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Report

A REVIEW AND ANALYSIS OF PRIMARY NITROGEN DIOXIDE EMISSIONS FROM ROAD VEHICLES IN SYDNEY

NSW ROADS AND MARITIME SERVICES

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EXECUTIVE SUMMARY

The presence of nitric oxide (NO) and nitrogen dioxide (NO_2) - collectively known as oxides of nitrogen (NOx)[a] - in the atmosphere is a cause for concern due to their impacts on human health and the environment, with NO_2 being the more important pollutant from a health perspective. Road transport is a major source of NO_X. Most of the NO_X in vehicle exhaust is NO, with NO_2 mainly being formed in the atmosphere. NO_2 is therefore generally considered to be a 'secondary' pollutant. Any NO_2 that is emitted directly in vehicle exhaust is usually referred to as 'primary NO_2', and the fraction of NO_X that is emitted as NO_2 is termed *f*-NO_2.

A number of major infrastructure projects are currently planned for Sydney, and these will require accurate and robust ambient air quality assessments, including a more informed characterisation of primary NO_2 compared with previous assessments. Road tunnels feature in some of these projects, and in-tunnel concentrations of NO_2 are an important consideration from a ventilation design perspective. Again, the use of correct assumptions for primary NO_2 is vital to this process. Historically, a fairly low value for *f*-NO_2 (5-10%) has been used in air quality and in-tunnel assessments. However, recent analyses of ambient NO_2 concentrations in urban areas - notably in the UK - have shown an increasing NO_2 proportion, and this has important implications for assessments. However, primary NO_2 emissions from vehicles on the road in Sydney are not well documented, either in terms of historical trends or future projections.

This Report summarises the issues associated with primary NO_2, and presents evidence based on the analysis of models and air quality monitoring data from Sydney. Several different data sets and analytical techniques are presented in the Report, including emission modelling, the analysis of ambient air quality measurements and the analysis of emissions from the ventilation stacks of operational road tunnels.

Although a range of different data sets and methods have been used in the study to determine *f*-NO_2, the level of agreement is good. The evidence suggests that there has been a gradual increase in *f*-NO_2 in recent years, from less than 10% before 2008 to around 15% in 2014. The increase has not been as pronounced as that observed in the UK, and it appears that although the value of *f*-NO_2 may increase further in the future, it will probably not increase much further and will eventually start to decrease.

The main reason for the increase in *f*-NO_2 recent years is likely to be the increased market penetration of diesel cars into the Sydney vehicle fleet. There is insufficient information on the types and distributions of exhaust after-treatment devices in use on vehicles in Sydney, and so it is not possible to determine the extent to which this is a contributing factor. Further information of this type would be beneficial to those studying the impacts of road transport on air quality in Sydney.

The findings of this work, whilst they should be viewed as indicative at this stage, could be used to support and refine calculations of primary NO_2 emissions from road traffic in Sydney, including calculations to support air quality impact assessments and studies of in-tunnel air quality. In particular, it appears from the measurements in Sydney that the approach of incorporating the European values for *f*-NO_2 in the NSW EPA inventory model has produced satisfactory results.

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[a] NO_X is, by convention, the sum of NO_2 and NO.

CONTENTS

TERMS AND ABBREVIATIONS

Term	Description
ABS	Australian Bureau of Statistics
ADR	Australian Design Rule
BTS	(NSW) Bureau of Transport Statistics
DPF	diesel particulate filter
Emission factor	A quantity which expresses the mass of a pollutant emitted per unit of activity. For road transport the unit of activity is usually either distance (i.e. g/km) or fuel consumed (i.e. g/litre).
Emission rate	A quantity which expresses the mass of a pollutant emitted per unit of time (e.g. g/second)
EU	European Union
GMR	(NSW) Greater Metropolitan Region
HDV	heavy-duty vehicle, which includes heavy goods vehicles, buses and coaches
HGV	heavy good vehicle
LCV	light commercial vehicle
LDV	light-duty vehicle, which includes cars and light commercial vehicles
NO	nitric oxide
NO_2	nitrogen dioxide
NO_X	oxides of nitrogen
NPI	National Pollutant Inventory
NSW EPA	New South Wales Environment Protection Authority
O_3	ozone
OEH	(NSW) Office of Environment and Heritage
ppb	parts per billion
ppm	parts per million
RMS	(NSW) Roads and Maritime Services
SCR	Selective catalytic reduction
VKT	vehicle-kilometres travelled
$\mu g/m^3$	micrograms per cubic metre

1 INTRODUCTION

1.1 Background

The presence of nitric oxide (NO) and nitrogen dioxide (NO_2) - collectively known as oxides of nitrogen (NOx)[2] - in the atmosphere is a cause for concern due to their impacts on human health and the environment, with NO_2 being the more important pollutant from a health perspective. Road transport is a major source of NO_X. Most of the NO_X in vehicle exhaust is NO, with NO_2 mainly being formed in the atmosphere. NO_2 is therefore generally considered to be a 'secondary' pollutant. Any NO_2 that is emitted directly in vehicle exhaust is usually referred to as 'primary NO_2', and the fraction of NO_X that is emitted as NO_2 is termed *f*-NO_2 (often stated as a percentage).

A number of major infrastructure projects are currently planned for Sydney, and these will require accurate and robust ambient air quality assessments, including a more informed characterisation of primary NO_2 compared with previous assessments. Road tunnels feature in some of these projects, and in-tunnel concentrations of NO_2 are an important consideration from a ventilation design perspective. Again, the use of correct assumptions for primary NO_2 is vital to this processes.

Historically, a fairly low value for *f*-NO_2 (5-10%) has been used in air quality and in-tunnel assessments. However, recent analyses of ambient NO_2 concentrations in urban areas - notably in the UK - have shown an increasing NO_2 proportion, and this has important implications for assessments. However, primary NO_2 emissions from vehicles in Sydney are not well documented, either in terms of historical trends or future projections. Few studies have focussed specifically on this topic, although data from models, ambient air quality measurements and tunnels been available for some time to allow primary NO_2 to be estimated.

NSW Roads and Maritime Services (RMS) has therefore commissioned Pacific Environment to review the available literature and data with a view to improving the understanding of primary NO_2 emissions in Sydney, how these have changed in the past, and how they are likely to change in the future. This Report describes the issue of primary NO_2 and presents evidence which should enable its more accurate representation in future studies.

1.2 Objectives and scope of work

The study is composed of the following elements:

- **Literature review (Section 2)**. This part of the study involves a brief review of the scientific and technical literature on primary NO_2 emissions. The issue of primary NO_2 is explained, with a summary of recent studies dealing with the quantification of primary NO_2.

- **Characterisation and emission behaviour of vehicles in Sydney (Section 3)**. The characteristics and emission behaviour of the vehicle fleet in Sydney have been summarised. This summary provides some context in terms of the importance of road vehicles as a source of NO_X.

- **Presentation of the evidence relating to primary NO_2 emissions from road vehicles in Sydney (Section 4)**. This includes the following:

[2] NO_X is, by convention, the sum of NO_2 and NO.

- Emissions modelling. Emissions of NO_X and NO_2 (and hence *f*-NO_2) for different vehicle groups and road types in Sydney have been estimated using the model developed for the NSW GMR inventory[3].

- An analysis of ambient air quality data. The historical trends in NO_X and NO_2 concentrations at roadside ambient air quality monitoring stations in Sydney have been investigated using different approaches, including some initial investigations using the *Openair* software **(Carslaw, 2015a)**. As far as we are aware, *Openair* has not previously been used to estimate primary NO_2 from monitoring data in Australia.

- An analysis of the air pollution monitoring data from operational road tunnels. The analysis has focused on the Lane Cove Tunnel and the M5 East Tunnel.

[3] An emissions inventory defines the amount (in tonnes per year) of each pollutant that is emitted from each source in a given area.

2 LITERATURE REVIEW

2.1 NO and NO_2 formation during combustion

The road transport sector, which relies significantly upon internal combustion engines, is a major contributor to global NOx emissions **(Boulter *et al.*, 2013)**. In an internal combustion engine energy is produced from the burning of fuel in air, with the main oxidation products being carbon dioxide (CO_2) and water vapour. However, some of the nitrogen in the combustion air is also oxidised, leading to the formation of NO. NO formation is enhanced by the high temperatures and pressures found in the combustion chamber and oxygen-rich fuelling conditions.

The main NO formation mechanism is known as the 'thermal' (or Zel'dovich) cycle **(Heywood, 2008)**, and this is responsible for more than 90% of emissions **(Vestreng *et al.*, 2009)**. The mechanism is described by the following reactions:

$$N_2 + O \rightarrow NO + N \qquad \textbf{Reaction 1}$$

$$N + O_2 \rightarrow NO + O \qquad \textbf{Reaction 2}$$

$$N + OH \rightarrow NO + H \qquad \textbf{Reaction 3}$$

The first reaction is the rate-determining step, influencing the amount of NO which is formed, and is highly dependent on combustion temperature; increasing the temperature from 1,200°C to 2,000°C increases the rate of this reaction by a factor of 10,000 **(Baulch *et al.*, 2004)**.

Whilst NO is the dominant NO_X component formed during engine combustion, significant amounts of NO_2 can also be produced under certain conditions.

2.2 Atmospheric transformation

An understanding of the transformation of NO and NO_2 in the atmosphere is vital when it comes to interpreting ambient air quality measurements. However, the ambient concentration of NO_2 is dictated by various complex processes and factors.

Under the majority of atmospheric conditions, the main mechanism for NO_2 formation in the atmosphere is through the rapid reaction of NO with ozone:

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad \textbf{Reaction 4}$$

If this were the only important reaction (as it is at night-time under most urban conditions) then NO will be transformed into NO_2 until either all the NO has been converted to NO_2 or until all the ozone has been used up (known as 'ozone titration'). At polluted locations comparatively close to sources of NO_x (such as roads) NO is in large excess and it is the availability of O_3 which limits the quantity of NO_2 that can be produced **(AQEG, 2004)**. The rate of this reaction is important for modelling, since the time it takes for exhaust emissions of NO and NO_2 to reach roadside location is only a few seconds, whilst the time it takes for emissions to reach an urban background location can be tens of minutes. As a result, the NO_2/NO_X ratio near to traffic sources is generally lower than at urban background locations **(Denby, 2011)**.

Other chemical processes can convert NO to NO_2, and these are discussed in detail in the literature **(**e.g. **AQEG, 2004)**.

Under normal ambient daytime conditions the reverse process also occurs – the destruction of NO_2 by photolysis to form NO and ozone, as shown in **Reaction 5** and **Reaction 6**:

$$\mathrm{NO_2 + sunlight \rightarrow NO + O} \qquad \textbf{Reaction 5}$$

$$\mathrm{O + O_2\,(+M) \rightarrow O_3\,(+M)} \qquad \textbf{Reaction 6}$$

where M is a third body, most commonly nitrogen.

Ozone is formed when there are sufficient amounts of NO_X and VOCs, adequate sunlight, and high enough temperatures. Concentrations therefore tend to peak during the summer months and when dispersion is constrained by meteorological conditions and local topography.

2.3 Quantifying NO_2 emissions and *f*-NO_2

2.3.1 Laboratory measurements

Laboratory measurements have shown that the proportion of NO_x in vehicle exhaust which is emitted as primary NO_2 varies according to factors such as vehicle group, operating condition and, indeed, the measurement method **(Latham *et al.*, 2001)**. An extensive review of recent NO_2 measurements in Europe was conducted by **Pastramas *et al.* (2014)**. The findings are not repeated here, but several of the studies mentioned that the types and combinations of after-treatment technology installed had a strong influence of on NO_2 emissions. It was noted that, in general, the increased market penetration of diesel vehicles and the emissions of NO_2 from technologies such as catalytically regenerative traps have led to a rise in the NO_2/NO_X ratio in vehicle exhaust.

2.3.2 Real-world remote sensing measurements

Vehicle emission factors can also be determined by remote sensing, whereby individual vehicle plumes are measured *in situ* using an UV/infrared beam. Although large numbers of vehicles can be sampled in this way, remote sensing has some limitations. For example, a bias is introduced to the dataset because sampling must be undertaken when vehicles are under load. In measurement campaigns in the UK, **Carslaw and Rhys-Tyler (2013)** produced NO_2 emission factors for a number of vehicle categories. In the United States, **Bishop and Stedman (2008)** observed an increase in *f*-NO_2 due to diesel particulate filters (DPFs).

2.3.3 Analysis of ambient monitoring data

In the late 2000s, several studies in the UK and Europe analysed the measurements from ambient air quality monitoring stations to determine long-term trends in NO_X and NO_2 at urban background and roadside sites. It is difficult to derive *f*-NO_2 accurately from ambient monitoring data because concentrations are determined by the overall equilibrium between ozone, NO and NO_2, of which the *f*-NO_2 ratio is only one of the drivers. In the UK the Netcen model is the main one used to estimate *f*-NO_2. The model is a one-dimensional representation of the interaction between *f*-NO_2 and NO_X, NO_2 and O_3 concentrations at roadside locations **(Grice *et al.*, 2009)**. It is underpinned by the basic assumption that an appropriate background site can be paired with a roadside monitoring site, such that the NO_X, NO_2 and O_3 measured at the background site are representative of the background concentrations at the roadside site.

Analyses have shown a downward trend in NO_X concentrations across urban areas, which is consistent with emissions from road traffic **(Carslaw *et al.*, 2011a; Grice *et al.*, 2009)**. However, this trend has not been observed for NO_2 concentrations at roadside locations **(Carslaw *et al.*, 2011b; Keuken *et al.*, 2012; Kurtenbach *et al.*, 2012; Tian *et al.*, 2011; Minoura and Ito, 2010)**. Analyses of ambient NO_2 concentrations in urban areas in the UK have indicated that a significant proportion of NO_2 must be emitted directly from the exhaust, and it has been found that the NO_2/NO_x proportion has increased as a result of the introduction of DPFs on buses (notably in London), the increased penetration of diesel vehicles in the passenger car fleet, the increased use of oxidation catalysts on diesel vehicles, and new light-duty and heavy-duty engine technologies **(Carslaw and Beevers, 2004; Carslaw, 2005)**.

