Guidance for the Management of Air Quality in Road Tunnels in New Zealand:
NIWA Research Report for NZTA

NIWA Client Report: AKL-2010-045
December 2010

NIWA Project: TRZ08104

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Executive Summary

This Report is designed to provide assistance to the New Zealand Transport Agency (NZTA), based upon scientific evidence, in setting guidelines for air quality in road tunnels. An accompanying report reviews actual air quality in the existing State Highway tunnels against the guidelines recommended in this report. It also discusses the practical implications of implementing such guidelines.

We briefly review the scientific evidence for the effects of exposure to road traffic emissions in the context of members of the public using typical road tunnels, and occupational exposure (e.g. maintenance staff). Most of the health evidence regarding exposure to traffic emissions is based on ambient exposure lasting hours, days or longer, and the significance of exposure lasting minutes or even seconds remains a major gap in scientific knowledge. The best understood pollutant for these timescales is carbon monoxide, although adverse effects arising from nitrogen dioxide and particulate matter are suspected and should be considered. Possible effects include aggravation of asthma, immediately or over subsequent hours, and risks of a cardiovascular event within a few days. Accrued effects from repeated tunnel use might include small increases in lifetime risk of cancer and potential for increased bronchitic events or respiratory infection.

Air quality in the outdoor environment around tunnel portals and stacks in New Zealand is subject to the National Environmental Standards for Air Quality. However, globally agreed or consistent guidelines for use **inside** road tunnels do not exist either in New Zealand or around the world. Air quality in road tunnels has conventionally fallen through a gap between the realms of ambient air quality and occupational exposure management in that a tunnel is neither fully indoors nor outdoors, that exposures usually last only a few minutes or less and protection is required for all potential users, including children, pregnant women and other vulnerable sub-groups, not just fit and healthy adults.

Conventionally, road tunnel air quality has been managed upon the basis of a maximum carbon monoxide exposure limit, usually expressed as the maximum concentration of the gas permitted within the tunnel averaged over 15 minutes. This approach has a long history and is proven to be relatively simple to implement.

We have reviewed the scientific basis of such an approach. As a consequence we recommend that a CO guideline equivalent to the World Health Organisation's 15-minute ambient air quality guideline 87 ppm be adopted by NZTA to provide protection from the acute effects of CO exposure for all tunnel users.

In the past a carbon monoxide guideline has been used to provide protection from the adverse effects on health of the full range of constituents of road traffic air emissions. Two recent trends have combined to threaten that protection. Firstly, emissions of carbon monoxide have fallen faster than emissions of other harmful pollutants, especially oxides of nitrogen from which the toxin nitrogen dioxide (NO_2) forms, so that there is relatively more NO_2 (and particulate matter) per CO in tunnel air than was previously the case. Secondly, health research has indicated toxic effects at lower exposures to NO_2 than previously thought, especially when inhaled alongside airborne particulate matter. Adverse respiratory effects, especially in asthmatics, have been observed in studies of exposures to levels likely to be found in some road tunnels. There is also evidence that road tunnel air exposure is a risk factor for adverse cardiovascular effects in vulnerable individuals. Consequently, a guideline based on CO alone can no longer be considered to automatically provide commensurate protection of tunnel-user's health.

We recommend that, in addition to carbon monoxide, any adopted guideline also considers the more demanding requirement to protect users from the combined effects of NO_2 and particles. However, the current state of health research prevents us from specifying an exposure limit for particles at this time. Recommendations are made for a nitrogen dioxide guideline, but this is based on more uncertain science than is the case for CO. We recommend that the guideline proposed by the World Road Congress (PIARC) of 1 ppm not to be exceeded more than 2 % of the time be adopted. More demanding NO_2 limits have been adopted in France and Hong Kong with the most demanding limits proposed in Sweden. These are based on a precautionary approach in view of evidence that asthmatics are more susceptible to NO_2. However, this is based on evidence based on exposures of 30 minutes or more and, unlike for carbon monoxide, the significance for much shorter duration exposures is currently unknown.

The New Zealand Workplace Exposure Standards (WES) apply to occupational exposure within tunnels. These Standards are less stringent than the NES or WHO ambient air quality guidelines as they are intended to apply to the typical healthy adult workforce, not the whole population. We find no basis to challenge the WES for short-term exposure to CO (200 ppm as a 15-minute average), which is widely adopted around the world for occupational exposure in road tunnels. An 8-hour time-weighted average limit of 30 ppm also has a long history and is widely adopted (e.g. in the UK, Germany and Sweden). We also find no basis to challenge this guideline and recommend it is adopted by NZTA. The World Road Congress (PIARC) has recommended a reduction in the 8-hour limit to 20 ppm, however the lack of justification provided by PIARC leaves us unable to determine whether the recommendation is based on medical evidence or other considerations, and we are therefore unable to endorse the recommendation on scientific grounds.

There is no internationally agreed short-term (15-minute average) occupational exposure limit for NO_2 in the same way as there is for 1-hour ambient exposures. Occupational NO_2 limits vary between countries and determining bodies, and have tended to change with time. On a precautionary basis, therefore we recommend that NZTA adopt the NIOSH (National Institute for Occupational Safety and Health) Recommended Exposure Limit (REL) of 1 ppm as a 15-minute average.

The 8-hour occupational CO guideline recommended provides more than adequate protection against benzene exposure, relative to the New Zealand 8-hour Workplace Exposure Standard which applies in all road tunnels. Lack of vehicle emission factors for formaldehyde, styrene and toluene prevent us from determining if the CO guidelines provide protection with regards to the WHO short-term exposure guidelines for these substances. However, the limited data available suggest that the emission reductions achieved for CO have generally been achieved also for other toxic species. Thus, we have no evidence to suggest that a CO guideline does *not* provide similar protection for these species.

Vehicle emissions are rapidly changing, but predicting current trends even a few years into the future is problematic. Health science is also making rapid advances and the effects of brief exposures and exposure to multiple pollutants in traffic exhaust are very active areas of research. For both these reasons we strongly recommend that any guidelines adopted are made subject to review at least every decade.

Road tunnel ventilation design demands a degree of future-proofing, hence the need to predict future emissions. A New Zealand-based modelling tool is available to predict the anticipated future changes in emissions. However, this capability is based on quite uncertain assumptions about the future vehicle fleet. More fundamentally it is not intended for predicting emissions on specific roads. We recommend that a modelling approach is augmented by a programme of long-term emissions monitoring in State Highway tunnels. This is achieved by supplementing in-tunnel air quality monitoring with airflow, traffic and ambient air quality monitoring. This will generate tunnel-specific data as well as feeding back valuable information to national emission tracking and modelling.

An exposure management approach (which applies any guideline to individuals using the tunnel taking into account the duration of their exposure, rather than applying solely to the tunnel air) may offer potential energy savings from reduced ventilation, the opportunities for which are greatest in shorter tunnels. Such an approach requires that a minimum speed can be assured. That speed is tunnel-dependent. Exposure management depends upon parameterisations which can be specified with acceptable confidence for CO, but their applicability to NO_2 guidelines is more uncertain.

Decreasing the rate of infiltration into a vehicle cabin can greatly reduce concentrations inside the vehicle. Setting air vents to ‘recirculate’ before entering the tunnel is a highly effective mitigation measure for individuals, but only if the vents are re-opened once the tunnel transit is complete. However, tunnel ventilation systems should be designed to the worst-case for exposure, which is low speeds and fully-open vehicles. Whether a vehicle cabin is open or sealed, the main determinant of the impact of carbon monoxide on health is the length of time spent in the tunnel.

How such guidelines are implemented is not obvious. This Report describes how pollutant concentrations vary within the tunnels volume and the consequences for monitoring. It also describes how nitrogen dioxide concentrations in particular, vary in a complex way that makes implementation of a NO_2 guideline more complex than for CO. This complexity can be addressed by well-designed observational campaigns which are highly recommended if a NO_2 guideline is to be implemented.

Report limitations

This Report does not recommend any guidelines for particulate matter. The current state of knowledge on the effects on human health of short-term exposure (order of minutes) to airborne particulate matter, and repeated exposure, as would be experienced in repeated passage through a road tunnel, is insufficiently established to propose a guideline at this time. We have found no evidence of such a guideline existing anywhere else in the world. This is also true for other toxic species present in vehicle exhaust, such as PAHs or benzene.

This Report does not explicitly cover visibility guidelines. This is because such guidelines are conventionally aimed at a safety objective. However, we do review some limited research into the potential use of visibility monitoring (which is common in many tunnels) to support compliance with air quality guidelines (for CO and NO_2). This Report also does not cover subjective responses to tunnel air, such as subjective responses to haze and visible smoke, or adverse reactions to odour.

Summary of NIWA's recommended guidelines for Occupational Safety for the protection of healthy adults working in a tunnel:

Contaminant	Threshold concentration	Averaging time	notes
CO	200 ppm	15 minutes	equivalent to NZ Workplace Standard
CO	30 ppm	8 hours	widely adopted abroad, PIARC 1995 recommendation
NO_2	1 ppm		equivalent to NIOSH Recommended Exposure Limit

Summary of NIWA's recommended guidelines for all non-occupational users:

Contaminant	Threshold concentration	Averaging time	notes
CO	87 ppm	15 minutes	equivalent to WHO ambient guideline, widely adopted in Australia
NO_2	1 ppm		PIARC proposal

Other road tunnel air quality guidelines adopted elsewhere (Occupational Safety guidelines in *bold italics*):

Contaminant	Threshold concentration	Averaging time	notes
CO	***20 ppm***	***8 hours***	***PIARC recommendation from 2010***
CO	***35 ppm***	***8 hours***	***US (NIOSH) Recommended Exposure Limit***
CO	100 ppm	15 minutes	PIARC
CO	70 ppm	15 minutes	PIARC from 2010
CO	100 ppm	5 minutes	Hong Kong
NO_2	1 ppm	5 mins	Hong Kong
NO_2	0.4 ppm	Unspecified	Norway
NO_2	0.2 ppm	1 hour	Sweden, Belgium
NO_2	0.11 ppm	1 hour	WHO ambient guidelines, NZ National Environmental Standard (ambient)
NO_2	0.5 ppm	20 mins	Belgium
NO_2	0.4 ppm	15 mins	France from 2010

Other Specific Recommendations:

- Due to the substantial remaining uncertainties regarding the health effects of brief exposure to NO_2, the emissions, determinants and levels of NO_2 in State Highway road tunnels, and technical challenges in monitoring NO_2 ,we recommend that an NO_2 guideline be implemented for the purposes of **design only** (rather than compliance monitoring) at this stage.

- We recommend that a research programme be conducted to determine the nature of NO_x emissions and resulting NO_2 concentrations and NO_2/NO_x ratios in New Zealand tunnels, and their determinants. This is required to facilitate the implementation of a NO_2 guideline, determine how demanding such a guideline would be, to better quantify the risks arising from road vehicle emissions, and specify mitigation options with greater certainty and confidence.

- Before such research can be conducted we recommend that a NO_2/NO_x concentration ratio of 0.1 be assumed in tunnels, rising to 0.2 within 200 m of the tunnel ends.

- We recommend that NZTA supports scientific research into the health impacts of brief exposures to vehicle emissions, and especially particulate matter.

- We recommend that any guidelines adopted are reviewed on a decadal basis in the light of new emission trend data, forecasts and new health research.

- We recommend that permanent internal monitoring be included as part of any tunnel design and that no system should rely on a single monitor.

- We highly recommended that a tunnel's ventilation system's performance is checked by monitoring at least once, and preferably on at least a decadal cycle.

- A pre-deployment study is recommended to assist in characterising the representativeness of any permanent monitor.

- We recommend the instigation of a long-term programme of vehicle emissions monitoring based in New Zealand's State Highway road tunnels to inform future demands on tunnel ventilation and feedback into tracking emission trends at a national scale.

Abbreviations

AER	air exchange rate
AQNES	air quality national environmental standards
CFK	Coburn-Forster-Kane model
CO	carbon monoxide
COHb	carboxyhaemoglobin
D_LCO	pulmonary diffusive capacity
HCV	heavy commercial vehicle
HDV	heavy duty vehicles
HGV	heavy goods vehicles
I/O	indoor/outdoor ratio
LDV	light duty vehicles
LGV	light goods vehicles
NIOSH	National Institute for Occupational Safety and Health
NMHC	non-methane hydrocarbons
NO	nitric oxide
NO_2	nitrogen dioxide
NO_x	oxides of nitrogen ($NO + NO_2$)
NZTER	New Zealand Traffic Emission Rates
PAHs	polycyclic aromatic hydrocarbons
PIARC	Permanent International Association of Road Congresses
PM	particulate matter
ppb	parts per billion
ppm	parts per million
RSD	remote sensing device
STEL	short-term exposure limit
TWA	time-weighted average
UFP	ultrafine particles (generally particles smaller than 0.1 μm)
ULSD	ultra low sulphur diesel

V_A	alveolar ventilation rate
V_B	blood volume
VEPM	Vehicle Emissions Prediction Model
VFEM	Vehicle Fleet Emission Model
VKT	vehicle-kilometres travelled
VOCs	volatile organic compounds
WHO	World Health Organisation

1. Introduction

1.1 Overview

- NZTA manage four major road tunnels: the Terrace and Mt Victoria tunnels in Wellington, the Lyttelton tunnel near Christchurch and the Homer tunnel which provides access to Milford Sound. A number of new tunnels are under construction or proposed.

- Air quality limits for use in tunnels can vary as a result of their objectives. These objectives can be characterised as a) maintaining safety, b) protecting health, and c) protecting wellbeing.

- In this context, maintaining safety means ensuring that the tunnel is safe to pass through. The objective is that tunnel users will not develop any adverse symptoms as a result of using the tunnel. This objective is conventionally met by limiting the exposure to carbon monoxide (CO).

- Safety is also maintained through the implementation of visibility limits, although this is intended to be independent of air quality impacts associated with visibility-reducing particles.

- Protection of health is a broader objective. This objective seeks to prevent adverse effects on people's health as a result of using a road tunnel. Meeting this objective requires a consideration of the effects of exposure to particles and nitrogen dioxide (NO_2).

- Protecting wellbeing encompasses the safety and health objectives, but additionally considers whether using a road tunnel is likely to be considered an unpleasant experience by the user. These outcomes are related to the perception of air quality which has strong links with odour, noise, the appearance of visibly smoky plumes and generally reduced visibility.

- Historically, road tunnel air quality has been regulated using limits on the allowable concentrations of carbon monoxide. Recent decreasing trends in emissions per vehicle, achieved through technological advances have translated into a decreased net ventilation demand per vehicle. Thus, it has become possible for ventilation energy demand to be reduced for some tunnels, whilst allowing more traffic for the same ventilation demand in other tunnels.

- Reduced mechanical ventilation will in most cases mean lower energy demand. The balance between protecting the health of tunnel users and minimising energy demand for ventilation is partly reflected in the choice of how stringent an in-tunnel concentration limit should be.

- Health-based limits are generally based on identifying the lowest-observed adverse effect level (LOAEL). However, not all persons have the same threshold of response to a given concentration of pollutant, and this implies that if these more vulnerable people are to be offered protection, more stringent limits are required.

1.2 Purpose of this Report

This report was initially commissioned by Transit New Zealand. Transit New Zealand amalgamated with Land Transport New Zealand in 2008 to form the New Zealand Transport Agency (NZTA). This report has now been finalised for the Highways and Network Operations group of NZTA.

NZTA is conducting a review and update of the management of their existing tunnels which will inform and set policy to cover new tunnels also. As part of the update, NZTA has commissioned NIWA to produce two reports with the aims of providing advice on the management of air quality both within road tunnels and the impact that tunnel emissions have on their immediate surroundings, covering both existing and new tunnels.

The reports are:

1. A review of the setting of air quality management guidelines and systems for road tunnels, and their implementation.

2. Scoping Assessments of the Mt Victoria, Terrace, Lyttelton and Homer tunnels, including scopes for detailed assessment.

This Report is the first of those two reports. It seeks to assist NZTA in the setting of Guideline values for air quality concentration limits for application to the interior of road tunnels operated by NZTA. It is intended to identify the key questions which must be answered in order to implement guidelines, and provides options from which solutions may be selected.

1.3 Scope of this Report

This Report consists of

- a discussion of the objectives and constraints of road tunnel air quality limits (this chapter),
- an overview of the composition of road vehicle exhaust (chapter 2),
- an overview of the principle known effects of emissions on human health in the context of road tunnel users (chapter 3),
- a review of the air quality standards, guidelines and targets currently in force in New Zealand in general, and applied in road tunnels around the world (chapter 4),
- a discussion of the options for limits for carbon monoxide (chapter 5) and nitrogen dioxide (chapter 6) and their basis,
- a review of the significance and implications of recent and future changes in vehicle emissions (chapter 7),
- a discussion of the implications of applying limits for monitoring (chapter 8),
- an introduction of an alternative approach to implementing guidelines – exposure management – and the consequences of adopting this approach (chapter 9)
- a discussion of the complications involved in implementing a nitrogen dioxide limit (chapter 10),
- a discussion of the use of monitoring for feedback and control and interactions with external air quality, energy demand and traffic management (chapter 11).

This Report does not explicitly review road tunnel visibility guidelines, as these are not conventionally considered to relate to air quality (and by implication respiratory or cardiovascular health), but to safety of tunnel users. However, visibility may act as a proxy for contaminated air and this is considered briefly in chapters 10 and 11. Furthermore, this report does not cover subjective responses to the tunnel environment, including odour.

1.4 Project Objectives

Table 1.1 below states the originally agreed project objectives and which sections of this report relate to each objective.

Table 1.1: Objectives of this report, and sections which relate to those objectives

Objective	Report sections
Review the air quality Standards, Guidelines, Targets and Limit Values which are relevant to road tunnels, covering ambient air, occupational exposure and those commonly adopted for road tunnel management around the globe, providing detail on the basis for each (e.g. health or otherwise) and any established uncertainties.	Chapters 4, 5 and 6
Review how different approaches are implemented and enforced and highlight known shortcomings and benefits of each, including conflicts and co-benefits with other considerations (including energy use)	Chapters 8, 9, 10, 11
Provide guidance on the development of approaches to manage air quality in new and updated tunnels	Chapters 8, 9, 10, 11
Reporting will be formatted so as to provide scientific advice to support decision-making by NZTA. It will offer a range of alternative strategies ranging from the simple but effective, to good practice and best practice. It may make scientific recommendations but will not make policy recommendations.	Whole report
The Study will cover…	
Occupational exposure of maintenance staff inside the tunnels	4.2, 4.4, 4.6, 5.9, 6.4
Consideration of protection of pedestrian tunnel users from poor air quality	9.3, 9.6, 12.1
Effects of ventilation stack/portal discharges on local ambient air quality.	Removed from scope
Consideration of protection of vehicle occupants from poor air quality.	9.6
An indication of the implications of air quality management for traffic management and vice versa.	9.3, 11.5
Emerging best practices at the forefront of technology	7.7, 7.8, 9, 10, 11

1.5 Context – road tunnels in New Zealand

NZTA manage four major road tunnels: the Terrace and Mt Victoria tunnels in Wellington, the Lyttelton tunnel near Christchurch and the Homer tunnel which provides access to Milford Sound. A number of new tunnels are under construction or proposed (see Table 1.2). The Waterview tunnel and possible Waitemata Harbour tunnel will be unprecedented in New Zealand in their length, complexity and potential environmental/public health impact. Due to the range of lengths of these tunnels, data from much longer tunnels in the world (10 km+) are not considered in this Report.

Table 1.2: Current and future road tunnels on State Highways in New Zealand

	Date of Opening	Length / m
Mt Victoria	1931	623
Homer	1953	1270
Lyttelton	1964	1945
Terrace	1978	460
Johnstone’s Hill	Estimated 2009	380
Victoria Park	From 2014	440
Waterview	From 2015	~ 3km
Waitemata Harbour	Feasibility Study	Up to 9 km

1.6 Objectives for AQ limits

Air quality limits for use in tunnels can vary as a result of their objectives. These objectives can be characterised as

a) maintaining safety

b) protecting health, and

c) protecting wellbeing.

In this context, maintaining safety means ensuring that the tunnel is safe to pass through. The objective is that tunnel users will not develop any adverse symptoms as a result of using the tunnel. Such an objective relates to the potential physiological reactions of the body to exposure to the air pollutants which may be present in the tunnel as the result of a single passage through the tunnel in its normal use. This objective is conventionally met by limiting the exposure to carbon monoxide (CO). This is mainly because the body reacts very rapidly to CO exposure in a way that is relatively well understood and quantifiable. Safety is also maintained through the implementation of visibility limits, although this is intended to be independent of air quality impacts associated with visibility-reducing particles.

Protection of health is a broader objective. This objective seeks to prevent adverse effects on people’s health as a result of using a road tunnel. The distinction with safety arises due to the possibility of health effects which do not produce identifiable symptoms or are delayed such that the effect cannot easily be related back to the tunnel. This objective also seeks to consider the effect not just of a single passage through a tunnel, but repeated tunnel use over an extended period of time. Meeting this objective requires a consideration of the effects of exposure to particles and nitrogen dioxide (NO_2). In principle, it also includes consideration of development of cancers as a result of exposure to carcinogenic compounds within vehicle exhaust; however, in practice, it is not practical to manage the contribution of road tunnel exposure in isolation to vehicle exhaust exposure in general.

Protecting wellbeing encompasses the safety and health objectives, but additionally considers whether using a road tunnel is likely to be considered an unpleasant experience by the user. Thus, adoption of such an objective could require a consideration of anxiety and stress. These outcomes are related to the perception of air quality which has strong links with odour and noise. There is no established means of quantifying the effect of these factors on feelings of anxiety in road tunnels and **they are therefore not covered in this Report**. However, perceptions of air quality are related to reduced visibility arising from haze and dust, and the appearance of visibly smoky plumes, and thus visibility limits can be used to also meet a ‘wellbeing’ objective.

1.7 Energy and sustainability issues

Historically, road tunnel air quality has been regulated using limits on the allowable concentrations of carbon monoxide. Recent decreasing trends in emissions per vehicle, achieved through technological advances (described in chapter 7) have translated into a decreased net ventilation demand per vehicle. Thus, it has become possible for ventilation energy demand to be reduced for some tunnels, whilst allowing more traffic for the same ventilation demand in other tunnels. Maximum tunnel length is constrained by ventilation and the maximum allowable airflow consistent with safety. Emission reduction trends have eased those constraints allowing longer tunnels for a given ventilation capacity.

Reduced mechanical ventilation will in most cases mean lower energy demand. Where this energy is sourced from non-renewable means the reduced demand translates to reduced greenhouse gas emissions. This will be associated with reduced emissions of toxic air pollutants also. These positive outcomes from minimising ventilation demand act as a counterbalance to the desire to provide maximum ventilation to provide the best possible air quality for tunnel users. A solution which balances these needs should always be sought.

Despite the potential for adverse health effects arising from use of road tunnels as well as effects on those living in the vicinity of portals and associated stacks, road tunnels offer the potential for a general improvement in air quality, especially in urban areas. Road tunnels remove traffic underground leading to potentially reduced emissions and concentrations at ground-level. Local roads can be relived of traffic, or congestion can be reduced. Tunnels have the potential to remove emissions from sensitive residential locations to less harmful locations. However, this always needs to be balanced against potential adverse effects caused by redistribution of traffic and congestion and induced traffic. On the whole a tunnel is unlikely to be unequivocally ‘good’ or ‘bad’ for air quality – the most likely result is an improvement for some and worsening for others. One of the over-riding objectives in road tunnel design should be to maximise the positive impacts and minimise the negative.

1.8 How demanding should air quality exposure limits be?

The balance between protecting the health of tunnel users and minimising energy demand for ventilation is partly reflected in the choice of how stringent an in-tunnel concentration limit should be. From a public health point of view we wish to reduce exposure as much as is possible, and apply the most stringent appropriate limits. Our bodies can generally tolerate a certain level of most contaminants, and health-based limits are generally based on identifying the lowest-observed adverse effect level (LOAEL). However, not all persons have the same threshold of response to a given concentration of pollutant, and this implies that if these more vulnerable people are to be offered protection, more stringent limits are required. Furthermore, adoption of the ‘protecting health’ objective (see above) is more demanding than the ‘maintaining safety’ objective.

More stringent limits, however, can require higher rates of ventilation. It must be remembered that ventilation does not destroy pollutants – it merely changes the rate at which they are displaced from the internal to the external atmosphere. Increased ventilation reduces internal and external concentrations, but can increase the spatial extent of the external impact, especially if vented via a stack.

2. Road tunnel air pollutants

2.1 Overview

- Emissions of air pollutants into road tunnel air are principally exhausts from vehicle tailpipes, but also include vehicle wear products (such as dust from brakes and tyres) and resuspended dusts.

- The major polluting tailpipe emissions are the gases carbon monoxide (CO) and nitrogen dioxide (NO_2, which largely forms indirectly from emissions on nitric oxide, NO), particles consisting mostly of elemental carbon (soot) and a wide range of complex organic compounds; and a wide range of toxic, irritant and carcinogenic compounds.

- Emission of lead compounds from the tailpipe has ceased to be a concern.

- Recent technological advances have led to a general reduction in the amount of CO, elemental carbon, and toxic organic compounds being emitted from the average vehicle.

- However, these emission reductions have not been matched by comparable cuts in emissions of oxides of nitrogen ($NO+NO_2=NO_x$).

- Concentrations of pollutants in urban tunnels > 1 km long will exceed those outside the tunnel by perhaps 10 – 100 times for pollutants with few non-traffic sources, such as carbon monoxide. For multi-source pollutants, such as PM_{10}, the ratio may be typically 1 – 10 times. Because of these large ratios internal concentrations are typically not significantly influenced by external conditions, except in very short tunnels. A possible exception is the potential for ambient ozone concentrations to influence in-tunnel levels of nitrogen dioxide, although this relationship is poorly understood.

2.2 Vehicle emissions in tunnels - overview

From an air quality point of view a road tunnel can be viewed as a chamber in which the emissions from a section of road, which would normally be dispersed into the atmosphere along the whole length of the road, are first collected rather than being instantly dispersed before being released at one or a few points. Compared to a surface road the air quality as experienced by road users is relatively poor whereas the impact on local residents is redistributed so that contaminated air is more concentrated near the points where tunnel air is released into the general atmosphere.

Air pollutants emitted from road vehicles are generally rapidly dispersed from the road by the wind and diluted by mixing with fresh air by the process of turbulence. The interior of a road tunnel is generally sheltered from the wind and although there will still be turbulence there is only a limited supply of fresh air to dilute the polluted air. In a given timeframe a certain mass of pollutants will be emitted into the tunnel air which depends upon

1. the number of vehicles in the tunnel, and

2. the emissions per vehicle (or emission characteristics of the fleet).

The emissions per vehicle are highly variable and will depend upon vehicle age, speed, size, fuel type, engine specifications, plus other factors even harder to quantify such as state of maintenance, engine temperature and driving style. Further influences may include correct tyre pressure and gradients on the road. Not only are emissions highly variable within a given vehicle fleet, but recent technological advances have meant that they are rapidly evolving.

Nevertheless average fleet emission characteristics can be, and are routinely, estimated. Broadly speaking emissions will be higher for an older vehicle fleet, a higher proportion of heavy duty vehicles (HDVs), uphill climbing and in congested conditions.

2.3 The composition of road vehicle emissions

The composition of the atmosphere inside a road tunnel is enhanced by road traffic emissions such that concentrations of pollutant trace gases and particles can be orders of magnitudes greater than what is typically present in the ambient atmosphere. Emissions are principally exhausts from the tailpipe, but also include vehicle wear products (such as dust from brakes and tyres). Vehicles also resuspend dusts from surfaces in the tunnel by both direct frictional contact of the tyres with the road and indirectly through the action of turbulent gusts generated by moving vehicles.

Chemically, these emissions can be characterised as having a range of reactivities. Within the timescale that these substances remain within the tunnel atmosphere, many vehicle emissions are effectively chemically inert. Carbon monoxide is a key example whose chemical atmospheric lifetime is on the order of months, far longer than the few minutes that air remains inside a tunnel. Other directly emitted species are more reactive so that some proportion of these emissions will be transformed into other substances before leaving the tunnel. This is the case for nitric oxide which will be rapidly converted into nitrogen dioxide if an oxidant such as ozone is present.

Physically, emissions can usually be characterised as either gases or particles. However, tailpipe emissions defy usual classifications. Vehicle exhausts are between 50 °C and 100 °C hotter than the ambient air and therefore, the plume cools very rapidly, providing conditions that favour gas-to-particle processes such as nucleation (new particle formation) and condensation onto pre-existing particles. Fresh vehicle emissions contain particles that span a range of sizes[1], but its population is typically dominated, as number concentration, by particles in the size range 10 - 100 nm, which is comparable to the size of small viruses. Particles in this range are known as ultrafine particles (or UFP). UFP are emitted directly, or form through nucleation of rapidly cooling volatile and semi-volatile compounds in the expanding exhaust plume. UFP will then commonly grow by condensation of further semi-volatile material and by the coagulation brought about by collision. Shortly after leaving the tailpipe (order of a few seconds) the rate of the processes of condensation and coagulation are greatly reduced by plume dilution. Whereas nearly all the gaseous material emitted from vehicles will be removed from the tunnel via the portals, the ventilation system (possibly including stacks) and inside vehicle cabins, some particles will be deposited to the tunnel surfaces.

2.4 The origin of road vehicle emissions

Tailpipe emissions arise because of the process of combustion. Ideally this process oxidises hydrocarbons, which are the core constituent of fossil fuels, to carbon dioxide and water. Two factors guarantee that there will be extra products: the presence of other compounds in the combustion chamber (fuel additives and nitrogen from air among many others) and the inherent inefficiency of combustion in even the best performing engines giving rise to incomplete combustion. The major products of incomplete combustion are the gas carbon monoxide, elemental carbon (soot) and a wide range of complex organic compounds, such as polycyclic aromatic hydrocarbons (PAHs) which are emitted in gas and particle phases, and a wide range of toxic, irritant and carcinogenic compounds. A few of these substances have occupational exposure limits associated with them, and fewer still have ambient air quality guidelines. For example, the WHO ambient air quality guidelines cover benzene, 1,3-butadiene and

[1] 'Size' is used here in a descriptive sense as particles vary in shape. In air quality management the size is usually taken to mean aerodynamic diameter, although other formulations of 'size' can be used (Hinds, 1999).

PAHs (represented by one PAH compound – benzo(a)pyrene) for their carcinogenic effects and formaldehyde, toluene and styrene for their annoyance or sensory effects. Tailpipe emissions also include particles of unburnt fuel and vehicles also emit droplets of lubricating oil, partly through the tailpipe (Fraser *et al*., 1998, Kirchtetter *et al.*, 1999). Relatively small amounts of toxic organics are derived from evaporation of fuel (McLaren *et al.*, 1996). PAHs and some other organic compounds are also emitted from road bitumen.

Removal of lead from petrol has meant that emission of lead compounds from the tailpipe has ceased to be a concern. More recently, sulphur levels in petrol and diesel fuel have been progressively reduced. Fuel sulphur leads to both the emission of toxic sulphur dioxide gas, but also sulphuric acid particles which promote the formation of ultrafine particles. The effective operation of particulate traps (which are generally required for diesel vehicles to meet the Euro IV emission standard) requires sulphur in fuel to be at 10 ppm or less (generally termed Ultra-Low Sulphur Diesel). Diesel vehicles operate at a higher temperature which should, in principle, lead to more complete combustion. However, this is compromised by the greater compositional complexity of diesel fuel, such that diesel exhaust is also generally more complex with typically a higher load of soot and PAHs than petrol exhaust (e.g. Allen *et al*., 2001, Laschober *et al.*, 2004, Phuleria *et al.*, 2006).

