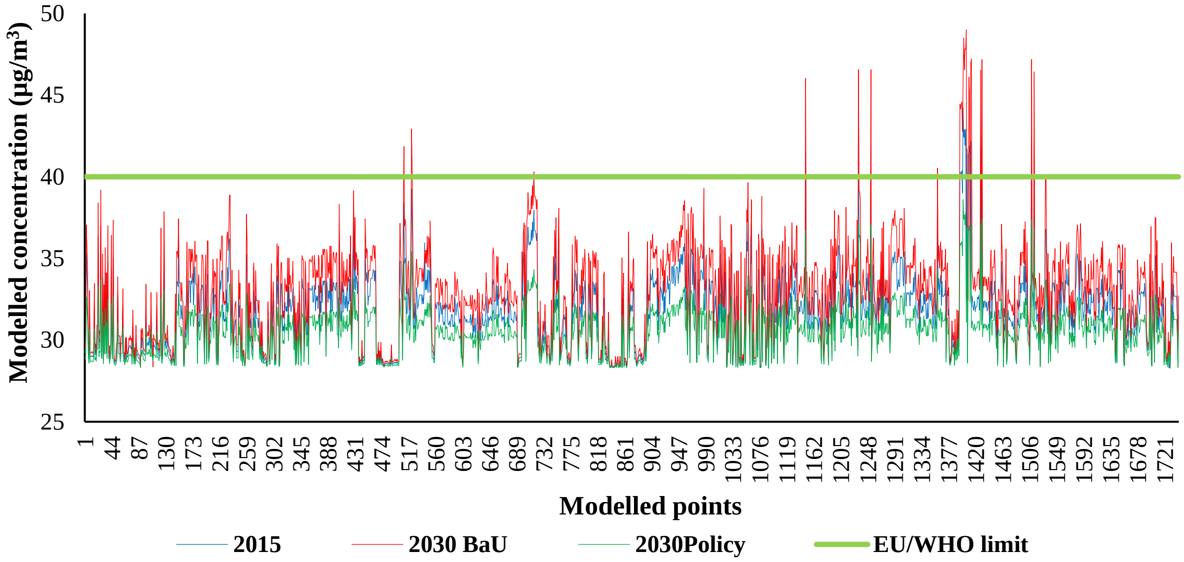


Figure 8.16. Modelled NO_2 concentrations and WHO/EU guideline value for annual average daily concentration

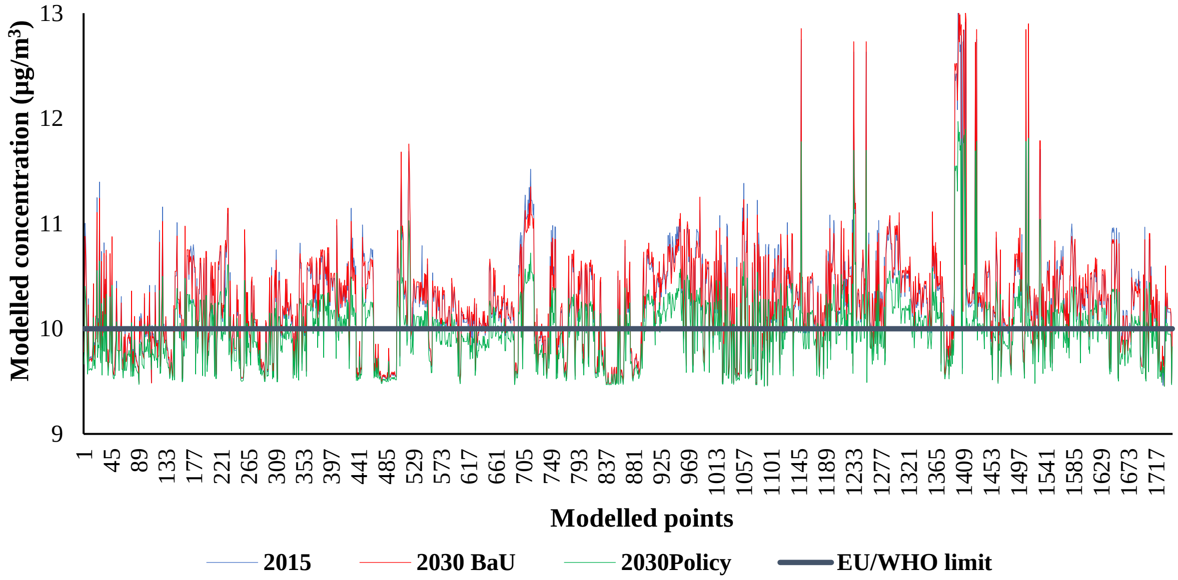


Figure 8.17. Modelled $PM_{2.5}$ concentrations and WHO/EU guideline value for annual average daily concentration

8.6. Comparison of emissions and concentrations on roads

In this section the street level modelled emissions and concentrations are compared in order to examine if the increase in emission levels leads to the same level of increase in concentrations. The modelled roads were ranked from high to low based on the quantity of NOx emissions and concentrations on the street. Top ten roads with the highest NOx emissions and concentrations were chosen and listed in the following table (Table 8.3). Both the total emissions (in tonnes) and emissions per unit length (in tonnes per metre) are listed in Table 8.3. As some roads are longer than others, it is more appropriate to compare the per unit length emissions with concentrations.

Table 8.3. Total NOx emissions (tonnes), emissions per unit length (tonnes/metre) and concentrations (μg/m^3) on roads

Rank	Road name	NOx emission (tonne)	Road name	NOx emission/ length (t/m)	Road name	NOx concentration (μg/m^3)
1	North circular road	30.94	Amiens street	0.035	Amiens street	84.1
2	South circular road	25.17	Tara street	0.015	Tara street	65.5
3	Amiens street	20.68	Dorset street lower	0.015	Dame street	63.3
4	Parnell road	10.11	Talbot memorial bridge	0.014	Dorset street lower	62.4
5	Pearse street	9.26	Bridge street lower	0.013	Portland row	61.6
6	Saint John's road west	9.26	Nicholas street	0.013	Nicholas street	61.0
7	Dorset street lower	7.91	Usher's quay	0.013	High street	60.6
8	Cork street	7.13	New street	0.013	Lord Edward street	60.2
9	Conyngham road	6.29	Bridge street upper	0.012	Church street	59.8
10	Grove road	6.28	Patrick street	0.011	College green	59.6

The results show that three among the ten streets correspond to the same level of emissions and concentrations. Also, it was found that out of around 150 modelled streets, only 6 streets correspond to the similar high or low levels of emissions and concentrations. As no specific pattern was observed in the study roads, it is fair to say that higher emissions may not always lead to higher concentrations or vice versa. In other words, an increase or decrease in concentrations are not directly dependent on the respective increase or decrease in emission levels. To investigate the relationship further, streets were chosen that have (a) higher emissions (HE) and higher concentration (HC); (b) higher emissions but lower concentrations (LC); (c) lower emissions (LE) but higher concentrations; (d) lower emissions and lower concentration. The four streets that fulfil these conditions were picked, namely, Amiens street (HE-HC), Usher's quay (HE-LC), D'Olier street (LE-HC), and Wilton terrace (LE-LC). Table 8.4 presents the NOx emissions, NOx concentrations, average speed, AADT on the streets, average building heights on the side of the streets, and orientation of the street with respect to north along with the width of the streets. Street views of the four roads are shown in Figure 8.18, 8.19, 8.20 and 8.21.

Table 8.4. NOx emissions, NOx concentrations, average speed, AADT, average building heights on the side of the streets, street orientation, and street width

Emission vs Concentration	Street name	Emission (t/m)	Concentration ($\mu g/m^3$)	Average speed	AADT	Building Height (m)	Street Orientation (degree)	Street Width (m)
HE-HC	Amiens street	0.035	84	50	> 40,000	7.3	20	51
HE-LC	Usher's quay	0.013	51	50	35,001 - 40,000	0 and 8.3	107	18
LE-HC	D'Olier street	0.008	59	30	20,001 - 25,000	11.9	136	22
LE-LC	Wilton terrace	0.004	43	50	10,001 - 15,000	0	62	51

Amines Street has high AADT leading to high levels of NOx emission, 0.035 tonnes/metre. Moreover, the average heights of buildings on both the side of roads are 7.3 meters. Therefore, it can be said that pollutants are is trapped at the street level for a while before they disperse. This leads to high NOx concentration levels in Amiens Street (Figure 8.18). In case of Usher's quay, though the AADT and resulting emission levels are high, concentrations are not proportionally high. Ushers quay has buildings only one side as shown in Figure 8.19.

Figure 8.18. Street geometry in Amiens Street

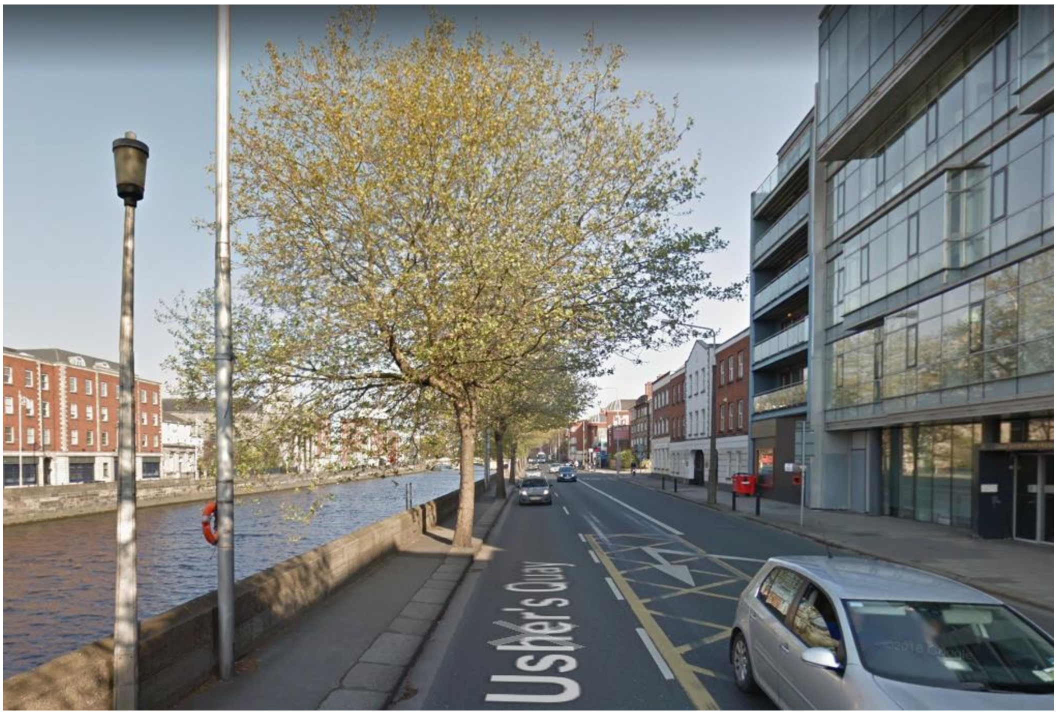


Figure 8.19. Street geometry in Usher's Quay

Therefore, the pollutants disperse at a quicker rate resulting in relatively lower concentration levels than those streets having similar or lower levels of emissions but higher concentration. D'Olier Street has AADT between 20,001-25,000 which is lower than Usher's Quay, therefore has lower NOx emission levels (see Table 8.4). However, the annual average daily NOx concentration is higher (59 $\mu g/m^3$) than that in Usher's Quay (51 $\mu g/m^3$).

Figure 8.20. Street geometry in D'Olier Street

Looking at the street geometry of D'Olier Street (Figure 8.20), it can be seen that it has relatively taller buildings of average height 11.9 meters (Table 8.4). Thus, pollutants are trapped for a longer duration than in Usher's Quay. In addition, D'Olier Street has bus stops and pedestrians crossing, leading to higher pollutant concentrations compared to Usher's Quay.

Figure 8.21. Street geometry in Wilton Terrace

For Wilton Terrace (Figure 8.21), where both the emissions and concentration levels are low, it can be observed that the AADT is in the range of 10,001-15,000 and that is why the emissions are relatively lower, 0.004 tonnes per metre. Also, it has a significantly low level of NOx concentration of 43 μg/m^3. Wilton Terrace does not have buildings on either side, therefore, the wind vortex is not formed, and the concentration level is low.

8.7. Conclusion

The research conducted in this chapter examined four main research questions, first, Do high NOx emission levels lead to high concentration levels of NOx and NO_2 in the base case scenario 2015?; second, How does the NOx emission levels for 2030 fleet under business usual situation and alternative 2030 fleet with policy interventions as described in Chapter 7 are going to impact the NOx and NO_2 concentration levels under those scenarios?; third, Do the modelled emissions have any relationship with concentrations?; forth, Do modelled $PM_{2.5}$ concentrations in 2015 and 2030 are below the acceptable limit?

The emissions and concentrations at the street level resulted from road transport fleet were modelled using COPERT Street Level and OSPM, respectively, on major urban roads in Dublin city. Although the emission levels were found to be about 45% and 100% higher than the respective Euro standard emission levels in 2015 and 2030 with no additional measures taken than the existing ones, NO_2 concentrations were found to meet WHO/EU safe guideline value in most of the modelling locations. In 2030, with alternative taxation and banning polices as detailed in Section 3.3.4 and Chapter 7, NO_2 concentrations in all the studied urban street canyons can be brought down below the recommended safe limit which is 40 μg/m^3.

However, $PM_{2.5}$ exposure levels are above the safe guideline value of 10 μg/m^3 at several locations in 2015, which is found to increase in 2030BaU. The number of locations exceeding the limit can be reduced in the 2030Policy scenario but will not be completely abated due to the increase in car ownership levels and significantly high share of petrol vehicles in the fleet. Nonetheless, the NO_2 and $PM_{2.5}$ concentration levels are higher than the baseline concentrations exposure above which is found to have health outcomes. In the next chapter, the health impacts of these modelled NO_2 and $PM_{2.5}$ pollution levels will be evaluated.

When the emissions and concentrations were compared for the same street, no specific pattern was noticed among the study roads to establish a relationship. Upon examining the data further, it was observed that the height of buildings on each side of the road has a strong influence on emission dispersion and thereby, on the street level concentration. Therefore, it can be concluded that the high level of NOx emissions may not lead to high levels of NOx or NO_2 concentrations.

Location specific wind direction and speed data were not available; therefore, the impact of these parameters could not be incorporated when comparing the NOx emissions and concentration levels. Future studies which investigate this topic further should consider the influence of wind speed and direction on the street emission to provide further insights into the correlation between emissions and concentrations. In the next chapter, health and cost impacts are estimated based on these modelled air pollution levels of NOx (NO_2) and $PM_{2.5}$. Although in Chapters 5 and 7, health impacts in terms of DALYs due to NOx emissions were calculated using damage factor approach, in Chapter 9, the health impacts were calculated from modelled concentrations and RR coefficients which is a more conventional approach followed in most of the related health impact assessment studies as discussed in Chapter 2.