In Rotterdam, **Keuken *et al.* (2009)** found that *f*-NO_2 for road traffic increased from 9% in1986 to 13% in 2005. This change was smaller than that observed in London **(Carslaw, 2005)**. It was suggested that this may have been related to the larger fraction of diesel vehicles in the UK fleet.

An analysis of air quality monitoring data from two sites near heavily trafficked roads in Helsinki by **Antilla *et al*. (2011)** showed that primary NO_2 emissions increased from below 10% in the 1990s to about 20% in 2009, with a more distinctive increase during the most recent years. Again, this change was considered to be due to the increase in the proportion of diesel cars in Finland.

Grice *et al.* (2009) assessed trends in *f*-NO_2 emissions in urban areas of ten European Union countries. The country-level values for *f*-NO_2 were derived using national emissions inventory data for NO_X, information on vehicle fleets, and estimates of *f*-NO_2 for different types of vehicle. The results - summarised in **Table 2.1** - show that *f*-NO_2 has increased since 1995 in each country, and that the rate of increase has been greatest since 2000. This trend is expected to continue to 2020 as a result of the further penetration of exhaust after-treatment technologies for diesel vehicles in the fleets.

Table 2.1: Average *f*-NO_2 in urban areas of European countries between 1995 and 2020 (Grice *et al.*, 2009)

	f-NO_2 (%)					
Country	**1995**	**2000**	**2005**	**2010**	**2015**	**2020**
Austria	6.5	7.3	11.1	20	29.1	31.8
Czech Republic	6.1	6.3	9.8	15.1	20.3	19.6
Finland	4.8	5.9	7.1	10.2	18.5	23.2
France	5.8	6.9	13.9	25.9	36.1	40.7
Germany	5.5	6.0	10.1	19.3	27.8	31.1
Greece	6.9	6.9	7.3	6.9	7.3	7.6
Italy	5.9	6.4	8.8	14.5	21.1	24.9
Netherlands	5.1	5.6	8.6	17.3	27.0	27.3
Spain	5.8	6.5	10.5	18.7	27.5	32.9
UK	5.4	5.9	10.2	21.0	31.8	35.6

Similar effects have been observed in the United States. For example, a number of studies conducted in Oakland, California and have linked an increase in *f*-NO_2 to a sharp rise in the use of DPFs; the use of DPFs on trucks in California increased from 2% to 99% between 2009 and 2013. **Dallman *et al.* (2012)** measured NO_2 emissions from heavy-duty diesel trucks in a road tunnel in Oakland. Pollutant concentrations in the exhaust plumes of individual trucks were measured with a high time resolution, with emission factors being calculated using a carbon balance method. Fleet-average emission factors for NO_x decreased by 30-37% between 2006 and 2010, whereas a 34% increase in the NO_2 emission factor was observed. The emission factors for NO_2 were found to be negatively correlated with those for all other pollutants, probably due to the use of DPFs. **Harley (2014)** assessed emissions from trucks with DPF retrofits at the Port of Oakland, and found that fleet-average NO_X emission factors had decreased by 53% between 2009 and 2013. However, direct emissions of NO_2 increased, and consequently *f*-NO_2 increased from 3% to 18%. The study found that DPF-equipped trucks had substantially higher NO_2 emission factors than trucks without DPFs. The newest trucks equipped with both DPFs and selective catalytic reduction (SCR) systems for NO_X control had an average NO_2 emission factor that was equal to that of the truck fleet in 2009. A summary of the observed emission factors from the study is provided in **Table 2.2**.

Table 2.2: Observed emission factors for NO_X, NO_2 and primary NO_2 for HDVs in the Port of Oakland

Fleet	NO_X (g/kg)	NO_2 (g/kg)	*f*-NO_2
2009 (2% DPF, 0% SCR)	33	1.0	**3%**
2011 (54% DPF, 2% SCR)	18	2.1	**12%**
2013 (99% DPF, 9% SCR)	15	2.8	**18%**

In Hong Kong, **Tian *et al.* (2011)** used roadside concentrations of NO_X and NO_2, and background ozone (O_3) to estimate the mean *f*-NO_2 value in vehicle exhaust between 1998 and 2008 in combination with the method described in **Carslaw and Beevers (2005)** (See **Section 4.2.2**). A summary of the *f*-NO_2 values obtained for three roadside monitoring stations (CB, CL, MK) is shown in **Figure 2.1**. At all three roadside stations there was a sustained upward trend in *f*-NO_2 from 2003. The grey shading shows the 95% confidence intervals on the trend line. The orange and yellow shaded areas represent periods with diesel retrofit programmes for the light-duty vehicles and heavy-duty vehicles respectively.

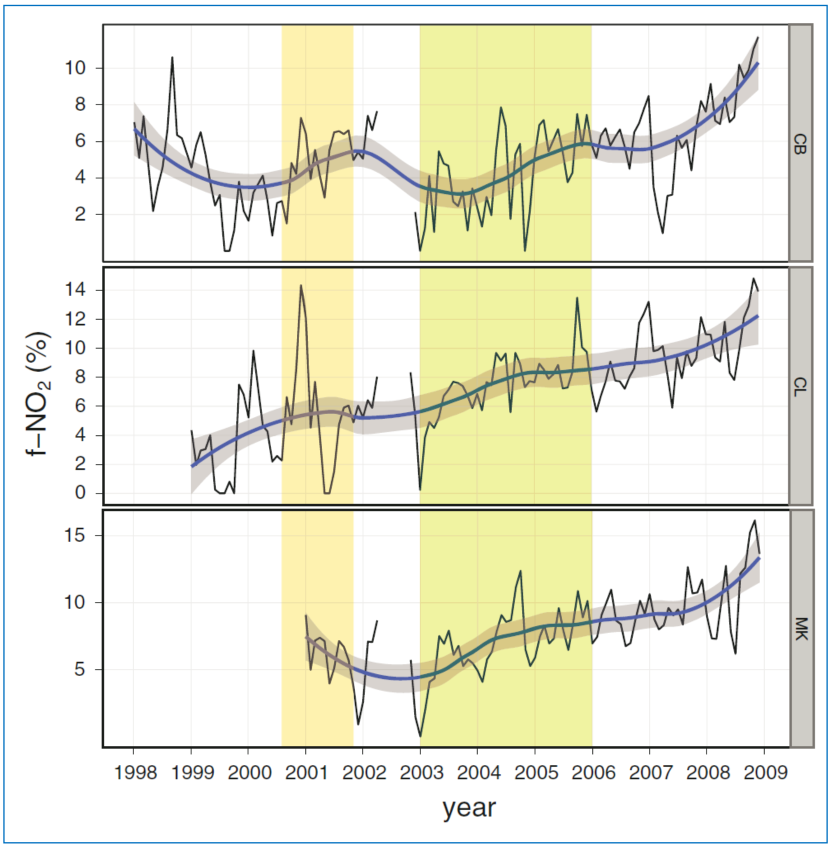

Figure 2.1: Monthly trends in the modelled *f*-NO_2 for three roadside monitoring stations in Hong Kong (1998-2008) (Tian *et al.*, 2011)

In Japan, **Minoura and Ito (2010)** developed a model for assessing *f*-NO_2 at roadside locations in Tokyo for the 2008 fleet. The model used real-time air quality monitoring data for NO, NO_X and O_3. At a key intersection, a *f*-NO_2 value of 7% was estimated.

2.3.4 Tunnel measurements

Due to the limited dispersion and dilution conditions in a tunnel environment, pollutant concentrations tend to be higher than in ambient air. As a result tunnel samples are used to derive tunnel emission factors. Inside a road tunnel there is usually a high concentration of NO_X from vehicle exhaust, and any available oxidant - principally ozone - should be removed relatively quickly. Once the ozone is removed, NO_2 formation via **Reaction 4** will stop. Inside a tunnel most of the NO_2 in the air is therefore primary in origin; it is emitted directly from vehicle exhaust pipes rather than being formed in the tunnel atmosphere. Whilst it is possible that certain reactions could lead to the formation of NO_2 in longer tunnels, the NO_2/NOx proportion in the air from tunnel ventilation outlets ought to provide a reliable indication of the overall average NO_2/NO_X proportion in vehicle exhaust. **Boulter *et al*. (2007)** used this approach to determine *f*-NO_2 values for light-duty and heavy-duty vehicles in the Hatfield and Bell Common tunnels in the UK.

3 SYDNEY VEHICLE FLEETS AND EMISSIONS

3.1 Sydney vehicle fleets

3.1.1 Overview

When estimating emissions from road transport it is important to distinguish between different types of vehicle, between vehicles using different types of fuel, and between vehicles conforming to different emission regulations. These distinctions are relevant to all pollutants, including NO_X and primary NO_2, and they are explored in this Section of the Report. The information on the composition of vehicle fleets in Sydney has been taken from the NSW GMR emissions inventory **(NSW EPA (2012b) and references therein)**. It should be noted that the fleet projections for future years were based on the knowledge at the time the inventory was compiled, and more recent work by NSW EPA has indicated some discrepancies between the projections and RMS statistical data. These discrepancies have been noted in this Report.

3.1.2 Vehicle group and fuel type

Nine vehicle groups have been defined in the inventory to reflect differences in emissions behaviour. These groups are summarised in **Table 3.1**.

Table 3.1: Vehicle groups in the NSW GMR emissions inventory

Vehicle group	Vehicles included
Petrol car[(a)]	Petrol car, 4WD[(e)], SUV[(f)] and people-mover
	LPG[(g)] car, 4WD
Diesel car[(a)]	Petrol car, 4WD, SUV and people-mover
Petrol LCV[(b)]	Petrol commercial vehicle < 3.5 tonnes GVM[(h)]
Diesel LCV	Diesel commercial vehicle < 3.5 tonnes GVM
Petrol HDV[(c)]	Petrol commercial vehicle < 3.5 tonnes GVM
Diesel Rigid HGV[(d)]	Diesel commercial vehicle 3.5 t < GVM < 25 t
Diesel Articulated HGV	Diesel commercial vehicle > 25 tonnes GVM
Diesel bus	Diesel bus > 3.5 tonnes GVM
Motorcycle	Any powered two-wheel vehicle

(a) Referred to as 'passenger vehicle' in the inventory
(b) LCV = light commercial vehicle
(c) HDV = heavy-duty vehicle
(d) HGV = heavy goods vehicle
(e) 4WD = four-wheel drive
(f) SUV = sports-utility vehicle
(g) LPG = liquefied petroleum gas
(h) GVM = gross vehicle mass

The typical ('default') composition of the traffic in terms of these vehicle groups varies according to the type of road, and some examples from the inventory are presented in **Figure 3.1**. The Figure shows the VKT-weighted composition of the traffic on residential roads and highways between 2003 and 2041. There is a larger proportion of HGVs on highways than on residential roads.

In recent years the refinement of light-duty diesel engines and their superior fuel economy relative to petrol engines has led to increased sales and a larger market share. **Figure 3.1** also shows that, as a consequence, there are projected increases in the proportions of diesel cars and diesel LCVs in the future. The petrol/diesel splits for cars and LCVs in the inventory are determined based on sales (registration) statistics, 'attrition' functions, and VKT. The assumptions underlying the fuel splits are summarised below.

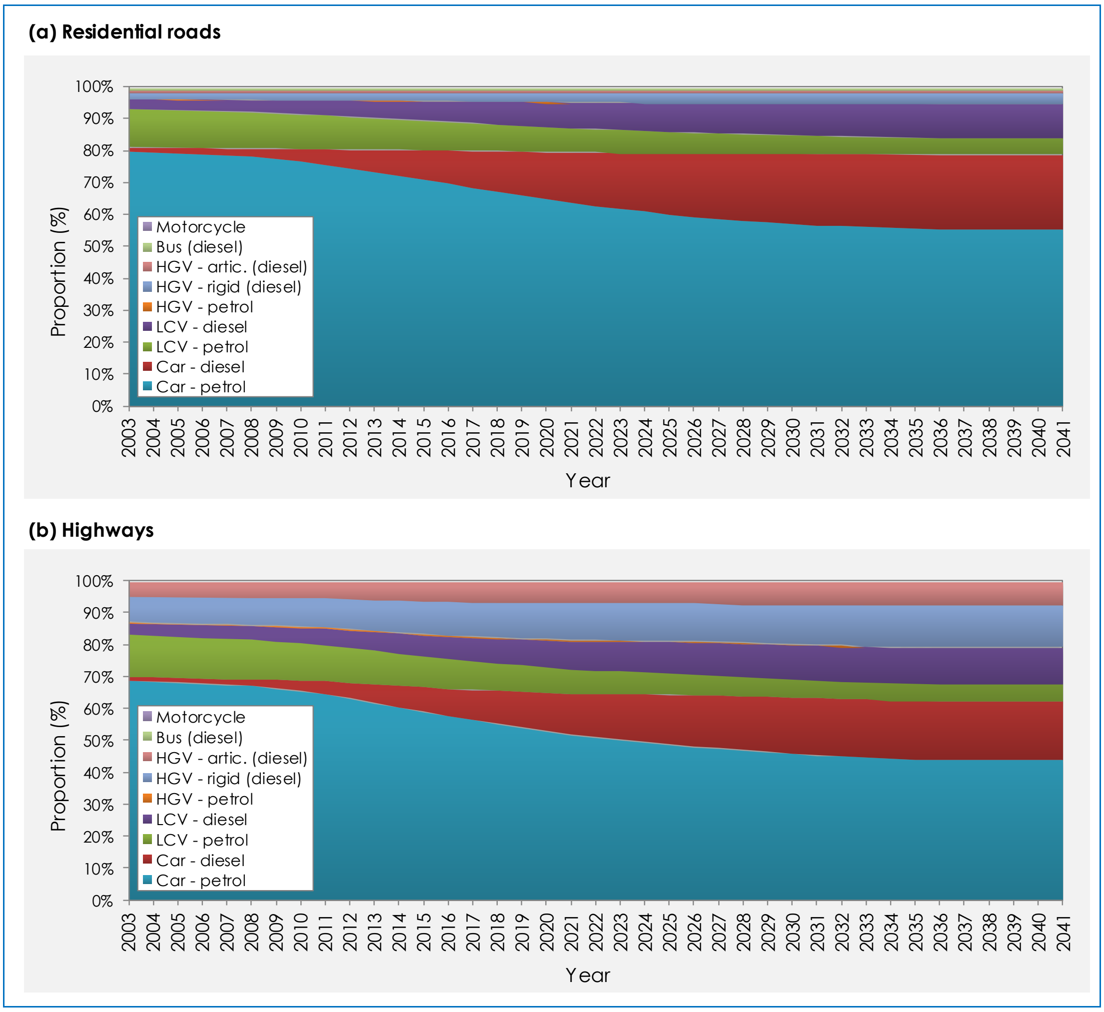

Figure 3.1: Composition of traffic on residential roads and highways between 2003 and 2041