As well as improving fuels, recent technological advances have focussed on improving combustion so as to reduce harmful emissions. As will be demonstrated in more detail below, this has led to a general reduction in the amount of CO, elemental carbon, PAHs and other toxic organic compounds being emitted from the average vehicle. One consequence is that non-fuel related emissions, including lubricating oil, are becoming relatively more significant.

However, the CO, PAHs and other organic compounds emission reductions have not been matched by cuts in emissions of oxides of nitrogen ($NO+NO_2=NO_x$). Nitrogen is not normally present in fossil fuels. Nitrogen and oxygen are highly stable molecules that, at ambient temperatures, co-exist without reacting and together form ~ 99% of the atmosphere. However, internal combustion engines operate at temperatures higher than 800C and at those temperatures nitrogen and oxygen do combine. Thus, vehicles emit the products of this combination in two forms: mostly nitric oxide (NO) with some in the form of nitrogen dioxide (NO_2). Although both potentially toxic, NO_2 is much more potently so and is the form for which ambient air quality guidelines exist, including a National Environmental Standard in New Zealand. Both forms are also quite reactive, such that most emitted NO is converted into NO_2 (the chemistry of NO_x, and its relevance to road tunnels is discussed in some detail in section 9.1). As the reaction between nitrogen and oxygen is a function of combustion temperature, regardless of fuel content, efforts to increase combustion efficiency by increasing combustion temperature will tend to increase NO_x emissions. This conundrum has led to NO_x emissions and NO_2 exposure becoming increasingly important issues in air quality management.

Road dusts are generally (but not entirely) made up of particles much larger than those emitted from the tailpipe. These particles are predominantly of a size larger than 1 μm. They are described as coarse or super-micron particles. Road dust also contains particles larger than 10 μm but these are generally considered less significant for air quality purposes as their atmospheric residence time is limited (their weight brings them to the surface relatively quickly) and they generally do not penetrate far in the human respiratory system. The toxicity of road dust is the subject of some debate. Although road dusts contain organic compounds (including cellular, plant and soil matter, and pavement material), they also contain a much higher proportion of heavy metals (derived from brake pads, tyres and other materials used in the construction of vehicles) and minerals (local dusts, pavement material and salt).

2.5 Road tunnel atmospheric composition compared to ambient air

Road tunnel air is naturally enriched in those substances emitted by road vehicles, compared to ambient air. The degree of this enrichment depends upon the length of the tunnel, the number of vehicles and the strength of their emissions, and the rate of ventilation. However, as a first approximation, concentrations of pollutants in urban tunnels > 1 km long will exceed those outside the tunnel by perhaps 10 – 100 times for pollutants with few non-traffic sources, such as carbon monoxide. For pollutants with multiple sources, such as PM_{10} (which has major industrial, domestic and natural sources) the ratio may be typically 1 – 10 times. Because of these large ratios internal concentrations are typically not significantly influenced by external concentrations, except in very short tunnels.

In a few subtle ways the atmosphere in a road tunnel may be different to the ambient air. Firstly oxidants, such as ozone and free radicals may have much lower concentrations due to reduced sources (low light) and increased sinks (depositional surfaces), although there is only limited evidence to support this. The chemicals control the rate of several relevant chemical reactions, especially those controlling concentrations of NO_2. Also the constant flow of vehicles and the constrained nature of a closed tube may enhance turbulence that resuspends road dust such that dust levels are generally higher than on the open road, but again there are only very limited observational data to confirm this.

3. General health effects of road traffic air pollutants

3.1 Overview

- Our scientific knowledge regarding the effects of road vehicle emissions on human health is far from complete. It is a rapidly growing and evolving field of research.

- Health effects can be classified as ‘acute’, ‘delayed acute’ and ‘chronic’.

- The mechanisms of carbon monoxide uptake in the body and some of the resulting effects, at least the short-term effects, are relatively well understood and this sound knowledge base provides the foundation for the extensive use of carbon monoxide limits in road tunnels.

- In the context of road tunnel users, an acute effect is one that we might expect to observe after a single passage through a tunnel. If the effect is manifest within minutes of exposure than it is likely that the effect can be causally ascribed to the tunnel exposure.

- Uptake of carbon monoxide by blood haemoglobin is rapid leading with little delay to hypoxia, with neurological effects being those most likely to be experienced or noticed by normal healthy subjects.

- The heart is exceedingly sensitive to hypoxia. Studies have shown that chronic angina patients will suffer symptoms sooner if exposed to CO during exercise.

- There are very limited data on the immediate acute response to short (minutes) exposure to particles.

- ‘Delayed acute’ effects refers to effects are not apparent until some hours post-exposure.

- Little is currently known about the short-term effects of particle exposure, especially exposures of a few minutes duration. Some research has indicated possible ‘delayed acute’ effects (hours after short exposures), however these studies do not yet amount to a sufficient understanding to either inform short-term exposure limits for particles, nor an evaluation of the risk presented by the kind of short occupational exposures or brief exposures experienced by road tunnel users.

- “Chronic effects” arise from repeated long-term exposure, typically of the order of years. In the road tunnels context chronic effects may arise from regularly repeated passage through a tunnel, or frequent occupational exposure.

- Serious chronic health effects have been related to city-wide poor air quality. For road tunnels this means that there is the potential for effects on the health of regular tunnel users that may not become manifest as acute effects, but will arise as the development of long-term illness. The problem for air quality management is that it will not be possible to trace the cause of such an illness to the tunnel, and the tunnel alone is unlikely to be the sole cause of the illness. The size or probability of such an effect is also very difficult to quantify or predict.

- The same problem arises in the case of the development of cancers as a result of exposure to the carcinogens in vehicle exhaust. Exposure to these compounds will arise from *all* exposure to traffic emissions and thus the impact of the time spent in a tunnel cannot be separated.

- In summary, potential health risks arising from exposure to in-tunnel air include aggravation of asthma, immediate or over subsequent hours, and exacerbation of cardiovascular disease within hours to days. Accrued effects from repeated tunnel use might include small increases in lifetime risk of cancer and potential for increased bronchitic events or respiratory infection.

3.2 Vehicle emissions as a toxic mixture

Our scientific knowledge regarding the effects of road vehicle emissions on human health is extensive, yet far from complete. It is a rapidly growing and evolving field of research. As discussed in chapter 2 the composition of vehicle exhaust is highly complex, not just in terms of chemically species, but also in terms of physical characteristics. Exhaust contains solids, liquids and gases of varying solubilities existing in internal and external mixtures. Some components are known carcinogens, others established irritants. The toxicity of other components is suspected but not proven. One of the greatest knowledge gaps and pressing questions is the effect of this combined exposure to multiple pollutants simultaneously. Do the components of vehicle exhaust act separately and independently, or do some exacerbate the effects of the others? Is the whole greater than the sum of the parts?

There is some experimental toxicological data that suggests that these synergistic effects do occur (e.g. Strand *et al*., 1998). In most of the rest of this report we shall be considering one pollutant at a time. However, that is a result of the bulk of our knowledge at this time being restricted to the effects of single pollutants. However, it should be borne in mind that these single pollutants will be acting as proxies for a large range of co-pollutants and their combined effects.

Experimental toxicological studies could, in principle, tease apart the independent and synergistic effects of the individual components of vehicle exhaust. Such studies, however, if they were truly comprehensive, would be major undertakings and would be necessarily limited by ethical considerations. Epidemiological studies of effects of traffic in the community are also necessarily limited to consider exposure to vehicle exhaust as it exists in all its complexity. As it is not practical to measure every component individually these studies rely on measurements of one or a few pollutants (usually NO_2, PM_{10} or $PM_{2.5}$, and occasionally CO or one or more organics species) to represent all traffic emissions. These studies have been crucially valuable in advancing our understanding of the impact of air pollution, but they are only able to draw limited indicative conclusions and are unable to unlock the toxicological mechanisms that explain observed links between traffic emissions and illness.

The most pertinent example of this is the conclusions one may draw from epidemiological studies based on NO_2 observations. A number of studies have shown an association between NO_2 concentrations in residential areas, either measured or modelled, and a number of adverse health impacts, such as reduced lung development in children (e.g. Gauderman *et al*., 2007). However, it is repeatedly noted that although these studies could indicate a direct causal link between NO_2 exposure and the observed effects, it is more likely that NO_2 is acting as a proxy for either another component of vehicle exhaust, or the combined effect of the components of vehicle exhaust. Thus, in setting the long-term NO_2 guideline (annual mean of 40 $\mu g\ m^{-3}$), the WHO made it very clear that …

> "Numerous epidemiological studies have used nitrogen dioxide as a marker for the air pollution mixture of combustion-related pollutants, in particular traffic exhaust or indoor combustion sources. In these studies, the observed health effects might also have been associated with other combustion products, such as ultrafine particles, nitric oxide, PM or benzene."

> "If … nitrogen dioxide is monitored as a marker for the concentrations and risks of the complex combustion-generated pollution mixtures, an annual guideline value lower than 40 $\mu g\ m^{-3}$ should be used instead." (WHO, 2006)

For example, some studies have identified that NO_2 sensitises those with allergic asthma to the effects of allergens, including particles (e.g. Svartengren *et al*., 2000). However, the degree to which the size, composition and physico-chemical nature of the particles is significant for this combined effect is yet unclear. Svartengren *et al*. (2000) suggested that an increased particle concentration may reduce the NO_2 concentration required to elicit the same order of response, indicating that NO_2 and PM are inter-dependent and setting guidelines for each cannot be considered in isolation. However, we have insufficient data at present to describe this inter-dependence adequately and must rely on independent guidelines for the time being.

The issue is further complicated by the reactivity of nitrogen oxides and their different emission sources. CO, black carbon, volatile organic compounds, PAHs and other particulate components have a common source (incomplete combustion) such that emissions per vehicle of each have broadly fallen in unison (see chapter 7), and we would expect them to dilute at the same rate in a tunnel such that their concentrations would exist in consistent ratios. This is not the case for NO_2, such that the ratio of NO_2 concentrations to particles, or any other pollutant measure, is likely to be variable within a tunnel, between tunnels, and with time on all scales (see chapters 7 and 9).

However, despite these limitations some of the health impacts of vehicle emissions are relatively well-established. The mechanisms of carbon monoxide uptake in the body and the resulting effects, at least the short-term effects, are extensively understood and this sound knowledge base provides the foundation for the extensive use of carbon monoxide limits in road tunnels.

3.3 Immediate acute effects

In the context of road tunnel users, an acute effect is one that we might expect to observe after a single passage through a tunnel. If the effect is manifest within minutes of exposure than it is likely that the effect can be causally ascribed to the tunnel exposure.

Uptake of carbon monoxide by blood haemoglobin is rapid leading with little delay to hypoxia, with neurological effects being those most likely to be experienced or noticed by normal healthy subjects. Carboxyhaemoglobin (COHb) levels of 20 – 40 % are associated with vividly experienced neurological effects such as headaches, dizziness, weakness, nausea, confusion, disorientation and decrements in fine motor control. At levels approaching 50 % collapse and coma may occur and levels above 50 % can lead to unconsciousness, cardiac arrest and death. The body purges itself of CO naturally over a time scale of hours (see chapter 5). It has generally been thought that the adverse effects will cease once COHb levels are back to normal. However, some recent evidence has suggested the possibility of some persistent symptoms remaining (including headache, lethargy and lack of concentration 2 – 40 days after exposure Townsend & Maynard, 2002).

The heart is exceedingly sensitive to hypoxia. As part of the natural defence against cellular hypoxia a compensatory increase in coronary blood flow may occur with associated tachycardia. If blood flow cannot meet oxygen demand, angina or other cardiovascular symptoms may be experienced. Studies have shown that chronic angina patients undertaking exercise whilst exposed to CO will experience debilitating chest pain much sooner than without CO exposure, or compared to healthy subjects (Allred *et al*., 1989). This and other studies have not identified whether CO exposure alone would cause chest pain (i.e. without exercise), but this cannot be ruled out in a more susceptible subjects than those used in these experimental studies.

The potential exists for the appearance of immediate symptoms in asthmatics as a result of NO_2 exposure. Experimental studies have indicated effects at concentrations over 200 $\mu g\ m^{-3}$, but for exposure times of 1 hour or more, leading to the WHO settling on this level for a 1-hour exposure guideline. The effects of NO_2 exposure at shorter timescales are essentially unknown, and a methodology for translating the 1 hour limit to shorter exposures is not yet established.

There are very limited data on the immediate response to short (minutes) exposure to particles. Nemmar *et al*. (2002) found that a radio-labeled ultrafine aerosol inhaled by 5 healthy volunteers passed rapidly into the blood stream, as radioactivity was detected in the blood after 1 minute, with levels remaining raised for at least an hour. This reveals the potential for an immediate response in the cardiovascular system. However, more recently this has been disputed by further experimental studies which found that most ultrafine particles are retained in the lungs (e.g. Möller *et al*., 2008). The effects of this retention or the effect of the small amount that does translocate to the bloodstream and other organs is still an area of active research.

3.4 Delayed acute effects

Some effects arising from air pollution exposure are not apparent until some hours post-exposure. The pro-inflammatory state observed by Svartengren *et al*. (2000) in subjects exposed to NO_2 and particles in a road tunnel persisted for over 4 hours – in fact the duration of the state is unknown but could be as long as 24 hours.

Little is currently known about the short-term effects of particle exposure, especially exposures of a few minutes duration. This is partly due to the predominant research focus on exposure to city-wide air pollution episodes lasting several hours to days and giving rise to breaches of 24-hour PM_{10} standards and increases in mortality. Several studies have shown that short-term exposure to diesel exhaust can lead to an inflammatory response in the lungs detectable in the blood without decrement in lung function (e.g. Salvi *et al*., 1999), such that the impact would not be noted by the subject. This lack of awareness may also apply to the increased bronchial responsiveness experienced by allergic asthmatics subsequent to NO_2 exposure, due to a similar lack of observed effect on lung function.

Effects subsequent to inflammation, especially in the cardiovascular system, have been observed in individuals in experimental studies of short-term particle exposure. For example, Brook *et al*. (2002) observed vasoconstriction, likely to be part of an immune response, in healthy subjects exposed to 150 $\mu g\ m^{-3}$ of particles and 120 $\mu g\ m^{-3}$ of ozone for two hours. The PM exposure would not be unusual for some busy tunnels, but such levels of ozone are highly unlikely in road tunnels.

Another recent study highly relevant to road tunnels is that of Riediker *et al*. (2004). North Carolina State Highway Patrol troopers took part in a thorough study among healthy young non-smoking men to assess effects of $PM_{2.5}$ in vehicles during a 9 hour shift. Physiological monitoring was undertaken and monitoring of $PM_{2.5}$ in the vehicles, although associations with fixed ambient and roadside $PM_{2.5}$ were also considered. The in-vehicle exposure was generally lower than the concentrations recorded at the outdoor fixed sites. This is not unexpected as the vehicles spent only a limited period of the shift driving in busy traffic, and due to the partial filtering of particles of outdoor source afforded by the vehicle envelope. Mean 9-hour exposure was 24 $\mu g\ m^{-3}$ of $PM_{2.5}$.

A few hours after exposure undesirable effects arose on vagal activity (ectopic beats), peripheral blood inflammatory markers (C - reactive protein) and coagulation markers (fibrinogen). Despite the lower in-vehicle concentrations, these effects were more strongly associated with in-vehicle $PM_{2.5}$ then external. The largest effect on heart-rate variability was seen on waking the morning after the in-vehicle exposure. This study is significant in terms of road tunnels, because it heralds cardiovascular effects that involve inflammation, coagulation and cardiac rhythm among a group at otherwise low risk for such outcomes. While the measured exposure was to $PM_{2.5}$, this arose in a setting of exhaust and highway air exposure within a vehicle and hence reproduces some of the aspects of combined pollutant exposure that might arise in a tunnel.

3.5 Chronic effects

The term “chronic effects” applies to the effects on the body of repeated long-term exposure, typically of the order of years. In the road tunnels context chronic effects may arise from chronic exposure, for example the twice-a-day travel of a commuter, or the more regular passage through a tunnel that may be experienced by professional drivers.

Detailed studies of acute effects of air pollution exposure have indicated that city-wide air pollution episodes tend to result in an increased number of vulnerable individuals. Longer-term cohort studies have revealed that persons living in more polluted environments are more likely to suffer long-term illness as a result, even if pollution levels are at relatively low levels (e.g. Dockery et al., 1993, Pope, 2000). For road tunnels this means that there is the potential for effects on the health of regular tunnel users that may not become manifest as acute effects, but will arise as the development of long-term illness. The problem for air quality management is that it will not be possible to trace the cause of such an illness to the tunnel, and the tunnel alone is unlikely to be the sole cause of the illness. The size or probability of such an effect is also very difficult to quantify or predict.

One of the most significant studies of recent years is the Southern California Children’s Health Study (Gauderman *et al*., 2007). This study’s power and impact has arisen from its size and scale, following over 3000 children in two cohorts over 8 years each. The study compared long-term exposure to traffic pollutants between children living in differently exposed communities over several years. The study found that children living in the more polluted communities (in terms of highway emissions) were significantly more likely to suffer from stunted lung development. The lungs stop developing around age 20. Gauderman and colleagues found significant lung function deficits in the cohorts at age 18 that were unlikely to be reversed. This deficit would be a strong predictor of chronic respiratory illness later in life and lead to more severe symptoms if a person were to develop an acute disease-related airway obstruction. These results translate most clearly to residential communities affected by road tunnel portals and vents, but could also equally apply to children who are repeatedly and regularly exposed to polluted road tunnel interiors.

Associations between air pollution and asthma have been difficult to isolate and prove or disprove. However, in recent years progress has been achieved through improved assessments of exposure to traffic emissions as opposed to air pollution in general. McConnell *et al*. (2006) found that residence within 75 m of a major road was associated with an increased risk of lifetime asthma or recent wheeze in a large cohort (over 8000 subjects) of first graders (aged 5 – 7) in Southern California. However, the association was not observed for children who moved to their residence after the age of 2, suggesting that exposure in the first 2 years of life are crucial in the development of asthma. In the Netherlands, Brauer *et al*. (2002) found wheezing, asthma, ENT infections and flu in a birth cohort by age 2 associated with modelled traffic air pollutants outside the place of residence. The strongest association was for asthma diagnosed before age 1.

Chronic effects also include the development of cancers as a result of exposure to the carcinogens in vehicle exhaust. These carcinogens include organic compounds, such as benzene and 1,3-butadiene, but also heavy metals found in wear products. The most difficult issue with regards to carcinogenic effects is that exposure to these compounds will arise from all exposure to traffic emissions and thus the impact of the time spent in a tunnel cannot be separated. Furthermore, epidemiological studies have been equivocal on a link between vehicle air pollution exposure and the development of cancers. The WHO review (WHO, 2000) considers benzene a genotoxic carcinogen and preferred a model for risk estimate that gave equal weight to concentration and duration of exposure. Hence the accumulation of dose becomes the determinant of cancer risk, such that risk is greater for repeated and regular tunnel users.

The implications for road tunnels from these findings about ultrafine particulate relate to the presence of adverse effects across sizeable at-risk groups in the community and the very high concentrations that can arise in an enclosed tunnel for the ultrafine particles. However, the current state of knowledge regarding ultrafine particles prevents the adoption of any guideline value or zero-effect threshold.

3.6 Risks to health of tunnel users – summary

For tunnel users, possible effects include aggravation of asthma, immediate or over subsequent hours, and risks of a cardiovascular event within a few days. Accrued effects from repeated tunnel use might include small increases in lifetime risk of cancer and potential for increased bronchitic events or respiratory infection.

4. Existing regulatory frameworks and applied limits

4.1 Overview

- Air quality limits applied inside tunnels around the world have generally been based upon either occupational exposure limits (particularly those developed by NIOSH in the US) or ambient air quality guidelines (particularly those of the World Health Organisation), but it is worth noting that there is no global agreement on which should apply.

- New Zealand's National Environmental Standards for Air Quality (AQNES) apply to ambient air, but explicitly do **not** apply inside tunnels. However, the New Zealand Workplace Exposure Standards apply to occupational exposure within tunnels. These Standards are less stringent than the AQNES or WHO ambient air quality guidelines as they are intended to apply to the typical healthy adult workforce, not the whole population.

- The WHO Guideline states that concentrations of CO averaged over a 15 minute period should not exceed 100 mg m^{-3} (87 ppm), the exposure at this level should not persist beyond 15 minutes, and should not be repeated within 8 hours.

- It is not uncommon for the WHO 15 minute guideline for CO (87 ppm) to be used as a criterion for assessing short term exposure to motor vehicle pollution, including indoor garages and is applied to road tunnels in Australia.

- The Permanent International Association of Road Congresses (PIARC) recommends a range of in-tunnel air quality limits. Their 'normal' operation limit of 100 ppm of CO is based upon the WHO 15-minute guideline of 87 ppm.

- Some national bodies have adopted or adapted the PIARC recommendations or the WHO guidelines directly, whereas elsewhere they have been adopted for individual tunnels. Some jurisdictions have adopted more stringent guidelines (such as Hong Kong which has a CO limit of 100 ppm over 5 minutes). In the recently built tunnels in Australia the WHO CO guideline has generally been adopted as the basis for in-tunnel CO limits set on a case-by-case basis as part of the Conditions of Approval for each scheme.

- A number of countries have taken a different approach, basing in-tunnel limits on the widely adopted 200 ppm short-term occupational limit for CO, rather than ambient guidelines.

- There is substantial evidence of the impact of exposure to particulate matter on human health for exposures of hours, days or years. The risk is particularly associated with ultrafine particles. However, there is far less evidence on the effects of the very short exposures likely to occur in road tunnels, and this evidence is insufficient at present to inform any guideline or standard. Nor is there any existing guideline for exposure to ultrafine particles for any exposure duration.

- An in-tunnel limit that is related to air quality is that of reduced visibility due to particulates. Loss of visibility is not related directly to effects on health, but has indirect effects, such as driver stress, as well as presenting a hazard to safe driving. PIARC recommends a set of 5 in-tunnel visibility limits corresponding to 5 traffic conditions.

4.2 Occupational exposure limits

Occupational exposure limits around the world have a wide variety of nomenclature, such as WELs (workplace exposure limits), MALs (maximum exposure limits), RELs (recommended exposure limits), STELs (short-term exposure limits), and so on. Despite the variety of names, however, two key facts are pertinent to this study: for most substances two exposure limits generally exist – one as an 8-hour average, intended to represent a typical workday exposure, and as a 15-minute average, intended to protect against peak short-term exposures. The 15-minute STEL is of interest to us as it approximately represents the order of exposure duration for tunnel users. [2]

Numerous bodies and agencies around the world set STELs, but there is wide agreement. One of the key organisations which set limits which tend to be adopted around the world is the National Institute for Occupational Safety and Health (NIOSH), part of the United States federal government's Department of Health and Human Services. For carbon monoxide the NIOSH 15-minute STEL of 200 ppm

[2] Urban tunnels are typically < 2km in length. Passage through a 2 km tunnel at an average 40 km/h will take 3 minutes. Passage through a 5 km tunnel would take 15 minutes at an average speed of 20 km/h.

seems to have been adopted universally. For NO_2, previous limits of 5 ppm or higher have generally been replaced by more stringent limits, such as the NIOSH REL of 1 ppm.

4.3 WHO ambient air quality guidelines

The World Health Organisation first published air quality guidelines for ambient situations in 1987. These guidelines were aimed at application in Europe, but were widely adopted globally as the basis of national and local guidelines. The Second Edition was published in 2000. In 2005 a Global Update reconsidered guidelines for ozone, particulate matter and nitrogen dioxide in the light of new research and consideration of global application.

The relevant guidelines for the road tunnel context are listed in table 4.1.

Table 4.1: WHO ambient air quality guidelines of relevance to road tunnels

Pollutant	Limit	Averaging period
CO	100 mg m^{-3}	15 minutes
Formaldehyde	0.1 mg m^{-3}	30 minutes
NO_2	200 µg m^{-3}	60 minutes
Styrene	70 µg m^{-3}	30 minutes
Toluene	1 µg m^{-3}	30 minutes

For particulate matter, WHO regard 24 hours a short-term exposure and have established guidelines of 50 µg m^{-3} and 25 µg m^{-3} for 24 hour average PM_{10} and $PM_{2.5}$ respectively. However, WHO makes it abundantly clear that the PM guidelines are not based on zero-effects thresholds (such thresholds have not been observed), and efforts should always be made to reduce PM concentrations as much as is practicable. There are no standards for assessing exposure to high levels of particulate matter at time scales less than 24 hours.

The WHO Guideline states that concentrations of CO averaged over a 15 minute period should not exceed 100 mg m^{-3} the exposure at this level should not persist beyond 15 minutes, and should not be repeated within 8 hours. An additional Guideline is set of 60 mg m^{-3} for 30 minutes (WHO, 2000). While the WHO criteria have been established for the general population, the risk to smokers may be greater than allowed for in the guidelines. Because smoking is a voluntary activity, guidelines for contaminants including CO were developed to protect non-smokers from environmental exposure (WHO, 1999).

4.4 New Zealand regulations and guidelines

4.4.1 New Zealand workplace exposure standards

New Zealand workplace exposure standards (WES) include time weighted averages (TWA), ceiling limits and short term exposure limits (STEL, applicable to 15 minute exposure). Where no STEL exists, using a TWA multiplied by a factor of 3 is recommended for 15 minute occupational exposure limits (DoL, 2002). MfE suggest that WES' may be used in place of ambient air quality guidelines for general population assessment criteria where none exist (MfE, 2006). To do this, WES should be divided by a factor of 40 to provide protection for sensitive members of the community because occupational criteria are established to protect healthy workers (MfE 2006).

Appropriate WES, as documented by OSH New Zealand (DoL, 2002), for contaminants that may be of concern in road tunnels include the following:

CO (15-minute STEL) = 200 ppm

NO_2 (15-minute STEL) = 5 ppm

Benzene (8-hour TWA) = 5 ppm

4.4.2 New Zealand ambient air quality standards, guidelines and targets

In New Zealand, the Ministry of Environment developed air quality guidelines which were informed by the WHO guidelines (MfE, 2002). The Resource Management (National Environmental Standards Relating to Certain Air Pollutants, Dioxins and Other Toxics) Regulations 2004 as amended in 2005 introduced the National Environmental Standards for Air Quality (AQNES), listed in Table 4.2. These Standards explicitly apply anywhere:

> “(a) that is in an airshed; and

(b) that is in the open air; and

(c) where people are likely to be exposed to the contaminant.

Areas which are **not** in the open air and where the standards do **not** apply include:

- inside a house
- **inside tunnels**
- inside vehicles." (MfE, 2005)

Table 4.2: National Environmental Standards for Air Quality (New Zealand)

Contaminant	Threshold concentration	Permissible excess
Carbon monoxide	10 mg m^{-3} as an 8-hr running mean	One 8-hour period in a 12-month period
Nitrogen dioxide	200 µg m^{-3} as an 1-hr mean	Nine hours in a 12-month period
Ozone	150 µg m^{-3} as an 1-hr mean	No exceedences allowed
PM_{10}	50 µg m^{-3} as an 24-hr mean	One 24-hour period in a 12-month period
Sulphur dioxide	350 µg m^{-3} as an 1-hr mean	Nine hours in a 12-month period
	570 µg m^{-3} as an 1-hr mean	No exceedences allowed

General ambient air quality management is the responsibility of Regional Councils. Several councils have adopted air quality limits that are more stringent than the AQNES. These limits are described as 'targets' and their intention is to provide councils with adequate time to respond if breaches of the AQNES are likely now or in the future. The target value is 66 % of the AQNES in at least Auckland, Waikato, Canterbury and Otago.

4.5 Is a tunnel indoors or outdoors?

We have noted that the AQNES very explicitly rules out their application for the purposes of the RMA to road tunnels. The WHO guidelines are generally interpreted to apply indoors and outdoors as the guidelines supposedly look at dose-response relationships only, as long as exposure is appropriately quantified.

It is not uncommon for the WHO 15-minute guideline for CO to be used as a criterion for assessing short term exposure to motor vehicle pollution, including indoor garages (e.g. Papakonstantinou *et al*. 2003) and is applied to road tunnels in Australia.

To determine what air quality criteria (if any) are applied to similar environments to road tunnels, local authorities were contacted in major New Zealand cities. This included environmental health officers, building inspectors and/or planners at Auckland City Council, Hamilton City Council, Wellington City Council and Christchurch City Council. These representatives were asked what air quality guidelines were applicable in environments such as enclosed bus stations, car parks and train stations.

Invariably the response was that Regional Councils were responsible for monitoring air quality, although this advice would appear to be misguided because Regional Councils do not have a mandate concerning indoor air quality. One council also advised that they relied upon air conditioning companies to install ventilation systems in these buildings and the Council requirements were satisfied if a certificate was produced to show that a system had been installed. The advice given was that property owners were responsible for any air quality monitoring after commissioning of the building.

A common response from Local Authorities was that occupational guidelines are appropriate in these environments and OSH is responsible for this. However, this is not appropriate in the case of carbon monoxide at least in road tunnels as vulnerable members of the population, who are not always expected to be protected by occupation guidelines, are likely to be exposed inside a road tunnel.

4.6 PIARC recommendations air quality in road tunnels

The World Road Association (PIARC) is a non-political and non-profit making association. It was granted consultative status to the Economic and Social Council of United Nations in 1970. PIARC's stated mission includes

- identifying, developing and disseminating best practice and giving better access to international information,

- developing and promoting efficient tools for decision making on matters related to roads and road transport.

To achieve these aims, PIARC creates and coordinates Technical Committees, including C.5 which deals with road tunnel operations. This committee regularly publishes Technical Reports, which, in recent years, have included:

1996: Road Tunnels: Emissions, Environment, Ventilation. Report 05.02.B

2000: Pollution by Nitrogen Dioxide in Road Tunnels. Report 05.09.B

2004: Road Tunnels: Vehicle Emissions and Air Demand for Ventilation. Report 05.14.B

2008: Road tunnels: a guide to optimising the air quality impact upon the environment .Report 2008R04

The key opportunity to manage the air quality within and around road tunnels is the setting of the in-tunnel concentration limits to which ventilation systems must be designed and subsequent operation can be aimed to maintain. The key international body that provides advice on such issues is PIARC (Permanent International Association of Road Congresses), which recommends a range of limits (see Table 4.3 below). The 'normal' operation limit of 100 ppm is based upon the WHO 15-minute guideline of 100 mg m^{-3} (i.e. 87 ppm).

Table 4.3: PIARC recommended in-tunnel pollutant limits

Traffic situation	CO-concentration Design year	
	1995 ppm	2010 ppm
Fluid peak traffic 50 - 100 km/h	100	70
Daily congested traffic, standstill on all lanes	100	70
Exceptional congested traffic, standstill on all lanes	150	100
Planned maintenance work in a tunnel under traffic	30	20
Closing of the tunnel	250	200

4.7 Ambient-based approaches

Some national bodies have adopted or adapted the PIARC recommendations or the WHO guidelines directly, whereas elsewhere they have been adopted for individual tunnels. For example, the French Ministry of Health has effectively adapted the 30 minute WHO Guideline in its ruling that CO concentrations in French tunnels should not exceed 50 ppm at any point in normal operation, or 150 ppm in emergency situations (referred to in CETU, 2003). In Hong Kong – one of the most densely tunnelled cities in the world with 17 km of new tunnels opening between 1989 and 2007 - a CO limit of 115 mg m^{-3} (i.e. 100 ppm) with a 5 minute averaging period has been established as one of three Tunnel Air Quality Guidelines (HKEPD 1995). The 5-minute averaging time is highly significant as it makes this guideline considerably more stringent than the WHO 15-minute average.