Chapter 9: Health Impact Assessment

9.1. Introduction

The results presented in Chapter 8 indicate that the annual average NO_2 concentrations are below the WHO guideline value on most of the study locations in both the study years, 2015 and 2030. Whereas, $PM_{2.5}$ concentrations are above permissible limit for 2015 and 2030BaU but could be brought down below the limit with the proposed policies in place in most of the street canyons. However, the concentrations are above the counterfactual concentration and therefore, can be associated with health consequences including premature deaths. There is a limited number of studies (see section 2.5.2) on estimating health impacts due to NO_2 exposure. There are only a few air quality monitoring stations in Ireland that record $PM_{2.5}$ and NO_2 concentrations. For example, air quality data for NOx and $PM_{2.5}$ are available, respectively, in 9 and 10 monitoring stations in the whole of Ireland in 2015. Therefore, information on actual population exposure to air pollution levels of $PM_{2.5}$ and NO_2 in different areas is not available. The recorded data at monitoring stations may not reflect the concentrations in other areas where monitoring is not carried out. Therefore, the population exposure and health impacts in different areas within the city cannot be calculated or compared.

In this Chapter, modelled NO_2 and $PM_{2.5}$ concentrations obtained at the points as presented in Chapter 8, were interpolated using kriging over the 49 Electoral Divisions (EDs) located within the canal cordon. EDs are the smallest legally defined administrative areas in the State (CSO, 2018). Spatial distribution in the form of GIS maps is presented for better visualisation of the results. Also, this gives the idea about the variations of population exposure to NO_2 and $PM_{2.5}$ in different areas within the city. Spatial variation of NOx concentrations is presented also. Based on the modelled concentrations in each ED, disease burden attributable to the NO_2 and $PM_{2.5}$ pollution due to road traffic were calculated as per WHO BOD method (WHO, 2004). The burden of disease is the impact of health problem and can be measured by mortality, morbidity, monetary value or other indicators. In this study, the burden of disease was estimated in terms of mortality, YLL and VSL. YLL is an estimate of the number of years someone would have lived if that person had not died early (EEA, 2017). YLL gives more assimilated information of health impact than death numbers as it takes into account the age at which the death occurred to every individual. In addition to computing health impacts for the baseline scenario (2015), the potential impacts on health in 2030 with the BaU situation and the possible reductions with the policy implications were also estimated.

The next section 9.2 provides an overview of the kriging interpolation method used to predict the concentrations in EDs in the study area. This is followed in section 9.3 by a description of the data used to calculate mortality incidences due to NO_2 and $PM_{2.5}$ pollution using WHO BOD method. The next section 9.4 presents the spatial variation maps of modelled pollutant concentrations of NO_2 and $PM_{2.5}$. Health and cost impacts were then assessed based on the methods described in sections 3.2.4 and 3.2.5 and presented in section 9.5. The final section 9.4 concludes the chapter.

9.2. Population Exposure Assessment

The concentrations modelled by OSPM at the study locations were interpolated to predict the concentrations in the ED population districts (Figure 9.1) within the study area. The concentrations at different locations within the study area where the concentrations are not known were predicted by kriging interpolation using Geostatistical analyst tool in ArcGIS. Kriging is an advanced geostatistical procedure that generates an estimated surface to predict values for unmeasured locations from a scattered set of points of known values. Therefore, kriging is a method of interpolation and expresses the spatial variation of the parameter in terms of variogram (Oliver and Webster, 1990).

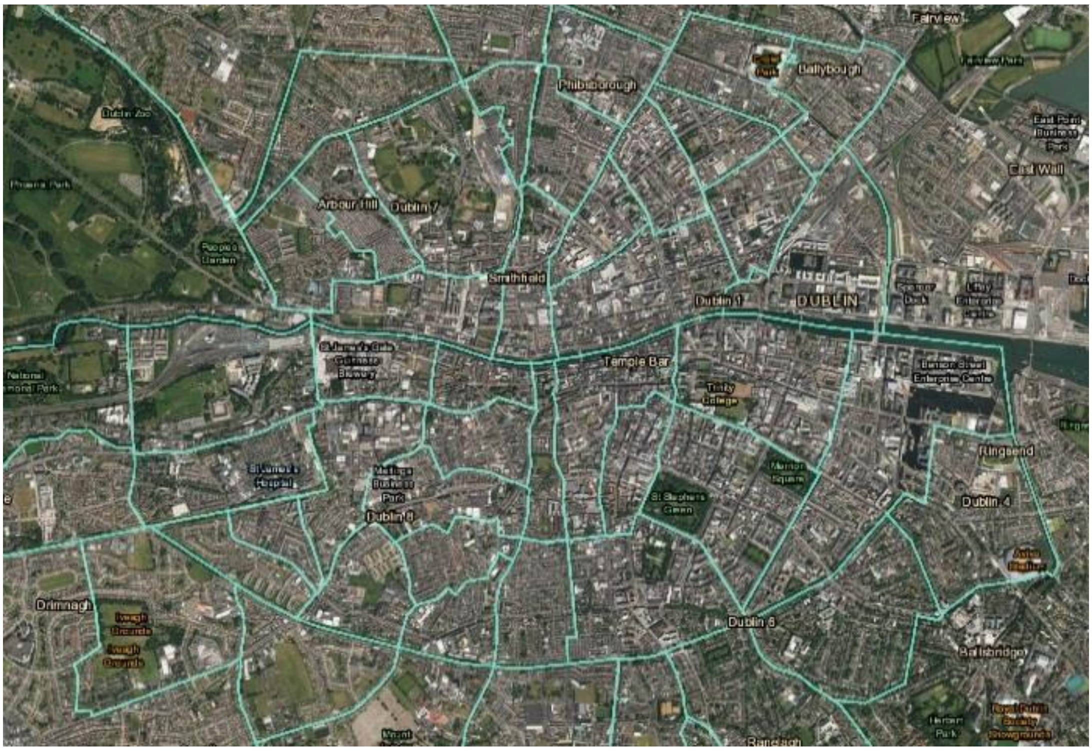


Figure 9.1. Electoral Division boundaries in Dublin city

The general kriging formula as used in ArcGIS can be presented by the following equation (ESRI, 2016),

$$\hat{Z}(s_0) = \sum_{k=1}^{K} \lambda_k Z(s_k) \tag{9.1}$$

Where $\hat{Z}(s_0)$ is the interpolated value; s_0 is the prediction location; K is the number of measured values; λ_k depends on the distance to the predicted location; $Z(s_k)$ is the measured value at the k^{th} location. The kriging tool in ArcGIS at first creates variogram and covariance functions to estimate the spatial correlation of the data and then makes the predictions. The number of minimum and maximum neighbouring points of known concentrations to be considered in predicting concentration at an unknown point can be specified by the user. The concentrations of NOx, NO_2 and $PM_{2.5}$ at other locations within the EDs were thereby predicted based on the modelled concentrations of NOx, NO_2 and $PM_{2.5}$ presented in Chapter 8. Separate maps were created for all the study scenarios, i.e. 2015, 2030BaU and 2030Policy for NOx, NO_2 and $PM_{2.5}$. Then the entire study area was divided into small stripes and overlaid on the interpolated concentrations to find the concentrations in each stripe (see Figure 9.2). Then the concentrations in each small stripe were extracted and the concentrations in the stripes lying within each ED were averaged to represent the concentration in that ED. The pollution exposure within that ED was estimated based on this averaged concentration and used to calculate the health consequences. Health outcomes in terms of premature mortality incidences and YLL, as a result of the modelled exposure levels of NO_2 and $PM_{2.5}$ pollution from road transport were calculated in each ED for 2015, 2030BaU and 2030Policy scenarios. Figure 9.2 shows an example of how the study area is divided into small stripes and overlaid upon the modelled concentrations in that area.

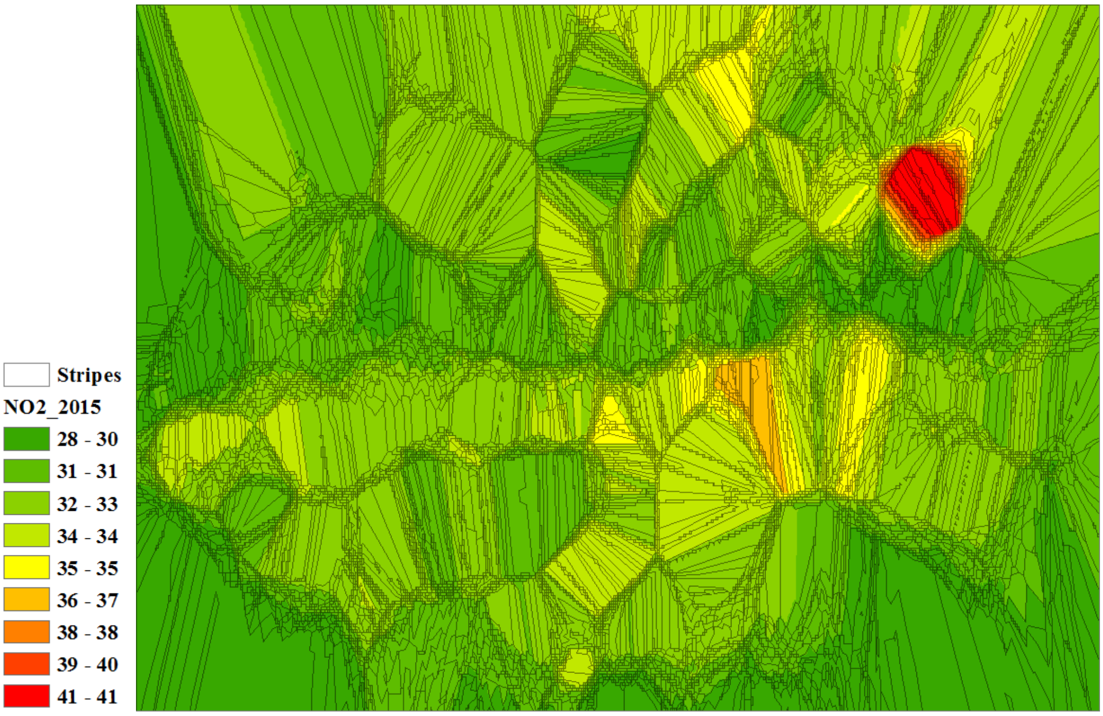


Figure 9.2. Example of interpolated concentrations (NO2_2015) and overlaid stripes on the study area

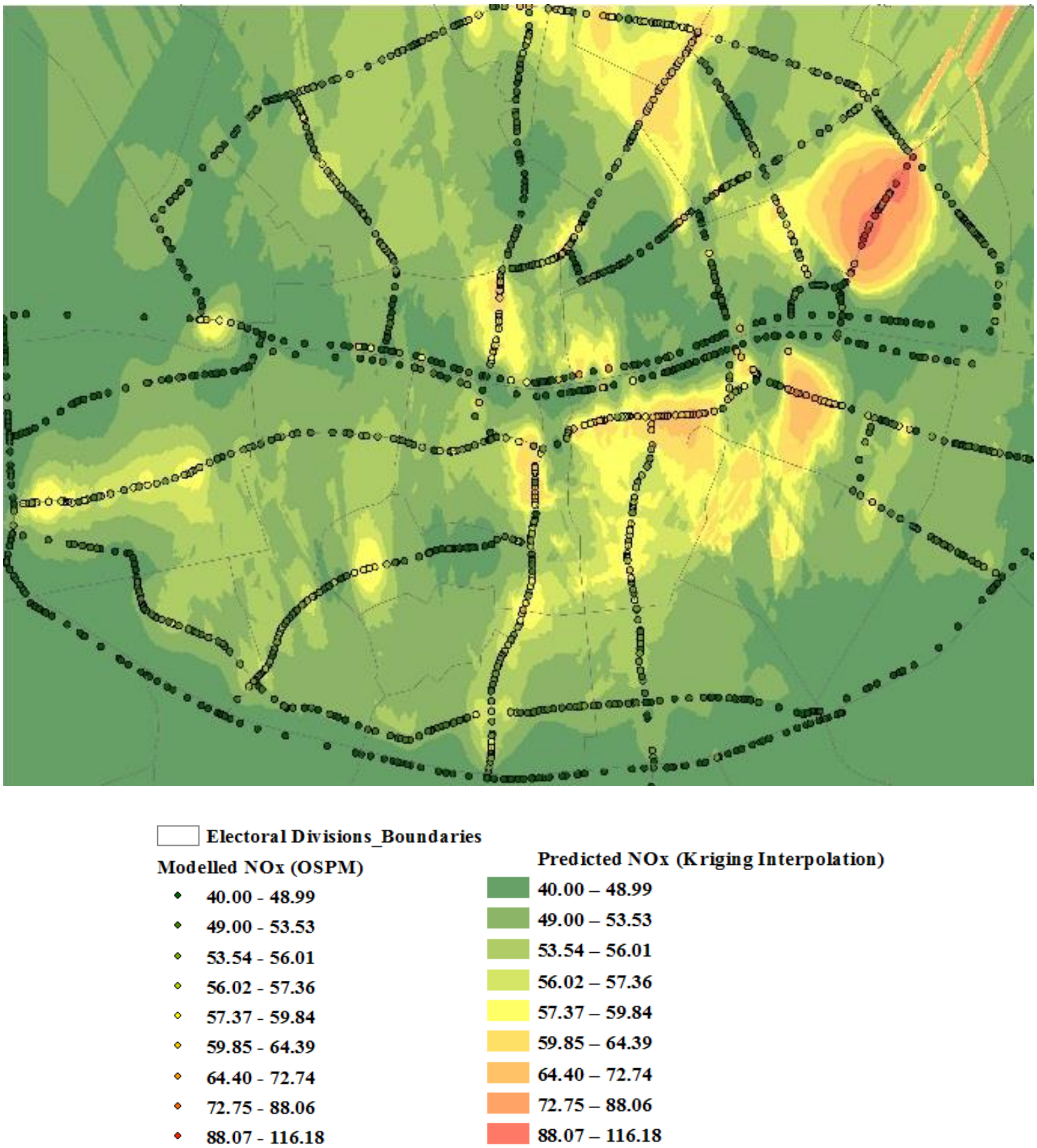
9.3. Data used for BOD estimation

The WHO BOD method is described in section 3.2.4. In this study, C_0 for $PM_{2.5}$ was taken as 2.5µg/m^3 which corresponds to the lowest concentration found in populated areas (Horálek et al., 2017). For NO_2, mortality was calculated based on baseline concentrations as suggested in the literature. In some previous European studies, C_0 for NO_2 was taken as 20 µg/m^3 following HRAPIE recommendation (WHO, 2013b). However, it was indicated by some of the recent studies (e.g. Héroux et al., 2015) that 20 µg/m^3 might be too high and 10 µg/m^3 was observed to be the lowest concentration (Raaschou-Nielsen et al., 2012) at which a significant correlation between NO_2 concentration and health outcomes were observed. Therefore, for NO_2, two separate estimates were made considering the counterfactual concentrations as 10 µg/m^3 and 20 µg/m^3. For all-cause mortality, β is taken as 1.039 for 10 µg/m^3 increase in NO_2 concentration (WHO, 2013b; EEA, 2017). This value is reduced by 30% from 1.055 to account for possible overlap with $PM_{2.5}$ (EEA, 2017; Walton et al., 2015). β value for $PM_{2.5}$ (WHO, 2013b) for all-cause mortality due to long term exposure is taken as 1.062 for 10 µg/m^3 increase in $PM_{2.5}$ concentration (WHO, 2013b). These β values represent the estimation at a 95% confidence interval. It can be observed that β value, i.e. the risk of an adverse health effect due to a unit change in ambient air pollution is higher for $PM_{2.5}$ than NO_2. This observation is comparable to the higher damage cost per tonne $PM_{2.5}$ emissions than per tonne NOx emissions (DTTaS, 2016a). The premature deaths attributable to short term changes of $PM_{2.5}$ are accounted for in estimating the effects of long term exposure (WHO, 2013b). Baseline mortality rate is taken as 8.1 per year per 1000 population (CSO, 2016). The current (2015) and predicted future (2030) population were obtained from CSO (2016) and Demographic and Economic Forecasting report (NRA, 2014) respectively.

