An Evaluation of the Potential Impacts of Traffic Management Strategies on Air Pollution, Emissions, and Public Health in Dublin

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Declaration

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Jiayi Tang

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Abstract

Nowadays, air pollution is a huge threat to human health. It is responsible for 6.4 million premature deaths worldwide per year, which is 72% of the 9 million deaths per year from all types of pollution. Vulnerable groups, e.g. children and the elderly, have been noted to be more affected by air pollution. In urban areas, traffic is one of the major sources of air pollution.

Traffic management strategies have been proposed and introduced in many cities as an important part of urban management. Some of these strategies are designed in order to solve the problem of congestion, to shape a better land use, to meet the needs of a more strategic plan of a city or a nation, etc. While others are aimed at improving the environment. These traffic management strategies can potentially have significant impacts on air quality and human health. Therefore, it is important to try and predict the impact of different traffic management strategies on pollutant emissions, air quality, and thus on public health. The analysis of traffic management strategies in relation to their potential health impact can help the public and policy makers to be aware of the effectiveness of these strategies, and to attract public attention about the human health impacts of traffic management strategies that are not designed to protect the environment.

There are few examples in the literature that compare different traffic management strategies with regard to their possible health impacts. In this thesis, using the city of Dublin as an example, the impacts of different traffic management strategies on traffic, emissions, air quality and public health were evaluated and compared. On the other hand, as transport is an important source of GHGs (Greenhouse Gases) and thus a significant contributor to global warming, the impact of traffic management strategies on the emission of GHGs were also discussed.

Four typical traffic management strategies were included in this research: an infrastructure construction, a traffic management regulation, speed limit changes and fleet composition changes. In order to assess the impacts of these strategies, a traffic model, emission model, dispersion model and health impact model were developed and integrated into a modelling chain to evaluate the traffic, air quality and health outcome changes brought about by these strategies against a baseline scenario. To achieve this goal, the evaluation of these strategies was divided into three steps. Each step served the

same goal of evaluating the impacts of different traffic management strategies, but each had its own focus.

In the first step, as presented in Chapter 3, a traffic model was established in VISUM and an emission model was developed by using COPERT to assess the impact of an infrastructural change and a traffic regulation change in Dublin. These were the opening of the Dublin Port Tunnel and the implementation of a city centre 5-axel HGV ban. The impacts on traffic conditions, total vehicle travelling distances, travelling speed distributions, total emissions and standardized emissions adjusted by travelling demand distances, brought about by these two strategies were evaluated. It was found that the infrastructural change and the traffic management regulation change, though only in operation in the city centre, had impact on the traffic of whole city. These strategies reduced the traffic in the city centre and improved traffic speed distribution for whole city. However, these strategies resulted in HGVs travelling further with increased total emissions. This chapter emphasised that the impacts of traffic management strategies on a local scale and on a whole city scale were different and thus it's important to find the suitable focus area for a holistic assessment of a traffic management strategies.

In the second step, as demonstrated in Chapter 4, VISUM again was utilized to develop a traffic model, and OSPM was employed to develop an emission and dispersion model to evaluate the impact of the changes in traffic speed limits and fleet compositions on air quality on roads near a school in Dublin city centre. This was conducted to investigate the influence of these changes on vulnerable population groups. The results suggested that traffic speed limit reduction and fleet changes can influence the traffic-induced air pollutant concentrations significantly, such that these strategies had larger impacts on air quality in streets with higher traffic volumes and hence higher pollutant concentrations. This study highlighted that while speed limit reduction regulation near the roads of schools has been widely adopted, its impact on air quality and health of vulnerable people (i.e. pupils) should not be neglected as it can influence the air quality notably. Also, petrol and diesel vehicles had different and sometimes contrasting impacts on different pollutants, therefore the health impacts of incentivising either type of vehicle are complex.

The final step of the evaluation was outlined in Chapter 5. The traffic model, emission and dispersion model developed in previous chapters were utilized. A Gaussian plume model was also developed in the form of finite line sources, and a health impact model developed in BenMAP was added to the modelling chain to estimate the air quality changes and health impacts brought about by four types of traffic management strategies throughout Dublin. It was predicted that the infrastructure construction and the traffic management regulation, which were estimated to increase the pollutant emissions in Chapter 3, had little health impacts on the city as a whole since they had contrasting effects, positive and negative, in different parts of the city. Speed limits were predicted to have the same order of impact on mortality incidences as fatalities caused by traffic accidents in Dublin in 2013, which again emphasised the consideration of health impact of air pollution in the design of traffic speed limit, especially for vulnerable population sub-groups. The fleet composition change from diesel to petrol vehicles was predicted to reduce total mortality incidence significantly.

Finally, the impacts of these strategies were discussed in Chapter 6, regarding their impacts on public health through affecting emissions of different air pollutants and on climate change through influencing emissions of GHGs. A cross-comparison of the cost and benefits between them was conducted. Practical suggestions for traffic management strategy implementation were derived from these analyses and cross-comparison.

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Chapter 1 Introduction

1.1 Background

The most important outdoor air pollutants at an EU-wide level are NO_2 (nitrogen dioxide), PM (particulate matter) and O_3 (ozone), and traffic is a major contributor to these pollutants (EEA, 2013a). About 39% nitrogen oxides (NO_X) and 11% PM emissions are from the transport sector in Europe (EEA, 2018a). The contribution of the transport sector to ambient NO_2 and PM concentrations, especially in urban areas, is considerably higher, because its emissions are close to the ground and are distributed over densely populated areas (EEA, 2019). As road transportation is a major contributor to NO_2, VOCs (volatile organic compounds) and PM, it is also a major contributor to the formation of O_3. Thus, road transportation has a large impact on outdoor air quality.

Air pollution has adverse health impacts, and it caused 6.4 million premature deaths worldwide per year (Landrigan et al., 2017). Many studies have shown that air pollution is detrimental to the respiratory, cardiovascular, nervous, urinary and digestive systems (Cui et al., 2015; Bourdrel et al., 2017; Landrigan, P. J., 2017; Ribeiro et al., 2019). Cui et al (2015) summarized that the most severe health risks from normal exposures to air pollution are related to particulates, especially $PM_{2.5}$. Vulnerable people are particularly affected by air pollution. For instance, when children were exposed to even very low-levels of air pollutants, they are more likely to develop disease, disability, and even death in childhood and in later life as they are in the period of developing their body (WHO, 2005; Landrigan et al., 2017). Therefore, the reduction of emissions from the transport sector is important.

Policies have been introduced to reduce traffic emission worldwide. Governments all over the world are taking actions to reduce traffic emissions and to build sustainable urban transport systems (Pojani & Stead, 2015). Commonly considered options in cities for these purposes include: (1) improved road infrastructure; (2) incentives for road-based public transport; (3) operating restrictions and pricing; (4) support for non-motorized travel modes; (5) technological solutions; (6) awareness-raising campaigns; (7) speed management; (8) traffic flow control and (9) control of land-uses (Pojani & Stead, 2015; Bigazzi & Rouleau, 2017).

These policies have impacts on many aspects of the transport sector, e.g. economics, convenience for the public, and pollutant and greenhouse gas emissions. It is essential for a policy to be scrutinised by taking as many aspects of its impacts as possible into consideration, to give sufficient information for policy makers. Studies have often focused on the economic impact when evaluating policy. However, how a policy benefits, or even adversely effects, public health is usually difficult to estimate.

Policies that are introduced to reduce air pollution emissions are complex. They vary in many ways, having different impact level, effect range and cost. Different traffic management strategies have different impact on air pollution and the emission of GHGs. One strategy can have impacts on emissions of different air pollutants and GHGs to differing extents, and sometimes having contrasting effects on different emissions. For example, diesel engines were reported to be more efficient than petrol engines, and thus consume less fuels and emit less CO_2 which is the major contributor to global warming. But diesel vehicles emit more NO_X than petrol vehicles which is harmful to human health (Speed & Whitmarsh, 2015). This is because diesel engines have higher air excess coefficients and operate at higher temperature inside the combustion chamber than petrol engines, which offer a more effective condition for the formation of thermal NO_X (Sher, E., 1998). Thus there has been a lot of analyses and arguments about the choice of incentivising diesel or petrol vehicles by governments. On the other hand, some policies that reduced both air pollutant and GHGs emissions at one location may have opposite impact at another connected location. For example, the policy of setting low emission zones (LEZ), which restrict vehicles entering certain areas, may reduce the emissions within the LEZ, but also may force some vehicles to choose alternative longer routes and thus increase the traffic volume at other areas. The impact of LEZ attracted a lot of investigation and is still unclear (Holman et al., 2015). Therefore, the impact of these strategies on public health and on climate changes are complex.

Some traffic management strategies are specially designed for vulnerable people, e.g. pupils, the elderly and disabled people. As these strategies influence traffic emissions, they should be given special focus as these vulnerable people are also sensitive to air pollution. For example, the speed limit near schools are lowered at many places for safety considerations and traffic speed can affect traffic emissions remarkably. It raises the importance to investigate the impact of these strategies on air pollution.

Because of the differences between traffic management strategies, it is difficult to make a cross-comparison regarding public health impact between air pollution control policies. How to evaluate policies using a single standard in terms of impact on health remains a problem. Therefore, knowing how to quantify health effects provides information for policy makers that enables them to have a more comprehensive consideration when making policy decisions.

To solve this problem, this thesis is intended to develop a modelling chain that involves traffic, emission, dispersion and health impact models to assess the potential impacts of different traffic management strategies on the environment and public health, and to discuss the nexus of the impact on public health, climate changes and ecosystem of these strategies.

Focusing on Dublin, Ireland, several traffic management strategies of different types have influences on traffic emission. Dublin city opened the Dublin Port Tunnel (DPT) in December 2006 as a designated route for Heavy Goods Vehicles (HGVs) to reduce the volume of HGVs travelling through the city centre. An HGV management strategy was also introduced in February 2007 to limit the volume of HGVs in city centre. These were conducted in order to improve congestion and the environment in the city centre.

There are also different speed limits at different areas in Dublin that affect traffic emission, especially for roads near schools, and also different fleet compositions influence the air quality. This research used these case study examples of policy implementation to estimate the environmental and public health impacts brought about by changes in infrastructure, traffic regulation, speed limits and fleet compositions.

1.2 Research objectives

The overall aim of this research was to conduct a holistic assessment of the impacts of different traffic management strategies and perform a cross-comparison between them.

In order to achieve this aim, the primary objectives listed below were established:

- Identifying a suitable traffic model, emission model, dispersion model and health impact model. Identifying suitable evaluation metric that for the scope and objectives focused by the study;

- Developing a baseline traffic model in line with the traffic conditions for the Dublin area. Calibrating and validating the model with the traffic counts data in Dublin;
- Integrating the emission model, dispersion model and health impact model with the traffic model into a modelling chain;
- Applying the scenarios that represent traffic management changes on the modelling chain to predict the impact of different types of strategies on traffic, air quality and public health;
- Conducting a holistic cross-comparison between different strategies, in terms of their costs, their impacts of changes in different air pollutants on public health and in changes in GHGs on climate and ecosystems.

1.3 Contribution to knowledge

The following gaps have been identified in the literature (see Chapter 2):

1. Lack of health impact assessments for traffic management strategies.

There has been little measurement of exposure or health impacts associated with the implementation of differing traffic management strategies. Current literature commonly estimates the traffic emission or air quality changes brought about by some traffic strategies, but limited studies take the population density and resulting health impact into consideration. Conversely, the broader epidemiology and environmental health literature assesses health effects of exposure to traffic-related air pollution (Health Effects Institute, 2010, Shah et al., 2012, Chen et al., 2017a), but rarely the effects of specific transportation interventions.

2. Lack of the direct comparisons among different traffic management strategies in terms of their costs, and impacts on traffic, emission, air quality and public health in a homogeneous condition needs to be addressed.

Assessing the impact of traffic management strategies alone excluding potential confounding elements that influence air quality are remaining challenges. In order to exclude potential confounding elements, the impacts of traffic management strategies are needed to be evaluated under the same base conditions for traffic, meteorology, etc. Other literature has compared the impact of different traffic management strategies implemented in different places. Some of them were found effective, others were not, but there is a lack of a direct comparison between them.

3. Lack of holistic analysis of traffic strategies in terms of the impact of different air pollutants on health and GHGs on climate change.

In accordance with the identified gaps in the literature, the following main contributions to knowledge were achieved in this research:

- A modelling chain was developed to assess the impact of different traffic management strategies in the way of enabling the hypothetical scenario of presence or absence of specific strategies to compare with a baseline scenario. This enabled us to see the impact on traffic, air quality and human health of individual traffic management policy changes without the influence of other elements.
- The evaluation of the health impact of these strategies was made with the same metric, i.e. mortality incidence changes, which enabled a cross-comparison among these strategies.
- The research found that different impacts, and sometimes even opposite impacts, of traffic management strategies on different air pollutants and GHGs, which disclosed the complex nature of traffic management strategies and highlighted connections and contradictions between protecting public health and combating climate change.
- The research found that local traffic management regulation and infrastructural construction could have impacts on a city-wide and local scales that were contradictory. Analysis of these strategies stressed the importance for a policy analysis of the consideration of all possible places that may be affected by the policy.
- The research found negative impacts of reducing speed limit on air quality near a primary school and emphasised that attention should be paid to negative impacts of traffic management strategies on air quality and potential health consequences that are closely related to vulnerable population.

- The research found that taking the population into account, the health impact result can be negligible compared to the results traffic emission changes, which suggest the fundamental difference between tackling the issues of the emission amount of GHGs and the health impact of air pollution.
- The research provided practical suggestions for traffic management strategy planners.
- The method is easy to extend to other strategies, allowing more comprehensive analysis and comparisons among different types of traffic management strategies when a lot of strategies are in the decision list and the comparison of their advantages and disadvantages for their effectiveness is required.

1.4 Thesis outline

This thesis consists of seven chapters. An introduction to the problem is presented in chapter 1 and the relevant background literature is discussed in chapter 2 in general. The results of the studies conducted are presented in chapters 3-5 in the form of papers, and each also contains elements of an introduction to a specific problem as well as relevant background literature.

In chapter 3, the impacts of an infrastructural change and a traffic regulation change on traffic and emission were evaluated by employing a traffic model and an emission model. In the context of this study, these changes were the opening of the Dublin Port Tunnel (DPT) and the implementation of the HGV management strategy in Dublin. This chapter has been published entirely in the journal "*Transportation Research Part D: Transport and Environment*", Tang, J., McNabola, A., Misstear, B., & Caulfield, B. (2017). The traffic conditions and emissions in 2006 and 2007 were modelled in line with the time of the opening of the DPT and the implementation of HGV management to investigate the impact of these strategies. The year of 2013 was also modelled in order to see the impacts of these strategies under different travel demand and vehicle fleet conditions. Moreover, the impacts of the change in travel demand and fleet composition can be inferred from this study.

Chapter 4 presents an assessment of the impacts of speed limit changes and fleet composition changes on the traffic emission and air quality around a school in the city centre of Dublin. A traffic model, and an emission and dispersion model were developed and utilized to evaluate these impacts. This chapter has been published in the journal "*International Journal of Environmental Research and Public Health*", Tang, J., McNabola, A., Misstear, B., Pilla, F. & Alam, M.S. (2019). This chapter gave a school in the Dublin city centre a special focus as pupils are particularly vulnerable to air pollution.

Chapter 5 evaluates the health impact associated with the air quality changes brought about by the opening of the DPT, the implementation of the HGV management strategy, speed limit changes and fleet composition changes in Dublin. The health impact model that was developed for this chapter, along with an integration of the data of traffic, weather, background concentrations, etc. and the traffic model, emission and dispersion model that were developed for Chapter 3 and Chapter 4 were employed to perform the health impact evaluation. In this chapter, all scenarios were implemented based on the traffic, weather, etc., condition in the Dublin city in 2013 in order firstly to estimate the impacts of these traffic management strategies on whole city instead of only a part of the city which may cause incomplete assessment and secondly to exclude the effect of other elements to estimate their impact alone.

The impacts of different traffic management strategies are discussed in Chapter 6. The impacts of different strategies on traffic emission, air quality and public health are compared to each other, and the potential impacts on climate and ecosystem are also discussed to enable a comprehensive assessment of the impacts of these strategies.

The overall conclusions of the research and recommendations for future work are included in Chapter 7, the final chapter of the thesis.

Chapter 2 Literature review

2.1 Air quality status in Europe

Compared to worldwide, air quality in Europe is relatively less polluted. The European Environmental Agency (EEA) reported that the population-weighted annual average concentrations in 41 European countries for $PM_{2.5}$ and NO_2, which are two main air pollutants that affect public health, are 14.1 $\mu g/m^3$ and 18.8 $\mu g/m^3$, respectively. These figures are below the EU standards, being 25 $\mu g/m^3$ and 40 $\mu g/m^3$ for $PM_{2.5}$ and NO_2, respectively. In spite of these, air pollution is still the single largest environmental health risk in Europe, causing around 400 000 premature deaths per year (EEA, 2018b). This is because the pollutant concentrations in Europe are beyond the counterfactual concentrations, above which the pollutant concentration is considered detrimental to human health and thus the health impacts are estimated. Although $PM_{2.5}$ has been found harmful to human health with a whole range of concentrations, the EEA has taken 2.5 $\mu g/m^3$ as a counterfactual concentration for $PM_{2.5}$, since it is the lowest annual mean concentration measured in Europe and could be considered as the natural concentration that would occur if all anthropogenic contributions were removed. The counterfactual concentration of 10 $\mu g/m^3$ has been taken for NO_2 by EEA, since it is the minimum concentration for which effects have been found (EEA, 2018c). Figure 2-1 shows the population of EU-28 countries that lives above counterfactual concentration, WHO air quality guide concentration and EU limit concentration for $PM_{2.5}$ and NO_2. For $PM_{2.5}$, The population exposure exceeding the EU limit value was about 6% of the total population, whereas 81% population exceeds the WHO air quality guide. For NO_2, about 3% population lived in areas with annual average concentrations above the EU limit value and WHO air quality guide.

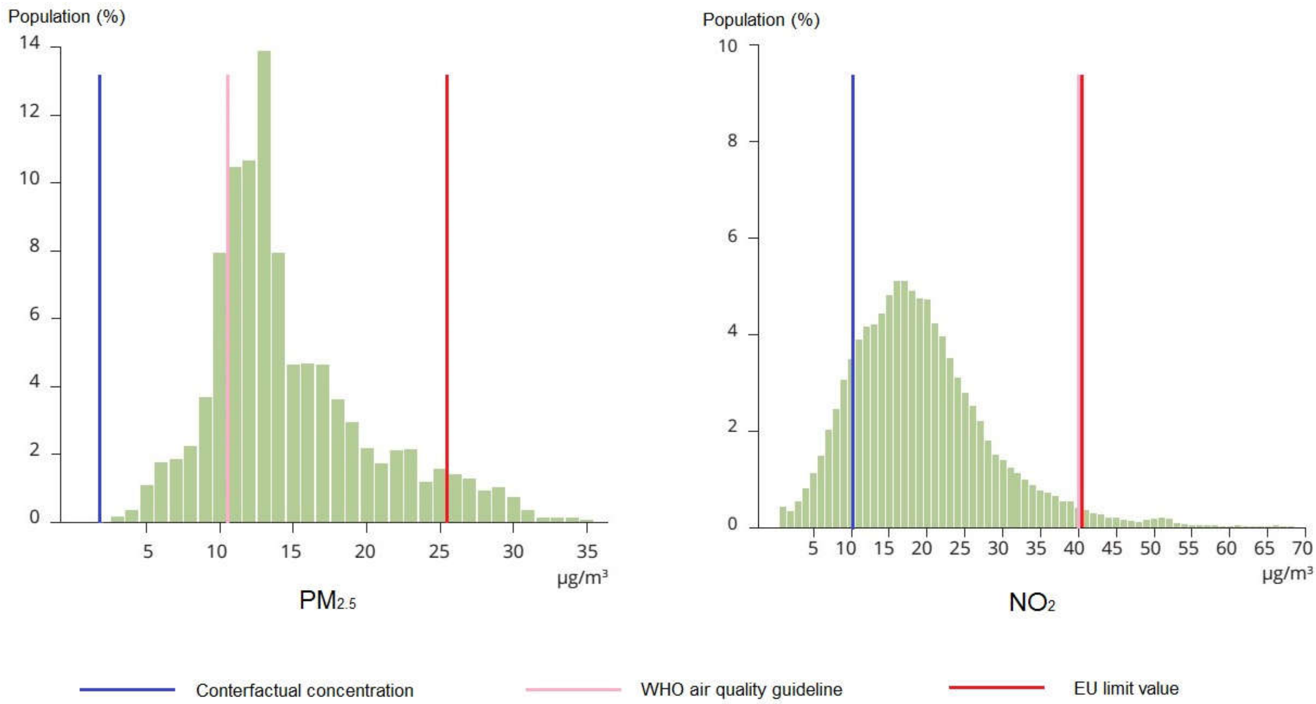

Figure 2-1: Frequency distribution of the total population exposure to $PM_{2.5}$ and NO_2 (annual mean) in EU-28 in 2015 (adapted from EEA, 2018b).

In Europe, the largest contributor for NO_X pollution is the road transport sector, accounting for 39% NO_X emissions. The largest contributor for PM and CO is commercial, institutional and households' combustion, accounting for 39% PM_{10} emission, 56% $PM_{2.5}$ emission and 48% CO emission. Transport sector is the second contributor for $PM_{2.5}$ and CO emissions, and the third contributor for PM_{10} emission. 11% $PM_{2.5}$ emission, 20% CO emission and 10% PM_{10} emission could be attributed to transport sector. Figure 2-2 shows the contribution of different main source sectors to pollutant emission in 28 countries in EU.

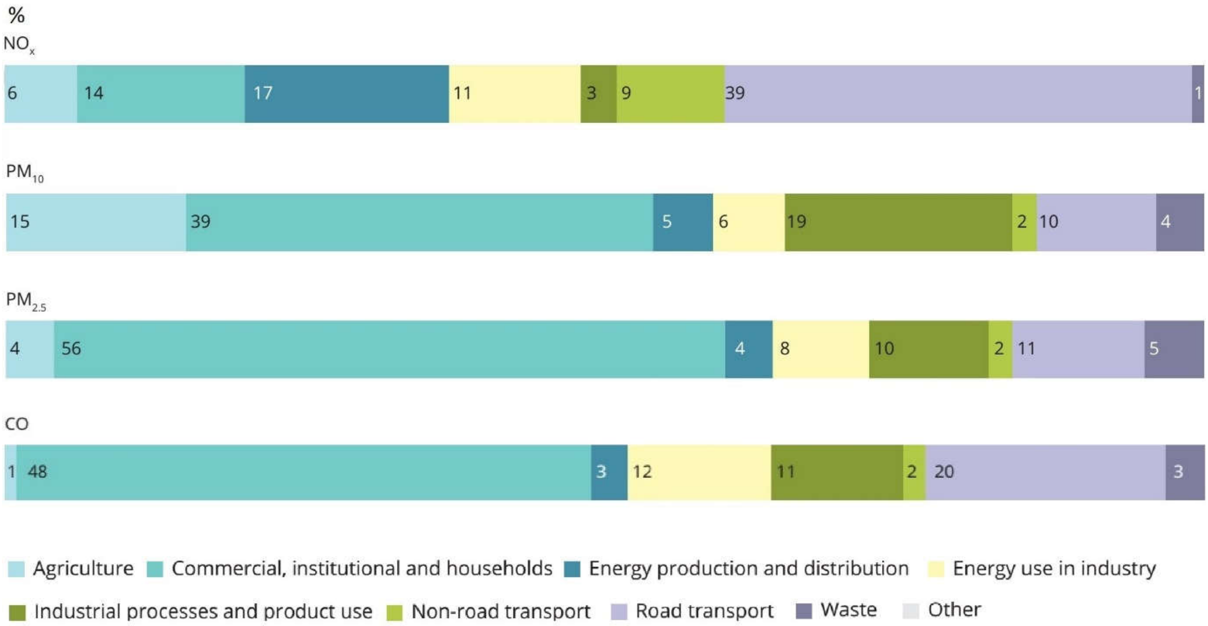

Figure 2-2: Contribution to EU-28 emissions from main source sectors in 2016 of NO_X, primary PM_{10} and $PM_{2.5}$ (source: EEA, 2018b).

The pollutant concentration has been reduced and the issue of the negative health impact due to poor air quality has been improved year by year since 1990, as environmental problems have drawn more and more public attention and relevant environmental policies have been put into effect. Figure 2-3 shows the trend of population averaged $PM_{2.5}$ concentrations from 2007 to 2015 and the premature death estimations due to $PM_{2.5}$ pollution over 1990 to 2016 using various data sets. From Figure 2-3 (a) we can see that from 2010 to 2015, the $PM_{2.5}$ population averaged concentrations had dropped a little every year. When this trend sustained for a longer time period, the outcome can be significant. From Figure 2-3 (b) we can see that the premature deaths due to exposure to $PM_{2.5}$ in Europe 1990 was around 1,000,000. This figure decreased to around 400,000 in 2016.

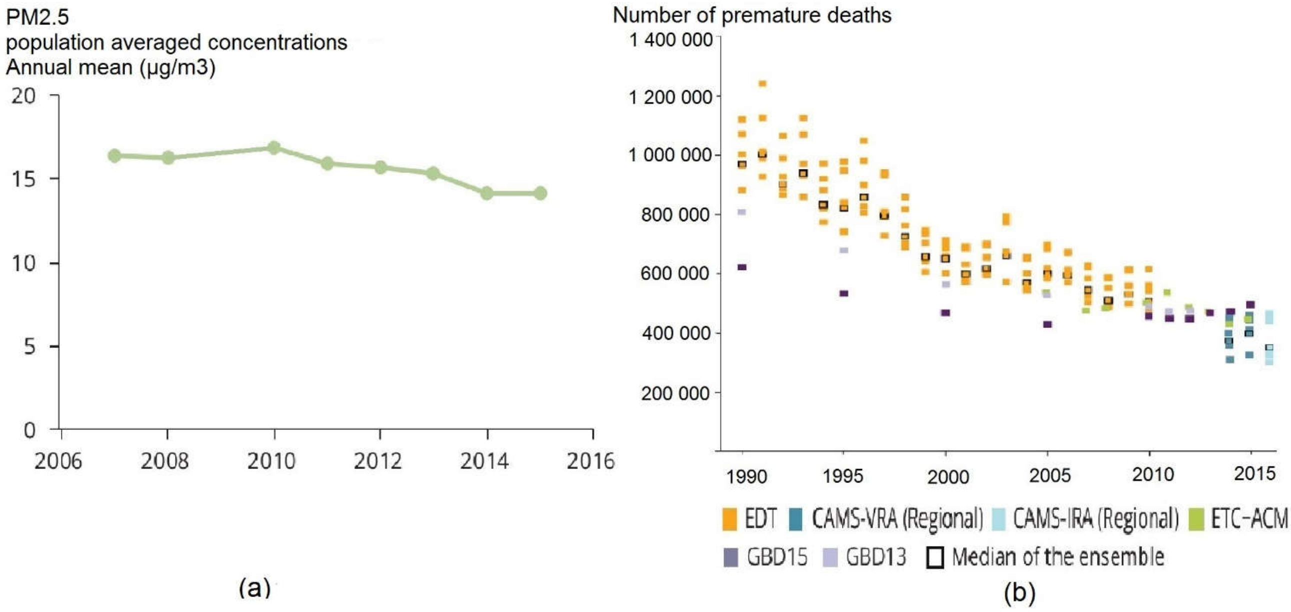

Figure 2-3: (a). Changes in total European population exposure to $PM_{2.5}$ (annual mean). (b). Premature deaths due to exposure to $PM_{2.5}$ (all-cause mortality) in Europe over the period 1990-2016 for various data sets of $PM_{2.5}$ concentration. Different colours of squares represent different data sets (source: EEA, 2018b).

Focusing on road transport section, the traffic emission is decreasing every year as shown in Figure 2-4. This is mainly due to the technology improvement and the introduction of traffic management strategies aimed to reduce traffic emissions.

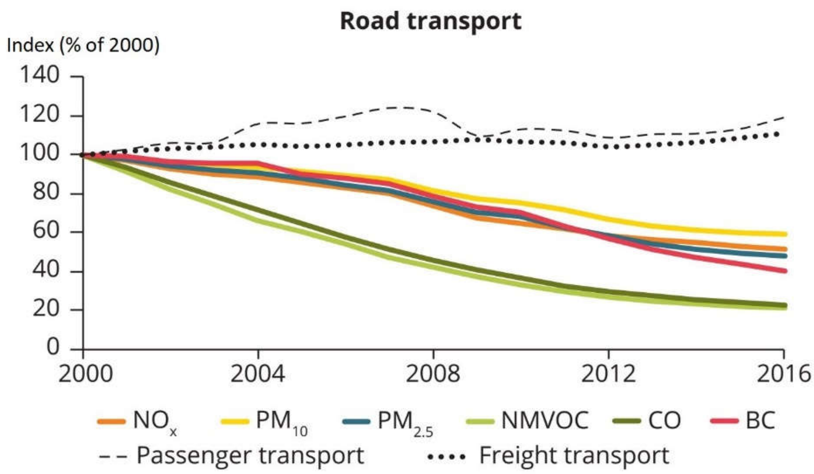

Figure 2-4: Development in EU-28 emissions from road transport, 2000-2016 (% of 2000 levels). Also shown for comparison are key EU-28 sectoral activity statistics (% of 2000 levels) (source: EEA, 2018b).

Although air quality has been improved over the years, challenges still remain. The environmental challenges are especially severe in urban settings. As mentioned above, 6% and 3% of the EU-28 population are still exposed on pollutant concentrations for $PM_{2.5}$ and NO_2 above the EU limit value. The percentage of the population exposure above the EU limit in urban is even higher, because higher pollutants concentrations were found in urban environments (EEA, 2018b). Worse air quality in urban area is mainly due to the high level of emissions from road traffic and residential combustion, and unfavourable conditions for the dispersion of emissions as a result of topography and meteorological conditions (EEA, 2018e). It is unlikely that the air quality standards for NO_2, PM and ground-level O_3 will be met in all EU Member States by 2020 because of continuing widespread exceedances in many urban areas (EEA, 2018e).

2.1.1 Air quality status in Ireland and Dublin

The air quality in Ireland is generally better than the average air quality of Europe. The population-weighted annual average concentrations in 41 European countries for $PM_{2.5}$ and NO_2, are 14.1 µg/m^3 and 18.8 µg/m^3, respectively. These values for 28 EU membership countries, being 13.9 µg/m^3 and 18.9 µg/m^3, have little difference with 41 European countries. However, these values for Ireland, being 6.5 µg/m^3 and 7.6 µg/m^3, are far below 41 or 28 European countries averages (EEA, 2018b).

Based on the monitoring records from 29 air quality monitoring stations across Ireland, there is no exceedance in the EU legislative limit values from 2010 onwards for annual average concentrations of NO_2, $PM_{2.5}$ and PM_{10} (EPA, 2018). However, a more detailed measurement about the air quality in Dublin using diffusion tube sampling method revealed that the NO_2 concentrations on the roadside were generally beyond the EU limit (EPA, 2019). Moreover, the concentrations of $PM_{2.5}$ and PM_{10} at some monitoring station exceeded the tighter WHO air quality guidelines, which are the new benchmarks for good air quality globally and it is possible that these guidelines will be adopted by EU in the coming years as legal standard (EPA, 2018).

Overall, according to Irish Environmental Protection Agency's (EPA) Air Quality Index for Health, the entire Ireland generally has good air quality index that allows the majority people to do outdoor activities without perceiving any health effects. Only people that are very sensitive to air pollution may experience health effects. Although the air quality for the entire Ireland is generally good, there are still micro-environments such as roadside places that face air pollution problems. According to Irish EPA, particulate matter from solid fuel burning remains the greatest threat to good air quality in Ireland. And this is closely followed by NO_2 from transport emissions in urban areas (EPA, 2018). It can be seen in Figure 2-5, for NO_2 concentration, the larger cities traffic area is generally higher than larger cities background area, followed by smaller urban area, and then by rural area, which demonstrate the phenomenon that area with the more traffic resulting in the more NO_2 pollution. The highest NO_2 concentration occurs in Dublin.

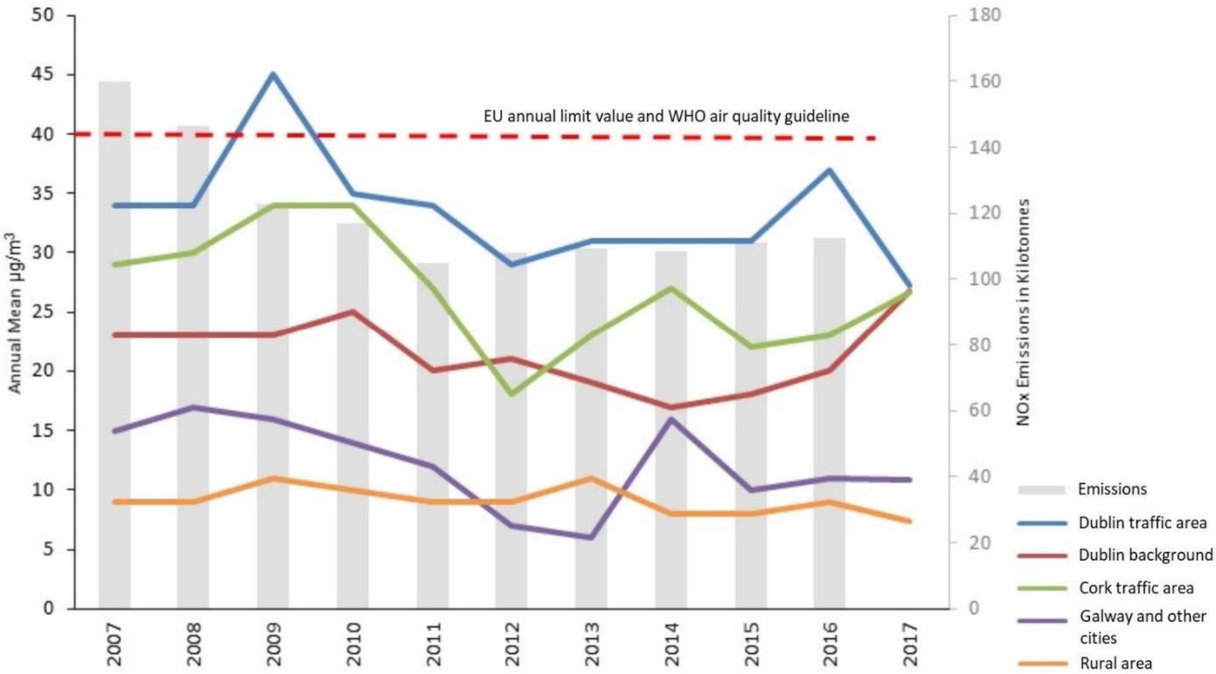

Figure 2-5: NO_2 emissions and concentrations for different area in Ireland from 2007 to 2017 (source: EPA, 2018).

Dublin is the capital of Ireland. Dublin has around 1.2 million population, accounting for 25% population of Ireland. The area of Dublin is around 120 km^2. The Dublin map is shown in Figure 2-6. From Figure 2-6 we can see that most area of the Dublin is surrounded by a motorway, M50, which is the busiest motorway in Ireland.

Averaged NO_2, $PM_{2.5}$, PM_{10} and CO concentrations in Dublin based on 12 monitoring stations are summarized in Figure 2-7 (EPA, 2018). The annual average NO_2 concentrations in Dublin met the EU limit value and the WHO air quality guideline of 40 $\mu g/m^3$ since 2007. The annual average PM_{10} concentrations also were below the EU limit value of 40 μg/m^3 and tighter WHO air quality guideline of 20 μg/m^3. The annual average $PM_{2.5}$ concentrations, although met the EU standard of 20 μg/m^3, exceeded the WHO air quality guideline of 10 μg/m^3 for some years. The maximum daily 8-hour mean CO concentrations were also under the EU limit value of 10 μg/m^3 since 2007.

From the above we can infer that the overall air quality in Dublin is qualified. The threats to Dublin air quality are mainly traffic emissions and emissions from commercial and residential combustion. There are also some stacks in Dublin that affect the air pollutant concentrations (Dublin City Council, 2009). The aforementioned 12 monitoring stations in Dublin are not representative for different micro-environments. For example, diffusion tube sampling results showed that NO_2 concentrations on the roadside generally exceeded the EU limit as mentioned before. At road side in Dublin, traffic influences the air quality the most, and other sources can be ignored (Pilla and Broderick, 2015). Moreover, the study of Beckerman et al. (2008) supported this statement. Beckerman et al. (2008) found that for the upwind side, the concentrations of NO_2, $PM_{2.5}$ and UFP at the distance of 0 m from a motorway in Toronto, Canada are about 2, 1.2, 5.8 times of the concentrations at 400m respectively, and at 200m from the motorway they are about 1.1, 1.1, 1.2 times of those at 400m; for the downwind side, the concentrations of NO_2, $PM_{2.5}$ and UFP at the distance of 0m from the motorway are about 1.8, 2.6, 5.3 times of the concentrations at 400m respectively, and at 200m from the motorway they are about 1.3, 1.2, 1.2 times of those at 400m. So, people who live close to the road are highly affected by the traffic pollution. The air quality for micro-environments in Dublin still need to investigate.

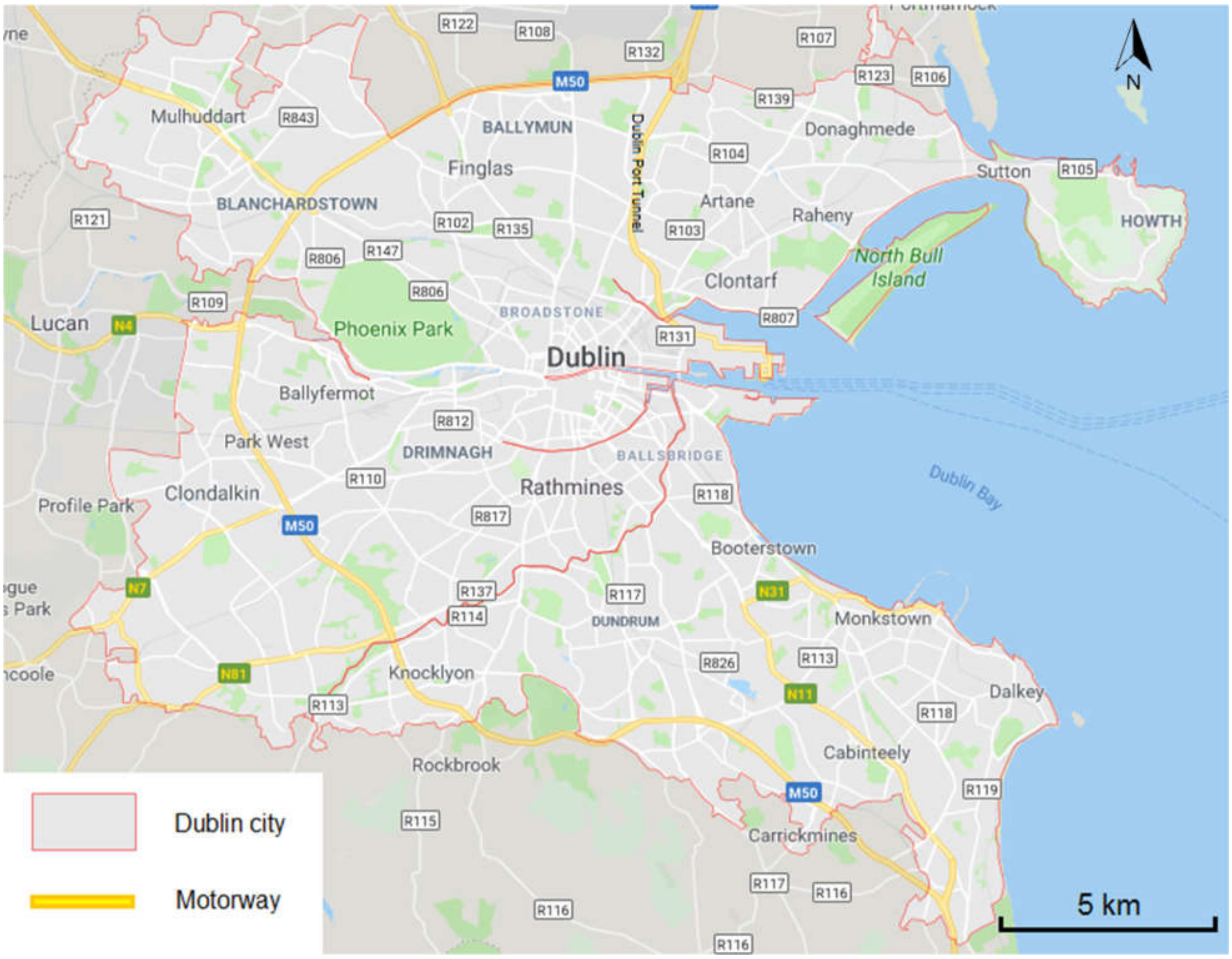

Figure 2-6: Dublin map.