In the recently built tunnels in Australia the WHO CO guideline has generally been adopted as the basis for in-tunnel CO limits set on a case-by-case basis as part of the Conditions of Approval for each scheme. In addition to the 15-minute 87 ppm limit an additional limit of 200 ppm as a peak measured as a 3 minute average has also been implemented in the Cross City tunnel.

4.8 Occupational-based approaches

A number of countries have taken a different approach, basing in-tunnel limits on the widely adopted 200 ppm short-term occupational limit for CO, rather than ambient guidelines.

Norway has an extensive network of mostly alpine tunnels. The Norwegian Public Road Administration (NPRA) has established 15-minute limits for the end and middle of tunnels of 200 ppm and 100 ppm respectively (NPRA, 2004). Similarly an NO_2 limit of 1.5 ppm is set for the tunnel end and 0.75 ppm for the middle of the tunnel. Both NPRA limits for CO and NO_2 are based on providing an acceptable working environment for maintenance workers.

The UK also bases its limits on the 200 ppm occupation CO limit, but within an exposure-management framework described below.

4.9 Exposure-based approaches

All of the examples cited above employ a hazard-management approach in which the air quality limits apply to the concentrations as measured within the tunnel air, regardless of the presence or absence of traffic, workers, pedestrians, vehicle speed or volume.

We believe that the UK is unique in having taken an exposure-based approach to in-tunnel air quality limits. Such an approach is based on strictly applying limits to exposure of individuals, rather than just air concentrations. At the foundation of the UK approach is the occupational CO limit of 200 ppm as a 15-minute average, which is applied to mean no person using the tunnel should be exposed to a level of CO, averaged over 15 minutes, above 200 ppm. The UK has only two tunnels longer than 2 km (both under the River Mersey with lengths < 4 km), such that tunnel transit times are generally well below 15 minutes. This suggests that the demand placed on a ventilation system could be excessive. Exposure to a higher concentration than specified in a 15-minute limit may be allowed if compensated by shorter exposure duration. This can be shown to be true for carbon monoxide using the CFK model (see chapter 5). Its applicability for NO_2 is essentially unknown but plausible and the UK approach applies the same methodology for NO_2 and CO.

Further details and the implications and data requirements involved in establishing an exposure-based approach to air quality management in tunnels are discussed in detail in chapter 9.

4.10 Particles

It is not practical to regulate the thousands (if not millions) of individual chemical compounds that may be present in the atmosphere in particle form. Instead, **ambient** regulation and monitoring is generally based on the PM_{10} metric. PM_{10} is effectively the mass concentration of all particles with an aerodynamic diameter smaller than 10 μm.

However, nearly all of our knowledge on the health effects of particle inhalation relates to longer exposure times than occur in tunnels, typically hours, days and years leading to effects on a few individuals amongst large populations. This research has led to the adoption of 24-hour PM_{10} standards for ambient air, such as New Zealand's National Environmental Standard.

The effects on human health of short-term exposure (order of minutes) to airborne particulate matter, and repeated exposure, as would be experienced in repeated passage through a road tunnel, are insufficiently established to propose a guideline. We have found no evidence of such a guideline existing anywhere else in the world. This is also true for other toxic species present in vehicle exhaust, such as PAHs or benzene.

There is substantial scientific evidence that exposure to elevated levels of particles derived from road vehicle tailpipe emissions is a risk factor for exacerbations of cardiovascular disease (including mortality), and for cancer. There is also substantial evidence for effects on respiratory health. However, the causal pathways linking exposures to these effects are still active areas of research. Although it is toxicologically plausible that very brief exposures to very high levels of vehicle-related particulates (as is the case in a road tunnel) poses a risk to health, there is currently insufficient scientific understanding to quantify any risk and there are no associated guidelines.

Road traffic particulate emissions are predominantly in the form of ultrafine particles. The current state of knowledge regarding ultrafine particles is currently insufficiently developed to lead to any guideline value or zero-effect threshold for this sub-set of particulate matter.

As a first approximation it is reasonable to assume that particle species and metrics (e.g. PM_{10}, PAHs, black carbon, etc.) and other key toxic gases (such as benzene) are non-reactive (i.e. there is no direct evidence from road tunnels to assume otherwise), and therefore levels within and between tunnels vary in a similar way to CO. For this reason, a CO guideline provides a potential indication of the possible relative risk from particle exposure.

4.11 Visibility

An in-tunnel limit that is related to air quality is that of reduced visibility due to particulates. Loss of visibility is not related directly to effects on health, but has indirect effects, such as driver stress, as well as presenting a hazard to safe driving. The visibility guidelines are strictly intended to manage safety by ensuring that vehicles have enough visibility to able to respond to an incident on the road ahead of them. However, the visibility in a tunnel is directly related to the presence of particles large enough to scatter visible light. This essentially means particles of diameter greater than 0.4 μm. As noted elsewhere in this report, such particles have known impacts upon human health, and so monitoring visibility also provides the potential for an alternative assessment of the air quality and health risk within a tunnel. This assessment is limited by the short duration of exposure in tunnels compared to the longer exposure times for which the health effects of particles are established.

Visibility guidelines are expressed in terms of the extinction coefficient in units of m^{-1}. PIARC (2000) provides these subjective impressions of different values as they apply to road tunnels:

- 0.003 m^{-1} clear air
- 0.007 m^{-1} haziness
- 0.009 m^{-1} foggy
- 0.012 m^{-1} uncomfortable, yet will allow a vehicle to stop safely.

The commonly applied visibility limits are listed in Table 4.5.

Table 4.5: PIARC recommended in-tunnel visibility limits

Traffic situation	Visibility: Extinction coefficient K (m^{-1})	Visibility: Transmission (beam length: 100 m) %
Fluid peak traffic 50 - 100 km/h	0.005	60
Daily congested traffic, standstill on all lanes	0.007	50
Exceptional congested traffic, standstill on all lanes	0.009	40
Planned maintenance work in a tunnel under traffic	0.003	75
Closing of the tunnel	0.012	30

4.12 Existing regulations – summary

In New Zealand, the National Environmental Standards (NES) apply everywhere that is outdoors and where people are likely to be exposed to the contaminant. These Standards do not apply inside tunnels or vehicles. Regional Air Quality Targets also apply to outdoor locations which are sometimes (but not always) more stringent than the NES.

The New Zealand Workplace Exposure Standards apply to occupational exposure within tunnels. These Standards are less stringent than the NES or WHO ambient air quality guidelines as they are intended to apply to the typical healthy adult workforce, not the whole population.

Around the world different ambient and occupational limits have been applied inconsistently to the interior of road tunnels. The most commonly applied limits have been 15-minute average CO limits based mostly on either an occupational limit of 200 ppm, or the WHO limit of 87 ppm. The UK has implemented an exposure-management approach in which an exposure limit is translated to a sensor limit which allows higher in-tunnel concentrations in shorter tunnels based on the assumption of shorter exposure duration. Visibility limits are also adopted in many tunnels for the purposes of safety, but which also provide some air quality protection.

5. Carbon Monoxide – toxicity and exposure limits

5.1 Overview

- The action of carbon monoxide (CO) in the body limits the oxygen transport in the blood and delivery to the organs. Adverse effects of carbon monoxide in the body have been related to the degree to which it replaces oxygen in the blood stream. This is conventionally assessed as a percentage of the total haemoglobin that exists as carboxyhaemoglobin, or %COHb. Various health effects have been associated with different levels of blood %COHb.

- CO exposure limits are set on the basis of a corresponding %COHb limit. Commonly, occupational exposure limits have been set based on a 5 % COHb limit (e.g. NIOSH, Australian and New Zealand). Ambient guidelines, such as those of WHO, have been based on a 2.5 or 2 % COHb limit.

- A 5 % COHb limit appears to be the highest acceptable limit for the protection of health of tunnel users. The question of whether to adopt a 2.5 % limit requires a consideration of whether it is believed that the target groups it protects are exposed in a tunnel. The cardiovascular effects at 2.5 – 5 % COHb are partly related to exercise-induced angina, and it may be argued that such persons will not be exercising within a tunnel. However, as WHO makes clear, the vulnerable group includes those with latent and/or undiagnosed heart disease and their lack of awareness of their own condition may make them more vulnerable. Furthermore, we must assume that along with this group, pregnant mothers are users of road tunnels. For these reasons we recommend that a 2.5 % or lower limit is adopted for road tunnels in New Zealand.

- %COHb limits can be translated into CO exposure limits if various physiological parameters are known, or assumed. We recommend a CO limit which is based upon physiological parameters that provide protection for all potential users of a road tunnel.

- We propose that a CO limit of 87 ppm averaged over 15 minutes, i.e. equal to the WHO ambient guideline, would appear to provide protection for all typical tunnel users, including walkers and cyclists for those tunnels that permit them, based on a COHb limit of 2.5 %. This limit would be more than adequate to protect healthy non-smoking adults in vehicles.

- We accept that an 87 ppm (15-minutes) guideline for CO is less demanding than a number of guidelines adopted for road tunnels elsewhere in the world, specifically the 70 ppm recommendation from PIARC (from 2010), the 50 ppm limit in France and the 100 ppm (5 minutes) guideline in Hong Kong. However, we find no sound **medical** evidence for recommending a guideline more demanding than the WHO limit at this time.

- An occupational safety limit of 200 ppm averaged over 15 minutes appears to have been universally adopted. We consider its scientific basis to be appropriate for road tunnel exposure, and see no reason to challenge the widespread adoption of this limit.

- An 8-hour time-weighted average limit of 30 ppm also has a long history and is widely adopted internationally. We consider its scientific basis to be appropriate for road tunnel exposure, and see no reason to challenge the widespread adoption of this limit. We note that PIARC recommend a reduction in the limit from 30 ppm to 20 ppm from 2010. However, this recommendation appears to not have any supporting documentation. We are therefore unable to comment on whether this reduction is based on medical evidence or other considerations.

- A CO limit may be used as a proxy for all other relevant air pollutants on the assumption that if CO concentration limits are not breached then potential limits for other substances will not be breached either. CO is appropriate for this on the basis of a) its well-established link to serious acute health effects, b) vehicle emission factors for CO being relatively well-constrained, allowing prediction at design stage, c) CO being unreactive in this context.

- However, the use of a CO limit as a single proxy to provide protection within all appropriate limit values for other substances has a number of weaknesses, including a) relatively poor emission data for other substances (including NO_2), b) the assumption that locally-relevant and accurate emission data are available, c) emission factors and their ratios (e.g. CO/NO_x) are not constant with time (see chapter 7), d) it is vulnerable to sudden changes in health-based limits arising from continuous improvements in scientific understanding of the effects of pollutants, and e) it is very sensitive to the correct characterisation of the number and nature of heavy duty diesel vehicles using the tunnel.

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5.2 CO and carboxyhaemoglobin in the body

The action of CO in the body arises from its much higher binding affinity to haemoglobin than oxygen. Thus the formation of carboxyhaemoglobin preferentially to oxyhaemoglobin limits the oxygen transport in the blood and delivery to the organs.

Adverse effects of carbon monoxide in the body have been related to the degree to which carbon monoxide has replaced oxygen in the haemoglobin molecules in the blood stream. This is conventionally assessed as a percentage of the total haemoglobin that exists as carboxyhaemoglobin, or %COHb. Various health effects have been associated with different levels of blood %COHb, some of which are listed in Table 5.1.

Table 5.1: Some effects on health associated with different levels of blood %COHb, adapted from HPA, 2006

Carboxyhaemoglobin in blood (%)	Signs and symptoms
<2	No significant health effects
2.5-4.0	Decreased short-term maximal exercise duration in young healthy men
2.7-5.2	Decreased exercise duration due to increased chest pain (angina) in patients with ischaemic heart disease
2.0 – 20.0	Equivocal effects on visual perception, audition, motor and sensorimotor performance, vigilance and other measures of neurobehavioural performance
4.0-33.0	Decreased maximal oxygen consumption with short-term strenuous exercise in young healthy men
20-30	Throbbing headache
30-50	Dyspnoea, dizziness, nausea, weakness, collapse, coma
> 50	Convulsions, unconsciousness, respiratory arrest, death

5.3 Zero-effects threshold

CO exposure limits are set on the basis of a corresponding %COHb limit. Commonly, occupational exposure limits have been set based on a 5 % COHb limit (e.g. NIOSH, Australian and New Zealand). Ambient guidelines, such as those of WHO, have been based on a 2.5 or 2 % COHb limit.

The choice of a CO limit for road tunnels is initially based upon the selection of the % COHb limit. The rationale for a 5 % limit derives from the fact that neurobehavioural effects and developmental toxicity effects in people have not been reported below COHb levels of 5%. The rationale for a 2.5 % limit derives from the observation of effects in some vulnerable groups. In particular some studies found that patients with angina pectoris experienced reduced time to onset of **exercise-induced** chest pain after of exposure to CO leading to %COHb levels of 2.5 – 5.9 % (WHO, 2000). From a review of the effects of smoking exposure on ischaemic disease patients Mennear (1993) set a lower limit of 2.5% as a conservative estimate for a zero-effects threshold.

A 2.5 % limit is also intended to provide extra protection for pregnant mothers and foetuses. This is because pregnancy leads to increased endogenous production of COHb so that an equal exposure to environmental CO will lead to higher %COHb levels in a pregnant mother than in a typical non-pregnant woman by 0.7 – 2.5 %.

Consequently, the WHO states:

> "To protect non-smoking, middle-aged and elderly population groups with documented or latent coronary artery disease from acute ischaemic heart attacks, and to protect the foetuses of non-smoking[3] pregnant women from untoward hypoxic effects, a COHb level of 2.5% should not be exceeded."

Allred *et al*. (1989) reported effects in exercising patients at COHb levels as low as 2%. This result was disputed by Mennear (1993), who argued that the gas chromatography method used to measure %COHb was inaccurate. However, it remains that 2 % COHb corresponds to the lowest observed effect level reported in the literature. Consequently, a 2 % limit has been adopted as the basis of ambient CO limits in the UK and Canada.

A significant knowledge gap remains regarding the effects of %COHb levels in children, and the appropriateness of adult-based limit values. Much less experimental

[3] 'Nonsmoking' has been specified because smokers expose themselves to much higher levels of carbon monoxide leading to higher %COHb levels (WHO (2000) reports up to 10 %). Raising the limit to the level experienced by smokers would deny protection to non-smokers.

data on children exist, partly because of the difficulty in obtaining measurements of children's blood. Children will not be pregnant, are much less likely to be smokers, and much less likely to suffer from coronary artery disease. On the other hand, their organs are in a developmental stage and will not necessarily respond to raised %COHb in the same way as adults. As discussed in more detail below, children are known to take up CO into their blood faster than adults, and may have higher baselines (% COHb levels pre-environmental exposure).

The WHO air quality guideline documentation makes no reference to whether the CO guideline provides protection for children. An extensive review of the effects of CO on children also found no evidence of whether a 2.5 % COHb limit was appropriate for children (Kleinman, 2000).

5.4 Adopting a %COHb limit for road tunnels

In-tunnel CO limits applied in road tunnels have been based on different %COHb limits. The United Kingdom, France and Norway have in-tunnel CO limits based upon a 5 % COHb limit. The United States and Australia have adopted a 2.5 % limit, as have the WHO for the ambient air quality guidelines.

A 5 % COHb limit appears to be the highest acceptable limit for the protection of health of tunnel users. The question of whether to adopt a 2.5 % limit requires a consideration of whether it is believed that the target groups it protects are exposed in a tunnel. The cardiovascular effects at 2.5 – 5 % COHb are partly related to exercise-induced angina, and it may be argued that such persons will not be exercising within a tunnel. However, as WHO makes clear, the vulnerable group includes those with latent and/or undiagnosed heart disease and their lack of awareness of their own condition may make them more vulnerable. Furthermore, we must assume that along with this group, pregnant mothers are users of road tunnels. For these reasons we recommend that a 2.5 % or lower limit is adopted for road tunnels in New Zealand.

5.5 CFK model and predicting %COHb

The key to implementing a CO limit that restricts the level of %COHb in the blood is a description of the way the body produces and removes COHb in response to environmental exposure and endogenous production. Such a description was developed in the 1960s. The Coburn-Forster-Kane (CFK) equation (Coburn *et al.*, 1965), although recognised as imperfect, provides a practical and reliable way to relate CO exposure to %COHb in the blood, and thus translate a %COHb limit to a CO exposure limit.

A variety of versions or formats of the CFK model exist, although the choice of version will not change our results and conclusions in this context. One such version is shown below.

$$\frac{d[COHb]}{dt} = \frac{1}{V_B}\left(V_{CO} + \frac{P_I CO}{\beta} - \frac{[COHb]}{[O_2 Hb]}\frac{P_C O_2}{M}\right)$$

Where

$$\beta = \frac{1}{D_L CO} + \frac{P_L}{V_A}.$$

Thus, the CFK equation is a differential equation linking the rate of change in [COHb], the volume of CO per volume of blood, with $P_I CO$, the partial pressure of CO in inhaled air. The equation consists of three terms representing (left to right) endogenous production, exogenous production and removal of COHb. The dependent parameters describe the existing fraction of carboxyhaemoglobin, $[COHb]/[O_2Hb]$, and a range of physiological parameters which govern the rate of gas exchange in the body.

These parameters are:

V_B: the blood volume,

V_{CO}: the rate of endogenous CO production,

$D_L CO$: the pulmonary diffusing capacity for CO,

V_A: the alveolar ventilation rate

Furthermore, three constants are included:

$P_C O_2$: the average partial pressure of carbon monoxide (100 mmHg),

M: the Haldane coefficient (218),

P_L: the pressure of dry gases in the lungs (713 mmHg)

McCartney (1990) conducted a sensitivity study to determine which parameters were most important for a range of exposure times. For an exposure time of 15 minutes the rate of COHb uptake is most sensitive to the blood volume and external CO concentration, such that faster COHb generation is achieved for higher CO concentrations and lower blood volumes. Following these parameters, the alveolar ventilation rate is the next most sensitive. COHb response has a relatively low sensitivity to initial %COHb and pulmonary diffusing capacity.

5.6 Physiological parameters for the CFK model and their values

To use the CFK equation to convert %COHb limits into CO exposure limits we need to select values for the physiological parameters. Scientific knowledge about what some of these values should be is limited.

5.6.1 Blood volume

Average human blood volume can be expressed as a function of body weight, such that:

V_B = 80 mL / kg of body weight for children,

V_B = 70 mL / kg of body weight for adults (Coburn, 1970)

Extensive body weight data are available from the National Center for Health Statistics in the United States (www.cdc.gov/nchs). A summary of some key data is presented in Table 5.2.

Table 5.2: Median body weight from US data, and estimated blood volume

age	Median Body weight / kg		Estimated blood volume / mL	
	male	female	male	female
2	7.8	7.2	620	580
5	18.5	18	1500	1400
10	32	33	2600	2600
20	71	58	5700	4600
Adult	86	74	6900	5900

5.6.2 Alveolar ventilation rate

We will estimate alveolar ventilation rate with the simple formulae:

V_A = alveolar volume x breathing frequency,

where alveolar volume = tidal volume – dead space volume,

dead space volume = 2.2 x body weight (Numa & Newth, 1996),

tidal volume = breath volume / breathing frequency.

Thus, alveolar ventilation rate is a function of breath volume and breathing frequency, which are dependent upon age, sex and activity, and on body weight, which is also a function of age and sex.

We adopt values of breath volume and breathing frequency from a large observational study (Adams, 1993). In Table 5.3 we have estimated the alveolar ventilation rate for a range of hypothetical persons which are relevant for road tunnels.

Table 5.3: Estimated alveolar ventilation rate for a range of hypothetical persons

subject	Activity	Breath volume / l min^{-1}	Breathing frequency / min^{-1}	V_A / ml min^{-1}
Adult male	Driving	10.8	16.8	7 600
Adult male	Passenger	9.8	16.2	6 800
Adult female	Passenger	8.2	17.6	5 300
Older adult male	Walking at 3.3 mph	33.4	22.5	29 000
Young adult male	Running at 5 mph	55.9	34.6	50 000
5 yr old child	Resting	8	20	7 200

A higher alveolar ventilation rate implies a faster transfer of CO to the blood and a faster production of COHb. This means that persons walking, running, or engaged in any strenuous activity whilst exposed to elevated levels of CO are more at risk. A child's smaller breath volume is partially compensated for by the higher frequency of breaths.

5.6.3 Pulmonary diffusing capacity

Data on the variability of the pulmonary diffusing capacity are limited. It is known to increase linearly with exercise, be lower in persons with pulmonary hypertension and vary slightly with age (in adults) and height. Some indicative values are given in Table 5.4.

Table 5.4 Some indicative values for the pulmonary diffusing capacity (mL CO (min mmHg)$^{-1}$) (from Huang *et al*., 2007 and Mitchell *et al*., 1998)

	At rest	During exercise
6 – 9 years	15	17
30 yr old male	30.1	38.2
30 yr old female	22.5	26.2
65 yr old male	25.3	32.4
65 yr old female	20.6	25.4

5.6.4 Initial (baseline) COHb concentration

Exposure to CO inside the tunnel will raise the concentration of COHb in the blood above that which was present before the subject entered the tunnel. This level is the baseline. It is governed by the endogenous production rate in the body and by the net exposure to environmental CO in the hours preceding arriving at the tunnel. The endogenous production rate varies between individuals. It is higher for those with a condition that stimulates CO production (such as haematoma, anaemia, thalassaemia). It is also higher for pregnant mothers. Both endogenous production and baseline concentrations can be raised at high altitude. The baseline COHb will also be higher for smokers. Conventionally, smoking exposure is not included in the derivation of a CO exposure limit and we follow that convention here. Pregnant mothers are offered protection by the adoption of a 2.5 % COHb limit (see above). As far as the literature allows us to check, other limit-setting agencies adopt a 'typical' endogenous production rate of 0.007 mL min^{-1}. We find no reason to adopt, nor any evidence to support any other values, and so retain this value.

To simulate pre-tunnel exposure we will use the CFK equation to predict the %COHb production arising from typical urban ambient concentrations of CO. In 2006, the annual mean CO concentrations recorded by permanent monitors run by Auckland Regional Council were typically 0.2 – 0.5 ppm at locations representative of residential areas. Adopting values of the CFK parameters which represent a mostly sedentary adult we arrive at a baseline COHb level of 0.26 % for an exposure to 0.2 ppm and 0.30 % for an exposure to 0.5 ppm. We therefore propose adopting a baseline value of 0.3 % for application to in-tunnel CO limits.

5.7 Formulation of CO limits

In the documentation supporting declaration of carbon monoxide air quality standards and guidelines around the world, for both ambient and occupational applications, the related %COHb is usually, but not always stated. The values of the physiological parameters adopted are often not quoted. Table 5.6 lists the values we have been able to source. The values for the NIOSH 15 minute occupational limit are representative of an adult undertaking strenuous activity in a relatively polluted environment, as appropriate for an occupational limit. However, a relatively high limit of 200 mg m^{-3} arises due to the limit being based on a 5 % COHb target as the limit is not expected to apply to pregnant mothers or coronary heart disease patents.

NIOSH recommends alternative values should be applied if the nature of the occupational exposure warrants it. For instance, values of the pulmonary diffusing capacity and alveolar ventilation rate may be selected from the list reproduced in Table 5.5.

Table 5.5: values of the pulmonary diffusing capacity and alveolar ventilation rate for use in the CFK equation based on level of activity of an occupationally exposed adult, from NIOSH

Level of activity	D_LCO / mL (min mmHg)$^{-1}$	V_A / mL min^{-1}
Sedentary	30	6 000
Light	40	18 000
heavy	60	30 000

Although we have not been able to source the values used in the derivation of the WHO guidelines, the guidelines do state that the short-term CO limits:

> "have been determined in such a way that the COHb level of 2.5% is not exceeded, even when a normal subject engages in light or moderate exercise". (WHO, 2000)

This implies that values similar to those specified by NIOSH for light activity (Table 5.5) were adopted.

Table 5.6: Values of CFK model parameters adopted in the formulation of some CO exposure limits

Body	Occupational or ambient	CO limit / ppm	Averaging time / min	%COHb limit	V_B / mL	V_A / mL min $^{-1}$	D_LCO / mL (min mmHg)$^{-1}$	$[COHb]_0$
NIOSH	occupational	200 ppm	15	5 %	5 500	30 000	60	0.75 %
USEPA AEGL-2	occupational	420 ppm	10	4 %	5 500	13 000	30	0.75 %
Canadian EPA	ambient	30 ppm	60	2 %	unspecified	18 000	unspecified	unspecified
US EPA (1980)	Ambient	35 ppm	60	2.5 %	5 500	20 000	30	0.5 %
WHO	ambient	87 ppm	15	2.5 %	unspecified	unspecified	unspecified	unspecified

5.8 Recommendations for a road tunnel CO limit for New Zealand

We recommend that a CO limit be chosen which is based upon physiological parameters that provide protection for all potential users of a road tunnel. Thus, in order to achieve a stringent limit, a blood volume of 5 500 mL has been adopted, in agreement with all other assessments identified in Table 5.7, despite this probably being below the average adult volume.

Table 5.7: Alternative 15-minute CO limits derived from protection of a range of subjects

subject	%COHb limit	V_B / mL	V_A / mL (min mmHg)$^{-1}$	D_LCO / mL min^{-1}	[COHb]$_0$	CO limit / mg m^{-3}
Driving adult	2.5 %	5 500	7 600	30	0.3 %	320
Driving adult	2.5 %	5 500	7 600	30	0.75 %	256
Driving adult	2.0 %	5 500	7 600	30	0.75 %	184
Child passenger (5 yrs)	2.5 %	1 400	7 200	15	0.3 %	110
Child passenger (5 yrs)	2.5 %	1 400	7 200	15	0.5 %	102
Adult, light exercise	2.5 %	5 500	20 000	40	0.3 %	157
Adult, walking	2.5 %	5 500	30 000	40	0.3 %	128
Adult, running or cycling	2.5 %	5 500	50 000	40	0.3 %	105

From this exercise we note that a CO limit of 87 ppm averaged over 15 minutes, i.e. equal to the WHO ambient guideline, would appear to provide protection for all typical tunnel users, including walkers and cyclists for those tunnels that permit them, based on a COHb limit of 2.5 %. This limit would be more than adequate to protect healthy non-smoking adults in vehicles, for whom a 5 % COHb level is a safe level not associated with any adverse effects. I.e. this group could probably be exposed to over 600 ppm for 15 minutes without adverse effect. The appropriateness of a 100 mg m^{-3} limit for the protection of children must be judged with caution due to the lack of evidence supporting a 2.5 % COHb limit for zero effects in children.

We accept that an 87 ppm (15-minutes) guideline for CO is less demanding than a number of guidelines adopted for road tunnels elsewhere in the world, specifically the 70 ppm recommendation from PIARC (from 2010), the 50 ppm limit in France and the 100 ppm (5 minutes) guideline in Hong Kong. However, we find no sound **medical** evidence for recommending a guideline more demanding than the WHO limit at this time.

5.9 Recommendations for a road tunnel Occupational Safety CO limit for New Zealand

An occupational safety limit of 200 ppm averaged over 15 minutes appears to have been universally adopted. Its basis is indicated in the choice of parameters shown for the NIOSH limit in Table 5.6. We find those parameters to be appropriate, the resulting limit to offer adequate protection, and see no reason to challenge the widespread adoption of this limit.

An 8-hour time-weighted average limit of 30 ppm also has a long history and is widely adopted (e.g. in the UK, Germany and Sweden, the NIOSH Recommended Exposure Limit is 35 ppm). It is based on CFK model parameters which we find no basis to challenge. We note that PIARC recommend a reduction in the limit from 30 ppm to 20 ppm from 2010. However, this recommendation does not have any supporting documentation to justify it. We find ourselves, therefore, unable to comment on whether we believe this reduction is based on medical evidence or other considerations.

5.10 Rationale for use of CO as a proxy for all tailpipe emissions

Although multiple concentration or exposure limit values (ambient and occupational) exist covering the air pollutants which may be found in road tunnels the bulk of these pollutants are all released from the same source – the traffic within the tunnel. Although the relative proportions of, say, benzene and carbon monoxide emitted from two different vehicles will vary, taking the traffic fleet using any given tunnel as a whole these relative proportions will tend towards an average fleet emission profile. This profile will be tunnel-specific, but if the mix of vehicles (sizes, ages, HDV mix, etc) is typical then the tunnel profile will tend towards the national emission profile. For example, the New Zealand Traffic Emission Rates model (NZTER) estimates the mean emission factors for petrol cars within the New Zealand fleet in 2008 in free-flowing (central urban) traffic are 5.5 g km^{-1} for CO and 0.69 g km^{-1} for NO_x. This shows that for any given distance driven (e.g. the length of a tunnel) the emission of CO is ~8 times greater than the emission of NO_x in terms of mass[4]. When emitted into a closed volume, such as a tunnel, this should mean that the resulting air concentration of CO due to cars in the tunnel alone (i.e. neglecting background contributions from outside the tunnel) will tend to 8 times the concentration of NO_x.

This indicates that if the relevant emission factor for more than one substance is known then one needs to know only the in-tunnel concentrations of any one of those substances in order to estimate the concentrations of the others, provided the background concentrations are known (unless they can be shown to be negligible).

I.e.

$$Ai = \left((Bi - Bb)\frac{EF_A}{EF_B} \right) + Ab$$

Where

A_i = internal concentration of substance A

B_i = internal concentration of substance B

A_b = background concentration of substance A

B_b = background concentration of substance A

EF_A = emission factor of substance A

EF_B = emission factor of substance B

[4] The recently released Vehicle Emission Prediction Model (VEPM) predicts higher CO emissions and lower NO_x emissions at lower speeds, so that CO emissions are greater than NO_x emissions by a factor of 14 at 50 km/h.

For example, if the background concentrations were 100 μg m^{-3} of CO and 10 μg m^{-3} of NO_x and the in-tunnel concentration of CO was 5000 μg m^{-3}, then, given the 2008 emission factors quoted above, the in-tunnel concentration of NO_x can be estimated thus:

$$= \left((5000 - 100) \frac{0.69}{5.5} \right) + 10$$

= 625 μg m^{-3}.

This approach provides one basic rationale for employing a limit for a single pollutant in tunnels, as only one need be monitored for the concentration of other pollutants to be calculated from it. The objective then is to select the most appropriate pollutant to regulate.

5.11 Rationale for use of carbon monoxide

The choice of which single proxy pollutant to use in air quality regulation of tunnels is informed by other considerations, primarily practicality and the scientific robustness of the known health impacts of that pollutant. Historically, carbon monoxide has been chosen. Carbon monoxide (CO) has several advantages:

- There is a very well-established body of scientific knowledge linking short-term exposure (order of minutes) to several serious health impacts,

- Road vehicle exhaust is its dominant ambient source, such that in-tunnel concentrations are generally very high compared to background concentrations (often by orders of magnitude), such that background concentrations can be generally disregarded (with an implicit cost saving as external monitoring is not required),

- Vehicle emission factors for CO are relatively well-constrained,

- On the time-scales relevant to road tunnels CO is chemically unreactive and conserved, unlike several other traffic pollutants, like NO.