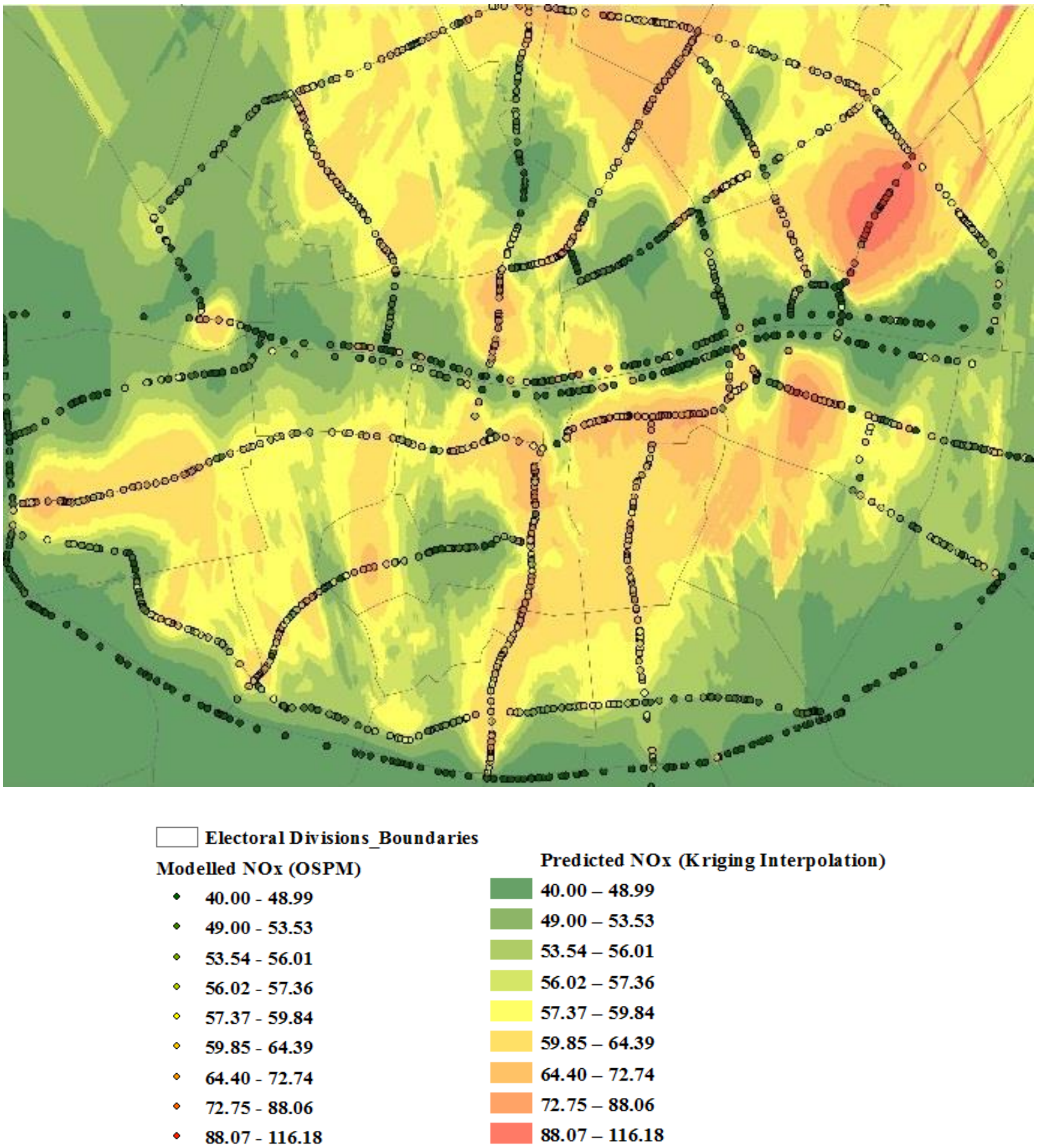
Standard life expectancy at age of death was taken as 11 years for Ireland (EEA, 2017) to calculate YLL. For Ireland, the given value is 3.75 million USD per incidence (WHO Regional Office for Europe, 2015). In this work, the total annual monetary impact due to premature deaths from NO_2 and $PM_{2.5}$ pollution was calculated by multiplying this unit value by estimated mortality values in 2015. The deviations in VSL in 2030BaU and 2030Policy with respect to 2015 were also calculated.

9.4. Population exposure to urban NOx, NO_2 and $PM_{2.5}$ pollution

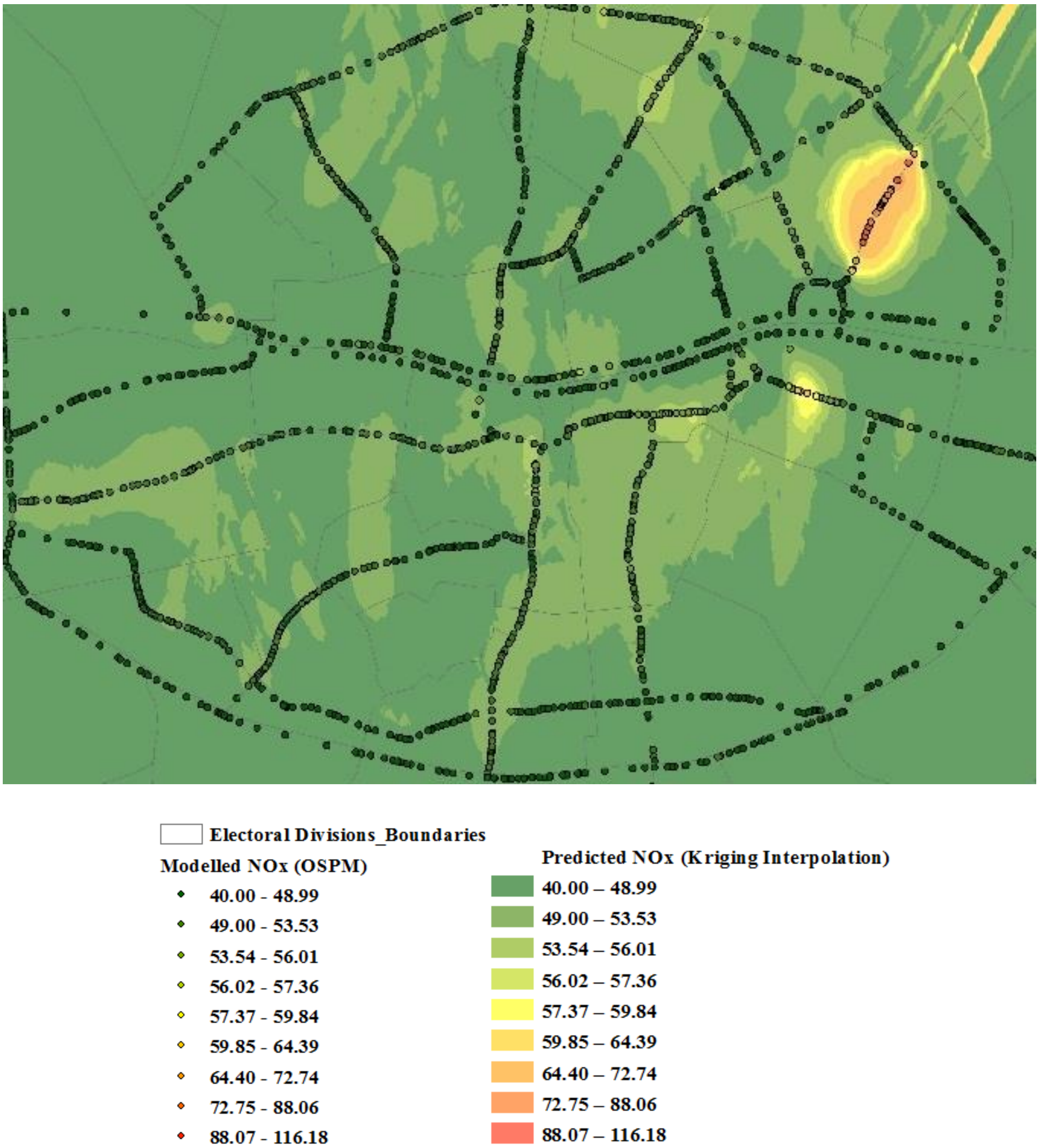
This section presents the modelled concentration results from OSPM for 2015, and future year cases, i.e. 2030BaU and 2030Policy. The modelled concentrations of NOx, NO_2 and $PM_{2.5}$ on the study roads are presented in GIS maps for better visualisation of the spatial variations of the exposure. Figure 9.3 (a), (b) and (c), respectively, shows the OSPM modelling outcomes and interpolated concentrations of NO_x within the EDs due to NO_x emissions from road transport fleet in 2015 road traffic, in 2030BaU and in 2030Policy.


Electoral Divisions_Boundaries
Modelled NOx (OSPM)
40.00 - 48.99
49.00 - 53.53
53.54 - 56.01
56.02 - 57.36
57.37 - 59.84
59.85 - 64.39
64.40 - 72.74
72.75 - 88.06
88.07 - 116.18
Predicted NOx (Kriging Interpolation)
40.00 – 48.99
49.00 – 53.53
53.54 – 56.01
56.02 – 57.36
57.37 – 59.84
59.85 – 64.39
64.40 – 72.74
72.75 – 88.06
88.07 – 116.18

(a) 2015


Electoral Divisions_Boundaries
Modelled NOx (OSPM)
40.00 - 48.99
49.00 - 53.53
53.54 - 56.01
56.02 - 57.36
57.37 - 59.84
59.85 - 64.39
64.40 - 72.74
72.75 - 88.06
88.07 - 116.18
Predicted NOx (Kriging Interpolation)
40.00 – 48.99
49.00 – 53.53
53.54 – 56.01
56.02 – 57.36
57.37 – 59.84
59.85 – 64.39
64.40 – 72.74
72.75 – 88.06
88.07 – 116.18

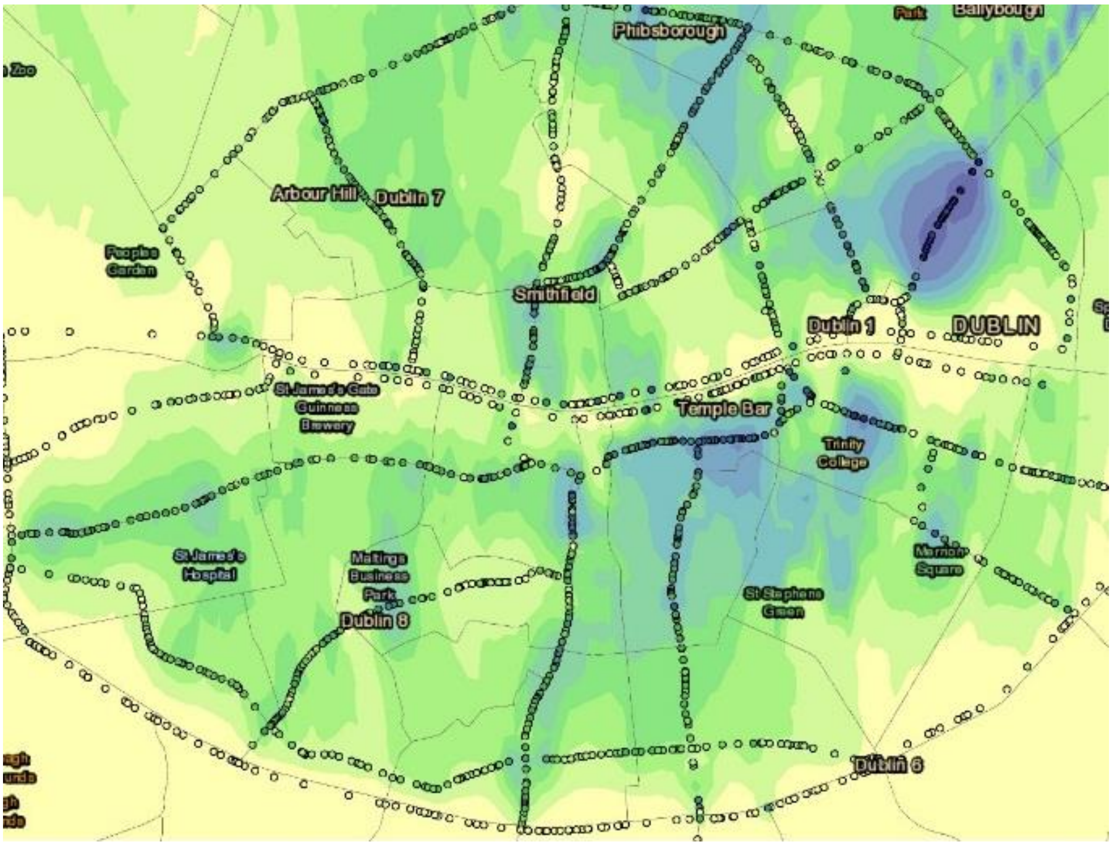
(b) 2030BaU



(c) 2030Policy

Figure 9.3. Modelled NOx concentrations and spatial variation in (a) 2015 (b) 2030BaU (c) 2030Policy