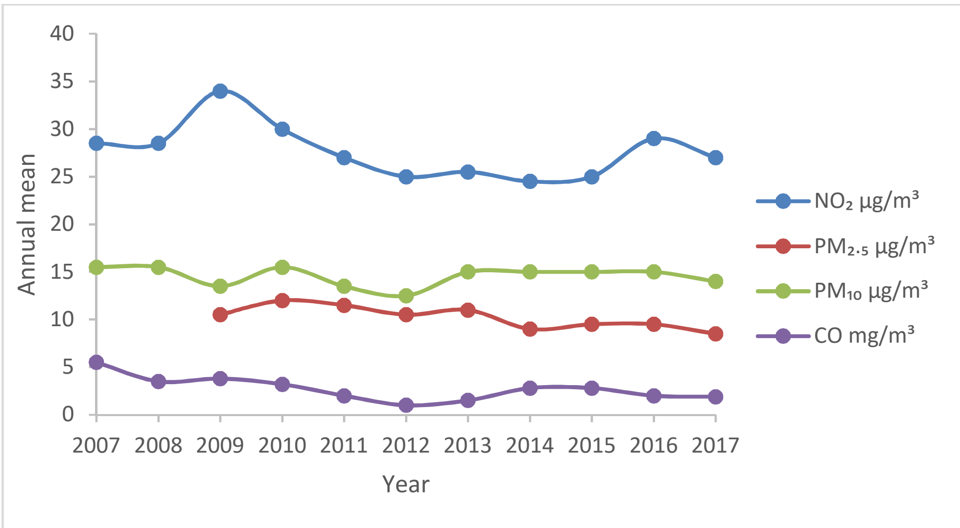

Figure 2-7: Dublin annual mean concentrations for NO_2, $PM_{2.5}$, PM_{10} and maximum daily 8 - hour mean CO concentrations from 2007 to 2017 (adapted from EPA, 2018).

2.2 Traffic emissions and impacts

Emissions from traffic includes a large amount of CO_2, CO, hydrocarbons (HC), NO_X, PM (particulate matter), and substances known as mobile source air toxics (MSATs), such as benzene, formaldehyde, acetaldehyde, 1,3-butadiene. Each of these pollutants, along with secondary by-products, such as ozone, can cause adverse effects on human health and the environment (Health Effects Institute, 2010). With the rapid urbanization process, traffic air pollution and its health impacts have raised more and more concerns.

2.2.1 Impacts of traffic emission on public health

Environmental epidemiology studies showed that traffic emissions affect public health in many aspects. Exposure to traffic-related ambient air pollutants is associated with an increase in all-cause mortality (Finkelstein et al., 2004; Beelen et al., 2008; Jerrett et al., 2013) and cardiovascular and cardiopulmonary mortality (Finkelstein et al., 2005). Also, traffic air pollution is associated with increased cardiovascular morbidity (Hoffmann et al., 2007; Tonne et al., 2007), and physiological changes in the cardiovascular system, such as changes in cardiac electrophysiological response (Schwartz et al., 2005; Adar et al., 2007).

The respiratory system is easily affected by air pollution, especially among children. For children, traffic air pollution is linked to increased prevalence of doctor-diagnosed asthma (McConnell et al., 2006; Kim et al., 2008), increased incidence of wheeze (Nordling et al., 2008), decreased lung function (Oftedal et al., 2008) and other respiratory symptoms (Morgenstern et al., 2007; Kim et al., 2008). The Health Effects Institute (2010) in the U.S. concluded that for children, there is sufficient evidence to infer a causal role for traffic-related pollution in the exacerbation of asthma and suggestive evidence to infer a causal role for traffic in the onset of asthma. For adults, increased asthma or respiratory symptoms and health-care utilization were reported to be connected to traffic air pollution (Sunyer et al., 2006; Schikowski et al., 2008). Exposure to traffic air pollution was also reported to be linked to decreased lung function for adults (Schikowski et al., 2008). The U.S. Health Effects Institution suggested that there is suggestive evidence to infer a causal role for traffic-related pollution in the exacerbation of symptoms in adults.

Other effects of exposure to traffic-related air pollution has included non-asthmatic respiratory allergy (Nicolai et al., 2003), low birth weight (Brauer et al., 2008) and cancer, especially childhood cancer (Reynolds et al. 2004).

The adverse health effects were widely reported for people who live or work near streets, e.g. street vendors and traffic policemen (Tamura et al., 2003; Kongtip et al., 2006; Noomnual & Shendell, 2017), which suggests the severity of the negative impact of traffic air pollution on human health. Besides, research shows that low-socioeconomic-status communities experience higher concentrations of air pollutants (Hajat et al., 2015; Brunt et al., 2016). This indicates that the variation of exposure to air pollution not only exists in people with different jobs, but also for groups with different socio-economic status. The inequality of the air pollution exposure and health impact due to socio-economic disparities cannot be neglected. Actions are needed to better protect the poor, the people with special jobs and the vulnerable people such as the elderly and children from air pollution.

People have become more and more concerned about traffic air pollution and have shown a strong willingness to pay to avoid health risks from road traffic related air pollution (Istamto et al., 2014; Zahedi et al., 2019). Therefore, traffic management strategies should be introduced to control traffic air pollution. A holistic assessment of traffic management strategies is essential for the constitution of traffic management strategies.

2.2.2 Impacts of traffic emission on environment

Besides health impact, the impact of traffic pollution on environment is also severe. Moreover, for some greenhouse gases (GHGs) emitted from vehicles, e.g. CO_2, CH_4 and N_2O, although having little impact on health, these emissions can affect the process of global warming and climate change, and consequently effect agriculture and ecosystems. In Europe, transport contributed 25.8 % of total greenhouse gas emissions (EEA, 2018). Table 2-1 summarizes the environmental impacts of some traffic-related air pollutants and GHG.

	Effect on ecosystems	Effect on Plants	Contribution to acidification	Contribution to eutrophication	Contribution to photo-chemical air pollution
CO_2	High	High	Low	Medium	None
NO	Low	Low	None	Low	High
NO_2	Low	Medium	High	High	High
O_3	High	Very high	None	None	Very high
PANs	Medium	Very high	None	None	High

Table 2-1: Impacts of traffic-related pollutants (Adapted from Fenger & Tjell, 2009).

2.3 Urban traffic management strategies to control traffic emissions

As traffic emissions bring significant impacts on human health, and contribute a lot to GHGs emissions, many traffic management policies and strategies were introduced worldwide in order to reduce traffic emission and its impacts. There are high-level and nationwide air quality control policies in place regulated by each government (Kuklinska et al., 2015), and road traffic emission control strategies are usually designed to support related legislation (Miranda et al., 2015). Also traffic management strategies exist designed for other functions, which could influence traffic emissions significantly as these strategies influence traffic conditions. Commonly implemented traffic management strategies are listed below (Gulia et al., 2015; Pojani & Stead, 2015; Bigazzi & Rouleau, 2017):

(1) Road infrastructure improvements, such as new road constructions, road re-builds, road maintenances;

(2) Road-based public transport related incentives, such as bus lane establishment or public transport ticket price reduction;

(3) Restrictions and pricing operation, which includes road or congestion pricing, setting low emission zones (LEZ), vehicle operating restrictions, parking management;

(4) Supports for non-motorized travel modes, for example constructing pedestrian and bike facilities;

(5) Technological solutions, such as alternative fuel options in vehicles, improvement in fuel quality, fitting of catalytic converter, optimizing fleet composition;

(6) Awareness-raising campaigns and supports, such as setting high occupancy lanes, encouraging eco-driving, building shared ride programs;

(7) Speed management, including lower speed limits or variable speed limits, speed control devices, speed enforcement devices, etc.;

(8) Traffic flow control devices that can improve traffic conditions and reduce traffic congestion, such as electronic toll collection, traffic signal timing optimization, intersection control devices;

(9) Control of land-uses, which is a more high-level strategic approach.

In order to be aware of the impact of these traffic management strategies on air quality and human health, assessments of their impacts are necessary. Some assessments of the strategies have been conducted broadly by previous investigators, and different conclusions were obtained, which were summarized in Section 2.4.

2.3.1 Traffic management strategies in European cities

In order to further improve air quality, many measures were introduced in Europe. EEA reported that measures in EU member states are always designed to reduce people's exposure to the two air pollutants: PM_{10} and NO_2, as these pollutants most commonly exceed air quality standards (EEA, 2018d).

For EU countries, the biggest NO_2 and PM_{10} sources are traffic emission and commercial and residential combustion. In line with the reasons for exceedances, 46% of the total number of PM measures implemented by EU member states target road transport, followed by the commercial and residential combustion sector (20%) and industry (17%). For NO_2, more than 60 % of the measures reported mainly target the road transport sector. Industry (13%) and the commercial and residential combustion sector (11%) are the second and third most targeted sectors (EEA, 2018d). Detailed assessment of the classification of the air quality measurements is shown in Figure 2-8.

For local governments in European cities, traffic management strategies are the most important measures designed to improve air quality, as traffic pollution is a remarkable problem for cities. EEA (2019) was reported that in 10 EU cities they investigated, more than 50% of the measures were related to traffic and urban mobility. The most common traffic measures implemented by the cities targeted reducing traffic volumes and a modal shift to cleaner modes of transport (e.g. improving public transport, adopting low-emission zones, promoting cycling).

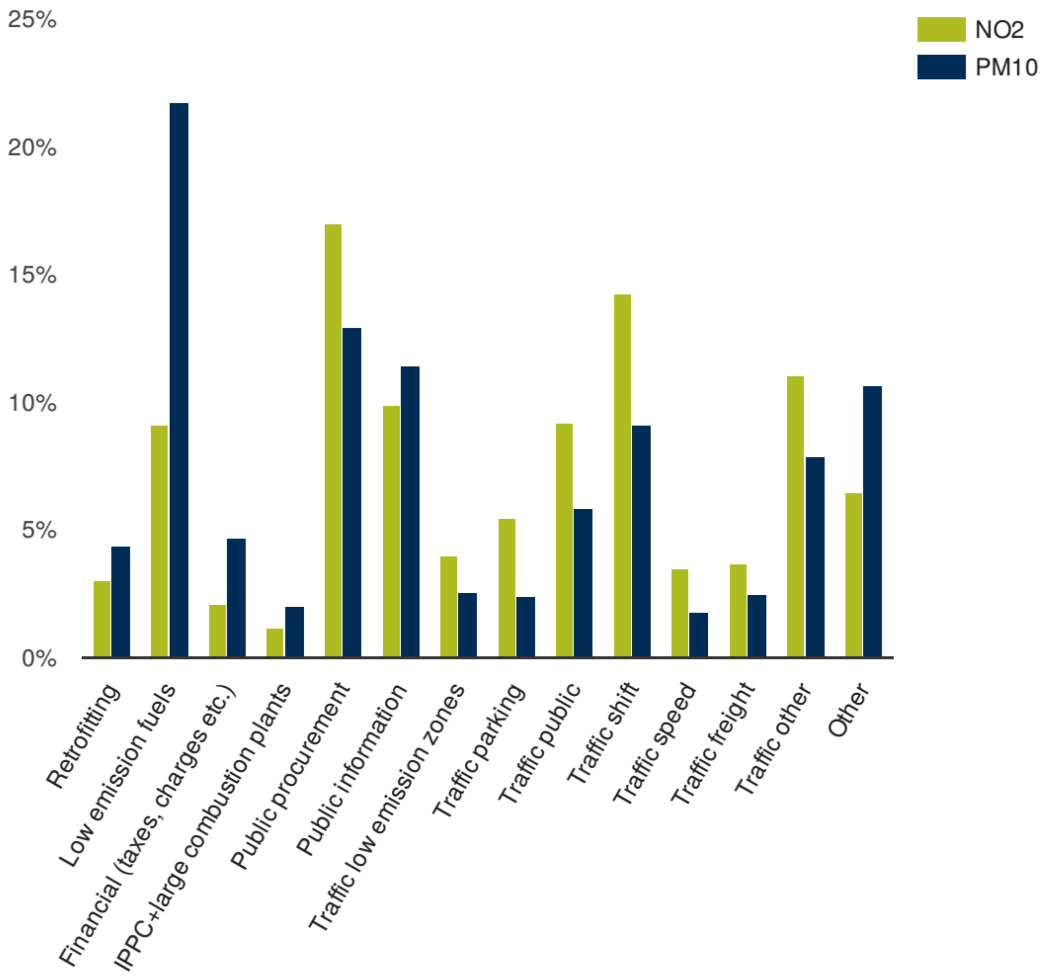

Figure 2-8: Classification of measures designed to reduce PM_{10} and NO_2 emissions in EU member states (source: EEA, 2018d).

Although positive outcomes were witnessed in Europe after introducing air quality strategies, as shown in Figure 2-3 (b), it is difficult to directly measure the impact of each individual strategy on air quality. Because there can be a series of air improvement strategies introduced at one place, and there are many confounding variables that influence air quality. Hence, it is difficult to analyse and compare directly the benefits brought about by different strategies.

Understanding the effectiveness of different air quality measures is important during the policy-making process. The majority of air quality measures is traffic management strategy, hence knowledge about the impacts of traffic management strategies is especially important. As it is hard to measure these impacts directly, many evaluation methods were introduced. For countries, national emission inventories evaluation can help to quantify the effectiveness of transport measures on reducing traffic emissions (EEA, 2010). For regional areas, the assessment methods for the impact of air quality improvement measurements are summarized in Section 2.4.

Focusing on Dublin, there are some policies that target on enforcement controls over illegal burning and excessive emissions from industry, but most of the air quality strategies are concerned with transport. Traffic policies that were introduced and are still ongoing include: 1) supporting public transport, such as introduction and expansion of the light rail network, continued expansion of quality bus corridors, provision of park and ride facilities adjacent to public transport; 2) issuing regulations to reduce traffic volume in the city centre, e.g. restriction on heavy goods vehicles in Dublin City Centre, limitation on private motor vehicles travelling on a thoroughfare in the city centre; 3) infrastructure constructions to provide alternative routes for traffic instead of passing through the city centre, such as the completion of Dublin port tunnel providing priority routes for heavy goods vehicles entering and exiting the port; 4) speed control, e.g. traffic calming installation in residential areas; 5) incentives for cycling, e.g. provision of cycle ways and application of tax incentive scheme in favour of cycling; 6) parking approaches, such as increased tariffs for pay and display for on-street parking in Dublin City and into suburban commercial areas (Dublin City Council, 2009).

The impact of these strategies was not evaluated on their own. In addition, Dublin does not have a local emission inventory (EEA, 2018e), which can help to estimate the impact of these strategies. Hence, further investigation of the impact of these strategies is needed.

2.4 Assessment of traffic management strategies

Two approaches were commonly employed for environmental assessments of traffic management strategies, i.e. 1) the measurement of traffic emission changes or air quality changes before and after the implementation of a strategy, and 2) the modelling of their impacts.

2.4.1 Measurement

The traffic management strategy of road pricing has been widely studied. In London, measurements found evidence of relative reductions in concentrations of NO, PM_{10} and CO, and relative increases in concentrations of NO_2 and O_3 in a congestion charging zone compared to a control zone. But as other traffic and emissions interventions were introduced concurrently with this congestion charging scheme (CCS), which might have had a more concentrated effect in central London, the study concluded that it is not appropriate to attribute these air quality changes solely to CCS (Atkinson et al., 2009). Other studies also observed air quality improvements in the congestion charging zone of London and other places after the implementation of CCS, but little evidence has been presented showing that these improvements were due to the CCS (Beevers & Carslaw, 2005; Burman et al., 2010; Givoni, 2012).

Stringent vehicle restrictions were found to be associated with notable improvements in air quality and public health. For instance, a significant correlation between aerosol optimal depth (AOD) improvement and the timing and location of plant closure and traffic restriction was observed in the time of the implementation of a stringent vehicle restriction policy during 2008 Beijing Olympics in China. Also the improvement of the officially reported air pollution index (API), the decrease of all-cause mortality, and the increase of baby birth weight was reported during that time, as the pollutant emissions reduced by 41% to 57%, depending on the pollutant in question (Wang et al., 2010; Chen et al., 2013; Rich et al., 2015; He et al., 2016). However, as plant closure and traffic restriction were implemented at the same time, it is difficult to quantify the proportion of these improvements contributed solely by the traffic restriction policy.

By comparison, the impacts of lenient vehicle restrictions are less certain. For example, in London, the policies of setting LEZ from which some polluting vehicles were restricted to enter, were found to be associated with PM concentration reduction, while no discernible difference was found in NOx concentration. But the reductions in PM were concluded to be more likely to be due to the introduction of zero sulphur diesel at the same time (Jones et al., 2012; Ellison et al., 2013). Studies in German and Danish cities concluded that the effect of LEZs on monitored PM concentrations was little (Jensen et al., 2011; Cyrys et al., 2014). However, Panteliadis et al. (2014) found statistically significant decreases in measured concentrations of NO_2, NO_X, PM_{10} and EC in the LEZ in Amsterdam, the Netherlands. Therefore, there is no conclusive agreement on the impact of lenient vehicle restrictions.

In other types of traffic strategies, it has been difficult to utilize monitoring air quality data alone to evaluate impacts as there are many confounding effects that are difficult to control (Holman et al., 2015). Also because of the constraints around air quality monitoring resources, a limited number of strategies have been assessed by measurement alone. Most of these assessments were conducted by modelling.

2.4.2 Modelling

The modelling approach to assess the impacts of traffic management strategies on emissions generally includes a traffic model established from or calibrated by traffic counts to evaluate the traffic condition changes brought about by traffic management strategies, and an emission model to evaluate the emission changes brought about by traffic condition changes. A dispersion model would be integrated with the two former types of models when the assessment of the impact of these strategies on air quality is needed. In addition, a health impact model can be added to estimate the health impacts associated with air quality changes (Mitchell et al., 2002; Bigazzi & Rouleau, 2017).

The impact of different intersection control approaches on traffic emissions have attracted a lot of attention. Roundabouts, stop signs, signal control and speed bumps were compared to each other by researches regarding their impacts on traffic emission. Using microscopic energy and emission models to estimate the emission from vehicles when encountering stop signs, roundabouts and speed bumps, Ahn & Rakha (2009) found that all intersection control measures increased traffic emissions, but among them roundabouts produced the least increases in vehicle fuel consumption and emissions, as roundabouts allow smoother driving patterns with mildest acceleration behaviour in general. Hu et al.

(2014) also found that fuel consumption and vehicle emissions could be reduced when other intersection control devices are replaced by roundabouts. However, on-road emission measurements found that roundabouts did not necessarily have lower emissions than traditional stop or signal control (Hallmark et al., 2011), which is not consistent with aforementioned modelling results. In addition, different types of roundabouts have different impact on traffic emissions (Fernandes et al., 2016). Thus there is no consensus on the benefits of these intersection control approaches to the environment.

Studies have also suggested that optimizing traffic signal timing and implementing intelligent transportation system can be effective in reducing traffic emissions, but the impacts have been found to be different regarding different pollutants (Chong-White et al., 2012; Battelle and Texas A&M Transportation Institute (TTI), 2014; Barth et al., 2015). For example, studies showed that the improvement of signal timing can reduce or have no effect on $PM_{2.5}$, PM_{10} and VOCs emissions, but CO and NO_X emissions can be increased or decreased (ICF International, 2006). Moreover, Kalra et al., (2012) suggested that the demand induced by these improvements has not been adequately addressed, which can generate extra emissions.

The impact of speed limit changes on motorways are complex. Studies in the Netherlands and in Barcelona, Spain found that speed limit change from 120 or 100 km/h to 80 km/h could reduce NO_X and PM_{10} in the range of 5% - 30% (Baldasano et al., 2010; Keuken et al., 2010). Whereas Bel et al. (2015) found that reducing speed limit from 120 or 100 km/h to 80 km/h has no impact on pollution levels on Barcelona's motorways, and only variable speed limit system can reduce NO_X and PM_{10} levels.

In addition, previous investigations have utilized modelling data compared with measurement data to assess traffic management strategies. For example, in Stockholm, Sweden, reductions in concentrations of PM_{10} by 15%-20%, NO_X by 10%, and CO by 15% on average were observed after the implementation of a congestion tax. However, the observed reduction in PM_{10} were found to be more affected by the meteorological conditions than by the congestion tax (Burman et al., 2010). The modelled calculation for the same congestion tax in Stockholm shows that the emission reduction and annual average NO_X concentrations would be lower by up to 12% along the most densely trafficked streets in the congestion tax zone. PM_{10} concentrations would be up to 7% lower (Johansson et al., 2009). The advantage of using measured data is that we can capture the real air quality changes after the introduction of a traffic management strategy. The disadvantage is that excluding confounding factors can be difficult. On the contrary, confounding factors can be controlled by using the modelled data to assess a strategy, but models intrinsically involve uncertainty.

From the traffic management strategy assessments mentioned above, we can infer that there is no cut-clear conclusion for most types of traffic management strategy of whether a strategy is beneficial or detrimental to the reduction of traffic emissions or the improvement of air quality. The impact of a strategy can be different at different places or for different pollutants (ICF International, 2006). Thus the comparison between different types of strategies can be even more difficult. Moreover, the assessments were usually conducted regarding their impact on traffic emission or air quality, and there are very limited number of health assessments of these strategies.

2.5 Models for the assessment of traffic management strategies

2.5.1 Traffic modelling

"*A model is a simplified representation of a part of the real world – the system of interest – which focuses on certain elements considered important from a particular point of view. Models are, therefore, problem and viewpoint specific*" (de Dios Ortuzar & Willumsen, 2011). Traffic models, according to their scope of space and time, can be categorised into three types in general, i.e. macroscopic, mesoscopic and microscopic traffic model. Each type of traffic model represents a trade-off between desired scope and analytical complexity.

Microscopic traffic models generally simulate the action of individual vehicles. According to traffic flow theories that the models are based on, microscopic traffic models can be classified into the general car following models and the safe-distance models. The general car following models assume that the following characteristic of a driver depends on his/her response to stimuli induced by the leading driver. Safe-distance models are based on the assumption that the following driver seeks to maintain a safe distance between him/her and the driver in front to avoid collision (Adebisi, 2017). The speed, travel time and other variables of each individual vehicle can be estimated by microscopic traffic model, and variables of traffic flow such as traffic flux and traffic density can also be assessed in real-time from the simulation.

Mesoscopic models are taking both features of microscopic models that describe the behaviour of individual vehicles, and macroscopic models that describe traffic as a continuum flow. Mesoscopic models describe vehicle behaviour in aggregate terms such as in probability distributions, but their vehicle behaviour rules are defined for individual vehicles (van Wageningen-Kessels et al., 2015). The most popular type of mesoscopic model is the gas kinetic model. This type of models were developed in analogy to models describing the motion of large numbers of small particles in a gas, which can be used to describe the dynamics of velocity distribution functions of vehicles when applied to the motion of vehicles in a traffic stream (van Wageningen-Kessels et al., 2015). Because the findings of mesoscopic models are still limited to the laboratory test, their practical applications are lagging in the field of transportation (Adebisi, 2017).

Macroscopic traffic models evaluate the mass action or the statistical properties of a large number of vehicles. Aggregated variables, such as average density, average traffic flux and average speed in roads, are determined in macroscopic models. Therefore, macroscopic models are suitable to traffic assessments that are focused on aggregated variables, not on each individual vehicles. The scope of the assessments are usually in a relatively large space scale (e.g. a city) and across a relative long time (e.g. a year).

In this thesis, as the impacts of traffic management strategies on the whole Dublin were assessed and aggregated traffic features were of concern, a macro traffic model is suitable for the scope of thesis.

There are several macroscopic traffic modelling tools, such as EMME, TransCAD, VISUM, Aimsun, CUBE, Saturn, etc. Among these tools, VISUM was shown to be technically appropriate for the type of work in the thesis, with a strong GIS interface, allowing high quality presentation of results. VISUM has also been shown to have a substantial user base. In particular, it was adopted by Transport Infrastructure Ireland (TII) for the development of the National Traffic Model (NTM) and its later version, National Transport Model (NTpM) (NRA, 2009; NRA, 2014). Besides, VISUM allows the use of Python script to control the modelling process, which is compatible with the batch processing that is required by the assessment of traffic management strategies in this thesis (See Chapter 4). Therefore, VISUM was selected as the traffic model to conduct the assessment.

The classical four steps model was adopted in VISUM to evaluate the travel demand and traffic conditions. The four steps model involves the steps of trip generation, trip distribution mode split and route assignment. In the trip generation step, the number of trips produced by and attracted to each traffic analysis zone (TAZ) was predicted, that is, total flows into and out of each TAZ in the study area were estimated. Trip distribution step is conducted to link the trip starts to the trip ends predicted in the trip generation step. In this step, an O-D (Origin-Destination) matrix is generated to represent each flow from one TAZ to another TAZ. In the mode split step, the percentages of travel flow using each of the available modes of transport between the origin and the destination were predicted, and an O-D matrix can be generated for each mode. In the route assignment step, the O-D flows for each mode are placed on specific routes of travel through the network. The route assignment is generally based on the equilibrium assignment. The basic principle of equilibrium assignment is Wardrop's first principle, which stated that under equilibrium conditions, traffic arranges itself in congested networks in such a way that no individual trip maker can reduce his path costs (time) by switching routes.

The traffic management strategies evaluated in the thesis mainly affected the procedure of route assignment. After these steps, the traffic condition such as traffic volumes, vehicle speeds and average vehicle travel times were obtained.

2.5.2 Emission modelling

Corresponding to different types of traffic models, emission models can also be classified as microscopic models, which estimate vehicle emissions based on second-by-second vehicle performance characteristics, and macroscopic models, which are based on the aggregated variables. Microscopic emission models take instantaneous driving modes (e.g. cruise, acceleration etc.), gear shift strategies, vehicle loadings, road gradients, and vehicle characteristics (mass, size, air resistance, etc.) into account. Examples include PHEM and MOVES which widely used in Europe and USA (U.S. EPA, 2014; iCET, 2015). Because aggregated emissions are essential for this thesis, and microscopic models require a lot of detailed inputs that are not relevant to this thesis, and need a lot of computing resources, a macroscopic emission model was selected as appropriate for this thesis.

Macroscopic emission models that are widely used worldwide can be generally categorized as average speed models, traffic situation models and aggregated emission factor models. Aggregated emission factor models (e.g. NAEI adopted in Europe and MOBILE adopted in USA) utilize area or road type to define vehicle operation conditions, which do not depend on vehicle speed or other vehicle specific characteristics. Aggregated emission factor models are at the simplest level among these three types of macroscopic emission model. On the contrary, average speed models take average trip speed into account and are specific to each vehicle type and technology, which is suitable for the traffic management strategy assessments in this thesis and is compatible with the output from the proposed traffic model. Examples of this emission model type include COPERT, which is widely adopted in Europe and in the Asia-pacific area. An alternative is VEPM which is adopted by the New Zealand government. There are also traffic situation models which estimate traffic emissions based on the traffic situation, such as type of road, speed limits, congestion level, etc. Widely-used models of this type include ARTEMIS, HBEFA, VFEM and VERSIT+. However, HBEFA, VFEM and VERSIT+ are only applicable in limited places, having limited traffic situations and are valid for limited vehicle types, respectively. ARTEMIS has the largest emission database available but some sub-models are very complex and may introduce unnecessary uncertainty (iCET, 2015; Carroll et al., 2016).

Balancing the computer resource cost, compatibility and model uncertainty, an average speed model COPERT was selected as the emission model for conducting the assessment in this thesis. COPERT was developed for official road transport emission inventory preparation in EEA member countries. The governments of majority countries in Europe (22 out of the EU27 member states) have adopted COPERT to evaluate traffic emission, including Ireland.

2.5.2.1 COPERT methodology

Emissions are produced from two sources, engine emissions, which distinguished into those produced during thermally stabilized engine operation (hot exhaust emissions) and emissions occurring during engine start from ambient temperature (cold-start and warming-up effects), and diffuse emissions, which include NMVOC (non-methane volatile organic compounds) emissions due to fuel evaporation and non-exhaust PM (particulate matter) emissions from tyre and brake wear. The traffic management strategies that were investigated in the thesis mainly affected the exhaust emissions, therefore the calculation of exhaust emissions was focused on in the thesis. In COPERT, the formula for estimating exhaust emissions for a given time period is displayed in Equation 2-1 (Leon and Zissis, 2014).

$$E_{i,k,r} = N_k \times M_{k.r} \times e_{HOT;i,k,r}(V) \times (1 + \beta_{i.k} \times ({e^{COLD}}/{e^{HOT}}_{i,k} - 1))$$

Equation 2-1: Exhaust emissions calculation formula.

Where, $E_{i,\,k,\,r}$ = exhaust emissions of the pollutant i (g), produced in the period concerned by vehicles of technology k driven on roads of type r;

N_k = number of vehicles (veh) of technology k in operation in the period concerned;

$M_{k,r}$ = distance per vehicle (km/veh) driven on roads of type r by vehicles of technology k;

$e_{HOT;\,i,\,k,\,r}(V)$ = hot emission factor in (g/km) for pollutant i, relevant for the vehicle technology k, operated on roads of type r, which is dependent on the average vehicle speed V;

$\beta_{i,\,k}$ = fraction of mileage driven with a cold engine or the catalyst operated below the light-off temperature for pollutant i and vehicle technology k;

e^{COLD} / $e^{HOT}{}_{i,k}$ = cold/hot emission quotient for pollutant i and vehicles of k technology.

Pollutant i represents CH_4, CO, CO_2, N_2O, NH_3, NO_X, PM, VOC, etc.

Vehicle technology k represents technologies presented in Table A-2 in Appendix A comprising the different fuel type, engine size and EURO class categories of vehicles available in the fleet. Road type r include three types of roads, i.e. urban roads, rural roads, and motorways. Speed (V)-dependant emission factors $e_{HOT;\,i,\,k,\,r}(V)$ can be expressed as the following:

$$e_{HOT;i,k,r}(V) = \frac{\alpha V^2 + \beta V + \gamma + \delta/V}{\epsilon V^2 + \zeta V + \eta}$$

Equation 2-2: Emission factors calculation formula.

Where, α, β, γ, δ, ε, ζ, and η are experimental parameters, and dependent on pollutant i, vehicle technology k and road type r.

The representativeness of emission factors is one of the main factors that determine the accuracy of estimations for traffic emissions. The emission factors are tested by measuring vehicle emissions when vehicles are operating. At present, the common practice is to operate vehicles on a chassis dynamometer. A chassis dynamometer is a stationary platform that can simulate the inertia of the vehicle, as well as the air drag resistance and the friction on the vehicle when the vehicle is operating. Then all emissions from the vehicle tailpipe are collected in sealed bags and were analysed subsequently. Thus, the emission results, measured in grams of pollutant per kilometre driven, are then determined. During the test process, each vehicle follows a pre-defined driving cycle, which is a pre-defined cycle of accelerations, gear changes, steady speeds, decelerations and idling in order to imitate real-world driving. The COPERT model adopts the Common Artemis Driving Cycles (CADC), which were developed based on statistical analysis of a large database of European real-world driving patterns. CADC include three driving schedules for different road types: urban, rural road and motorway. For different road types, these driving cycles includes different driving features to imitate the driving conditions on different road types, i.e. they have different speed distributions, different proportions of acceleration, deceleration, idling and cruising etc. Thus COPERT as an average speed emission model, captures the average trend of emissions on urban area, rural area and motorways at different average vehicle speeds (EEA, 2016).

Although COPERT is a widely used tool for calculating traffic air pollutant emissions and GHG emissions, and it incorporates driving cycles that captures real-world driving features, there is still gap between real world and test cycle emissions. Therefore, when using a traffic emission model, attention to the gaps and uncertainties should always be given.

The existing gap between real world and test cycle emissions is mainly due to three factors (EEA, 2016): 1) the difficulties for a test procedure to fully reflect real-world driving conditions; 2) flexibilities in the test procedures, such as wheel and tyre specification and condition, vehicle loading mass, and test track design for resistance; 3) several in-use factors which are driver dependent, e.g. driving style, or independent, e.g. operation temperature and other environmental conditions.

Measuring emissions from vehicles as they are driven on roads directly can be achieved. A Portable Emissions Measurement System (PEMS) is a transportable measurement system containing a variety of instruments that can be carried on board a vehicle to monitor the real-time emissions of selected pollutants. PEMS is still a relatively new technology, but is considered relatively simple and inexpensive to purchase and maintain compared with a chassis dynamometer. Its main limitations are the reduced range of pollutants that can be measured during a test compared with laboratory testing on chassis dynamometer, as well as the mass (30-150 kg) it adds to the vehicle, which can affect the fuel consumption and hence measurements of the different pollutants. Furthermore, the lower repeatability of measurements encountered when testing, owing to real-world sources of variability, can be challenging to ensure consistency of measurements between different vehicles tested. These limitations of the real-time emission measurement restrict it for being as the method to measure emission factors. However, the gap between test and real-world emissions is still needed to be addressed. The European Union has recently agreed a Real Driving Emission (RDE) test procedure for cars and vans. Following its introduction, the EU will become the first region in the world to use on-road emissions testing methods for legal compliance purposes (EEA, 2016).

Therefore, currently, adopting COPERT to estimate traffic emissions is an ideal choice, yet the uncertainties of the emission model due to the nature of the difficulties in determining emission factors need attention (EEA, 2016).

2.5.3 Dispersion modelling

A dispersion model describes the movement of pollutants in the air and can be used to evaluate the impact of pollutants from traffic sources on air quality. There are different types of model with different assumptions that are suitable for different situations, which are summarized below (Zannetti, 1990; Holmes & Morawska, 2006; De Visscher, 2013).

Box models are based on the conservation of mass. The modelled area is treated as a box into which pollutants are emitted and undergo chemical and physical processes. It requires the input of simple meteorology and emissions and the movement of pollutants in and out of the box is accounted for. The inside of the box is not defined, and the air mass is treated as if it is well mixed and concentrations uniform throughout. One advantage of the box model is because of the simplified meteorology box models can include more detailed chemical reaction schemes. But the disadvantage is that the assumption is too simplified and thus only suitable for conditions that pollutants are well-mixed. Box modelling tools include AURORA, CPB, PBM, etc (Holmes & Morawska, 2006).

Gaussian type models are widely used in atmospheric dispersion modelling, in particular for regulatory purposes, and are often "nested" within Lagrangian and Eulerian models. Gaussian models are based on a Gaussian distribution of the plume in the vertical and horizontal directions under steady state conditions. The normal distribution of the plume is modified at greater distances due to the effects of turbulent reflection from the surface of the earth and at the boundary layer when the mixing height is low. The width of the plume is determined by σ_y and σ_z, which are defined either by stability classes or travel time from the source. One limitation of plume models with regards to modelling particle dispersion is that since the plume models use steady state approximations, they do not take into account the time required for the pollutant to travel to the receptor. This type model is recommended for long term average concentration assessments. Gaussian models include CALINE4, HIWAY2, CAR-FMI, OSPM, CALPUFF, AEROPOL, AERMOD, UK-ADMS, SCREEN3 (Holmes & Morawska, 2006).

The above-mentioned box model is a Eulerian model. There is another model, Lagrangian models, which are similar to box models in that they define a region of air as a box containing an initial concentration of pollutants. The Lagrangian model then follow the trajectory of the box as it moves downwind. The feature of the Lagrangian model is that it uses a reference system that moves with the air parcel whereas Eulerian models use a fixed reference system. The concentration is a product of a source term and a probability density function as the pollutant moves from x to x'. Lagrangian models incorporate changes in concentration due to mean fluid velocity, turbulence of the wind components and molecular diffusion. Their accuracy is better than the Gaussian plume model for short term predictions. Sometimes a Lagrangian model would incorporate an Eulerian approach to help its calculation. Lagrangian/Eulerian Models include GRAL, TAPM, ARIA Regional (Holmes & Morawska, 2006).

Computational fluid dynamic (CFD) models provide complex analysis of fluid flow based on conservation of mass and momentum by resolving the Navier-Stokes equation using finite difference and finite volume methods in two or three dimensions, which consumes the most computer resources among all models mentioned. CFD models are recommended for detailed and instantaneous prediction within limited spatial scales. CFD models include ANSYS, ARIA Local, MISKAM, MICRO-CALGRID (Holmes & Morawska, 2006).

Factors which are critical to the choice of the model include: the complexity of the environment, the dimensions of the model, the nature of the pollutant source, the computing power and time that is required, and the accuracy and time scale of the calculated concentrations desired. Even with the most accurate model fluctuations in the wind flow and emission strengths mean that the results generated are only an approximation of the actual concentrations (Holmes & Morawska, 2006). Focusing on average pollutant concentration, box model and Gaussian type models are adequate for the air quality assessment required in this thesis.

In the urban context, buildings along streets influence the dispersion of traffic pollutants, with partially enclosed street forming a relatively closed place, causing the street canyon phenomenon, which can be represented by a box model. A dispersion model that is suitable for evaluating air quality in streets canyons is necessary. OSPM (the Operational Street Pollution Model), a combination of box model and Gaussian model, was chosen to evaluate air quality on streets in Dublin. Outside street canyon spaces a Gaussian plume model was selected.

2.5.3.1 OSPM

The ratio of the height of buildings along a street and the width of the street (H/W ratio) plays an important role in air pollutant dispersion and thus affects the pollutant concentration (Oke, 1988). Closer spacing between buildings (H/W > 0.7) results in a skimming flow regime where a stable circulatory vortex is established in the street canyon and the ambient flow is decoupled from the street flow (Oke, 1988). In the urban context, where H/W ratio for streets can regularly exceed 0.7, the impact of the circulatory vortex on pollutant concentration should be considered.

In order to model the impact of a circulatory vortex, in OSPM, concentrations of exhaust gases in the street are calculated as a sum of the direct contribution calculated by a Gaussian plume model and the recirculating part of the pollutants calculated by a box model as shown in Figure 2-9. Street geometry, reactions between NO, NO_2 and O_3 and wind direction were taken into consideration of OSPM. For the NO-NO_2-O_3 chemistry, the direct emission of NO_2 is set to 5% of NO_X (Berkowicz et al., 1997). Previous investigations have evaluated OSPM against measured data in urban street canyons for NO_X, NO_2, O_3, CO, PM and benzene, and overall good performances were found, with the values of the correlation coefficient between the observed and the modelled air quality generally in the range of 0.8 to 1.0 (Kakosimos et al., 2010).

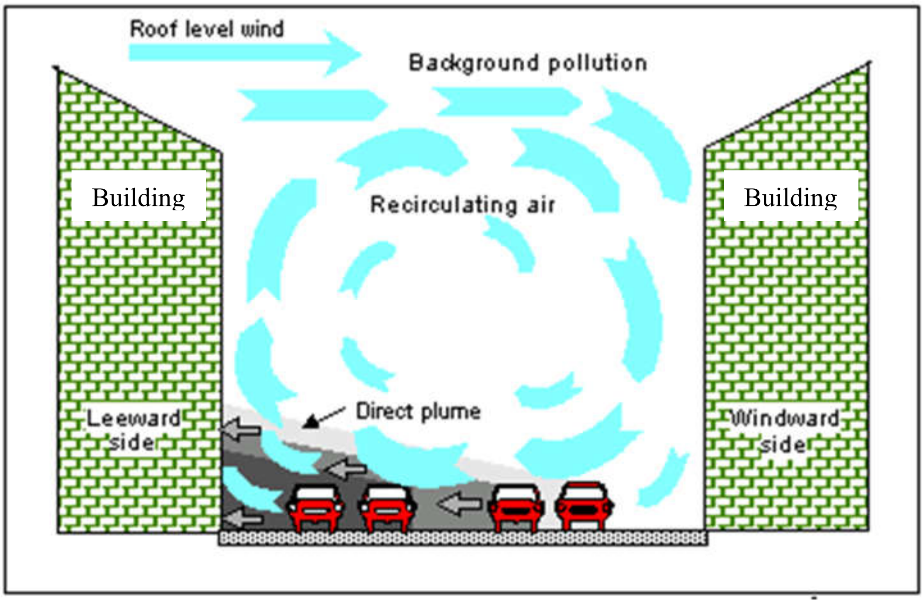

Figure 2-9: Basic model principles in OSPM (from Berkowicz et al. , 1997).

2.5.4 Health impact modelling

As epidemiological studies show that traffic air pollution relates to incidence rates of mortality and many diseases, the health impact of traffic management strategies can be measured by changes in mortality and morbidity incidence of certain diseases or in general associated with different pollutant concentration changes brought about by different strategies. By definition, the incidence rate is the number of new cases of a given health outcome per population at risk in a given time period, and the incidence is a measure of the probability of occurrence of a given medical condition in a population within a specified period of time. For a given timeframe, the incidence is the multiplication of the incidence rate and related population.

A large part of past research associated pollutant concentration increases with risk ratio (RR, also called relative risk), which compares the two risks (or incidence rate) as an index (Jewell, 2009). Consider a group of the population under different air pollution levels, with the incidence rate under each of these pollution levels, y_1 and y_0. The relative risk, by definition, is y_1/y_0.

For research that has associated pollutant concentration increases with RR, a log-linear relationship can be assumed between the pollutant concentration increase and incidence rate as shown in Equation 2-3 (1), and the incidence change for different pollutant concentration changes can be evaluated as shown in Equation 2-3 (2) and (3) (U.S. EPA, 2015).

$$\ln(y) = \ln(B) + \beta \times Q \qquad (1)$$

Therefore,

$$\Delta I = y_0 \left(1 - e^{-\beta \Delta Q}\right) \times Pop \qquad (2)$$

Where, $\beta = \frac{\ln(RR)}{\Delta Q_r} \qquad (3)$

Equation 2-3: Relationship between incidence changes and RR.

Where, RR represents the RR derived from epidemiology literature, ΔQ_r represents the change in concentration Q associated with this RR in epidemiology literature, ΔI represents the changes of incidence that is needed to investigate (baseline incidence - ‘current’ incidence), y_0 represents the baseline incidence rate concerned in the research, β represents the estimation of the effect of the Q concentration on the incidence, ΔQ represents the change of the concentration Q concerned in the research (baseline concentration - ‘current’ concentration), and Pop represents the related population in the research.