Thus, until very recently, road tunnel ventilation systems have been designed so that CO concentration limits are not breached on the understanding/assumption that if this goal is achieved, potential limits for other substances will not be breached either.

5.12 Is CO an adequate proxy for benzene?

As noted in section 4.4 above, a Workplace Exposure Standards for benzene is in force in New Zealand, and explicitly applies inside tunnels. The standard is 5 ppm as an 8-hour time-weighted average. The equivalent CO guideline recommended by NIWA is the NIOSH 8-hour guideline of 30 ppm. Applying the same approach as for NOx above we estimate the probable concentration of benzene in a road tunnel if the concentration of CO equals 30 ppm.

$$benzene_i = \left((CO_i - CO_b) \frac{EF_{benzene}}{EF_{CO}} \right) + benzene_b$$

Where

$benzene_i$ = internal concentration of benzene

CO_i = internal concentration of CO = 30 ppm (= 35 mg m^{-3})

$benzene_b$ = background concentration of benzene, for which we estimate a typical value of 1 µg m^{-3}

CO_b = background concentration of CO, which is negligible compared to 30 ppm

$\frac{EF_{benzene}}{EF_{CO}}$ = emission factor mass ratio of benzene to CO. This ratio is not readily available from observational data in New Zealand. We therefore estimate this as the product of the published estimate of the fleet-average fraction of total VOCs (volatile organic compounds) emitted from vehicle tailpipes as benzene of 0.07 (MfE, 2008) and the fleet-average emission ratio of VOCs to CO as predicted by the VEPM emission model (0.08 at an average speed of 40 km/h[5]).

[5] This relatively low speed gives a conservative result relative to higher speeds.

Hence,

$$benzene_i = ((35)0.07 * 0.08) + 0.001 = 0.20 \text{ mg m}^{-3} = 0.06 \text{ ppm}$$

Thus, when the air in the tunnel is sufficiently contaminated to breach the 8-hour CO guideline, levels of benzene are still 83 times lower than the 8-hour benzene guideline.

The same approach cannot be applied to other air toxics associated with vehicle emissions for which short-term WHO guidelines apply, i.e. formaldehyde, styrene and toluene, because vehicle emission factors or emission ratios are not available for New Zealand.

5.13 Where and when a single proxy may be inadequate

The use of a CO limit as a single proxy to provide protection within all appropriate limit values for other substances has a number of major weaknesses:

1. It assumes that the appropriate national fleet emission factors are known, but for many substances these data are subject to large uncertainty or is effectively unknown. Even the better-constrained emission factors possess significant uncertainty.

2. It assumes that national emission factors and fleet characteristics are applicable in specific tunnels. Advice and data exist to correct for local variations (reviewed below in chapter 7 and covered in a forthcoming report from NIWA) but this is also subject to uncertainty and is dependent upon the quality (or even existence) of local data.

3. It assumes that the emission factor ratios (e.g. CO/NO_x) are constant with time. It is well-established that this is not so, especially over the previous and forthcoming decades when major changes in engine technology are leading to a significant evolution in the composition of vehicle exhaust (see a full discussion in chapter 7).

4. It is vulnerable to sudden changes in health-based limits arising from continuous improvements in scientific understanding of the effects of pollutants.

5. It is very sensitive to the correct characterisation of the number of heavy duty vehicle vehicles using the tunnel, the quality of their emissions (which is partly related to fuel quality, but also age and state of maintenance).

5.14 Carbon monoxide limits – summary

A carbon monoxide limit can be set on the basis of the relatively well understood toxicology of that compound. Basing the limit on a carboxyhaemoglobin limit of 2.5 % (or 2 %) provides protection for all non-smoking adult tunnel users, including pregnant mothers and coronary disease patients.

This may be translated into a carbon monoxide limit of 87 ppm as a 15-minute average which will protect non-smoking tunnel users from a COHb level above 2.5 % in the case of light to moderate activity. It is also likely to provide protection for children travelling in vehicles.

A carbon monoxide limit alone, however, may not provide adequate protection to tunnel users from the effects of nitrogen dioxide or particles. The former use of CO as a proxy for all harmful traffic emissions in road tunnels is no longer appropriate.

6. Nitrogen dioxide – toxicity and exposure limits

6.1 Overview

- NO_2 is associated with the various adverse impacts on health. These include important effects among children: increased respiratory symptoms, onset of respiratory symptoms among infants, increased bronchitic symptoms for children with asthma.

- The WHO has published two guidelines regarding annual and 1-hour exposure to NO_2. No guidelines exist for shorter ambient exposures.

- There is no internationally agreed short-term (15-minute average) occupational exposure limit for NO_2 in the same way as there is for longer ambient exposures. Occupational NO_2 limits vary between countries and determining bodies, and have tended to change with time.

- In the US, ACGIH and NIOSH have recommended a short-term exposure limit (STEL) of 5 ppm. This value has subsequently been adopted in many other countries, including Japan, Australia and New Zealand. However, NIOSH also state a recommended exposure limit (REL) of 1 ppm.

- With the coincidence of new health research data, the NIOSH 1 ppm REL and the rise of NO_2 emissions relative to CO from road vehicles, increased attention has been paid to NO_2 limits in tunnels.

- The Permanent International Association of Road Congresses (PIARC) has recommended a limit value for road tunnels of 1 ppm averaged over the length of the tunnel not to be exceeded more than 2 % of the time. No averaging time was specified.

- Recently experimental studies of short-term exposures to tunnel air have been conducted in Sweden showing adverse health effects following exposures to 160 ppb of NO_2 over 30 minutes. However, they were based on very small numbers of subjects; have yet to be repeated and the implications for development of ill-health are unknown. A report to the Swedish Road Administration argued that a precautionary approach might be to consider an

NO_2 limit corresponding to half this concentration (80 ppb) or half this time (15 minutes). We have found no evidence of this recommendation having been adopted in Sweden or anywhere else. Nevertheless, it seems plausible that, if such experimental data are corroborated, the argument for such demanding guidelines may become stronger in the future.

- International precedent permits a wide choice when selecting an NO_2 limit. We believe that a value of 5 ppm should be considered too high, as it is above the NIOSH recommended exposure limit for occupational purposes, and has been abandoned by the UK Health & Safety Executive because it gave insufficient protection. A 1 ppm limit, based on the NIOSH REL could be adopted, with the knowledge that such as value has been recommended by PIARC for use in road tunnels. However, whilst it appears that a 1 ppm limit gives sufficient protection to healthy individuals, there is now some evidence that it does not provide full protection for asthmatics and sufferers from chronic obstructive pulmonary disease. To protect such individuals within a road tunnel, a precautionary approach would suggest that a 0.4 ppm limit is appropriate, as adopted in France. We also note that there is some argument for a limit as stringent as 0.2 or 0.1 ppm to provide a factor of safety, although such a limit has yet to be adopted.

6.2 Effects of NO_2 on the body

The recent update summary from the WHO Air Quality Guidelines (released 2006) re-iterates that NO_2 is associated with the various adverse impacts on health. These include important effects among children: increased respiratory symptoms, onset of respiratory symptoms among infants, increased bronchitic symptoms for children with asthma. Also other demonstrated effects among people with asthma include: direct effects on lung function, increased bronchial responsiveness at levels upwards of 200 $\mu g\ m^{-3}$ (0.11 ppm), or above 1900 $\mu g\ m^{-3}$ (1.01 ppm) in non-asthmatics (Folinsbee, 1992)

6.3 Ambient NO_2 guidelines and standards

The WHO has published two guidelines regarding long-term and short-term exposure to NO_2:

40 $\mu g\ m^{-3}$ (21 ppb) as an annual mean,

200 $\mu g\ m^{-3}$ (106 ppb) as a 1-hour mean.

The 1-hour guideline is largely based on the observation of increased bronchial responsiveness amongst asthmatics following short-term exposures above 200 $\mu g\ m^{-3}$ (WHO, 2006). The guideline is thus intended to provide protection for those individuals found to be most susceptible to short-term NO_2 exposure: asthmatics. This 1-hour mean guideline has been adopted as one of the Air Quality National Environmental Standards (AQNES) in New Zealand, with 9 exceedences allowed in 12 months.

In New Zealand, a further air quality guideline exists for intermediate exposure:

100 $\mu g\ m^{-3}$ (53 ppb) as a 24-hour mean.

6.4 Occupational short-term NO_2 exposure limits

There is no internationally agreed short-term (15-minute average) occupational exposure limit for NO_2 in the same way as there is for longer ambient exposures. Occupational NO_2 limits vary between countries and determining bodies, and have tended to change with time.

In the UK a limit of 8 ppm was introduced in 2000, reduced to 5 ppm in 2002. The limit was withdrawn in 2003 as it was felt that it did not provide adequate protection, and a new limit is due to be announced soon.

In the US, ACGIH and NIOSH have recommended a short-term exposure limit (STEL) of 5 ppm. This value has subsequently been adopted in many other countries, including Japan, Australia and New Zealand. However, NIOSH also state a recommended exposure limit (REL) of 1 ppm.

6.5 NO_2 limits applied in tunnels

Unlike with CO, the WHO guideline for NO_2 cannot be directly or simply applied to the case of the interior of road tunnels, due to the inappropriateness of the 1-hour averaging time. With the coincidence of new health research data, the NIOSH 1 ppm

REL and the rise of NO_2 emissions relative to CO from road vehicles, increased attention has been paid to NO_2 limits in tunnels.

The Permanent International Association of Road Congresses (PIARC) reviewed the evidence and recommended a limit value for road tunnels of 1 ppm averaged over the length of the tunnel not to be exceeded more than 2 % of the time (PIARC, 2000). This was stated as applying to "healthy people", based on the Folinsbee (1992) study. In order to protect others, PIARC made no recommendation other than to defer this decision to local bodies. No averaging time was specified. It is our opinion that this recommendation was based on minimal health evidence. Nevertheless, the PIARC recommendations are generally the default reference for road tunnel design and are generally adopted in the absence of alternatives. For example, the North-South Bypass Tunnel in Brisbane is Australia's first tunnel to consider an NO_2 limit where the PIARC recommendation will be adopted.

Table 6.1 lists the few alternative NO_2 limits which have been adopted at a national level for road tunnels around the world.

Table 6.1: Various limit values for in-tunnel NO_2 applied around the world

	Limiting concentration	notes
PIARC	1 ppm	proposal
Hong Kong	1 ppm	5 minutes
Norway	0.4 ppm at mid-point 0.8 ppm anywhere 0.22 ppm if pedestrians or cyclists use tunnel	
Sweden, Belgium	0.2 ppm	1 hour, same as WHO ambient guideline
Belgium	0.5 ppm	< 20 minutes
France	0.4 ppm (from 2010)	15 minutes

A review in Sweden (Sandström *et al*., 2003) considered recent experimental studies. In particular, it reports an experimental study of 20 adults with mild asthma sat in a stationary car inside the Söderledstunnel in Stockholm for 30 minutes during peak hour (Svartengren *et al*., 2000). The exposures within the car were described as: NO_2 range 203 – 462 $\mu g\ m^{-3}$, PM_{10} range 103 – 613 $\mu g\ m^{-3}$ and $PM_{2.5}$ range 61 – 218 $\mu g\ m^{-3}$. It may be significant that PM_{10} is much higher than $PM_{2.5}$ indicating the presence of a large concentration of coarse particles. This experiment took place in winter 1997/8, when pollen levels were low, but the widespread use of studded tyres generally leads to a large emission of coarse road dust. Smell and irritant symptoms were reported within the tunnel, but no symptoms from increased airway resistance. Once out of the tunnel an allergen challenge test measured an enhanced response, but there was no statistically significant difference from similar results on a control day. Later in the evening after the tunnel exposure more asthma symptoms were reported than for a control period, using the same subjects but with urban air exposure.

The study concluded:

> "It is thus reasonable to assume that exposure to air pollutants for half an hour in a road tunnel can increase the bronchial response to allergens several hours after the exposure in individuals with allergic asthma. The findings suggest that exposure to car exhaust initiates a pro-inflammatory or inflammatory process in the bronchial mucosa. This state persists for ≥4 h and gives an extra impetus to the allergic reaction with accompanying deterioration of lung function. Such an interpretation, implying a pro-inflammatory effect of exposure to NO_2 and particles, is supported by data obtained in human and animal exposure experiments. If this interpretation is correct, it is also likely that the inflammation increases bronchial responsiveness to not only allergens but also non-specific agents such as cold air and tobacco smoke, as well as exercise. The increase in bronchial responsiveness in the present study occurred without changes in lung function during exposure or in the interval before allergen exposure. There are other studies reporting similar findings with no observed effect on lung function during NO_2 exposure, but which induced an increase in airway responsiveness to agents like histamine, methacholine or ozone. **This makes it difficult for the exposed individual to be aware of the risk."** [*Our emphasis*]

This study presents an interesting and relevant comparison with previous studies of the influence of NO_2 exposure on allergen response conducted in the laboratory in the absence of above-ambient levels of particulate matter. For instance, Strand *et al*. (1998) exposed 16 subjects with mild asthma to 500 $\mu g\ m^{-3}$ of NO_2 followed four hours later by exposure to Birch or Timothy pollen repeatedly over four days. Compared to a control with zero NO_2 exposure, the asthmatic response was significantly increased. This study involved NO_2 exposure duration of 30 minutes each time. The lower NO_2 concentration in the Svartengren tunnel study indicates that a lower NO_2 concentration may elicit the same order of response in the presence of a raised particulate concentration.

In summary, this key tunnel exposure study observed a significant increased allergenic response in asthmatics after exposure to NO_2 at levels > 300 µg m^{-3} (160 ppb) which could be experienced in a single transit of many tunnels around the world. In that study exposure lasted 30 minutes. The differences in the NO_2 limits listed in Table 6.1 appears to relate to the uncertainty regarding the significance of exposures at this level and for this duration, but especially the much shorter exposures (a few minutes) typically experienced in most road tunnels. The Swedish experimental studies were based on very small numbers of subjects, have yet to be repeated, and the implications for development of ill-health are unknown. A report to the Swedish Road Administration argued that a precautionary approach might be to consider an NO_2 limit corresponding to half this concentration (80 ppb) or half this time (15 minutes) (Sandström *et al*., 2003). We have found no evidence of this recommendation having been adopted in Sweden or anywhere else. Nevertheless, it seems plausible that, if such experimental data are corroborated, the argument for such demanding guidelines may become stronger in the future.

6.6 NO_2 limits – options for NZTA

International precedent permits a wide choice when selecting an NO_2 limit. We believe that a value of 5 ppm should be considered too high, as it is above the NIOSH recommended exposure limit for occupational purposes, and has been abandoned by the UK Health & Safety Executive because it gave insufficient protection. A 1 ppm limit, based on the NIOSH REL could be adopted, with the knowledge that such as value has been recommended by PIARC for use in road tunnels. However, whilst it appears that a 1 ppm limit gives sufficient protection to healthy individuals, there is now some evidence that it does not provide full protection for asthmatics and sufferers from chronic obstructive pulmonary disease. To protect such individuals within a road tunnel, a precautionary approach would suggest that a 0.4 ppm limit is appropriate, as adopted in France. We also note that there is some argument for a limit as stringent as

0.2 or 0.1 ppm to provide a factor of safety, although such a limit has yet to be adopted.

On a practical level, we note that a limit of 0.4 ppm may offer the added advantage of keeping NO concentrations sufficiently low to prevent the production of nitrogen dioxide in a tunnel through the termolecular reaction of NO with oxygen (see section 9.1), leading to a more stable control of in-tunnel NO_2. This assumption is based on very limited foreign observations and requires more detailed observational research to be verified.

7. Recent and future trends in vehicle emissions and their implications for road tunnel guidelines

7.1 Overview

- Technological advances over the last few decades have led to a rapid reduction in the average emission of pollutants per vehicle. However, these reductions have not applied to the different components of vehicle exhaust at equal rates. In particular, in the context of road tunnels, reductions in emission of carbon monoxide (CO) have been achieved at a faster rate than reductions in oxides of nitrogen (NO_x). This is significant as changes in the emission ratios between pollutants (particularly the NO_x/CO ratio) changes the ability of one guideline to provide proxy protection against guidelines for other pollutants.

- For the purposes of this report, the UK emissions database was used to analyse trends in emission factors and ratios from the last decade on the assumption that they are indicative of trends in the New Zealand fleet over the next decade.

- Accurate determination of the emissions from heavy duty vehicles is crucially important to the success of an air quality assessment and tunnel design. Unfortunately, data for HDVs are harder to obtain and generally subject to more uncertainty. In New Zealand the situation is made worse by the presence of a large number of diesel HDVs in the fleet built overseas to unknown (or no) emission specifications. Predicting the fleet turnover impact on emissions in New Zealand is much harder due to the lack of knowledge about the emission standard of the existing fleet, and uncertainties regarding the nature of the imported car market.

- The net effect of the changes in NO_x and CO emissions is that in the decade since 1996 fleet-averaged emission factors in the UK have fallen by over 80 % for CO, but only ~ 60 % for NO_x. Crucially for road tunnels this means that the NO_x/CO emission ratio has approximately doubled from 0.25 to 0.5 (or 0.6 to 1.5 for motorways) in one decade.

- The PM_{10}/CO ratio has followed the same general rising trend as the NO_x /CO ratio, whereas the VOC/CO ratio rose initially at the same time only to peak in the early 1990s. The steepest rise in each ratio began in the late 1980s consistent with the widespread penetration of three-way catalysts into the vehicle fleet.

- The data from VEPM indicate that the NZ fleet, in terms of emissions, is approximately a decade behind the UK. It follows that, as newer vehicles, built to Euro II, III and IV or equivalent standards penetrate the New Zealand fleet over the next decade, the fleet-average emission trends in New Zealand are likely to follow similar trends as experienced in the UK over the last decade – although probably not at the same rate. This rate will be dependent upon the rate of fleet turnover, the timing of adoption of new emission standards, and the timeline for improving fuel specifications in New Zealand.

- These general trends describe 'average' traffic on 'average' roads. For any specific road, the emission rates, and the NO_x/CO emission ratios, are sensitive to the fleet mix, driving conditions and vehicle speeds. This is particularly relevant for NO_x because its emissions are dominated by HDVs and highly dependant on vehicle speed, and thus are sensitive to the correct description of the number of HDVs and vehicle speed on any road of interest.

- A rapid increase in the NO_2/NO_x emission ratio has been reported across Europe, probably related to new emission technologies.

- NO_2 'effective emissions' (i.e. concentrations arising directly from vehicle emissions or indirectly from the oxidised emissions of NO) in a road tunnel may still be roughly estimated by assuming an NO_2/NO_x ratio of 0.1, whilst bearing in mind that this ratio is tunnel-specific and may be rising with time. To retain conservatism a ratio of 0.2 should probably be considered. We hope that the uncertainty surrounding this important ratio is reduced in the near future by gathering detailed in-tunnel observations.

- Recent research has lowered the acceptable exposure concentrations for NO_2 during the same period that reductions in NO_x emissions have failed to keep up with faster reductions in CO emission.

- The consequence is that for a given in-tunnel CO concentration the NO_x concentration derived from vehicle emissions has more than doubled over the last three decades. Once NO_2/NO_x ratio rises are considered, the NO_2 concentration has increased even more.

- It can clearly be concluded that whereas a CO limit of 87 ppm alone did provide sufficient protection against the effects of NO_2 in the past, that is not necessarily the case now and increasingly so in the near future.

- An occupational limit of 30 ppm of CO (as an 8 hour average) provides more than adequate protection against the New Zealand Workplace Exposure Standard for benzene.

7.2 The significance of vehicle emission trends and ratios

As will be detailed below, technological advances over the last few decades has led to a rapid reduction in the average emission of pollutants per vehicle. However, these reductions have not applied to the different components of vehicle exhaust at equal rates. In particular, in the context of road tunnels, reductions in emission of carbon monoxide (CO) have been achieved at a faster rate than reductions in oxides of nitrogen (NO_x). We shall express this in the form of the NO_x/CO emission ratio. This allows us to consider CO as a reference species which is appropriate due to the convention to use CO guidelines to provide protection to tunnel users from all vehicle-related pollutants. Recent research has lowered the acceptable exposure concentrations for NO_2 during the same period that reductions in NO_x emissions have failed to keep up with faster reductions in CO emission. Assuming NO_2 represents a fixed proportion of NO_x, the NO_x/CO ratio indicates the NO_2 concentration that would be present in a tunnel for any given CO concentration, once background levels are considered, have increased in time. However, we have also indicated how NO_2/NO_x ratios may be increasing also.

7.3 Vehicle emissions – general features

7.3.1 Variability in emissions

Emissions from vehicles are complex and difficult to quantify. For any given vehicle, emissions will vary with engine and vehicle speed, acceleration, driving style, fuel quality, road surface roughness and gradient and state of maintenance. Variability between vehicles can be attributed to the same causes, but also differences in the emission standard that the vehicle was built to (if any), the vehicle's history and age. Emission databases try to capture the key features of this variability, but it must always be kept in mind that emission databases **at best** represent an average but are rarely able to capture the effect of the most polluting vehicles. The limiting practicalities of measurement also cast doubt over how well emissions data can even represent the average. Consequently, emission factors represent a major cause of uncertainty in all air quality assessments related to traffic emissions, including in road tunnels.

7.3.2 Dynamometer, remote sensing, tunnel, chase and live-tailpipe studies.

There are five key methods for measuring emissions from road vehicles. The core method is the use of dynamometer tests, running vehicles through a typical "drive cycle" and collecting the exhaust stream for analysis with a bank of instruments. This approach has the advantage of a high degree of controllability and the relative ease of direct quantitative measurement of exhaust composition. Most emission databases are based largely or entirely upon dynamometer studies. Dynamometer studies cover a wide range of driving cycles and need to be repeated. Consequently, their major weakness is that they are costly and time-consuming placing practical limitations on the number of vehicles that can be sampled. From these measurements, extrapolations are made to the whole fleet, or to particular scenarios. Recent studies, however, show that such methods tend to under-estimate real-world emissions. This may be due to a number of possible factors such as: not adequately representing a true drive cycle, not estimating emissions properly, or not accounting for all vehicles. However, the main reason is that the bulk of emissions generally come from a small proportion of vehicles known as the "gross emitters" and it is difficult to capture the effect of gross emitters adequately in a selected dynamometer testing programme.

One alternative method is remote sensing (RS), in which vehicle exhaust plumes are sampled optically as they drive past a sensor on a roadside. NIWA has been pioneering the use of RS emissions monitoring in New Zealand and has amassed a significant database. Remote sensing has the advantage of measuring real vehicles as driven by their normal drivers on real roads. This RS monitoring takes less than one second per vehicle and up to 3000 vehicles can be sampled each hour. This compares to approximately 30 minutes to complete a single IM240 set up and test. The open path monitoring is also unobtrusive because there is no physical connection to the vehicle and no specific behaviour is required of the driver. The RS monitoring is therefore very cost effective – typically $2-3 per vehicle.

The weaknesses of this approach are the limited number of pollutants which can be measured and difficulties in verifying the accuracy of the measurements. The methodology assumes all plumes will intersect the beam (not the case for raised exhausts as on some HDVs), and that the measurement site allows observation of a vehicle sample locally (or nationally) representative of the fleet (e.g. not biased towards one type of vehicle). One measurement site is also unlikely to allow thorough investigation of the effect of speed, acceleration, gradient, road surface, etc. The RSD measures a vehicle's emissions at one point in time (generally under slight acceleration) and provides little detail on how the emissions from an individual may vary under differing driving conditions. The monitoring sites used in NIWA campaigns were single lane on- or off-ramps, or one way streets. For this reason the emissions monitored will reflect driving conditions that predominate on these types of roadway and will not be entirely representative of emissions generated on other roadway types, e.g. at busy intersections.

A tunnel presents an opportunity for measuring average emissions from the fleet that use it. This is because the tunnel acts as a controlled dilution chamber. If the airflow through the tunnel is accurately measured then the increase in concentration of any substance between the entrance and the exit of the tunnel (assuming no air enters or exits the tunnel elsewhere, or those separate ports are monitored) can be directly related to the rate of emission from vehicles within the tunnel. This approach is subject to some uncertainties regarding the representativeness of measurements of airflow and concentration. It also requires detailed data regarding the ventilation flow rate and the traffic using the tunnel (type, speed, maybe origin). The data obtained are specific to the tunnel, i.e. specific to the vehicle fleet that use it and the fuel that fleet uses, whether it has gradients and what the traffic conditions are like. This approach works best when these issues are well-constrained (e.g. no congestion at any time, relatively constant vehicle speed, constant ventilation rate). Tunnel emission studies are well-suited to studies of long-term emission trends where a campaign is repeated some years after an initial one so that changes can be observed.

Two further approaches to characterising emissions are chase and live-tailpipe studies, although data from these techniques have not been used in this Report. Concerns over capturing the real difference between dynamometer drive cycles and the way real people drive real vehicles can be partly addressed by chase studies. An instrumented vehicle literally chases a target vehicle sampling its exhaust plume. This has the advantage of being able to collect a great deal of data about individual vehicles, but is limited in the number of vehicles that can be sampled and is not as easily repeatable as the other methods. This method is principally used in the research community but the data are generally considered insufficiently robust to be included in general emission databases. A variation on the chase study is for a vehicle to be instrumented to sample its own exhaust. Such systems are also generally used only as research systems. However, this research is increasingly being used to complement dynamometer data and to generate more representative emission databases.

7.3.3 Gross emitters and heavy-duty vehicles

Generally, an accurate prediction of vehicle emissions from any given road is dependent upon the correct description of emissions from *gross emitters* – the few vehicles that emit the most pollution.

Figure 7.1 shows the contributions of the vehicle fleet as observed in a NIWA remote sensing campaign, to the cumulative total emissions. The measured vehicles are sorted by descending emissions. The cumulative total emissions (the vertical axis) is normalised to the total emissions from all the measured vehicles. The skewed distribution is demonstrated by the curvature of the line (an even distribution would lead to a straight line, and a normal distribution to symmetrical curve).

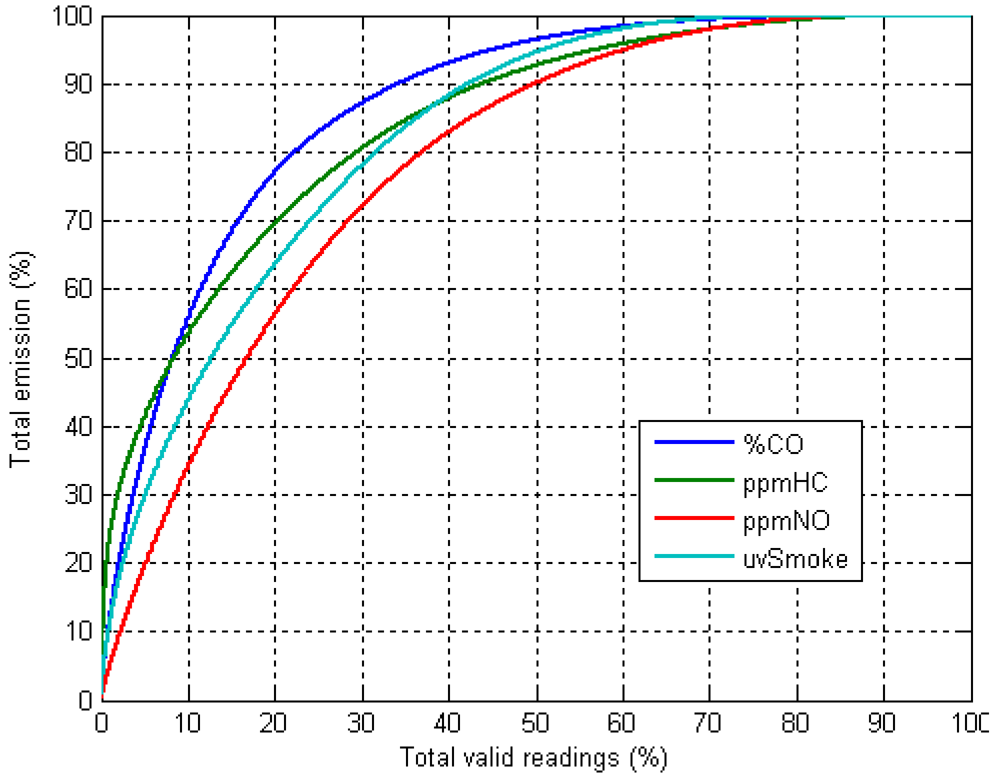

Figure 7.1: Relative fleet cumulative contribution to total emissions. Source: Bluett and Dey, 2006. On-Road Measurement of Vehicle Emissions in the Wellington Region. NIWA report number CHC2007-123.

Figure 7.1 indicates that 'most polluting' 10 % vehicles are responsible for about 35 % of NO emissions, 45 % of uvSmoke emissions and about 55 % of CO and hydrocarbon (HC) emissions. On the other hand, the cleanest 50 % of the fleet contribute only between 5 % and 10 % of the total emissions for each pollutant.

Between vehicle classes heavy-duty[6] diesel vehicles (HDVs) may be considered gross emitters. Accurate determination of their emission factors is crucially important to the success of an air quality assessment and tunnel design. Unfortunately, data for HDVs are harder to obtain and generally subject to more uncertainty. This is due to any sub-class of the fleet representing a small sample of vehicles, leading to inherently larger statistical uncertainties. Dynamometer testing of HDVs requires a specialised installation which is less common, and HDV testing is more expensive than car or van

[6] Heavy-duty refers to vehicles with a gross vehicle mass greater than 3,500 kg

testing. Consequently, less HDV dynamometer data exist, further increasing uncertainties. In New Zealand the situation is made worse by the presence of a large number of diesel HDVs in the fleet built overseas to unknown (or no) emission specifications.

7.3.4 Emissions in congestion

The nature of real-world vehicle emissions in congested traffic is a developing area of research. Existing emission databases are generally too crude to predict emissions in congestion, partly because it is unclear which measure of congestion to use, and a general lack of emission factors for congested driving. European databases (more detail below) indicate that emissions of some pollutants from vehicles are highly sensitive to speed, such that the net emission on a congested road is dependent on the proportion of these vehicles on the road. However, emissions at low speed are as much related to acceleration as speed itself, and existing databases do not properly distinguish between congested driving which is smooth, and that incorporating stop-start driving. Recent and future research data are intended to provide some further insights, but care must be taken over such data as they are inevitably location-, and thus fleet-specific.

7.3.5 Official emission inventories in EU and US.

Globally, the most detailed emission databases come from the European Union, especially the United Kingdom, and the United States.

The UK national database (www.naei.org.uk) provides vehicle emissions for 7 vehicle classes, and for each of the prevalent emission standards (pre-Euro, Euro I, Euro II, Euro III and Euro IV). Data for cars are also broken down by engine size (2 classes for diesel and 3 for cars). Formulae are then provided for each of these sub-classes to derive emission factors for CO, NO_x, PM and other pollutants as a function of vehicle speed. Cold-start weightings are also provided as well as factors for calculating emissions from previous years and forecasting to future years to account for age and maintenance factors and changes in fuel quality. All this builds up to a formidable database of 85 emission factors per contaminant per year per speed. Implementation of the database requires equally detailed data on the prevalence of each of the 85 sub-class of vehicles on the roads now, and in the future if forecasts are required.