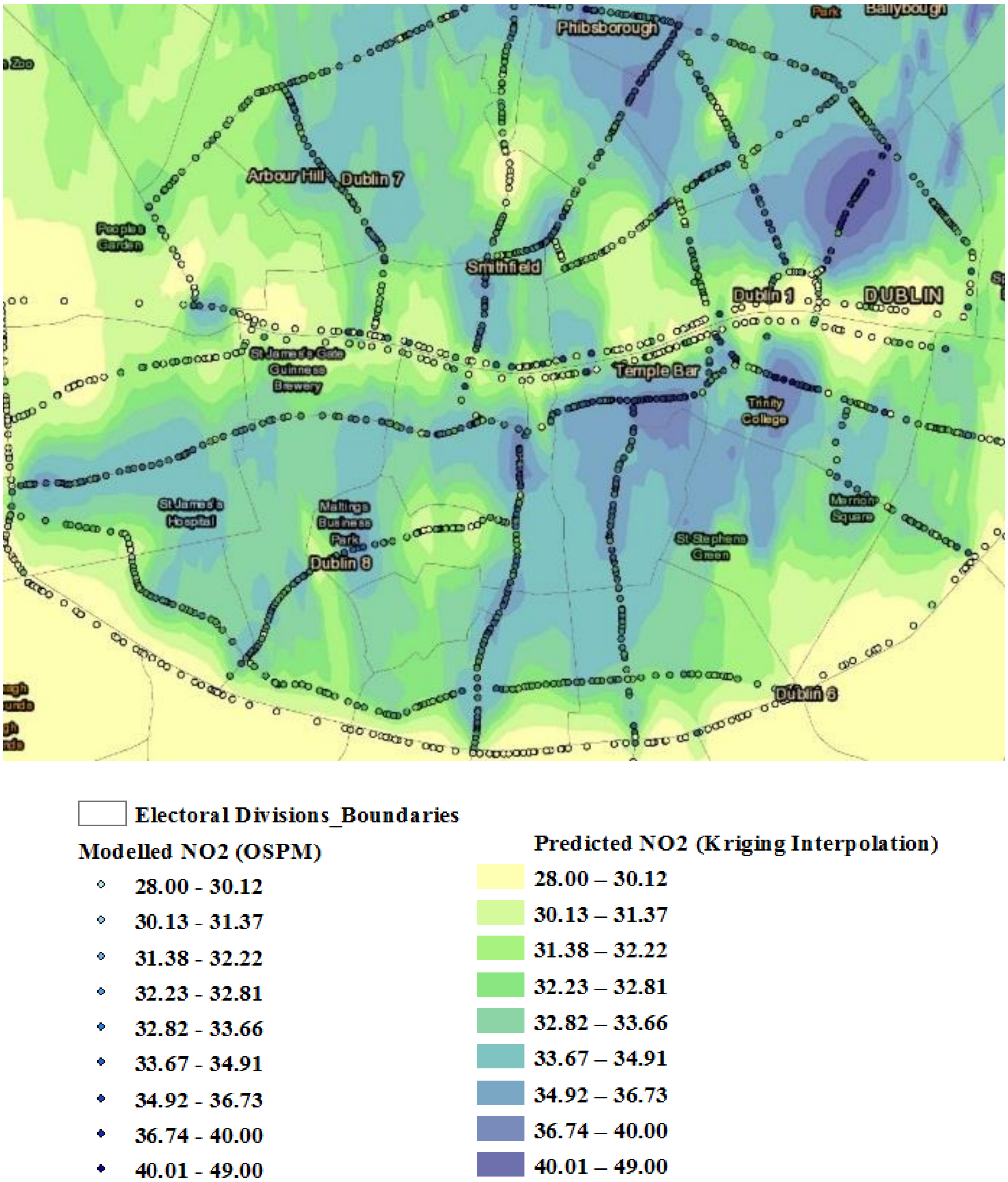
It can be observed that NOx concentrations are expected to rise above 59.85 µg/m^3 in many areas within the city. But it can possibly be reduced and brought down 53.53 µg/m^3 with the diesel vehicle ban and alternative taxation scenarios. There is no health guideline value for NOx in general, therefore the harmful level of concentration cannot be distinguished. But how this can be linked to increasing in NO_2 exposure, can be visualised in Figure 9.4. NO_2 concentrations in different parts over the study area are presented in Figure 9.4 (a), (b) and (c) for 2015, 2030BaU and 2030Polciy respectively.



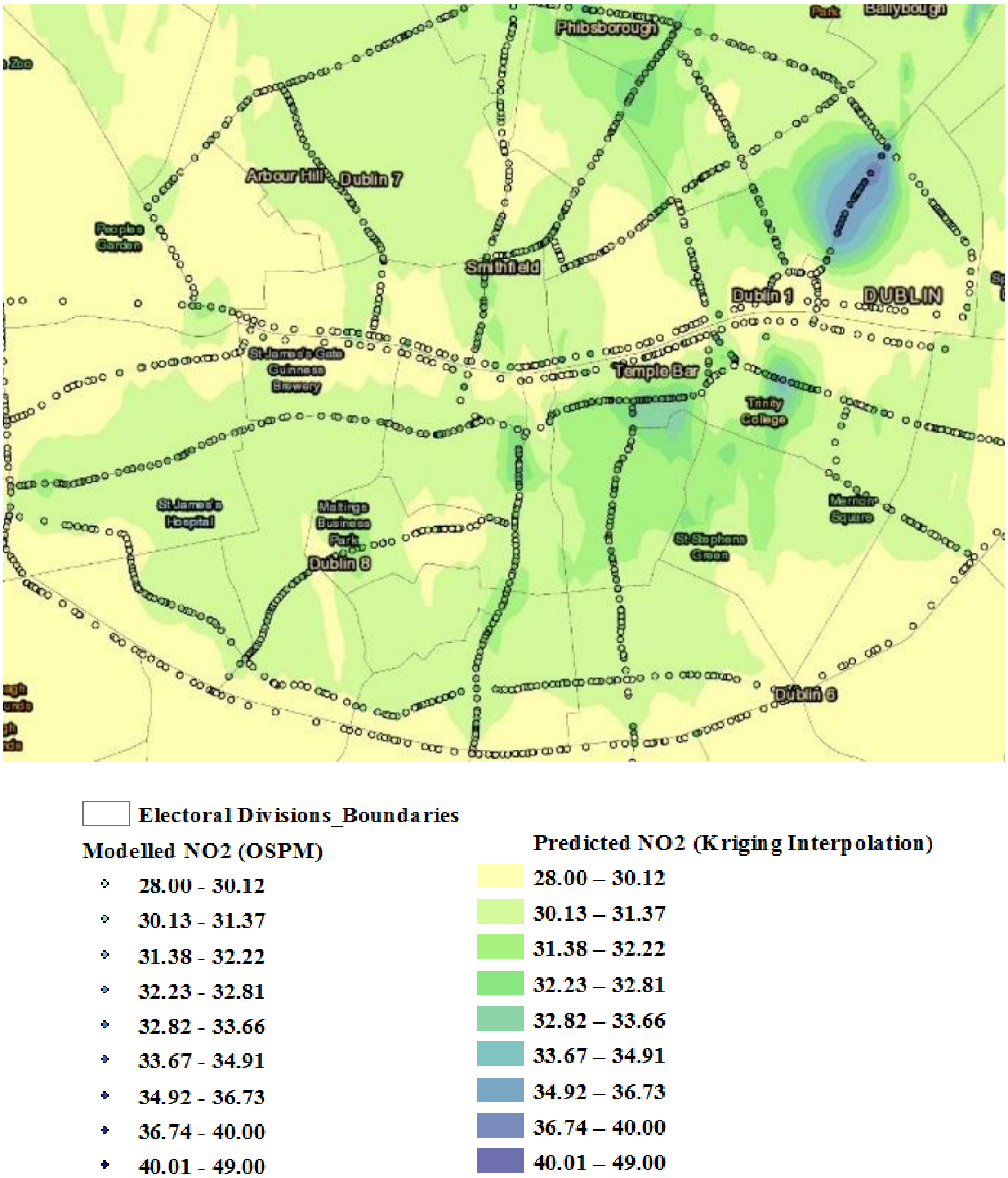
Electoral Divisions_Boundaries

Modelled NO2 (OSPM)	Predicted NO2 (Kriging Interpolation)
28.00 - 30.12	28.00 – 30.12
30.13 - 31.37	30.13 – 31.37
31.38 - 32.22	31.38 – 32.22
32.23 - 32.81	32.23 – 32.81
32.82 - 33.66	32.82 – 33.66
33.67 - 34.91	33.67 – 34.91
34.92 - 36.73	34.92 – 36.73
36.74 - 40.00	36.74 – 40.00
40.01 - 49.00	40.01 – 49.00

(a) 2015


Phibsborough
Ballybough
Arbour Hill
Dublin 7
Peoples Garden
Smithfield
Dublin 1
DUBLIN
St James's Gate Guinness Brewery
Temple Bar
Trinity College
St James's Hospital
Dublin 8
Merrion Square
St Stephens Green
Dublin 6
Electoral Divisions_Boundaries
Modelled NO2 (OSPM)
28.00 - 30.12
30.13 - 31.37
31.38 - 32.22
32.23 - 32.81
32.82 - 33.66
33.67 - 34.91
34.92 - 36.73
36.74 - 40.00
40.01 - 49.00
Predicted NO2 (Kriging Interpolation)
28.00 – 30.12
30.13 – 31.37
31.38 – 32.22
32.23 – 32.81
32.82 – 33.66
33.67 – 34.91
34.92 – 36.73
36.74 – 40.00
40.01 – 49.00

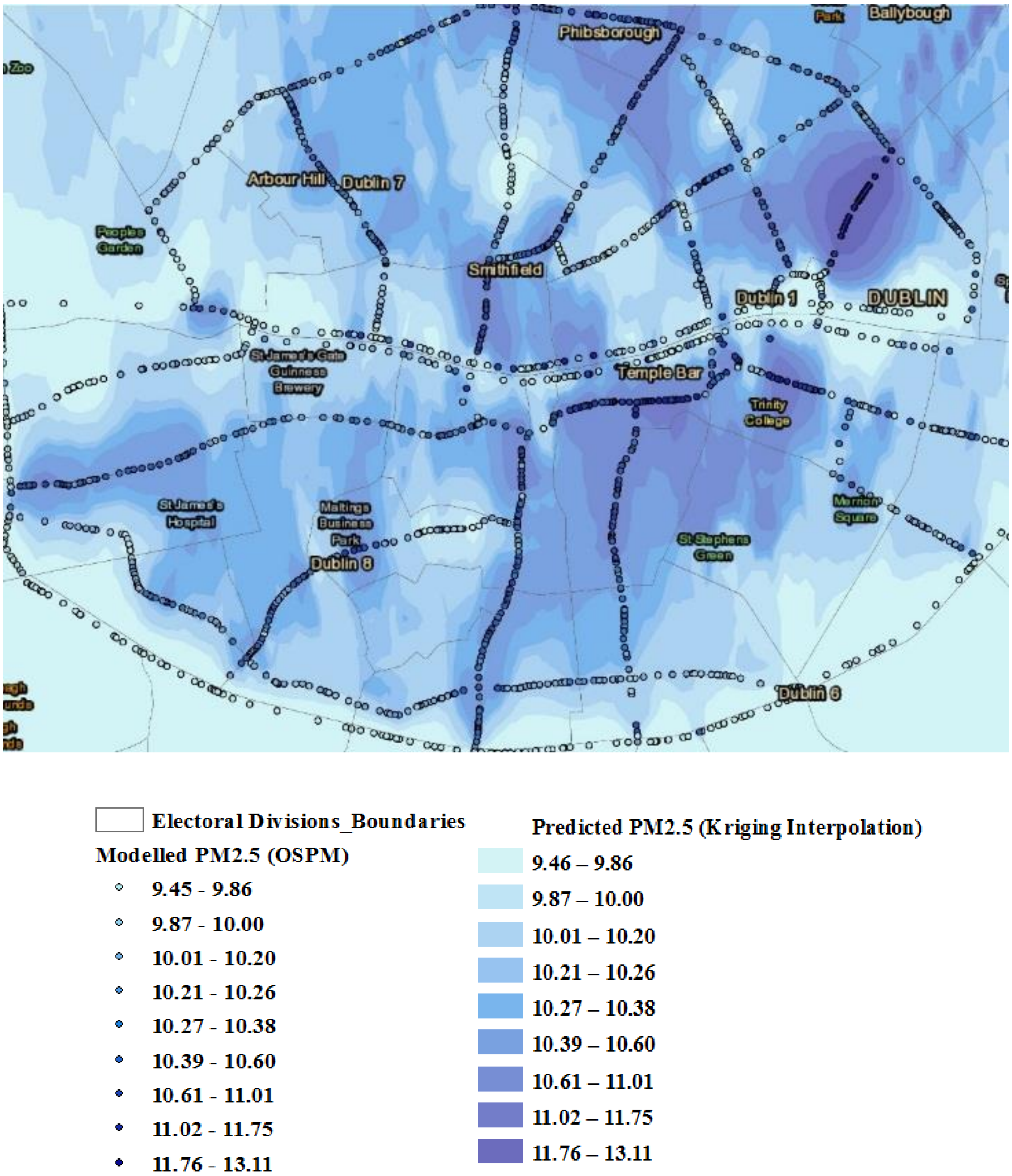
(b) 2030BaU



(c) 2030Policy

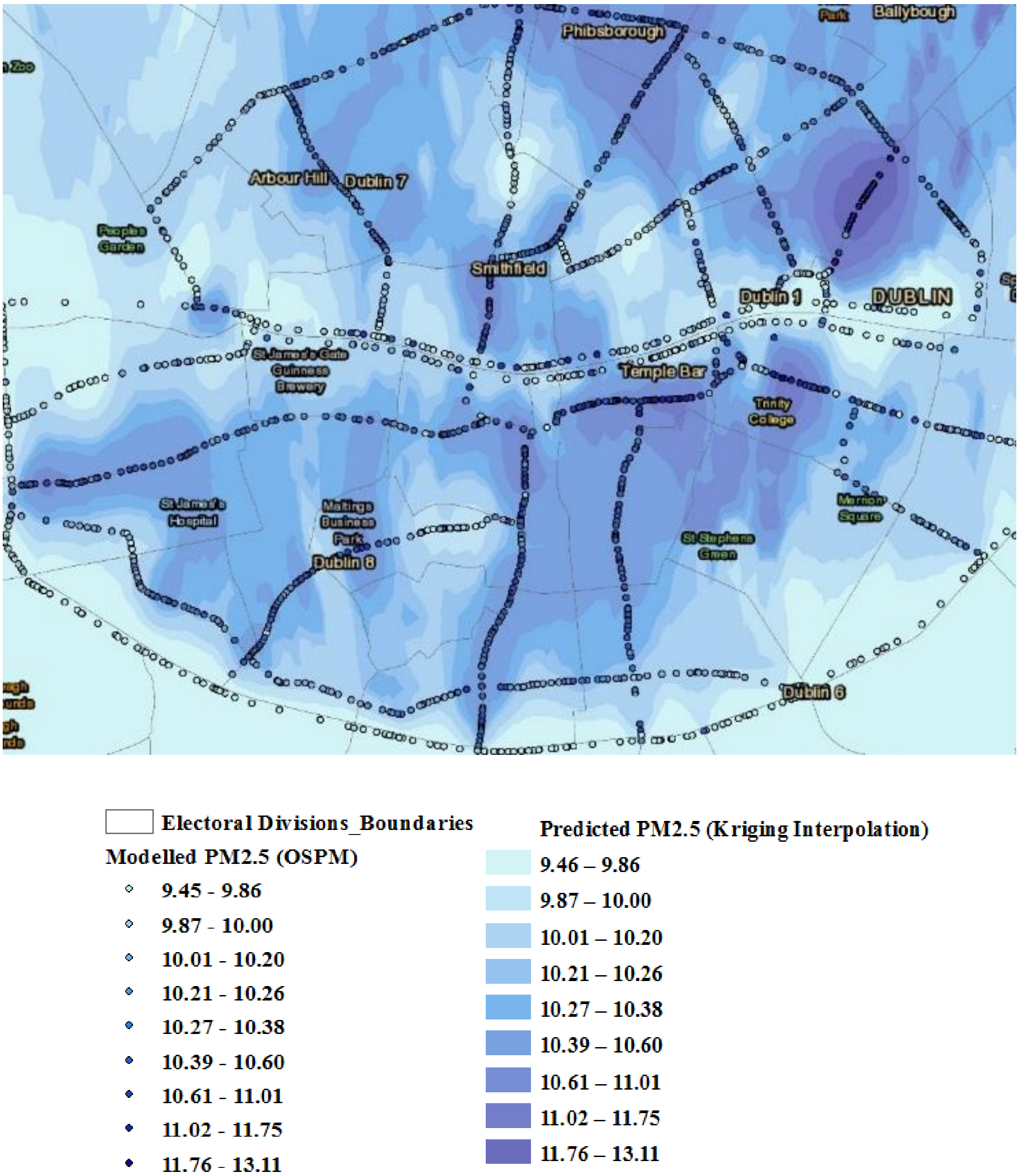
Figure 9.4. Modelled NO_2 concentrations and spatial variation in (a) 2015 (b) 2030BaU (c) 2030Policy

It can be observed from Figure 9.4 that, in 2015, NO_2 concentration levels are above the WHO/EU specified guideline value of 40 µg/m^3 in some parts. With business as usual situation i.e. with no additional policies than existing, there will be more areas facing unsafe levels of NO_2. However, with the policy interventions, the pollution level can be lowered to below the guideline limit in all the areas. Figure 9.5 (a), (b) and (c) presents the modelled $PM_{2.5}$ concentrations and predicted concentrations in the EDs for 2015, 2030BaU and 2030Policy scenarios, respectively.