There are several tools that can be utilized for assessing and visualizing ambient air pollution health risk, e.g. AIRQ2.2, BenMAP-CE, EBD, GMAPS (Anenberg et al., 2016). Among these tools, BenMAP-CE (Environmental Benefits Mapping and Analysis Program -- Community Edition, hereinafter to be referred as BenMAP) was selected to conduct the health impact assessment in this thesis because of its flexibility in scope and it allows user-defined resolutions, concentration-response relationships, baseline incidences and demographic data.

BenMAP is a software that can calculate the number of air pollution-related deaths and illnesses. The key and basic health impact function used in BenMAP is the Equation 2-3. The workflow of BenMAP is showed in Figure 2-10.

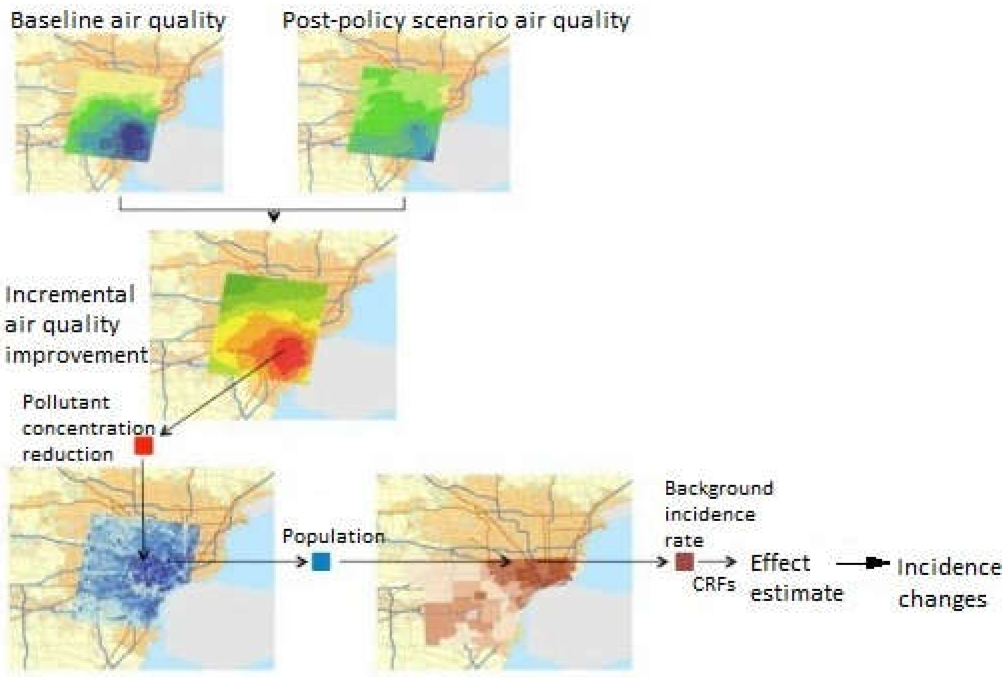

Figure 2-10: Air Pollution health impact estimation process. (Adapted from U.S. EPA, 2015)

2.6 Summary

The impacts of many traffic management strategies are still unclear, as there are insufficient or even contradictory results for the evaluation of the impact of traffic management strategies. Due to unclear results for the impact of one traffic management strategy, it is more difficult to compare the impacts of different strategies as the impact of these strategies are highly contextual. Also previous studies have rarely included health impact assessment. In addition, although vulnerable population groups such as children are more prone to be affected by air pollution, previous traffic management assessment studies have not given sufficient attention and targeted vulnerable groups. Moreover, although traffic management strategies have influenced both air pollution and GHG emissions, previous assessments have not always included the impacts on both of them.

Therefore, in this thesis, four types of traffic management strategies were selected to investigate their impacts on air pollution, GHG emissions and public health, under the same traffic conditions, which is based on the conditions in Dublin, Ireland in 2013, using a modelling approach. These strategies were infrastructure construction, traffic management regulation, speed limits and fleet composition changes. The infrastructure construction and traffic management regulation changes were chosen for the purpose of assessing two of the main air quality improvement strategies implemented in Dublin. For speed limits and fleet composition changes, special focus was given to their impact on school children, as speed limit changes were often found near schools, and in order to in line with several recent proposals regarding fleet compositions in European cities.

Chapter 3 evaluates the impacts of the infrastructural change and the traffic management regulation change on traffic condition and traffic emission in the entire Dublin. It focusses on the pollutant emissions of NO_X, total PM and CO, and GHG emissions of CO_2 and CH_4, as the impacts of these strategies on emissions of both traffic pollutants and GHGs can be significant regarding the whole city. Chapter 4 assesses the impacts of speed limit changes and fleet composition changes on air quality near a school in Dublin city centre. The pollutant concentrations of NO_2, PM_{10}, $PM_{2.5}$, CO and Benzene were assessed. These pollutants can be harmful to people, especially to vulnerable group such as school children. The changes of these pollutant concentrations are therefore of concern to the public and this thesis. The changes of GHG emissions concerning about only Dublin city centre can be limited, thus GHG emission changes assessment were not included in Chapter 4. Chapter 5 evaluates the health impact of four aforementioned strategies in Dublin. Because NO_2 and $PM_{2.5}$ are considered the most health-harmful traffic air pollutant, the health impact of changes these pollutions were covered by Chapter 5.

VISUM was chosen as the traffic model, COPERT was utilized as the emission model, OSPM was employed as the dispersion model to evaluate the pollutant concentrations on roads, and a Gaussian Plume model in a finite line source form was used to estimate the air quality outside roads. BenMAP was utilized to incorporate the air quality and population to evaluate the health impact. Chapter 3, 4 and 5 include introductions for each specific strategy and further explanation of each models.

Chapter 3 An evaluation of the impact of the Dublin Port Tunnel and HGV management strategy on air pollution emissions

This chapter has been published entirely in the journal "*Transportation Research Part D: Transport and Environment*", Tang, J., McNabola, A., Misstear, B., & Caulfield, B. (2017).

Abstract

Heavy Goods Vehicles (HGVs) contribute a large proportion (about 40%) of the emissions of air pollutants while only representing a small proportion (about 10%) of all transport operations. In Ireland, the Dublin Port Tunnel (DPT) was opened in 2006 as a dedicated route for HGVs between Dublin Port and the motorway system in order to reduce the HGV volume in the city centre. An HGV management strategy to restrict HGVs travelling through the city centre was also introduced. The aim of this study was to estimate the emission changes brought about by these infrastructural and regulatory changes. A transport model built in VISUM was utilised. Emissions were calculated using COPERT 4. The results showed that the DPT and HGV management strategy reduced the traffic in the city centre, and the HGV management improved traffic speed distribution. However the DPT and HGV management resulted in HGVs travelling further (travel distance increased by 16% and 51%, respectively) and increased the total emissions (increased by 8% and 21% in NO_X, respectively). Total traffic and emission changes over time in Dublin were also estimated in this study. The traffic conditions and emissions in 2006, 2007 and 2013 were evaluated and the results indicated that a travel demand reduction in 2013 could also improve speed distribution. Emissions reduced from 2006 to 2013 and the fleet technology improvements had a positive impact on this reduction. The study shows that a traffic management policy and/or infrastructure change may bring about some localised environmental benefits within the management area; however, such a policy does not always reduce the total traffic emissions in the network as a whole and the impact to the wider environment could be negative in some circumstances.

3.1 Introduction

Traffic is one of the major users of energy and one of the major polluting sectors. It is considered a significant cause of the monitored exceedances of ambient air quality limit values in urban areas (EEA, 2013a). In 2011, the contribution of road transport emissions to nitrogen oxide (NO_x) and carbon monoxide (CO) in Europe amounted to 40% and 26%, respectively (EEA, 2013b). Traffic is also a major source of particle emissions (PM_{10} and $PM_{2.5}$) (Pant and Harrison, 2013).

Compared to industrial and other air pollution causes, air pollution from road transport is more likely to affect people, because the source of air pollution from transport, i.e. vehicles are often within close proximity to residential and workplace locations, in addition to exposure during commuting. Among all traffic modes, Heavy Goods Vehicles (HGVs) are a significant contributor to traffic pollution. An OECD report highlights that trucks can produce over 40 percent of the pollution where they only account for 10 percent of all transport operations in urban areas (OECD, 2003).

Governments all over the world are taking actions to reduce traffic emissions and to build a sustainable urban transport system. Some commonly considered options in cities for these purposes include: road infrastructure, public transport, technological solutions, vehicle access restrictions and control of land-uses (Pojani & Stead, 2015).

In cities, regulations aimed at restricting vehicle access have had an important impact on traffic emissions and air quality. In London, a low emission zone (LEZ) was implemented in 2008, which restricted some vehicles entering the zone. Ellison et al. (2013) concluded that the LEZ may have reduced PM_{10} emissions by 2.47-3.07% within the zone whereas by only 1% outside the zone. In Munich, after the implementation of a LEZ, PM_{10} concentrations in the LEZ were found to be reduced by 5-12% at almost all the monitoring sites (Cyrys et al., 2009). In China, some cities have implemented a license plate restriction policy, which prohibit a portion of cars entering the restriction zone at a particular time. Pu et al. (2015) found that in the license plate restriction zone in Hangzhou city in China, emissions decreased by 6.9%.

Transport infrastructure changes influence emissions and air quality as well as influencing traffic flows. Lozano et al. (2014) found a new toll highway had positive effects on emissions in the short term in Mexico City. Bandeira et al. (2013) estimated the emissions impact of vehicles choosing different routes. They found that faster intercity routes tended to reduce fuel use and CO_2 emissions, however they increased emissions of carbon monoxide, nitrous oxides, and hydrocarbons by up to 150% (Bandeira et al., 2013).

Focusing on Ireland, the national government proposed a set of strategies to reduce transport air pollution and CO_2 emission, including: regulating vehicle standards, implementing compulsory measures to restrict large vehicles, encouraging people to shift transport mode, and launching large infrastructural projects (Department of the Environment Heritage and Local Government, 2007).

In Dublin city, strategies taken that have had impacts on traffic emissions include improved infrastructure and vehicle restriction access: the Dublin port tunnel (DPT) was opened on December 20th 2006 as a dedicated route for HGVs between Dublin Port and the national road network to remove trucks from the city centre. An HGV management strategy was introduced on the 19th of February 2007 in Dublin. This strategy implemented a ban on 5+ axle HGVs prohibiting them from entering the city cordon area (roads that within the cordon area are shown in Figure 3-1 in red). It aimed to encourage maximum use of the Port Tunnel by port related traffic and thus to minimize the numbers of trucks on the city streets. This would in turn enhance the city centre environment through reduced congestion, noise and air pollution (O'Brien and Bolger, 2009).

Significant changes in traffic and air quality have been observed after the opening of the DPT and the implementation of the HGV management strategy. Decreases in 5+ axle vehicles of between 33% - 90% were recorded in the city centre. There were also reductions of 3 axle and 4 axle vehicles (Finnegan et al., 2007). Three years after the operation of the HGV Management Strategy dramatic reductions of 5+ axle vehicles were also observed within the city centre, between 88-96% and over 3,582 5+ axle vehicles used the tunnel per day in 2009 (O'Brien and Bolger, 2009). An air quality monitoring site installed in the city centre witnessed a 26% drop in average daily PM_{10} concentration after the opening of the DPT, from 35.5ug/m^3 to 26.2ug/m^3. It also recorded a 36% decrease in average daily PM_{10} concentration after the introduction of the HGV management strategy, from 35.5ug/m^3 to 22.7ug/m^3 (Finnegan et al., 2007). However, this monitoring site only recorded PM_{10} concentrations over a short period of three months at a fixed location within the city centre. The full impact of this infrastructure and regulatory change on air quality and emissions in the city of Dublin as a whole is not fully understood. Since a significant traffic change has been witnessed after the opening of DPT and the implementation of the HGV management strategy, a holistic evaluation of the emission change brought by this traffic change is needed to improve our understanding of the impacts of infrastructure and regulatory changes on air quality.

The DPT and HGV strategy had significant impacts on vehicle route choices and traffic conditions, as the DPT provided an extra route and the HGV strategy restricted some other routes. As one of the design functions of the DPT and the HGV strategy was to improve the environment of the city centre, it is important to know how these perform regarding reducing emissions.

In this paper an evaluation of the impacts of changes in transport infrastructure and policy on air pollution emissions was conducted. The evaluation focused on the role of HGV transport in the urban environment and highlights methods to improve environmental impact and adjust existing policy. Macroscopic traffic models and emission models were used to evaluate the impact of the DPT and a proposed new HGV management strategy on total emissions in the Dublin city region. Since its inception the HGV strategy has brought about changes in traffic and air quality, and it has been proposed that the strategy would be extended to include all types of HGVs (Finnegan et al., 2007). The current study estimated the impact of such a scenario where all HGVs are prohibited from entering the city centre. This study also examined the traffic and emission changes over time in order to evaluate the impact of different travel demands and the effect of vehicle technology improvements alongside infrastructure and regulatory changes. The paper aims to develop appropriate policy suggestions regarding reducing traffic congestion and emissions through the evaluation of these impacts.

3.2 Methodology and data

The traffic change brought about by the DPT and a proposed HGV management strategy was simulated using a traffic model built in VISUM (PTV company, 2014). The emission change brought about by this traffic change was then estimated using COPERT 4 (Leon and Zissis, 2014). The data requirements and sources for the traffic and emission models are summarized below:

1. Road network information was derived from the map in VISUM and the National Transport Model (NTpM) of Ireland (NRA, 2014; PTV company, 2014);

2. Origin-Destination matrices were extracted from the National Traffic Model (NTM) of Ireland and NTpM (NRA, 2014);

3. Traffic count records were obtained from Dublin City Council (DCC);

4. Fleet composition data were obtained from the Environmental Protection Agency of Ireland (EPA, 2015).

The NTpM has been constructed and updated in 2011 and 2013 respectively by Irish National Roads Authority (NRA, 2014). However, NTpM being an all-Ireland multi-modal transport model was calibrated and validated at a strategic level and therefore the model required recalibration in order to reflect accurately the situation at the regional level in Dublin city (NRA, 2014).

3.2.1 Traffic model

In order to simulate the traffic change brought about by the DPT and a proposed HGV management strategy, and also to investigate the effects of traffic and emission changes over the time, five scenarios in the Dublin area were included in this paper. Four scenarios were based on real conditions and one scenario was hypothetical. These scenarios were simulated by the traffic model and were compared with each other, as shown on Table 3-1 below:

No.	Scenario	Description
1	2006 no DPT	Taking one average hour traffic from 7 to 9am from 1/1/2006 to 20/12/2006 for the whole year of 2006
2	2006 DPT	Taking one average hour traffic from 7 to 9am from 21/12/2006 to 19/02/2007 for the whole year of 2006
3	2006 DPT+Ban	Hypothetical scenario, taking one average hour traffic volume from 7 to 9am from 21/12/2006 to 19/02/2007 for the whole year of 2006
4	2007 DPT+Ban	Taking one average hour traffic from 7 to 9am from 20/02/2007 to 31/12/2007 for the whole year of 2007
5	2013 DPT+Ban	Taking average hour traffic from 7 to 9am across 2013 for the year of 2013

Table 3-1: Times and dates represented by 5 scenarios. DPT+Ban represents DPT & HGV management scenarios.

These scenarios contained the following conditions: 1) before the tunnel was opened, i.e. 2006 no DPT; 2) after the tunnel was opened but before the HGV strategy was implemented, i.e. 2006 DPT; 3) after the HGV strategy was implemented, i.e. 2006 DPT+Ban, a hypothetical scenario; and 4) and 5) the 2007 and 2013 DPT+Ban scenarios. The traffic model simulated one average hour from the period of the AM peak (7:00-9:00am). Scenarios in the years 2006, 2007 and 2013 utilized the traffic data and fleet composition data of the year 2006, 2007 and 2013 respectively. The traffic model consisted of a network model based in VISUM, a demand model and various impact models.

3.2.1.1 The network model

Figure 3-1 below displays the network model built in VISUM representing the nodes, zones and links in Dublin city. The links in the network included the main roads in Dublin, i.e. motorways, national roads, regional roads and some local roads. As small roads and alleys had limited capacity for allocating trips, and thus little impact on the final results, these were excluded. 764 links were included in this study.

Travel information was recorded according to electoral divisions (EDs) from census data in Ireland. The Dublin area was divided into 30 zones according to EDs or amalgamations of EDs. The model also included internal zones covering the modelled area and external zones representing travel between the modelled area and the rest of Ireland.

For the scenarios before the DPT was opened, the network models did not have the link of the DPT. For the scenarios after the HGV strategy was implemented, a proposed HGV management strategy was used to evaluate the impact of a full HGV ban on the city centre. This prohibited HGVs entering the cordon area in the city centre. Links within the cordon area were blocked for HGVs in the network models, as shows in Figure 3-1.

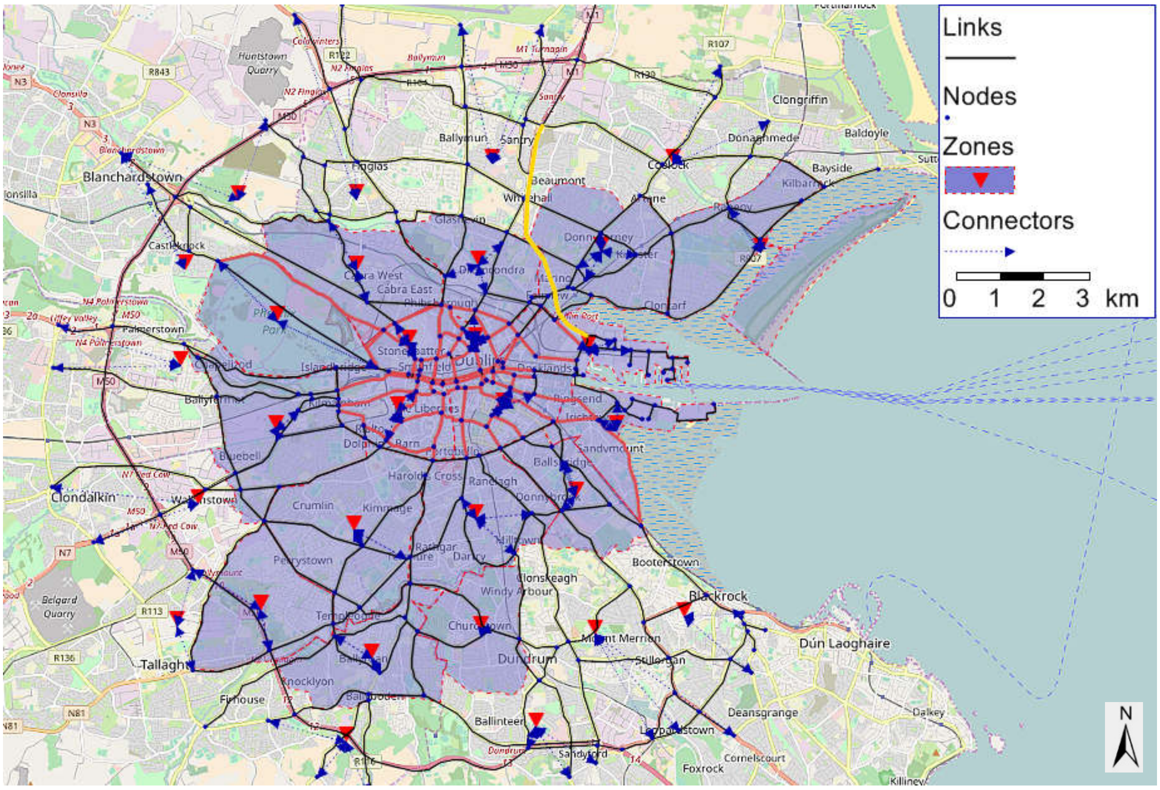

Figure 3-1: The VISUM network and prohibited links for HGVs within cordon area (red links). The yellow link represents the location of the DPT.

3.2.1.2 The demand model

The demand model contained the travel demand data in the form of trip matrices information for cars and HGVs, derived from NTM and NTpM. For 2006 scenarios, the trip matrices, i.e. Origin-destination (OD) matrices information for the year 2006 were used and calibrated with the traffic count data of 2006. For the scenario 2007 DPT+Ban, trip matrices for 2006 were used and calibrated according to the traffic count data of

2007. For the 2013 DPT+Ban scenario, trip matrices for the year 2013 was used and calibrated with the traffic count data of 2013.

The trip matrices for cars and HGVs in 2006 were extracted from NTM. NTM is a strategic traffic model of Ireland's national road network. The data used for travel matrix development for light vehicles and heavy vehicles of NTM were observed trip data obtained from road side interviews (RSI). It also made use of information contained in an Irish census of 2006 Place of Work - Census of Anonymous Records (POWCAR). NTM modelled an average hour in the morning peak between 07:00 and 09:00 (AM hour) for the base year 2006 (NRA, 2009).

A modified version of the NTM was used during this study. It was downscaled to 30 zones according to the zones of the study area. The zones outside the Dublin city in the NTM model were aggregated and regarded as an external zone representing the travel between the Dublin city and the rest of Ireland.

The trip information for 2013 was obtained from NTpM which was established in 2011 and updated in 2013. NTpM is the successor of the NTM. The demand of NTpM for travel by car and HGVs was constructed based on data from the: 2011 POWCAR survey (CSO, 2011); 267 NRA Traffic Monitoring Units; Surveys and information from the previous version of the NTM; National Survey of Transport of Goods by Road by CSO; and various traffic survey data collected by the NRA in 2013 (NRA, 2014).

An extracted model of NTpM was also downscaled to 30 zones according to the zones of the study area. The zones outside Dublin city in the extracted NTpM model were amalgamated and regarded as external zones representing the travel between the Dublin city and the rest of Ireland.

After trips of cars and HGVs were assigned to roads, trip matrices were calibrated to adjust the assigned volume to real traffic counts. Because the NTM and NTpM are national models which reflect the trips at a national level rather than a local level, re-calibration against traffic count data within Dublin city was conducted.

3.2.1.3 Calibration

The data for calibrating the demand model was obtained from Dublin City Council and the National Roads Authority. The TFlowFuzzy technique was used to calibrate OD matrices (PTV company, 2014). The calibration process was designed to automatically manipulate the OD matrices to match a counted volume along a particular link or multiple links, making the difference between the assignment volume and the actual volume less than a tolerance. Detailed process of TFlowFuzzy is described in 4.2.1.1.

As this study simulated situations for one average hour from the period of the AM peak, the traffic count data being used was correspondingly the traffic volume of one average AM hour for the year. As mentioned in Section 3.2.1, scenarios in the years 2006, 2007 and 2013 utilized the traffic counts data of the corresponding year. As the scenario of 2006 DPT existed for a very short time (about 11 days in the end of 2006), the annual traffic counts for 2006 actually represents the conditions of 2006 no DPT scenario. Therefore the 2006 no DPT scenario was calibrated with 2006 traffic counts, and the 2006 DPT scenario and the hypothetical 2006 DPT+Ban scenario utilized the same OD matrices as the 2006 no DPT scenario.

The UK Design Manual for Roads and Bridges (UK DMRB, 1997) specifies the acceptable values for modelled and observed flow comparisons and suggests how calibration should be conducted in this context. These calibration criteria were observed in this study and are summarized in Table A-1 in Appendix A. 17 links selected to calibrate the model, based on available data, as shown in Figure 3-2. The modelled traffic volumes were set to meet the criteria using the TFlowFuzzy approach.

3.2.1.4 Validation

Validation used independent traffic data from that used in the calibration process. Validation criteria for tolerance implemented in this study were the same as the calibration criteria. For each link, the validation process checked whether the difference between the assignment volume and the actual volume was within the tolerance range, which is regulated by Table A-1 in Appendix A.

Five links were selected for validation and all of these links met the validation criteria. Figure 3-2 also shows the links that have met the validation criteria.

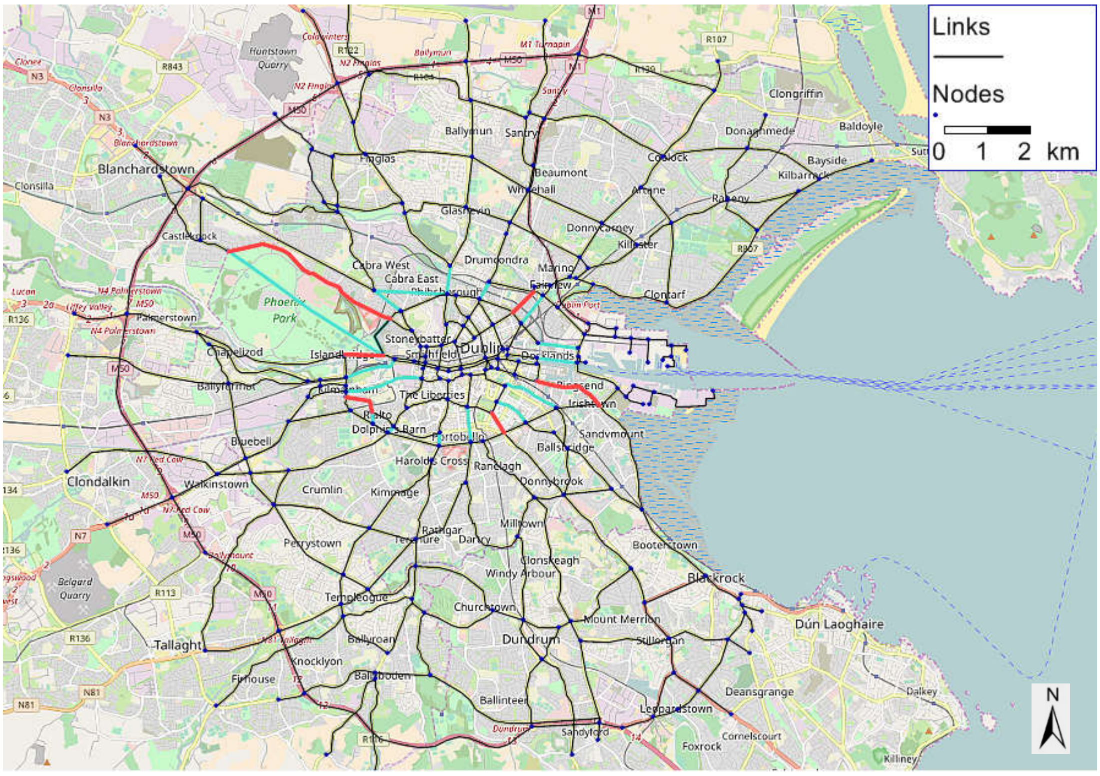

Figure 3-2: Links of calibration (blue line) and links that met the validation criteria (red line).

3.2.1.5 The impact model

The impact model used input data provided by the network model and the demand model to calculate the impact of traffic in order to analyse and evaluate transport supply. The attributes of network objects such as the vehicle congested speed on each link and the length of each link, were calculated and used in the subsequent emission calculations.

3.2.2 Emission model

Emissions were estimated using COPERT 4. The emission factors were obtained from those used in the COPERT 4 emission model. COPERT has been developed for official road transport emission inventory preparation in EEA member countries. These emission factors are suitable for EU conditions. The COPERT methodology is part of the EMEP/EEA air pollutant emission inventory guidebook for the calculation of air pollutant emissions and is consistent with the 2006 IPCC Guidelines for the calculation of greenhouse gas emissions (Leon and Zissis, 2014). Therefore, it is appropriate to use COPERT 4 to estimate emissions in Irish conditions.

Emissions can be classified into exhaust emissions and evaporative emissions for non-methane volatile organic compounds (NMVOCs). For particulate matter (PM) road vehicle tyre/brake wear and road wear caused by vehicle's motion emissions are also included (Leon and Zissis, 2014).

For exhaust emissions, these were calculated as the sum of hot emissions (when the engine is at its normal operating temperature) and emissions during transient thermal engine operation (termed 'cold-start' emissions). As the trip amount and the type of vehicles are not affected by the opening of the DPT, and the simulation hour of different scenarios are all in December and thus have a similar temperature, the effect of cold emission was not considered.

Emissions were calculated by combining activity data for each vehicle category with appropriate emission factors. These emission factors are pertinent to the technology standards available in Ireland (e.g. EURO Class 1, 2, 3, 4, 5, 6) and the classification of the Irish vehicle fleet was also conducted according to these technology standards. The emission factors also varied according to other input data (driving situations, climatic conditions).

The number of vehicles per category and class was obtained from national vehicle fleet composition data. National fleet composition of 2006, 2007 and 2013 were derived from Environmental Protection Agency of Ireland (EPA, 2015) which included the number of vehicles for every exhaust emission legislation class of each vehicle category. In this study, it was assumed that the proportion of each vehicle category and class to the total amount of vehicles within simulation area equalled the proportion of that for Ireland as a whole. More than a quarter of the population of Ireland is located in the Dublin city, and Dublin as a sample of Ireland as a whole, is a quite large sample. The Irish national fleet therefore represents the fleet composition in Dublin very well. Details of the National fleet composition of 2006, 2007 and 2013 are listed in the in Table A-2 in Appendix A.

Not taking evaporative and cold-start emissions into account, the formula for estimating emissions for a given time period, and using experimentally-obtained emission factors, is expressed by Equation 3-1:

Emission [g] = emission factor [g/km] × number of vehicles [veh] × distance per vehicle [km/veh]

Equation 3-1: Hot emissions calculation.

Different emission factors, numbers of vehicles and distance per vehicle were present for each vehicle category and class. The emission factors depended on the vehicle class and emissions control technology the vehicle applies. Within each vehicle class, the emission factor also depended on the vehicle speed (see Chapter 2, section 2.4.2.1 for more details of the emission calculation and emission factors). The information for vehicle speed and distance per vehicle were derived from the traffic model.

For each vehicle class, the two items on the right side of Equation 3-1 of *number of vehicles [veh] (in the traffic model) × distance per vehicle [km/veh] (in the traffic model)* represented the total distance that vehicles of this class travelled in the traffic model. In this study, because the total distance travelled by cars and HGVs respectively could be derived from the traffic model and the proportion of each vehicle class in the simulation area was assumed to be the same as the national fleet composition, these two items for each class were expressed by Equation 3-2.

$$\#\ of\ vehicles \times distance\ per\ vehicle = total\ distance \times \frac{\#\ of\ vehicles\ of\ a\ class}{total\ \#\ of\ vehicles}$$

Equation 3-2: Distance travelled by one class of vehicles calculation.

Where, the total distance refers to the total distance that every vehicle travelled in the traffic model; the total number of vehicles refers to the total number of vehicles in the Irish fleet; and the number of vehicles of a class refers to the number of vehicles with the same technology in the Irish fleet.

In Equation 3-2, the vehicles were divided into two general types, cars and HGVs. For each type there were different categories and classes. Total distance was calculated for each type respectively.

Vehicle speed was introduced into the calculation via three average speeds and their shares, which influenced the emission factor. For cars, the speed range and average speed are displayed in Table 3-2 as follows:

Group	1	2	3
Speed range (km/h)	5 - 35	36 - 65	66 - 95
Average speed (km/h)	20	50	80

Table 3-2: Average speed for each speed range for cars.

For HGVs, because the speed limit on the motorway was 90 km/h (LEO, 2012), the average speed and speed range were 78 km/h and 66-90 km/h respectively. The speed range and average speeds are displayed in Table 3-3.

Group	1	2	3
Speed range (km/h)	5 - 35	36 - 65	66 - 90
Average speed (km/h)	20	50	78

Table 3-3: Average speed for each speed range for HGVs.

The percentage distance that a vehicle driving with a speed within each speed range was given by Equation 3-3.

Share = Distance vehicles travelled with the speed within the range of each group [km] / total distance vehicles travelled [km]

Equation 3-3: Distance percentage calculation for each speed range.

Equation 3-3 was applied to each vehicle category and class. Different emission factors, numbers of vehicles and distance per vehicle, for each vehicle category and class were calculated with the data and method mentioned above, for each of the five scenarios examined. Emissions of CO, CH_4, NO_X, PM, CO_2 were calculated.

3.3 Results

3.3.1 Traffic modelling

Differences in traffic between different scenarios are displayed in Figure 3-3 to Figure 3-6. These figures show the results of the traffic volume difference for each link between a number of the scenarios, as follows:

- scenario 2006 DPT minus 2006 no DPT (Figure 3-3), (scenario No. 2 minus No. 1), representing the impacts of the DPT
- 2006 DPT+Ban minus 2006 DPT (Figure 3-4), (scenario No. 3 minus No. 2), representing the impacts of the HGV management strategy
- 2007 DPT+Ban minus 2006 DPT+Ban (Figure 3-5), (scenario No. 4 minus No. 3), representing the traffic differences between 2007 and 2006 under the implementation of the DPT and HGV management strategy
- 2013 DPT+Ban minus 2007 DPT+Ban (Figure 3-6), (scenario No. 5 minus No. 4), representing the traffic differences between 2013 and 2007 under the implementation of the DPT and HGV management strategy

The volume differences for cars are represented by a red bar. Light red shows that the volume difference is less than 0 and dark red shows the opposite. For example, in Figure 3-3, links with dark red bars represent the links where the traffic volumes in the scenario 2006 DPT are more than in 2006 no DPT. The width of the link bar indicates the magnitude of volume difference. The blue bar on each link represents the volume difference for HGVs. Light blue shows that volume difference is less than 0 and dark blue shows the opposite.

Figure 3-3 indicates that after the opening of the DPT, cars and HGVs in the city centre decreased by approximately 400-500 cars/hr and around 100-200 HGVs/hr. In the DPT, cars and HGVs increased by about 2000-2500 cars/hr and 800-900 HGVs/hr. Cars and HGVs travelled through the DPT and avoided travelling through the city centre, therefore traffic volume (veh/hr) in the city centre decreased.

Figure 3-4 indicates traffic changes for the hypothetical scenario 2006 DPT+Ban minus the scenario of 2006 DPT. This result allows us to see the traffic changes brought about by the HGV management implementation. It shows a decrease of the HGV volume by about 100-200 HGVs/hr within the city centre and a rise of HGV volume on the DPT and some links outside the city centre by around 200-400 HGVs/hr. On the M50, a motorway that encircles Dublin, the HGV volume increased by about 200-800, demonstrating that the HGVs travelled to the motorway network through the DPT without travelling through the city centre. This shows that the HGV management changed the route travelled by HGVs but had little impact on cars.

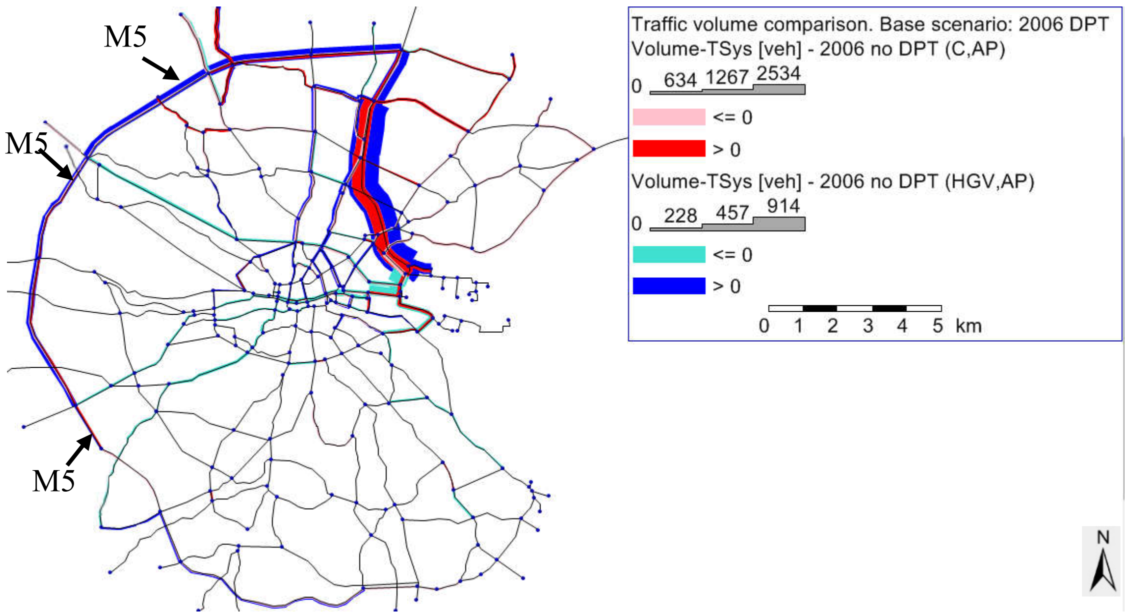

Figure 3-3: Traffic volume comparison. 2006 DPT minus 2006 no DPT.

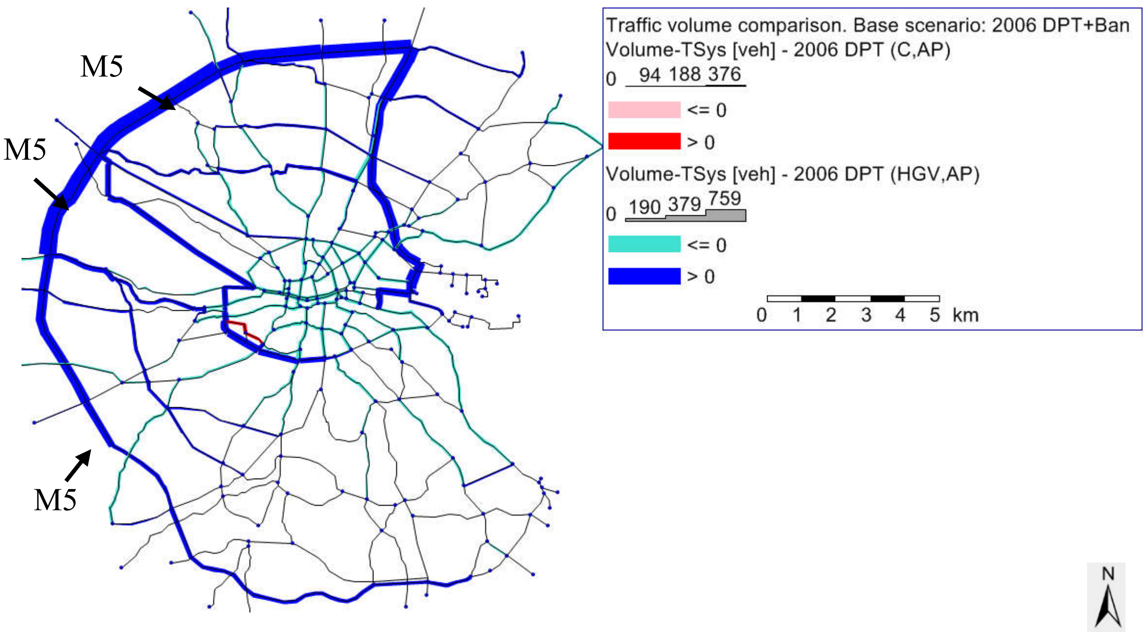

Figure 3-4: Traffic volume comparison. 2006 DPT&HGV strategy minus 2006 DPT.

Figure 3-5 shows scenario 2007 DPT+Ban minus 2006 DPT+Ban. It represents the traffic difference of a real situation which took the travel demand changes between 2006 and 2007 into account. There were rises and drops for traffic volumes for different links, with changes of less than 1000 cars/hr and less than 100 HGVs/hr.

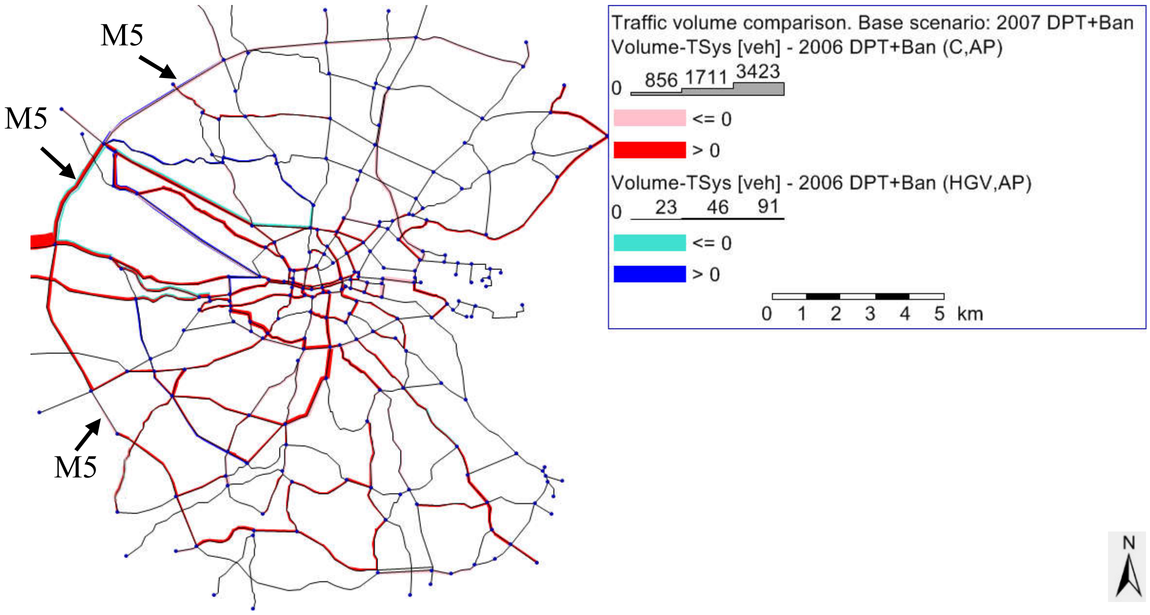

Figure 3-5: Traffic volume comparison. 2007 DPT&HGV strategy minus 2006 DPT&HGV strategy.