For the purposes of this report, the great value of the UK database is that it allows the investigation of trends which may be compared to those in New Zealand. Much of the data that are commonly used in New Zealand for emissions assessment and prediction are based upon the UK database.

7.3.6 NZTER and VEPM

The New Zealand Traffic Emission Rates database is an output of the Ministry of Transport's Vehicle Fleet Emission Model (VFEM). This model was published as part of the Ministry's Vehicle Fleet Emission Control Strategy in 1998. It provides emission factors of CO, NO_x, PM and VOCs for a road as a function of a given fleet mix, for four classes of road (Central Urban, Suburban, Rural Highway and Motorway) and for five driving conditions (Idle, Cold Start, Free Flow, Interrupted and Congested). It also provides estimates for years up to 2021 based on expected technological improvements. There is no explicit consideration of speed. It is the primary vehicle emissions modelling tool used in NZ over the last 5 years.

Auckland Regional Council has recently developed a Vehicle Emissions Prediction Model (VEPM). This model is intended to supersede NZTER. VEPM is based upon a combination of EU, Japanese and Australian databases. It provides year-specific emissions for CO, CO_2, NO_x, PM and VOCs as a function of a particular fleet mix and speed.

Intercomparison of these databases with each other and external sources of data, including NIWA's remote sensing data, will be covered in a forthcoming NIWA Report. At this point we will merely note that VEPM emission factors are generally lower than in NZTER, especially for NO_x emissions from heavy duty vehicles. A preliminary NIWA remote sensing campaign indicated that the NZTER HCV NO_x factors, especially pre-1999, are probably over-estimates (Bluett & Fisher, 2004). NIWA intend to conduct further research in the near future to refine these data and provide further detail.

7.3.7 Non-tailpipe emissions

Vehicle emissions include not only emissions from the tailpipe, but also evaporative emissions and resuspended road dust. Evaporative emissions are primarily associated with volatile organic compounds, but the quantities emitted are low compared to tailpipe emissions. For example, McLaren *et al.* (1996) decomposed emissions of non-methane hydrocarbons (NMHC) into direct tailpipe and evaporative emissions in the Cassiar tunnel near Vancouver. For benzene, 71 % of the measured tunnel concentration was related to combustion emissions, 27 % to unburned fuel and 2 % to evaporative losses. Emissions of evaporative emissions are poorly quantified and not included in this analysis due to our focus on gross trends. Similarly, emissions of resuspended road dust are poorly constrained and an active area of current research. Furthermore, the impacts of such dust on health are disputed and difficult to quantify, especially in the context of road tunnel exposure, and as such they are not included in this analysis.

7.3.8 Technological innovation in emission control

Improvements in engine and exhaust technology have led to dramatic changes in vehicle emissions in the last two decades. The impact of these changes is most clearly seen in the change in the trend of total emissions from all vehicles. Figures 7.5 – 7.7 show that the rising trend in total emissions until the 1990's has turned to a falling trend since then, despite continual rises in traffic flow (figure 7.2).

In Europe, Japan and the United States a series of emission standards have been adopted which have been highly successful in reducing the net emission from vehicles. Each standard has been made possible by the development of one or more new technologies. The Euro I standard for petrol engines (introduced 1994), for instance, was largely made possible by the introduction of three-way catalytic converters. Further improvements were achieved, apparent in vehicles built in the 1990s, by a combination of electronic management, improved fuel quality and lean-burn technology, all aimed at improving the efficiency of combustion and thus reducing emissions of volatile organic compounds, soot and carbon monoxide. The Euro IV standard (introduced in 2005 for HDVs) required a large cut in PM emissions from heavy duty vehicles from 100 mg/kWh (Euro III) to 20 mg/kWh. Similarly in 2005 the Japanese standard for HDVs fell from 180 mg/kWh to 27 mg/kWh. These reductions have largely been achieved through the fitting of particulate traps to tailpipes, but the reduction is dependent upon the use of ultra low sulphur diesel (ULSD). From September 2009 Euro V and the corresponding Japanese standard will ensure a similarly large reduction in PM for diesel cars, both requiring a limit of 5 mg/kWh.

Reductions in NO_x emissions have also been achieved, but at a slower rate than PM. Exhaust Gas Recirculation (EGR) reduces NO_x emission by reducing the temperature of combustion. The introduction of ULSD has enabled the additional use of NO_x catalytic reduction by burning extra fuel in a NO_x trap. Japanese standards for NO_x from HDVs have been consistently ahead of European standards by about 3 years. The 2005 Japanese standard for heavy-duty diesels of 2 mg/kWh, equivalent to the forthcoming Euro V standard (from 2008), is met mostly by the introduction of Selective Catalytic Reduction (SCR). This technology offers a similar dramatic reduction in NO_x emissions as particulate traps offer for PM reduction, however with the shortcoming of an increase in ammonia emissions. A Japanese standard of 0.7 g/kWh for HDVs and 0.08 g/kWH for diesel and petrol cars will be introduced in 2009. European petrol cars already meet this standard (via Euro IV) whereas a Euro VI standard is also proposed (effective from 2013 for HDVs and 2014 for LDVs) which will require further NO_x reductions from diesel vehicles to below the 2009 Japanese standards (0.4 g/kWH for HDVs and 0.06 g/kWh for cars).

7.3.9 The significance of emission trends for road tunnels

Due to the use of CO as a proxy for all traffic pollutants, the change in CO emission relative to other traffic pollutants is of crucial importance to road tunnels. Specifically, due to concerns noted above about the effects of NO_2, we are most interested in changes to the NO_x/CO emission ratio. Although most NO_2 arises from the rapid conversion of NO to NO_2, some NO_2 is directly emitted. Data regarding this partitioning between NO and NO_2 in the tailpipe are rare, and emissions data are usually reported as NO_x (the sum of NO and NO_2). The significance of assumptions regarding NO_x partitioning is covered in more detail below.

7.4 Emission trends in UK and EU fleets

7.4.1 Traffic and vehicles in the UK and NZ – similarities and differences

Due to the European Auto-Oil Programme, and equivalent measures in the US, and a higher vehicle turnover rate, we consider the hypothesis that vehicle emissions have fallen in advance in Europe and the US compared to New Zealand. Consequently, past trends in these countries may be seen as indicative of possible trends in New Zealand into the future, although differences in the New Zealand fleet must be borne in mind.

Firstly, figure 7.2 illustrates that traffic rates (VKT) have been growing continuously and remarkably uniformly in the EU, US, Australia and New Zealand, at a rate of around 2 % per year. Figure 7.3 shows that the total number of cars registered grew faster in the UK up to the 1980s, but the rate of growth has been similar in both countries in the last two decades. Of particular interest are the differences in the fleet composition between New Zealand and the UK. For example, 22 % of the UK cars in 2006 were diesel powered, compared to only 8.5 % of New Zealand cars (LTNZ, 2007). Furthermore, the average vehicle engine size in New Zealand in 2006 was 2.2 litres (MfE, 2007) and the distribution of engine sizes is clearly biased towards larger sizes in New Zealand compared to the UK (figure 7.4).

This has implications for comparing emission databases as the UK database reports emission factors for three size classes of petrol car: < 1.4 litres, 1.4 – 2.0 litres and > 2.0 litres. Whereas approximately half of all cars in both countries fit in the middle range, the larger category accounts for only 13 % of the UK fleet but almost 40 % of the NZ fleet, including a very significant proportion of vehicles over 3 litres. In terms of emissions, the relatively high emissions from these large engine vehicles may be under-represented in the UK database. This is relevant because the VEPM emission factors are based upon its UK counterpart and specified for UK-relevant engine size classes which are then mapped onto the New Zealand fleet, leading to the possibility of it under-representing the emissions of large petrol cars.

(UK data from Transport Statistics 2007, Department of Transport. NZ data from Land Transport New Zealand.)

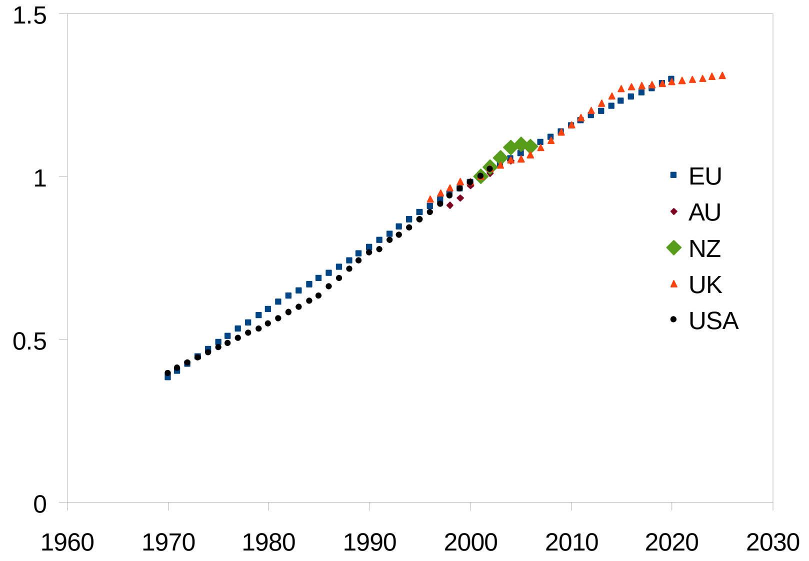

Figure 7.2: Total vehicle kilometres travelled (VKT) in 5 regions (normalised to year 2001).

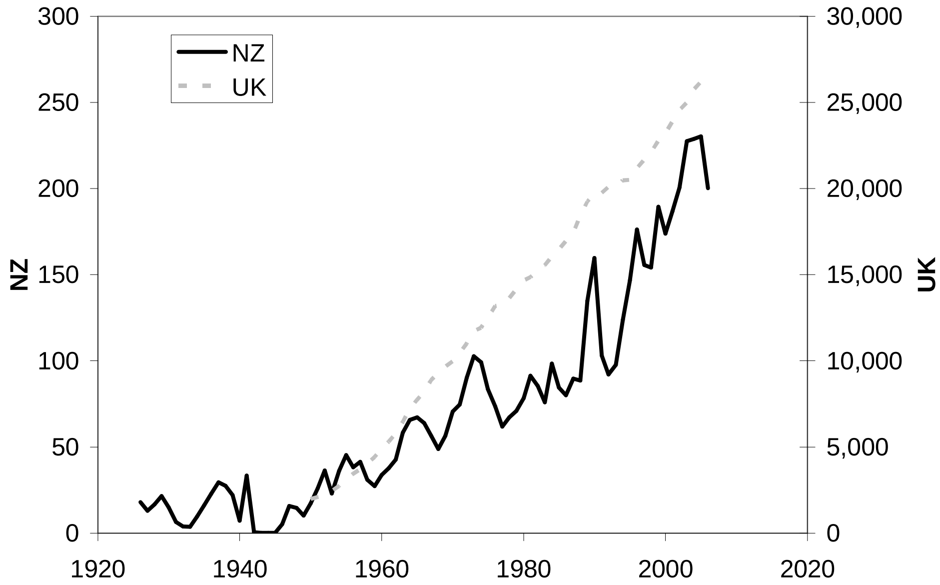

Figure 7.3: Number of cars registered (thousands) in New Zealand and the UK by year (LTNZ, 2007, DfT, 2007).

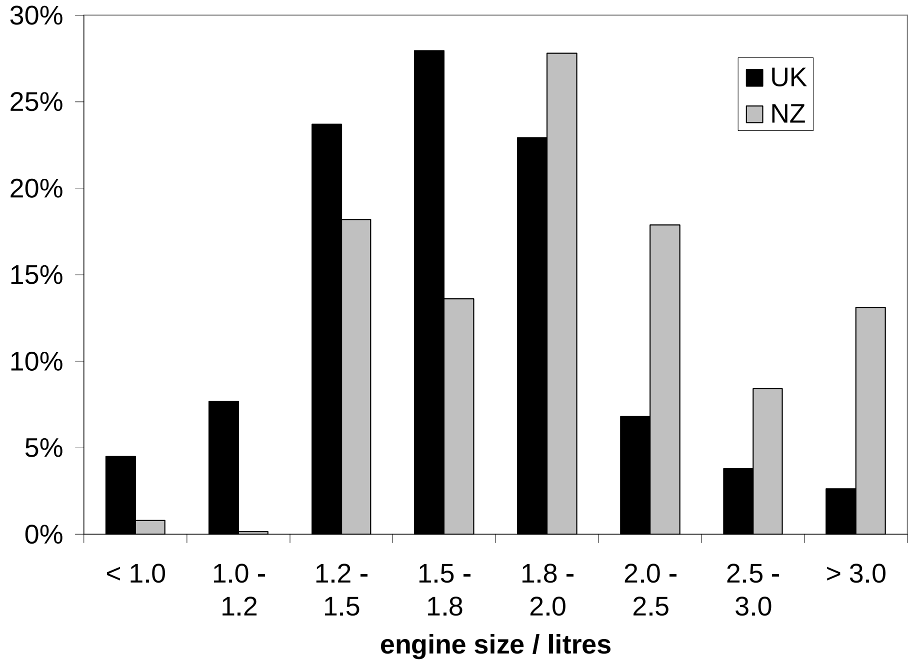

Figure 7.4: Distribution of car engine sizes in the UK and New Zealand in 2006 (DfT, 2007, MoT, 2007).

7.4.2 Fleet turnover

Key factors in the trends of emissions are the rate at which people replace older vehicles with newer ones, where they source those newer vehicles from and the emission standards of those new vehicles. These factors are influenced by economic factors (personal wealth, the cost of new, used and imported vehicles) and cultural factors. Amongst the EU15[7], the average vehicle age is approximately 7 years (Eurostat, 2003) compared to 12 years in New Zealand (MfE, 2007). Studies by NIWA found median ages of vehicles monitored on the road side to 9.2 years in Auckland and 9.3 years in Wellington (Bluett & Dey, 2006).

There are a number of reasons that may explain why the monitored fleet is significantly newer than the average age of the vehicle fleet. Newer vehicles tend to be

[7] The EU15 are those states which were members of the European Union before its expansion on 1st May 2004. They are: Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, Netherlands, Portugal, Spain, Sweden and the United Kingdom.

driven a greater number of kilometres each year than do older vehicles. Hence they have a greater chance of driving through a monitoring site than older vehicles that do less kilometres. Another possible explanation is that the two monitoring campaigns were undertaken in cities, where the average age of vehicles may be lower than the country wide average that includes smaller towns and rural areas, where older vehicles may be more commonly used.

Figure 7.5 shows the projected number of cars meeting the Euro standards in the UK and its evolution as more new vehicles complying with more recent standards penetrate the market.

The result of this fleet turnover in the UK, and similarly across the EU and the US is that total emissions of traffic pollutants have fallen, despite the increase in traffic, as shown in figures 7.6 and 7.7. Predicting the fleet turnover impact on emissions in New Zealand is much harder due to the lack of knowledge about the emission standard of the existing fleet, and uncertainties regarding the nature of the imported car market.

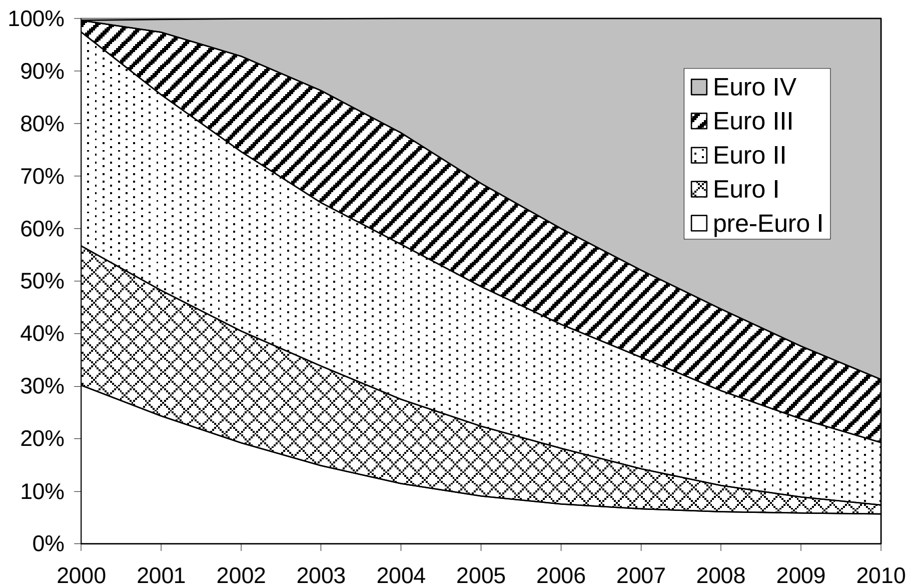

Figure 7.5: Predicted evolution of composition of UK car fleet by emission standard over this decade as a result of fleet turnover (assumes no new standard introduced beyond Euro IV). (Data: UK Department for Transport).

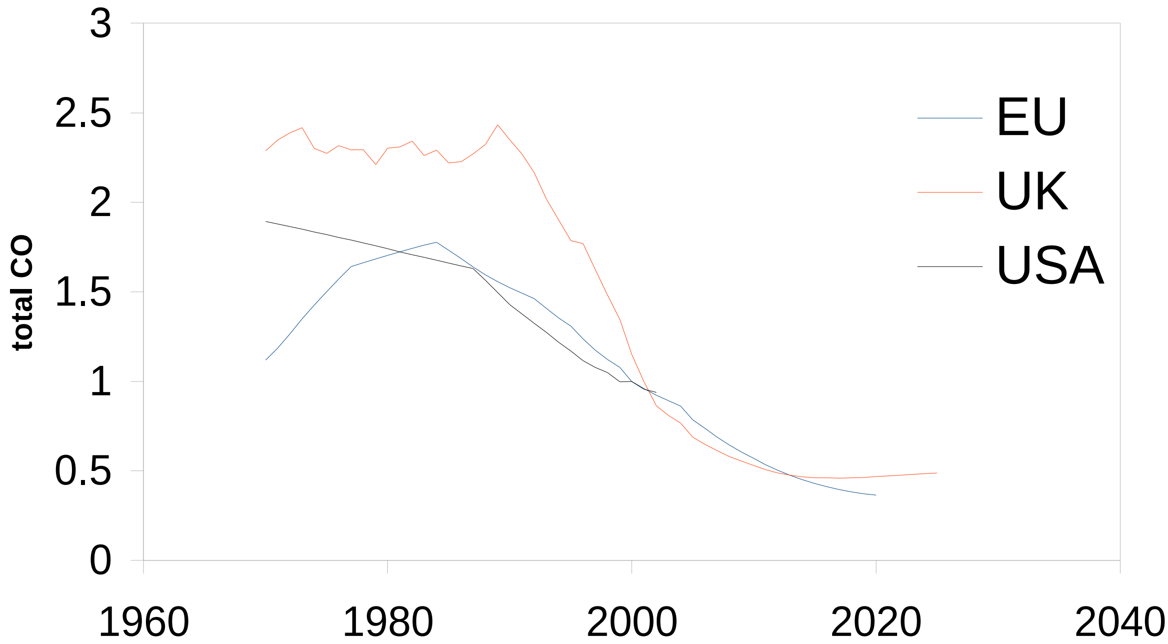

Figure 7.6. Total CO emissions from road transport, normalised by year 2001.

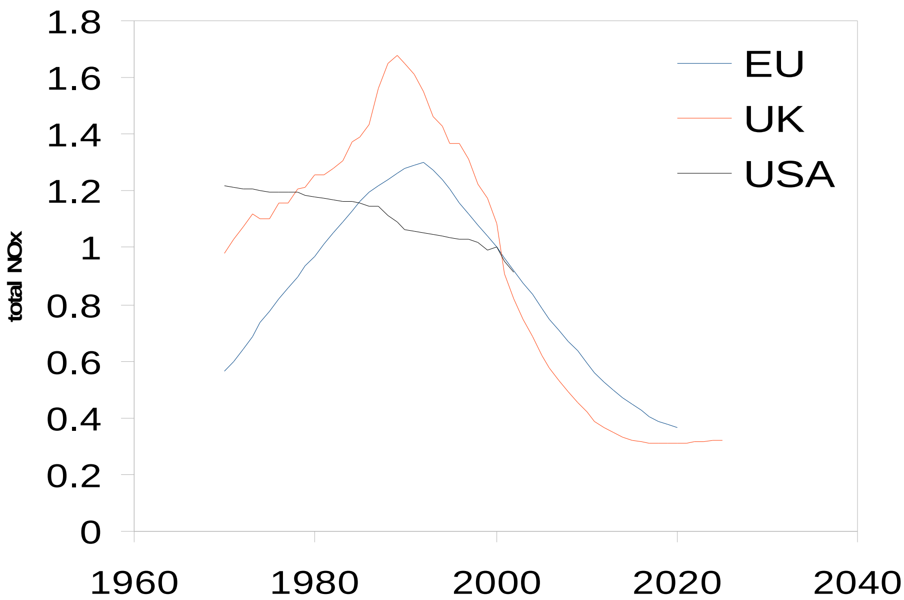

Figure 7.7 Total NO_x emissions from road transport, normalised by year 2001.

The staggered introduction of emission standards in Europe, which primarily tackled the products of incomplete combustion first, led to a reduction in carbon monoxide in advance of reductions in NO_x and PM, as shown in figure 7.8.

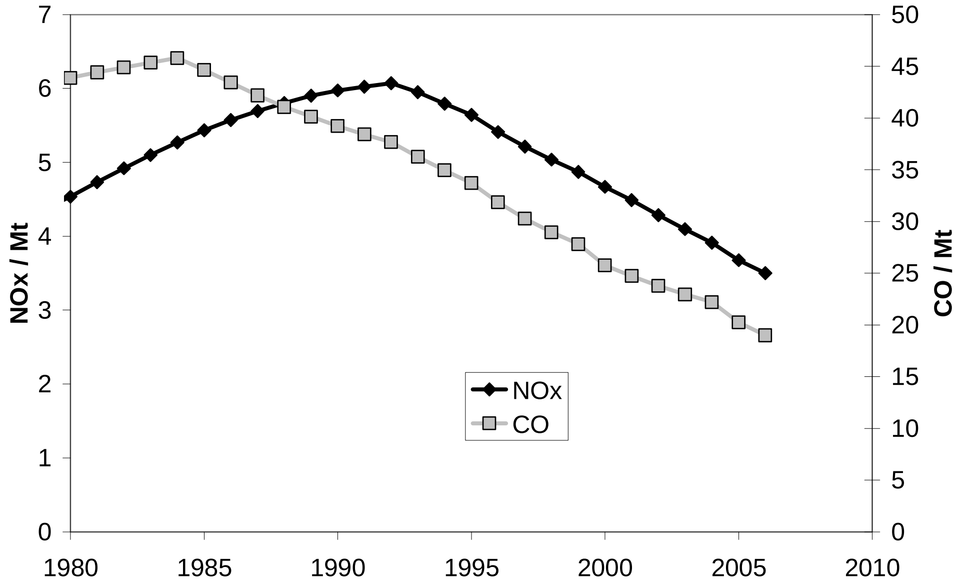

Figure 7.8: Total CO and NO_x emissions from road transport in the EU15.

7.4.3 Major trends in UK emission factors

Carbon monoxide

In 1996, petrol cars were responsible for 88 % of CO emissions on urban roads and 75 % on motorways. In the 10 years from 1996 to 2006, the UK national emission database reports that there have been major reductions (80 %+) in the emission factors of CO from petrol cars, LGVs and buses, explaining the reduction in total emissions in this period. However, in 2006, petrol cars still contributed similar proportions to total CO emissions from roads as 10 years before, due to their being the dominant source of CO.

Oxides of nitrogen

The evolution of NO_x emissions is more complex. Emission factors for heavy duty vehicles (in 1996 and now) are an order of magnitude higher than for light-duty vehicles. In 1996 the higher number of cars meant that on urban roads petrol cars made a higher contribution (56 %) to total road NO_x emissions than HGVs (22%).[8] In the following 10 years NO_x emissions from petrol cars fell by ~ 80 %, but only by 40 – 50 % from HGVs (Figure 7.9). The net result was that by 2006 petrol cars and HGVs are reported to have a similar fleet-weighted average emission per kilometre travelled on urban roads (Figure 7.9).

Further, as NO_x emissions are greater at higher speeds, the shift to HGV dominance was advanced on motorways. In fact, HGVs NO_x emissions already exceeded those from cars in 1996, with HGVs contributing approximately one third of all motorway NO_x emissions by 2006.

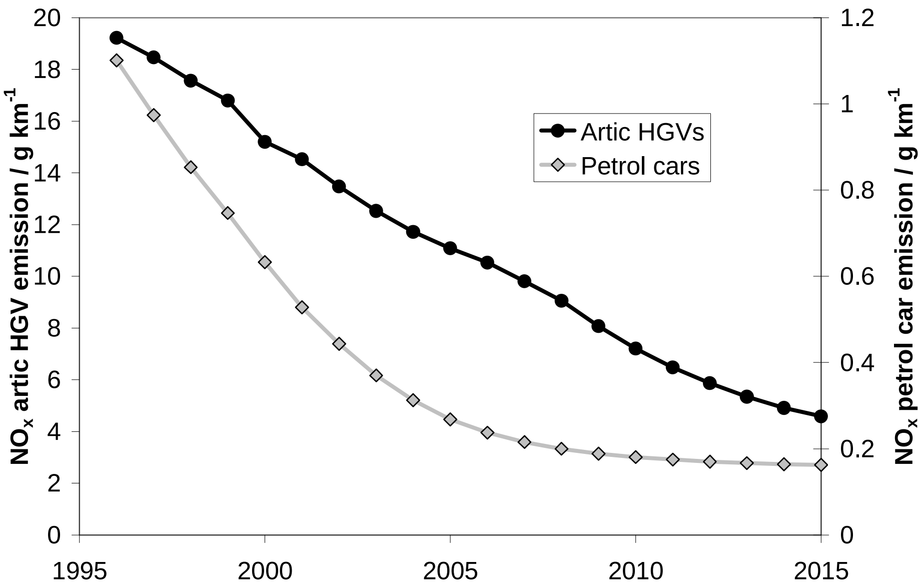

Figure 7.9: UK emission factors for NO_x for the two largest contributors to total NO_x emissions ("Artic HGVs" = articulated heavy goods vehicles).

[8] The remaining 22 % was due to buses (11 %), light goods vehicles (8 %) and diesel cars (3 %).

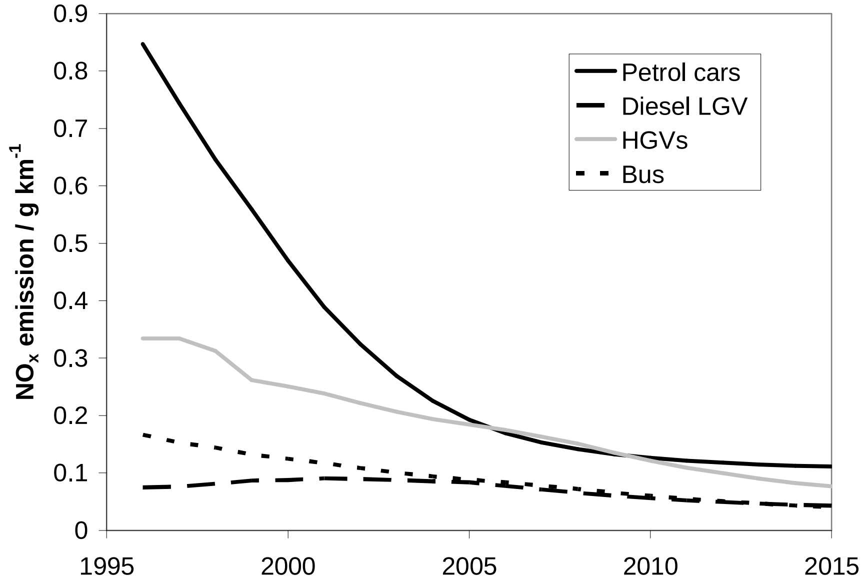

Figure 7.10: Evolution of the contribution to total NO_x emissions on central urban roads (UK) of the four highest contributing vehicle classes.

NO_x/CO emission ratio

The net effect of the changes in NO_x and CO emissions is that in the decade since 1996 fleet-averaged emission factors in the UK have fallen by over 80 % for CO, but only ~ 60 % for NO_x (Figure 7.11). Crucially for road tunnels this means that the NO_x/CO emission ratio has approximately doubled from 0.25 to 0.5 (or 0.6 to 1.5 for motorways) in one decade (figure 7.12).

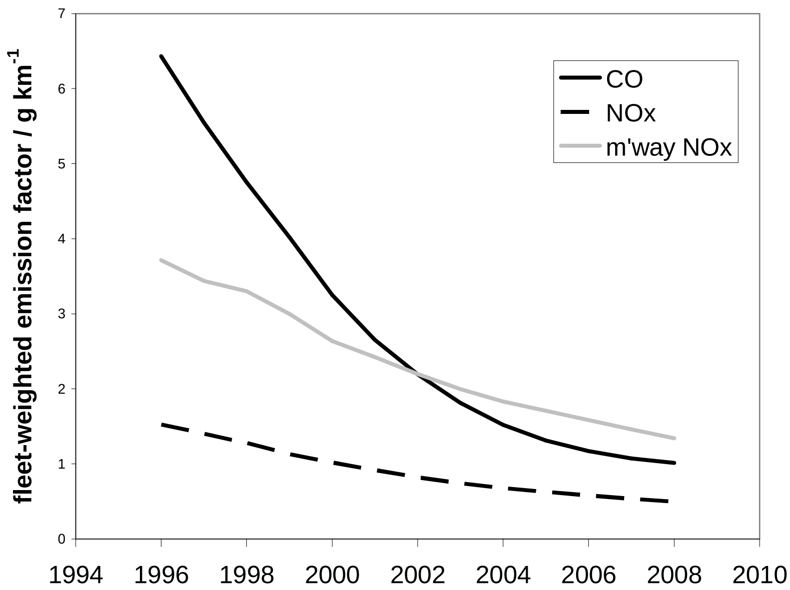

Figure 7.11 Evolution of fleet-weighted CO and NO_x emission factors on UK urban roads and motorways (Motorway data for CO is not shown as it is very similar to urban road data).

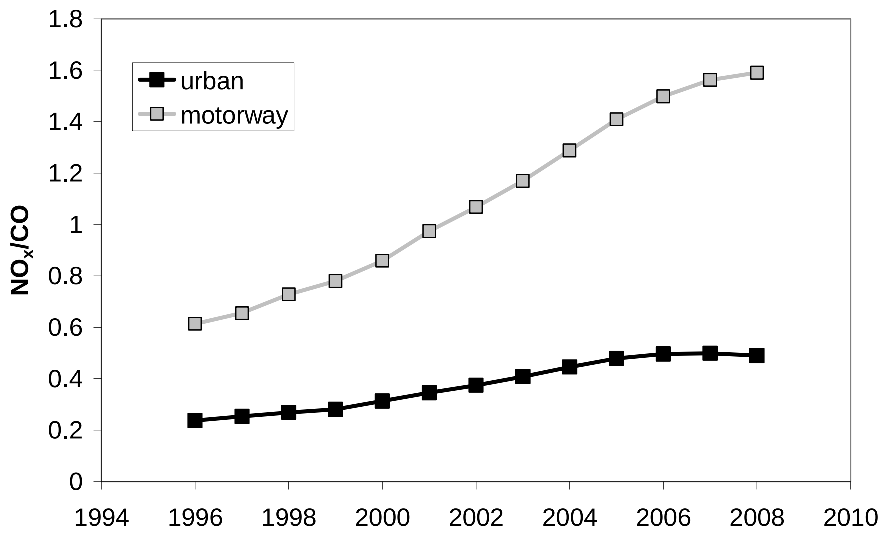

Figure 7.12 Evolution of UK fleet-averaged NO_x/CO emission ratio for urban roads and motorways.