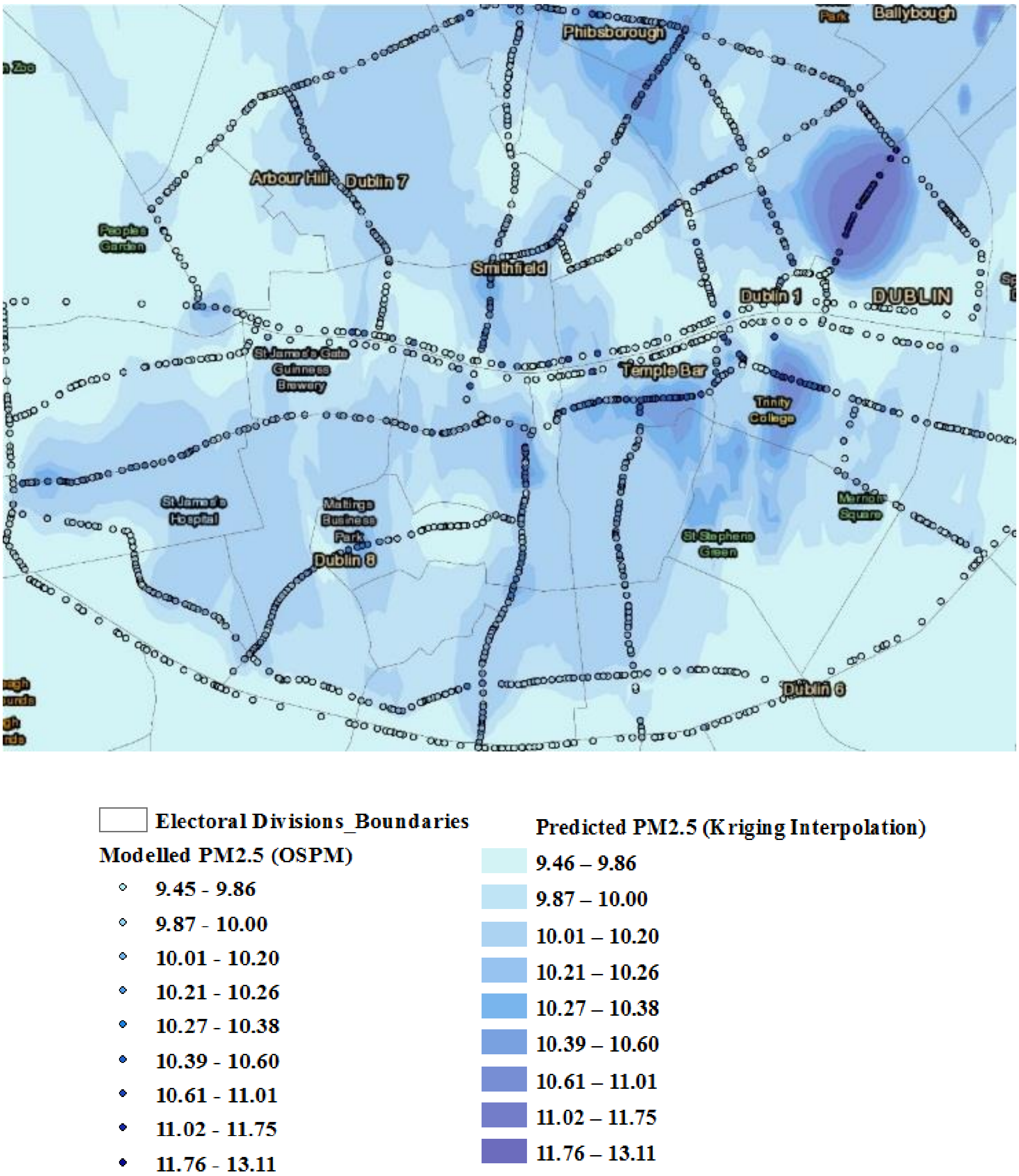
Phibsborough
Park
Ballybough
Zoo
Arbour Hill
Dublin 7
Peoples Garden
Smithfield
Dublin 1
DUBLIN
St James's Gate Guinness Brewery
Temple Bar
Trinity College
St James's Hospital
Maltings Business Park
Dublin 8
Merrion Square
St Stephens Green
Dublin 6
Electoral Divisions_Boundaries
Modelled PM2.5 (OSPM)
9.45 - 9.86
9.87 - 10.00
10.01 - 10.20
10.21 - 10.26
10.27 - 10.38
10.39 - 10.60
10.61 - 11.01
11.02 - 11.75
11.76 - 13.11
Predicted PM2.5 (Kriging Interpolation)
9.46 – 9.86
9.87 – 10.00
10.01 – 10.20
10.21 – 10.26
10.27 – 10.38
10.39 – 10.60
10.61 – 11.01
11.02 – 11.75
11.76 – 13.11

(a) 2015



Phibsborough
Park
Ballybough
Arbour Hill
Dublin 7
Peoples Garden
Smithfield
Dublin 1
DUBLIN
St James's Gate Guinness Brewery
Temple Bar
Trinity College
St James's Hospital
Maltings Business Park
Dublin 8
St Stephens Green
Merrion Square
Dublin 6
Electoral Divisions_Boundaries
Modelled PM2.5 (OSPM)
9.45 - 9.86
9.87 - 10.00
10.01 - 10.20
10.21 - 10.26
10.27 - 10.38
10.39 - 10.60
10.61 - 11.01
11.02 - 11.75
11.76 - 13.11
Predicted PM2.5 (Kriging Interpolation)
9.46 – 9.86
9.87 – 10.00
10.01 – 10.20
10.21 – 10.26
10.27 – 10.38
10.39 – 10.60
10.61 – 11.01
11.02 – 11.75
11.76 – 13.11

(b) 2030BaU



(c) 2030Policy

Figure 9.5. Modelled $PM_{2.5}$ concentrations and spatial variation in (a) 2015 (b) 2030BaU (c) 2030Policy

The WHO and EU guideline value for $PM_{2.5}$ is 10µg/m^3. The results reveal that with the expected road transport fleet composition in 2030, $PM_{2.5}$ concentrations will be higher than 10µg/m^3 in most of the EDs within the study area. With alternative fleet compositions as a result of new policy measures (2030Policy), the levels can be reduced. But a few areas with concentrations above the WHO specified safe limit will remain. The reason is, unlike NOx, $PM_{2.5}$ equally results from both petrol and diesel vehicles. For example, for diesel Euro 3, Euro 4 and Euro 5 passenger cars NOx euro standard emission levels are 0.5, 0.25 and 0.18 g/km respectively, whereas, for the same Euro emission standards for petrol passenger cars are 0.15, 0.08 and 0.06 g/km respectively. On the other hand, for $PM_{2.5}$, emission specification for both petrol and diesel Euro 5 passenger cars is 0.005 g/km. Under the alternative 2030 scenario, there still will be 51% petrol vehicles in the fleet resulting in the discharge of fine particulate matter. Therefore, the results are promising with new policy applications and $PM_{2.5}$ pollution are expected to reduce in all the area as those policies continue to be effective.

9.5. Burden of Disease

The health outcomes in terms of premature death and YLL due to NO_2 and $PM_{2.5}$ pollution associated with road traffic are described in this section. The total mortality values computed using BOD method from the modelled concentrations are listed in Table 9.1 for 2015, 2030BaU and 2030Policy scenarios. Table 9.1 also includes YLL that were calculated assuming that the average YLL per premature death incidence is 11 years (EEA, 2017). There are two air quality monitoring stations (at Winetavern street and at Coleraine street) within the study area (Figure 9.6), out of which only one station (Coleraine street) records $PM_{2.5}$ concentrations.

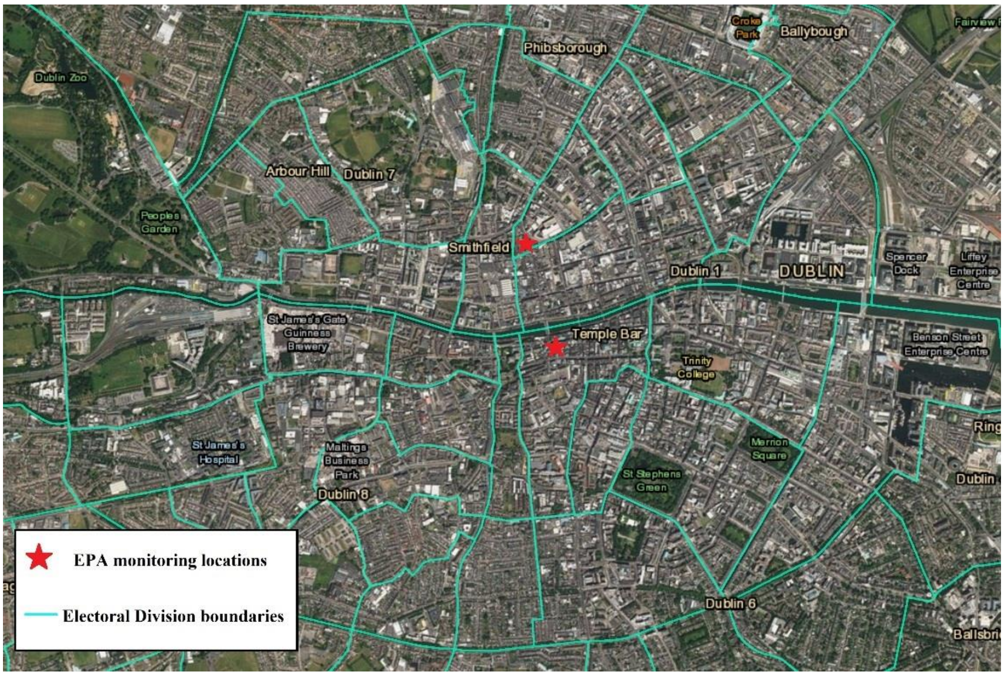


Figure 9.6. EPA air quality monitoring stations and Electoral Division boundaries within the study area

There could be possible over- or under-estimation in assessing the health burden when only monitoring station data is used for the whole area. In order to examine the fluctuation, total premature death incidences, YLL and VSL were calculated based on the captured data at the air quality monitoring station in 2015 (See Table 9.1). The annual average daily concentration of

$PM_{2.5}$ and NO_2 recorded at the monitoring station is 9.44 µg/m^3 and 28.25 µg/m^3 (31.1 µg/m^3 in Winetavern street and 25.4 µg/m^3 in Coleraine street) respectively. These values are below WHO and EU specified values but higher than the baseline concentration levels therefore possess health effects. VSL values, calculated by multiplying the number of deaths by VSL (3.75 million USD) per incidence (WHO Regional Office for Europe, 2015), are also listed in Table 9.1.

Table 9.1. The burden of disease due to long-term exposure to NO_2 and $PM_{2.5}$ pollution

Burden of Disease	2015			2030 BaU			2030 Policy		
	$PM_{2.5}$	NO_2		$PM_{2.5}$	NO_2		$PM_{2.5}$	NO_2	
	*C_0=2.5	C_0=10	C_0=20	C_0=2.5	C_0=10	C_0=20	C_0=2.5	C_0=10	C_0=20
	As per modelled concentrations								
Premature deaths	155	142	59	173	162	71	169	153	60
VSL (billion US$)	0.58	0.53	0.22	0.65	0.61	0.27	0.63	0.57	0.23
VSL (billion €)†	0.50	0.46	0.19	0.56	0.52	0.23	0.55	0.49	0.19
YLL	1705	1562	649	1903	1782	781	1859	1683	660
	As per monitoring station data								
Premature deaths	147	130	45	-	-	-	-	-	-
VSL (billion US$)	0.55	0.49	0.17	-	-	-	-	-	-
VSL (billion €)	0.47	0.42	0.15	-	-	-	-	-	-
YLL	1595	1430	495	-	-	-	-	-	-

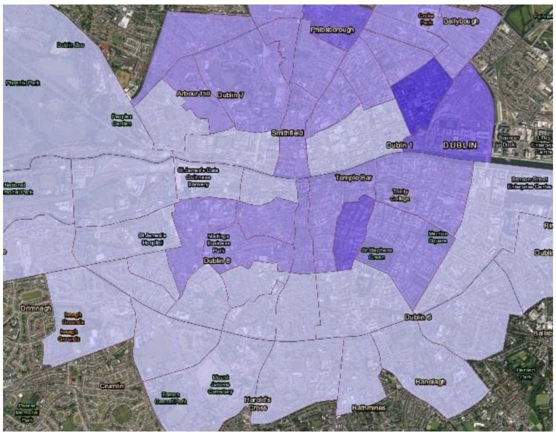
*C_0 units are in µg/m^3; †considering 1 US$ = 0.86€ (27.09.2018)

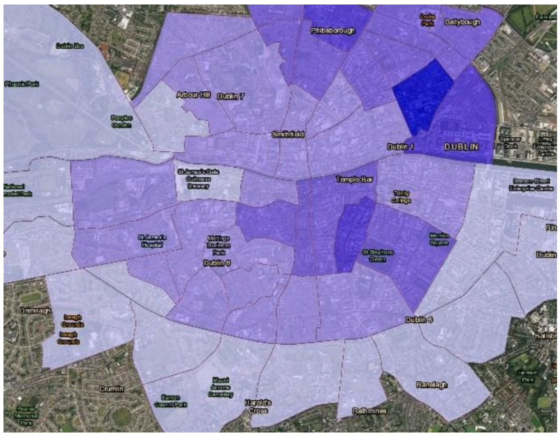
In the whole modelled area, total 297 premature deaths can be related to NO_2 and $PM_{2.5}$ pollution in 2015. This number is expected to rise to 335 in 2030 relative to 2015, resulting in 3,685 potential YLL. With the policy measures, in 2030, 13 premature deaths can be reduced per year. This is a significant reduction considering that the population is predicted to increase by approximately 10.4% in 2030 compared to 2015. The spatial distribution maps of the mortality incidences in the EDs in 2015, 2030BaU and 2030Policy scenarios are presented in Figure 9.7.