Figure 3-6 displays traffic volumes of 2013 DPT+Ban minus 2007 DPT+Ban, illustrating the impact of the infrastructure and policy change under significantly different travel demand and vehicle fleet conditions. A quite obvious decrease of HGVs can be seen in the DPT and on the M50. This is due to the drop of the total HGV travel demand in 2013 which can also be seen in the Figure 3-7.

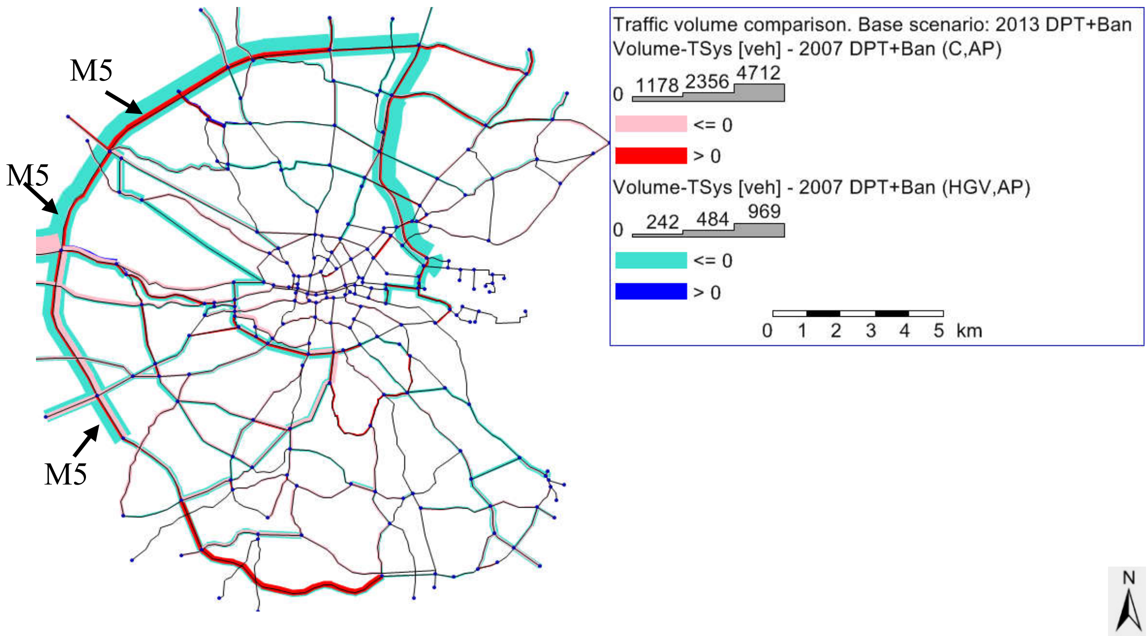

Figure 3-6: Traffic volume comparison. 20013 DPT&HGV strategy minus 2007 DPT&HGV strategy.

Figure 3-7 shows the total distance travelled by every vehicle (i.e. vehicle kilometers travelled) and the direct distance for all the trips in the five different scenarios. Direct distance is referred to here as the sum of each trip multiplied by the direct distance from origin to destination of this trip. It represents the shortest distance to be travelled for a given OD matrix. Figure 3-8 shows the ratio of the total distance to the direct distance. A larger ratio means that on average vehicles had to detour more, i.e. vehicles had to travel more for a given origin and destination.

As can be seen in Figure 3-7, the total distance rose for cars and HGVs after the opening of DPT, from 917713km to 917663km and from 72755km to 84605km, respectively. The implementation of the HGV management strategy also caused the total distance to rise further, from 917663km to 917673km and from 84605km to 110012km for cars and HGVs respectively. On the other hand, the direct distance required to meet travel demands for these three scenarios were the same, because these scenarios all used the same OD matrices. In the scenario 2007 DPT+Ban, total distance and direct distance all increased while in the scenario of 2013 DPT+Ban the total distance and direct distance all decreased. As direct distance can reflect the level of travel demand, we can infer from the figure that the travel demand increased in 2007 and decreased in 2013.

As shown in Figure 3-8, ratio of total distance to direct distance didn't change after the DPT was opened and after the HGV strategy was implemented. In the 2007 and 2013 scenario, this ratio only had a very small change for cars compared to the ratios for HGVs. This shows that the DPT and the HGV management strategy did not have a significant impact on cars. However, for HGVs, this ratio increased from 1.50 to 1.74 and then surged to 2.27 after the opening of the DPT and the implementation of the HGV strategy. This was an increase of by 16% and 51%, respectively. In the 2007 and 2013 scenarios, the ratio for HGVs were both higher than 2.0. These ratios show that the DPT and the HGV strategy affected the route of HGVs a lot and resulted in HGVs detouring a lot.

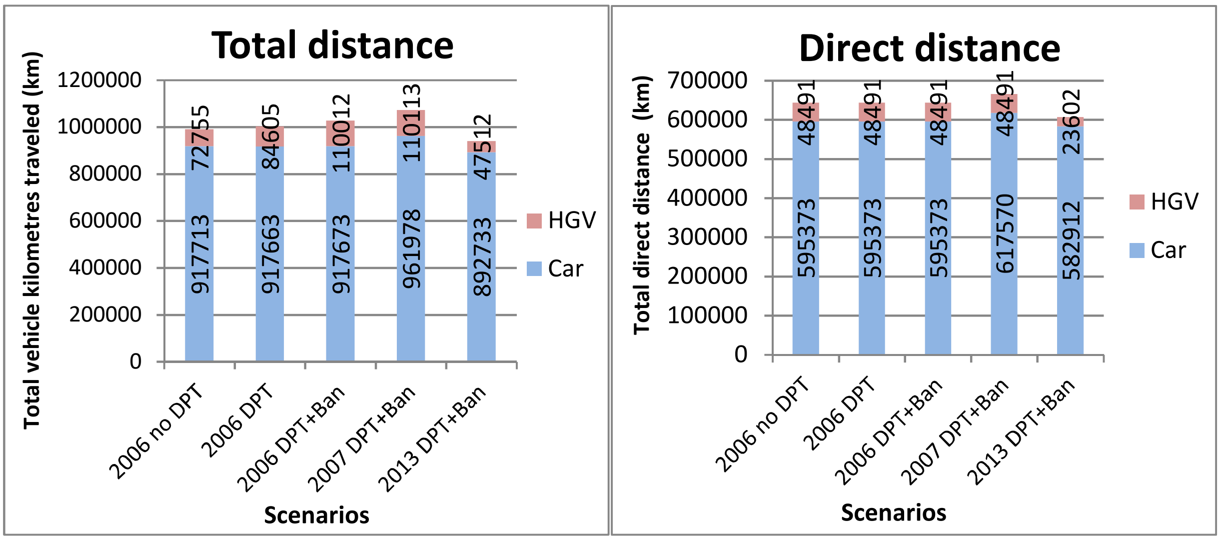

Figure 3-7: Total distance and direct distance.

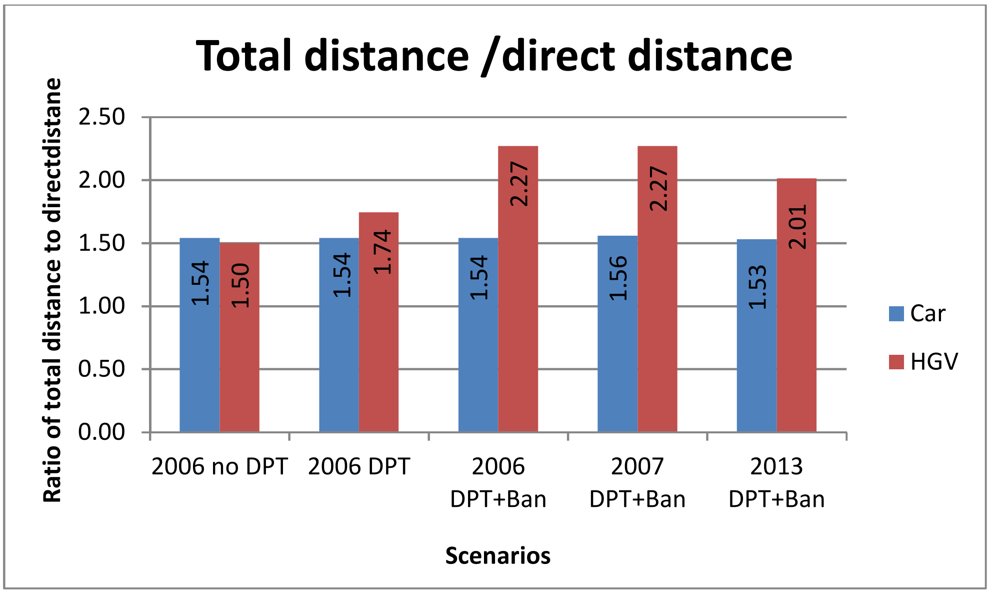

Figure 3-8: The ratio of total distance to total direct distance.

The speed distributions of the different scenarios are depicted in Figure 3-9. Comparing 2006 no DPT, 2006 DPT, 2006 DPT+Ban and 2007 DPT+Ban scenarios, it can be observed that the proportions of low speed range (5-35 km/h) for cars for all these four scenarios are around 0.53-0.55. However, for HGVs, the proportions of low speed range declined after the HGV management, from about 0.46-0.48 for the former two scenarios to about 0.33-0.36 for the latter two scenarios. This means the latter two scenarios have higher proportions of middle and high speed range (35-65 and 65-95 km/h, respectively) so that there was less congestion for HGVs after the implementation of the HGV strategy. This implies that the DPT with the HGV management strategy together has a function to improve the average speed of the network by reducing congestion in the city centre, which would have obvious air pollution benefits. As for the scenarios in 2013, the proportion of middle and high speed ranges rose for cars and HGVs compared to previous scenarios, from more than 0.52 to 0.40 for cars and from more than 0.32 to 0.20 for HGVs. This was due in part to the fact that travel demand dropped in 2013 as we discussed earlier, due to economic factors.

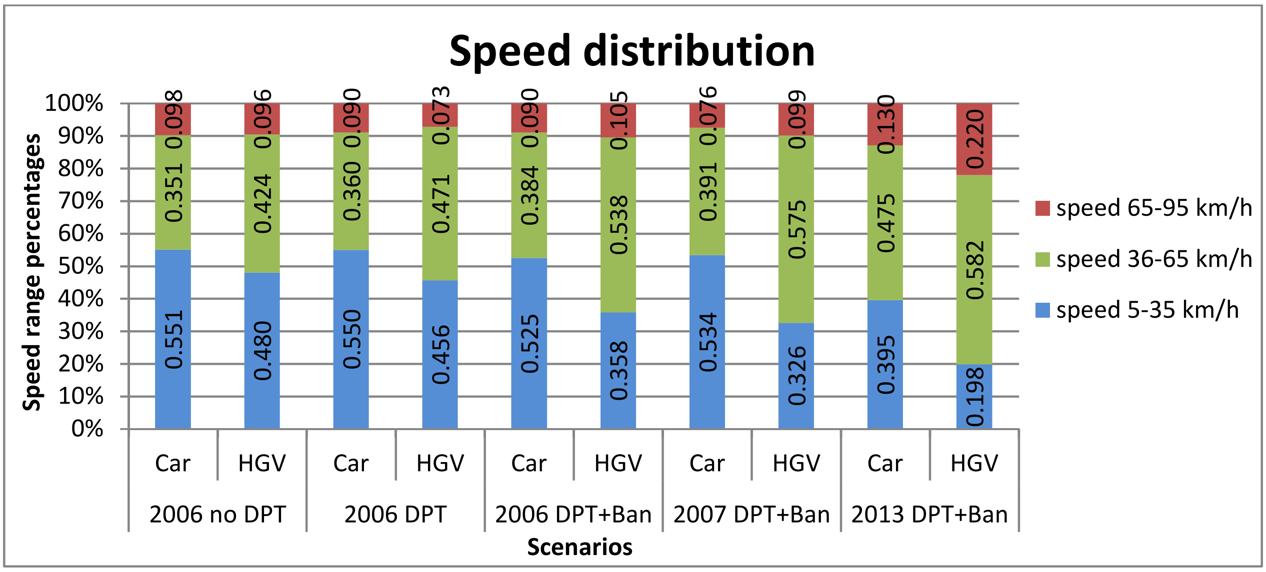

Figure 3-9: Speed distribution.

3.3.2 Emissions calculation

Figure 3-10 illustrates the total traffic emission trends for all scenarios. In the 2006 scenarios, in general, all pollutions were increased after the DPT was opened and after the

HGV management was implemented. Using NO_X as an example, the emission was 719.05 kg in the 2006 no DPT scenario. Then it increased to 775.36 kg in 2006 DPT scenario and again increased to 869.28 kg in the DPT+Ban scenario. It increased by 8% and 21%. This implies that the DPT and the HGV strategy would make the total emission rise. This was due to the DPT and the HGV strategy forcing HGVs detour significantly during their trips. Although DPT with HGV strategy reduced the congestion in the city centre, the emission reduction brought about by this benefit was not compensated for by the emission increase brought about by the additional detours.

In the 2007 DPT+Ban scenario, the emission generally dropped for all pollutions except for CO_2. Using NOx and PM as examples, comparing to 2006 DPT+Ban scenario, the NOx and PM emissions declined from 869.28 kg to 839.65 kg and from 30.71 kg to 28.82kg, which are 3% and 6% decrease, respectively. However, emission of CO_2 rose from 240.75 t to 250.58 t (4% increase). The speed distribution of this scenario was similar to the 2006 DPT+Ban, but the total distance travelled by vehicles was more than the total distance in the scenario of 2006 DPT+Ban. The fleet distribution used in this scenario was for the year of 2007 while the 2006 DPT+Ban used 2006 fleet distribution. Comparing 2006 and 2007 DPT+Ban scenario, the similar speed distribution should have similar impact on emissions, and increased total distance should produce more emissions. Therefore, as only travel demand and fleet composition were changed between these two scenarios, we can infer that the reduction in emissions was mainly caused by the technology improvement of fleet between these two years. Comparing to other pollutants, the emission of CO_2 is less sensitive to the technology improvement and relies more on the travel distance.

The emissions in the 2013 scenario fell sharply. For example, for CH_4 and CO, the emissions of the 2013 scenario decreased to less than a half of the emissions of the 2007 scenario, from 8.07 kg to 2.74 kg and from 0.84 t to 0.36 t, which are 66% and 57% decrease, respectively. This could be a reflection of the joint impacts of the drop of total distance travelled, the improvement of speed distribution, and technology improvement of the fleet.

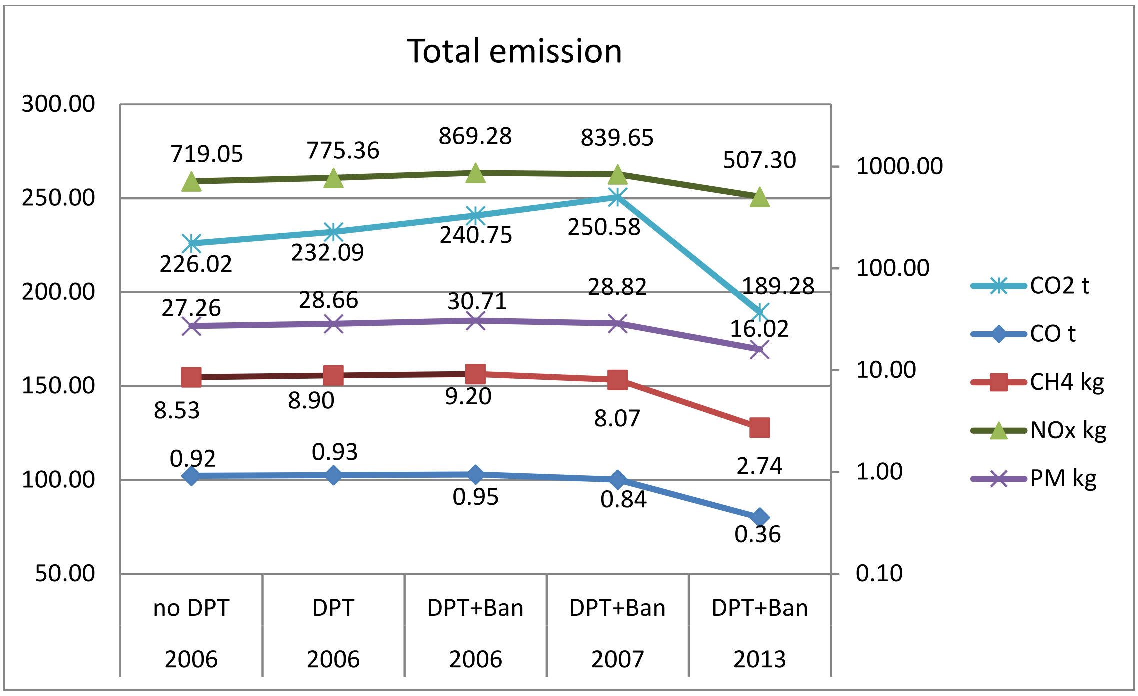

Figure 3-10: Total emission trend.

The standardized emissions shown here were calculated as the ratio of total emission to direct distance. The result of the standardized emissions is displayed in Figure 3-11.

Comparing the three 2006 scenarios (i.e. 2006 no DPT, 2006 DPT and 2006 DPT+Ban) with each other, we can find that emissions for all pollutants remained similar for cars, about 1.4 g/km for CO, 10.2 mg/km for CH_4, 0.6 g/km for NOx, 30.5 mg/km for PM, 317 g/km for CO_2. This was because the total distance travelled by vehicles and speed distributions were similar for cars across the scenarios and the fleet compositions were the same. However, although the three scenarios had the same fleet compositions for HGVs, the emissions for HGVs were different. Compared to the 2006 no DPT scenario, all the pollutants emissions were increased in the 2006 DPT scenario, and again increased further in the 2006 DPT+Ban scenario. Taking PM emissions from HGVs as an example, it was 185.8 mg/km in the 2006 no DPT scenario, 215.4 mg/km in the 2006 DPT scenario, and was 261.0 mg/km in the 2006 DPT+Ban scenario, which were 16% and 40% increases, respectively. Although the proportion of low speed range for HGVs declined a little in the scenario of 2006 DPT+Ban, which implies that the congestion for HGVs abated, the standardised emissions of the 2006 DPT+Ban scenario was still the highest among the three 2006 scenarios. From this we could infer that the total distance travelled by vehicles plays a major role in affecting the standardised emissions in our three scenarios. For a given origin and destination, the more a vehicle detoured, the more pollutions the vehicle emitted. The ratio of the total distance to the direct distance for HGVs were 1.50, 1.74, and 2.27 for 2006 no DPT, 2006 DPT and 2006 DPT+Ban scenario, respectively, equalling to 16% and 51% increases in 2006 DPT and 2006 DPT+Ban scenario comparing to 2006 no DPT scenario. Therefore, due to the significantly longer travel distances, the improvement of speed and the reduction of congestion in the city centre could not offset the extra emission brought on by the extra distance for 2006 scenarios.

Regarding the three DPT+Ban scenarios, the results showed a decline of the emissions from cars for CO, CH_4 and PM. For instance, the CO emissions for the scenario of 2006, 2007 and 2013 DPT+Ban were 1.4 g/km, 1.2 g/km (14% decrease) and 0.6 g/km (57% decrease) respectively. As for NO_X, the three scenarios had the same emissions of 0.6 g/km. The emissions of CO_2 for 2006, 2007 and 2013 scenario were 315.3 g/km, 321.2 g/km (2% increase) and 288.2 g/km (9% decrease) for cars. The ratio of total distance to the direct distance was similar for these three scenarios. The speed distributions for the scenarios of 2006 and 2007 DPT+ Ban were also similar and both of them had a larger proportion of low speed range vehicles than the scenario of 2013 DPT+Ban. Thus, part of the reason that the 2013 DPT+Ban scenario had lower standardised emissions could be due to the improvement of speed distribution in the network. Another reason could be that the vehicle fleet technology was improved.

Although the ratio of total distance to direct distance and the speed distribution in the 2006 DPT+Ban scenario were similar to 2007 DPT+Ban, the emission of CO, CH_4 and reduced. This could be due to the technology improvement in the vehicle fleet between 2006 and 2007. It can be seen that these reductions in emissions were quite modest at 3-6%.

For the emissions from HGVs, emissions of all the pollutants decreased from 2006. As the scenario of 2013 DPT+Ban had small ratio of total distance to direct distance, optimized speed distribution and improved technology for HGVs, all of these could result in the lowest standardised emission for HGVs among the DPT+Ban scenarios.

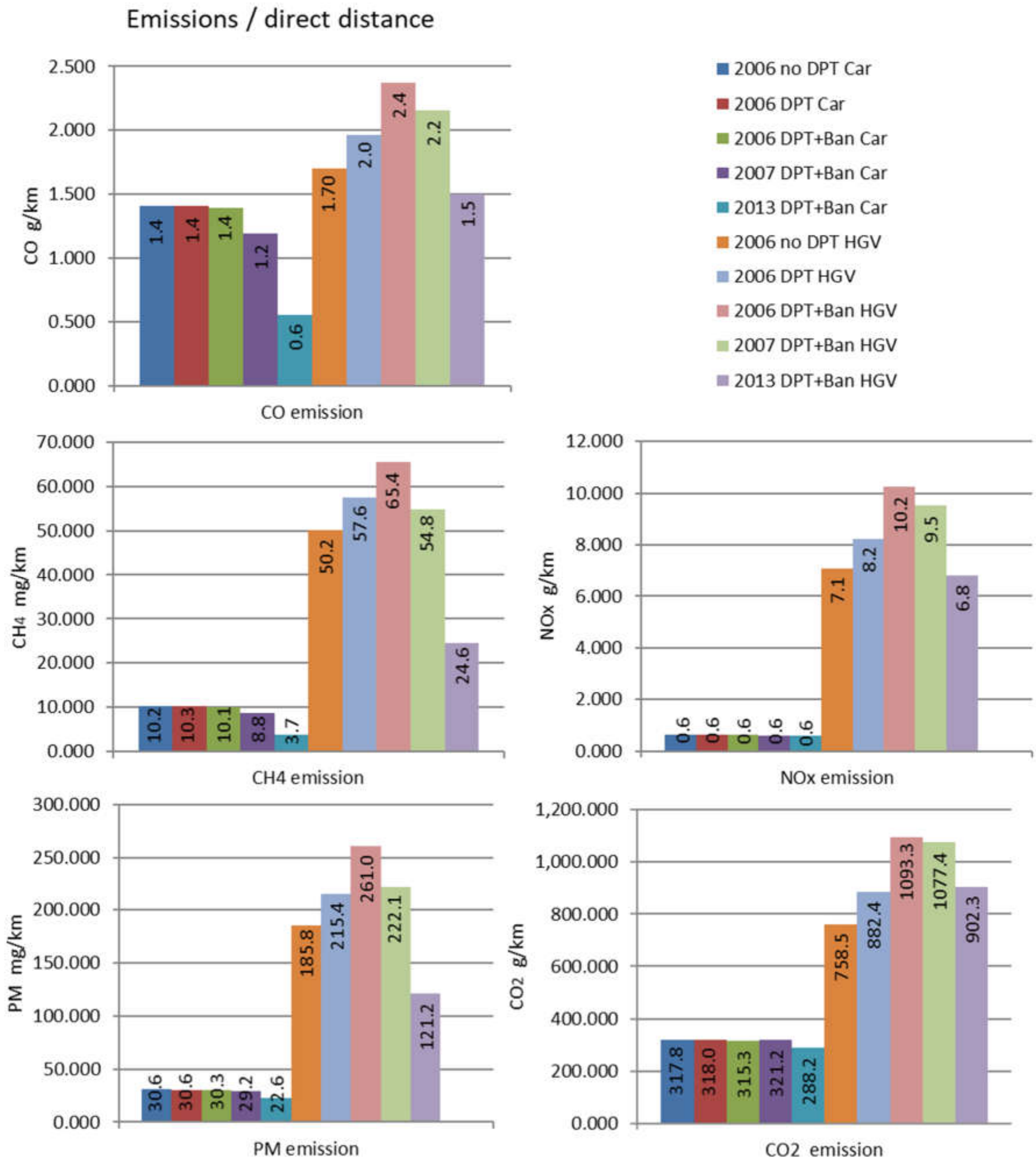

Figure 3-11: Standardized emissions for different scenarios.

3.4 Discussion

The impact of the DPT and HGV management strategy can be inferred from the comparison of the different scenarios simulated in this paper. The impact of DPT on traffic and emission in Dublin city were estimated by comparing the scenario of 2006 no DPT to 2006 DPT. The impact of HGV management strategy was evaluated by the comparison of the hypothetical 2006 DPT+Ban scenario and 2006 DPT scenario.

Scenarios for 2007 and 2013 were also simulated in the paper to investigate the effect in this context of emission changes over time and the impact of vehicle technology improvements.

From the comparison of the 2007 DPT+Ban scenario and the 2013 DPT+Ban scenario, we can infer that for the same road network capacity, less travel demand can improve speed distribution and average speed in the network. From this aspect, in the situations when the travel demand is difficult to reduce, it can be useful to construct more roads to improve the network capacity and thus reduce congestion. On the other hand, from the comparison of 2006 DPT and 2006 no DPT, a new road could cause vehicles to detour and travel longer. A trade-off between these two potential impacts brought by a new road should be carefully considered.

By comparing the three 2006 scenarios, we can infer that the DPT and the HGV management strategy both had an impact on the average distance travelled by vehicles. The HGV strategy could improve the speed distribution of vehicles in the network. However, the benefit of the emission reduction brought about by this improvement could not offset the disadvantage of the emission growth brought about by the longer travel distances. Therefore, in total, the DPT and the HGV strategy increased the total traffic emissions, considering the whole road network of the Dublin city.

By comparing the three DPT+Ban scenarios, we found that the vehicle fleet technology improvement from 2006 to 2013 could have a positive impact on emission reduction. As discussed in section 3.3.2, the standardized emission for HGVs for the scenario 2013 DPT+Ban was lower than 2007 DPT+Ban and HGV emission for the scenario 2007 DPT+Ban was lower than 2006 DPT+Ban. This could be caused by two reasons: vehicle emissions technology improvements and speed improvement. It can be seen in Table A-2, compared to 2006, the proportions of vehicles that applied Euro 3 and Euro 4 standards in 2007 and vehicles that applied Euro 4 and Euro 5 standards in 2013 have increased, while vehicle numbers applying Euro 1 and Euro 2 fell. Euro 3, 4, and 5 standards implement more strict emission controls than Euro 1 and Euro 2, and significant efforts have been made to reduce emissions from vehicles during this period (e.g. introduction of diesel particle filters).

An obvious impact of these emissions is the potential health effects on the urban population. As can be seen from our traffic model, after the opening of the DPT and the implementation of HGV management strategy, traffic reductions have been found in the city centre where population density is relatively high. Also, as mentioned in the introduction to this paper, an air quality monitor in the city centre witnessed a significant reduction in average daily PM_{10} concentration after the opening of the DPT and the implementation of HGV management strategy. From this aspect, the DPT and HGV strategy therefore could have resulted in some health benefits to the population. However, the total emissions after the opening of the DPT and the implementation of HGV management strategy were increased. Thus, it is difficult to assert whether the DPT and the HGV strategy have brought about benefits to health of the populations or not. Hence, the health impacts of the DPT and HGV management strategy are evaluated in Chapter 5. Such an outcome is a further example of the nexus between climate change and air pollution policy (Bollen & Brink, 2014), where in this case the regulatory and infrastructure changes implemented in Dublin have resulted in improved city centre air quality but at the cost of increased CO_2 emissions and potential climate change impacts, as well as increased air pollution outside the city centre.

Some cities like London have implemented low emission zones (LEZ) which charge vehicles that do not meet the LEZ emissions standards travelling within the LEZ. This policy can make some vehicles that are originally planning to travel within the LEZ choose another route, causing detours. The impact of such a policy is similar to the HGV management strategy which reduces the traffic within the strategy implementation area whilst causing some vehicles to travel for a longer distance. As mentioned in Section 3.1, the air quality improvements within the LEZ were observed after the implementation of LEZ. However, like the HGV strategy, if we consider the whole area that is potentially affected by the LEZ, it may not always the case that the LEZ will improve the environment as a whole. Dias et al. (2016) supported this finding when they modelled the emissions change that could be brought about by using an LEZ in the city of Coimbra in Portugal. They found that PM_{10} and NO_2 emissions from private cars would decrease significantly inside the LEZ (63% and 52%, respectively). However, in contrast, total emissions would increase and a deterioration of air quality was expected to occur at city level.

This chapter showed that infrastructural or policy changes that affect traffic flow should be appraised for their effects on the road network as a whole. Emissions from detours are a major factor that could influence the outcome of policy or infrastructure change.

Examples of these changes include new road construction, LEZ, tolling systems, license plate restriction policy, HGV management strategy, and so on. This paper also demonstrated an appraisement method that could be used when estimating the traffic and the emission changes of a city.

3.5 Conclusion

According to the results of this study, the opening of the DPT and the introduction of the HGV strategy influenced the speed distribution and travel distance of vehicles, which all have an impact on the emission of air pollutants. While the HGV management strategy improved speed distributions, it made HGVs travel further as the ratios of total distance to direct distance increased.

From the study results, it can be seen that the DPT and HGV management strategy could reduce the traffic in the city centre while also forcing vehicles to travel further, thereby increasing the total emissions. The HGV management could improve the overall speed distribution of the network. Travel demand reductions from 2006 to 2013 could also improve the speed distribution. Emissions reduced significantly from 2006 to 2013 and fleet technology improvements could have a positive impact on this reduction.

Regarding the emissions, the construction of a new road should be appraised carefully. An analysis of the traffic condition of the existing road network is necessary. For a heavily congested city, the benefit of speed optimisation brought about by a new bypass may surpass the disadvantage of adding extra distance for the same trip. For a city which does not have heavy traffic, a bypass could bring about a negative effect.

A bypass that helps to reduce the emissions in a city centre location, for example, may not reduce the total emissions in a network as the travel demand remains, resulting in detours. The findings of this study are also relevant to other transport policies whereby travel restrictions are placed on small areas of a road network (e.g. the city centre) with the potential to cause significant detouring of traffic, such as low emissions zones.

Chapter 4 Assessing the impact of vehicle speed limits and fleet composition on air quality near a school

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Abstract

Traffic is a major source of urban air pollution that affects health, especially among children. As lower speed limits are commonly applied near schools in many cities, and different governments have different policies on vehicle fleet composition, this research estimated how different speed limits and fleet emissions affect air quality near a primary school. Based on data of traffic, weather and background air quality records in Dublin from 2013, traffic, emission and dispersion models were developed to assess the impact of different speed limits and fleet composition changes against current conditions. Outside the school, hypothetical speed limit changes from 30km/h to 50km/h could reduce the concentration of NO_2 and PM_{10} by 3% and 2% (corresponding to 22% and 15% of traffic-induced NO_2 and PM_{10} reduction); shifts in the fleet from diesel to petrol vehicles could reduce these pollutants by 4% and 3% (corresponding to 35% and 22% of traffic-induced NO_2 and PM_{10} reduction), but would increase the traffic-induced concentrations of CO and Benzene by 63% and 35%. These changes had significantly larger impacts on air quality on streets with higher traffic volumes and hence higher pollutant concentrations. Findings suggest that both road safety and air quality should be considered when determining speed limits; besides, fleet composition has different impacts on different pollutants and there are no clear benefits associated with incentivising either diesel or petrol engine vehicles.

4.1 Introduction

Research has shown significant associations between traffic pollution and adverse human health effects related to lung, heart, psychological and other body systems, e.g. increasing lung cancer, heart disease, dementia, and other health problems (Rosenlund et al., 2008; Health Effects Institute, 2010; Andersen et al., 2012; Estarlich et al., 2016; Raaschou-Nielsen et al., 2016; Bowatte et al., 2017; Chen et al., 2017b). Air pollution is a major environmental problem that causes 6.4 million premature deaths worldwide per year, which is 72% out of 9 million deaths from all types of pollution. Moreover, outdoor air pollution is responsible for 65% premature deaths caused by air pollution, causing 4.2 million premature deaths per year (WHO, 2018). Children in particular have been noted to be at high risk of pollution-related disease. Even very low-level exposures to air pollutants during windows of developmental vulnerability can result in disease, disability, and death in childhood and in later life (Landrigan et al., 2017). Traffic related air pollution is not only linked to worsening health problems in children, but is also connected with the development of asthma (Chiang et al., 2016; Ding et al., 2016). Traffic related air pollution is harmful to the development of children's nervous system, causing neurodegeneration, neuro-inflammation and problems connected to cognition (Calderón-Garcidueñas et al., 2016; Chiu et al., 2016; Saenen et al., 2016). Many other problems such as allergy and autism spectrum disorders have been found to be associated with traffic air pollution (Lee et al., 2015; Flores-Pajot et al., 2016).

Many strategies have been investigated aimed at addressing the aforementioned health risks from transport sources and improving air quality. These have involved traffic management strategies, vehicle fleet composition upgrades, land use and infrastructure optimisation, etc. Traffic management strategies include implementing road/congestion pricing, setting up low emission zones, executing vehicle operating restrictions, optimising traffic signal timing, changing speed limits, encouraging eco-driving, providing pedestrian/bike facilities and so on. A recent review found some limited evidence suggesting that these strategies can reduce total traffic emissions or improve local air quality (Bigazzi & Rouleau, 2017). However, Tang et al. (2017) illustrated that traffic management strategies implemented in Dublin, Ireland including changes to the heavy goods vehicle management and road infrastructure, had both positive and negative impacts on air pollution and greenhouse gas (GHG) emissions. Alam & McNabola (2014) (2018) highlighted the limitations and potential negative impacts of Eco-driving on fleetwide emissions. Ghafghazi & Hatzopoulou (2015) found that traffic calming schemes (speed bumps) could increase both NO_2 and NO_X concentrations. As such, the impact of traffic management strategies on air quality is not always clear and requires careful investigation.

Changes in the vehicle fleet composition have also been shown to result in changes in emissions and air quality. For example, increasing the number of electric vehicles in the fleet has been shown to reduce the emissions of both CO_2 and $PM_{2.5}$ (Alam et al., 2018). Other strategies, such as low emission zones have led to notable beneficial impacts on air quality at one place but may not have the same impacts at another place. Low emissions zones have been found to bring about positive effects on reducing air pollutant concentrations in Germany (Jiang et al., 2017), but no obvious effect can be found elsewhere (Holman et al., 2015). Research has found that both positive and negative impacts can be brought about by similar strategies (Bigazzi & Rouleau, 2017). It was suggested that the impact of similar strategies may vary from case to case. Therefore, the impacts of traffic management should be scrutinized in each case.

As children are especially vulnerable to traffic pollution, many strategies have been implemented in order to reduce the exposure of children to traffic pollution in school. A person's exposure to air pollution is generally determined by the concentration of pollutants and the length of exposure. Strategies to lessen traffic pollution exposure usually focus on decreasing pollutant concentrations and reducing the duration of exposure. Moreover, pathways of exposure to traffic pollution for children relating to school include exposures during their travel to school, exposures due to idling traffic emission during drop off and pick up period, exposures in schools because of pollutant dispersions from streets, etc. Thus, strategies were introduced worldwide to reduce pollutant concentration and duration of exposure regarding these pathways. Strategies that have been adopted by governments to reduce children's exposure to traffic pollution include: 1) reducing traffic volumes through infrastructure, e.g. governments in UK created many traffic-free routes that keep children away from traffic emissions when travelling to school by foot or bike; 2) supporting children to choose lower-traffic route for travelling to school, e.g. in Washington State, US, and Cambridge, UK, maps or online information about low-traffic routes to school were provided; 3) reducing traffic emissions by encouraging people to shift car trips to walking and bicycling, for example, a Europe-wide campaign has been launched to encourage children to walk and bicycle to school and educate children about the relationship between sustainable transportation and the environment; 4) limiting idling on school grounds, e.g. many schools in the US have no-idling signage inside school; 5) reducing diesel vehicles exhausts around schools, e.g. some local governments in the US provided funds for replacing or retrofitting diesel school bus, and some governments routed diesel trucks away from schools; 6) sitting schools in low-pollution locations, e.g. constructing schools away from motorways, etc (McAuley & Pedroso, 2012). The effects of these strategies were not fully investigated. Since it is evident that traffic air pollution is harmful to human health, especially in children, the impact of traffic management strategies on air quality near schools is worthy of investigation. Reductions in speed limits near schools are commonly implemented for traffic safety reasons yet the air quality impact of this measure is unclear and not often considered. Also, as reducing diesel exhausts is a common approach for schools to reduce air pollution around schools, and European cities are proposing to reduce the prevalence of diesel vehicles (Knight et al., 2015), the impact of this proposal is needed to be explored. Considering these two factors, the impacts of different speed limit settings and fleet changes on air quality near a school were evaluated in this Chapter.

Traffic conditions are important for the accuracy of evaluating traffic management strategies. In this paper, we utilized a traffic model and an emission and dispersion model to estimate the air quality conditions in the vicinity of a school in Dublin (Ireland) across 2013, based on traffic count records, background pollutant concentrations, meteorological data and the topography of the area near the school. Predicted air quality impacts of traffic management changes (i.e. different speed limit settings and fleet changes) were compared against current conditions.

Assessing the effect of these elements helps increase the awareness of policy makers as to the extent to which these strategies influence air quality, and enables informed evaluation of whether it is worthwhile to apply these strategies to improve children's health.

4.2 Research methodology

A modelling chain approach was applied to evaluate the traffic and air quality conditions near a school, and to estimate potential impacts of changes in speed limits and fleet composition on air quality. The year 2013 was chosen for the analysis as it had the most complete and up to date dataset available. The area selected for the case study contained a primary school (children aged 4-11) located in Dublin City centre in Ireland (Figure 4-1). Road-side air quality monitoring has been conducted 200m to the east of the school for many years, as part of the fixed site air quality monitoring network within Dublin (Winetavern monitoring site) (EPA, 2017b). The recorded air quality data of this site was used as observed data to compare with modelling data in order to see the model performance. Section 4.2.1 discusses the model development and the data used for the estimation of air quality. Section 4.2.2 describes hypothetical scenarios that were used to estimate the impact of speed limit and fleet composition changes.

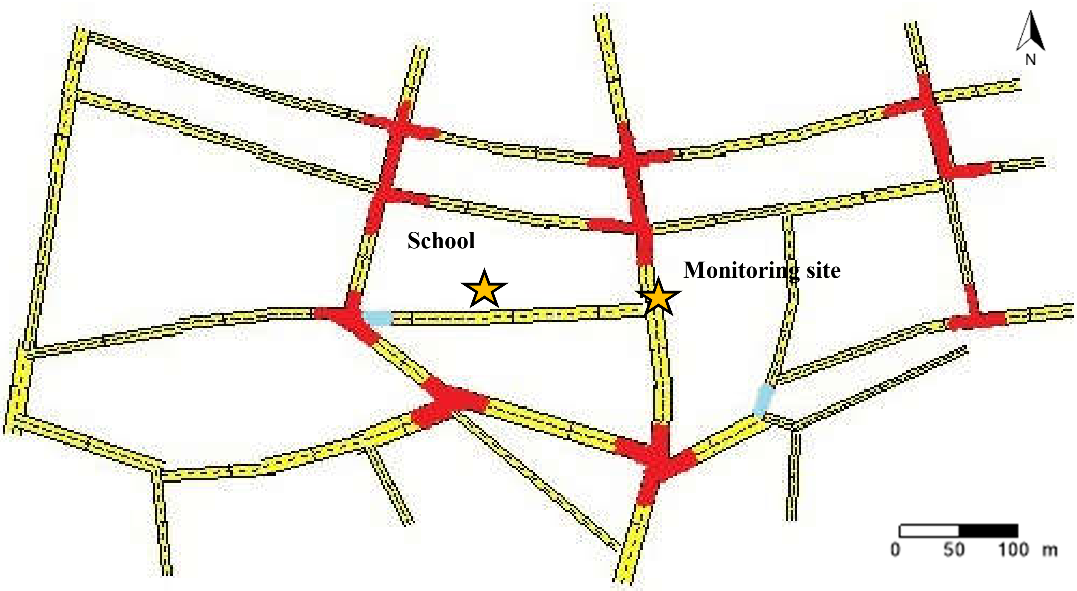


Figure 4-1: Modelling area. (Calibrated roads (red) and validated roads (cyan))

4.2.1 Model development

The modelling chain comprised a traffic model and an emission and dispersion model (See Figure 4-2). The size of the modelled area was approximately 40 hectares. A primary school was located at the centre of the modelling area. This area was chosen to represent the vicinity of the school, and the traffic and air quality conditions around the school. Distance-decay studies on traffic air pollutant concentrations have found that motorways have impacts on concentrations of NO_X and PM typically up to 200m away from the roadside (Health Effects Institute, 2010). Roads in the city centre generally have lower traffic volumes than motorways, and a street canyon topography is often present, which concentrates the air pollution and prevents it from dispersing freely. Therefore, roads in the city centre have a smaller area of influence compared to motorways. In addition, many pupils of the school live within the modelled area, and thus its air quality is crucial for their health.

The traffic model was developed in VISUM (PTV, 2015) and was used to assess the traffic volume and vehicle speed on the road network. Traffic volumes from the year 2013 were used to calibrate and validate the model. The traffic condition outputs from the traffic model were used as an important input to the emission and dispersion model, which was developed using the Operational Street Pollution Model (OSPM) (Berkowicz, Olesen, & Jensen, 2003). The methodology for emission calculations and emission factors for different types of vehicles that were applied in OSPM were based on the European Emission Model COPERT4 (Kakosimos et al., 2010). The study focused on hourly pollutant concentration predictions across a full year, and the implementation of a traffic assignment model (VISUM) and emission calculation method based on hourly average speed (COPERT4) complied with the objective of the study. The details of the development of the traffic model, and the emission and dispersion model, are explained in the following subsections.