7.4.4 Other emissions and broader European and American trends

Similarly to what the UK NO_X/CO emission ratio shows, the emissions of other traffic related pollutants have changed at different rates than CO emissions. Data from the European Union show that despite continuous increases in total vehicle kilometres travelled, the **total** emission from road traffic peaked in 1984 for CO, 1989 for VOCs, 1992 for NO_x and 1993 for PM_{10}. We can divide total emissions by total vehicle-kilometres travelled (VKT) to derive an estimate of the average total emissions per kilometre driven across the whole EU15. For this measure emissions (per VKT) peaked for VOCs in 1976, for CO in 1977 for NO_x in 1985 and for PM_{10} in 1992. I.e. any rise in total emissions since these dates have been caused by rises in traffic rather than emission rates per vehicle. From the early 1995 to 2005, total emissions per vkt in the EU15 have fallen fastest for VOCs (average 9 % per year), followed by CO (7 % per year), NO_x (6 % per year) and PM_{10} (5 % per year). The consequence for emission ratios is shown in figure 7.13. In general it can be seen that the PM_{10}/CO ratio has followed the same general rising trend as the NO_x /CO ratio, whereas the VOC/CO ratio rose initially at the same time only to peak in the early 1990s. The steepest rise in each ratio began in the late 1980s consistent with the widespread penetration of three-way catalysts into the vehicle fleet which aim to convert CO and hydrocarbons into carbon dioxide and water. By the end of this decade these data indicate that VOC emission reductions will have caught up with CO reductions such that the VOC/CO ratio will have returned to the kind of values apparent in the 1970s, but that the NO_x /CO and PM_{10}/CO ratios, whilst levelling off, will have substantially increased since the 1970s.

The situation is more uncertain when considering hydrocarbons. Several tunnel studies from the US show that the NMHC/CO emission ratio increased by 60 % between the early 1990’s and the early 2000’s (Gertler and Pierson, 1996; Sagebiel *et al*., 1996; Gertler *et al*., 1997; Kirschstetter *et al*, 1999; Kean *et al*, 2000 and McGaughey *et al*, 2004). For total HCs, results from a large scale remote sensing campaign in several cities in the US show that the emission ratio increased more than 100 % in the same period, indication that methane emissions have not been reduced in the same way as other hydrocarbons. However, both ratios (HC/CO and NMHC/CO) show a very large scatter in time and the indicated trends are indicative only (figures 7.14 and 7.15).

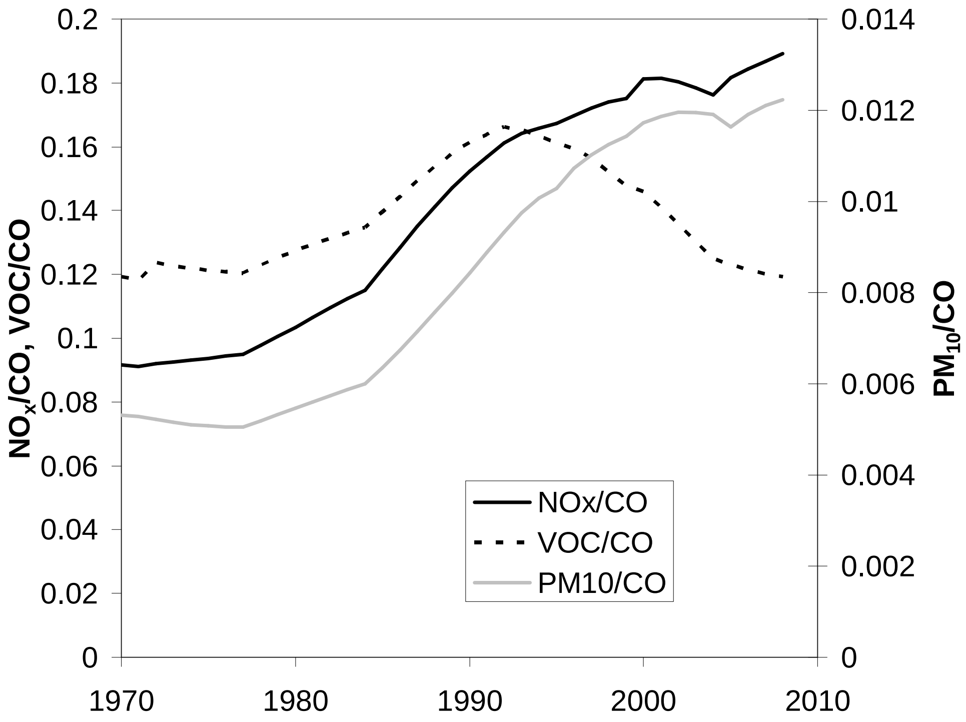

Figure 7.13 Trends in ratios of total emissions from road vehicles per total vkt in the EU15.

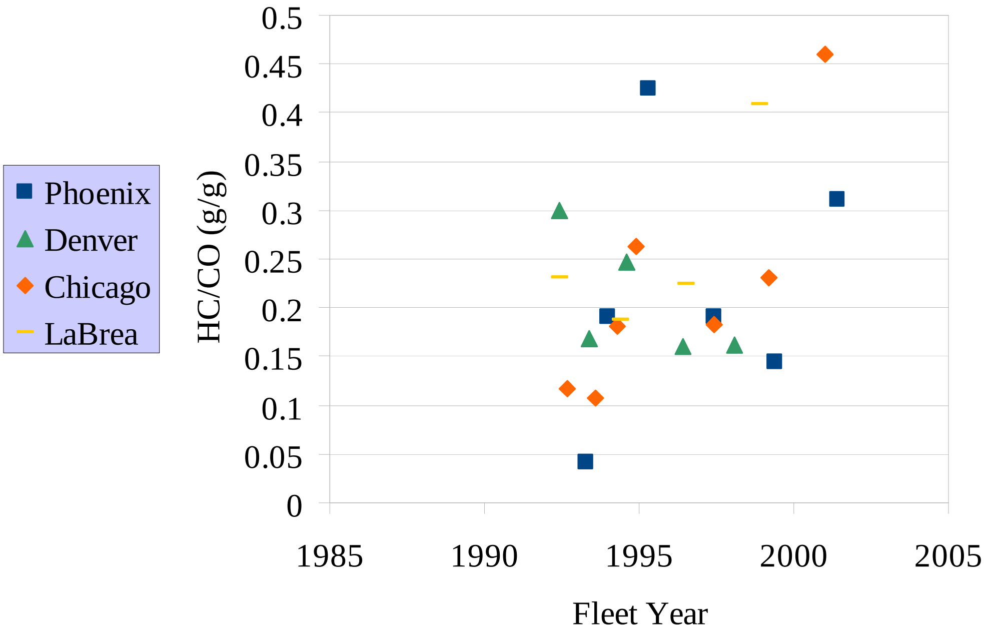

Figure 7.14 Evolution of fleet-averaged HC/CO emission ratio from remote sensing studies in several cities in the US.

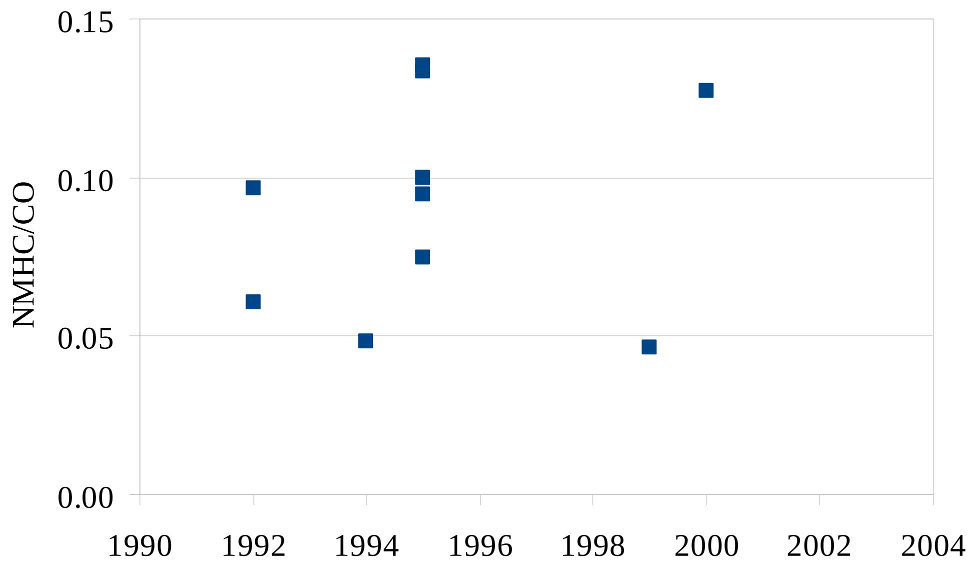

Figure 7.15 Evolution of fleet-averaged NMHC/CO emission ratio from tunnel studies in the US.

Long-term reductions in measured emission factors have been reported where measurements have been repeated in the same tunnel some years apart. Schmid *et al.* (2001) reported major reductions of emission factors measured in the Tauerntunnel in Austria between 1988 and 1997 (see figure 7.16), including an 85 – 87 % fall in CO emissions per km. Stemmler *et al.* (2005) reported falls in benzene and toluene emission per km in the Gubristtunnel (Zurich) of 80 % and 76 % respectively between 1993 and 2002.

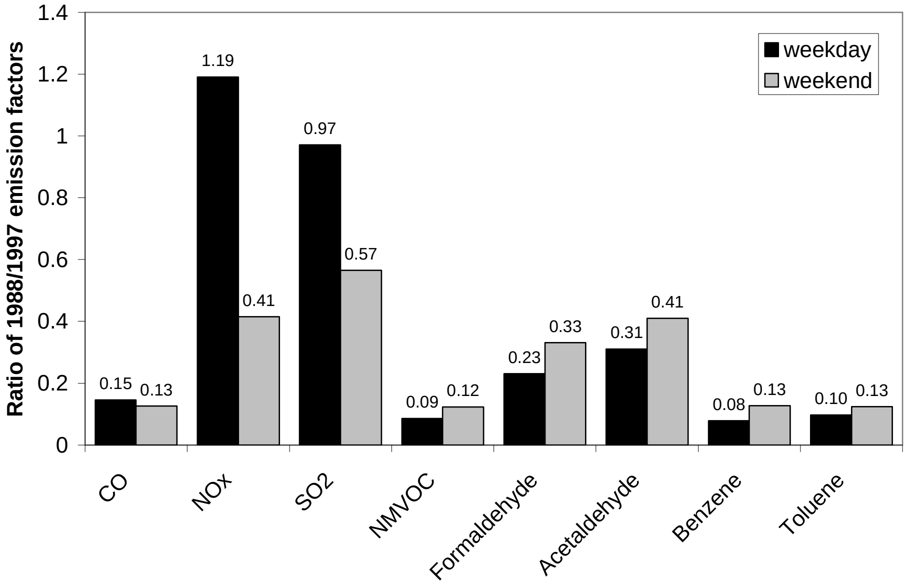

Figure 7.16: Changes in road tunnel emission factors between 1988 and 1997 in the Tauerntunnel, Austria (from Schmid *et al.*, 2001).

7.5 New Zealand trends

The relative lack of data regarding the New Zealand fleet, its origin and emission standards of individual vehicles make tracing such trends as discussed above for the UK and EU15, much more difficult and uncertain. We shall illustrate similar trends as identified in the NZTER emission database but the shortcomings of that database must be borne in mind.

By comparing emission factors for the respective UK and NZ fleets it is possible to crudely estimate "how far behind" the NZ fleet is in terms of emissions. Using NZTER 'free flow', 'central urban' factors and comparing with UK and VEPM factors for a speed of 40 km h^{-1} we find the following equivalences:

- An average NZ petrol car in 2008 has equivalent CO emissions to an average UK petrol car in 1998 (using NZTER) or 1996 (using VEPM).

- An average NZ petrol car in 2008 has equivalent NO_x emissions to an average UK petrol car in 1999-2000 (NZTER) or 2001-2002 (VEPM).

A similar comparison cannot be given for HGV emissions. This is because each database classifies heavy duty vehicles differently (e.g. VEPM and NZTER by weight, but in different bands, and the UK by articulated or rigid) and with greater emissions arising from fewer vehicles estimation of fleet averages are more uncertain. Also, NZTER's predictions of NO_x emissions from HCVs are several times higher than those from VEPM.

We must reiterate that this is a crude assessment and is highly dependent upon the accuracy of both databases. In general, the data from both NZTER and VEPM indicate that the NZ fleet, in terms of emissions, is approximately a decade behind the UK. It follows that, as newer vehicles, built to Euro II, III and IV or equivalent standards penetrate the New Zealand fleet over the next decade, the fleet-average emission trends in New Zealand are likely to follow similar trends as experienced in the UK over the last decade – although probably not at the same rate. This rate will be dependent upon the rate of fleet turnover, the timing of adoption of new emission standards, and the timeline for improving fuel specifications in New Zealand.

For example, new emission standards in New Zealand will require that by 2009 all new and newly imported used vehicles to be compliant with Euro IV or equivalent standards[9]. This represents an attempt to accelerate emissions reduction in New Zealand as the equivalent introduction of Euro IV would be only four years after its introduction in Europe in 2005.

Despite NZTER's weaknesses, it can still be used to trace recent trends in CO and NO_x emissions. Based on a simplified fleet mix assumption (80 % petrol cars, 15 % diesel LCVs and 5 % large HCVs) and using 'free flow' factors NZTER reports a decrease in fleet-averaged emission factor by three-quarters since 1980 for CO and by a half for NO_x. This means that the NO_x/CO ratio has doubled in the same time frame, with an upward trend expected to continue over the next decade (Figure 7.17). Predictions of absolute values should not be relied upon, due to the errors inherent in our simplified fleet assumption, the inaccuracies in NZTER and especially the known over-prediction of NZTER for NO_x emissions from HDVs.

Running a similar prediction with VEPM (Figure 7.18) results in lower absolute values of the NO_x/CO ratio, but similar trends with growth in the ratio peaking around 2013 – 2014. NIWA's initial remote-sensing campaigns (Bluett & Fisher, 2004)

[9] The exception being used light petrol vehicles for which a Euro III standard will be required from 2009 and Euro IV from 2012.

indicated that NZTER over-estimates CO emissions from petrol cars manufactured before 2002, with the error increasing for older vehicles, whilst also suggesting that NZTER significantly over-estimates NO_x emissions from heavy duty vehicles. The consequence for the tend in the NO_x/CO ratio is similarly to predict lower absolute values, especially in the past, but the same general increasing trend with a potentially more rapid increase in recent years.

More recent remote-sensing data captured by NIWA has cast some doubt as to whether absolute NO_x emission factors are falling in New Zealand, as suggested in the UK and EU databases, and in NZTER. This is indicated by results from sampling of NO emissions only from petrol vehicles in 2003 and 2005 during which time a significant rise in NO emissions was observed (Kuschel & Bluett, 2007). At the time of writing it is unclear if this was a 'real' increase or an artefact of the experimental design or instrumentation.

Intercomparison between emission factors, and those derived from remote sensing campaigns in New Zealand, will be treated in more depth in a forthcoming NIWA report.

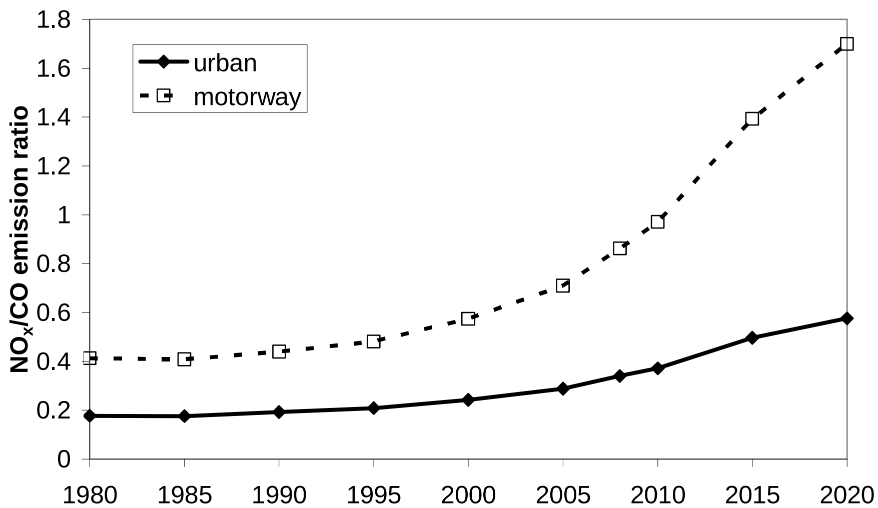

Figure 7.17: Trend in NO_x/CO fleet-averaged emission ratio on New Zealand roads (assuming 'free flow') as predicted using NZTER.

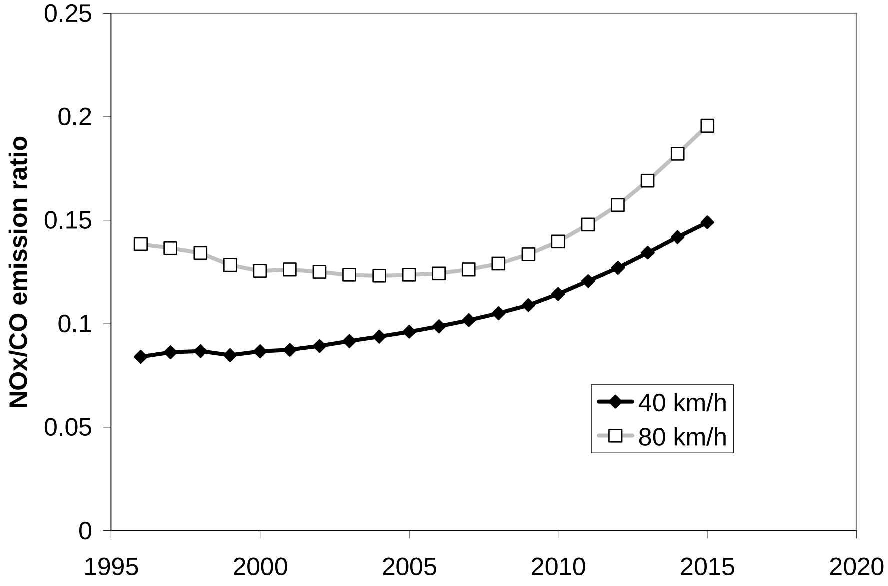

Figure 7.18: Trend in NO_x/CO fleet-averaged emission ratio on New Zealand roads as predicted using VEPM.

Note that the discussion above concerned 'average' traffic on 'average' roads. For any specific road, the emission rates, and the NO_x/CO emission ratios, are sensitive to the fleet mix, driving conditions and vehicle speeds. This is particularly relevant for NO_x because its emissions are dominated by HDVs and highly dependant on vehicle speed, and thus are sensitive to the correct description of the number of HDVs and vehicle speed on any road of interest. This is illustrated in Figure 7.19 showing the trend predicted by NZTER for three different proportions of large HCV (all other traffic assumed to be petrol cars).

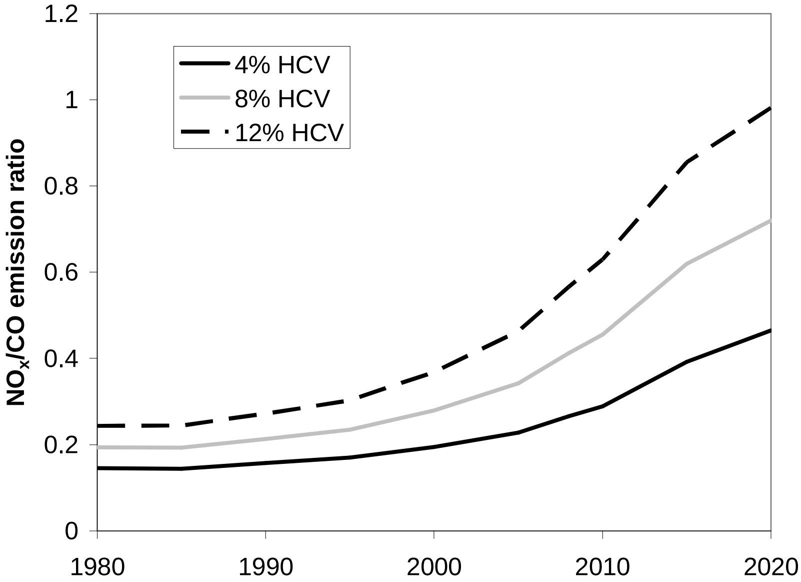

Figure 7.19: Hypothetical trend in corridor-averaged NO_x/CO emission ratio for petrol cars plus three different proportions of large HCV, derived from NZTER.

Other local variations may apply which could be significant. For instance, the Mount Victoria tunnel carries a much larger than normal proportion of taxis due to it being the principal route linking the city centre (and much of the surrounding country) with the airport. The fleet mix in the Homer Tunnel is also likely to be significantly different to that on urban roads, while the Lyttelton tunnel near Christchurch carries high levels of HDVs to service the container and bulk loading facilities at the port.

Using NZTER, higher NO_x/CO emission ratios are found using the 'free flow' factors than the 'interrupted' factors, with 'congested' factors giving the lowest ratio. This reflects the fact that CO emission factors generally increase with congestion as NO_x emission factors decrease.

In summary, higher NO_x/CO ratios are related to free-flowing motorways with high proportions of heavy duty vehicles. Lower NO_x/CO ratios are related to urban roads with congestion and low proportions of HDVs.

7.6 Partitioning of oxides of nitrogen

Interpreting the significance of NO_x emissions, their reduction, and the increasing trend in NO_x/CO emission ratios requires an understanding of the way NO_x emissions are partitioned between the less-toxic compound NO and the more toxic NO_2.

Conventionally in air quality assessments it is assumed that when NO_x is emitted **at the tailpipe** it is overwhelmingly in the form of NO, but quantifying this proportion for traffic in general is difficult due to the scarcity of experimental data. Conventionally, values such as 5 % have been assumed. Almost three decades ago, Hilliard & Wheeler (1979) reported values of 2 – 5 % for non-catalyst petrol cars, but more recent data suggests that more recent vehicles equipped with three-way catalysts, and more recent emission-reduction technology may have much higher ratios. For instance, Soltic & Weilenmann (2003) investigated emissions from Euro II vehicles, observing ratios of 2 – 8 % (with one outlier above 10 %) from cars, rising towards 20 % at cold start and 10 – 40 % from light-duty trucks.

If ozone is present (as it generally is in the ambient air) some of the NO will rapidly react to form NO_2, such that the NO_2/NO_x concentration ratio will rapidly rise as the exhaust plume dilutes and disperses. At roadside monitoring stations when traffic is busy NO_2/NO_x concentration ratios are typically 10 – 30 %, with lower values for higher NO_x concentrations. A value of 20 % is sometimes adopted in air quality assessments at locations in the range 20 – 50 m from a major road (e.g. MfE, 2008), although adopting a single ratio that applies at all times does not adequately describe the complexity of nitrogen chemistry and can lead to significant errors.

The net NO_2/ NO_x ratio from a stream of traffic may be estimated from measurements of NO_2 and NO_x inside a road tunnel, if the concentrations are corrected by removal of external background concentrations. There have been remarkably few examples of this reported in the open literature. Measurements inside a short road tunnel in Birmingham (UK) conducted in 1992 and 1993 (note that the Euro I standard for cars and HGVs only came into force in 1992) indicated a net NO_2/ NO_x emission ratio of 3 – 8 % with the higher values observed during traffic congestion (Harrison & Shi, 1996). Measurements in longer tunnels are more complicated, and discussed further in chapter 9.

More recent research has investigated the variability in the NO_2/ NO_x emission ratio as a function of traffic. Carslaw & Beevers (2005) summarised dynamometer measurements by Latham *et al*. (2001) to show how the NO_2/ NO_x emission ratio for London traffic in 2001 varied with vehicle type and speed. The ratio was shown to be a minimum at speeds above 60 km h^{-1}, whilst peaking at low speeds for petrol cars and at 40 – 50 km h^{-1} for diesel LGVs, buses and HGVs. For the mean London fleet, NO_2/ NO_x was estimated as 10 – 12 % below 50 km h^{-1} falling to below 5 % above 80 km h^{-1}.

Subsequently, UK research has identified a recent and rapid rise in the NO_2/NO_x emission ratio, rising from 5 – 7 % in 1997 and reaching 30 % by 2006 (Carslaw, 2005, Abbott, 2005, Carslaw *et al*., 2007, AQEG, 2007). This rise has been attributed principally to the penetration of Euro III light duty diesel vehicles fitted with oxidation catalysts and the fitting of regenerating particle traps to buses. However AQEG noted that these explanations do not seem sufficient to describe all of the observed trends. A similarly rapid increase in the NO_2/NO_x emission ratio has been observed at a motorway monitoring site in Switzerland (Hueglin *et al*., 2006) and analysis from other European sites is beginning to confirm the same trend.

Secondly, in certain limited conditions (very high NO emissions coupled with poor ventilation) another chemical reaction can be triggered in which NO can oxidise to NO_2 without the need for ozone, by reacting directly with oxygen. This is a high-order reaction and can, in principle, lead to rapid production of NO_2 within a road tunnel. However, this event can only be described in principle as there are hardly any documented instances of it happening (due to the scarceness of in-tunnel NO_2 measurements), and it may be limited to very long (several km) tunnels.. This possibility is discussed in more detail in chapter 9.

In summary, NO_2 'effective emissions' (i.e. concentrations arising directly from vehicle emissions or indirectly from the oxidised emissions of NO) in a road tunnel may still be roughly estimated by assuming an NO_2/NO_x ratio of 0.1, whilst bearing in mind that this ratio is tunnel-specific and may be rising with time. To retain conservatism a ratio of 0.2 should probably be considered. We hope that the uncertainty surrounding this important ratio is reduced in the near future by gathering detailed in-tunnel observations.

7.7 Predictions of future emission trends in NZ

7.7.1 Indications from the UK and EU15

We have indicated how New Zealand vehicles are equivalent to UK vehicles 6 - 12 years ago in terms of emissions on average. Current and proposed emission standards in New Zealand are intended to follow the developments in Europe such that we may expect broadly similar emission trends here in the next decade to those experienced in the UK and EU15 over the previous decade, albeit not necessarily at the same rate.

The most pertinent trend is the impact of the most recent vehicles meeting Euro IV and in some cases Euro V standards. Euro V especially (and the proposed Euro VI standard) is intended to allow NO_x reductions to 'catch up' with the CO, PM and VOC reductions already achieved. Figure 7.20 predicts the impact of penetrations of standards **up to Euro IV only** on the crucial NO_x/CO ratio. The ratio is seen to peak in 2007 or 2008 before falling again. Predicted trends in total emissions per vkt in the EU15 (see figure 7.13 above) lead to a forecasted peak in NO_x/CO (and PM_{10}/CO) around 2012 to be followed by a gradual fall. In the NZTER data (figure 7.17) the NZ NO_x/CO ratio appears to be rising towards 2020. Beyond that, however, any prediction would be uncertain and speculative at this point. As noted above, the consequence of this cannot be clearly judged until we have better information on which to predict any changes in the NO_2/NO_x ratio.

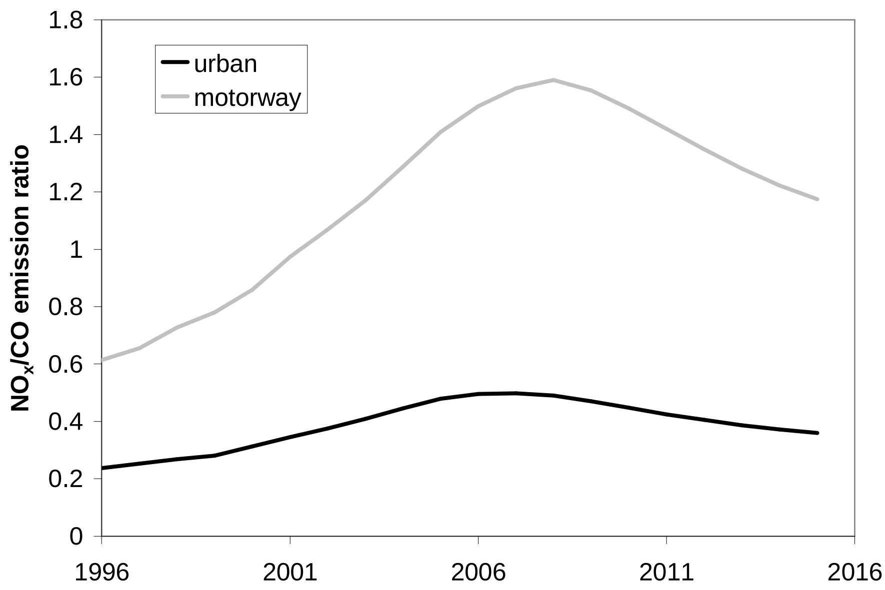

Figure 7.20: Predicted evolution of the UK fleet-averaged NO_x/CO emission ratio.

7.8 Consequences of emission evolution for road tunnel air quality limits

From around 1985 it appears that penetration of new technology vehicles into the New Zealand fleet has begun to raise the average NO_x/CO emission ratio. The rate of increase in that ratio has been subsequently accelerating, and is predicted to continue to do so until around 2015. Data from Europe suggest that the rise may be temporary. Newer technologies aimed at ensuring NO_x emission reductions catch up with CO reductions may halt the NO_x/CO ratio rise some time beyond 2020. Beyond that a reduction is predicted, but forecasts on this timescale are quite uncertain as they depend upon predictions of fleet technology mix and turnover, plus the introduction and effects of future emission standards (such as Euro V and VI).

The focus on the NO_x/CO ratio arises from the convention to use in-tunnel CO limits to provide protection to tunnel users from all vehicle-related pollutants. Recent research has lowered the acceptable exposure concentrations for NO_2 during the same period that reductions in NO_x emissions have failed to keep up with faster reductions in CO emission. Assuming NO_2 represents a fixed proportion of NO_x, the NO_x/CO ratio indicates the NO_2 concentration that would be present in a tunnel for any given CO concentration, once background levels are considered, have increased in time. However, we have also indicated how NO_2/NO_x ratios may be increasing also.

The consequence is that for a given in-tunnel CO concentration the NO_x concentration derived from vehicle emissions has more than doubled over the last three decades. Once NO_2/NO_x ratio rises are considered, the NO_2 concentration has increased even more.