These deaths correspond to NO_2 and $PM_{2.5}$ exposure levels above 10 µg/m^3 and 2.5 µg/m^3 respectively.

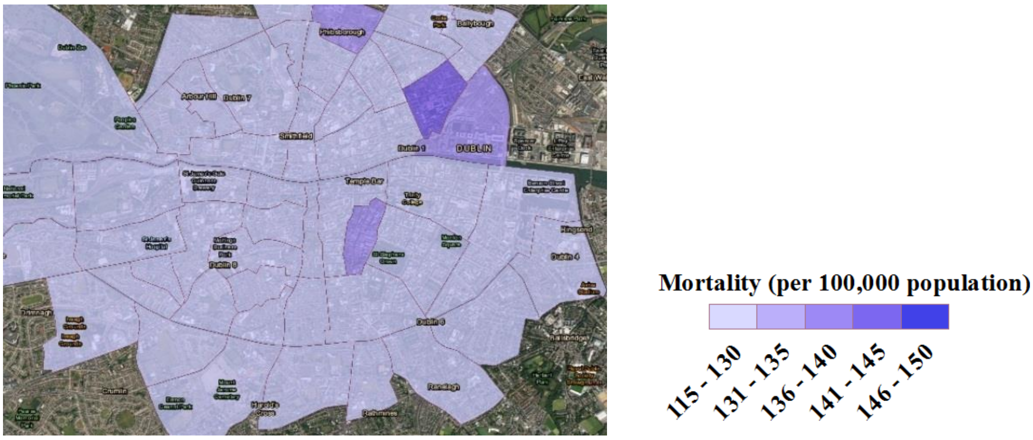
Figure 9.7. Total premature deaths due to NO_2 (C0=10µg/m^3) and $PM_{2.5}$ (C0=2.5µg/m^3) pollution in (a) 2015 (b) 2030BaU (c) 2030Policy

It can be seen from Figure 9.7 that the health impacts in EDs are improved with the policy implications in 2030. Death incidences in each ED considering the counterfactual concentration of $PM_{2.5}$ as 2.5 µg/m^3 and NO_2 as 20µg/m^3 are presented in Figure 9.8. In 2015, there are 15 EDs with $PM_{2.5}$ and NO_2 pollution related mortality more than 130 (per 1,000,000 people). The maps in Figure 9.8 show that in 2030BaU, 36 out of 49 EDs will possibly be having the number of deaths higher than 130 (per 1,000,000 people). Whereas, in the 2030Policy scenario, the mortality numbers can be reduced to below 130 (per 1,000,000 people) in all EDs expect 4. The resulting health consequences when converted to monetary term, it was found that a value of worth €0.96 billion can be linked to premature morality due to long-term NO_2 and $PM_{2.5}$ pollution exposure (when $PM_{2.5}$ C0=2.5 µg/m^3 and NO_2 C0=10 µg/m^3) in the study area in 2015. The overall impacts are underestimated by 20 deaths and approximately €0.07 billion when calculated based on the air quality monitoring station data.





(a) **(b)**



(c)

Figure 9.8. Total premature deaths due to NO_2 (C0=20µg/m^3) and $PM_{2.5}$ (C0=2.5µg/m^3) pollution in (a) 2015 (b) 2030BaU (c) 2030Policy

The estimates based on the monitoring station data not only underestimates the impacts but also does not represent a true picture of the areas having more impacts. More air quality monitoring and impact assessments should be carried out at a range of locations for better policy making and investments. The assessment of morbidity i.e. years lived with disability due to adverse health effects was not carried out in this study due to the lack of availability of required data. Efforts for developing the morbidity dataset are needed in order to improve the evaluation of the burden of disease as it imposes a significant health burden. However, it was observed in many studies that total external costs due to mortality outweighs those from morbidity (EEA, 2017). For instance, for $PM_{2.5}$, long-term exposure and mortality account for around 99% of the total, whereas morbidity is only 1% of damage costs (Walton et al., 2015).

9.6. Conclusion

This chapter evaluated the environmental and health impacts due to the high level of NOx emissions from road transport by modelling NO_2 concentrations in 155 streets in Dublin city using OSPM. Road transport is one of the major sources of $PM_{2.5}$ pollution, concentrations of $PM_{2.5}$ were modelled also on the same streets. Health impact in terms of premature deaths and YLL due to long-term exposure to NO_2 and $PM_{2.5}$ were estimated. Additionally, NO_2 and $PM_{2.5}$ concentrations were modelled on those street canyons for 2030 under BaU situation to evaluate and compare the modelled values with the WHO guideline values considered to be the safe limit. Also, the possible improvements in the pollution levels and subsequent health impacts with additional policy implications were assessed. This was achieved by modelling NO_2 and $PM_{2.5}$ for 2030 with alternative fleet compositions in presence of *dieselban* policies and alternative taxation scenarios.

The results show that the annual average daily NO_2 concentrations in 2015 are below the air quality guideline limit value in most of the areas, indicating that the air quality in Dublin is good. However, it was found that the concentration level will increase in 2030 exceeding 40 $\mu g/m^3$ safe limit in many areas and thereby, resulting in a rise in a number of premature deaths and other health consequences due to short-term and long-term exposure. For $PM_{2.5}$, although in 2015, the recorded $PM_{2.5}$ concentrations at the monitoring locations are below the guideline value of 10 $\mu g/m^3$, the concentrations exceed the value in approximately 68% of the modelled street segments. The average of all the modelled $PM_{2.5}$ concentrations in 2030BaU was found to be 10.24 $\mu g/m^3$ which is above the WHO/EU guideline value. Whereas, in the 2030Policy scenario it was found to be 9.98 $\mu g/m^3$. Therefore, air quality at human exposure level can be brought down below the safe limit with the additional policy measures in place.

Though the NO_2 concentrations are below the safe limit, it is higher than the counterfactual concentration and therefore, still possess some health threats. It was found that 142 premature deaths within the study area can be related to long-term exposure to NO_2 pollution from road transport, whereas, for $PM_{2.5}$ the number is 155. This together, lead to a potential life loss of 3267 years. This is expected to increase by 418 in 2030 with no additional measure taken than the existing ones. Also, VSL is expected to rise by about €0.12 billion in 2030BaU compared to 2015. This study sets an example of how controlling air pollution can be prioritised, compared to other policy interventions aimed at improving public health. Irish fleet has an increasing trend of diesel use and it was estimated that it is not until 2050 when there will be a notable reduction of ICEVs (DTTaS, 2015b). However, the results indicate that the policy measures described in Chapter 7 will significantly improve air quality and public health. This chapter also answers the question of whether the high level of NOx emissions will be equally harmful to human health, which was obscure in the case of Ireland. The findings of the chapter provide policy makers with actual information on how NO_2 and $PM_{2.5}$ emissions from road transport impact the air quality and health.

Chapter 10: Conclusion

This chapter concludes the thesis by summarising the work done, discussing the main contributions of this research, providing the limitations and suggesting some directions for future work. Some of the areas where more information is required for better policy making, better strategy making to reduce emission levels and to meet emissions target were identified based on the current trends in emission levels, increasing transport demand, expected levels in future emissions, Irish government's targets in relation to road transport and emissions, major policies taken up by other European countries to minimise the emission levels, current research papers, reports and findings on emissions and its impacts. This thesis aimed to provide information on those areas in the Irish context.

10.1. Summary of the work

In this thesis, present emission levels from the road transport fleet and expected emission levels from the future road transport fleets were modelled. Additionally, several scenarios were examined to assess how the emission levels can be mitigated with new policy measures and resulting possible changes in the fleet. Further, the environmental, health and cost impacts of the overall emissions were assessed for both the present and future years. The potential reduction in those impacts from reduced emission levels as a result of the additional policy measures were also examined. Considering a wide range of applications of emissions, potential uncertainty associated with emission estimates were characterised through a sensitivity analysis of input parameters. The findings of the analysis emphasized the importance of precise consideration of not only the activity data but also the meteorological data when using COPERT. Apart from the uncertainty associated with input parameters, there could be uncertainty related to the vehicle itself. While the total number of vehicles as per fuel and legislation standards can be obtained accurately, there could be uncertainty associated with emission factors, for example, as was found in *dieselgate* for diesel Euro 5 and Euro 6 diesel vehicles. This will have an effect on emission estimates and its applications. Therefore, the impacts of certain vehicles not obeying the emissions standards were quantified.

Looking at the expected increase in car ownership levels, emission levels of all the major air pollutants for the predicted future car fleets in Ireland and potential emissions reduction with increased market penetrations of electric cars were estimated. In addition, potential emission reductions from changes in public transport bus fleet were examined as it offers an opportunity to reduce emissions on a large scale. A set of policies were designed with a view to eliminating some of the causes linked to high emission levels such as (a) diesel fuel and VRT tax reliefs which led to huge increase in car ownership and diesel vehicle purchase, (b) more polluting older vehicles more than 20 years, and finally, (c) diesel which is causing high level of NOx pollution and health impacts. The policies were proposed such that it can be implemented in a realistic way. Then, NOx and $PM_{2.5}$ concentrations were modelled at the population exposure level at all the major roads in Dublin city for the base year and for 2030 under BaU and with policy scenarios. Based on the concentrations, premature mortality incidences attributable to long term exposure to NOx and $PM_{2.5}$ were calculated. Further, economic impacts due to premature mortality were calculated.

10.2. Main contributions

One of the notable contributions and uniqueness of this thesis is that it provides with a complete information on emissions and their impacts. There have been several studies carried out, as described in the literature review chapter, separately assessing the emissions and their potential environmental, health and cost impacts. But this research carried out an overall assessment by not only examining air pollution levels attributable to vehicular emissions and subsequent health effects, but also quantifying the health cost of air pollution. Other main contributions of this research are discussed as follows.

Although many researchers in the past studied the effect of variations of input parameters on emission levels estimated by different software tools including previous versions of COPERT. However, the present research is unique in terms of place of application, data used, and the approach followed to design the scenarios. Additionally, multifactor interaction and uncertainty modelling including all the scenario results add more strength to the findings. The data used in this study and the level of disaggregation is consistent with the data used in preparing national emissions inventory in Ireland. The emissions that are being reported and based on which several policies are being made could be an underestimation or overestimation of the actual levels. Therefore, the findings of this study provide important and very useful information to the question if the assumptions of averaging the input parameters required by COPERT can be neglected or more attention should be paid to eliminate the potential errors. The impact assessment of emissions from diesel cars and light commercial vehicles provide with a timely information on emissions which is again very necessary for vehicular emissions-based policy design and its successful application. Taking these into account will provide the policy makers with better emission estimates, thereby, more effective policy making, and implementation related to quantification of actual emission levels, emission projections, emission prevention and control measures, environmental impact assessment. On the basis of the results, it is also recommended that rather than reporting a single value of emissions, a range of emissions should be reported along with the full information on characteristics of uncertainty.

Though there have been studies carried out in Ireland to quantify road transport emissions in smaller or larger scales (Alam et al., 2015; Brady and O'Mahony, 2011), there has not been any study carried out yet examining the potential emissions reductions from the future fleets with alternative fleet compositions, in other words, examining what fleet Ireland will actually require in its traffic to reduce expected future emissions and meet emissions target. Emissions mitigation is one of the prime concerns in Europe now and, the targets set by EU are strict. Therefore, the research findings presented in terms of estimating future emission levels and examining alternative fleets to reduce emissions contain very useful information showing the consequences of both a realistic scenario and an optimistic scenario. Also, a damage cost analysis for all the scenarios have been provided which has not been quantified before but is an essential component of policy decisions. A feasible way of meeting 2030's emissions goal with reduced car ownership and mixed percentage of alternative options was proposed by backcasting Ireland's targeted emission level for 2030. From this research and findings, it can be said that without a reduction in car ownership levels in 2030 GHG emissions target as set by EU cannot be met. Therefore, policies should be made such that not only the alternative fuel uptake increases but also the usage of the sustainable mode of transport increases resulting in a shift from private car usage, especially for shorter distances.

The policies proposed and evaluated are a major strength of this thesis. The policies are designed looking at the very cause of the increase in emissions as mentioned in the previous section. The work also provides a timeline which can possibly help in implementing those policies in a realistic and effective way. Also, the reduction in overall emission levels and benefits in terms of revenue that can be drawn by those policy implications were estimated. It was found that as a consequence of these police measures, emission levels are expected to reduce significantly in 2030. However, CO_2 levels will not improve much due to the increase in car ownership and with the huge percentage of petrol vehicles already present in the fleet. Nonetheless, this will slowly get better as electric vehicle percentages will increase. Through the additional policies, the monetary damage due to the emissions could be saved. In addition to this, a sizable amount of revenue will be generated as a result of the withdrawal of fiscal incentives, which can be utilised in improving EV infrastructure to increase EV uptake and also towards the betterment of infrastructure for walking and cycling.

Assessing the environmental impacts of vehicular emissions through dispersion modelling is another key strength of this research. Although researchers in Ireland have previously aimed to model urban PM and NOx concentrations at street level, concentrations have not been modelled at a large scale as have been done in this research. In addition, concentrations were modelled for BaU and policy scenarios in 2030. This gives a whole visualisation of the current pollution level, the expected pollution level in future and the reduced pollution level that can be achieved. On comparing the modelled NOx emissions with NOx and NO_2 concentrations, no pattern was observed to strongly conclude that high NOx emissions will lead to high NO_2 concentrations at population exposure level. Upon examining the building geometry, speed and traffic volume, it was identified that building heights on either side of the roads play a major role in pollutant concentrations as it influences the dispersion of pollutants by trapping it for a longer duration. Also, based on an overall evaluation, it can be concluded that Ireland's overall air quality is good in terms of meeting air quality guidelines. However, in some areas, NO_2 and $PM_{2.5}$ pollution levels are very high indicating special attention needed for those areas. The concentration levels were expected to get worse in 2030 under BaU, however, with policy interventions, the situations can be significantly improved. Another important observation was that modelled concentrations at exposure level were found to be higher than the recorded pollutant concentrations in monitoring locations. This deviation may lead to inefficient policy making related to air quality management, air pollution control programme, development of policies to prevent and control emissions.

The work done in terms of assessing the health and economic burdens due to the environmental impacts completes the overall research by providing an overall assessment of emissions and its subsequent impacts on health and economy. This is also a major strength of this thesis. The presentation of the spatial distribution of the impacts adds more value to it as it points out the areas with higher impacts. The impacts are also estimated for 2030 under BaU and with new policy measures. A considerable number of premature deaths occurs annually with the current level of pollution which is expected to be more in 2030 due to an increased level of pollution. Consequently, this results in a huge financial loss which will increase with the increase in mortality in 2030. With the new policies, the health damages and relevant costs can be significantly improved. The findings on loss of lives and its associated monetary values indicate how government policies and investments should be designated in improving air pollution levels and public health.