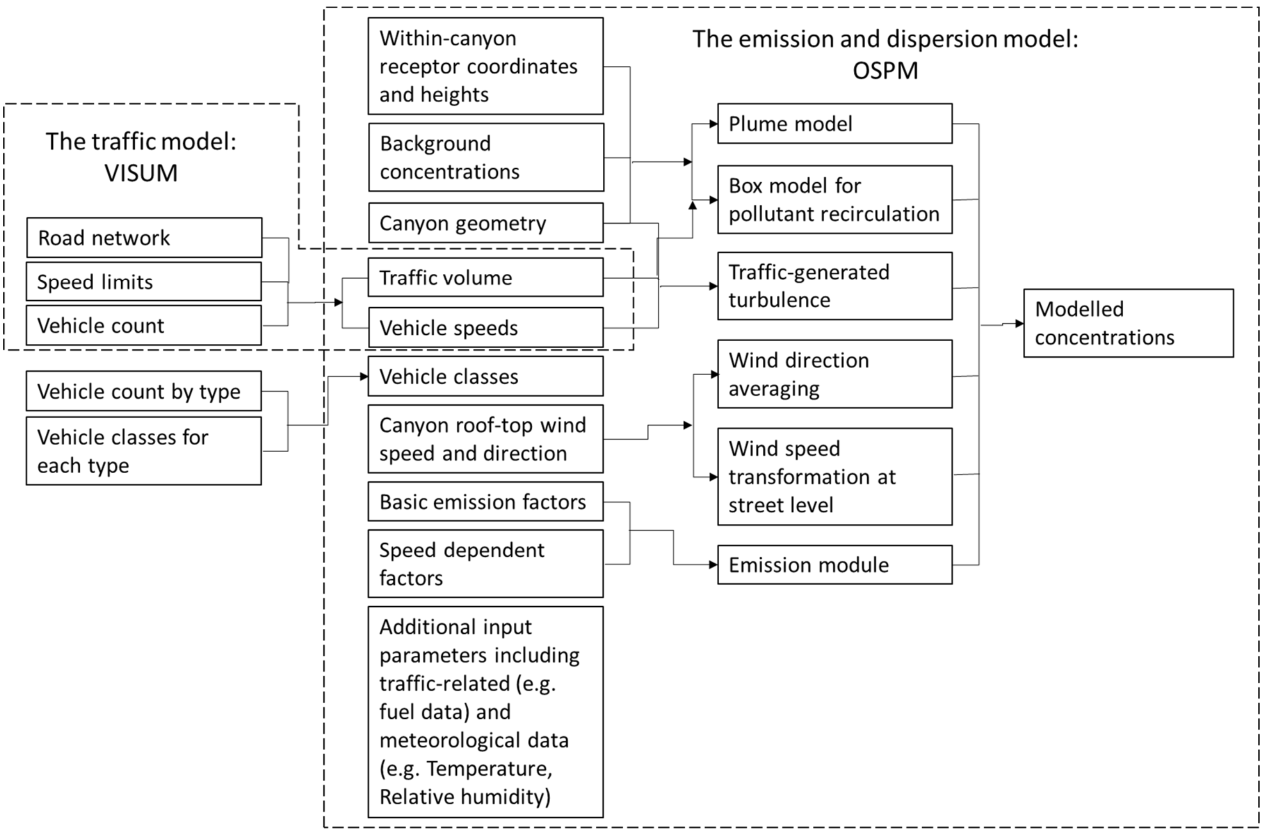


Figure 4-2: Schematic representation of the principal modules of VISUM & OSPM modelling chain. (Adapted from: Aquilina & Micallef, 2004)

4.2.1.1 Traffic model

The flow chart of the traffic model is illustrated as part of the modelling chain in Figure 4-2. The traffic model estimated traffic volume and traffic speed based on speed limit and traffic condition inputs on each road in the network. VISUM was chosen as the traffic model because of the flexibility to adjust traffic volume based on traffic count, and the ability to use Python script to control the modelling process.

The road network was developed in VISUM to represent the roads of the modelling area shown in Figure 4-1. The original travel demand was presented as an OD matrix for the modelled area and was derived from a traffic model for the city, developed by Tang et al. (2017). This traffic model for the city was obtained by extracting Dublin city from the National Transport Model (NTpM) of Ireland, and was calibrated with annual average traffic count records of 2013 for roads in Dublin.

The original OD matrix was then calibrated with traffic count data throughout the year 2013 to obtain a more precise travel demand. The traffic count data were derived from hourly traffic counter data from Dublin City Council (DCC). The traffic conditions for weekdays for ten months of 2013 - i.e. excluding July and August - were modelled to represent the days when the traffic pollution in the modelled area were of most relevance to pupils in the school (pupils did not attend school during weekends or in the summer holiday months of July and August). The traffic model was calibrated using one week of hourly traffic data from every two-month period in 2013, excluding weekends and July and August. Analysis was conducted using a Python script because of the large amount of data involved in hourly travel data across a full year. The traffic data involved in the calibration were chosen to be representative of various traffic conditions throughout the school year, i.e. they included all weekdays, rainy and sunny days, spring, autumn and winter periods.

The TFlowFuzzy technique was utilized to calibrate OD matrices as mentioned in section 3.2.1.3. During the calibration process, the original OD matrix was assigned to roads to obtain traffic volumes using a user-equilibrium assignment approach. Then the assigned volumes were compared to traffic counts and adjustments were made to OD matrices to obtain more accurate results. This step of the assignment of the OD matrix to roads, the comparison between the assigned volumes and traffic counts, and the adjustment of the OD matrix was repeated several times to obtain an accurate matrix to reflect the traffic condition for each hour corresponding to traffic count data. In this process, traffic count data had to be set as an input to VISUM for each hour at several roads. A second Python script was established to conduct this iterative process automatically. A validation of the traffic volume was performed after the calibration process. Hourly traffic count data for 600 hours of 37 roads were derived from DCC, among which 35 were chosen for calibration and the remaining 2 were reserved for validation. Roads that were calibrated and validated are indicated in Figure 4-1 with red and cyan colours, respectively.

The Geoff Havers (GEH) statistic was chosen to evaluate the validation. A guidance regulated in the Design Manual for Roads and Bridges (DMRB) in UK was chosen as the criteria for the validation (UK DMRB, 1997). The calculation of GEH and the criteria is summarized in Table 4-1.

Measure	Criteria	Guideline
$\boldsymbol{GEH = \sqrt{\frac{2(M-C)^2}{M+C}}}$	GEH < 5	> 85% of cases

Table 4-1: Validation criteria (Adapted from UK DMRB (1997)).

Where, In GEH calculation formula, M is the hourly traffic volume from the traffic model and C is the real-world hourly traffic count.

During the assignment process, the traffic volume on each road was determined not only by the OD matrices but also influenced by the relationship between traffic volume and speed, which was represented by a volume-delay function (VDF). A logistic form of the VDF was adopted as it was found to be more accurate when applied to the prediction of speed measurements in this study, as shown in Equation 4-1:

$$t_{cur} = t_0 + \frac{a}{1+f\cdot e^{b-d\cdot sat}}$$

Equation 4-1: Volume-delay function (VDF).

Where, $sat = \frac{q}{c\cdot q_{max}}$, t_{cur} represents travel time modelled, t_0 represents the free flow travel time, q represents the traffic volume and q_{max} represents the road capacity. The values of the parameters in Equation 4-1 are listed below:

$$a = 0.1, \qquad b = 9, \qquad c = 2, \qquad d = 28, \qquad f = 1$$

The comparison between the modelled and recorded traffic average speeds and the result of traffic volume validation are presented in Section 4.3.1.

4.2.1.2 Dispersion and emission model

The flow chart of the emission and dispersion model is also shown in Figure 4-2. The traffic emissions were calculated in OSPM based on the predicted traffic flow in streets (vehicles/hour), the traffic speed, and the emission factors at certain speeds for particular types of vehicles (g/vehicle/km). Only the exhaust emission was considered in this study.

The fleet composition was derived from traffic count data and the Irish national fleet composition. Fleet proportions for each category (e.g. cars, buses and trucks) were derived from the count data across 2013 in Dublin city centre. Within each category, the percentages for each vehicle sub-category (e.g. petrol >2.0 l with PC Euro 3 and Diesel >2.0 l with PC Euro 4) were assumed to be in proportion with the Irish national fleet composition (Duffy et al., 2015), following the same methodology applied by (Tang et al., 2017). Summarized fleet data are presented in Table 4-2. Details of the fleet data are given in Table A-2 in the Appendix A. A 1% composition of benzene in petrol was assumed, in line with the EU regulation (European Parliament, 2009).

Category	% of fleet	Fuel type	Percentage in each category by scenario (scenario number)				
			Baseline	30% diesel cars to petrol (iv)	60% diesel cars to petrol (v)	100% diesel cars to petrol (vi)	100% diesel cars and vans to petrol (vii)
Passenger car	82%	Petrol Diesel	63% 37%	74% 26%	85% 15%	100% 0%	100% 0%
Van	12%	Petrol Diesel	0.30% 99.70%	0.30% 99.70%	0.30% 99.70%	0.30% 99.70%	100.00% 0%
Truck	1%	Petrol Diesel	0 100%	0 100%	0 100%	0 100%	0 100%
Bus	5%	Petrol Diesel	0 100%	0 100%	0 100%	0 100%	0 100%

Table 4-2: Fleet data for each category and fuel type for different fleet change scenarios.

Information on traffic conditions, which included the traffic volume and speed for each hour of the year and fleet composition information, was developed using a third Python script to gather information from the traffic model, and then formatted and inputted into OSPM.

Besides emissions, other inputs to OSPM that influenced pollutant concentrations included road and building geometry information, weather information and background pollutant concentration. These data were obtained from Google Maps, the Irish meteorological service, and the air quality monitoring network of the Environmental Protection Agency of Ireland, respectively. A summary of these elements is shown in Table 4-3.

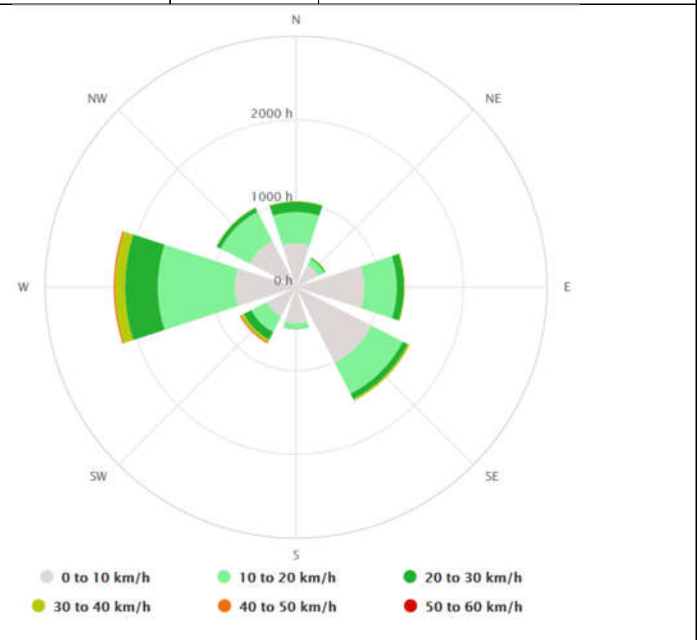
Input type	Element	Descriptive statistics: Mean	Descriptive statistics: Standard deviation	Notes
Background concentration	PM_{10}	13.58 µg/m^3	9.43	Data source: PM_{10} monitoring site located at a park in Dublin
	NO_X (including NO_2 and NO)	28.06 µg/m^3	38.37	Data source: NO_X monitoring site located at inner suburb in Dublin
Building geometry	Building height / road width (H/W)	0.98	0.41	Data source: Google Maps
Weather condition	Wind speed	5.63 m/s	2.86	Data source: the Irish meteorological service.
	Temperature	9.52 °C	5.46	
	Relative humidity	0.82	0.12	
	Windrose			

Table 4-3: Input data for background concentration, building geometry and weather condition.

The background concentration for PM_{10} is derived from a monitoring site located in a park in Dublin (Phoenix park). The distance between this site and the school considered in this study is about 4 km. Phoenix park is one of the largest enclosed parks within any European capital city, and most of its area is away from traffic emission source. This site can provide a good approximation for background air quality. However, as this site does not monitor NO_X concentration, the background NO_X concentration is derived from a monitoring site located in an inner suburb on the southside of Dublin (Rathmines), which is around 3 km from the school. The traffic nearby this site is low and thus it is the most representative site for NO_X background concentration in Dublin. The data for building geometry along the roads is derived from Google Maps. The heights of the buildings along the roads were calculated by counting the storeys with the assumption that the height for each storey was 3m. The meteorology data was obtained from the Irish national meteorological service. The data was recorded by a weather station at the Dublin airport, which is the closest weather station to the study area. Wind speed measurements were taken at the height of 10m and the data is capable to represent roof-level wind speed required by the model in the study area.

In order to facilitate a holistic assessment of traffic emissions, the major harmful pollutants that originate from traffic were included, namely NO_X, PM_{10}, $PM_{2.5}$, CO and Benzene. For NO_X, a special focus was given to the modelling concentrations of NO_2 (in the Modelling results section), as NO_2 is major harmful pollutants to health. The modelled total concentrations consisted of a background concentration component and a component arising from the local traffic in each road by taking the sum of them. Whilst the impacts of speed limit and fleet composition changes on the concentrations of NO_X, PM_{10}, $PM_{2.5}$, CO and Benzene on each road were estimated, the background concentrations were only available for PM_{10} and NO_X. Therefore, whereas total modelled concentrations of NO_X and PM_{10} were estimated, only the traffic-induced modelled concentrations of $PM_{2.5}$, CO and Benzene were evaluated.

In OSPM, concentrations of traffic emissions were calculated using a combination of a plume model for the direct contribution of traffic pollution, and a box model for the recirculating part of the pollutants in the street, taking street canyon geometry into account. The receptor height was set to 1.2m on both sides of the road, because the age of children in the school in the modelling area was in the range of 4 to 11, and the average height for children in this age range is around 1.2m in Ireland (Heinen et al., 2014). The receptor height of the street on which the air quality monitoring site was located was set to 2m, the same height as the as the monitor.

4.2.2 Scenarios

The air quality conditions estimated from traffic counts data in 2013 near the school were considered as the baseline scenario. The air quality of hypothetical speed limit changes and fleet composition changes were evaluated as alternative scenarios and compared to the baseline. The baseline scenario and six hypothetical scenarios are evaluated as outlined in Table 4-4.

At present, diesel cars account for only 3% of total passenger vehicles in the United States and less than 1% in China, whereas this is about 50% in Europe, and it is 45% in Ireland (Chambers & Schmitt, 2015; Duffy et al., 2015). Compared to petrol cars, diesel cars have better fuel economy than petrol-powered cars, thus their emission of CO_2 may be lower. However, diesel remains a major source of harmful pollutants e.g., ozone-forming gases, including NO_X and PM (Chambers & Schmitt, 2015). Therefore, in line with several recent proposals to reduce the prevalence of this source of air pollution in European cities (Knight et al., 2015), this paper compares the impact on air quality outside a school in an urban setting of converting diesel cars to comparable petrol cars.

Also, Dublin City Council reduced the speed limit within the modelled area from 50 km/h to 30 km/h in 2009 because of road safety considerations within the city centre. This paper, therefore, also assesses the impact of the speed change by modelling the impact of two hypothetical speed limits on air quality.

Scenario type	Scenario	Notes
Baseline	i. 30 km/h with Irish national fleet composition in 2013	Reflection of the current condition of 2013.
Speed limit	ii. 40 km/h iii. 50 km/h	Baseline fleet composition was applied; speed limit was changed for these scenarios.
Fleet composition	iv. 30% of diesel cars converting to petrol cars v. 60% of diesel cars converting to petrol cars vi. 100% of diesel cars converting to petrol cars vii. 100% of diesel cars and vans converting to petrol vehicles	30km/h speed limit was applied; detail percentages of petrol and diesel vehicles for these scenarios is shown in Table 4-2 with the scenario number corresponding to each scenario number in this table.

Table 4-4: Scenarios information.

4.3 Modelling results

4.3.1 The traffic model

4.3.1.1 Volumes and speeds

The average modelled traffic volumes and speeds within the modelled area are summarised in Figure 4-3, showing the average hourly traffic data and average speed for all road segments within the baseline scenario (speed limits of 30km/h) during different hours of the day. The average modelled speed at AM peak hour (8 to 9 am) and PM peak hour (5 to 6 pm) of 12 km/h and 21km/h was very similar to the measured average traffic speed in the city centre of 13 km/h and 19 km/h (National Transport Authority, 2016). During off-peak hours the average modelled speed returned towards the relevant speed limit. The modelled average speed under speed limit 40km/h and 50km/h are also depicted in Figure 4-3.

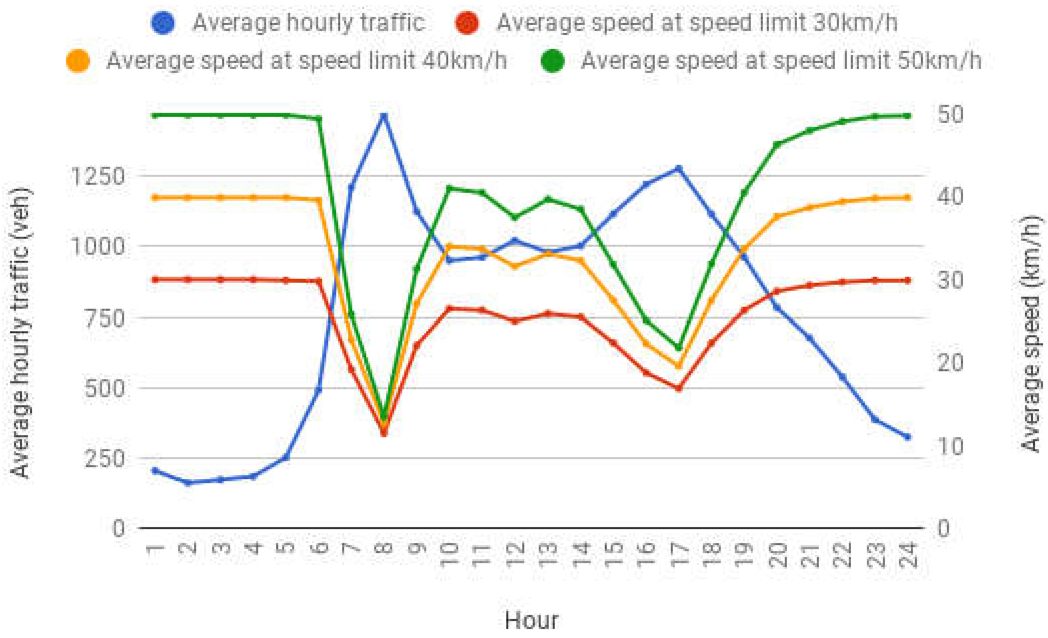


Figure 4-3: The average hourly traffic and average speed at the speed limit of 30, 40 and 50 km/h at different hours of a day for all road segments across 2013.

4.3.1.2 Result of validation for traffic volumes

A summary of validation results, using the GEH statistics approach, is presented in Table 4-5. The validation satisfied the DMRB criteria shown in Table 4-1, resulting in a robust model which represented the travel demand after being calibrated using the recorded volumes.

	# of links and turns	# of hours	# of cases	# of cases with GEH < 5	% of cases with GEH < 5
Validation	2	600	1200	1041	87%

Table 4-5: Summary of the result of traffic volume validation.

4.3.2 Dispersion model verification

Studies elsewhere have compared and validated local-scale emission and dispersion models (including OSPM) with in-situ measurements and have found a good fit with this modelling approach especially for annual average concentrations (Aquilina & Micallef, 2004). Figure 4-4 shows the results of modelled NO_X and PM_{10} concentrations compared against the observed concentrations in Dublin. Both modelled and observed concentrations represent daily average concentrations of ten months of weekdays across 2013. The modelled concentrations were acquired from the particular road on the network where the monitoring site was located, and also on the side of that road where the monitor was located.

The modelled concentration fitted the observed concentration, with an R^2 = 0.84 and 0.82 for NO_X and PM_{10}, respectively. The modelled PM_{10} concentration had good accuracy, with a slope of 1.07. However, the modelled NO_X data underestimated the measured concentration, where the slope was 0.69. Previous investigations have found similar underestimations of NO_X using the OPSM model (Fallah-Shorshani, Shekarrizfard & Hatzopoulou, 2017).

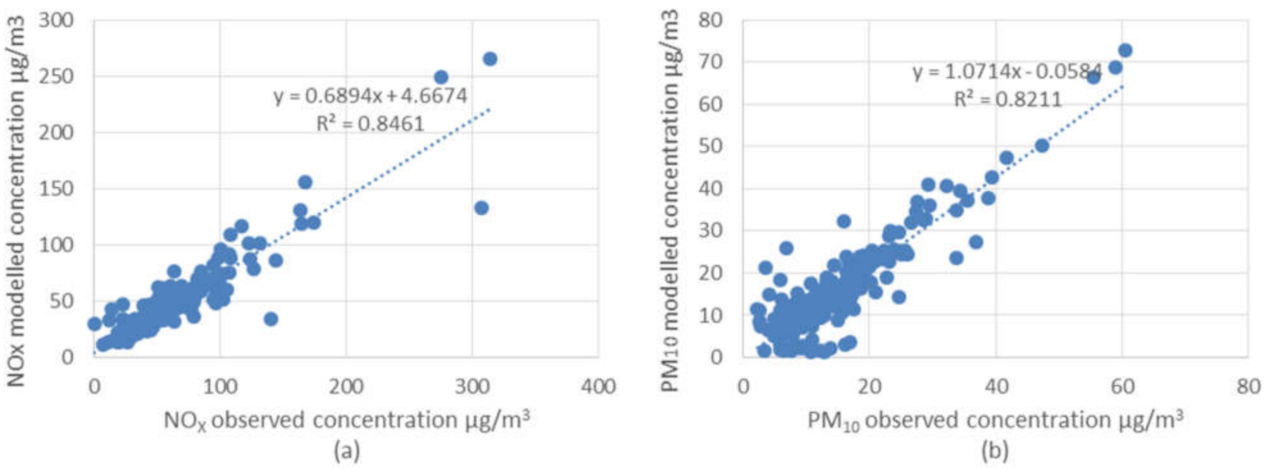


Figure 4-4: Modelled vs observed concentrations for a) NO_X and b) PM_{10} daily average in 2013.

The comparison of modelled traffic-induced concentration and observed concentration minus measured background concentration is needed (LAQM, TG., 2018) and is presented in Figure 4-5. The modelling traffic-induced concentration does not fit well with the subtraction of measured background concentration from observed concentration, with an R^2 = 0.25 and 0.11 for NO_X and PM_{10}, respectively. This comparison is based on two assumptions: 1) the air quality of the observed site nearby the school was only influenced by traffic except background air quality, and 2) pollutant sources that influence the background air quality would influence the observed site to the same degree. However, the concentration for PM_{10} in Ireland was affected by many factors, such as solid fuel burning and natural sources, and traffic is not the major source which contributing 9% of PM_{10} pollution (EPA, 2017a). Also, the annual average PM_{10} concentration in rural area can be larger than in the traffic-intensive area (EPA, 2018). Thus, the above assumptions were not firm, causing the modelling traffic-induced concentration and the subtraction of measured background concentration from observed concentration incapable to be compared.

For NOx concentration, the correlation is relatively better than PM_{10}. NO_X concentration is mainly affected by traffic emissions in Dublin, and the first assumption mentioned above were relative firm. From Figure 4-5 (a) we can see that NO_X concentrations of background site were less than the observed site near the school for the majority days. This means that the degree of the affection by traffic on the background site is far less than the observed site. In spite of this, background concentration site still has a little traffic that influence its concentration, and thus the second assumption were not firm. Besides, the OSPM has its limitations that influence the correlation. The OSPM does not account for traffic emission dispersion from other roads. Although emissions from a road is the predominant factor that influence pollutant concentrations of this road, emissions from other roads nearby would still contribute to pollutant concentration of the road considered. The long distance between the background site and the observed road site increased the modelling error brought about by the limitation of OSPM. Despite this limitation, previous studies found that OSPM can still perform well with background site close to the observed road site (Aquilina & Micallef, 2004; Berkowicz et al., 2008; Ketzel et al., 2012).

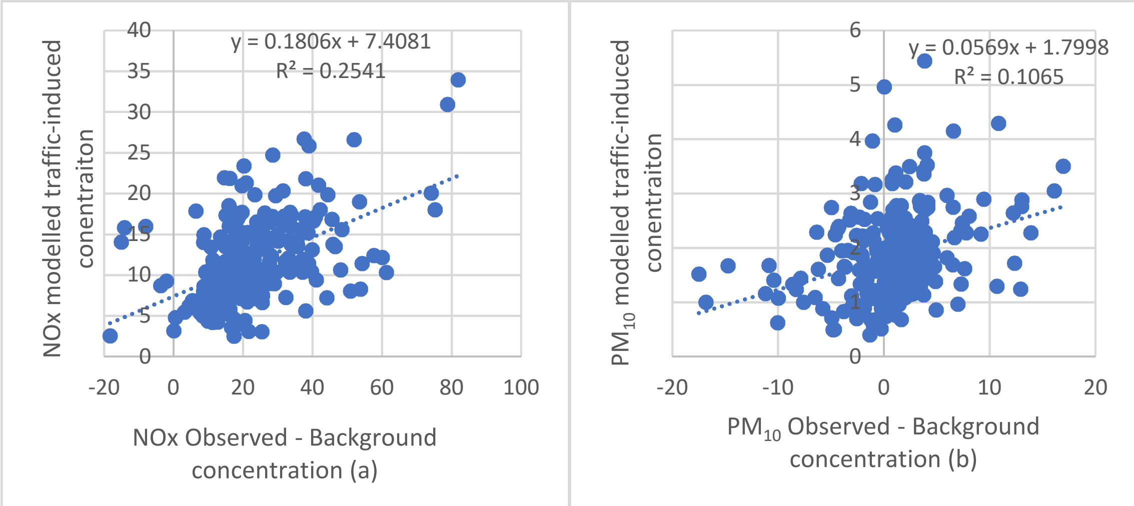


Figure 4-5: Modelled traffic-induced concentrations vs observed concentrations minus background concentrations for a) NO_X and b) PM_{10} daily average in 2013.

4.3.3 The effect of speed limit changes

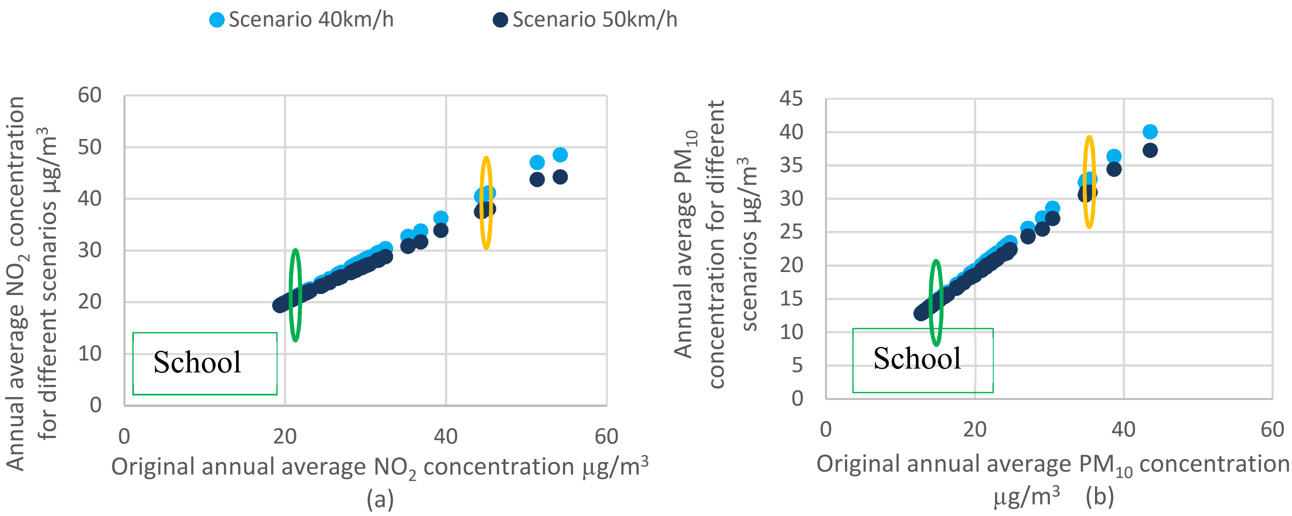


Figure 4-6: Predicted change in NO_2 and PM_{10} total street concentrations on each road segment in the model domain for varying speed limits.

The modelled impacts of speed limits on the total concentrations of NO_2 and PM_{10} in each street in the modelled area are summarised in Figure 4-6 (a) and (b). Each individual data point in Figure 4-6 represents the concentration on a specific road link for the speed limit change scenarios compared to the baseline scenario. The x-axis shows the baseline scenario concentration, and the y-axis shows the concentrations for two speed limit scenarios. A green oval highlights the road on which the school was located. Traffic volumes and consequently air pollution concentrations on this street were less than for the majority of road links in the rest of modelled area. The impact of speed limits on the traffic-induced concentrations of $PM_{2.5}$, CO and Benzene are summarised in Figure B-1 (a), (b) and (c), respectively, in the Appendix B.

Figure 4-6 shows a decline in both NO_2 and PM_{10} concentration for scenarios with hypothetical increases of speed limits from 30km/h to 40km/h and 50km/h. The alteration of the speed limit from 30 km/h to 50 km/h could reduce NO_2 by up to 18% depending on the original concentration in the street in the baseline scenario. This action could also reduce PM_{10} by up to 15%, also depending on the original concentration. The general trend shows that NO_2 and PM_{10} concentrations could be reduced more for the streets that originally had higher NO_2 and PM_{10} concentration. Regarding traffic-induced concentration, the alteration of the speed limit from 30 km/h to 50 km/h could reduce traffic-induced concentration of NO_2 and PM_{10} by 21% to 30% and 14% to 22% depending on the street in question, respectively. As background concentration for different streets was assumed to be the same, the higher the NO_2 and PM_{10} concentration was on the street, the higher the pollutant concentration that was generated by traffic, and therefore, the speed limit change strategy that influences traffic conditions would have a relatively large effect on the pollutant concentration.

As the school was located in a place with low traffic volumes and was thus less polluted, the effect of the speed change on air quality was relatively small compared to other streets with heavier pollution. Comparing the speed limit change impact in front of the school with a busier street (shown in Figure 4-6 with yellow oval), for example, the predicted concentrations of NO_2 and PM_{10} could only be reduced by a maximum of 3% and 2% (corresponding to 22% and 15% reduction of traffic-induced NO_2 and PM_{10}), respectively, on the street in front of the school, whereas these figures could be reduced by up to 15% and 12%, respectively, on the street with higher original pollutant concentration.

4.3.4 The effect of fleet composition

The impact of fleet composition on the total concentrations of NO_2 and PM_{10} in each street is shown in Figure 4-7 (a) and (b), whilst the impact of fleet composition on the traffic induced concentrations of CO and Benzene is shown in Figure 4-7 (c) and (d). Traffic induced concentrations instead of total concentration of CO and Benzene were estimated because of the absence of background concentration information for these gases. The impact of fleet composition on the traffic induced concentrations of $PM_{2.5}$ is shown in Figure B-2 in the Appendix B.

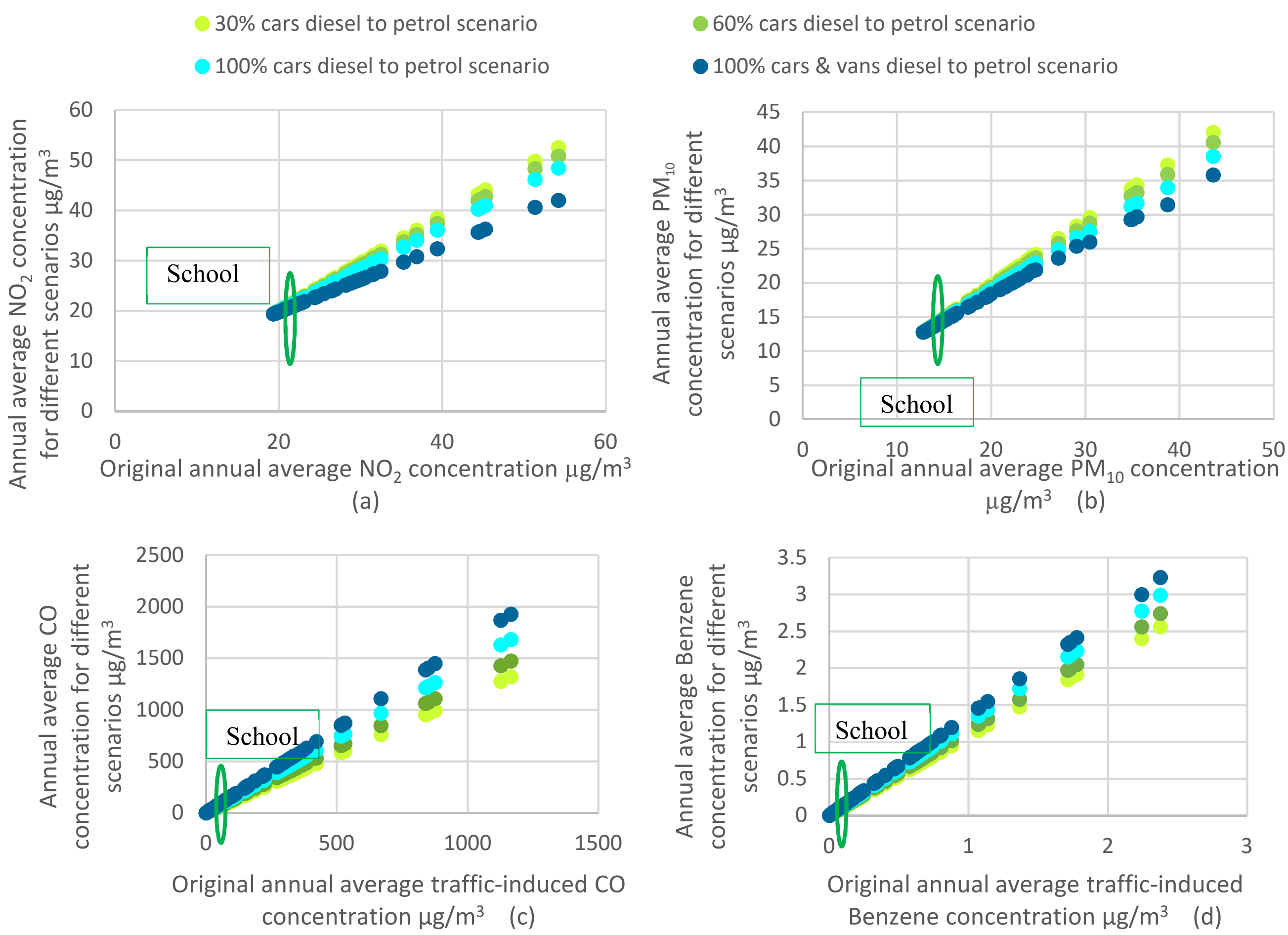


Figure 4-7: Predicted percentage change in a) NO_2 and b) PM_{10} total street concentrations and c) CO and d) Benzene traffic induced concentrations on each road segment in the model domain for varying fleet compositions.

Figure 4-7 (a) and (b) show declines in NO_2 and PM_{10} concentration for the modelled fleet composition change scenarios. This demonstrates that increasing the proportion of petrol vehicles can reduce the NO_2 and PM_{10} by up to 23% and up to 19%, depending on the street in question. This hypothetical strategy can reduce the traffic-induced NO_2 and PM_{10} by up to 35% and up to 28%. For similar reasons as outlined above, the general trend for NO_2 and PM_{10} concentrations was that they could be reduced more for the streets that originally had higher concentrations. Among all these scenarios, including speed limit change scenarios, replacing all diesel cars and vans with petrol cars and vans had the greatest predicted impact on NO_2 and PM_{10} concentration.

The effect of fleet composition change on the street in front of the school was relatively small compared to other streets with heavier pollution (Table 4-6). Regarding the reduction of the concentration of NO_2 and PM_{10} on the street in front of school, the scenario of changing all diesel cars and vans to petrol vehicles was the most efficient, reducing the NO_2 and PM_{10} pollution concentrations by 4% and 3%, respectively. When compared to the 3% and 2% reductions found for the scenario of increasing the speed limit to 50km/h, both interventions produced similar levels of impact.

		Street in front of the school	A street with high original concentration
Original concentration	NO_2	22 μg/m^3	45 μg/m^3
	PM_{10}	14 μg/m^3	35 μg/m^3
Change in concentration with 30% diesel changing to petrol cars scenario (change in traffic-induced concentration)	NO_2	-1% (-5%)	-3% (-5%)
	PM_{10}	-1% (-4%)	-3% (-5%)
Change in concentration with 60% diesel changing to petrol cars (change in traffic-induced concentration)	NO_2	-2% (-10%)	-6% (-10%)
	PM_{10}	-1% (-8%)	-6% (-10%)
Change in concentration with 100% diesel changing to petrol cars (change in traffic-induced concentration)	NO_2	-2% (-16%)	-9% (-17%)
	PM_{10}	-2% (-14%)	-11% (-17%)
Change in concentration with 100% diesel changing to petrol cars and vans (change in traffic-induced concentration)	NO_2	-4% (-35%)	-20% (-35%)
	PM_{10}	-3% (-22%)	-16% (-26%)

Table 4-6: The impact of fleet composition changes on the concentration of NO_2 and PM_{10} comparison between the street in front of the school and a street with high original pollutant concentration.

With regard to the traffic-induced concentrations of CO and Benzene, Figure 4-7 (c) and (d) depict that, in general, increasing the proportion of petrol cars and vans would increase these concentrations appreciably. Thus, in contrast to NO_2 and PM_{10}, increasing the proportion of petrol vehicles would exacerbate the CO and Benzene pollution. By changing all the diesel cars and vans to petrol cars and vans, the traffic-induced concentration of CO would be increased by 65% at maximum. This strategy would also increase traffic-induced Benzene concentration by 36% at maximum. The scenario of changing diesel vehicles to petrol vehicles could increase the traffic-induced CO and Benzene concentration at the school by 63% and 35%, respectively.

4.4 Discussion

This investigation showed that speed limits and fleet composition changes could have notable impacts on air quality. These scenarios could potentially affect the air quality at the location of the school in question here, albeit to a lesser extent than they would affect the air quality along the more polluted streets away from the school.

Regarding speed limits, the hypothetical increase of speed limits from 30km/h to 50km/h was predicted to reduce the concentrations of all the pollutants examined here, i.e. NO_2, PM, CO and Benzene. Ghafghazi & Hatzopoulou (2015) found that speed bumps that are used to reduce speeds, could lead to an increase in both NO_2 and NO_X concentrations in urban areas, which is in line with the results of the present study. Panis et al. (2011) estimated that a speed limit change from 50 km/h to 30 km/h would increase NO_X and PM emissions by 2% to 5% and 3% to 8%, respectively, using the COPERT methodology. Carsten et al. (2008) found that PM and NO_X would increase by 1% and 0.7% when the number of vehicles conforming to a speed limit of 30 mph was increased. These results are in line with the results of the present study. Some studies have also found a decrease in traffic emissions with speed limit changes from 50 km/h to 30 km/h in Belgium, using microsimulation (Panis et al., 2011; Madireddy et al., 2011). To date there have been few ex-post studies that can support the conclusions made by the different analyses of speed limit changes, and clear-cut conclusions about the impact of these strategies on air quality do not exist (Bigazzi & Rouleau, 2017).

The results also showed that for the streets that experienced more severe air pollution, the changing of speed limits would have more impact on traffic-related air pollution. Regarding the speed limit decrease from 50 km/h to 30 km/h in 2009, it would not have affected the children significantly in terms of the air quality health impacts as the school within this study area is located along a less-busy, less-polluted street. However, this study does also indicate that, for a busy and heavily polluted area, a strategy of decreasing the speed limit could potentially lead to a reduction in air quality, with consequent implications for health.

In assessing the potential risks of speed to human health, it is necessary that all risks are taken into account. In this respect, it is perhaps worth pointing out that air pollution is responsible for a considerable amount of premature deaths worldwide. In Ireland in 2014, 193 people were killed in road traffic accidents (Road Safety Authority, 2015b), while in the same year, air pollution was associated with 1,510 premature deaths (EPA, 2016). However, transport is only one of many contributors to this figure. Therefore, it does not necessarily suggest that the influence of air pollution on human health outweighs that of traffic accidents. On the other hand, in terms of the relationship of speed limit and car accidents, a review suggests that "studies of the effectiveness of school zone limits, have generally found poor driver compliance, particularly when the limits are set very low, and no relationship between pedestrian crashes and the special limits" (National Research Council (U.S.), 1998).

The emission calculation method used in the study is based on average speed, which, although complying with the objective and time scope of the study, confined the function of evaluating the impact of accelerating and decelerating on traffic emission. The driving cycles that are adopted to develop COPERT EFs includes the condition of driving in real urban roads, which infer that these EFs incorporate the emission related to acceleration and deceleration to some extent. In order to research the impact of speed limit in more detail in future, a traffic micro-simulation in an area is required. However such micro-simulations are often limited in the size of study area and length of time scale which can be covered, limiting the results of such investigations in contrast to the current approach.