To illustrate this, consider a hypothetical tunnel designed in 1980 with an ambient CO concentration limit of 87 ppm, based on the WHO ambient guideline. We assume background concentrations are negligible for CO and 0.01 ppm for NO_2. What would the NO_2 concentration be if the CO limit were reached? Let's apply two sets of assumptions. The first assumptions would be what may have been acceptable for 1980, i.e. a relatively low NO_x/CO ratio of 0.08[10], and a NO_2/NO_x ratio of 0.03[11]. The second set of assumptions could be described as "modern-pessimistic", i.e. a higher

[10] Value adopted from observational data (1980/1) in Mt Victoria tunnel, see Task 2 report
[11] Value adopted from observational data (1980/1) in Mt Victoria tunnel, see Task 2 report

NO_x/CO ratio of 0.15[12] and a NO_2/NO_x ratio of 0.15[13]. The resulting NO_2 concentrations would be:

$$NO_{2i} = \left(\left((COi - COb) \frac{EF_{NOx}}{EF_{CO}} \right) \frac{NO_2}{NO_x} \right) + NO_{2b}$$

1980 assumptions	Modern assumptions
$NO_{2i} = (((100-0)*0.08)*0.03)+0.02 = 0.14\,ppm$	$NO_{2i} = (((100-0)*0.15)*0.15)+0.02 = 1.2\,ppm$

If NO_2 limits were set at 1 ppm or higher this would have indicated that the CO limit was the more stringent in 1980, but no longer. If we assume that a NO_x/CO ratio of 0.1 represents a best estimate for the current value, then the guideline of 1 ppm of NO_2 is reached before 87 ppm of CO if NO_2/CO is greater than 0.019, or NO_2/NO_x is greater than 0.19 if the NO_x/CO ratio is 0.1. Unfortunately there is considerable uncertainty regarding the NO_2/NO_x ratio in tunnels. Nevertheless, it can clearly be concluded that whereas a CO limit of 87 ppm alone did provide sufficient protection against the effects of NO_2 in the past, that is not necessarily the case now and increasingly so in the near future.

Recommendation: A CO limit for in-tunnel concentrations does not provide sufficient protection to tunnel users from the impacts of nitrogen dioxide. We recommend that a separate and additional NO_2 limit is also adopted. It is unclear if both limits will still be needed beyond 2020. As a precautionary measure we recommend that both limits be adopted on the assumption that both will still be required beyond 2020, but that the limits be reviewed on a decadal basis in the light of new emission trend data, forecasts and new health research.

Chapter 10 will consider how an NO_2 limit might be adopted and practically implemented alongside a CO limit.

[12] Based on increase predicted by VEPM emission model, plus ‘pessimistic’ margin
[13] Based on assumption of NO_2 production from termolecular reaction with oxygen at anticipated NO_x levels based on Indrehus & Vassbotn (2001) and/or increased direct NO_2 emission

8. Monitoring

8.1 Overview

- We highly recommended that a tunnel's ventilation system's performance is checked by monitoring at least once and preferably on at least a decadal cycle.

- A post-opening campaign of monitoring can verify whether the system is functioning within acceptable limits, whether adjustments are necessary, or whether the ventilation is excessive permitting the possibility of energy savings.

- Long-term changes in traffic and emissions, which were only estimated at the design stage, may be evaluated from real data and will provide the best information for ventilation upgrades also providing valuable NZ-specific information for the design of future tunnels.

- The issues of monitor representativeness, failure or inaccuracy can be overcome to some degree by using multiple monitors through the tunnel – ideally, control systems should not rely on a single monitor, especially considering the maintenance, reliability and accuracy issues.

8.2 Rationale and objectives for monitoring

Once in-tunnel concentration or exposure limits are set and the ventilation system designed so that the limits will not be breached it is highly recommended that the system's performance is checked by monitoring once the tunnel is opened to traffic. The design procedure is based on emissions estimates which are inherently uncertain, and is sensitive to traffic predictions which may be in error. A post-opening campaign of monitoring can verify whether the system is functioning within acceptable limits, whether adjustments are necessary, or whether the ventilation is excessive permitting the possibility of energy savings. Ideally, monitoring for the purpose of verifying the efficacy of the ventilation design should be conducted according to the following criteria:

- Temporal resolution at least as fine as the averaging time of the applied limit (e.g. 15 minutes) and preferably much finer (e.g. 1 minute).

- Monitoring should be conducted throughout the daytime hours during that traffic peaks, and preferably continuously (24 hours a day). Continuous monitoring is required if the ventilation rate is designed to vary during the day.

- Monitoring should cover whole weeks, as traffic conditions are often substantially different on Sundays, Saturdays and weekdays (and may differ between weekdays also).

- If only a single monitor is used, it should be located at or as near as practical, the location of expected maximum concentrations (see below).

- Multiple monitors are preferred (see discussion below).

We recommend that permanent monitoring be included as part of the tunnel design. In this way compliance with the requirement of acceptable air quality may be monitored in all conditions, including congested, accidents, poor external air quality episodes, ventilation malfunction, etc. Long-term changes in traffic and emissions, which were only estimated at the design stage, may be evaluated from real data and will provide the best information for ventilation upgrades also providing valuable NZ-specific information for the design of future tunnels.

A more advanced system will utilise live monitoring data as a feedback into ventilation control, or traffic management. In this case, very fine temporal resolution is required, as well as a high degree of accuracy and multiple monitors to provide a continuous, reliable service, with consequent demands on maintenance and calibration.

8.3 In-tunnel monitoring

8.3.1 Air quality monitoring in a hostile environment

The interior of road tunnels present a challenging environment for the measurement of air quality. Optical surfaces that are essential for measurements of opacity, and which are also unavoidable elements of the chemi-luminescence instruments generally used to measure NO_2, are exposed to a very dirty, dusty and sooty environment for long periods of time. Tunnels are often cleaned using jet sprays often including strong solvents or acids. Hence monitors require high levels of maintenance, protection and regular calibration. However, maintenance of any kind in a road tunnel is difficult and expensive due to the need to greatly reduce concentrations when personnel are likely to be spending prolonged periods of time in the tunnel. Analytical measurement techniques for atmospheric gases and particles are generally not designed for use in tunnels and are usually deployed and operated by non-specialists.

8.3.2 Sensor limits and representativeness

As concentrations can vary significantly with not just depth into the tunnel (see section 8.3 below), but potentially across the cross-section, the choice of instrument location is crucial.

To illustrate some of these issues it is interesting to consider two studies that have compared independently monitored data with data from the tunnel's own monitoring systems. These permanent CO sensors are sometimes located in the roof of the tunnel. In the case of any potential negative vertical concentration gradient, roof-mounted CO sensors will under-represent the concentration at exposure heights (typically 1 m for cars, or 1 - 3 m for other vehicles).

This possibility was studied by HKPU (2005) in the Shing Mun and Tseung Kwan O tunnels in Hong Kong. CO and NO data from the tunnel company were compared to that gathered by monitoring conducted by the research team at 1.5 m height above the roadway. Small differences were measured in the Tseung Kwan O tunnel, but in the Shing Mun tunnel mean exposure-height concentrations were 1.8 times higher than the tunnel company's data in the northbound tube in winter, 1.6 times higher in the south tube in summer and 3.5 times higher in the south tube in winter. There were insufficient data in the relevant report to assess the degree to which this discrepancy could be explained by other factors (e.g. sensor accuracy, lateral location, differences in averaging time, etc.).

A further study compared in-tunnel roadside CO measurements with those measured externally on a vehicle passing through both the Plabutsch tunnel (near Graz, Austria) and the Mersey Kingsway tunnel in Liverpool, UK (Boulter *et al*., 2004). This study concluded that the CO concentration measured outside the vehicle was typically 50 % higher than that measured at fixed points at the tunnel roadside. In the UK exposure-management approach (see chapter 9 for more details) the CO exposure limit (200 ppm) is translated into a sensor limit using, in part, an assumed relationship between what a fixed sensor in the tunnel detects and what the concentration actually is on the carriageway at a height that air enters a vehicle cabin. This factor is taken to be 3 – 6 based on studies outside of tunnels, leading Boulter *et al*. (2004) to suggest that the current factors used in the UK are excessively conservative.

8.3.3 Accuracy and reliability

A comparison between CO concentrations made on the roof of a moving vehicle and tunnel company monitors was made in the M5 East tunnel in Sydney (SESPHU, 2003). Out of 32 eastbound transects in the morning only one fixed monitor out of 8 had a correlation with mobile measurements. All 8 monitor values were highly correlated with mobile values on westbound journeys. The highest correlation was for one sensor and eastbound afternoon journeys, with a relationship of

$$\text{Monitor value} = 1.36 \times \text{mobile value} + 20.02.$$

This 20.02 ppm offset is highly significant if the sensor is supposed to be representing exposure of persons in the tunnel, and cannot be simply accounted for.

In September 2002 the NSW Department of Infrastructure, Planning and Natural Resources requested that they be notified of any exceedences of 200 ppm of CO as a 3 minute average at any monitor in the M5 East tunnel. In order to provide such information in July 2003 the operating scales of CO monitors were reset from 0 – 100 ppm to 0 – 300 ppm. This change potentially reduced the accuracy of the data from the monitors, although this conclusion is dependent upon a disputed interpretation of manufacturer's data. Auditors considered that accuracy reduced from ±8 ppm to ±24 ppm, although this was disputed by the tunnel operators (NSW Planning, 2005). An intercomparison between one of the CO monitors and an infra-red absorption instrument was conducted over 1^{st} – 5^{th} April 2004. If the IR instrument is assumed to have been accurate, then it gave readings between –5 ppm and +1 ppm of the monitor value. It was noted that in-tunnel monitors are more prone to drift errors than in non-tunnel environments. Consequently, the maintenance schedule was amended for monitors in the M5 East tunnel requiring monitors to be removed and calibrated every

6 months, protection of the monitors during tunnel wall washing (followed by another calibration) and weekly checks by comparison with the nearest monitor.

Airflow sensors are critical items in a ventilation system. Evidence of 10 exceedences of the CO guideline in the M5 East tunnel (87 ppm averaged over 15 minutes) was found by auditors between 5th March 2002 and 13th May 2003 (NSW Planning, 2005). On five occasions malfunctions of airflow sensors was cited as the cause. Four incidents were related to traffic congestion (one vehicle breakdown incident and once due to flooding), and one incident related to incorrect operation of the ventilation system during a shutdown of the stack for maintenance work.

The issue of monitor representativeness, failure or inaccuracy can be overcome to some degree by using multiple monitors through the tunnel – ideally, control systems should not rely on a single monitor, especially considering the maintenance, reliability and accuracy issues. For example, the 4 km M5 East tunnel in Sydney contains 4 CO monitors in each tube. The Cross City tunnel contains a total of 18 CO monitors throughout the tunnel system. The Shing Mun tunnel contains 10 sets of CO, NO, NO_2 and visibility monitors (Yao *et al*., 2005). Data should always be validated by regular calibration. In the literature we have noted an 8-month calibration schedule in the Bomlafjord tunnel in Norway (Indrehus & Aralt, 2005).

8.4 Tunnel ventilation schemes and concentration variability

8.4.1 Air quality variability in tunnels and implications for limit implementation

Once one or more limits are set for concentrations of pollutants within tunnels they need to be implemented. A key decision is whether implementation will include enforcement and whether it will include a feedback loop in which measured concentrations control or inform the operation of mechanical ventilation or traffic management. Both enforcement and feedback-control will require monitoring of in-tunnel concentrations. This is far from straight-forward as these concentrations are highly variable in space and time. This chapter will give a brief, general overview of those variabilities to help identify the implications for monitoring and implementation.

8.4.2 Passive and reactive species

As noted in chapter 2, the constituents of vehicle exhaust are a highly complex mixture. Within this mixture are substances which are more and less chemically reactive. The presence of a large particulate component mixed with rapidly cooling and condensing vapours means that an exhaust plume is also likely to undergo physical transformations, such as interactions between gases and particles. For the purposes of limit implementation, however, we are primarily concerned with carbon monoxide and nitrogen dioxide, whilst we also wish to be mindful of the impacts of all particles and toxic gases.

On the timescale on which any parcel of air remains in a road tunnel, carbon monoxide is effectively inert. Thus we may assume that all of the CO injected into the tunnel air volume will dilute with the entrained ambient air and then be removed from the tunnel along with that air. CO is described in this manner as ‘conservative’ (i.e. total mass emitted is conserved) or ‘passive’. Nitrogen dioxide, however, is mostly emitted as nitric oxide (NO) which then forms NO_2. Both NO and NO_2 are chemically reactive on the relevant timescale. This makes levels of NO_2 much more unpredictable and the consequences of this are explained in section 9.1 below. As a first approximation it is reasonable to assume that particle species and metrics (e.g. PM_{10}, PAHs, black carbon, etc.) and other key toxic gases (such as benzene) are non-reactive (i.e. there is no direct evidence from road tunnels to assume otherwise), and therefore behave in a similar way to CO.

Concentrations just outside tunnel portals are similar to those in the ambient (roadside) air. Concentrations inside the tunnel are higher due to the reduced opportunity for mixing and dilution with ambient air. However, the concentrations gradient between inside and outside, and the variability along the tunnel length is dependent upon the way in which the tunnel is designed and ventilated.

8.4.3 Tunnel ventilation schemes

In-tunnel concentrations and the emission from the tunnel openings will depend upon the rate of ventilation. This rate will vary within a range that is unique for each tunnel and is effectively constrained at the design stage. Three basic options exist:

a) passive ventilation

b) longitudinal ventilation

c) transverse ventilation (including semi-transverse)

Vehicles moving through a tunnel induce their own air flow through drag (‘traffic-induced turbulence’). In the immediate vicinity of the vehicle this induces mixing which helps to disperse exhaust emissions. In very short tunnels (< 300 m) this induced airflow is generally assumed to be sufficient to entrain enough air from outside the tunnel to sufficiently dilute the vehicle emissions within the tunnel to levels below the relevant concentration limits.

In a longer tunnel, the movement of vehicles through the tube induces a more organised airflow in the direction of vehicle motion. This phenomenon is known as the ‘piston effect’ and it is the basis of passive ventilation. Passive ventilation requires no installation other than the tunnel itself, making it a lower cost option than having to install mechanical ventilation. The piston effect is only effective if all of the traffic is proceeding in the same direction, and this is the principle reason why most road tunnels longer than a few hundred metres have two tubes, one for each direction of travel. The inevitable consequence is that contaminated air is transported to both tunnel exits leading to two emission point sources within the tunnel’s surrounding community, although this can be avoided by having air mechanically extracted to an alternative vent or stack.

Longitudinal ventilation refers to installations in which the piston effect is boosted or assisted by fans increasing the ventilation rate (see figure 8.1). The word ‘longitudinal’ refers to the general direction of diluting airflow along the tunnel’s length. This arrangement represents both a capital cost, plus an operational cost which needs to be justified. Longitudinal ventilation is very common for tunnels over a few hundred metres. A major advantage of longitudinal ventilation is that operational costs can be reduced by not running the fans when unassisted (i.e. passive) ventilation is sufficient to maintain adequate air quality.

Transverse ventilation consists of a system to deliver fresh air and remove contaminated air at points along the full length of the tunnel (figure 8.1). Commonly fresh air enters via the roof and contaminated air leaves through the floor, hence the use of the word ‘transverse’ to describe the direction of airflow across the bore of the tunnel, perpendicular to vehicle motion. However, such a fully-transverse system is not so common (examples include the Lion Rock tunnel, Hong Kong, Plabutsch tunnel, Graz, Central Artery and Ted Williams tunnels, Boston, Caldecott tunnel, Oakland and the Tauerntunnel, near Salzburg). More often semi-transverse ventilation is employed. This involves either the provision of fresh air or the removal of contaminated air only, with the former being more common. Air enters or exits the tunnel at a separate opening, stack or stacks as well as the tunnel portals, and the system can be designed so that no air leaves via the tunnel portals. Such a system demands a much larger capital investment due to the extra ventilation shafts and equipment. One published estimate suggests that ventilation represents 30 % of the total costs of a semi-transverse tunnel compared to 5 – 10 % for a longitudinal tunnel (CETU, 2003). Electrical power consumption for major tunnels can be of the order of megawatts per km (Jacques & Possoz, 1996). The environmental gains in ventilating tunnels should ideally be balanced against the environmental costs in terms of energy consumption.

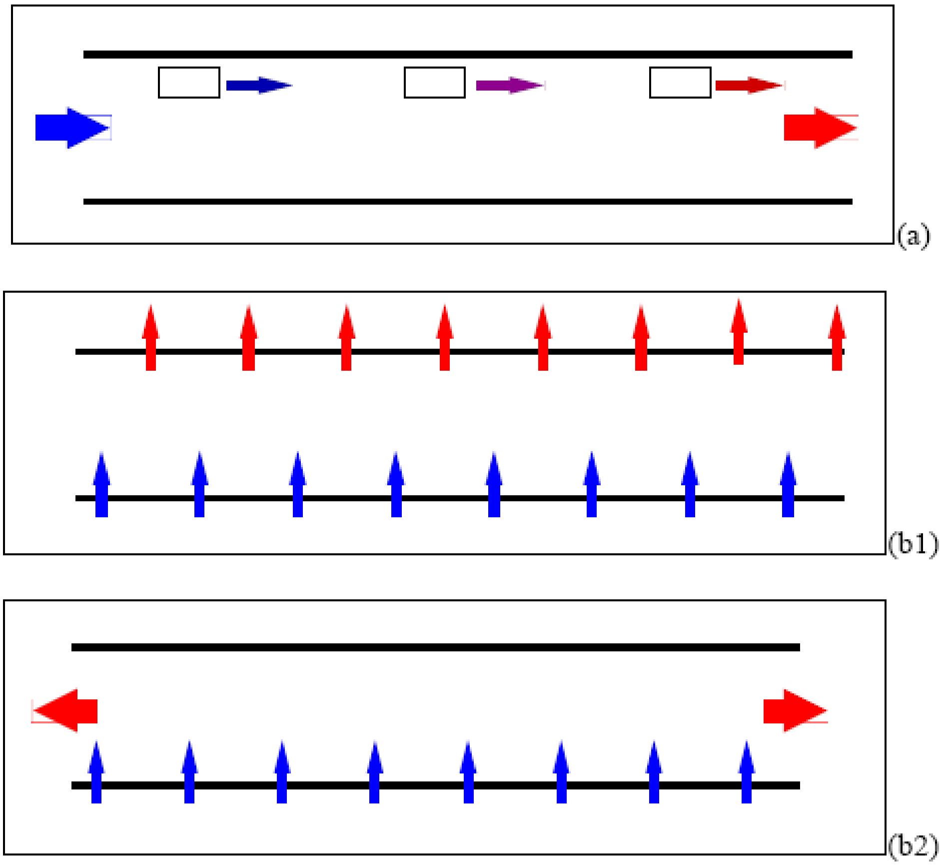

Figure 8.1 Illustration of the air flow in a) longitudinal, b1) transverse and b2) semi-transverse ventilation systems. Blue represents fresh air, red vitiated air (from CETU, 2003).

The selection of ventilation system is a complex engineering process, but in broad terms, more complex systems have been applied to longer tunnels. A guidance document produced by the French Centre for Tunnel Studies (CETU) reports that longitudinal systems are generally used where recurring congestion is not expected and transverse systems where it is. Table 8.1 below shows some of the recommended maximum tunnel lengths for each system found in a range of literature.

Table 8.1: Recommended tunnel length limits by ventilation type

	Tunnel length
Natural	< 300 m[1,3]
Longitudinal (uni-directional only for urban or high-traffic tunnels)	< 600 m[1] < 500 m[3] Any length with mass extraction[3] Recommended for uni-directional non-urban tunnels > 500 m[3]
Semi-transverse	< 1000 m[1] < 1500 m[2]
Transverse	> 1500 m[1]

[1] El-Fadel & Hashisho, 2001

[2] ASHRAE

[3] CETU, 2003

Recently longitudinal systems have been installed in long, busy urban tunnels. Examples include the M5 East (Sydney), Cross City (Sydney), Tate’s Cairn (Hong Kong) and Shing Mun (Hong Kong) tunnels, all over 2 km long and opened since 1990. This has been made possible by the general long-term reduction in vehicle emissions and the use of mass-extraction ventilation systems and ventilation stacks.

High levels of pollutant emissions from the portals of long, busy tunnels may not be acceptable if the portals are in residential areas. In this case, fans can direct most of the tunnel air out of a separate ventilation stack at an elevated height rather than out of the portals at ground level.

8.4.4 The 'piston effect' and the operation of longitudinal ventilation

The size of the piston effect of air flow induced by vehicles in the tunnel is a complex function of traffic volume, speed, fleet mix and the tunnel dimensions. It is, however, limited and its effect diminished in longer tunnels by increased pressure losses due to friction, etc. It is recommended by CETU that air velocity is limited to a maximum of 8 m s^{-1} in a bi-directional tunnel and 10 m s^{-1} in a uni-directional tunnel for the purposes of pedestrian safety and to allow for effective smoke removal in the case of a fire. If one has a fixed upper concentration limit to design to, this effectively puts a limit on the tunnel length, unless multiple opportunities for air exchange, other than the portals or a single stack, are introduced to the design. In the case of low traffic in the tunnel a minimum air flow (provided either passively or mechanically if necessary) should be included in the design so as to cope with the transient effects of gross polluting vehicles or tunnel road blockage.

8.4.5 Variation in concentrations along passively or longitudinally ventilated uni-directional tunnels

The concentration of any traffic-related pollutant at any point in a uni-directional longitudinally or passively ventilated tunnel is dependent upon the cumulative emissions from the tunnel entry up to that point, i.e. the concentration increases with distance along the tunnel. In the simplest notional case of a passively ventilated tunnel with evenly distributed emissions, no entrainment of fresh air and no pollutant removal mechanisms, the concentration of CO, NO_x and PM will increase linearly with depth into the tunnel (CETU, 2003, Chang & Rudy, 1990). Entrainment and removal (such as deposition) will cause concentrations to level off near either end. Transect studies, a continuous measurement of pollutant concentrations made from a normal vehicle moving through the tunnel (e.g. SEHA, 1994, 1995, SESPHU, 2003), confirm this general picture (see figure 8.2 below). Repeated measurements made at 100 m and 1000 m into the 1500 m long Söderledstunnel in Stockholm consistently show large concentration increases at 1000 m compared to 100 m. This study indicated that the average concentration during a transect of this tunnel is similar to the concentration at one third to one half depth, which in this case can be approximated by the mean of the 100 m and 1000 m values.

8.4.6 Variation in concentrations along semi-transverse or transverse ventilated tunnels

In theory the concentrations in a semi-transverse tunnel should increase initially and then level off as the accumulation of emissions is countered by dilution by the fresh air injected along the tunnel length. This theoretical situation is described by the model of Chang & Rudy (1990). The difference between semi-transverse, longitudinal and natural ventilation for a hypothetical tunnel according to Chang & Rudy (1990) is shown in figure 8.2.

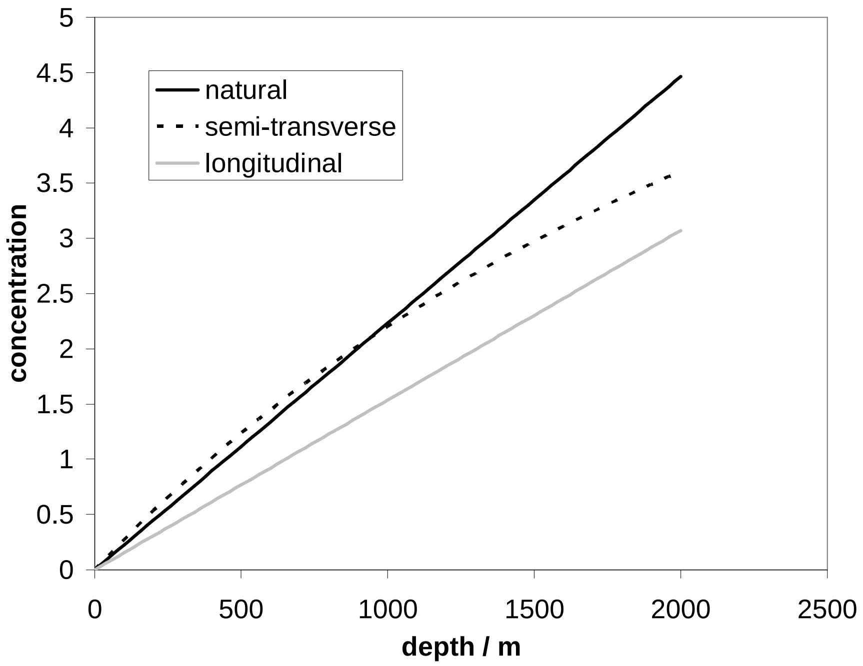

Figure 8.2: Variation in concentration of a conservative species with zero ambient concentration with depth of hypothetical tunnels with three different ventilation schemes.

Very few data exist to verify this model. In one study transects were made through the Lion Rock tunnel in Hong Kong. This tunnel has fully transverse ventilation and a high traffic flow of ~ 95 000 vehicles per day. In the transect concentrations of CO concentrations were approximately constant along the tunnel length, except in the first 50 metres of the tunnel where they were up to 100% higher (Chow & Chan, 2003). This may have been due to traffic congestion and the acceleration of vehicles away from the tunnel toll plaza.

8.4.7 Bi-directional tunnels

Bi-directional tunnels have a single tube and the ventilation capacity of the piston effect is lost. In such tunnels longitudinal mechanical ventilation is inefficient as it has to push against the traffic moving in one direction. Transverse ventilation is favoured. In theory concentrations will be reasonably constant with depth, but in reality an external wind or pressure gradients are likely to bias the internal airflow in one direction leading to an increase in concentration towards one end of the tunnel.

8.5 Monitor location and representativeness – summary and recommendations

Accurate monitoring of CO, NO, visibility and especially NO_2 in a road tunnel are very challenging. Calibration and maintenance protocols must be demanding and rigorously followed in order that the monitoring is not rendered ineffective. NO_2 may be estimated indirectly through its relationship with NO, CO and/or visibility. Such relationships are complex and such a system is best tuned using live data once a tunnel and monitoring system is already operational.

Whether monitors are placed near the roof of a tunnel, at the sides or near the floor may make a significant difference to the concentrations reported. The monitors are ideally intended to represent the concentration on the carriageway at a height of up to 1 metre, i.e. the height at which air typically enters most vehicle cabins. Efforts should be made to characterise any consistent difference between concentrations in this ‘exposure zone’ and those reported by the monitor. If persistent differences are found NZTA may wish to consider a conversion of the air quality limit into a ‘sensor limit’ (i.e. if the sensor consistently under-represents concentrations in the exposure zone by half, the sensor limit should be half the exposure limit). This step must not be taken lightly as it must be shown that a reduction in protection of tunnel users does not occur as a result.

A summary of the dependence of concentrations on depth as a function of ventilation follows:

- Longitudinal, uni-directional tunnels: CO rises with depth initially linearly, and then the rate of rise falls and concentrations levels off at a peak, which will occur in the second half of the tunnel. NO_2 rises less steeply than CO, and may actually fall in the first half of a long (> 2 km) tunnel. Peak CO and NO_2

concentrations will both occur in the second half of the tunnel, but not necessarily at the same point.

- Longer tunnels with congestion or poor airflow: NO_2 may rise rapidly and locally if NO_x exceeds a few ppm.

- (Semi-)Transverse, uni-directional tunnels: CO rises less steeply than in a similar longitudinal tunnel and level off sooner. Peak concentrations occur in the second half of the tunnel. Insufficient data to make conclusions regarding NO_2 (probably dependent on rate of ozone injection).

- (Semi-)Transverse, bi-directional tunnels: CO relatively uniform along depth, but slightly higher towards downwind end.

A survey of concentration variation can be undertaken in an existing tunnel which is to have a monitoring system retrofitted and this is highly recommended. In a new tunnel, it is recommended that numerical modelling be employed to predict the likely spatial variation in concentrations. It needs to be made clear whether the location is being chosen so as to be representative of the peak or average concentration in the tunnel. The peak is in many ways simpler to interpret, but the location of the peak may not be consistent (especially in a bi-directional tunnel). In many cases the location of any monitor is dictated by logistical practicalities. In this case it is important to survey how the concentrations at this location relate to concentrations elsewhere in the tunnel, and whether the relationships are consistent or vary in some predictable way.

9. An exposure-management approach for tunnel users

9.1 Overview

- Exposure management for road tunnels is a more technically demanding approach to tunnel air quality management which is subject to more uncertainties than the conventional ‘hazard management’ approach. However, it offers the potential advantage of energy savings from reduced ventilation, the opportunities for which are greatest in shorter tunnels and those with a higher minimum speed.

- Whereas hazard management applies an air quality guideline to the tunnel air, exposure management applies any guideline to *individuals* using the tunnel, who, in most cases, will be exposed for less than 15 minutes.

- Exposure management is based primarily on concept recognition that exposure to a higher concentration of carbon monoxide for a shorter period can have an equivalent effect on the body (represented by blood carboxyhaemoglobin levels) as exposure to a lower concentration for a longer period.

- The same assumption has been applied to exposure to other pollutants, although this is based on weaker scientific evidence. This means that the implications of permitting elevated levels of other pollutants (on the basis of shorter exposures) on the health of tunnel users cannot be specified. This is one reason why an exposure management approach may be rejected on precautionary grounds.

- This means that a given air quality guideline applied as a personal exposure limit may translate to a higher sensor limit (i.e. permissible concentration in the tunnel air) if it is possible to confidently specify a maximum probable exposure duration (i.e. journey time through the tunnel). This may be assisted if a minimum speed limit can be successfully implemented and enforced in the tunnel.

- The restriction on infiltration of air into a vehicle cabin reduces occupant exposure relative to pollutant concentrations as indicated by tunnel sensors.

However, tunnel ventilation management systems must be designed to protect the health and safety of all users. Consequently this extra protection cannot be assumed as some tunnel users will not have the protection of a cabin (motorcyclists, occupants of open-top cars, cars with windows fully down, etc.).

- For individual drivers and passengers, closing vehicle windows, switching off a vehicle's ventilation fans, and especially setting the its vents to 'recirculate', are very effective in reducing internal concentrations in a vehicle cabin to a small fraction of the concentration inside the tunnel, and is probably the most effective action individual tunnel users can take to reducing their exposure to air pollutants in a tunnel.

- Tunnel ventilation systems should be designed to the worst-case for exposure, which is low speeds and fully-open vehicles. Whether a vehicle cabin is open or sealed, the main determinant of the impact of carbon monoxide on health is the length of time spent in the tunnel.

9.2 What is exposure management?

It is the convention to manage air quality inside road tunnels on the basis of a hazard management approach. This means that a ventilation criterion for design (and operation if a variable ventilation system is installed) is based on maintaining a certain level of acceptable air quality represented by a concentration limit (typically of carbon monoxide). The ventilation is intended to prevent pollutant concentrations exceeding the limit at all times. A vital part of such an air quality limit is the specification of a timescale, usually stated as an averaging time. For example, NIWA recommend a carbon monoxide limit of 87 ppm as a 15-minute average, which is the same as the ambient air quality guideline of the World Health Organisation (WHO, 2000) and based upon the same principles.

A potential criticism of such an approach is that it can be unnecessarily over-demanding on ventilation if persons are not exposed for the full 15 minute period. This is because the CO limit is inherently an exposure guideline. The guideline is based on limiting the build up of carboxyhaemoglobin in the blood of an exposed person to below 2.5 % as a result of their exposure to carbon monoxide in the tunnel. That physiological response occurs in response to both the concentrations that the tunnel user is exposed to and to the duration of that exposure. The consequence is that exposure to a higher concentration for a shorter period can have an equivalent effect on the body (represented by blood carboxyhaemoglobin levels) as exposure to a lower concentration for a longer period.

For a road tunnel this means that a given fixed internal air concentration does not present an equal risk to all users. Rather, that risk is dependent upon the duration of exposure. That duration depends upon tunnel length and vehicle speed (as well as trapping of air in a vehicle cabin, discussed below). For a given concentration (e.g. the 87 ppm CO guideline) the total exposure, and hence health risk, is lower for a shorter tunnel or for faster speeds, both leading to shorter duration of exposure.