10.3. Limitations of this research

The limitations or weaknesses of this research are discussed in this section. A comprehensive analysis was carried out to assess the uncertainty in emissions estimates with variation in vehicle activity and meteorological parameters used in COPERT 5. The effect of mileage degradation on emission estimates was taken into account by using country specific default mileage degradation factors in COPERT 5. Using Ireland specific values for taking into account mileage degradation due to age or total kilometres travelled may have an influence on emissions of CO, NOx and VOC. However, those data were not available for Ireland.

Another weakness of this work is regarding the determination of future fleet composition. The model used in this research to determine future fleet composition predicts that new diesel car share will continue to increase until 2030 and after that will start to decrease. But diesel cars have shown a drop in the purchase in 2017. However, this may not have much influence on the overall share of diesel cars in the fleet in 2030, also, the most updated version of the available model was used in this research.

The weakness of the results presented to show the spatial distribution of emission impacts is that it was done on the basis of vehicle densities in the counties. Though this gives an estimate about the possible impacts, the distribution based on kilometres travelled by the vehicles in each county will give more accurate estimates. However, this was not possible as county wise mileage data to the required disaggregated level is not available in Ireland at this moment. One limitation of this research could be that same death rate as 2015 was considered to calculate premature death incidences in 2030. Also, even though mortality due to long-term exposure to NOx and $PM_{2.5}$ exposure was assessed estimation of morbidity could not be included in this research due to the unavailability of appropriate data.

10.4. Future research

The research presented in this thesis offers a good scope for further work in relation to the uncertainty of emissions, emissions mitigation, dispersion of emissions, and health and cost effects of that. Hereunder, some recommendations that should be considered for future work are presented. The effect of mileage degradation can be examined in future to quantify the extent of variations in emission estimates as a result of mileage degradation depending on age or total kilometres travelled. The impacts of *dieselgate* on emission levels in Ireland were quantified based on the findings of Ntziachristos et al. (2016) who reported emissions factors obtained from extensive on road and lab test results. However, on road emissions data can be collected using PEMS and lab tests can be carried out on a variety of vehicle makes and models to gather more Ireland specific data.

Alternative low emission options for passenger cars, light duty vehicles and public transport bus fleets were examined and their potential in reducing emissions was evaluated. Future research covering heavy duty vehicles can be carried out to examine their potential in reducing emission levels from changes in the fleet. Also, while assessing the relationship between NOx emissions and NO_2 concentrations, the impact of wind speed and direction could not be taken into account due to the lack of street specific data. In future, information on wind profile on roads can be collected and the impacts of wind on concentrations can be investigated as the wind plays a significant role in recirculation and dispersion of the pollutants. This study focused on assessing urban air quality as the impacts of air pollution is more severe in urban areas. However, further studies can be done on examining air quality in rural areas and spatially distribute to identify the areas wi1th higher impacts. Morbidity constitutes a key part of health impact assessment as this gives an estimate of the number of years lived with disability resulted due to air pollution. Therefore, dedicated research should be carried out in developing the compatible dataset and morbidity attributable to NOx and $PM_{2.5}$ pollution from road transport should be calculated.

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Appendix A: European emission standards for vehicles

Table 11.1. Technology class vehicle active years (Alam et al., 2015)

Technology	**Passenger car**	**LDV**	**HDV**	**Buses/Coaches**	**Moped and Motorcycle**
Pre-ECE	Up to 1969	--			
ECE 15/00-01	1970-1978				
ECE 15/02	1979-1980				
ECE 15/03	1981-1985				
ECE 15/04	1986-1991				
Conventional vehicles*	--	Upto 1993	Upto 1994	Upto 1993	Upto 1999
Euro-I	1992-1996	1994-1997	1995-1997	1994-1996	2000-2003
Euro-II	1997-2001	1998-2001	1998-2001	1997-2001	2004-2006
Euro-III	2002-2005	2002-2005	2002-2005	2002-2006	2007-to date
Euro-IV	2006-2010	2006-2010	2006-2010	2007-2009	--
Euro-V	2011-2015, September	2011-2015, September	to date	2010-to date	--
Euro-VI	2015- to date	2015- to date			

*Before introduction of emission standards

Appendix B: Passenger Car Unit (PCU) factors

Table 11.2. Passenger Car Unit factors (Dublin City Council)

Vehicle type	PCU factor
Car	1.0
LGV	1.0
HGV 2X	2.0
HGV 3X	2.0
HGV 4x	2.0
HGV 5+X	2.0
Dublin bus	2.0
Other bus	2.0
Taxi	1.0
Motor cycle	0.5

Appendix C: Screenshots from COPERT Street Level software

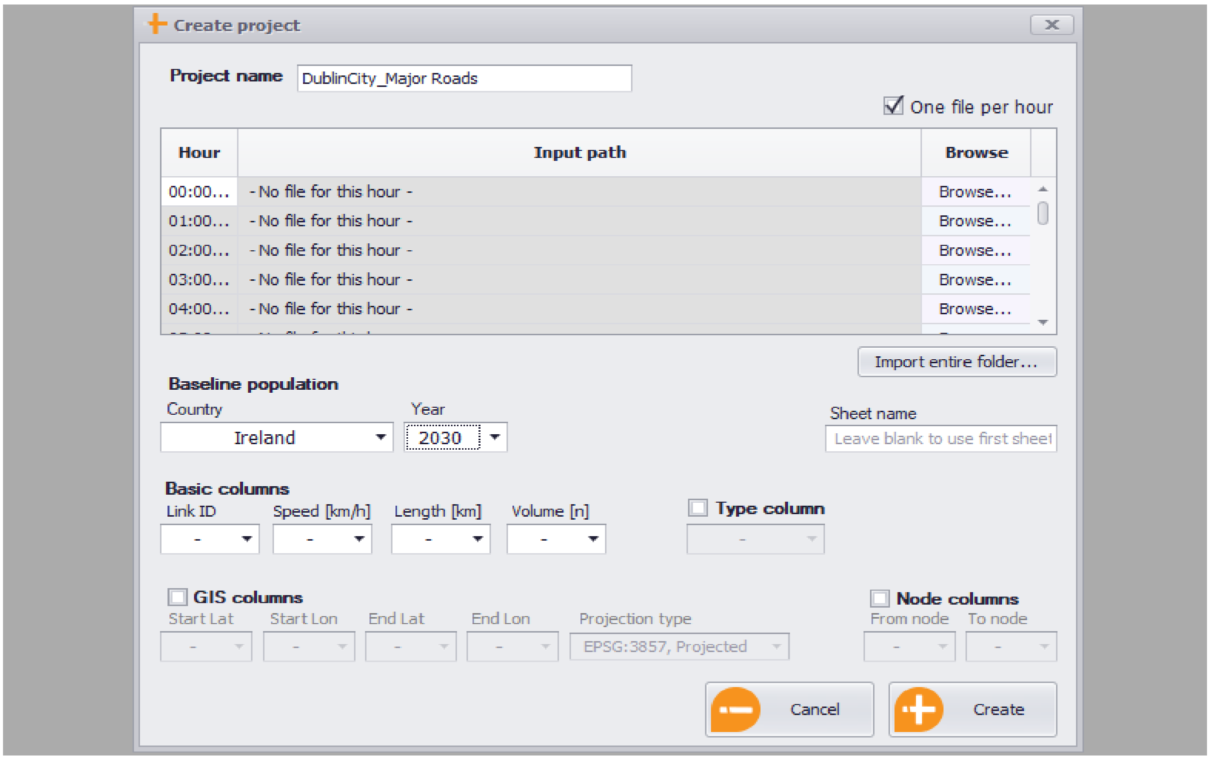


Figure 11.1. Creating a new project in COPERT Street Level

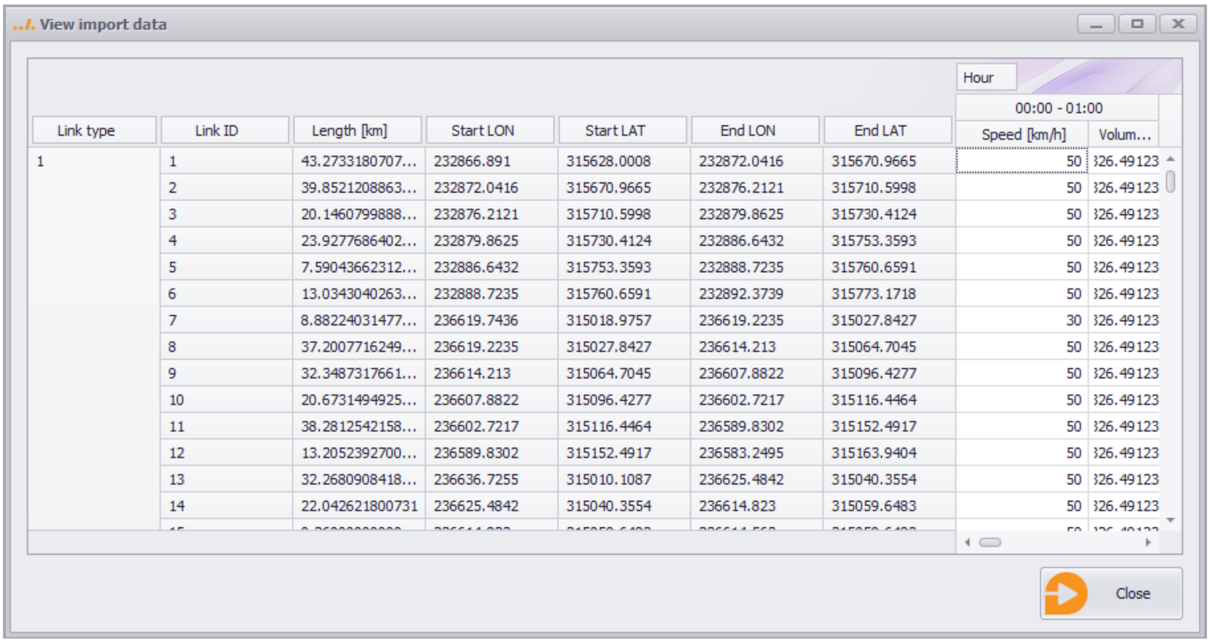


Figure 11.2. Imported hourly input data

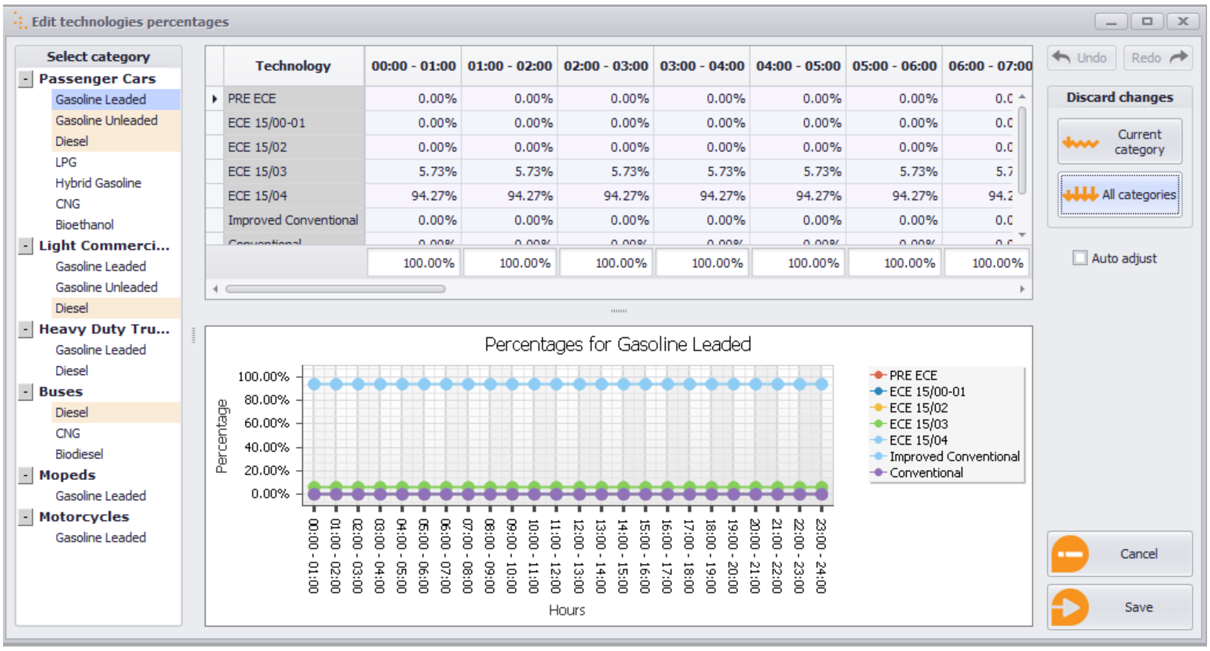


Figure 11.3. vehicle population percentage per engine technology

Appendix D: Screenshots from OSPM software and geometry of recirculation zone formation

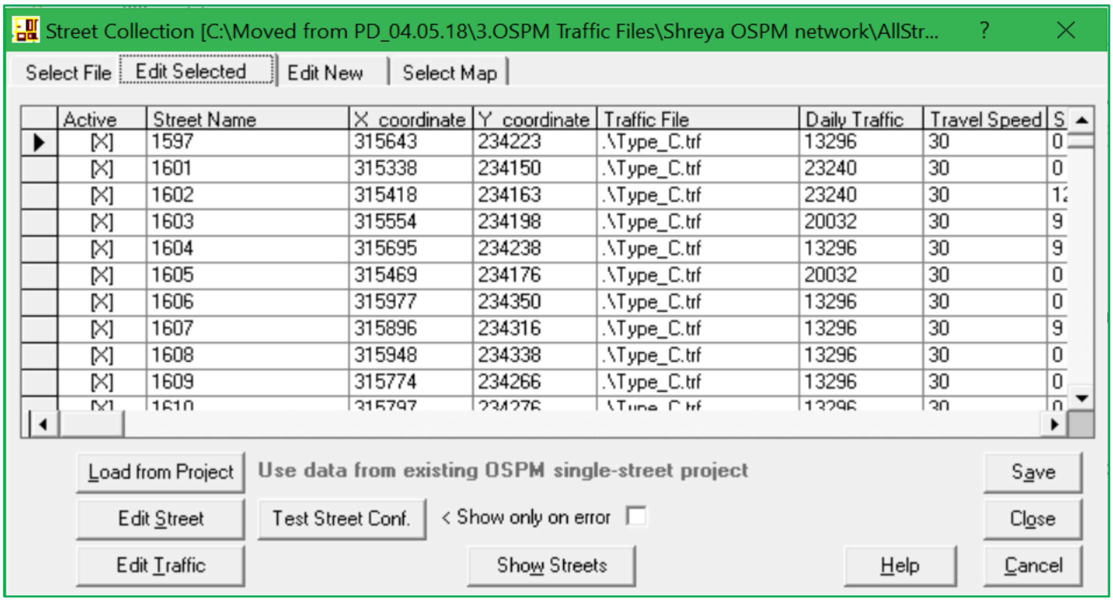


Figure 11.4. Screenshot of the window to define street configuration and traffic characteristics in OSPM