The sales data for light-duty passenger vehicles (cars, people-movers and 4WD/SUV) are shown in **Figure 3.2**. The fuel splits up to 2008 were based on an analysis of RMS registration data for the GMR. Historically, diesel vehicles have formed only a small proportion of the car fleet and VKT. However, there was a strong growth in diesel passenger vehicle sales between 2004 and 2008, increasing from 0.2% to 6.3%, while diesel 4WD/SUV sales increased from 9% to 25%. Beyond 2008, the growth was projected to continue strongly, with the diesel proportion of passenger vehicles stabilising at 25% in 2026, and the diesel proportion of 4WD/SUV stabilising at 42% in 2020. The proportion of 4WD/SUV in the total passenger vehicle fleet increased from 17% in 2001 to 24% in 2008. Further growth was projected, with stabilisation at 34% in 2026.

The equivalent data for LCVs are shown in **Figure 3.3**. Diesel LCVs have historically made up 15-30% of total LCV sales, but as with cars the sales have risen rapidly in recent years **(NSW EPA, 2012b)**. The data show that the proportion of diesel vehicles increased from 16% in 2000 to 45% in 2008, and was projected to continue to grow further, stabilising at 70% in 2026.

For each vehicle group an attrition function was derived from historical RTA registration records. The attrition function gives proportion of vehicles surviving per year as a function of age. A fixed VKT as a function of vehicle age was also applied to each vehicle group, with newer vehicles accumulating greater VKT per year than older vehicles **(NSW EPA, 2012b)**.

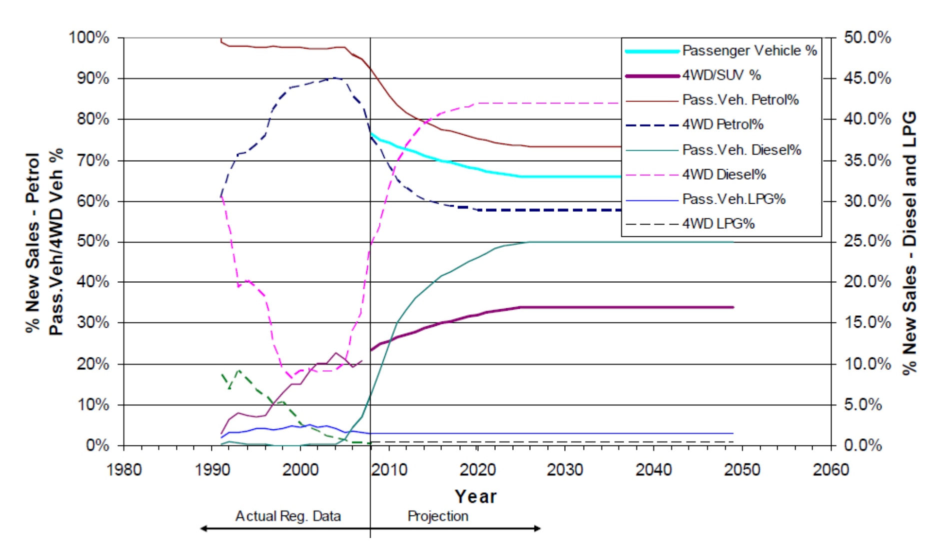

Figure 3.2: Passenger vehicle fleet sales proportions projections (NSW EPA, 2012b)

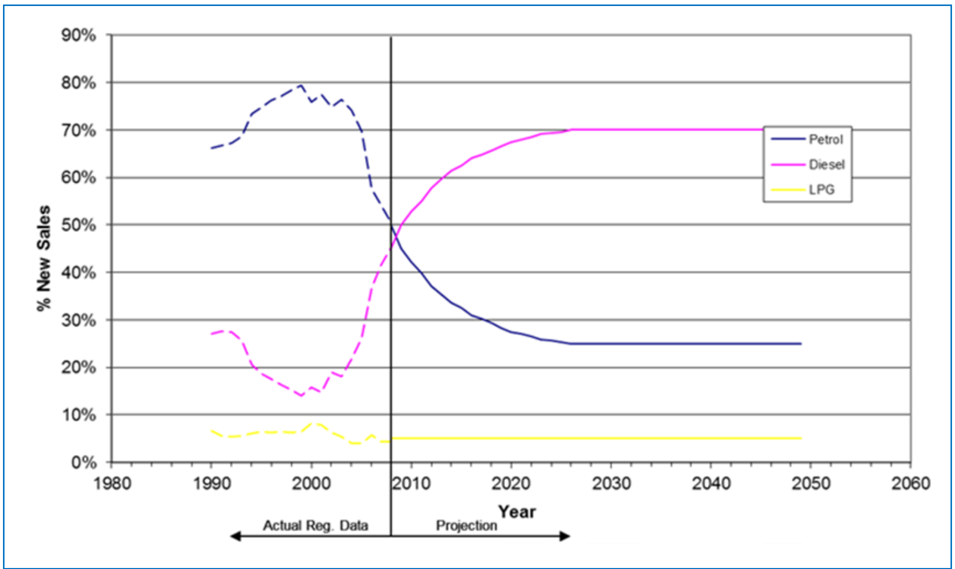

Figure 3.3: Light commercial vehicle fleet sales proportions projections (NSW EPA, 2012b)

For each vehicle group the growth in the total fleet size was estimated based on population projections for passenger vehicles, and economic activity (state gross product) for commercial vehicles.

VKT data were generated by the NSW Department of Transport's Bureau of Transport Statistics (BTS) using two demand models:

- Strategic Travel Model (STM), which estimates passenger vehicle activity.
- Freight Movement Model, which estimates commercial vehicle activity.

The resulting overall VKT-weighted petrol/diesel splits used for cars and LCVs in the inventory are given in **Figure 3.4**.

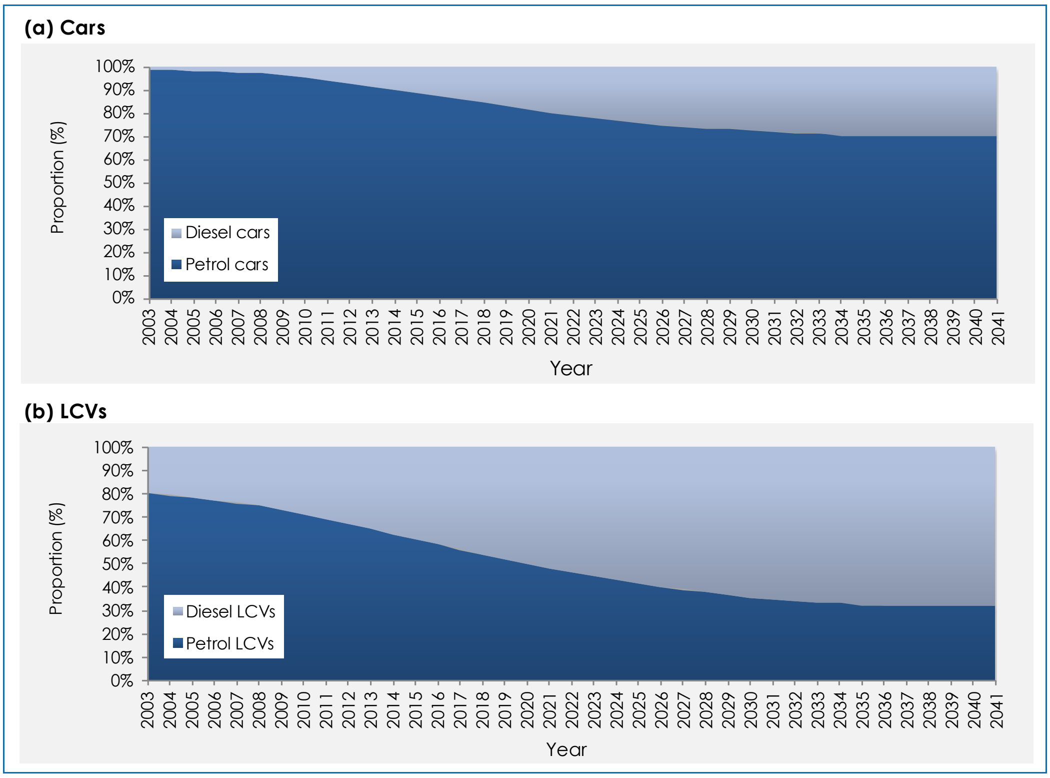

Figure 3.4: Fuel splits for cars and LCVs (weighted for VKT)

A more recent examination of vehicle registration data has been conducted by NSW EPA, and the initial findings are summarised in **Appendix A**. The EPA analysis has highlighted some discrepancies between the actual vehicle sales figures and the inventory projections. The actual growth in diesel car sales has been lower than projected in the inventory. For example, the actual NSW-wide proportion of sales in 2014 was 8%, compared with a projection for 2014 of 19%. On the other hand, the actual diesel proportion of LCVs in 2014 (89%) was substantially higher than the projection (61%). It was beyond the scope of this Pacific Environment study to include these discrepancies. NSW EPA is currently updating the emissions inventory, and the fleet information will be revised as part of this process. The emission modelling work presented here should also be updated once the new inventory has been finalised.

3.1.3 Age distribution and emission standards

The age distribution of fleet within a vehicle group is a major determinant of the emissions from that group in a given year. This is because new vehicles are required to comply with more stringent emission regulations than older vehicles. The allocation of vehicle model years to specific emission standards allows this to be taken into account, and some specific examples for three different calendar years are shown in **Figure 3.5**

Australia has had road vehicle emission standards for new vehicles in place since the early 1970s, and these have been progressively tightened. Australia's vehicle emission standards are set nationally in the Australian Design Rules (ADRs). The current standards reflect Australia's commitment to harmonise the ADRs with the international standards.

For light-duty vehicles the Euro 5 emission standards commenced in Australia for new vehicle models from 1 November 2013. The Euro 6 standards will commence for new models from 1 July 2017 **(Australian Government, 2010)**. With full implementation of Euro 6 the World Harmonized Light-duty Vehicle Test Cycle (WLTC) will replace the current test cycle, and a variety of test parameters will be adjusted to close loopholes and address shortcomings of the existing procedure **(Mock *et al.*, 2014)**.

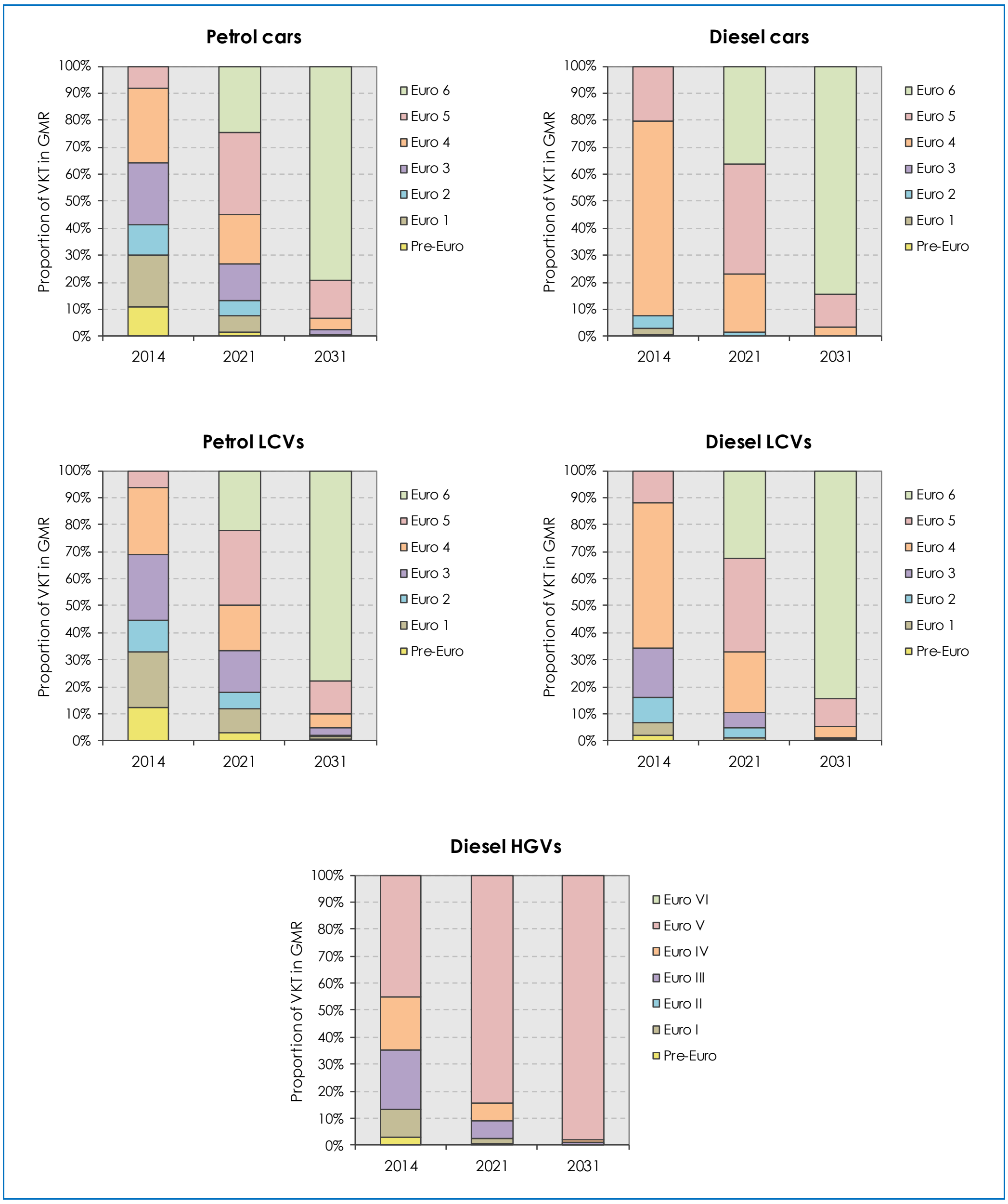

Figure 3.5: Fleet breakdown by emission standard (VKT weighted)