The health impact of the traffic emissions resulting from reducing speed limits should be taken into consideration, especially for schools located on busy roads. In contrast, outside an urban setting, road transport may contribute significantly less than other sources of air pollution to total exposure, and in such circumstances a speed limit reduction would be especially beneficial. A health impact assessment was performed for speed limit changes and is presented in Chapter 5.

As petrol and diesel vehicles are still and will be predominant in the market for some while and there is no alternative that can replace these two types of vehicles worldwide in a short time, it is worthwhile to evaluate these two types of vehicles in terms of their impact on air quality. Also the emission change estimation of the fleet conversion to hybrid or electric is worthy of research in the future in line with the development of the technology.

For different air pollutants, the effects of the changes of speed limit and the fleet composition were different. Increasing the proportion of petrol vehicles could reduce the on-street concentration of NO_2 and PM, but this strategy would lead to a rise in the concentration of CO and Benzene. Thus, although the most significant modelling result for reducing the pollution of NO_2 and PM was obtained by replacing all diesel cars and vans with petrol cars and vans, when considering the pollution of CO, benzene and other pollutants that were not covered by our paper, this strategy appears inconclusive, with positive and negative impacts. Policy makers should take as many pollutants as possible into account in order to make a full assessment of the advantages and disadvantages of using petrol and diesel cars. The results of this study highlight that the incentivisation of one fossil fuel type over another has advantages and disadvantages. The negative impacts of dieselisation of the European vehicle fleet are well publicized (Cames & Helmers, 2013). However, a similar incentivisation of petrol engines would have resulted in problems of a different nature. Determining which policy is effective in tackling traffic-related health impacts should involve a health impact assessment to quantify the impact of increases and decreases in different pollutants with different levels. Chapter 5 presents a health impact assessment of the fleet composition changes.

Similar to the effect of changing speed limits, for the streets that experienced more severe air pollution, the strategies of increasing the proportion of petrol cars were more effective in dealing with NO_2 and PM pollution. The improvement in air quality of implementing different traffic management strategies depends on how much the traffic contributes to the air pollution. Therefore cities with severe traffic pollution will get larger benefits when applying these traffic air pollution reduction strategies, than those found in Dublin where air quality is relatively good.

4.5 Conclusion

This study analysed several strategies and related factors that could affect air pollution in a city centre near a school. Traffic counts data were used and several vehicle fleet and speed limit scenarios were modelled. The results showed that both strategies can influence the traffic-induced air pollutant concentrations significantly. Changing diesel cars and vans to petrol cars and vans had both advantages and disadvantages: this strategy would lead to benefits in terms of reducing NO_2 and PM pollution, but would increase the pollution from CO and Benzene. Although the health impact of CO and Benzene could be small, comprehensive health impact assessment is still needed when policy makers plan to provide incentives in favour of either petrol or diesel vehicles. Policy makers should seek to strike an appropriate balance of both in the fleet at present, whilst looking to the future, strategies should be implemented that are aimed at phasing out both types of vehicles, and replacing these with e.g. electric vehicles or hybrid vehicles.

Decreasing speed limits near a school, although justified in terms of road safety, was shown to have potentially negative impacts in terms of air quality, especially for streets that are heavily polluted; thus these impacts should be considered and balanced with the benefits to safety brought about by reducing the speed limit.

Chapter 5 Cross-comparison of the potential air pollution related health impacts of differing traffic management strategies

Abstract

Traffic management strategies can affect air quality significantly and thus affect public health as traffic is one of the most important air pollution sources in urban areas. In this study, the impact of four types of traffic management strategy on air quality and public health were assessed. These included a change in transport infrastructure, a traffic regulation change, speed limit changes and fleet composition changes. A modelling chain was adopted consisting of a traffic model, emission model, dispersion model and the health impact model. The modelling chain was employed to evaluate the health impacts of the four strategies. Hypothetical scenarios were created to assess these impacts based on the traffic conditions of 2013 in Dublin, Ireland. In the study, the construction of new infrastructure and changes in traffic management were predicted to have contrasting effects on different parts of the city, bringing little benefit to the city as a whole. The mortality incidence change brought about by changes in speed limits were predicted to have the same order of health impact as fatalities caused by traffic accidents in Dublin in 2013. The fleet composition change from diesel to petrol vehicles were predicted to reduce the $PM_{2.5}$ and NO_2 concentration and relevant mortality incidence for the whole city. A cross comparison among these strategies concluded that finding an optimum speed limit for different areas that balances traffic accidents, congestion and environmental impacts was the most efficient and economic approach of the four, to reduce traffic pollution and associated health impacts in the short term.

5.1 Introduction

Traffic is a major source of air pollution in urban areas, which contributes 39% of nitrogen oxides (NO_X) emissions and 11% of Particulate Matter (PM) emissions in Europe (EEA, 2018a). Focusing on Ireland, road transport contributed 46% of NO_X, 26% of $PM_{2.5}$ and 9% of PM_{10} (EPA, 2017a). As air pollution is a major environmental problem that was estimated to cause 6.4 million premature deaths worldwide per year (Landrigan et al., 2017; WHO 2018), and as traffic pollution contributes a lot to poor outdoor air quality, traffic has a significantly negative impact on public health.

With more importance and urgency given to protecting the environment, many traffic management strategies were implemented by governments worldwide in order to alleviate air pollution and protect public health (Bigazzi & Rouleau 2017; Ferreira et al. 2015; Holman et al. 2015; Morfeld et al. 2014). However, many traffic management strategies were not designed to protect the environment e.g. speed limits are often designed for safety or congestion considerations; new road constructions are designed to meet increasing travel demand. The impact of these strategies on air quality and public health should be evaluated as they impact emission and air quality.

Many studies examined the relationship between traffic management strategies and air pollution, but there is no clear-cut conclusion about the impact of these strategies on air quality. For example, a review of the efficacy of low emission zones (LEZ) found that LEZs may have reduced PM_{10} and NO_2 concentrations by a few percent in Germany but no clear effects were found elsewhere (Holman et al. 2015). Another review found limited evidence that such strategies can reduce total traffic emissions or improve local air quality (Bigazzi & Rouleau 2017).

The health impact of traffic-related air pollution has been widely researched, and there is a consensus on the impacts. For example, $PM_{2.5}$ is well-known related to the all-cause mortality, lung cancer mortality and morbidity, and heart disease mortality and morbidity (Pope et al. 2015; Krewski et al. 2009). NO_2 is mainly associated with respiratory mortality and morbidity (Hoek et al. 2013; Luginaah et al. 2005). However, a limited number of investigations have focused on the health impact brought about by traffic management strategies considering air pollution. As a result, policy makers are not in possession of a full picture of the impacts of new traffic management initiatives to enable a robust comparison between the costs and benefits considering public health and transportation impacts.

In this chapter, the potential health impacts of four different types of traffic management strategies were evaluated, considering $PM_{2.5}$ and NO_2. These strategies included a change in road infrastructure, regulation of heavy goods vehicles (HGVs) within the city, speed limit changes and fleet composition changes. In order to enable a direct comparison of the health impacts of these different types of traffic management strategies, a modelling chain of the traffic, emission, dispersion and the health impact was employed. Hypothetical scenarios depicting these strategies were modelled based on the traffic conditions of Dublin, Ireland in 2013. The modelling of these traffic management strategies enabled a cross comparison of the costs and benefits of each to be conducted.

5.2 Research methodology

This study aimed to assess the health impact of four traffic management strategies in terms of their influence on air quality and public health. The study enabled a cross-comparison and discussion of these strategies regarding to their feasibility, cost and potential benefits/damages to public health. Details on the traffic management strategies and modelling chain are depicted in the following sub-sections.

5.2.1 Health impact model

Mortality incidence was chosen as a health indicator to evaluate the impact of traffic management strategies. The environmental Benefits Mapping and Analysis Program community edition (BenMAP-CE) was utilized to evaluate the mortality incidence associated with traffic air pollution. BenMAP-CE is a software that can evaluate and visualise the number of air pollution-related deaths and illnesses in a given location (U.S. EPA 2015). The evaluation process accounted for the population of a certain area, the changes in air pollutant concentration, the baseline mortality incidence rate, and the relationship between the change in air quality and the change in incidence rate that was estimated by previous epidemiological research (Hoek et al. 2013).

As traffic management strategies were in focus here, air pollutants that are mainly generated or influenced by traffic and that have been found to have significant health impacts were considered. $PM_{2.5}$ and NO_2 which are widely reported to have significant impact on mortality were evaluated. Concentration response functions (CRFs) developed in previous research to quantify the relationship between pollutant concentration and mortality incidence rates of different kind, were used here and are outlined in Table 5-1.

Study	Author	Krewski et al. 2009	Hoek et al. 2013
	Study location & age groups	116 U.S. cities, 500,000 participants, 30-99 years	Pooled estimation from 12 studies, 18-99 years
Pollutant		$PM_{2.5}$	NO_2
Unit		μg/m^3	μg/m^3
Health effect		Mortality, all cause	Mortality, all cause
Concentration--response function formula		$\Delta Y = Y_0 \left(1 - e^{-\beta \Delta Q}\right)$	
Incidence change		$\Delta I = \Delta Y \times Pop$	
Regression coefficient (β)	Mean estimate	0.005827	0.0054
	Std. error	0.000963	0.00123

ΔY: change in incidence rate between baseline and control scenario; Y_0: baseline incidence rate; ΔQ: change in estimated air pollutant concentration between baseline and control scenario; ΔI: change in incidence between baseline and control scenario; Pop: affected population.

Table 5-1: Summary of the main features of selected concentration–response functions (CRFs) for health impact assessment analysis.

Because air quality and population density vary within a city, using data with a relatively high spatial resolution was desirable. In Dublin, the smallest legally defined administrative areas are Electoral Divisions (EDs). EDs were the smallest area units for which population size data were available. Air quality levels were also estimated for each ED.

Figure 5-1 shows the EDs and major roads in Dublin that were considered by the study.

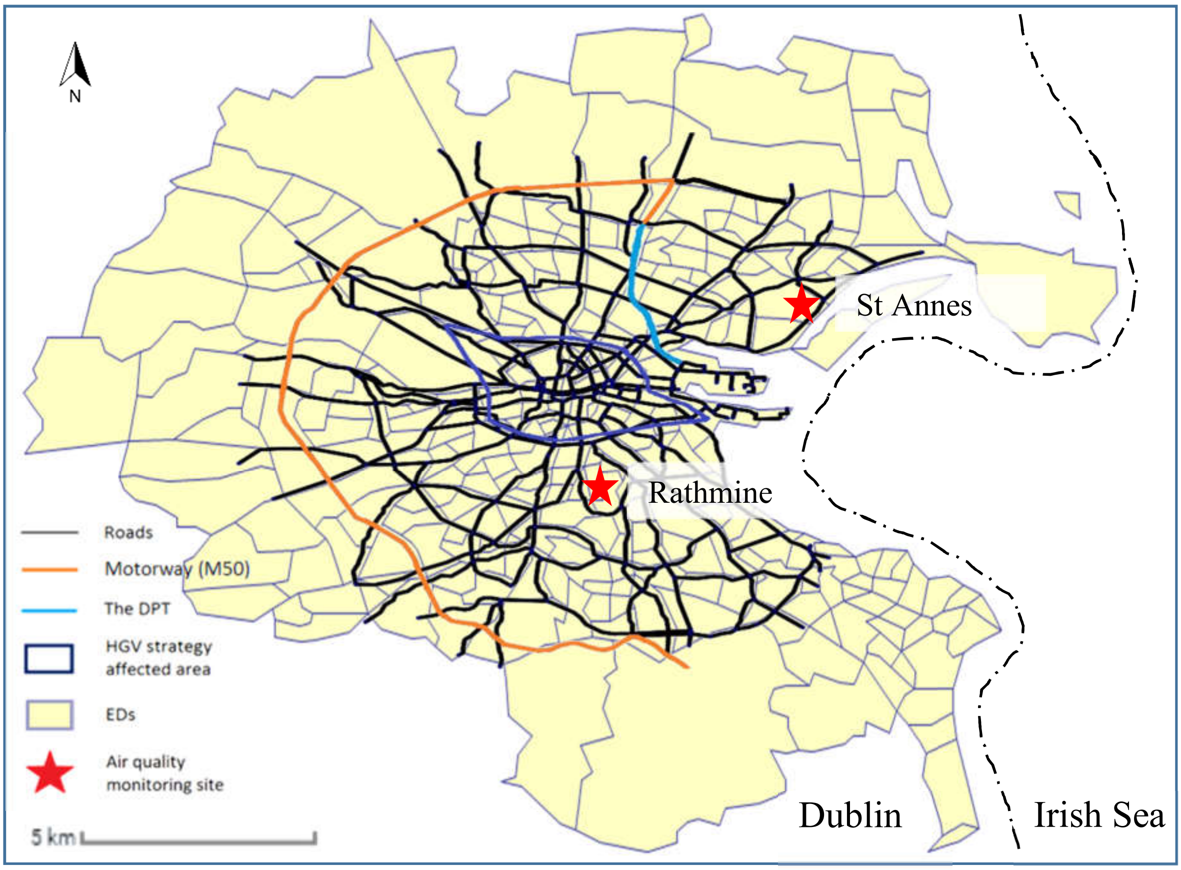


Figure 5-1: Electoral Divisions and roads in study area (the blue line represents the DPT and the area being enclosed in the dark blue line was affected by the HGV management strategy).

5.2.2 Traffic management strategies

Infrastructure & Traffic Regulation Change

Details of the four traffic management strategies evaluated are summarised in Table 5-2. In December 2006, the Dublin Port Tunnel (DPT) was opened as a dedicated route for heavy goods vehicles (HGVs) between Dublin Port and the national motorway network. The aim was to remove HGVs from the city centre to reduce congestion. An HGV management strategy was also introduced in February 2007. This introduced a restriction on 5+ axle HGVs, prohibiting them from entering the city centre area (see Figure 5-1). The impact of the DPT and HGV management strategies on traffic volumes, composition and emissions were previously evaluated by Chapter 3 (Tang at al., 2017). In the present study, both of these traffic management strategies were further evaluated for their impact on air quality and public health, using the travel demands for Dublin in 2013, as the year 2013 had the most complete and up to date dataset available. The OD matrices generated by Chapter 3 estimated the travel demand of an average hour between 7am and 9am across the full year 2013. Chapter 4 (Tang et al., 2019) subsequently modelled hourly traffic data for each hour in 2013 in order to evaluate the impact of traffic strategies in the Dublin city centre area. These analyses were further expanded here to cover a part of the Greater Dublin Area (120 km^2) rather than just a part of Dublin city centre (0.4 km^2), as shown in Figure 5-1. Based on the average one-hour travel demand reported in Chapter 3, the travel demands for other hours were calibrated in the present study according to hourly modelled traffic data and traffic counts across 2013, derived from Chapter 4 and Dublin City Council (DCC). The methodology for the calibration and the development of the traffic volume profile for each road was the same as described in Chapter 4. These traffic data were used as inputs to the emission, dispersion and health impact models in the present study. The three scenarios relating to this aspect of the study are denoted as the baseline, no-HGV and no-DPT scenarios in Table 5-2.

Speed Limit Change

The default speed limit in the majority of the Dublin area in 2013 was 50 km/h, excluding motorways. Areas in the city centre adopted a reduced speed limit of 30 km/h for safety reasons and there are proposals to expand this 30 km/h speed limit area. Chapter 4 evaluated the impact of speed limit changes on air quality on streets in the city centre. The current study evaluated the impact of speed limit changes on air quality and public health for the whole city. The present study assessed the air pollution and health impact of the default 50 km/h speed limit and hypothetical speed limits of 40 km/h (scenario Speed-40) and 30 km/h (scenario Speed-30) applied across the entire city. Travel demand data from traffic models representing these speed limit change scenarios in 2013, were again adopted from Chapter 3 and Chapter 4.

Fleet Composition Change

Scenarios that examined fleet composition changes were assessed primarily in the dispersion and health impact models, as the traffic speed and volume were assumed to be unaltered. Since around one quarter of the population of Ireland is located in the modelling area, the Irish national fleet composition was assumed to represent the fleet composition in Dublin. Therefore, the scenario that represented the actual fleet composition took the assumption that Dublin fleet composition was the same as the national fleet composition in 2013 (listed in Table A-2 in Appendix A). The national fleet composition was also utilized in the other three strategy assessments. Scenarios assessed here included the impact of shifting 50% and 100% of diesel cars to petrol cars (scenario Cars-50% and Cars-100%), and shifting 100% of diesel cars and vans to petrol (scenario Cars&Van-100%).

Scenario code	Scenario condition explanation	Traffic strategy type
Baseline	with DPT opened, HGV management implemented, default speed limit at 50km/h, actual fleet composition in 2013	Baseline scenario, representing actual conditions in 2013
no-HGV	all conditions the same as baseline scenario, except with HGV management not implemented	Traffic management regulation change
no-DPT	all conditions the same as baseline scenario, except with HGV management and DPT not present	Infrastructural change
Speed-30	all conditions the same as baseline scenario, except with default speed limit at 30km/h	Speed limit changes
Speed-40	all conditions the same as baseline scenario, except with default speed limit at 40km/h	
Cars-50%	all conditions the same as baseline scenario, except with 50% diesel cars converting to petrol cars in fleet composition	Fleet composition changes
Cars-100%	all conditions the same as baseline scenario, except with 100% diesel cars converting to petrol cars in fleet composition	
Cars&Van-100%	all conditions the same as baseline scenario, except with 100% diesel cars and vans converting to petrol cars and vans in fleet composition	

Table 5-2: Summary of the traffic strategies and scenarios evaluated by the study.

5.2.3 Dispersion model and air quality

The Operational Street Pollution Model (OSPM) (Berkowicz et al. 2003) and a Gaussian dispersion model were used to estimate the concentration of $PM_{2.5}$ and NO_X in Dublin. The concentration of NO_2 were estimated from the concentration of NO_X using experience functions (see section 5.2.3.3) (Clapp & Jenkin 2001). The estimation of both traffic emission and air pollution from traffic on individual roads were conducted using OSPM (Berkowicz et al. 1997). The overall pollutant concentrations on the streets were estimated as the combination of the estimated contribution from a given street, from OSPM, and the contribution from other roads in the vicinity, from the Gaussian model. The use of OSPM combined with a Gaussian model was previously reported to reduce 16%–25% of the error compared with using OSPM or Gaussian model alone (Fallah-Shorshani et al. 2017; Mensink & Cosemans 2008).

5.2.3.1 OSPM

OSPM took traffic volume and speed data estimated from the traffic model (Tang et al.,2017) as inputs to determine the resulting change in air pollutant concentrations. OSPM calculated the traffic emission based on the traffic flow in the street (vehicles/hour), the traffic speed and the emission factors at a certain speed for particular types of vehicles (g/vehicle/km). The data that represented the traffic condition for each street derived from the traffic model were formatted and inputted into OSPM in batch via a Python script. A Python script was required to process the large volume of hourly data for each road modelled for the year 2013 (764 road segments and 24 hours). OSPM adopted the methodology for calculations of emission for different types of vehicles from the European Emission Model COPERT4 (Berkowicz et al., 1997). During the emission calculation process, the fleet compositions were changed to model different fleet change scenarios (scenarios Cars-50%, Cars-100% and Cars&Van-100%).

Road and building geometry information, along with weather information were important data that influenced the pollutant concentration. These data were obtained from Google Maps and the Irish meteorological service. The background concentration was also a factor that influenced the estimated concentration. The background concentrations of $PM_{2.5}$ were sourced from an air quality monitoring site in a suburb in Dublin (Rathmines). The background concentration of NO_X was sourced from an air quality monitoring site in a park area in Dublin (St. Annes Park). The data from air quality monitoring sites was provided by the Environmental Protection Agency of Ireland. The receptor height in OSPM was set to 1.7m to represent average breathing height in Ireland.

5.2.3.2 Gaussian dispersion model

A second Python script was developed for the Gaussian plume model. Equation 5-1 shows the function used for the Gaussian model for a finite line source (De Visscher 2013).

$$\bar{c} = \frac{Q}{2\bar{u}L\sqrt{2\pi}\sigma_z}\left[\operatorname{erf}\left(\frac{y_2}{\sqrt{2}\sigma_y}\right) - \operatorname{erf}\left(\frac{y_1}{\sqrt{2}\sigma_y}\right)\right]\left\{\exp\left[-\frac{1}{2}\frac{(z-h)^2}{\sigma_z^2}\right] + \exp\left[-\frac{1}{2}\frac{(z+h)^2}{\sigma_z^2}\right]\right\}$$

Equation 5-1: Gaussian plume equation for line source.

Where:

$\bar{c}$ represents the average concentration of the pollutant; Q represents the mass emission rate; $\bar{u}$ represents the average wind speed; L represents the length of the line source section; σ_z and σ_y represent spread parameters in the vertical and lateral direction respectively, which are linked to the downwind component of the distance from the receptor to the line section, and which adopted the calculation equations and empirical calculation parameters proposed by Briggs (1973); y_1 and y_2 represent the crosswind component of distances from the receptor to the two ends of the line section; z represents the vertical component of the distance from the receptor to the line section; h represents the effective source height which adopted 2m in this study in line with the assumption made in OSPM.

5.2.3.3 The relationship between NO_X, NO_2 and O_3

The concentrations of NO_2 were estimated from the concentration of NO_X using experience functions (Clapp and Jenkin, 2001). Because NO_2 and O_3 were reactive in the air, the Gaussian dispersion model was not suitable to estimate their concentration. However, the concentration of NO_X, in the form of NO and NO_2 which convert to each other, remain constant with the unit of ppb. Therefore, the concentration of NO_X were first estimated by the Gaussian model, and experience functions of the relationships among NO_X, NO_2 and O_3 were used to assess the concentration of NO_2.

The experience functions were chosen from a study which analysed the relationship among ambient concentrations of O_3, NO_2, NO and NO_X at different sites in the UK (Clapp & Jenkin 2001). The study derived a general expression of the function of the relationship between the annual average concentration of the "oxidant" (noted as OX, taken to be the sum of O_3 and NO_2) and NO_X, and a general function between the ratio of NO_2 to OX and NO_X, with different parameters at different sites.

The parameters for a site in London were chosen in this study. The reactions among NO, NO_2 and O_3 were influenced by the original concentration of NO_X, VOC and O_3, the original ratio of NO_2 and NO in NO_X, solar radiation, and weather conditions. The site from which the parameters were chosen shared many similarities in these respects with Dublin. As this site is in London, it has similar geography, weather and solar radiation to the places in Dublin, and traffic is the main cause of the NO_X, VOC and O_3 pollution for both regions. Also the range of NO_X concentration and the "oxidant" concentration of the site were similar to this study. The annual average NO_X concentrations of the site were reported to range from 10 to 90 ppb (Clapp and Jenkin, 2001). From the records of air monitoring sites in Dublin, the annual average NO_X concentrations in Dublin from 2001 to 2013 were also in the range of 10 to 90 ppb. Moreover, the annual average "oxidant" concentrations of the site ranged from 35 to 50 ppb, and the annual average "oxidant" concentrations for Dublin also fell within this range. Therefore, the choice of the parameters of the site could ensure that these parameters were representative of the conditions of Dublin.

The experience functions used in this study are shown in Equation 5-2 and Equation 5-3.

$$[OX] = 0.1272\ [NO_X] + B$$

Equation 5-2: Relationship between annual average concentration of OX and NO_X (Clapp & Jenkin 2001).

$$\frac{[NO_2]}{[OX]} = 1.015 \times 10^{-1} + 1.367 \times 10^{-2}[NO_X] - 6.127 \times 10^{-5}[NO_X]^2 - 4.464 \times 10^{-8}[NO_X]^3$$

Equation 5-3: Relationship between the ratio of NO_2 to OX and the annual average concentration of NO_X (Clapp & Jenkin 2001).

Where, *[OX]* represents the concentration of the sum of O_3 and NO_2. *B* represent the background O_3 concentration. In this study, *B* = 22.3 ppb. The background O_3 concentrations were obtained from an air quality monitoring site in a suburb in Dublin

(Rathmines). Units of NO_X, NO_2 and O_3 are in ppb. The applicable range of the equations is 10-90 ppb NO_X.

An annual average pollutant concentration (2013) was estimated for each point within a 40m × 40m grid throughout Dublin. There were 228,000 points in total. In order to represent the concentration in each ED, the concentrations at each point within each ED were averaged to represent the mean concentration of each ED. For Dublin, the mean concentration and the population size of each ED were utilized to calculate the population-averaged pollutant concentration.

5.2.4 Population data and baseline incidence rates

The population data for each ED were obtained from Central Statistics Office (CSO) of Ireland. As CSO conducted a census in 2011, only the data for 2011 were available. Therefore, an annual population growth rate of 0.3% was implemented to estimate the population for each ED in 2013, which corresponded to the real situation between 2011 and 2013. The baseline incidence rate for mortality in Dublin was assumed to be the same as the incidence rate in Ireland. The baseline incidence rate data for mortality in 2013 were also obtained from CSO. Table 5-3 lists the population and baseline mortality incidence rates for each pollutant pertaining to each epidemiological study in Table 5-1.

Baseline scenario code	Pollutant	Health indicator	Population affected (age group)	Number of baseline incidence	Number of baseline incidence per 100,000
DB	$PM_{2.5}$	All-cause mortality	585013 (30-99 yrs)	6164	1054
	NO_2	All-cause mortality	732460 (18-99 yrs)	6239	852

Table 5-3: Population and baseline mortality incidence information.

Figure 5-2 shows the population for each ED in 2013. The EDs with the populations artificially indicated as <1 in Figure 5-2 indicates that they were not considered by this study as they were not covered by the road network examined by this study.

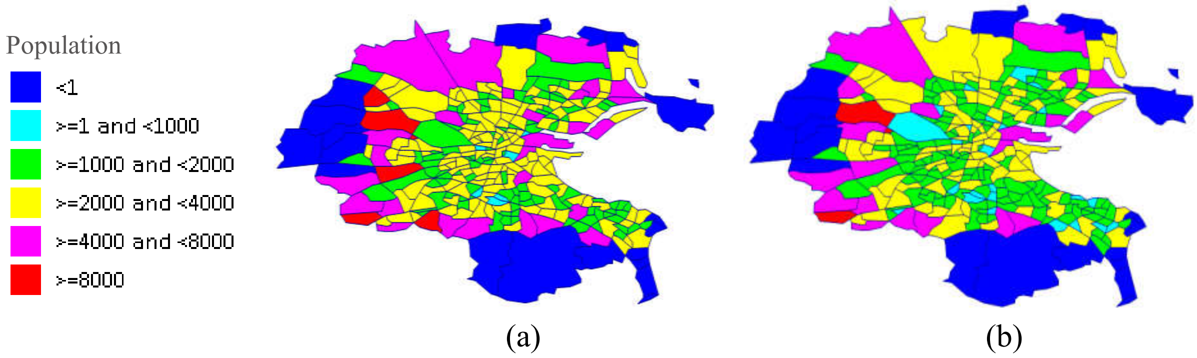


Figure 5-2: Dublin population aged over 18 (a) and over 30 (b) in 2013 for each ED.

5.3 Result

5.3.1 Fleet composition changes

The predicted changes in NO_2 concentration and consequential health impacts brought about by fleet composition changes are shown in Figure 5-3 and Figure 5-4. The predicted change in concentrations for $PM_{2.5}$ and consequential health impacts by fleet composition changes are shown in Figure C-1 and Figure C-4 in the Appendix C.

Figure 5-3 depicts the NO_2 concentration changes induced by fleet composition changes (scenarios Cars-50%, Cars-100% and Cars&Vans-100%). From Figure 5-3 we can see that the more diesel vehicles were replaced by petrol vehicles, the more the NO_2 concentration decreased. With 50% of diesel cars shifting to petrol cars (scenario Cars-50%), the decreases in average NO_2 concentration for most of the Dublin area were in the range of 0 to 0.2 μg/m^3 (see Figure 5-3 (a)). This amounted to a 0 to 1% decrease in NO_2 concentration. Some EDs with a motorway passing through them had an average concentration drop of 0.2 to 1 μg/m^3, giving a 1% to 7% decrease in NO_2 concentration. This was because the traffic volumes on the motorway were high and the fleet composition change affected the motorway emission more than the roads within the city centre. With 100% of diesel cars being replaced by petrol cars (scenario Cars-100%), the NO_2 concentrations in some EDs in the city centre, and some EDs that had motorways passing through, were lowered by 0.2 to 2 μg/m^3 (see Figure 5-3 (b)). This represented a 1% to 13% decrease in NO_2 concentration. The concentrations in other EDs were predicted to decline by less (0 to 1%). These results show that air pollution concentrations in the city centre area and the areas that are in the vicinity of the motorway were affected to a greater extent by this hypothetical strategy.

In Figure 5-3 (c), the impact of 100% diesel cars and vans converting to petrol vehicles was predicted to have a greater influence on NO_2 concentration. The NO_2 concentration for the majority of areas was decreased by 1% to more than 13%. Compared to Figure 5-3 (b), this demonstrates that shifting all the diesel vans to petrol vans had a significant impact on NO_2 concentrations, even with the amount of vans being far less than cars (ratio of 7:1).

From Figure 5-4 we can see that the overall trend of health impacts of fleet composition changes for different EDs was predicted to be a combination of the trend of NO_2 concentration changes and the feature of population distribution. EDs with larger NO_2 concentration changes and more populations were predicted to experience greater health impacts.

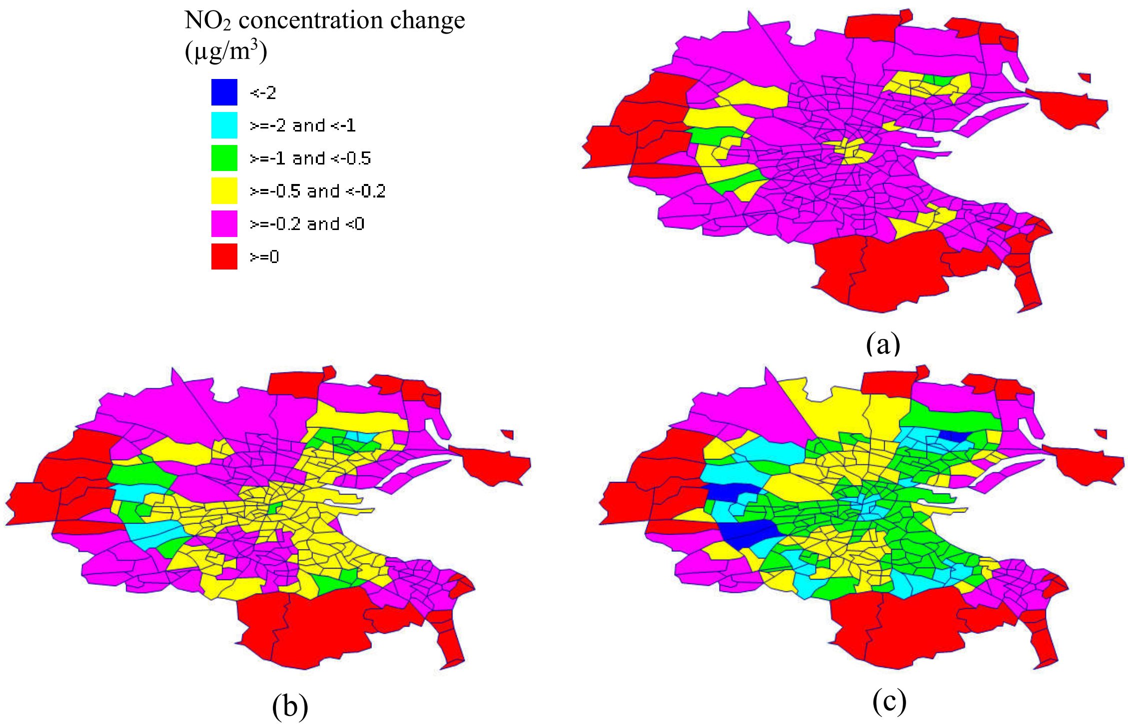


Figure 5-3: Predicted increase in annual average NO_2 concentration ($\mu g/m^3$), averaged across each ED, and brought about by the scenarios of (a) 50%, and (b) 100% of diesel cars converting to petrol cars, and (c) 100% diesel cars and vans converting to petrol vehicles, compared to baseline scenario.

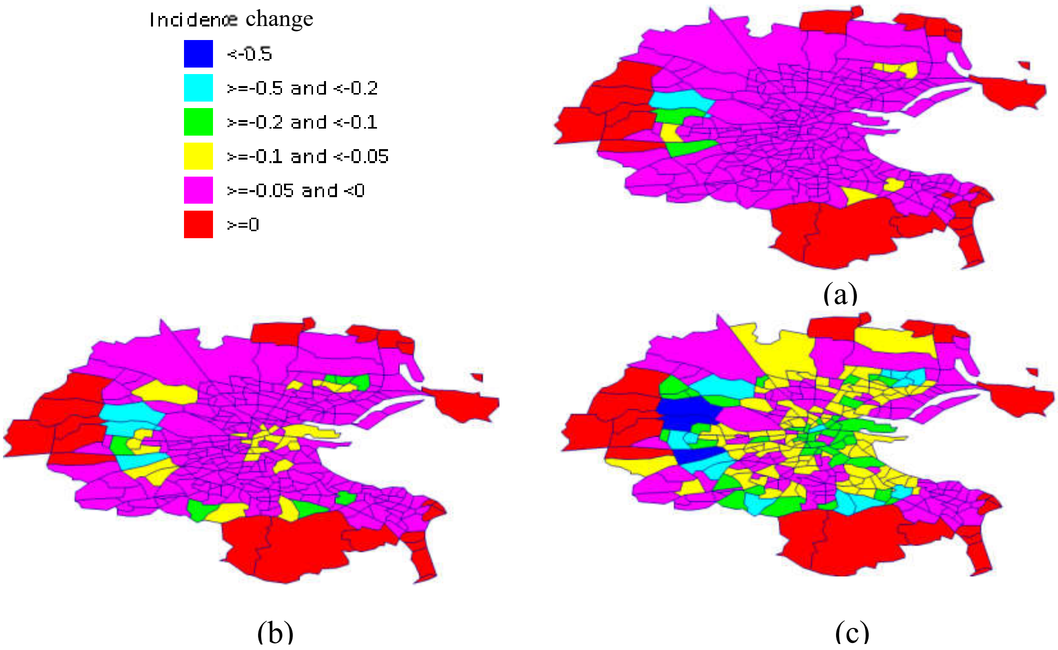


Figure 5-4: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased NO_2 brought about by the scenarios of (a) 50% and (b) 100% of diesel cars converting to petrol cars, and (c) 100% diesel cars and vans converting to petrol vehicles, compared to baseline scenario.

5.3.2 Infrastructure and traffic regulation changes

In Figure 5-5, the impacts of the infrastructure and traffic management regulation on NO_2 concentration are depicted (scenarios no-HGV and no-DPT). The consequential health impacts are shown in Figure 5-6. The predicted change in concentrations for $PM_{2.5}$ and consequential health impacts brought about by infrastructure and traffic regulation changes are shown in Figure C-2 and Figure C-5 in the Appendix C.

From Figure 5-5 (a) we can see that without the HGV management strategy, NO_2 concentration in the city centre was predicted to be over 1% higher. In contrast, it was predicted to decrease NO_2 concentration in most EDs in the north and west and some EDs in the south of Dublin without this strategy. These decreases were also modest at 0 to 1%. This illustrates that the implementation of the HGV management may help to improve city centre NO_X concentrations marginally, but it may also marginally exacerbate the pollution elsewhere outside the city centre due to the diversion of HGV traffic to other areas. Diversions of HGV traffic to the outer ring round require some of these vehicles to travel greater distances when traveling south or west, as they must first travel north and then circumvent the city. In both cases the change of NO_2 was predicted to be very small, however it should be noted that levels of traffic and NO_2 pollution in Dublin are low in comparison to many larger cities. As such, the findings of the impacts of such traffic diversion may be more significant in other locations.

In the scenario assessing Dublin air quality without the HGV management strategy and the DPT in place (scenario no-DPT) (see Figure 5-5 (b)), the concentration of NO_2 was predicted to increase over a larger area compared to Figure 5-5 (a). The concentration in the city centre was increased by more than 1%, and the concentration of some EDs in the north-west, north-east and in the south were increased by up to 1%. The concentrations of some EDs near the DPT were declined by more than 3%, and the concentrations of some EDs in the north and south of Dublin away from the city centre were decreased by up to 3%. This demonstrates that the presence of the DPT may result in reallocation of travel routes to the benefit of a larger area in Dublin regarding to NO_2 pollution. It may also worsen the environment in the vicinity of the DPT itself. The absolute values of the predicted changes were again small, considering that the annual average NO_2 concentrations in Dublin is generally low (15.78 $\mu g/m^3$).

Similar to the health impact trend of fleet composition changes, Figure 5-6 that EDs with larger NO_2 concentration changes and more populations were predicted to have greater health impacts.

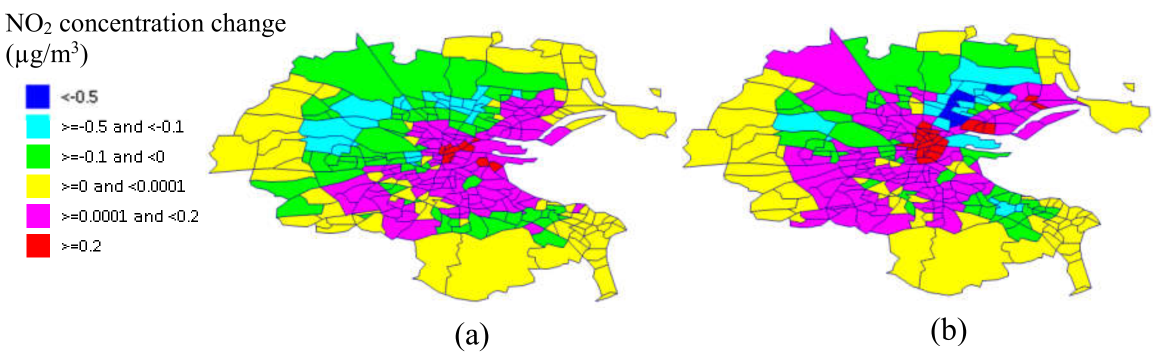


Figure 5-5: Predicted increase in annual average NO_2 concentration ($\mu g/m^3$), averaged across each ED, and brought about by the scenarios of (a) HGV management not implemented and (b) HGV management not implemented plus DPT not opened, compared to baseline scenario.

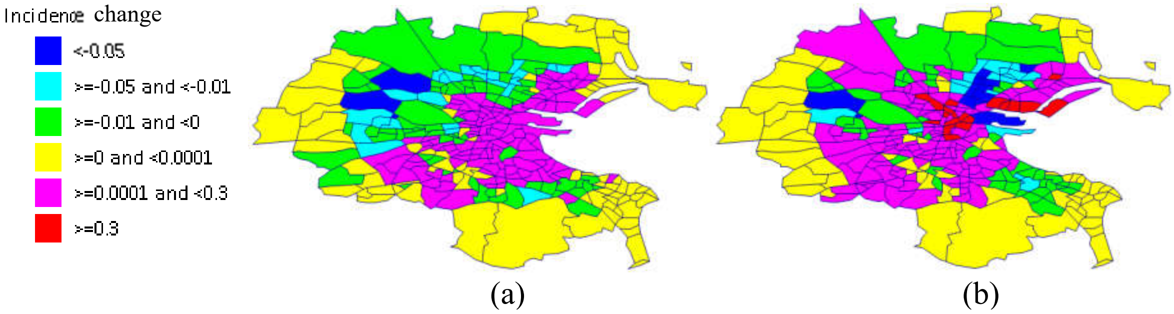


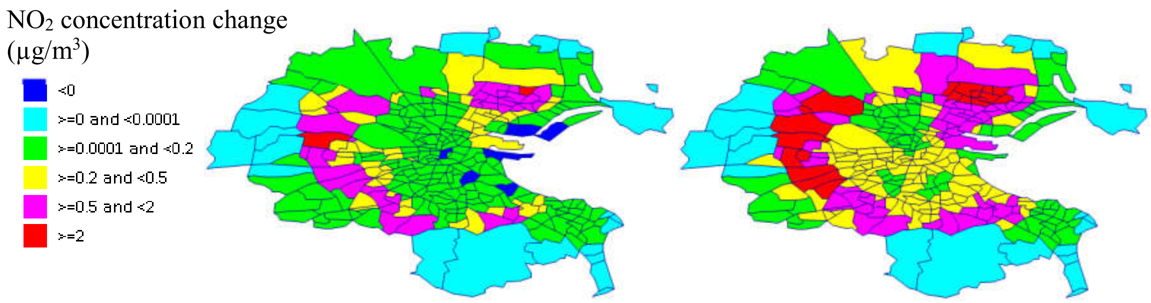
Figure 5-6: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased NO_2 brought about by the scenarios of (a) HGV management not implemented, and (b) HGV management not implemented plus DPT not opened, compared to baseline scenario.

5.3.3 Speed limit changes

Figure 5-7 and Figure 5-8 shows the impacts of the speed limit changes (scenarios Speed-40 and Speed-30) on NO_2 concentration and their consequential health effects. The predicted change in concentrations for $PM_{2.5}$ and consequential health impacts brought about by speed limit changes are shown in Figure C-3 and Figure C-6 in the Appendix C.