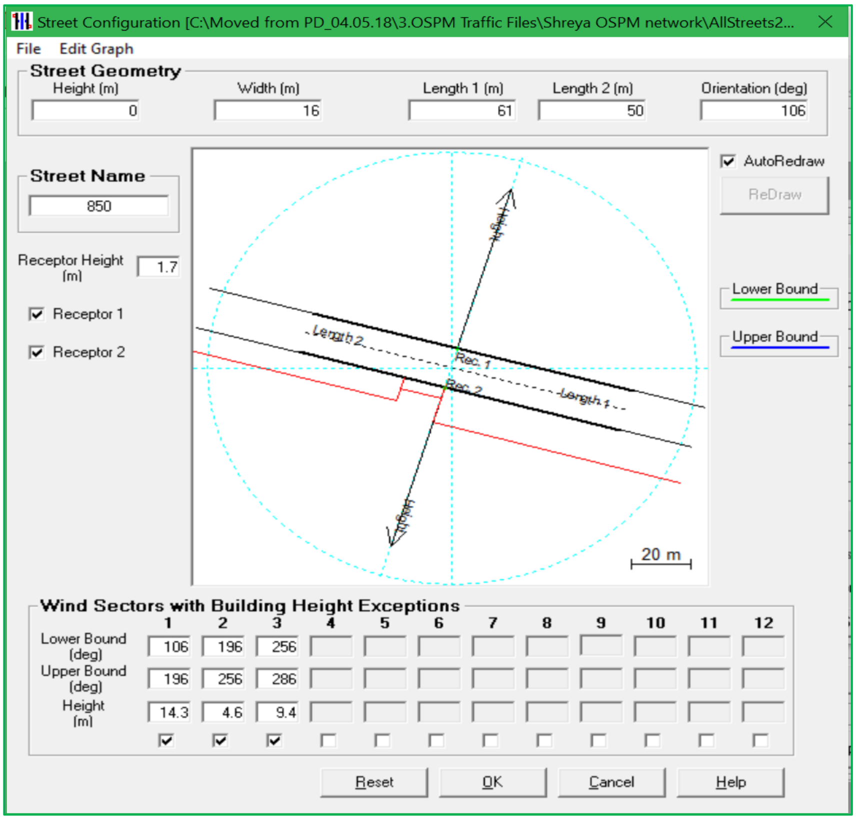


Figure 11.5. "Street configuration" window in OSPM

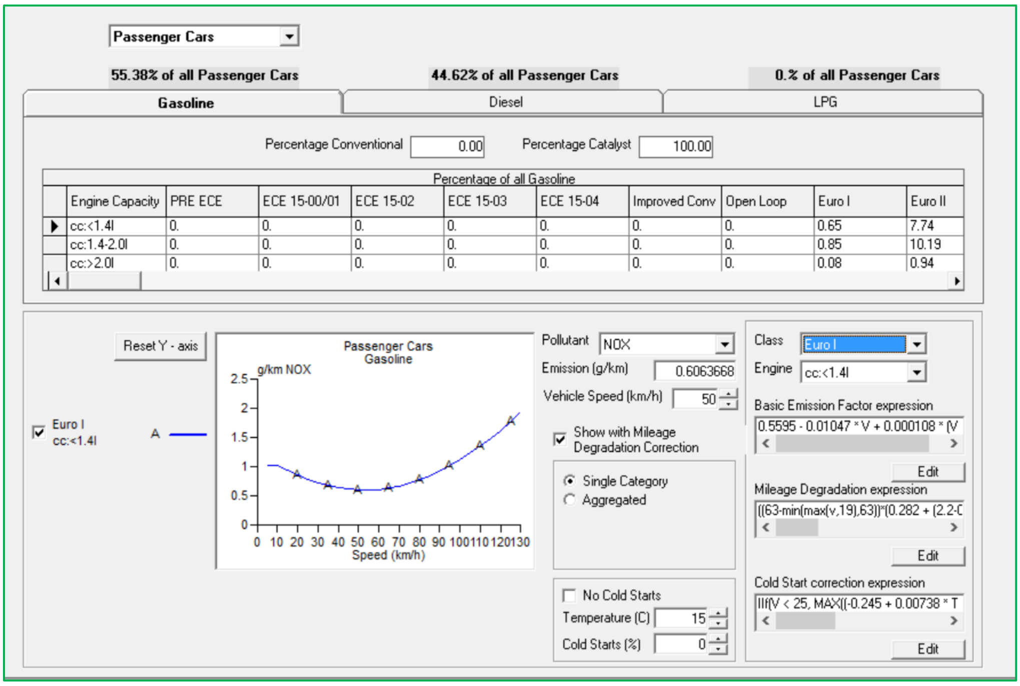
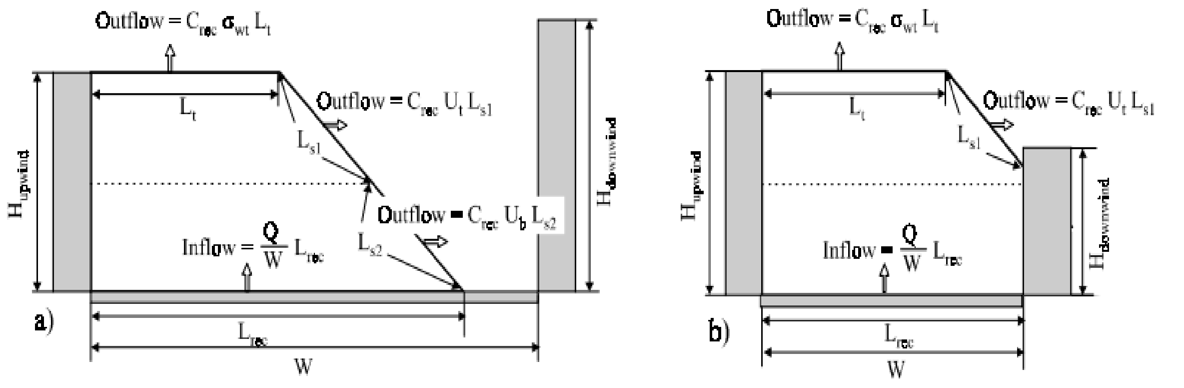


Figure 11.6. Traffic composition Ireland



(a) The recirculation zone totally inside the canyon

(b) The downwind building intercepts the recirculation zone

Figure 11.7. Geometry of the recirculation zone (Berkowicz et al., 1997)

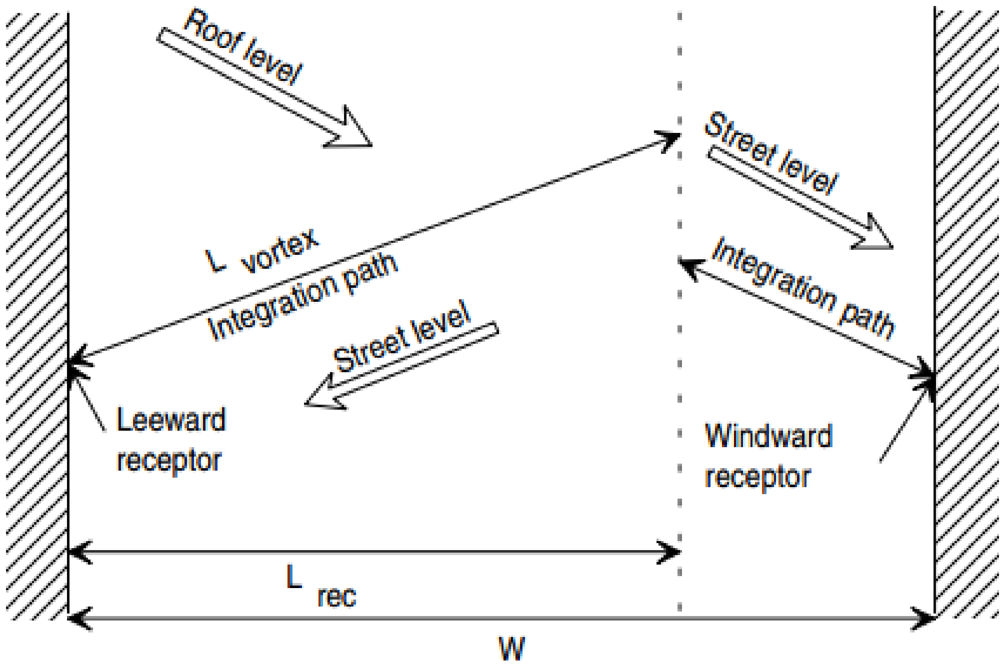


Figure 11.8. Illustration of the wind flow and formation of the recirculation zone in a street canyon (Berkowicz et al., 1997)

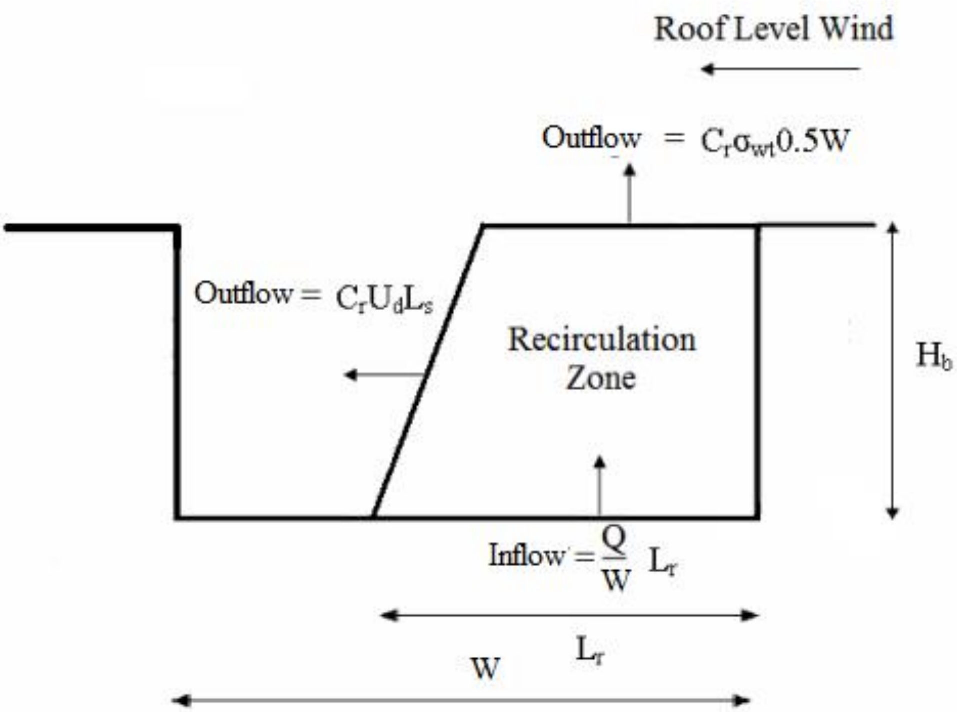


Figure 11.9. Assumed geometry of the recirculation zone (Berkowicz et al., 1997)

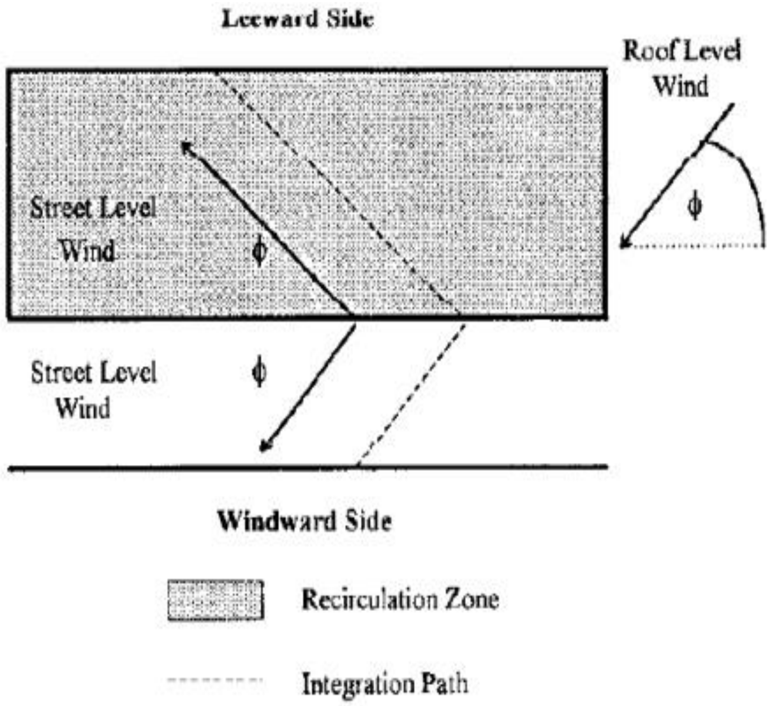


Figure 11.9. Wind flow and formation of recirculation zone in a street canyon (Berkowicz et al., 1997)

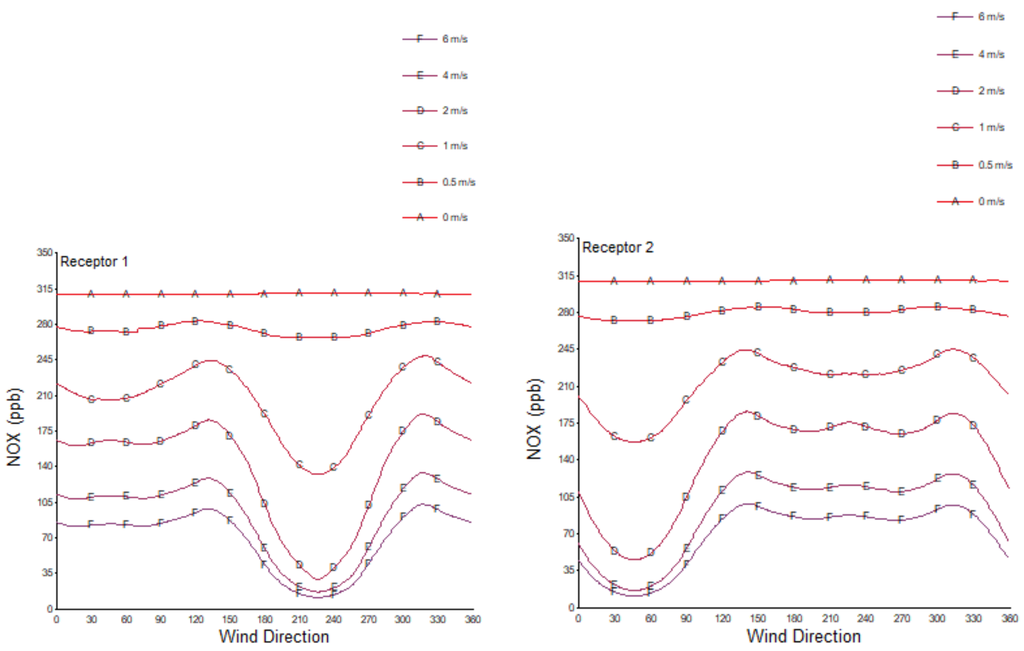


Figure 11.10. Effect of wind speed and wind direction on NOx concentration