For heavy-duty vehicles, whilst the Euro VI standards reduce the limit on NO_X emissions by 77% relative to Euro V, and by 89% relative to Euro IV, advanced test protocols that improve real-world conformity to NO_X limits result in reductions that are closer to 95% **(Muncrief, 2015)**. The NO_X reductions will be achieved with combustion improvements (high-pressure fuel injection and advanced air/fuel management), exhaust gas recirculation, and a closed-loop SCR system. However, at present there is some uncertainty concerning the adoption of Euro VI in Australia.

It is worth adding that several shortcomings of the regulations have been identified in the EU. For heavy-duty vehicles the Euro V standards have not achieved the anticipated reductions in NO_X emissions **(Ligterink *et al.*, 2009)**. Whilst the Euro 5 standards have resulted in dramatic reductions in PM emissions from light-duty diesels, real-world NO_X emissions from Euro V trucks and buses have continued to far exceed certification limits **(Carslaw *et al.*, 2011a)**. Therefore, given the availability of ultra-low sulphur fuel, it is argued that greater air quality benefits can be gained by from a leap from Euro IV to Euro VI standards, particularly for countries that are at prior standards for heavy-duty vehicles **(Chambliss and Bandivandekar, 2015)**.

3.2 NO_X emissions from road transport in Sydney

The importance of road transport as a source of NO_X in Sydney can be illustrated by reference to sectoral emissions. The most detailed and comprehensive source of information on emissions in the Sydney area is the EPA's emissions inventory **(NSW EPA, 2012a)**. The base year of the latest inventory is 2008. Projections are currently available for 2011, 2016, 2021, 2026, 2031, 2036 and 2041, and the calculations also go back to 2003. The data for anthropogenic and biogenic emissions in Sydney, and also for road transport in Sydney, have been extracted from the inventory by EPA[4] and are presented here.

Figure 3.6 shows that in Sydney during 2011 road transport was the single largest sectoral contributor to emissions of NO_X (57%). The breakdown of NO_X emissions in 2011 from the road transport sector by process and vehicle group is presented in **Figure 3.7**. Petrol passenger vehicles (mainly cars) account for a large proportion of the VKT in Sydney, and they were responsible for 45% of NO_X from road transport in Sydney in 2011. Heavy-duty diesel vehicles are disproportionate contributors to NOx emissions due to their inherent combustion characteristics, high operating mass (and hence high fuel usage) and level of emission control technology **(NSW EPA, 2012b)**.

The projections of road transport NO_X and NO_2 emissions are broken down by vehicle group in **Figure** 3.8. This Figure assumes no adoption of Euro VI in Australia, but even so there are projected to be substantial reductions in emissions NO_X between 2011 and 2036 due to the effects of emission-control technology.

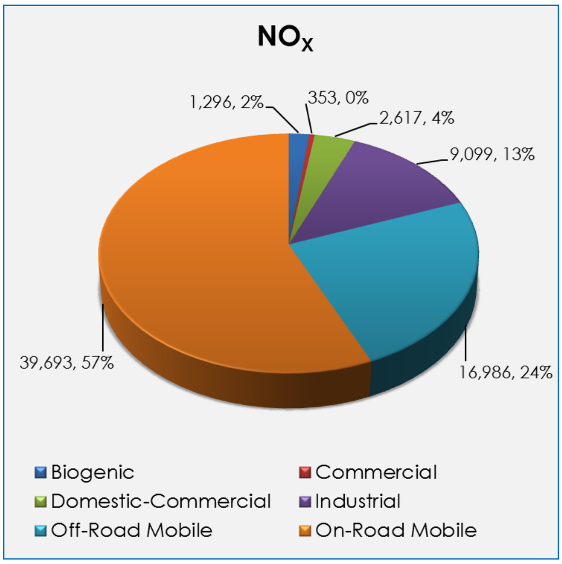

Figure 3.6: Sectoral emissions of NO_X in Sydney, 2011

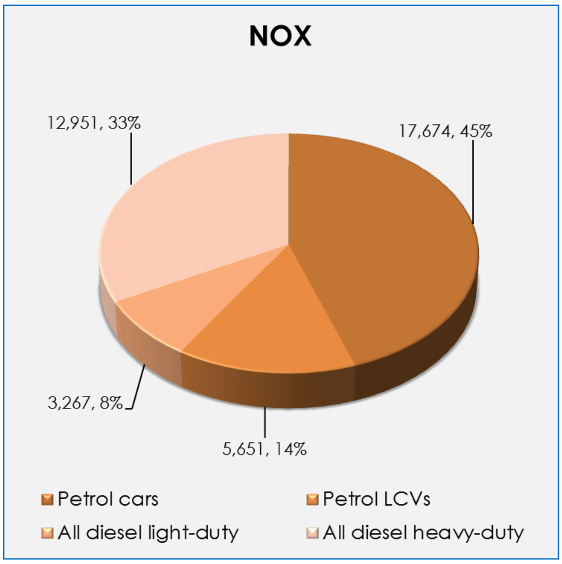

Figure 3.7: Breakdown of road transport emissions of NO_X in Sydney, 2011[5]

[4] The data were provided for the project *Economic Analysis to Inform the National Plan for Clean Air (Particles)*, undertaken by Pacific Environment on behalf of the NEPC Service Corporation.

[5] Vehicle types having a very small contribution – heavy-duty petrol vehicles and motorcycles - have been excluded to simplify the presentation.

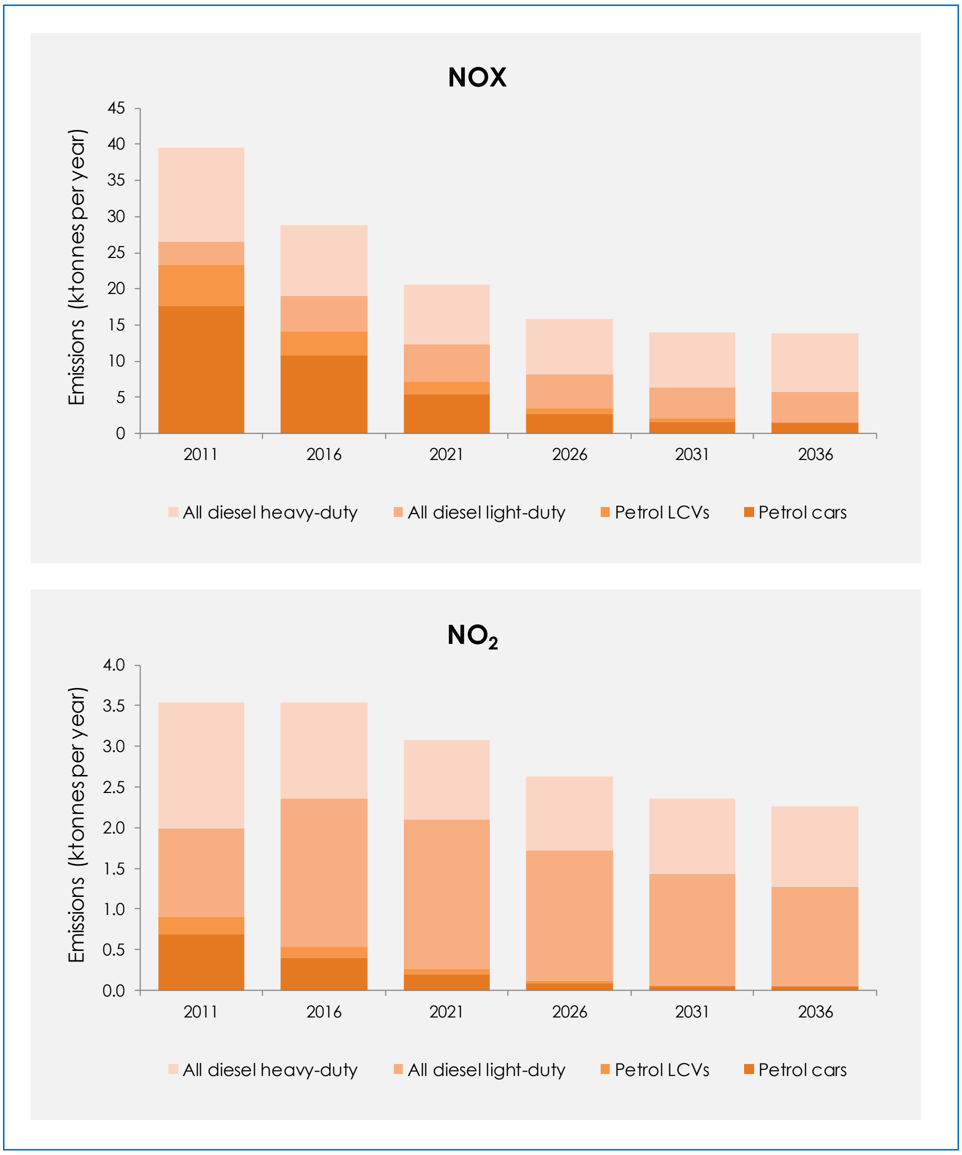

Figure 3.8: Projections of road transport NO_X and NO_2 emissions – Sydney, 2011-2036

4 EVIDENCE RELATING TO PRIMARY NO_2 EMISSIONS IN SYDNEY

4.1 The EPA inventory model

4.1.1 Method

The method for calculating hot running emissions for the inventory involved the use of matrices of base 'composite' emission factors for the following cases:

- The nine vehicle groups identified earlier, with the composite emission factor for each vehicle group taking into account VKT by age and the emission factors for specific emission standards.
- Five road types (residential, arterial, commercial arterial, commercial highway, highway/ freeway), as defined in the NSW GMR emissions inventory. Further information on the mapping of these categories to the RTA road network road types and the RTA's EMME2 model is provided in the inventory report **(NSW EPA, 2012b)**. Some types of road – notably arterial roads – are associated with higher emissions for a given average speed than others.
- Nine calendar years (2003, 2008, 2011, 2016, 2021, 2026, 2031, 2036 and 2041). The year defined the composition of the fleet for each type of vehicle, allowing for technological changes.

Each base composite emission factor is defined for a VKT-weighted average speed (the base speed) associated with the corresponding road type. Dimensionless correction factors – in the form of 6th-order polynomial functions - are then applied to the base emission factors, taking into account the actual speed on a road. According to EPA, the speed correction factors are valid up to 110 km/h for light-duty vehicles, and up to 100 km/h for heavy-duty vehicles.

Emission factors have also been provided by EPA for heavy-duty vehicles both with and without the implementation of the ADR80/04 regulation (Euro VI).

The emission factor for a given average speed[6] is calculated as follows:

$$EF_{HotSpd} = EF_{HotBasSpd} \times \frac{SCF_{Spd}}{SCF_{BasSpd}}$$

Equation 1

Where:

EF_{HotSpd}	is the composite emission factor (in g/km) for the defined speed
$EF_{HotBasSpd}$	is the composite emission factor (in g/km) for the base speed
SCF_{Spd}	is the speed-correction factor for the defined speed
SCF_{BasSpd}	is the speed-correction factor for the base speed

Each speed-correction factor is a 6th order polynomial: $SCF = aV^6 + bV^5 + \ldots + fV + g$, where a to g are constants and V is the speed in km/h.

[6] 'Average speed' should not be confused with 'constant speed'. The former is calculated for a driving cycle which includes periods of acceleration, deceleration, cruise and idle, as encountered in real-world traffic.

No emission factors for NO_2 were available for Australian vehicles. Primary NO_2 emissions were therefore built into the inventory model using the *f*-NO_2 values for the various vehicle categories and emission standards that have recently been developed for the EMEP/EEA Air Pollutant Emission Inventory Guidebook and COPERT 4 model **(Pastramas *et al.*, 2014)**. These values, which were based on a review of the literature, are shown in **Table 4.1**. The *f*-NO_2 values are independent of road type and vehicle speed.