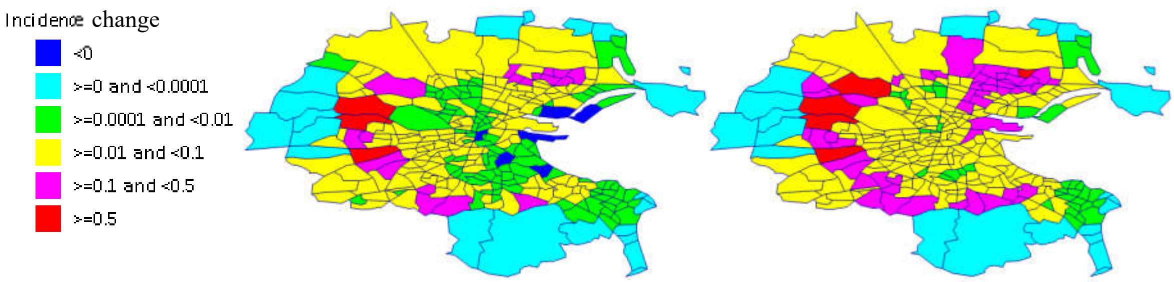
Figure 5-7 (a) represents the change of NO_2 concentration for the default speed limit changing from 50 km/h to 40 km/h (scenario Speed-40). This predicts that concentrations would decrease in a few EDs, and increase in the most of the EDs by up to 13%. The concentration in the majority of Dublin rose by 0 to 1%. Although the default speed limit change didn't affect the speed limit on the motorway, some EDs close to the motorway were affected by the speed limit change even more, with the concentration increased by 1% to more than 13%. In Figure 5-7 (b) (scenario Speed-30), the average concentration for most areas was increased by 1% to more than 13% and the concentration of all of the EDs was increased to some extent. This shows that the speed limit reducing from 50km/h to 30km/h had negative impact on NO_2 pollution. Also for this scenario, the concentration increases for some EDs close to the motorway were in the range of 3% to more than 13%, and this was more than for the majority of other areas. This phenomenon suggests that the change of default speed limit in the city not only affected the area in which the speed limit changed, but also could redistribute the traffic resulting in more vehicles detouring around the city centre, worsening the traffic emissions on the motorways.

Again, Figure 5-8 shows that EDs with larger NO_2 concentration changes and more populations were predicted to have greater health impacts.



(a) (b)

Figure 5-7: Predicted increase in annual average NO_2 concentration (µg/m^3) averaged across each ED and brought about by the scenarios of the default speed limit of (a) 40 km/h and (b) 30 km/h compared to baseline scenario.



(a) (b)

Figure 5-8: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased NO_2 brought about by the scenarios of (a) the default speed limit of 40 km/h, and (b) 30 km/h compared to baseline scenario.

5.3.4 Aggregated health impact evaluation

The integrated health impact for Dublin regarding to the air quality changes of $PM_{2.5}$ and NO_2 relating to each scenarios are summarised in Table 5-4 and Table 5-5, respectively.

Table 5-4 illustrates the impact of $PM_{2.5}$ changes on all-cause mortality incidence for the population over 30 years old in Dublin in 2013 brought about by different scenarios. The fleet changes could reduce the average $PM_{2.5}$ concentration in Dublin by 0.24 µg/m^3 at maximum, which is about 2%, and could diminish the all-cause mortality incidence by 8.52 (95% CI: -5.75 to -11.27) at maximum, with the scenario of all the diesel cars and vans converting to petrol vehicles (scenario Cars&Van-100%). The scenarios of the traffic regulation change and this change plus the infrastructural change (scenarios no-HGV and no-DPT), which increased the concentration and mortality incidence in city centre and decreased these numbers on some other places, had little influence on the average air quality and mortality incidence when considering the whole city, with the increases of 0.04 (95% CI: 0.02 to 0.05) and 0.02 (95% CI: 0.01 to 0.02) in mortality incidence respectively. The speed limit changes would marginally increase the $PM_{2.5}$ concentration and mortality incidence. The speed limit changes from 50km/h to 40km/h and 30km/h (scenarios Speed-40 and Speed-30) increased the $PM_{2.5}$ concentration by 0.08 $\mu g/m^3$ and 0.18 $\mu g/m^3$, accounting for 1% and 2% increases respectively, and increased the mortality incidence by 2.98 (95% CI: 2.01 to 3.94) and 6.45 (95% CI: 4.36 to 8.52), respectively.

From Table 5-5 we can see that compared to $PM_{2.5}$, the concentration of NO_2 was more sensitive to traffic strategies. The fleet change could reduce the NO_2 concentration by 0.69 $\mu g/m^3$ at maximum with the hypothetical strategy of 100% diesel cars and vans converting to petrol vehicles (scenario Cars&Van-100%), which was a 4% decrease. This change was connected to a 23.55 (95% CI: 13.03 to 34.00) decrease in the all-cause mortality incidence for the population over 18 years old in Dublin. The infrastructural change and the traffic regulation change (scenario no-HGV and no-DPT) had little influence on the average concentration of NO_2 and associated mortality incidence. Besides the influence on the $PM_{2.5}$ concentration, the speed limit changes (scenarios Speed-40 and Speed-30) could also increase the average NO_2 concentration significantly, by 0.65 $\mu g/m^3$ at maximum, accounting for a 4% increase, and linked to a mortality incidence increase by 22.41 (95% CI: 12.40 to 32.35) at maximum.

The mortality incidence changes brought by the changes of different pollutants should not be simply added or deducted to get the total incidence change for each scenario. Because epidemiological researches from which the CRFs were drawn on different pollutants are not all exclusive to other pollutants.

Scenario code	Pollutant	Health indicator	Population-weighted concentration / change (µg/m³)	Population at risk (age group)	Number of predicted increased deaths (95% CI)	Number of predicted increased death per 100,000 (95% CI)
Baseline	$PM_{2.5}$	All-cause mortality	11.55	585013 (30-99 years)	n/a	
Cars-50%			-0.06		-2.33 (-1.57 to -3.08)	-0.40 (-0.27 to -0.53)
Cars-100%			-0.13		-4.84 (-3.27 to -6.40)	-0.83 (-0.56 to -1.09)
Cars&Vans-100%			-0.24		-8.52 (-5.75 to -11.27)	-1.46 (-0.98 to -1.93)
no-HGV			0.0010		0.04 (0.02 to 0.05)	0.01 (0.00 to 0.01)
no-DPT			0.0005		0.02 (0.01 to 0.02)	0 (0 to 0)
Speed-30			0.18		6.45 (4.36 to 8.52)	1.10 (0.74 to 1.46)
Speed-40			0.08		2.98 (2.01 to 3.94)	0.51 (0.34 to 0.67)

Table 5-4: Integrated predicted impact of $PM_{2.5}$ change on all-cause mortality for Dublin brought about by all the scenarios.

Scenario code	Pollutant	Health indicator	Population-weighted concentration / change (µg/m³)	Population at risk (age group)	Number of predicted increased deaths (95% CI)	Number of predicted increased death per 100,000 (95% CI)
Baseline	NO_2	All-cause mortality	15.78	732460 (18-99 years)	n/a	
Cars-50%			-0.13		-4.43 (-2.45 to -6.39)	-0.60 (-0.33 to -0.87)
Cars-100%			-0.27		-9.11 (-5.04 to -13.15)	-1.24 (-0.69 to -1.80)
Cars&Vans-100%			-0.69		-23.55 (-13.03 to -34.00)	-3.21 (-1.78 to -4.64)
no-HGV			0.01		0.19 (0.11 to 0.28)	0.03 (0.01 to 0.04)
no-DPT			0.01		0.18 (0.10 to 0.25)	0.02 (0.01 to 0.03)
Speed-30			0.65		22.41 (12.40 to 32.35)	3.06 (1.69 to 4.42)
Speed-40			0.30		10.13 (5.61 to 14.63)	1.38 (0.77 to 2.00)

Table 5-5: Integrated predicted impact of NO_2 change on all-cause mortality for Dublin brought about by all the scenarios.

5.4 Discussion

5.4.1 Cross-comparison

The study assessed the possible impacts of different types of traffic strategies on air quality and public health. These strategies include infrastructural change, traffic management regulation change, speed limit changes and fleet composition changes.

From the result we can see that the infrastructural change and traffic management regulation change might redistribute the traffic, but the influence was limited and had little impact on the predicted city-wide averaged pollutant concentration and overall mortality incidence change, compared to other strategies. The absence of the HGV management strategy was associated with a predicted increase of all-cause mortality of 0.04 due to the increase on $PM_{2.5}$ and 0.19 due to the increase on NO_2, which suggests that the HGV management had very little impacts on improving the population-weighted average air quality. Without the DPT however, the HGV management could not be implemented because the DPT provides an alternative route for connecting Dublin port to the motorway network. Without the DPT, the HGVs would have to travel through the city centre of Dublin. This example shows that a good road network design together with a good traffic management regulation policy could bring about a positive effect on the environment. However, the influence of the DPT and the HGV management was limited. This is because the DPT is only one road additional to the original road network, and HGV management was only effective in the city centre and for only HGVs, not for all the vehicles.

The most important role for infrastructural construction is to satisfy the travel requirement of people, reduce the congestion and boost the economy, and infrastructural construction is not an ideal approach to reduce the traffic pollution in an old, already-shaped city. Because the construction of new infrastructure can be highly expensive, and it can take a long time to complete the construction, moreover, the environmental improvement of this strategy on an old city is limited. However, land use planning and road network planning for a new city is worthwhile because a holistic approach to planning can help optimize traffic conditions and distribution, and reduce unnecessary travel demand, which is important for reducing the traffic-related pollution.

The results suggest that fleet composition can be an important factor that influences the traffic pollution and relevant mortality incidence. Shifting diesel vehicles to petrol vehicles were predicted to have positive impacts on reducing the $PM_{2.5}$ and NO_2 concentration, and these impacts brought about notable benefits on reducing the all-cause mortality incidence. As $PM_{2.5}$, NO_2 are pollutants that affect the public health and are most related to traffic, this study focused on the health impact of these pollutants. However, it is important to include as many pollutant types as possible to make a comprehensive assessment of the health impact of changing fleet composition, because the impacts of fleet composition on different pollutants can be varied. Although converting diesel vehicles to petrol vehicles could have positive impacts on mortality incidence changes related to $PM_{2.5}$ and NO_2, studies have found that for some other pollutants (e.g. CO and benzene (Subramanian & Babu 2013)), petrol cars can emit more of these pollutants than diesel cars. Hence it is crucial to evaluate the level of the impacts of changing fleet composition on these pollutants and on relevant health outcomes, and to compare these impacts with the impacts of $PM_{2.5}$ and NO_2.

Regarding feasibility, changing fleet composition can take a lot of time and involves significant management resources. But as fleet composition can influence the traffic pollution substantially, it is important for the government to stimulate people to choose the types of vehicles that are less harmful to public health (in terms of air pollution). Both petrol and diesel vehicles have their disadvantages in producing pollutant emissions, thus in the long run it is worthwhile to phase out both of them and encourage alternative fuel vehicles. The results also show that the larger vehicles influence the air quality more notably (vans were predicted to have more impact than cars). Therefore, investing in upgraded technology and improving the performance of larger vehicles is important.

The change of a traffic speed limit is usually made on the basis of safety considerations. The fatalities caused by traffic accidents in Dublin in 2013 were 19 people (Road Safety Authority, 2015a), which is in the same order of magnitude as the predicted increase in the mortality incidence associated to the predicted NO_2 increase brought about by speed limit reduction from 50km/h to 30km/h (22.41 deaths). This suggests that the joint consideration of traffic accidents and health impact from traffic pollution, should be conducted when finding optimum speed limits for the city. Moreover, the results showed that there would be decreases in $PM_{2.5}$ concentration in some EDs with the scenario of speed limit changing from 50km/h to 40km/h, which demonstrated that 40km/h speed limit could be a better choice for those EDs regarding traffic pollution. This implies that implementing a different speed limit in different areas, according to the traffic conditions of that area, could be an effective approach to reduce the health impact of traffic pollution.

Compared to other traffic strategies, planning and changing speed limit regulation is an easily implementable control, the most economic and the quickest approach of the scenarios examined here.

To sum up, among all these strategies, the determination and implementation of optimized speed limits for different areas is the most feasible and efficient way for the government with a main target to reduce the traffic pollution in the short term.

All these traffic strategies had greater impact on the average concentration of NO_2 than of $PM_{2.5}$. Air quality in Dublin is influenced by numerous sources. PM pollution is known to be dominated by solid fuel burning (Ovadnevaite et al. 2017), with transport as the second largest contributor (EPA, 2016), whereas the major source of NO_2 is transport (EPA, 2016). Governments can mainly focus on the design and implementation of different traffic strategies in order to achieve the target of NO_2 concentration reduction, but the reduction of PM pollution must rely on the cooperation of different sectors that produce particulates.

Traffic management strategies would have greater impact when traffic is heavier. With the recent economic growth and development of the city, traffic management strategies would have a larger impact on air quality and public health now than in 2013, the year for which the modelling scenarios were run; this was a relatively depressed year economically, and so traffic volumes were less and air quality in Dublin was generally good. The condition of the traffic level and air quality would be different again for Dublin in the future, and for other larger cities. The evaluation of traffic management strategies in the future for Dublin and other cities with heavy traffic levels is therefore necessary.

5.4.2 Uncertainty

Uncertainty of the result includes uncertainty from the input data and from the uncertainty embedded in the modelling. This study employed a chain of models, which could expand the uncertainty and made it difficult to estimate. This study reduced the uncertainty to a minimum by employing the models that were widely-used and complying with the scope of the study and collecting the data in detail from authorized institutions.

The traffic model was developed in VISUM, a macroscopic model, which was complied with the objective of the study, i.e. focusing on a city for a year. The traffic volume was calibrated and validated with the data of traffic counts in 2013 in Dublin (Tang et al. 2017). The traffic emissions were calculated by implementing the method and the emission factors of European Emission Model COPERT4, which is one of the leading European Emission models and is widely-used (Sjödin & Jerksjö 2008).The emission factors of COPERT4 were suitable to estimate the emissions of 2013, as it was the most updated emission factors in 2013 and thus was able to be used to represent the condition of 2013. The method of the calculation of traffic emissions of COPERT4 was based on average speed, not on second-by-second simulation. Although an average-based emission model is suitable for a larger area (e.g. a city) and across a relatively long time (e.g. a year), research has found that aggressive acceleration and deceleration are associated with more traffic pollutant emissions and these instantaneous effects were not captured in this modelling approach (Alam & McNabola 2018; Huang et al. 2018). The consideration of acceleration and deceleration by implementing second-by-second simulation would be desirable for more precise results when estimating the speed limit impacts in the future.

The performance of OSPM had been widely researched and a good fit between the modelled and observed pollutant concentrations was found (Ketzel et al. 2012; Berkowicz et al. 2008). Also research has found that the combination of the Gaussian dispersion model and the OSPM could reduce the model error of the Gaussian dispersion model or the OSPM on its own (Fallah-Shorshani et al. 2017).

The studies from which the CRFs and their parameters were derived were chosen that had the same metrics of time as this study (i.e. focusing annual or long-term impact), and involving as much population as possible to get reliable CRFs. Besides these standards, the epidemiological studies that relied on the measured pollutant concentrations instead of modelling concentrations to estimate the parameters of CRFs were preferred in this study, in order to reduce uncertainty. For example, the reason that we chose the US study of Krewski et al. (2009) instead of a European study such as Cesaroni et al. (2013), was because of their reliance on modelled concentrations of $PM_{2.5}$. For the health impact of $PM_{2.5}$, the US cohort studies were chosen because the reliable European cohort studies were few (Cesaroni et al. 2013). Also, this study belongs to American Cancer Study (ACS), the result of which was also adopted by a UK authority COMEAP (Ayres, 2010). It should be pointed out that although the CRFs were chosen as the best in the literature, the nature of the time and space variance of the CRFs could contribute to the uncertainty of the modelling results.

The input data, e.g. traffic counts, fleet compositions, population data, baseline mortality incidence data, etc., were derived from official institutions in order to ensure the quality and reduce the uncertainty.

A significant limitation is that the study only evaluated the mortality incidence changes associated to the changes of $PM_{2.5}$ and NO_2 concentration because of the lack of reliable CRFs and baseline incidence data for other pollutants. Other health impact indicators, e.g. morbidity incidences for diseases that are linked to these pollutions, and the health impact of other pollutants that are influenced by traffic, are also worth investigating in order to obtain a more comprehensive analysis.

5.5 Conclusion

This study evaluated air quality and the potential health impacts of different traffic management strategies. A modelling chain containing the traffic model, emission model, dispersion model and the health impact assessment model was employed. Strategies of the infrastructural change and the traffic regulation change were found to have influences on the local air quality and mortality incidence rather than on a larger area. In this study, these two strategies were predicted to reduce the traffic pollutant concentrations and mortality incidence at one place while increase the concentrations and incidence at another place. When considering the influence on the whole city, the benefits brought about by these two strategies at one place were cancelled out by the disadvantage at another place, and made little contribution on the improvement of the environment for the whole city. The increased mortality incidence associated with increased NO_2 concentration brought about by the scenario of changing speed limit from 50km/h to 30km/h, was predicted to be of the same order of magnitude as fatalities as caused by traffic accidents in Dublin. This emphasizes the importance for a government to consider the possible air quality impacts when determining traffic speed limits. The fleet composition changing from diesel vehicles to petrol vehicles was predicted to have positive impact on the environment and mortality incidences in this study. As diesel vehicles and petrol vehicles both have their advantages and disadvantages regarding the emission of different pollutants, and these two vehicles are predominant in the market - and it would be difficult to find an alternative that can be widely introduced in a short time - a more comprehensive assessment is needed to analyse the health impact of both types of vehicles.

Considering the efficacy, feasibility, economy and the time, finding an optimum speed limit that takes the factors of traffic accidents, traffic condition and the environment into account is the most efficient and the quickest strategy to alleviate traffic pollutions among all these strategies.

Chapter 6 Discussion

6.1 Traffic strategies, GHGs emissions and global warming

The impacts of traffic management strategies on public health were discussed in Chapter 5, this section is intended to discuss the impacts of these strategies on GHG emissions and the environment broadly.

6.1.1 Traffic strategies and CO_2 emissions

The health impact of the construction of the DPT, the HGV traffic management regulation, speed limit changes and fleet composition changes were estimated in Chapter 5. These traffic management strategies also had significant impacts on GHGs emissions and thus influence climate change.

In Chapter 3, the opening of the DPT was estimated to bring about a 3% increase in CO_2 emission, from 226 tonnes to 232 tonnes, and the DPT with the HGV management strategy together caused a 7% increase in CO_2, from 226 to 241 tonnes, mainly because of vehicle re-routing and detouring. This estimation was conducted only for traffic emissions of an average hour in morning rush hours (7 to 9 am) across the year of 2006. Accounting for the traffic emission for a full year, a 7% increase in road transport in a city would be a notable increase, assuming the amount of detouring remains unchanged during off-peak hours.

Some researchers have found similar impacts from similar traffic management strategies to this research. Mitchell et al. (2002) predicted that the completion of two roads in Leeds, UK would bring about a 1.4% increase in CO_2 emission, again because vehicles were travelling further as they were re-routing to take the advantage of the new infrastructure. Lozano et al. (2014) estimated that the construction of three bridges in Mexico City could reduce vehicle travel time, but would increase the vehicle travel distance by 2% in the long term. Increases in travel distances and total emissions were also found for the implementation of a LEZ in Coimbra, Portugal (Dias et al., 2016).

Previous research also found some positive impacts of road construction and traffic regulations on environment. For example, the licence plate restriction policy, which a number of cities in developing countries have used to reduce urban air pollution and traffic congestion, prohibiting vehicles from entering the restriction zone at a particular time based on the last digits of their vehicle's licence plates, were found to cause total emissions to decrease by 4.7% (Pu et al., 2015). This predicted emission decrease was due to the assumption of a 20% reduction in travel demand. This shows that traffic regulations which reduce travel demand could have positive impacts on environment. However, as reported in this thesis, unlike private cars, HGVs usually have low travel demand elasticity. The negative impacts of detours overwhelmed the benefits of improved speed distributions brought about by the HGV management strategy on emissions; there was no travel demand decrease, and so no additional benefit for the environment. On the contrary, the DPT and HGV management strategy stimulated the travel demand of cars and thus brought about additional CO_2 emission (comparing 2006 DPT+Ban scenario and 2007 DPT+Ban scenario in Figure 3-7 and Figure 3-10).

Therefore, the impact of infrastructural construction and traffic regulation on the total emission of GHG is complex. Their impact on travel demands, traffic speeds, and travel distance should be all taken into consideration in the assessment of these traffic management strategies.

The impacts of speed limit and fleet composition on GHG emissions were not assessed in this thesis, as these impacts have already been thoroughly investigated. There is a consensus in the literature that diesel vehicles, on average, consume less fuel (per litre) and emit less CO_2 (in g/km) than comparable petrol vehicles and therefore are favourable from a GHG perspective (IARC, 1989; EEA, 2017; Fontaras et al., 2017). However, with the improvement of vehicle technology, both diesel and petrol vehicles emit less and less CO_2 per kilometre every year (on average), and the difference in the CO_2 emission between diesel and petrol vehicles has reduced. In 2016, diesel-fuelled passenger cars emitted an average 116.8 g CO_2/km in Europe, which is 4% (4.9 g CO_2/km) less than from the average petrol vehicle. Whereas in the year 2000, the CO_2 emission of diesel vehicles was 160.3 g/km on average in Europe, and the difference between diesel and petrol vehicles was over 10% (17.1 g CO_2/km), much larger than 2016 (EEA, 2017).

For all types of fossil fuel powered vehicles, the relationship between the emission rate of CO_2 and vehicle speed presents a "U" shape, with notably higher emissions at low and high speeds. The lowest emissions rates are achieved at speeds between 40 to 80 km/h, depending on fuel type, engine size and Euro class of the vehicle. So therefore low speed limits and low speed traffic can affect overall CO_2 emissions. Previous research highlighted field trials of 30 cars showing that on average, 30 and 40 mph (48 and 64 km/h) speed limits would cause CO_2 emission decreases by 17% and 26% compared to a 20 mph (32 km/h) speed limit (Carsten et al., 2008). However different research on this topic has produced different results leaving a lack of consensus in the literature.

Microsimulation results demonstrated that both increases and decreases in the emission of CO_2 can be observed in the action of a speed limit reduction (50 km/h reducing to 30 km/h speed limit), depending on the conditions (Panis et al., 2011). Although lower speed is associated with higher emissions, higher speed limits can also involve more actions of acceleration and deceleration in an urban context which would also increase the CO_2 emissions significantly. Therefore, field tests of emissions for different speed limits, different traffic conditions, involving more vehicles are desirable in order to better understand the impact of speed limits on emissions. In this thesis, the speed limit change from 30km/h to 50km/h were predicted to increase the average speeds (see Figure 4-3) and made these speeds close to the range of 40 to 80 km/h, which, based on average speed method, would reduce the CO_2 emissions.

6.1.2 Traffic strategies and other GHGs

Aside from CO_2, there are other GHGs that have more significant impacts on global warming and climate change. They have much larger Global Warming Potential (GWP), which estimates how much heat a greenhouse gas traps in the atmosphere and is expressed as a factor of CO_2 (whose GWP is standardized to 1). Two main GHGs that are related to road transport are CH_4 (methane) and N_2O (nitrous oxide). CH_4 has a relatively short lifetime of 12 years, with a GWP of 28 over 100 years. N_2O has even greater impacts on global warming, as it has a GWP of 265 over 100 years and has a longer lifetime of 121 years (Myhre et al., 2013).

In the thesis, the DPT was estimated to increase the total emission of CH_4 by 5% (8.5 to 8.9 kg, i.e. 238 to 249.2 kg CO_2 equivalent or CO_2e considering GWP), and the DPT together with HGV management strategy would increase CH_4 by 8% (8.5 to 9.2 kg, i.e. 238 to 257.6 kg CO_2e). These increases in CH_4 emissions were mainly due to the impacts of vehicle detours (see section 3.3). The impacts of vehicle speed on the emission of CH_4

and N_2O were found to be limited. The emission rates of CH_4 and N_2O were relatively constant for varying vehicle speeds in the modelling calculation process and in the national inventory evaluation (IPCC, 2006; Ntziachristos et al., 2018). Thus impact of the speed limit changes and the improved speed distribution (see Figure 3-9) brought about by the DPT and HGV management strategy had almost no effect on the emission of CH_4 and N_2O.

The fleet composition had notable predicted impact on the emission of CH_4 and N_2O. On average petrol passenger cars can emit CH_4 at the rate of 10.8 mg/km (i.e. 302.4 mg/km CO_2e) and N_2O at 2.3 mg/km (i.e. 609.5 mg/km CO_2e), whereas diesel passenger cars emit much less GHGs, with CH_4 at the rate of 0.3 mg/km (i.e. 8.4 mg/km CO_2e) and N_2O at 0.6 mg/km (i.e. 159 mg/km CO_2e) (U.S. EPA, 2016).

Although modelled CH_4 and N_2O emission rates of vehicles were significantly less than the emission rate of CO_2, the impact of CH_4 and N_2O from traffic on the climate should not be ignored in view of their high global warming potential. A more comprehensive assessment of traffic management strategies should include the evaluation of their impacts on the emission of CH_4 and N_2O.

6.2 Comparison of strategies and future directions

6.2.1 Cross-comparison of traffic management strategies

A cross-comparison between the infrastructural change, the traffic management regulation change, speed limit changes and fleet composition changes in terms of their health impact was conducted in Chapter 5. Based on this, a cross-comparison of these four types of strategies is discussed below in this section in terms of their impacts on air pollutant emissions, GHG emissions and pollutant concentrations with the summary of their health impact.

Table 6-1 summarized all scenarios in Chapter 3, 4 and 5 from which we can infer the impacts of traffic management strategies. The baseline scenarios were different in Chapter 3, 4 and 5, and consequently the scenarios designed to evaluate the impacts of strategies were different. This is because baseline scenarios were designed to be in line with the actual traffic regulation and traffic condition at the modelling time and place, in order to perform calibration and validation for traffic model using traffic count data accurately. Although scenarios were designed differently in Chapter 3, 4 and 5, from Table 6-1 we can see that the impact of four types of traffic management strategies can be inferred by comparing the scenarios within each chapter. For the purpose of comparing the impacts among these four types of traffic management strategies directly in the same frame, the relative impacts of these strategies were shown in Figure 6-1 and Figure 6-2. Figure 6-1 and Figure 6-2 illustrate the relative impact of different traffic management strategies on GHG emissions, air pollutant emissions and public health.

	Scenario comparison	Change description	Impacts				
Chapter 3		scenarios in 2006	Emission changes				
			NO_X	PM	CO	CO_2	CH_4
	2006 DPT - 2006 no DPT	Opening of the DPT	+	+	+	+	+
	2006 DPT&HGV Ban - 2006 DPT	Implementation of the HGV management strategy	+	+	+	+	+
Chapter 4		scenarios in 2013	Concentration changes				
			NO_2	PM	CO	Benzene	
	Speed 40 - Speed 30	Speed limit change from 30 to 40 km/h	-	-	-	-	
	Speed 50 - Speed 30	Speed limit change from 30 to 50 km/h	-	-	-	-	
	30% Cars - 0% Cars	30% diesel cars shifting to petrol cars	-	-	+	+	
	60% Cars - 0% Cars	60% diesel cars shifting to petrol cars	-	-	+	+	
	100% Cars - 0% Cars	100% diesel cars shifting to petrol cars	-	-	+	+	
	100% Cars & Vans - 0% Cars & Vans	100% diesel cars and vans shifting to petrol cars and vans	-	-	+	+	
Chapter 5		scenarios in 2013	Mortality incidence changes				
			NO_2		$PM_{2.5}$		
	no HGV - DPT & HGV	Without HGV management strategy	+ (little)		+ (little)		
	no DPT - DPT & HGV	Without DPT and HGV management strategy	+ (little)		+ (little)		
	Speed 40 - Speed 50	Speed limit change from 50 to 40 km/h	+		+		
	Speed 30 - Speed 50	Speed limit change from 50 to 30 km/h	+		+		
	50% Cars - 0% Cars	50% diesel cars shifting to petrol cars	-		-		
	100% Cars - 0% Cars	100% diesel cars shifting to petrol cars	-		-		
	100% Cars & Vans - 0% Cars & Vans	100% diesel cars and vans shifting to petrol cars and vans	-		-		

Table 6-1: Summary of the impact of all scenarios on pollutions.

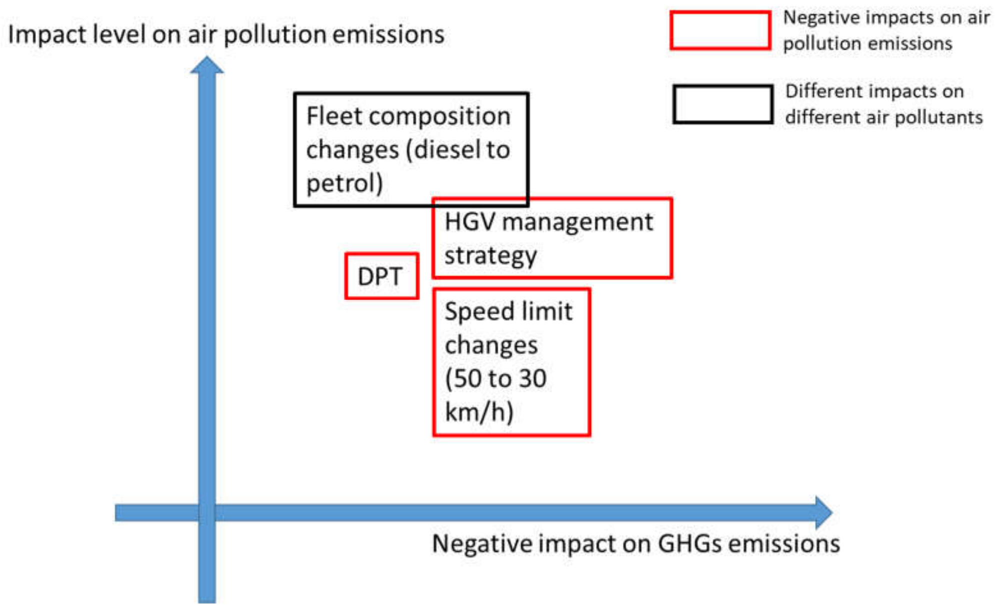


Figure 6-1: Traffic management strategies and their impact on GHGs emissions and air pollution emissions.

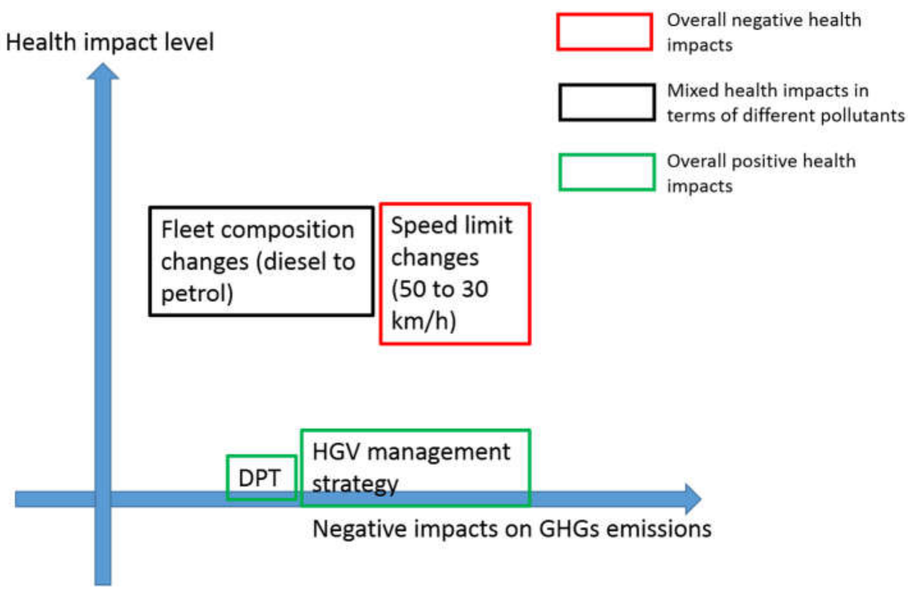


Figure 6-2: Traffic management strategies and their impact on GHGs emissions and health.

Infrastructure and traffic management regulation changes

The infrastructure construction and traffic management regulation changes were estimated to have impact on travel demands, traffic re-distributions, speed distributions and traffic emissions of air pollutants and GHGs. These strategies led to predicted increases in emissions for all types of pollutants and GHGs considered in this thesis (NO_X, PM, CO, CH_4, CO_2), mainly due to the impact of longer vehicle travel distances. Although it is difficult to quantify the impact of increasing emission of GHGs, there is no doubt that these strategies brought about negative effects on global warming. However, taking the population density into consideration, the health impact of these increasing air pollutants were found to be negligible. This phenomenon demonstrated the insufficiency of the estimation of only emissions in respect of air pollutant impacts, and the difference between the evaluation of the impacts on global warming and public health.

Speed limit changes

Speed limit changes were predicted to have influences on pollutant emissions and traffic-induced pollutant concentrations. Students in schools are particularly vulnerable for both air pollutants and traffic accidents. A trade-off was identified between reducing risk of traffic accidents and reducing the potential harmfulness of air pollution exposure for pupils. Traditional speed limit decisions usually take only the safety issues into consideration in this context. However, the predicted mortality increases caused by concentration of NO_2 increases brought about by speed limit changes from 50km/h to 30km/h were estimated to have the same order of magnitude as fatalities caused by traffic accidents in Dublin in 2013, which highlights the importance of this trade-off.

Fleet composition changes

Fleet composition changes also can affect emissions and traffic-induced pollutant concentrations. Unlike the three traffic management strategies stated before, which had relatively consistent impacts on different air pollutants and GHGs, the fleet composition had very different predicted impacts on different air pollutants and GHGs. The operating condition of petrol and diesel vehicle engines and the components of petrol and diesel fuels are different in many ways, thus the impact of different types of vehicles varied regarding different pollutants. To summarise, replacing diesel vehicles by comparable petrol vehicles could reduce the emissions and concentrations of $PM_{2.5}$, PM_{10}, NO_2, but would increase the emissions and concentrations of CO, Benzene, CO_2, CH_4 and N_2O and energy consumption. Considering two pollutants that are affecting people the most and causing illness and deaths the most, i.e. $PM_{2.5}$ and NO_2 (Health Effects Institute, 2010), shifting diesel to petrol vehicles were evaluated to save 9 and 24 people, respectively, from premature deaths in Dublin in 2013. Changes in other pollutant concentrations such as CO and benzene may have relative minor predicted impacts on public health, but may still offset these benefits to some extent. In addition, shifts from diesel to petrol would increase the GHG emissions and place extra pressures on global warming. This suggests a trade-off between considerations of the health impacts of different pollutants, and between the local health impacts and the effects of global warming and climate change.

Therefore, considering the high cost of new infrastructure construction, the potential for vehicle detours brought about by road restrictions, the negative impacts of both petrol and diesel vehicles on traffic emissions and also the relatively high management cost in the transition of fleet composition, the management of speed limits is the most efficient approach to control traffic emissions of pollutants and GHGs, and to reduce traffic impacts on air quality and public health, subjected to careful consideration of safety.

6.2.2 Future directions

A trade-off between reducing GHGs emissions/energy consumptions and local pollutants/health impacts has been identified in this thesis, which has also been highlighted by some other analyses of traffic management strategies e.g. (Mitchell et al., 2002) and (Bandeira et al., 2013). These trade-offs indicate that the policy-makers need to treat these strategies carefully and should endeavour to find more desirable solutions that are beneficial to both public health and the environment, and thus eliminating these trade-offs.

It was demonstrated in the thesis that all of these traffic management strategies have different impacts on modelled emissions of GHGs and local air quality. Therefore, future policies should be aimed at finding optimized strategies in terms of all these aspects. As traffic infrastructure and traffic management regulation were found to influence travel demands and travel distances which were the main reasons for total emission changes, and population density affected traffic pollutants' impacts on public health, a strategic land use planning and traffic management regulation design for a city that aims to reduce both travel demands and distances, and to optimize population distribution, is needed. Good city planning can achieve both goals of reducing total traffic pollutants and GHGs emissions and reducing the health impact of traffic pollution. Also, as diesel and petrol vehicles both have their own disadvantages in influencing public health and GHGs emissions, future policies can be aimed at phasing out both vehicle types and replacing them with alternative fuel vehicles or electric vehicles. In addition, trade-offs between safety and traffic emissions were identified and regulations for setting different speed limits according to local traffic conditions were suggested in Chapter 5. In the future, with the technological improvement of the Internet of things (IoT) and the development and refinement of the intelligent transportation system (ITS), dynamic speed limits that change according to real-time traffic, roads and weather conditions potentially offer a better solution to reduce traffic pollutions, GHGs emissions and traffic accidents.

Besides the impacts of traffic management strategies mentioned above, there are also impacts brought about by other traffic pollution and other effects of the same pollutants discussed in this thesis. For example, NO_2 not only affects public health, it also has significant impact on acidification and eutrophication of water bodies, and contributes to the production of photochemical air pollution. In addition, ground level ozone has notable impacts on ecosystems, plants and also contributes to photochemical air pollution aside from its health impact. Moreover, volatile organic compounds (VOCs), a pollutant group that was not analysed in this thesis, is another important traffic-related pollution. Combustion emissions include numerous complex VOCs, but little study had elaborated the health impact in terms of all traffic-related VOCs that have been identified. Barriers to this progress are mainly due to incomplete characterization of the composition and concentration of traffic-related VOCs. Certain components of air toxics, such as benzene and formaldehyde, have known carcinogenic potential (HEI Air Toxics Review Panel, 2007). Another group of VOCs, PANs (Peroxyacyl nitrates), are powerful respiratory and eye irritants present in photochemical smog. They also have negative impacts on plants (Fenger & Tjell, 2009).

This thesis has advanced knowledge of the impact of traffic management strategies surrounding traffic management, environmental pollution and public health, but there are still many topics to tackle in future research.

6.3 Modelling assumptions and uncertainties

There are many types of errors in models that causes uncertainty. The effect of some errors can be reduced or minimized whereas some are intrinsically embedded in models when simplifying assumptions were made. Common errors that are associated to the establishing, calibrating and predicting processes of models include specification errors, measurement errors, sampling errors, transfer errors, aggregation errors and computational errors (de Dios Ortuzar & Willumsen, 2011). Because of there are many types of errors, increasing model complexity would reduce some types of error, e.g. the error associated with simplification assumptions, while increase other types of error, e.g. measurement errors. A model with more complexity does not necessarily mean that it is more accurate. The general relationship between model complexity and associating errors is presented in Figure 6-3. Thus, choosing a suitable model with appropriate assumptions, instead of the most complicated model, is important. The reason for model selection is performed in Chapter 2 after literature review. Different types of errors in this thesis is discussed below.

1. Specification errors. This type of error is specified with each model, as simplification assumptions need to be taken when developing models, and sometimes the phenomenon being modelled is not well understood. In the thesis, simplification assumptions that may generate this type of error are summarized in Table 6-2. In the procedure of traffic emission calculation, the simplification assumption of only evaluating exhaust emissions was taken. Although traffic management strategies evaluated in the thesis only affected exhaust emissions for most of the pollutants, they would affect non-exhaust emissions for PM, such as emissions from tyre, break and surface wear. Thus, this assumption could cause underestimation of the impact of traffic strategies on PM. Future research could reduce the errors brought about by this assumption by incorporating non-exhaust emissions into evaluation. Besides errors brought about by assumptions, there are also errors associated with each model due to the modelling phenomenon is not well modelled. For COPERT model, there is a gap between real world and test cycle emissions, from which COPERT emission factors were derived. Emissions calculated from emission factors are usually underestimated. The error associated with emission factors is inevitable because of the measuring technique for emission factors nowadays. For OSPM, the pollution from only each road that was focused was considered. Pollution from other roads that could influence the concentration of the road that was focused was not

considered. The errors that were brought about by this factor cannot be neglected. In order to take emissions from other roads into account, Gaussian dispersion model was utilized as a supplementary model to OSPM to reduce this error.

In order to reduce assumptions, the strategies evaluated in the thesis were all compulsory or infrastructural strategies instead of other types of strategy, such as behavioural interventions. The data that could be used to infer the impact of behavioural interventions would involve more assumptions and indirect data, which would arise more uncertainties. Other strategies such as behavioural interventions are recommended to evaluate when enough data with high quality can be collected.

2. Measurement errors. This type of errors is associated with the measurement of data. Data sources, from which the measurement errors emerge, are also summarized in Table 6-2. Because there was a lot of data used in the thesis and data sources were from different authorities, the measurement errors are unpreventable.

3. Sampling errors. This type of errors arises because the models must be estimated using finite data sets. In the procedure of traffic model calibration, as available traffic counts were not cover all the roads in the modelling network, the sampling errors could not be eliminated.

4. Transfer errors. This type of errors is because modelling parameters developed in one context (time and/or place) transferred to a different one. In the thesis, CRF parameters were taken from the epidemiology studies in other cities to represent Dublin, which would cause transfer errors. As Dublin does not have its own research about the relationships between $PM_{2.5}$ and NO_2 concentrations and mortality incidences, this type of errors is unavoidable.

5. Aggregation errors. As a modelling chain involving four types of models was used in this thesis, a trivial error in the output of a model could be amplified in another model. Model aggregation error could not be neglect.

6. Computational errors. This type of errors always arises in the calibration procedure for OD matrices in the traffic model, where iterative procedures for which the exact solution were not found for reasons of computational costs, and an approximation solution were taken. Computational errors are typically small.