Table 4.1: Values of *f*-NO_2 from Pastramas *et al.* (2014)

Light-duty vehicles Emission standard/ technology	*f*-NO_2 Petrol cars and LCVs	*f*-NO_2 Diesel cars and LCVs	Heavy-duty vehicles Emission standard/ technology	*f*-NO_2 Diesel HGVs and buses
Pre-Euro	0.07	0.15	Pre-Euro	0.11
Euro 1	0.06	0.13	Euro I	0.11
Euro 2	0.05	0.13	Euro II	0.11
Euro 3	0.04	0.27	Euro III	0.14
Euro 3 + DPF	-	0.51	Euro III + CRT	0.36
Euro 4	0.05	0.46	Euro IV	0.10
Euro 4 + DPF	-	0.42	-	-
Euro 5	0.03	0.33	Euro V	0.17
Euro 6	0.03	0.30	Euro VI	0.08

For petrol cars and LDVs, *f*-NO_2 is quite low and does not vary greatly by emission standard. The efficiency of the three-way catalyst has led to a reduction in NOx emissions over the consecutive Euro level vehicles, and at the same time has kept *f*-NO_2 levels low **(Pastramas *et al.*, 2014)**. For diesel cars and LDVs the range of values is larger, and varies with emission standard and configuration of the emission-control technology. For diesel HGVs and buses the values are generally between those of petrol and diesel LDVs, but are relatively high for vehicles equipped with continuously regenerating traps (CRTs).

Pastramas *et al.* (2014) note that, due to the fact that few measurements of *f*-NO_2 from Euro 6/VI vehicles have been made and the real-world behaviour of such vehicles is uncertain, the final values include an assumption regarding the after-treatment technologies that are expected to be used. This assumption is necessary because NO_2 emissions are highly dependent upon the after-treatment technology. For exmple, SCR can limit *f*-NO_2 to10-20% in real-world driving, but if a catalysed DPF follows the SCR then this could increase *f*-NO_2 to 50%. The values in **Table 4.1** assume that SCR will be the dominant NO_X-reduction technology, with some 70% of SCRs preceding the DPF and 30% of SCRs following the DPF.

4.1.2 Results

The time series (2003-2041) of the NO_X and NO_2 emission factors for the main vehicle groups in the inventory model, and the associated values of *f*-NO_2, are shown for the 'highway/freeway' road type in **Figure 4.1**. These emission factors were calculated for a speed of 80 km/h, and for each vehicle group they take into account the age distribution of the fleet. Emission factors are presented for situations with and without the adoption of the Euro VI regulation. Whilst the total NO_X emission factors are predicted to decrease with time, the value of *f*-NO_2 exhibits different patterns. The plots show that the model predicts a slight decrease in *f*-NO_2 for petrol vehicles in the future. In the case of light-duty diesel vehicles there is a sharp increase in *f*-NO_2 after 2008, with a peak between 2010 and 2020, and a gradual reduction further into the future. The *f*-NO_2 values for heavy-duty diesel vehicles are relatively stable with time, at around 10-12%. **Figure 4.2** shows the equivalent emission factors for the highway/freeway road type, weighted for the default traffic mix for this type of road in each year.

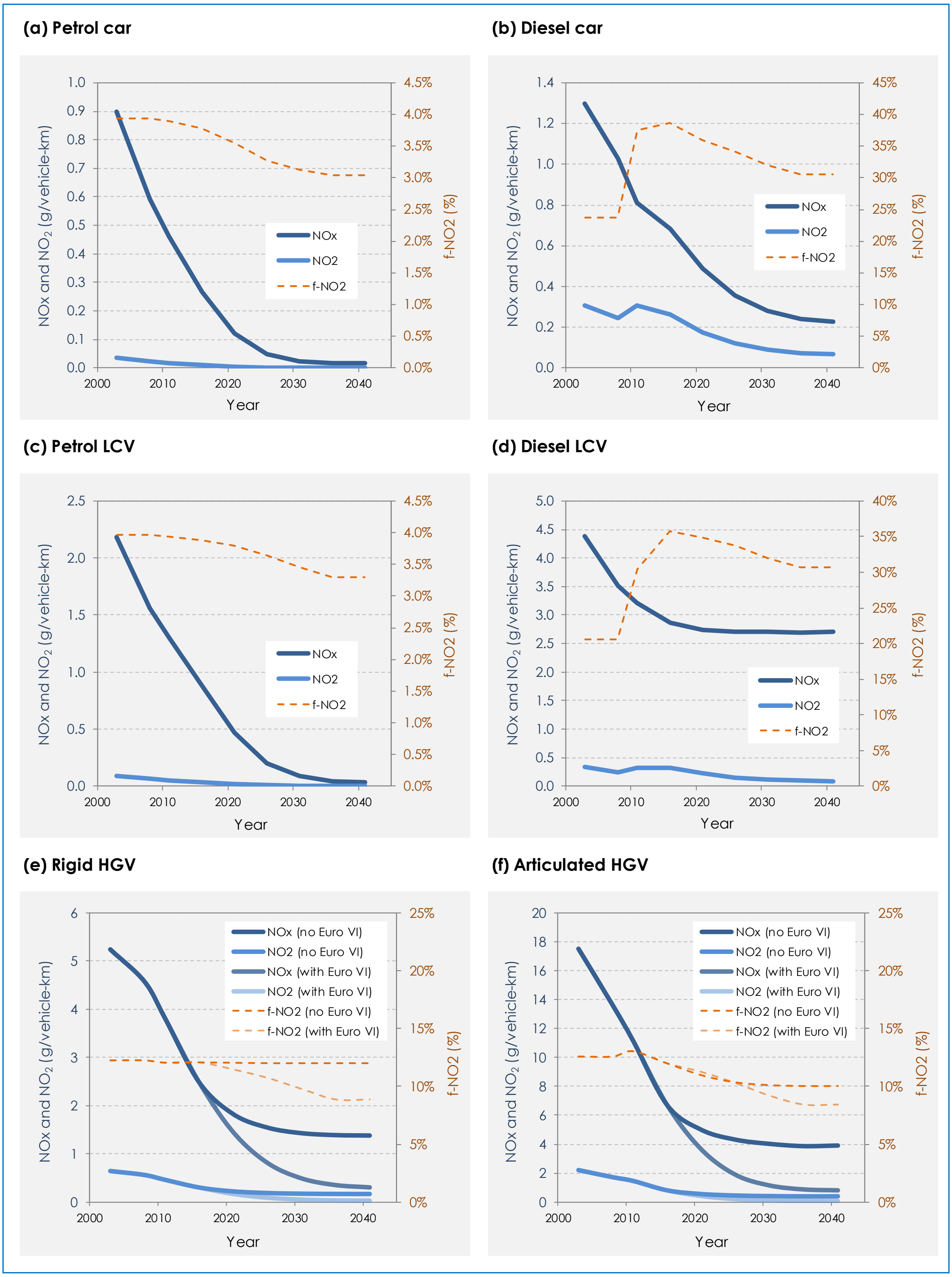

Figure 4.1: Emission factors for NOx, NO_2 and *f*-NO_2 from the GMR emissions inventory model. The information relates to the driving conditions and fleet for highways/freeways.

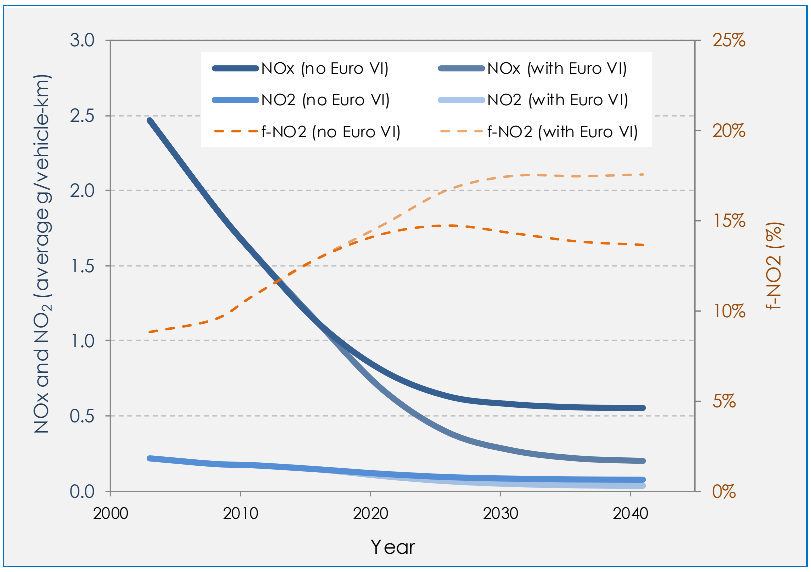

Figure 4.2: Emission factors for NOx, NO_2 and *f*-NO_2 from the GMR emissions inventory model for highways/freeways, weighted for default traffic mix.

The contributions of the different vehicle groups to the traffic-weighted emission factors for NO_X and NO_2 are shown in **Figure 4.3** (assuming no adoption of Euro VI). These graphs show the disproportionate contribution of diesel heavy goods vehicles to the fleet-weighted emission factor in each year. Moreover, as a fraction of the total emission factor the contribution of HGVs is projected to increase in the future.

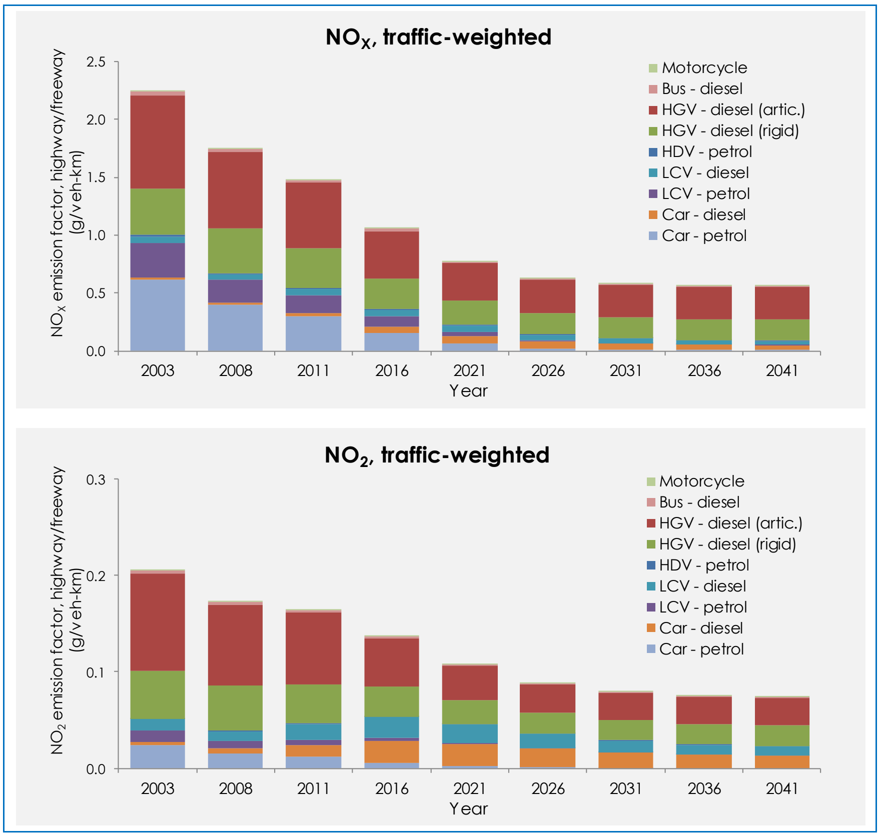

Figure 4.3: Emission factors for NOx and NO_2 and *f*-NO_2 by vehicle group for highways/freeways, weighted by traffic mix (no Euro VI)

Figure 4.4 shows how *f*-NO_2 varies by road type in the inventory model, based on a nominal traffic composition and speed in each case (assuming no adoption of Euro VI). It can be seen that the values for highway/freeway conditions are somewhat different from those for other road types. This is essentially linked to the higher proportion of heavy-duty vehicles on highways and freeways. In addition, the traffic-weighted *f*-NO_2 value for highways is predicted to peak at around 14% between 2020 and 2025, whereas on other types of road the value is predicted to peak at a slightly higher value (around 16%) and at a later date (around 2030).

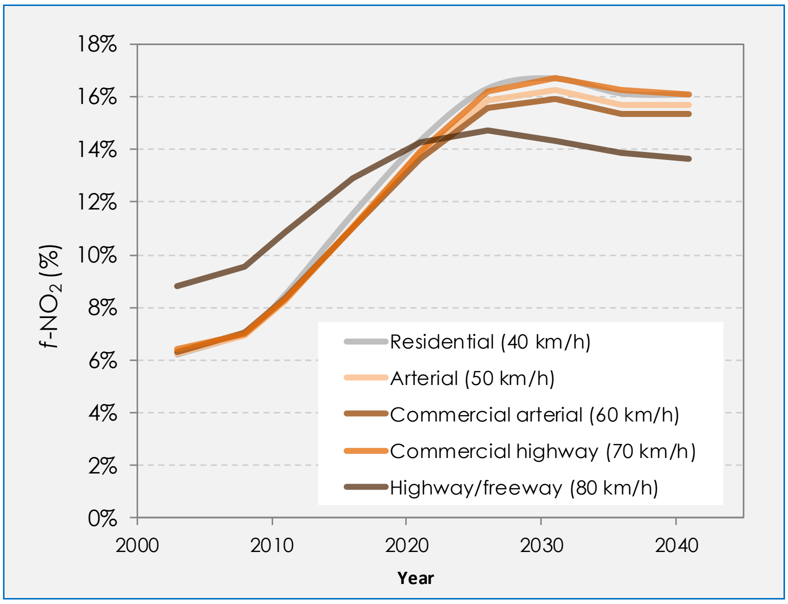

Figure 4.4: *f*-NO_2 by road type from the GMR emissions inventory model (no Euro VI)

It was noted in **Section 3.1.2** that there are potentially some discrepancies between the acutal vehicle fleets in the GMR and the projected fleets in the EPA emissions inventory. These discrepancies are likely to relate mainly to the fuel split for LDVs.

The sensitivy of the inventory projections to the fuel split for cars was investigated. Four different cases were considered (highways only):

- The default projection for the uptake of diesel cars, reaching a maximum of around 18%.
- An accelerated uptake of diesel cars after 2015, with the diesel share gradually increasing to 100% by 2030.
- All cars running on petrol in all years (lower bound NO_X and NO_2 emissions).
- All cars running on diesel in all years (lower bound NO_X and NO_2 emissions).

The results are shown for the ‘no Euro VI’ case in **Figure 4.5**, and for the ‘with Euro VI’ case in **Figure 4.6**.

The average traffic-weighted NOx emission factor is not especially sensitive to the petrol/diesel split for light duty vehicles. For example, in the ‘no Euro VI’ case the assumption that all cars were running on diesel resulted in NO_X emissions that were between around 15% and 25% higher than in the default case.

NO_2 emissions were more sensitive to the diesel car proportion. The ‘all cars diesel’ case resulted in NO_2 emissions that were around 130% higher than the base case in 2011 and around 60% higher in 2031.

For f-NO_2 the sensitivity to the car fuel split was similar to that for NO_X emissions. In the default case the long-term value of f-NO_2 stabilised at just below 15%. With all cars running on diesel f-NO_2 peaked at around 25% in 2016, but stabilised at around 17% in the loger term. The results for NO_X and NO_2 in the 'with Euro VI' case are broadly similar to those for the 'no Euro VI' case. The main difference is that f-NO_2 stabilises at a higher value and, because of the lower emissions from HDVs, it is more sensitive to the car fuel split.

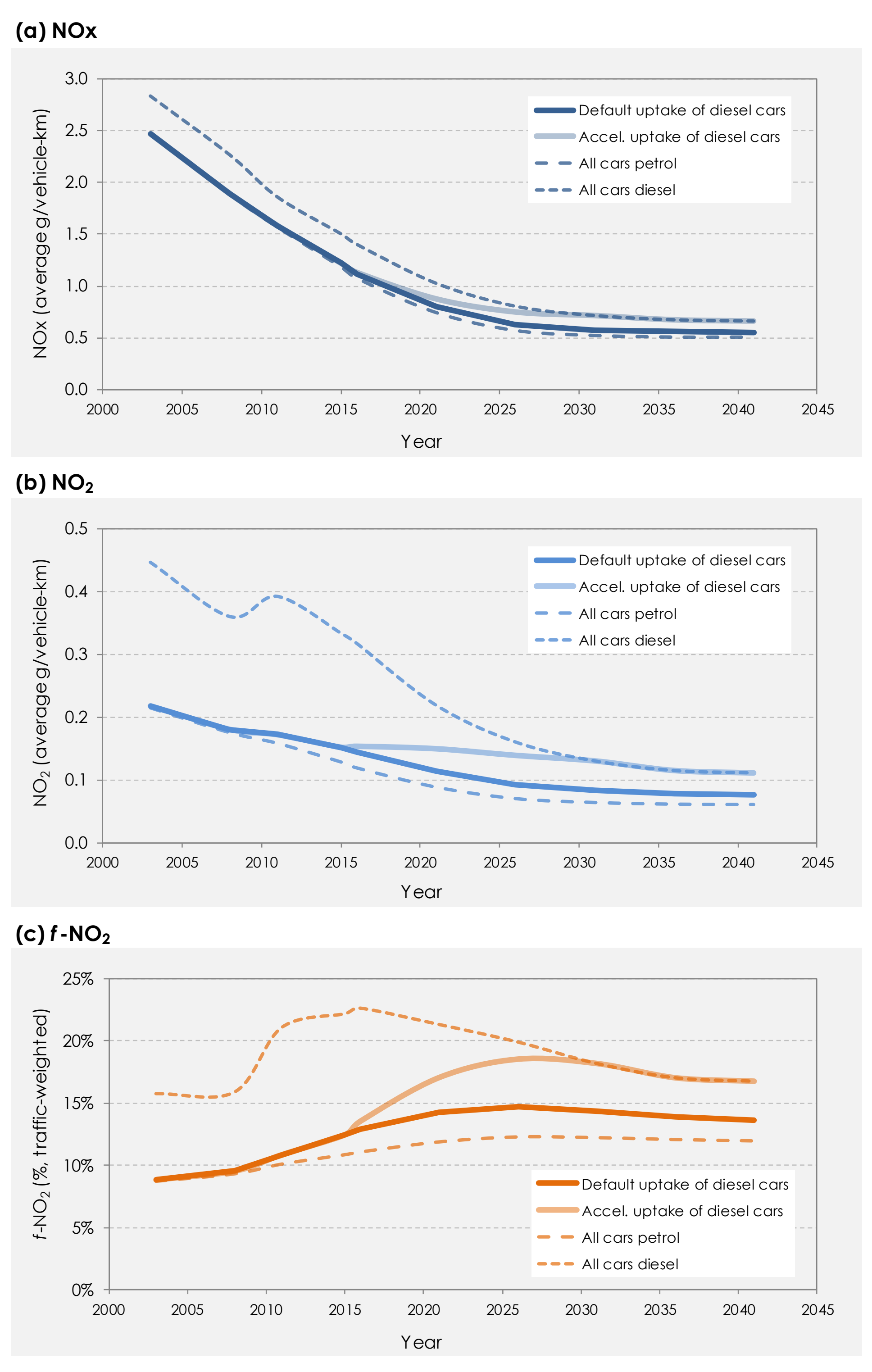

Figure 4.5: Sensitivity of f-NO_2 to car fuel split (no Euro VI)

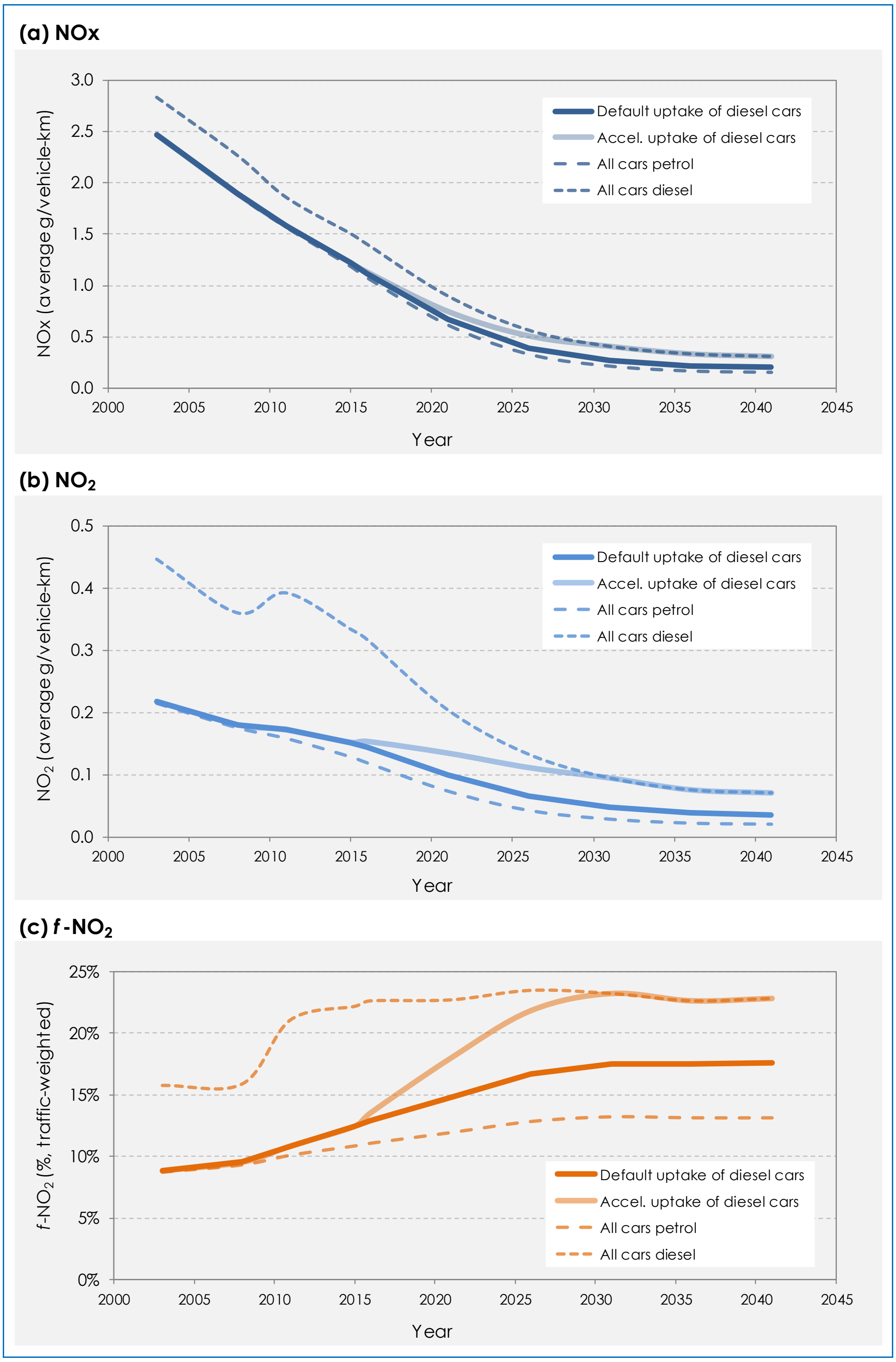

Figure 4.6: Sensitivity of *f*-NO_2 to car fuel split (with Euro VI)

4.2 Analysis of ambient air quality data

4.2.1 Upwind/downwind method

There are few long-term roadside monitoring sites in Sydney. One monitoring station which has been in place for several years is operated by RMS near to the M5 East Freeway. RMS has named this site 'F1'. The monitoring station is located approximately 20 metres north of the Freeway, near Flat Rock Road. Given the volume of traffic on the Freeway (around 100,000 vehicles per day), there ought to be a reasonably strong influence of the traffic on the measured concentrations. It is worth noting, however, that the station is in a dip and close to trees, and there is also a noise barrier between the station and the Freeway. These factors will influence the dispersion characteristics of the site.

The influence of the M5 East Freeway traffic on pollutant concentrations at the RMS F1 monitoring site was firstly investigated using the *Openair* software **(Carslaw, 2015a)**.

Figure 4.7 to **Figure 4.10** on the following page show two plots for NO_X and NO_2:

- A 'weighted mean' polar frequency plot. This shows the wind directions that contribute most to the overall mean concentration of each pollutant. The 'weighted mean' option in *Openair* provides the percentage contribution by wind direction.

- A bivariate polar plot. This shows how pollutant concentrations vary by wind speed and wind direction, with smoothing techniques giving a continuous surface. The monitoring station is located at the centre of each plot. The axes show the directions from which the wind is coming, and the distance from the origin indicates the wind speed; the further from the centre that concentrations appear, the higher the wind speeds when they were measured. Calm conditions appear close to the centre. The polar plot is a useful diagnostic tool for understanding potential sources of air pollutants at a given site. For many situations an increasing wind speed generally results in lower concentrations due to increased dilution through advection and increased turbulence. Ground-level, non-buoyant sources - such as road traffic – therefore tend to have highest concentrations under low wind speed conditions.

The concentrations of NO and NOx are most strongly influenced by light southerly winds (*i.e* from the direction of the M5 East Freeway). For NO_2 there are additional contributions from higher speed winds from the east.

The influence of the Freeway traffic on NO_X and NO_2 concentrations at the RMS F1 monitoring site was investigated further by filtering the concentrations by wind direction. The mean concentrations when the monitoring station was downwind and upwind of the Freeway were calculated. The Freeway has an approximately east-west alignment near the monitoring station (a line perpendicular to the road axis was actually 8^o to the west of due north). The monitoring station was considered to be upwind of the Freeway when the wind direction was within $\pm 45^o$ of 352^o. Similarly, the monitoring station was considered to be downwind of the Freeway when the wind was within $\pm 45^o$ of 172^o. These wind sectors are also shown in **Figure 4.11**. Other roads in the area would have a negligible impact on concentrations at the monitoring site within thise wind sectors.

The results are shown in **Figure 4.12**. It is clear that the concentrations measured when then station is upwind of the road are similar to the concentrations measured at the background sites, especially for NO_X. Moreover, there is a marked road traffic contribution when the monitoring statiton is downwind of the road.

The net road traffic contribution – determined as the difference between the mean downwind and upwind concentrations - is given in **Table 4.2**. The traffic contribution to NO_X has decreased with time. The traffic contribution to NO_2 has increased and then stabilised. The value of *f*-NO_2 (calculated from the traffic increments for NO_X and NO_2) reflects these changes, increasing from 9% in 2008 to 14% in 2012, and then reducing slightly in 2013. This method for estimating *f*-NO_2 is, of course, quite simplistic, and ignores the complex chemistry of NO, NO_2 and ozone in the vicinity of roads. The resulting values - which are assumed to relate to tailpipe emissions - should therefore be considered as indictive and potentially overestimates given that some of the 'road increment' NO_2 may be secondary in nature.