There were only 12 air quality monitor sites in Dublin, only 3 of them were located in relatively less traffic area, such as park or suburb area. Also, the pollutant types measured by these sites were limited. This caused the lack of background concentration monitors and limited the viability of model verification. Although the differences of the pollutant concentrations were focused to quantify the impact different strategies, and the background concentrations would not influence the conclusions of these impacts, the inadequacy of model verification limited our knowledge about the model performance and uncertainties. Although efforts were made to reduce all these types of errors in the thesis, there is still room for improvement for the model. Recommendations for model refinement regarding optimizing model selection and simplification assumptions are discussed in section 7.3.

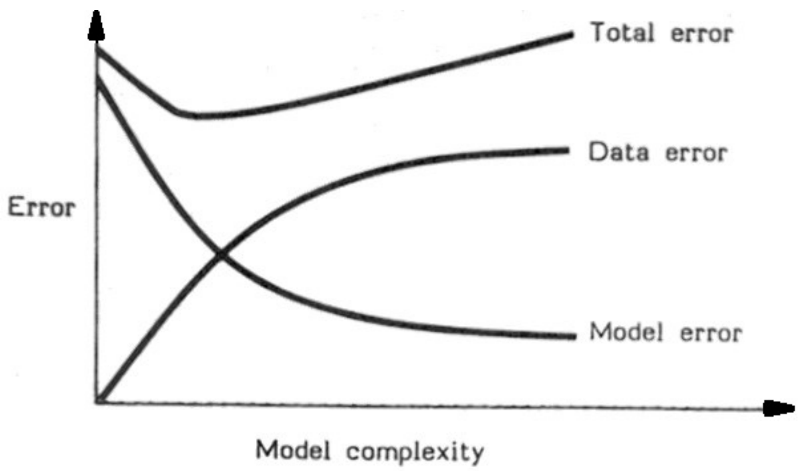


Figure 6-3: Variation of error with model complexity (Adapted from de Dios Ortuzar & Willumsen, 2011).

	Data sources	Assumptions	Reason for making the assumptions
Traffic model: VISUM	OD matrices: NTpM. Traffic counts: DCC. Speed records: DCC.	Logistic form of VDF was taken	Logistic form for VDF was the most suitable form after try-and-error and comparing with speed records.
Emission model: COPERT	Fleet composition: Irish EPA. Fuel contents: EU regulation.	1) Irish fleet composition was taken to represent Dublin fleet. 2) Only exhaust emissions were evaluated. 3) 1% composition of benzene in petrol was assumed.	1) Dublin fleet composition was not available and Irish fleet composition was the most suitable data source as Dublin population accounts for about 25% of Ireland. 2) Traffic strategies mainly affects exhaust emissions. 3) In line with the EU regulation.
Dispersion model: OSPM	Weather information: national meteorology service (Met Éireann). Road topography: Google Maps. Background concentration: Irish EPA.	Building height along roads was calculated by counting the storeys assuming each storey is 3m.	By approximation.
Gaussian Dispersion model	Gaussian dispersion model parameters: Briggs, 1973. NO_X to NO_2 ratio experience functions: Clapp & Jenkin, 2001.	NO_X to NO_2 ratio in London was assumed to be the same as in Dublin	The NO_X concentration range, original ratio of NO_2 and NO in NO_X, geography, solar radiation, and weather conditions that influence the ratio of NO_X to NO_2 were similar between two places.
Health impact model: BenMAP	CRF parameters: Krewski et al., 2009; Hoek et al., 2013. Population information: CSO. Baseline mortality incidences: CSO.	1) CRF parameters in literature were taken to represent Dublin situation. 2) Baseline mortality incidence rates in Dublin were assumed to be the same as in Ireland.	1) Dublin did not have its own research about CRF for $PM_{2.5}$ and NO_2. For NO_2, a review result for 12 European cities was chosen. It was a reflection of the average situation in Europe and thus representative for Dublin. For $PM_{2.5}$, a study of an American authority American Cancer Study (ACS) was chosen, the result of which was also used by a UK authority COMEAP (Ayres, 2010). 2) Dublin did not have its own data and Irish incidence data was the most suitable data source as Dublin population accounts for about 25% of Ireland.

Table 6-2: Summary of the data sources and assumptions.

Chapter 7 Conclusions and recommendations

7.1 Conclusions

In this thesis, the impact of the construction of the DPT; HGV traffic management regulation; speed limit changes; and fleet composition changes; on air pollutant and GHGs emissions, air quality and public health were estimated. A modelling chain including a traffic model, emission model, dispersion model and health impact model was employed to evaluate the impacts of these strategies.

The DPT and the HGV management strategy were estimated to increase the travel demands, to redistribute traffic across the network causing significant vehicle detours, to improve speed distributions, and to increase air pollutant emissions of PM, NO_X, and CO, and GHGs emissions of CO_2 and CH_4. For air pollutants, the opening of the DPT was predicted to increase PM, NO_X and CO emissions by 5%, 8% and 1%, respectively, and DPT together with HGV management strategy were predicted to increase these pollutants by 13%, 21% and 3%, respectively. For GHGs emissions, the DPT was predicted to increase CO_2 and CH_4 emissions by 3% and 5%, respectively, and two strategies together were predicted to increase CO_2 and CH_4 emissions by 7% and 8%, respectively. The increase in emissions were mainly due to the longer average travel distances for the vehicles.

The traffic redistribution resulted from the opening of the DPT and the implementation of HGV management strategy were predicted to result in decreases in $PM_{2.5}$ and NO_2 concentrations in the city centre, where population density was relative high, and increases in these pollutant concentrations in the place where population density was relative low. Therefore, although the DPT and HGV management strategies increased the total emissions of PM and NO_X by a relatively significant amount (by 13% and 21%), the predicted overall health impact of the concentration changes of $PM_{2.5}$ and NO_2 brought about by these strategies were found to be negligible. However, overall GHGs emissions were increased and thus these two strategies were predicted to have negative impacts on global warming and climate change. The assessment of these two strategies demonstrates that the impact of traffic management strategies can be very different in terms of air pollutant emissions, health impacts by traffic pollutants and GHGs emissions. The consideration of all these aspects is essential for a holistic assessment.

The fleet composition shifting from diesel vehicles to comparable petrol vehicles was found to worsen the pollution of CO and Benzene, but would reduce the pollution of $PM_{2.5}$, PM_{10} and NO_2. The thesis mainly focused on the health impact of $PM_{2.5}$ and NO_2, which are the main transport-related air pollutants and affect the public health most severely. In terms of these pollutants, the fleet composition shifting from diesel vehicles to petrol vehicles was predicted to have positive health impacts. However, the negative health impact of CO and Benzene, although it could be relatively minor compared to the impact of $PM_{2.5}$ and NO_2, would still offset the benefits brought about by the reduction of $PM_{2.5}$ and NO_2 to some extent and thus need to be further investigated. Nonetheless, the fleet composition shifting from diesel vehicles to petrol vehicles would increase the emission of GHGs emissions and place threat to the global warming and the environment.

Speed limit changes from 30km/h to 50km/h were found to increase the concentrations of NO_2, PM_{10}, $PM_{2.5}$, CO and Benzene. The health impact assessment found NO_2 and $PM_{2.5}$ concentration changes that were brought about by speed limit change from 50km/h to 30km/h were predicted to increase the all-cause mortality incidence in Dublin in 2013 by 22 death cases in the over-18 age group and 6 death cases in the over-30 age group, respectively. It shows that the health impact of traffic-related pollutions should not be neglected. Speed limit changes from 30km/h to 50km/h were predicted to increase the average speed, which may reduce the GHGs emissions and benefit to the environment. The control of speed limit was found the most effective and an easy to control approach to reduce traffic emission compared to the other three approaches.

Trade-offs between human health impacts of traffic-related air pollution and road safety were found to exist in this context. This indicates that policy makers need to consider health externalities resulting from traffic-related pollutions when introducing traffic management strategies, especially for vulnerable people who are more likely to be adversely affected by air pollutants, and regarding busy cities and busy roads, where the air quality changes and health impacts brought about by the traffic management strategies would be greater. Also trade-offs of introducing these strategies between health impacts of different pollutants, and between health impacts and impacts on global warming and ecosystems, were identified. The thesis emphasized the importance of a holistic assessment taking utmost consideration of these trade-offs. It also suggested that future directions of traffic management strategies should be aimed to reduce the negative impacts on both public health and the environment.

The evaluation of the impact of traffic management strategies on traffic emission, air quality and public health is a complex process. Employing models to estimate their impacts on real world scenarios involves many simplifying assumptions. These are due to data availability and computational resources, and thus models have intrinsic errors within them. Although the best efforts were made to insure the robustness of the data and the results, further refinement and investigation could be conducted. Recommendations for future research are listed in section 7.3.

7.2 Reflections on research objectives

The research objectives raised in Chapter 1 were achieved in this thesis. A holistic assessment of the impacts of four types of traffic management strategies, i.e. the infrastructural change, the traffic management regulation change, speed limit changes and fleet composition changes, were conducted by establishing a modelling chain consisted of the traffic model with VISUM, the emission model with COPERT, the dispersion model with OSPM and Gaussian dispersion model, and the health impact model with BenMAP. These models were suitable to the time and space scale for scenarios considered in this thesis. Also, a cross-comparison among these strategies was performed regarding the air quality impacts and the health impacts. All-cause mortality changes in Dublin brought about by these strategies were employed as a common metric for the evaluation of these strategies to enable different impacts brought about by different types of strategies to compare.

During the impact assessments of different traffic management strategies, a baseline traffic model was developed to represent traffic conditions in Dublin, and the baseline model was calibrated and validated with the recorded traffic counts. Scenarios that represented different traffic management strategies were created and incorporated into the modelling chain to evaluate the conditions of traffic, emission, air quality and mortality incidence under different strategies. These scenarios were compared and the differences of them were calculated to assess the impact of these strategies on traffic, emissions, pollutant concentrations and health. The impacts of traffic strategies were analysed and cross-compared regarding their costs and impacts on air quality, public health and GHG emissions.

The method utilized in the thesis is easy to extend to evaluate other strategies and to employ in other cities, which enables a more comprehensive analysis and comparisons among different types of traffic management strategies, and can provide practical suggestions for traffic policy makers for different places.

7.3 Recommendations for future research

1. More inclusive assessments of traffic management strategies: There are other traffic management strategies or even more strategic transport initiatives that influence traffic activities, emissions, population density and thus influence public health and the environment (e.g. LEZs, parking restrictions, tolls roads, emission taxes, development of intelligent transportation systems and even land use planning). The assessment conducted here can be extended to other traffic strategies and obtain more comprehensive information about the advantage and disadvantages of a wider range of strategies. Besides these compulsory, technical or infrastructural strategies, behavioural interventions can also have impacts on traffic emissions and air quality. Behavioural interventions such as campaigns about environmental education or eco-driving are worthy of investigation in future research on the condition that relevant data is available and qualified. Furthermore, air quality changes brought about by traffic condition changes even not because of traffic management strategy also deserve to be studied. For example, the impact of the reducing traffic of not driving children to school during summer holiday time can be investigated to understand the potential impact of driving children to school and to provide information for pertinent strategies such as introducing school buses.

2. More comprehensive assessments of pollutants and GHGs emissions: As mentioned, the health impact of CO and Benzene, although it could be relatively minor compared to the impact of $PM_{2.5}$ and NO_2, is still important to investigate. Regarding GHGs emissions, although it is difficult to estimate their local impact, their impacts on a global scale need to be taken into consideration. Also the impacts of other pollutants (e.g. VOCs) are worthwhile to analyse in order to acquire more complete knowledge for the impact of a traffic management strategy.

3. Assessments with expanded criteria: Aside from the mortality impacts of pollutants that are covered by this thesis, traffic-related pollutants are also associated to other health

issues, e.g. impacts of $PM_{2.5}$ and O_3 on cardiovascular diseases and on respiratory systems, impacts of NO_2 on respiratory systems. Pollutants' impacts on morbidities and other health issues need to be assessed. In addition, the impacts of pollutants on ecosystems and materials damages, and the impacts of GHGs on global warming and climate change also remained to be examined.

4. Model refinements: For the traffic model, the calibration and validation process could be further optimized if there were more available data. For the assessment of the impact of speed limit, the emission estimation in OSPM was based on the average speed, which could underestimate the impact of acceleration and deceleration. A traffic micro-simulation and emission model based on second-by-second speed and acceleration profiles could be developed for future investigation. However such micro-simulations are often limited in the geographical and time scope they can cover. Compared to the current approach a micro-simulation for the entirety of Dublin across a year would not be feasible. Micro-simulations are recommended for the investigation of a smaller scale in time and area. Future research also can employ a more precise dispersion model that allows the calculations of chemical reactions in the air rather than utilizing a Gaussian plume dispersion model and experience functions to estimate the chemically active pollutants. Moreover, if there are epidemiological studies in the future that focus on the pollutant concentration-response relationships in Dublin, the parameters of health impact functions can be adopted from these studies to be more representative of the conditions in Dublin. Furthermore, many assumptions were taken when developing the model that affects increased the uncertainty of the model. Efforts can be made to reduce the number of assumptions or the uncertainty brought about by these assumptions. For instance, fleet composition in the thesis was assumed to be the same as national fleet composition of Ireland. If Dublin fleet composition is available in the future, this assumption can be avoided, and the uncertainty of the model can be diminished. Also, traffic strategies were assumed to only influence exhaust emissions, thus only exhaust emissions were evaluated. While this assumption was valid for most of the pollutants, it would influence the estimation of PM emission. Non-exhaust emissions such as brake wear emissions that would affect the PM pollution can be evaluated in the future research.

5. Assessment beyond Dublin: The assessment of traffic management strategies is important, especially to cities with a heavy traffic burden and more severe air quality problems. The assessment method in the thesis can be expand to other cities and the

assessment of traffic management strategies in terms of health and environmental impact can attract attentions of policy makers on environmental and health externalities.

Appendix A for Chapter 3

Calibration criteria shown in Table A-1:

Criteria and Measure	**Guideline**
Modelled Hourly Flows vs. Observed traffic counts:	
Individual flows within 15% for flows 700 – 2700 vph Individual flows within 100 vph for flows <700 vph Individual flows within 400 vph for flows > 2700	> 85% of cases

Table A-1: Calibration criteria (Adapted from UK DMRB, 1997).

Ireland fleet distribution shown in Table A-2:

Sector	Subsector	Technology	Vehicle # in 2006	Vehicle # in 2007	Vehicle # in 2013
Passenger Cars	Gasoline 0,8 - 1,4 l	PRE ECE	0	0	0
Passenger Cars	Gasoline 0,8 - 1,4 l	ECE 15/00-01	0	0	0
Passenger Cars	Gasoline 0,8 - 1,4 l	ECE 15/02	0	0	0
Passenger Cars	Gasoline 0,8 - 1,4 l	ECE 15/03	0	0	0
Passenger Cars	Gasoline 0,8 - 1,4 l	ECE 15/04	15904	11489	949
Passenger Cars	Gasoline 0,8 - 1,4 l	Open Loop	0	0	0
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 1 - 91/441/EEC	107123	77929	7739
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 2 - 94/12/EEC	336491	307589	201061
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 3 - 98/69/EC Stage2000	266772	258354	276634
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 4 - 98/69/EC Stage2005	73236	143395	251944
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 5 - EC 715/2007	0	0	39297
Passenger Cars	Gasoline 0,8 - 1,4 l	PC Euro 6 - EC 715/2007	0	0	0

Passenger Cars	Gasoline 1,4 - 2,0 l	PRE ECE	0	0	0
Passenger Cars	Gasoline 1,4 - 2,0 l	ECE 15/00-01	0	0	0
Passenger Cars	Gasoline 1,4 - 2,0 l	ECE 15/02	0	0	0
Passenger Cars	Gasoline 1,4 - 2,0 l	ECE 15/03	0	0	0
Passenger Cars	Gasoline 1,4 - 2,0 l	ECE 15/04	12265	9511	504
Passenger Cars	Gasoline 1,4 - 2,0 l	Open Loop	0	0	0
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 1 - 91/441/EEC	82616	64507	4434
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 2 - 94/12/EEC	259509	254611	106786
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 3 - 98/69/EC Stage2000	205741	213856	146907
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 4 - 98/69/EC Stage2005	56481	118698	136307
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 5 - EC 715/2007	0	0	21812
Passenger Cars	Gasoline 1,4 - 2,0 l	PC Euro 6 - EC 715/2007	0	0	0
Passenger Cars	Gasoline >2,0 l	PRE ECE	0	0	0
Passenger Cars	Gasoline >2,0 l	ECE 15/00-01	0	0	0
Passenger Cars	Gasoline >2,0 l	ECE 15/02	0	0	0
Passenger Cars	Gasoline >2,0 l	ECE 15/03	0	0	0
Passenger Cars	Gasoline >2,0 l	ECE 15/04	1621	1353	33
Passenger Cars	Gasoline >2,0 l	PC Euro 1 - 91/441/EEC	10921	9174	425
Passenger Cars	Gasoline >2,0 l	PC Euro 2 - 94/12/EEC	34305	36208	7055
Passenger Cars	Gasoline >2,0 l	PC Euro 3 - 98/69/EC Stage2000	27197	30413	9699
Passenger Cars	Gasoline >2,0 l	PC Euro 4 - 98/69/EC Stage2005	7466	16880	9733
Passenger Cars	Gasoline >2,0 l	PC Euro 5 - EC 715/2007	0	0	1723
Passenger Cars	Gasoline >2,0 l	PC Euro 6 - EC 715/2007	0	0	0
Passenger Cars	Diesel 1,4 - 2,0 l	Conventional	5187	4309	182
Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 1 - 91/441/EEC	34938	29228	1452

Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 2 - 94/12/EEC	109746	115363	38544
Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 3 - 98/69/EC Stage2000	87007	96897	132268
Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 4 - 98/69/EC Stage2005	23886	53781	288016
Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 5 - EC 715/2007	0	0	160045
Passenger Cars	Diesel 1,4 - 2,0 l	PC Euro 6 - EC 715/2007	0	0	0
Passenger Cars	Diesel >2,0 l	Conventional	844	760	26
Passenger Cars	Diesel >2,0 l	PC Euro 1 - 91/441/EEC	5688	5158	209
Passenger Cars	Diesel >2,0 l	PC Euro 2 - 94/12/EEC	17866	20358	5545
Passenger Cars	Diesel >2,0 l	PC Euro 3 - 98/69/EC Stage2000	14164	17100	19029
Passenger Cars	Diesel >2,0 l	PC Euro 4 - 98/69/EC Stage2005	3888	9491	41435
Passenger Cars	Diesel >2,0 l	PC Euro 5 - EC 715/2007	0	0	23025
Passenger Cars	Diesel >2,0 l	PC Euro 6 - EC 715/2007	0	0	0
Passenger Cars	LPG	Conventional	0	0	0
Passenger Cars	LPG	PC Euro 1 - 91/441/EEC	196	183	72
Passenger Cars	LPG	PC Euro 2 - 94/12/EEC	151	137	57
Passenger Cars	LPG	PC Euro 3 - 98/69/EC Stage2000	83	81	57
Passenger Cars	LPG	PC Euro 4 - 98/69/EC Stage2005	83	81	57
Passenger Cars	LPG	PC Euro 5 - EC 715/2007	0	0	0
Passenger Cars	LPG	PC Euro 6 - EC 715/2007	0	0	0
Passenger Cars	Hybrid Gasoline <1,4 l	PC Euro 4 - 98/69/EC Stage2005	0	0	0

Passenger Cars	Hybrid Gasoline 1,4 - 2,0 l	PC Euro 4 - 98/69/EC Stage2005	0	0	0
Passenger Cars	Hybrid Gasoline >2,0 l	PC Euro 4 - 98/69/EC Stage2005	0	0	0
Light Commercial Vehicles	Gasoline <3,5t	Conventional	89	49	4
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 1 - 93/59/EEC	217	134	28
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 2 - 96/69/EEC	550	381	101
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 3 - 98/69/EC Stage2000	661	496	236
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 4 - 98/69/EC Stage2005	223	337	292
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 5 - 2008 Standards	0	0	78
Light Commercial Vehicles	Gasoline <3,5t	LD Euro 6	0	0	0
Light Commercial Vehicles	Diesel <3,5 t	Conventional	14153	10621	1441
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 1 - 93/59/EEC	34690	29131	10951
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 2 - 96/69/EEC	87696	82841	39482
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 3 - 98/69/EC Stage2000	105457	107724	92220
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 4 - 98/69/EC Stage2005	35522	73131	113834
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 5 - 2008 Standards	0	0	30260
Light Commercial Vehicles	Diesel <3,5 t	LD Euro 6	0	0	0
Heavy Duty Trucks	Gasoline >3,5 t	Conventional	53	44	24
Heavy Duty Trucks	Rigid <=7,5 t	Conventional	785	589	108
Heavy Duty Trucks	Rigid <=7,5 t	HD Euro I - 91/542/EEC Stage I	1205	986	287

Heavy Duty Trucks	Rigid <=7,5 t	HD Euro II - 91/542/EEC Stage II	3593	3282	1228
Heavy Duty Trucks	Rigid <=7,5 t	HD Euro III - 2000 Standards	4321	4267	2868
Heavy Duty Trucks	Rigid <=7,5 t	HD Euro IV - 2005 Standards	1467	2897	3541
Heavy Duty Trucks	Rigid <=7,5 t	HD Euro V - 2008 Standards	0	0	932
Heavy Duty Trucks	Rigid <=7,5 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 7,5 - 12 t	Conventional	1159	830	129
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro I - 91/542/EEC Stage I	1781	1389	345
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro II - 91/542/EEC Stage II	5308	4626	1477
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro III - 2000 Standards	6384	6015	3450
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro IV - 2005 Standards	2167	4084	4258
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro V - 2008 Standards	0	0	1121
Heavy Duty Trucks	Rigid 7,5 - 12 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 12 - 14 t	Conventional	542	401	66
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro I - 91/542/EEC Stage I	832	671	176
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro II - 91/542/EEC Stage II	2481	2234	754
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro III - 2000 Standards	2983	2905	1761
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro IV - 2005 Standards	1013	1972	2173
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro V - 2008 Standards	0	0	572
Heavy Duty Trucks	Rigid 12 - 14 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 14 - 20 t	Conventional	226	187	43
Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro I - 91/542/EEC Stage I	347	314	114

Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro II - 91/542/EEC Stage II	1036	1044	487
Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro III - 2000 Standards	1245	1358	1137
Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro IV - 2005 Standards	423	922	1403
Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro V - 2008 Standards	0	0	370
Heavy Duty Trucks	Rigid 14 - 20 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 20 - 26 t	Conventional	1	0	0
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro I - 91/542/EEC Stage I	1	1	0
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Rigid 20 - 26 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 26 - 28 t	Conventional	1	0	0
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro I - 91/542/EEC Stage I	1	1	0
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Rigid 26 - 28 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid 28 - 32 t	Conventional	1	0	0
Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro I - 91/542/EEC Stage I	1	1	0

Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Rigid 28 - 32 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Rigid >32 t	Conventional	1	0	0
Heavy Duty Trucks	Rigid >32 t	HD Euro I - 91/542/EEC Stage I	1	1	0
Heavy Duty Trucks	Rigid >32 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Rigid >32 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Rigid >32 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Rigid >32 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Rigid >32 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Articulated 40 - 50 t	Conventional	1	0	0
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro I - 91/542/EEC Stage I	1	1	0
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Articulated 40 - 50 t	HD Euro VI	0	0	0
Heavy Duty Trucks	Articulated 50 - 60 t	Conventional	1	0	0
Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro I - 91/542/EEC Stage I	1	1	0

Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro II - 91/542/EEC Stage II	3	2	1
Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro III - 2000 Standards	3	3	3
Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro IV - 2005 Standards	1	2	4
Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro V - 2008 Standards	0	0	1
Heavy Duty Trucks	Articulated 50 - 60 t	HD Euro VI	0	0	0

Table A-2: Ireland fleet distribution in 2006, 2007 and 2013 (EPA, 2015).

Appendix B for Chapter 4

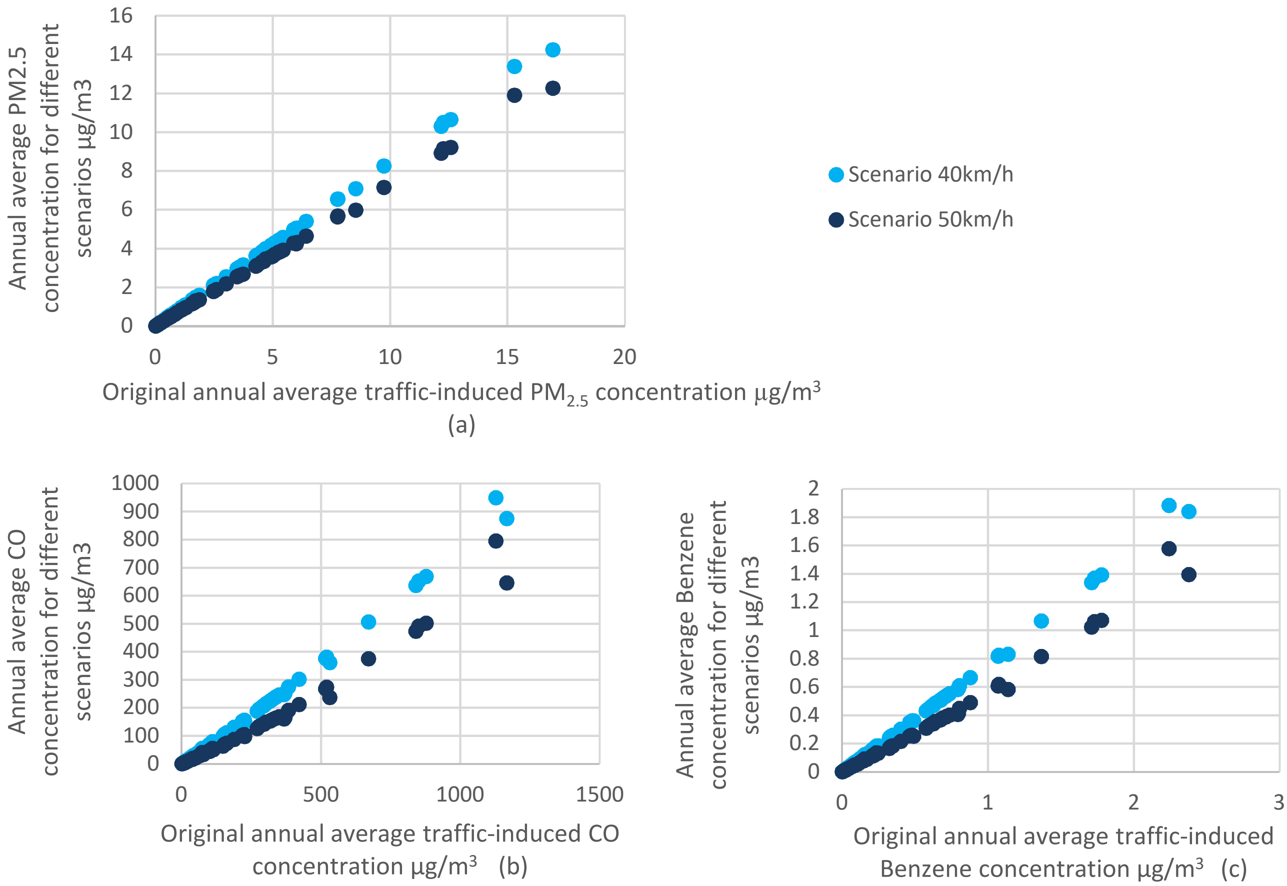


Figure B-1: Predicted change in $PM_{2.5}$, CO and Benzene traffic induced concentrations on each road segment in the model domain for varying speed limits.

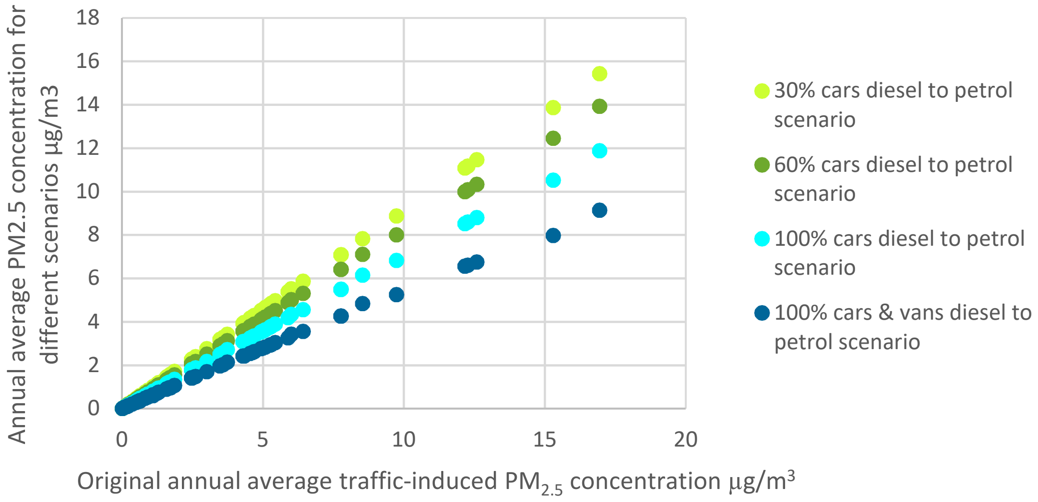


Figure B-2: Predicted change in $PM_{2.5}$ traffic induced concentrations on each road segment in the model domain for varying fleet compositions.

Appendix C for Chapter 5

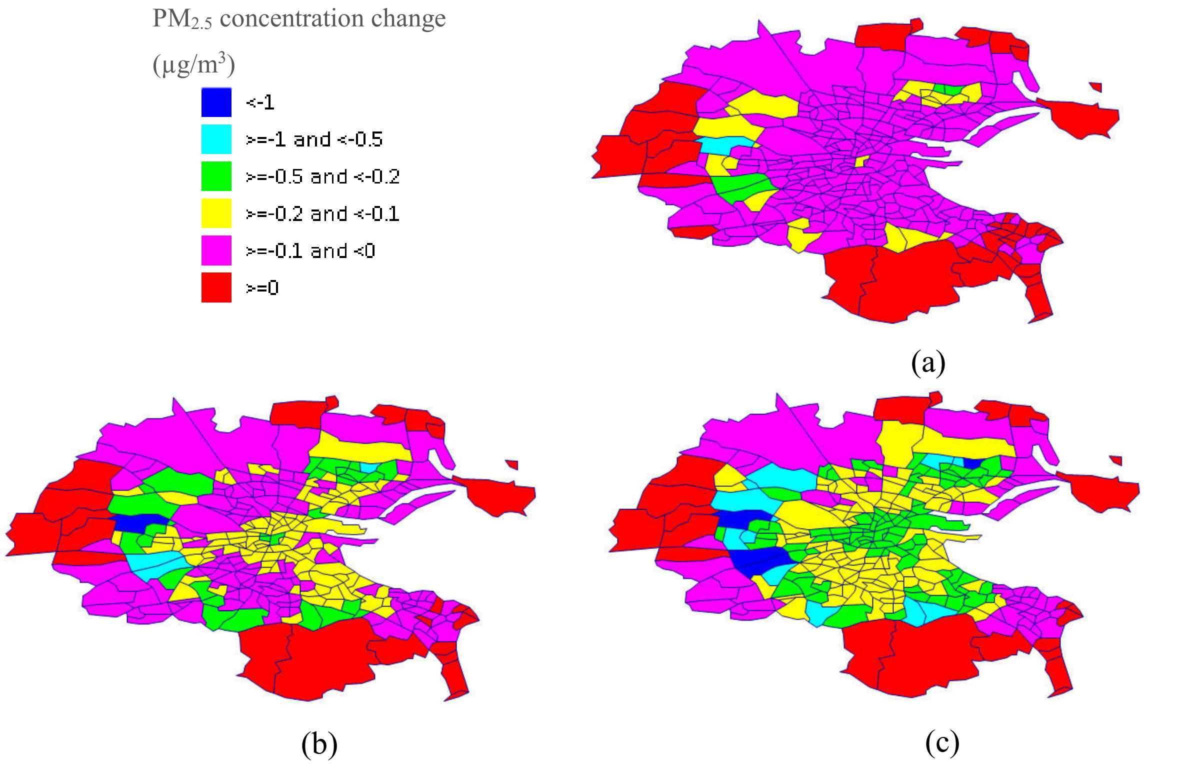


Figure C-1: Predicted increase in annual average $PM_{2.5}$ concentration (µg/m^3), averaged across each ED, and brought about by the scenarios of (a) 50%, (b) 100% diesel cars converting to petrol cars and (c) 100% diesel cars and vans converting to petrol vehicles, compared to baseline scenario.

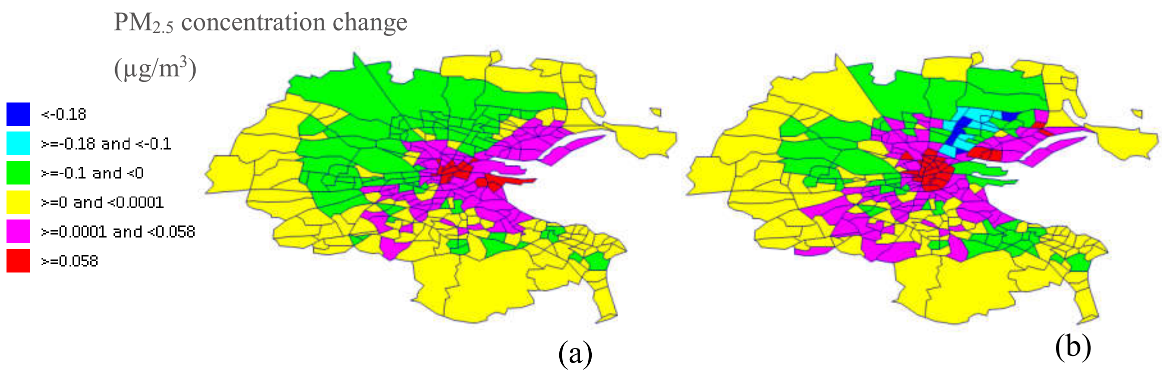
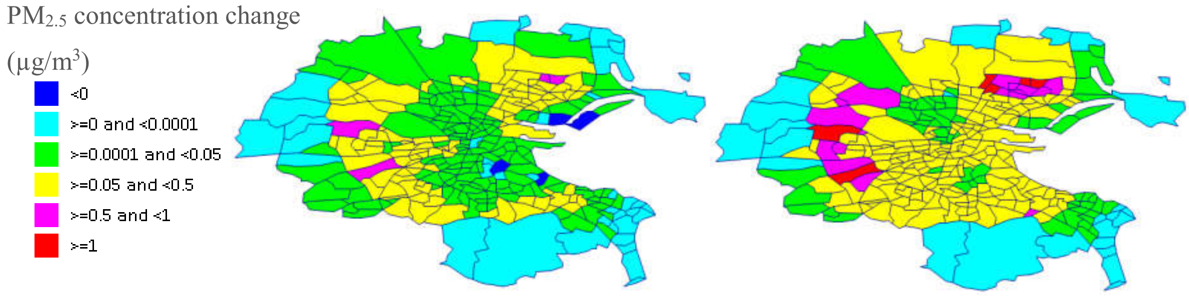
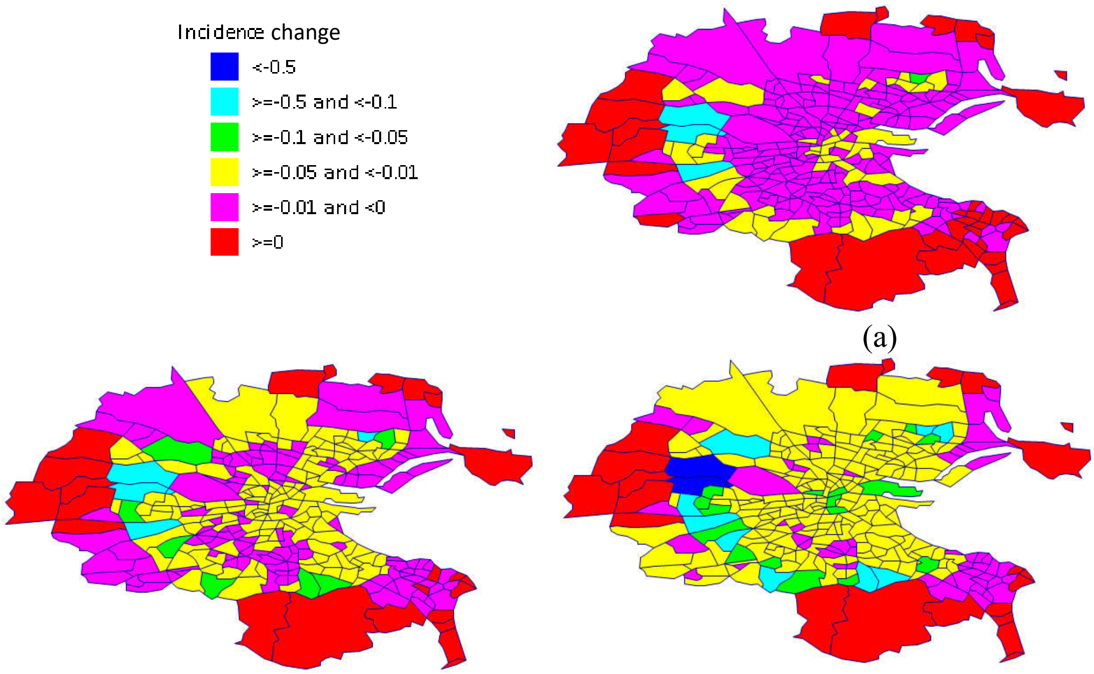


Figure C-2: Predicted increase in annual average $PM_{2.5}$ concentration (µg/m^3), averaged across each ED, and brought about by the scenarios of (a) HGV management not implemented and (b) HGV management not implemented plus DPT not opened, compared to baseline scenario.



(a) (b)

Figure C-3: Predicted increase in annual average $PM_{2.5}$ concentration ($\mu g/m^3$), averaged across each ED, and brought about by the scenarios of the default speed limit of (a) 40 km/h and (b) 30 km/h, compared to baseline scenario.



(b) (c)

Figure C-4: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased $PM_{2.5}$ brought about by the scenarios of (a) 50% and (b) 100% of diesel cars converting to petrol cars, and (c) 100% diesel cars and vans converting to petrol vehicles, compared to baseline scenario.

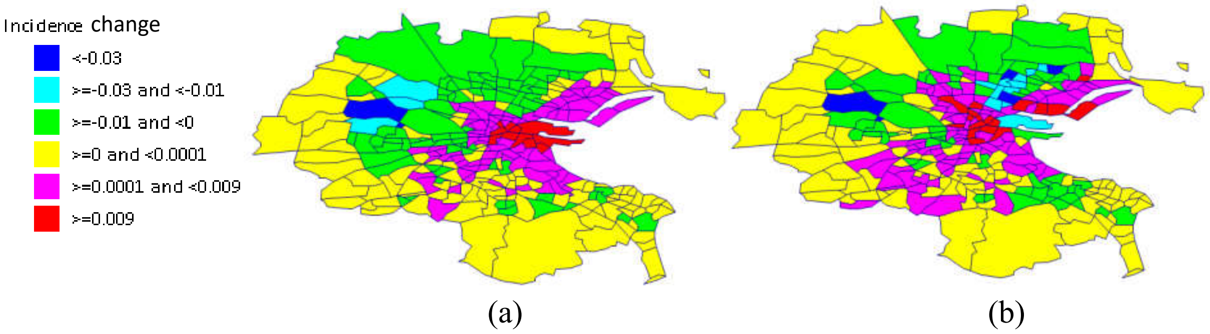


Figure C-5: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased $PM_{2.5}$ brought about by the scenarios of (a) HGV management not implemented, and (b) HGV management not implemented plus DPT not opened, compared to baseline scenario.

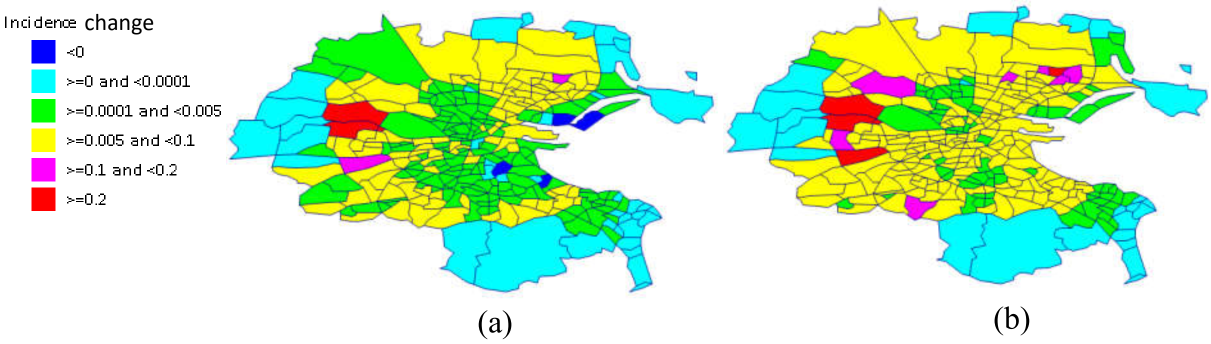


Figure C-6: Absolute number of predicted increased all-cause mortality incidence (death cases) with increased $PM_{2.5}$ brought about by the scenarios of the default speed limit of (a) 40 km/h, and (b) 30 km/h, compared to baseline scenario.

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