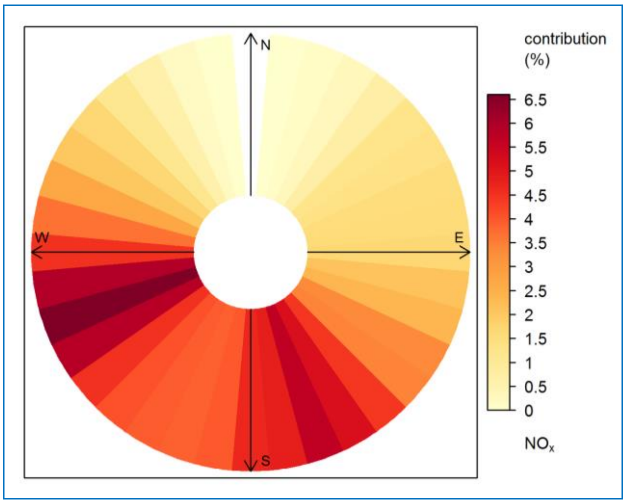

Figure 4.7: Polar frequency plot – NO_X (RMS F1)

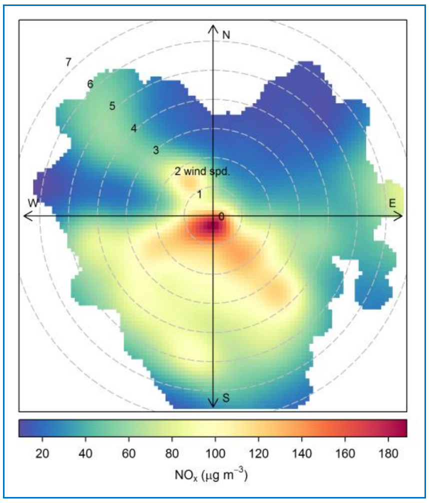

Figure 4.8: Bivariate polar plot – NO_X (RMS F1)

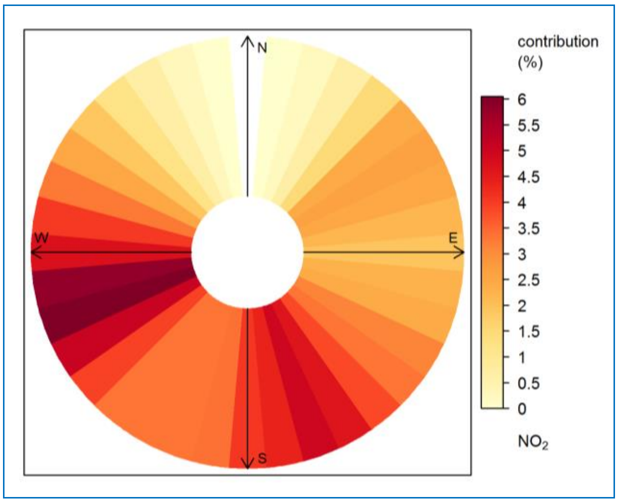

Figure 4.9: Polar frequency plot – NO_2 (RMS F1)

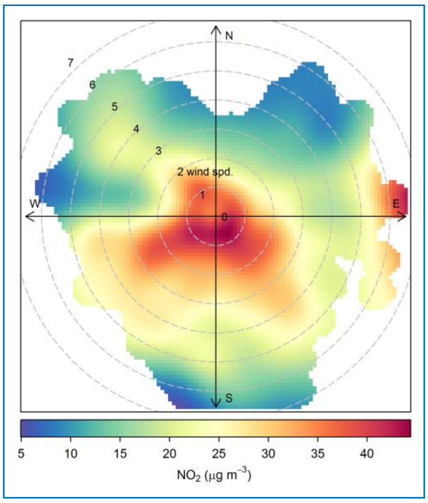

Figure 4.10: Bivariate polar plot – NO_2 (RMS F1)

Figure 4.11: Location of RMS F1 monitoring station and wind direction criteria

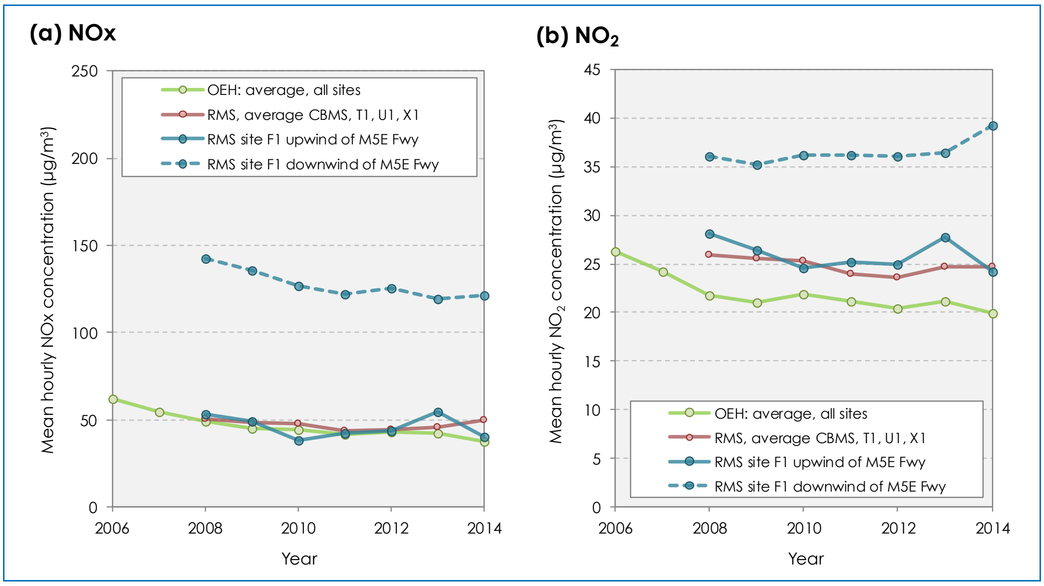

Figure 4.12: Mean concentrations at the RMS F1 monitoring station by wind direction

Table 4.2: Road contribution at RMS site F1

Year	Downwind-upwind concentration		
	NOx (µg/m³)	NO_2 (µg/m³)	*f*-NO_2 (%)
2008	89.3	7.9	9%
2009	86.7	8.7	10%
2010	88.9	11.5	13%
2011	79.9	10.9	14%
2012	81.6	11.1	14%
2013	65.1	8.6	13%
2014	81.5	15.0	18%

4.2.2 *Openair* 'calcFno2' function

The *Openair* software contains a simple constrained chemistry model (as described in **Carslaw and Beevers, 2005**) that can be used to calculate the NO_2/NO_x ratio in vehicle exhaust based on ambient air quality measurements. This is provided in *Openair*'s 'calcFno2' function. The method works by trying to distinguish between NO_2 that is directly emitted from vehicles and that derived through the reaction between NO and O_3. There are several assumptions and criteria that need to be considered when using this function. These include the following:

- The method is most reliable at estimating *f*-NO_2 when the roadside concentration is much greater than the background concentration.
- The function works best if the chosen background site is reasonably close to the roadside site and not greatly affected by local sources (which it should not be as a background site).
- As a simplifying assumption, the time available for chemical reactions to take place is set to 60 seconds.

The *Openair* calcFno2 function was applied to the data from the RMS F1 monitoring site. There is no background monitoring station in the immediate vicinity of the F1 site. In the *Openair* analysis, the data from the RMS F1 monitoring site were therefore used in conjunction with data from the nearest OEH urban background site, located at Earlwood, approximately three kilometres away. A closer background site would have been better suited to the analysis.

The results obtained are shown in **Figure 4.13**. There seems to be a pronounced seasonal effect in the results. This suggests that the method is struggling to distinguish between the NO_2 sources. In these situations both the NO + O_3 and the primary NO_2 tend to vary linearly with NO_X, making it difficult to partition between them. The calcFno2 function has been developed primarily for application in the UK. It is possible that the stronger photochemical activity in Sydney makes it more challenging to obtain meaningful results **(Carslaw, 2015b)**. Nevertheless, the general trend obtained using the function is broadly consistent with the results from the other data sets presented in this Report.

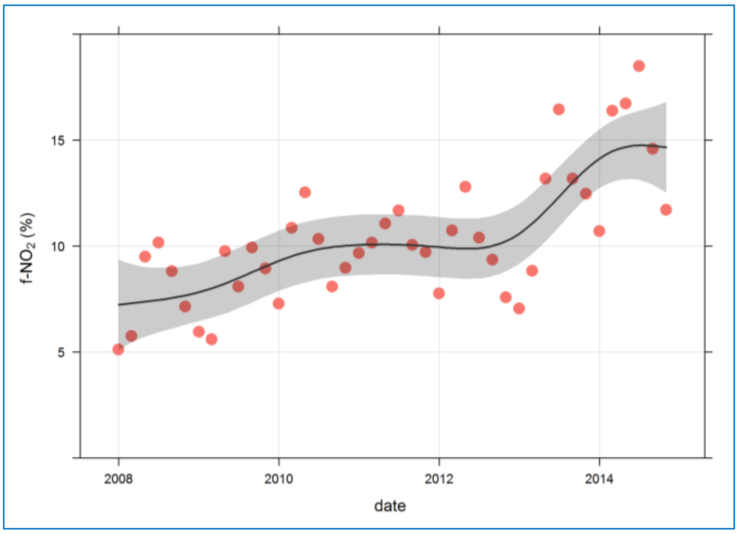

Figure 4.13: Trend in *f*-NO_2 at the RMS F1 monitoring site, as determined using the calcFno2 function in *Openair*

4.2.3 *Openair* 'linearRelation' function

Most of the ambient air quality monitoring at roadside locations in Sydney has not included the measurement of O_3. However, roadside O_3 measurements have been included in recent monitoring campaigns for the NorthConnex and WestConnex projects. In such cases the linearRelation function in *Openair* can be used to estimate *f*-NO_2. This function considers the relationships between two pollutants, based on linear regression fits to the data over different time periods. If the dataset contains NO_X, NO_2 and O_3, the y axis in the regression can be chosen as total oxidant 'OX' (NO_2 + O_3). At roadside sites the oxidant slope provides a good indication of the likely ratio of NO_2/NO_X in vehicle exhaust **(Carslaw, 2015a)**.

The following monitoring sites were included in this analysis:

- WestConnex M4 East, Concord Oval. This site was next to Parramatta Road. Data were available between November 2014 and January 2015.
- NorthConnex, Brickpit Park. This site was next to Pennant Hills Road. Data were available between December 2013 and January 2015.
- NorthConnex, Observatory Park. This site was next to Pennant Hills Road. Data were available between December 2013 and January 2015.

The results are shown in **Figure 4.14**, **Figure 4.15** and **Figure 4.16**. For the Concord Oval and Brickpit Park sites there again appears to be a seasonal effect on primary NO_2, with higher values in spring and winter. At Observatory Park, on the other hand, there is no strong seasonal influence. Further investigation would be required to determine whether these effects are real or artefacts of the method. There are also some differences in the absolute *f*-NO_2 values. At Concord Oval these vary between 8% and 15%, and at Observatory Park the value is around 10%. However, lower values (3% to 8%) were observed at Brickpit Park.

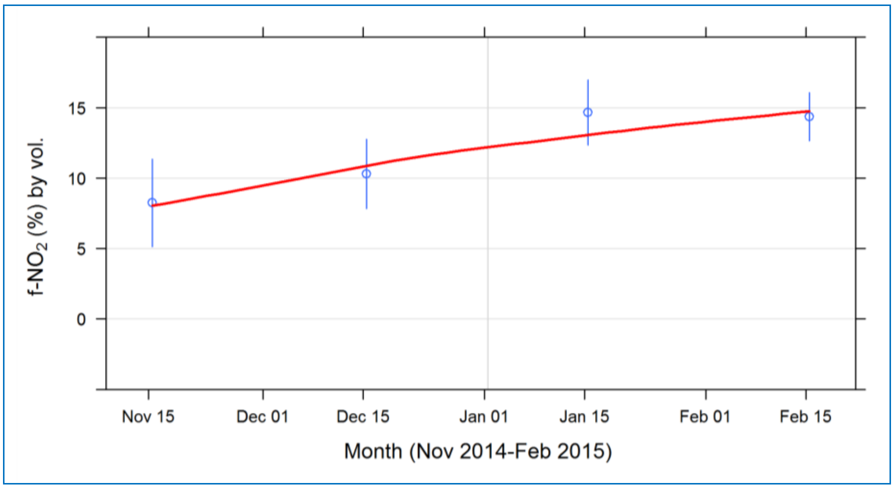

Figure 4.14: *f*-NO_2 at the WestConnex Concord Oval site between November 2014 and February 2015

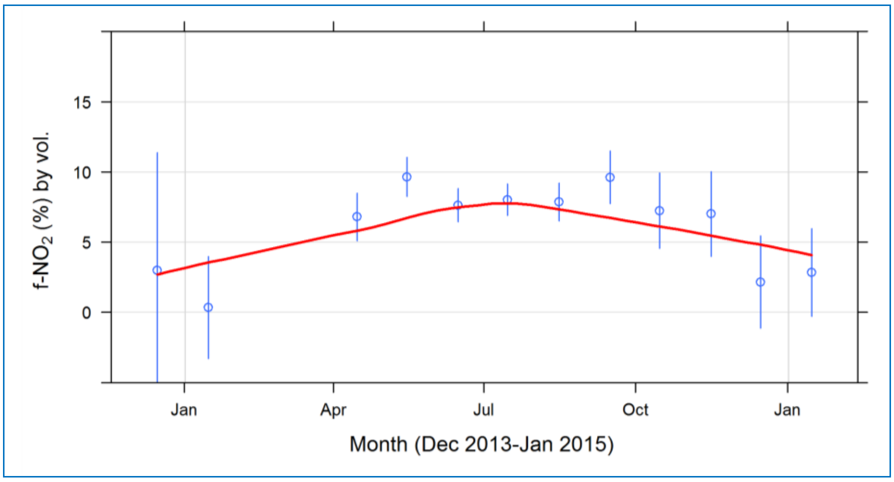

Figure 4.15: *f*-NO_2 at the NorthConnex Brickpit Park site between December 2013 and January 2015

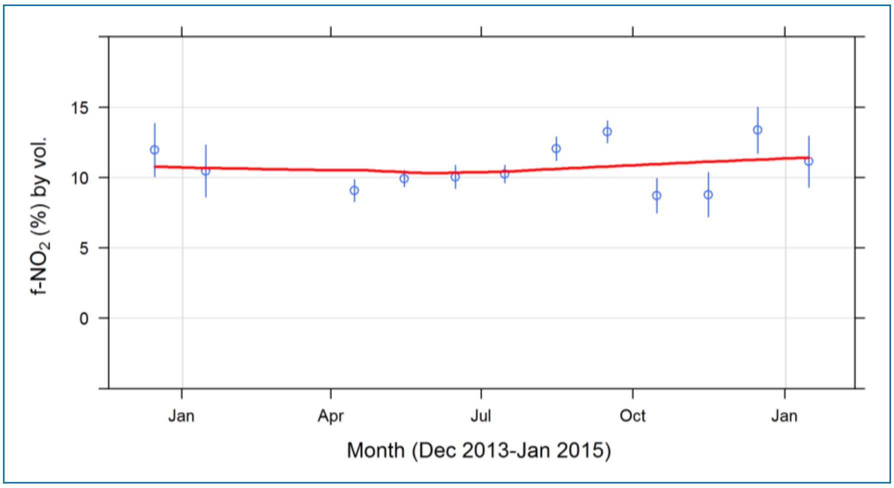

Figure 4.16: *f*-NO_2 at the NorthConnex Observatory Park site between December 2014 and January 2015

4.3 Analysis of data from Sydney tunnels

4.3.1 Lane Cove Tunnel

Vehicle emissions in the Lane Cove Tunnel (LCT) were recently investigated by **Boulter and Manansala (2014)**, and the original work has been updated and refined for this Report to provide further insight into primary NO_2 emissions.

The ventilation system in the LCT results in all vehicle emissions being released from the stacks. No pollution is released from the tunnel portals. This makes it relatively straightforward to estimate emission factors for the vehicles in the tunnel. The in-stack measurements at the LCT provided useful information on primary NO_2 emissions from the vehicles in the tunnel, but in this case the analysis was conducted using hourly data from October and November of 2013 (the only period for which data had been provided). A multiple linear regression (MLR) approach was used to determine mean emission factors for NO_X and NO_2 (in g/km) based on the LCT ventilation stack emission rates. The inputs to the MLR were the hourly mean emission factor for the traffic (dependent variable) and the corresponding numbers of LDVs and HDVs in the tunnel ach hour (independent variables). A similar MLR method has been used in previous studies to derive emission factors **(e.g. Imhof *et al*., 2005; Colberg *et al*., 2005)**.

The following regression model was applied to derive the emission factors:

$$EF_{total} = (N_{LDV} \times EF_{LDV}) + (N_{HDV} \times EF_{HDV}) + c \quad \text{Equation 2}$$

where:

EF_{total} = the hourly mean emission factor for all traffic in the tunnel, as determined from the in-stack measurements [g/km/h]

N_{LDV} = the number of LDVs in the tunnel per hour [vehicles/hour]

N_{HDV} = the number of HDVs in the tunnel per hour [vehicles/hour]

EF_{LDV} = the emission factor per LDV in the tunnel [g/vehicle.km]

EF_{HDV} = the emission factor per HDV in the tunnel [g/vehicle.km]

c = a constant (intercept on *y*-axis)

As the stack emission rates had already been adjusted to allow for the background contribution, and there were no other in-tunnel emission sources, it was considered acceptable to run the regression model with the constant constrained to zero.

The observed values of *f*-NO_2 calculated from the MLR analysis are given in **Table 4.3**. The average measured traffic-weighted values of *f*-NO_2 for all vehicles were 13% in the eastbound tunnel (in which the road is predominantly uphill) and 17% in the westbound tunnel (in which the road is predominantly downhill). Whilst the *f*-NO_2 values for each vehicle group was similar in both the eastbound and westbound tunnels, the much lower contribution from HDVs in the westbound tunnel meant that LDVs had a larger impact on the traffic-weighted *f*-NO_2 value.

Table 4.3: Observed emission factors for NO_X, NO_2 and primary NO_2 (October-November 2013)

Vehicle group	Eastbound (EB)			Westbound (WB)		
	NO_X (g/vkm)	NO_2 (g/vkm)	*f*-NO_2	NO_X (g/vkm)	NO_2 (g/vkm)	*f*-NO_2
LDV	0.29	0.06	**19%**	0.13	0.03	**19%**
HDV	8.42	0.37	**4%**	1.07	0.03	**3%**
All vehicles[(a)]	0.61	0.08	**13%**	0.18	0.03	**17%**

(a) Weighted for traffic volume.

4.3.2 M5 East Tunnel

Concentrations of NO and NO_2 have be measured in the ventilation stack of the M5 East tunnel for more than a decade. For the purpose of this study, hourly average in-stack measurements and the corresponding traffic data were examined for the period between 2004 and 2014. Unfortunately, the in-stack data supplied by RMS had not been validated, and were of very variable quality. Whilst an attempt was made to render the data useable, it was concluded that it would not be possible to determine robust emission factors from the measurements.

A sample of monitoring data from inside the M5 East Tunnel was also provided by CSIRO. The monitoring data was collected as part of a baseline study to evaluate the effectiveness of the introduction of smoky vehicle fines in the Tunnel **(Halliburton *et al.*, 2014)**. However, the dataset only covered a short period of time (three days of November 2014), and measurements were only conducted at a single location in the tunnel. The extent to which the data were representative of average traffic and concentrations in the tunnel was therefore quite uncertain, and therefore the data have not been presented in this Report.

5 SUMMARY AND CONCLUSIONS

This Report has summarised the issues associated with primary NO_2 emissions from road vehicles. Most of the evidence to date has been collected overseas. In previous overseas studies primary NO_2 has been quantified using laboratory measurements, remote sensing measurements, ambient air quality monitoring data and tunnel air quality data. A general finding of these studies has been that the NO_2/NO_X ratio in vehicle exhaust (f-NO_2) has increased due to increased market penetration of diesel vehicles and the emissions of NO_2 from technologies such as regenerative particle traps.

The Report has provided new evidence of values and trends in primary NO_2 for vehicles on the road in Sydney. Several different data sets and analytical techniques have been presented in the Report, including emission modelling, the analysis of ambient air quality measurements, and the analysis of emissions from tunnel ventilation stacks. The 'highway/freeway' road type is likely to be the most relevant to tunnels in Sydney, as it will most accurately reflects the corresponding traffic composition and driving conditions. In addition, more data are available for this type of road than other types. The values of f-NO_2 obtained for highway-type roads in the various analyses are presented in **Table 5.1**. The exceptions are the NorthConnex and WestConnex datasets, which relate to trunk roads rather than highways.

Table 5.1: Summary of f-NO_2 values (for highway traffic, unless stated otherwise)

Year	GMR inventory model	f-NO_2 by approach (values are traffic-weighted)			
		Ambient: downwind-upwind (F1)	Ambient: Openair (F1-Earlwood)	Ambient: Openair (WestConnex, NorthConnex)[b]	LCT ventilation stacks
2003	9%	-	-	-	-
2008	10%	9%	8%	-	-
2009	10%	10%	8%	-	-
2010	10%	13%	10%	-	-
2011	11%	14%	10%	-	-
2012	11%	14%	10%	-	-
2013	12%	13%	12%	-	13% (EB), 17% (WB)[a]
2014	12%	18%	15%	~5-10%	-
2020	14%	-	-	-	-
2025	15%	-	-	-	-
2030	14%	-	-	-	-

(a) Based on data for October and November of 2013 only.
(b) Not highways

Although a range of different data sets and methods have been used, the level of agreement in both the f-NO_2 values and the trend is good. The evidence suggests that there has been a gradual increase in f-NO_2 in recent years for highways, from less than 10% before 2008 to around 15% in 2014. The increase has not been as pronounced as that observed in the UK, and it appears that although the average value of f-NO_2 for highway traffic may increase further in the future, it will probably not increase much further and will eventually start to decrease. The recent NorthConnex and WestConnex datasets have shown lower f -NO_2 values (less than 10%) , but the monitoring stations are not near highways, and therefore traffic speed and composition may be atypical.

The main reason for the increase in f-NO_2 recent years is probably linked to the increased market penetration of diesel cars into the Sydney vehicle fleet. There is insufficient information on the types and distributions of exhaust after-treatment devices in use on vehicles in Sydney, and so it is not possible to determine the extent to which this is a contributing factor. Further information of this type would be beneficial to those studying the impacts of road transport on air quality in Sydney.

The findings of this work, whilst they should be viewed as indicative at this stage, could be used to support and refine calculations of primary NO_2 emissions from road traffic in Sydney, including calculations to support air quality impact assessments and studies of in-tunnel air quality. In particular, it appears from the measurements in Sydney that the approach of incorporating the European values for *f*-NO_2 in the NSW EPA inventory model has produced satisfactory results.

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Appendix A: NSW EPA analysis of vehicle registration data

An examination of vehicle registration (sales) data in NSW has recently been conducted by NSW EPA **(Jones, 2015)**. Whilst there are some differences[g] between the geographical coverage and the definition of vehicle groups, the EPA analysis has highlighted some discrepancies between the actual vehicle sales figures and the projections. These are shown in the Figures below.

Figure A1 shows the diesel proportions for cars and SUVs. NSW EP note the following:

- The % of new passenger cars (PC = cars and people-movers) that are diesel is over estimated in the 2008 inventory projections for 2014 at 19%, versus 8% 2014 RMS data 8% (blue lines)
- The % of new 4WD that are diesel is in good agreement (red lines);
- The proportion of passenger vehicles (PV = cars + people-movers + SUV) that are SUVs is underestimated in the inventory projections for 2014 at 29%, vs RMS data 37% (green lines).

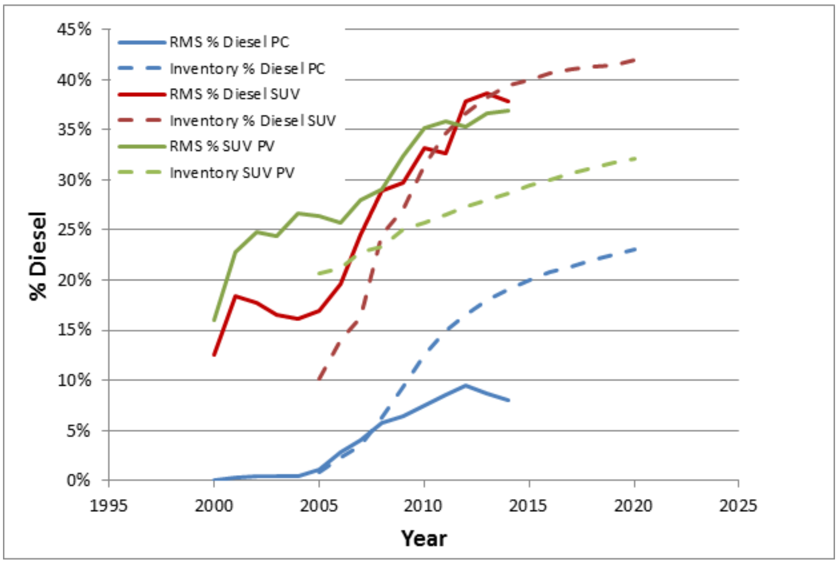

Figure A1: Diesel proportions for cars and SUVs

Figure A2 shows the total passenger vehicle fleet (cars + people-movers + SUV) and light commercial vehicles. NSW EPA note:

- The % of total new passenger vehicles (cars, people-movers and 4WD/SUV) that are diesel is overestimated in the 2008 inventory projections for 2014 at 25% vs RMS 19%;
- The % of new LCV that are diesel is underestimated in the 2008 inventory projections for 2014 at 61% vs RMS 89%.

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[g] The vehicle registration data to the end of 2014 were NSW-wide, whereas the inventory projection were for the GMR only. There may be some bias due to the different geographical areas (e.g. there may be more diesel vehicles and more SUV/4WD in regional areas). In addition, for LCVs the inventory defines these as GVM <=3500kg in line with the ADR's (i.e. primarily utes and vans), while the RMS rego data lists light trucks as those commercial vehicles with GVM <=4500 kg; thus there may be a bias again to diesel in the RMS as those vehicles 3.5t – 4.5t will be almost exclusively diesel.

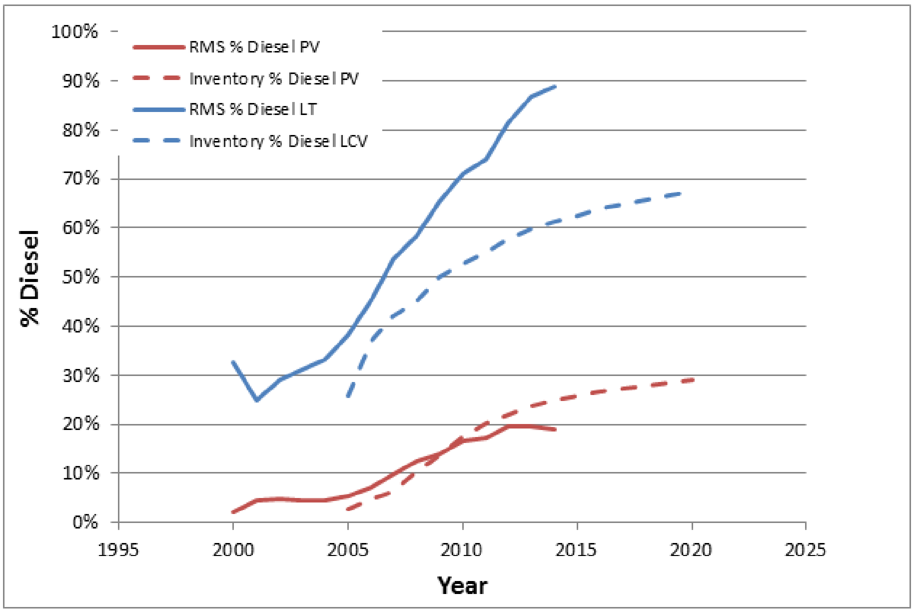

Figure A2: Diesel proportions for all passenger vehicles and light commercial vehicles