An exposure management approach seeks to explicitly manage the risk associated with exposure of tunnel users, rather than just the concentration of pollutant in the air within the tunnel. Implementation depends upon translating the **exposure limit**, i.e. the air quality guideline as applied to any individual's personal exposure, regardless of the vehicle they are in or the speed they are travelling at, into a **sensor limit**, i.e. the maximum allowable concentration in the tunnel as detected by a fixed sensor which will ensure compliance with the exposure guideline for all exposed individuals.

Predicting exposure of tunnel users moving at a range of speeds in a range of vehicles through a variable pollution field is considerably more complicated than the conventional hazard management approach. Exposure management is more technically demanding and subject to more uncertainties than hazard management. Implementation of exposure management must, therefore, offer significant advantages over hazard management to justify its adoption.

The carboxyhaemoglobin limit upon which CO concentration limits are based is implicitly an exposure limit. There may also be potential energy savings to be made by avoiding the over-ventilation of short tunnels.

The success of such a system is entirely dependent upon the assurance of maximum exposure duration. This has significant implications for occupational exposure, where exposure times are likely to be much longer than 15 minutes. However, a degree of freedom is permitted by the fact that the CO guidelines for occupational exposure are less stringent than for the public, e.g. 200 ppm compared to 87 ppm over 15 minutes (the rationale for this is explained in chapter 5). If tunnel maintenance is to be carried out it is necessary that the implementation of any exposure management system for public users does not lead to CO levels exceeding either the 15-minute or 8-hour occupational exposure guidelines.

In applying an exposure management approach the following factors need to be taken into consideration:

1. The variability in vehicle speeds and journey times through the tunnel, and determining factors (such as probability and timing of congestion, gradients, etc.),

2. The difference between concentrations where a fixed monitor is located and concentrations on the carriageway where air enters a vehicle cabin,

3. The difference between concentrations inside and outside a vehicle, i.e. the partial 'protection' of a vehicle occupant from outside pollutant concentrations due to limited infiltration of air,

4. The variability in this infiltration rate (due to 'leakiness' of vehicle design, opening of windows, operation of vents, fans and air conditioning).

Each of these points is discussed in detail below.

9.3 Exposure as a function of speed and length

Exposure duration is a function of journey speed and tunnel length. Figure 9.1 shows how exposure to a given average concentration of carbon monoxide and the duration of that exposure are related for a given physiological response (i.e. predicted rise in blood carboxyhaemoglobin from a baseline of 0.3 % to the threshold level for adverse health effects of 2.5 %, based on typical physiological parameters). Firstly this figure illustrates that the carboxyhaemoglobin response is dependent upon some age-dependent physiological parameters (principally blood volume) and on activity rate (which determines the rate at which CO passes into the blood stream). The NIWA recommended CO guideline for tunnels (and the WHO ambient air quality guideline for CO) is based on the target of preventing blood COHb exceeding 2.5 % for any person who is likely to be exposed (with some exceptions, such as smokers – see chapter 5 and WHO (2000) for further details). In the case of a road tunnel this translates to a guideline of 87 ppm as a 15-minute average in order to provide protection for children using the tunnel as vehicle passengers. If a tunnel is open to pedestrians, then the same guideline applies. Adult drivers (and passengers) have a slower response to carbon monoxide exposure than runners (because of their lower breathing rate and volume) and children (who have a smaller blood volume) and therefore can tolerate higher concentrations (approximately 3 times higher for 15-minutes exposures) for the same impact.

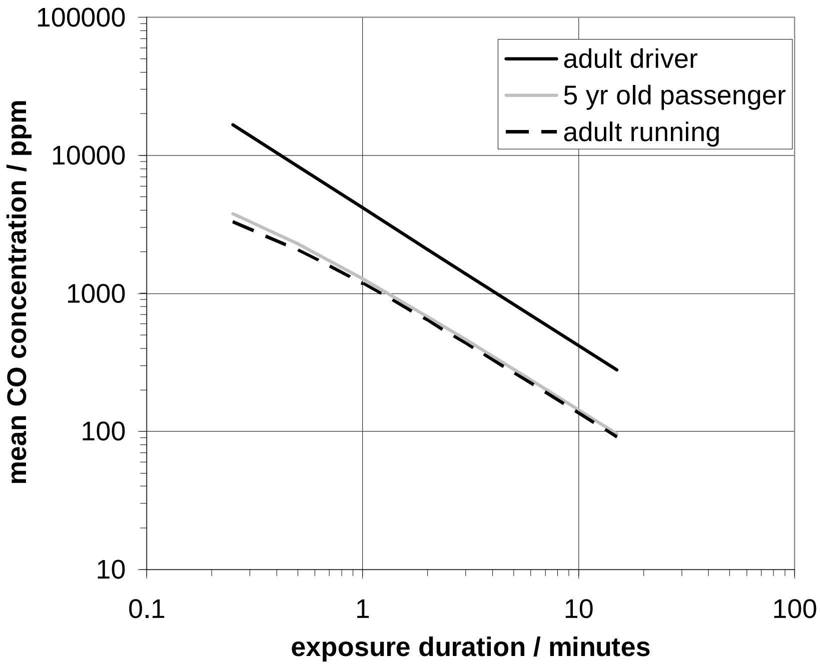

Figure 9.1. Relationship between mean CO concentration and exposure duration giving rise to equivalent effect on health (i.e. rise in blood carboxyhaemoglobin from baseline of 0.3 % to threshold level for adverse health effect of 2.5 %). Note log-log scales.

What figure 9.1 also shows is that where exposure durations are considerably shorter than 15 minutes, much higher concentrations than 100 ppm of CO may be tolerated as they lead to the equivalent physiological response. For example, exposure to 100 ppm for 15 minutes is equivalent to an exposure to 500 ppm for 3 minutes. In a hypothetical tunnel in which exposure durations of less than 3 minutes **could be assured**, the 15-minute **exposure limit** of 100 ppm would translate to an equivalent **sensor limit** of approximately 500 ppm. In this case, the 100 ppm limit applied in a hazard management mode would be 5 times more demanding than necessary.

The key to implementing an adjusted sensor limit based on this approach is being able to confidently specify a maximum probable exposure duration. For a given tunnel this corresponds to the minimum probable speed. The potential energy savings from reduced ventilation are greatest in shorter tunnels and for a higher minimum speed. Where road tunnels are closed to pedestrians and cyclists the minimum speed might be specified with ease or it might vary systematically with time of day. In tunnels where congestion results in a more unpredictable pattern of stop-go pulsing traffic there may be more uncertainty in the minimum speed or exposure duration and the advantages of exposure management may be lost. Similarly, if a tunnel is open to pedestrians, the range of possible exposure durations is longer and more randomly distributed over a wider range. **If the minimum speed cannot be confidently assessed, an exposure management approach based on higher sensor limits compensating for shorter exposures cannot be confidently implemented.**

On the other hand, exposure management may be assisted if a minimum speed limit can be successfully implemented and enforced in the tunnel. This may require traffic management measures upstream of the tunnel entrance, but this is highly desirable in any tunnel due to avoiding the increased emissions from congestion, increasing the inter-vehicle separation, which influences infiltration of pollutants into vehicle cabins (see below) and increasing ventilation by the ‘piston effect’.

The success of such a system is entirely dependent upon the assurance of maximum exposure duration. This has significant implications for occupational exposure, where exposure times are likely to be much longer than 15 minutes. This would require an alternative or additional ventilation regime.

There are currently insufficient health and biomedical research data to develop a comparable relationship between exposure concentration and duration as illustrated in figure 9.1 for other pollutants, such as nitrogen dioxide or particulate matter. This means that the implications of permitting elevated levels of other pollutants (on the basis of shorter exposures) on the health of tunnel users cannot be specified. This is one reason why an exposure management approach may be rejected on precautionary grounds.

9.4 Relating sensor data to road-level concentrations

There are limited data available on the relationship between concentrations measured by fixed tunnel sensors, which will necessarily be installed closed to the walls, roof or floor of the tunnel, and concentrations on the roadway at a height where air typically enters a vehicle cabin. Fixed measurements cannot be made in this zone when vehicles are moving. Measurements can, instead, be conducted from moving vehicles. However, the problem then becomes that only brief measurements can be made that will be more representative of an average along a length within the tunnel rather than a point measurement as in the case of the fixed sensor. For instance, an instrument that reports concentrations with a 1-minute resolution (or data that are averaged up to 1 minute) will have been carried 500 m – 1 km in that time at typical road tunnel speeds.

Boulter *et al*. (2004) report on such measurements made in the ~10 km long Plabutsch tunnel in Austria and the ~2.5 km long Kingsway tunnel in the UK. At the time of the experiment the Plabutsch tunnel was a bi-directional tunnel with transverse ventilation. This leads to an expectation that pollutant concentrations do not vary too much along the tunnel's length and that a single sensor mid-tunnel is reasonably representative of the whole tunnel on average. During an experimental campaign 2-minute average CO concentrations at a single in-tunnel monitor were compared to the average of 2-minute average CO concentrations measured outside a moving vehicle along the whole tunnel length. It was found that the sensor under-represented the road-level concentration by about 50 %, i.e. concentrations measured outside the vehicle were 50 % higher than those registered by the fixed sensor. The Kingsway tunnel has two uni-directional tubes with semi-transverse ventilation, leading to significant concentration gradients along the tunnel's length, confirmed by monitoring. Journey times are typically of the order of 2 minutes. During the campaign, CO concentrations were measured at various points along the tube. The average vehicle-based concentration was approximately 50 % higher than the average of the fixed point concentrations, a result consistent with the findings from the Plabutsch tunnel.

Another study compared measurements made from a moving vehicle with those made by a fixed instrument inside the M5 East Tunnel in Sydney (Holmes Air Sciences, 2005). This study focussed on NO and NO_2 and did not include observations of CO. The M5 East tunnel is 4 km long and heavily trafficked such that congestion regularly occurs within the tunnel. 80 transits of the tunnel were conducted in each tube over 39 days at different times of the day. Unlike in the studies cited above, the fixed instrument (a DOAS, or Differential Optical Absorption Spectroscopy device) was deliberately located at a point where maximum NO and NO_2 concentrations were expected, i.e. near one of the tunnel's permanent sensors (designated CP2) near the exit of the westbound tube. However, the data captured from the tunnel's sensors during the campaign indicate that this assumption may not have been correct. During the morning and mid-day periods the CP2 sensor reported concentrations within ±25 % of the average of all sensors in the westbound tube for both NO and NO_2. In the afternoon periods, the CP2 sensor data were up to 2 times higher than the average for all westbound sensors. The afternoon periods were characterised by regular congestion indicated by a more than doubling in average journey time compared to other periods. However, the accuracy of the CP2 sensor was not reported.

With a large dataset it was possible to analyse the statistics of the ratio between the fixed data and the average transit data for NO_2. The assumption that the DOAS was located at a point representative of maximum concentrations in the tunnel (despite evidence to the contrary from the permanent sensors) appears to be supported by this comparison. This analysis showed that on ~98 % of transits the moving NO_2 concentration was lower than that measured by the DOAS at the fixed point. The average fixed point NO_2 was larger than the transit peak 30-second average 75 % of the time.

Extensive experiments in two tunnels in Hong Kong considered this issue (HKPU, 2005) but compared the tunnel company's fixed permanent sensors with the output of additional campaign measurements. The campaign measurements were made in access tunnels at a height of 1.5 m above the roadway and compared with the nearest roof-mounted permanent sensors in the Tseung Kwan O and Shing Mun (northbound and southbound tubes) tunnels. This study highlighted some significant problems with inaccuracy in the permanent sensors, partly caused by drift. In general, average road-level concentrations of CO exceeded those recorded by the fixed sensors by less than 50 % in the Tseung Kwan O tunnel, and between 50 and 100 % in the Shing Mun tunnel, with the exception of the southbound tube in winter, when road-level concentrations exceeded roof level by a factor of 3.5. There were insufficient data in the relevant report to assess the degree to which this discrepancy could be explained by other factors (e.g. sensor accuracy, lateral location, averaging time, etc.).

Data were also gathered for NO, which is also monitored routinely in these tunnels. The results were much less consistent. Concentrations at road level exceeded those reported by the fixed monitors on average by a factor ranging from 2 to 30. The linear correlation between the measurements was good in the Shing Mun tunnel (southbound) in summer but very poor in winter. Although it is possible that these differences may be in part explained by variations in nitrogen compound emissions and chemistry it is likely that unreliable and inaccurate sensors played a larger role.

It is difficult to generalise these relationships due to the lack of data, and the large number of influencing and confounding variables. Sensor reliability and accuracy is a major issue preventing confident conclusions being drawn in many cases. Beyond instrumental issues, the variability in concentration around the tunnel's cross-section is related to the degree of mixing of exhaust plumes. This mixing is strongly influenced by traffic-induced turbulence and will thus depend upon traffic volume, speed, the number of tall vehicles, and the tunnel geometry (shape, cross-sectional area, vertical and lateral clearance). A consequence of this is that the ratio between sensor data and road-level concentration could be quite dependent upon exact sensor location.

Implementation of an exposure-management approach in any given tunnel would benefit from an observational campaign to constrain the relationship between road-level and sensor data. Road-level data are most easily captured by instruments mounted on a moving vehicle, although a large number of transits encompassing the full spectrum of expected traffic conditions is recommended (not just during the daytime – concentrations can peak at night if ventilation is operating at a reduced or zero flow). More sophisticated techniques involving remote sensing could, in principle, be applied to this problem.

In conclusion, there is a tendency in the reported data towards a general finding that road-level concentrations are 50 % higher than those observed by a fixed sensor. In the interests of conservatism a value of 100 % could be applied. However, we recommend that tunnel-specific data (quality assured for a high level of accuracy) are gathered to better constrain this factor.

9.5 An example – the UK approach (DMRB)

The UK has implemented a national exposure management approach for road tunnels. This approach is detailed in the Design Manual for Roads and Bridges document BD 78/99 (Highways Agency, 1999). In the case of carbon monoxide, it provides for the implementation of a single guideline to three cases of tunnels of differing lengths. The base guideline adopted is based on the occupational exposure limit of 200 ppm[14] as a 15-minute average. This exposure limit is translated to a 'sensor limit' for each case,

[14] This more than twice the NIWA-recommended CO exposure limit of 87 ppm (100 mg m^{-3}). This is because it is an occupational limit based on a carboxyhaemoglobin limit of 5 %, appropriate for healthy adult workers, but not providing full protection for pregnant women or those susceptible to heart disease, for which a 2.5 % limit is adopted.

i.e. the concentration measured by a sensor installed in the tunnel that must not be exceeded in order to keep user's exposure within the exposure limit.

The translation combines two factors. The first is an increase in the permissible in-tunnel concentration for shorter tunnels based on the shorter exposure duration. To provide a conservative factor of safety it is assumed that all vehicles travel at 10 km h^{-1}. The second factor allows for in-vehicle concentrations being 3 – 6 times greater than those measured externally to a vehicle, e.g. by a tunnel sensor. The resulting sensor limits are listed in Table 9.1 For example, maintaining CO concentrations, as measured by in-tunnel sensors, below 100 ppm should ensure that no tunnel user is exposed to a level of CO equivalent (in terms of carboxyhaemoglobin response) to 200 ppm over 15 minutes, even though they will normally spend less than 2 minutes in the tunnel. A similar translation is performed for NO and NO_2 using the equivalent UK occupational exposure limits[15] of 35 ppm and 5 ppm respectively.

Table 9.1: Sensor limits for CO applicable in the UK (Highways Agency, 1999)

Tunnel length / m	Sensor limit / ppm	
	CO / ppm	NO_2 / ppm
< 500	100	4
500 – 1000	50	3
1000 - 2500	35	1.5
> 2500	Derived from first principles	Derived from first principles

It may be noted that the UK example leads to sensor limits which are more demanding than a hazard management limit would imply (200 ppm in the UK case, or 87 ppm following NIWA's recommendation) for tunnels longer than 500 m. However, it has been noted that some of the factors used in determining the sensor limits may be over-conservative and lead to excessive demands on ventilation. In particular, it has been noted experimentally that the factors relating concentrations inside a vehicle to that outside are based on ambient roadside measurements rather than in-tunnel measurements. Observations in three European tunnels (Boulter *et al.*, 2004) suggest

[15] These limits have since been withdrawn as occupational limits in the UK. Their use as the basis of the DMRB tunnel sensor limits is under review at the time of writing.

that the current factors may be 2 – 4 times too large (and hence sensor limits 2- 4 times higher could be tolerated).

9.6 In-vehicle exposure

9.6.1 Cabin protection and air quality management

Except in cases where pedestrians and cyclists are permitted to use a road tunnel, exposure **of the public** to pollutants inside road tunnels occurs within a vehicle cabin. This cabin is partially sealed to external air with infiltration restricted by openings including windows, vents and bodywork leaks. Although highly variable, this restriction introduces a lag in the response of internal concentrations to rapid rises in external concentrations as occurs when a vehicle enters a road tunnel. Consequently, internal concentrations of vehicle pollutants within a vehicle cabin will generally be lower than the concentrations outside the vehicle. Thus true exposure to those pollutants will be lower than that predicted by assuming the person is exposed to external road-level concentrations.

The management of air quality in a tunnel must be based upon the protection of the most vulnerable. In the case of cabin protection, the most vulnerable are those who have none – e.g. pedestrians, cyclists, occupants of convertible or open vehicles – followed by those who have limited protection – e.g. those who choose to travel through a tunnel in a vehicle with the windows open.

For those travelling in vehicles with a partially or well-sealed cabin, the reduced exposure in the tunnel may be offset by the extended exposure post-tunnel caused by the retention of some of the tunnel air that penetrated into the cabin. The air restriction now limits the infiltration of cleaner air so that air of tunnel origin, with elevated concentrations of pollutants, remains trapped in the vehicle cabin for an extended period. Thus exposure to tunnel air persists for some time after the vehicle exits the tunnel.

The following section considers how much extra protection is provided by a vehicle cabin whilst in the tunnel (and thus how much lower true exposure is compared to that assumed in an air quality management scheme) and to what degree this is compensated by post-tunnel exposure to trapped air.

9.6.2 Vehicle infiltration

The peak concentration arising inside a vehicle as it passes through a tunnel, and the time it takes for contaminated air to be replaced post-tunnel, are both dependent upon the air exchange rate (AER) of the vehicle. The AER, in turn, is determined by

a) whether the windows are opened or closed,

b) the operation of the vents and air conditioning,

c) the leakiness of the bodywork,

d) the speed of the vehicle,

e) the external wind speed and level of air turbulence.

Limited experimental data exist reporting air exchange rates of moving vehicles. While stationary, rates of 1 – 4 $hour^{-1}$ have been measured in cars with windows closed rising to 13 – 26 h^{-1} with windows open (Park *et al.*, 1998). There are fewer data for moving vehicles. However, Ott *et al.* (1994) reported rates of 1 – 2 min^{-1} at 20 mph with the driver's window open, the passenger's window open 3 inches and other windows closed. Rodes *et al.* (1998) reported rates of 13.5 – 39 h^{-1} at 55 mph with windows closed and vents on low. Zhu *et al*. (2007) estimated rates of 1 – 2 min^{-1} in a car with windows closed, but various ventilation settings during freeway driving in Los Angeles, averaging 50 – 60 mph. Batterman *et al.* (2006) reported 1.5 min^{-1} in a car at 100 km h^{-1} under 'low-to-medium' vent conditions. A study of AERs in a horse trailer (Purswell *et al.*, 2006) indicated a maximum AER of 1.42 min^{-1} at 97 km h^{-1} with all windows and vents open. Research conducted by NIWA in Auckland (Longley *et al.*, in preparation) found that air exchange rates in a typical car were of the order of 1 min^{-1}, but that this rate could be increased by an order of magnitude if windows were opened and decreased by almost an order of magnitude if the vents were closed and air recirculated.

In summary it appears that for road tunnel users in vehicles with small cabins values of 1 – 2 min^{-1} may be typical, perhaps biased towards lower values where driving with closed windows is more common.

In extensive controlled tracer studies in school buses exchange rates of 0.2 – 0.8 min^{-1} have been measured at 40 km h^{-1} with windows closed and 1.6 – 5 min^{-1} with windows open (Fitz *et al.*, 2003). Experimental studies on in-service buses in the UK have led to estimated rates of 0.14 min^{-1} (windows closed) and 0.35 min^{-1} (windows open) (Longley & Leavey, in preparation). These values are substantially lower than those above for smaller cabins, and presumably reflect the larger volume of air to be mixed.

In summary, the more sealed a vehicle cabin is, the lower its air exchange rate will be. A lower AER implies a slower internal response to changes in external concentrations, inevitably leading to lower internal concentrations whilst inside a tunnel, but a longer persistence of contaminated air within the vehicle after leaving the tunnel. In the following hypothetical case studies we investigate the effect of two scenarios which might be considered typical – a relatively well-sealed vehicle (windows and vents closed) with an AER of 0.2 min^{-1} and a relatively leaky vehicle (vents open, fans on, maybe a window slightly open) with an AER of 2.0 min^{-1}.

9.6.3 Effect of air exchange rate on indoor/outdoor ratios

The indoor/outdoor ratio (or I/O ratio) describes the extra degree of protection afforded to vehicle occupants by the vehicle cabin and the resistance it offers to infiltration of external contaminants. It can be approximately predicted if the external concentration and air exchange rate are specified on the basis of an infiltration model. The simplest model is a first-order model in which the change in internal concentration in a given time step is directly proportional to the difference between internal and external concentrations and the air exchange rate. For the following examples we will assume that upon entering the tunnel the external and internal concentrations are zero.

If we assume that AER is linearly proportional to speed (there is some evidence to show that this is approximately true – see AER references above) then, according to this simplified model, the I/O ratio at any point within a tunnel is independent of vehicle speed. This is because the decreased infiltration into a slower vehicle is compensated for by the extra length of time spent in the tunnel. Thus, the I/O ratio becomes a direct function of the constant of proportionality between AER and speed, i.e. it depends upon the porosity or 'leakiness' of the vehicle cabin. Figure 9.2 shows two scenarios intended to be representative of the ends of a wide range of typical vehicles and indicates that vehicle leakiness has a major impact on I/O. After 500 m, the I/O ratio is approximately 0.1 for the sealed vehicle and 0.5 for the leaky vehicle.

After 2 km concentrations in the leaky vehicle would be more than 90 % of the external concentrations in the tunnel air, whereas concentrations in the sealed vehicle would still be less than a quarter what they are outside the cabin. If we break the assumption that AER is directly proportional to speed, (provisional NIWA data suggest a power law, such that AER = kV^n where $n < 1$), then speed becomes a minor determinant of I/O, such that slower speeds will lead to slightly higher I/O ratios due to higher exchange than that predicted by the proportionality assumption.

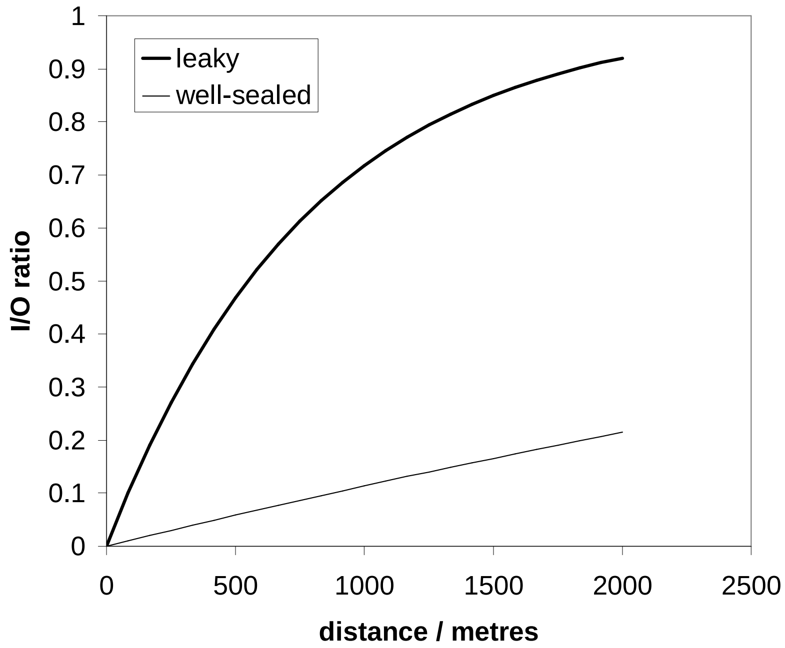

Figure 9.2: Dependence of I/O ratio on distance travelled along a tunnel, based on constant external concentration, zero initial concentration, a simple first-order infiltration model and vehicle AER directly proportional to vehicle speed for two cases (leaky: AER = 2 min^{-1} at 100 km h^{-1}, sealed: AER = 0.2 min^{-1} at 100 km h^{-1}).

The conclusion to be drawn from this is that the sealing of a vehicle cabin can be remarkably effective in reducing the true exposure of road tunnel users **whilst inside the tunnel**. Simply closing windows, closing vents (set to recirculate) and switching off ventilation fans can reduce internal concentrations in a vehicle cabin to a small fraction of the concentration inside the tunnel. This gain, however, must be considered in the context of the trapping of tunnel air in the vehicle cabin and subsequent post-tunnel exposure, as discussed below.

9.6.4 Post-tunnel exposure

After exiting the tunnel the tunnel air trapped in the vehicle is gradually replaced by 'fresh' air. For illustrative purposes we will assume that the removal is described by the same first-order model as pollutant infiltration. We initially model the effect of passage through a 2 km long tunnel with constant concentrations along its length. If we furthermore assume that vehicle AER is a direct linear function of vehicle speed then we can make the following broad conclusions. Firstly that for the 'leaky' vehicle internal concentrations return to close to zero on a similar timescale that the vehicle spent in the tunnel, independently of speed. For the 'sealed' vehicle, internal concentrations fall to 10 % of the peak value (which is lower than in the 'leaky' case) approximately after a period 10 times the time spent in the tunnel. I.e. if the tunnel passage took 2 minutes, then tunnel air could be trapped in the vehicle for around 20 minutes after it left the tunnel (albeit at reduced concentrations). A longer initial tunnel transit leads to a proportionally longer period of post-tunnel exposure. It is worth remarking that for a typical tunnel of length 1 – 4 km length, this retention of pollutants extends the exposure duration of the occupants towards the order of 15 minutes, i.e. the averaging time of conventional short-term exposure limits, including that recommended for road tunnels in New Zealand.

We can judge the biological significance of this post-tunnel exposure (for carbon monoxide only) using the Coburn-Forster-Kane model to predict the effect on blood carboxyhaemoglobin. We model the effects of passing at a constant speed through a tunnel containing a constant CO concentration of 100 ppm throughout. The CFK model parameters are set for a typical driving adult with baseline %COHb set to 0.3 %.

We firstly consider the **peak** %COHb response as a result of this transit and the subsequent trapping of CO in the cabin post-tunnel, as it is the peak which determines the health effects. The modelling firstly shows that peak %COHb is directly related to tunnel length. Figure 9.3 illustrates the model's prediction of the effect of vehicle speed for three vehicle infiltration scenarios (a 'well-sealed', a 'leaky' and an 'open' vehicle with I/O = 1) for passage through and beyond a 2 km tunnel. This shows that the totally open case encompasses the other cases. In the open case there is no retention of pollutants and so no post-tunnel exposure, yet the higher exposure in-tunnel more than compensates for this. Above 30 km h^{-1} the result is relatively insensitive to speed, but the effects of total exposure to **carbon monoxide only** are progressively worse at speeds below 30 km h^{-1}. A slow-moving totally open vehicle is therefore the worst-case for total exposure, judged in terms of carboxyhaemoglobin response, if one assumes that post-tunnel vehicle speed is the same as within the tunnel. In that sense, vehicle infiltration and pollution retention need not be considered for tunnel management purposes as an assumption that all vehicles are open is conservative.

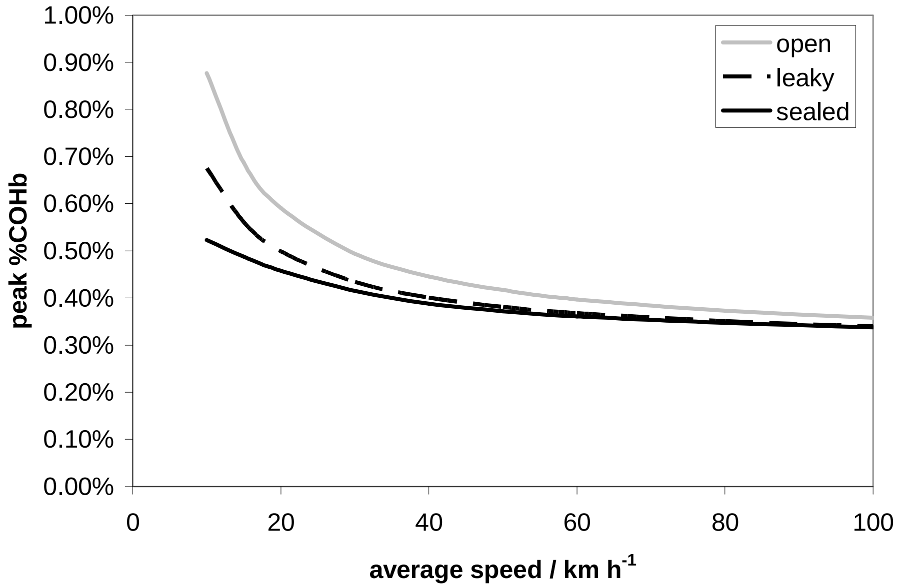

Figure 9.3: Peak %COHb reached, as predicted by the CFK model, by an adult in vehicles with three different air exchange rates as a function of average speed through a hypothetical 2 km long tunnel.

The high sensitivity to low speeds is useful for management purposes. Vehicle speed, in many tunnels, is less variable and more amenable to control, than the ventilation properties of vehicles. However, this is less true of urban tunnels where the journey of any given vehicle post-tunnel may vary more than in the case of a motorway or rural tunnel. Exposure management requires predictability of exposure, and figure 9.3 shows that the effect of total CO exposure, represented by the peak COHb response, is relatively predictable for all vehicles, regardless of air exchange rate, if a minimum speed of approximately 30 km h^{-1} can be assured.

Nevertheless there are many simplifying assumptions in this analysis which may need to be re-examined in the case of specific individual tunnels. The assumption that an open vehicle represents the worst-case is invalidated in the case where vehicle speed falls after exiting the tunnel. In this case, air accumulated in the vehicle during the tunnel transit is removed from the cabin post-tunnel at a slower rate, thus prolonging exposure.

Figure 9.4 shows the time series of %COHb response in a well-sealed, leaky, and open vehicle from the same modelling exercise, assuming a constant speed of 30 km h^{-1}. At this speed the passage through the tunnel takes 4 minutes, and the end of the tunnel can be observed in the sudden change in slope in the 'open' line. A fourth scenario is included; "sealed-leaky'. This is intended to describe a passage through the tunnel under 'sealed' conditions, but changing to 'leaky' conditions at the tunnel exit. This could be represented by opening the driver's window and switching on the fan. This scenario shows a dramatically lower exposure and consequent %COHb response. A more dramatic reduction can be achieved by setting the vehicle vents to 'recirculate' (with windows also closed) whilst in the tunnel, as long as the vents are re-opened upon exiting. This re-opening is crucial as otherwise the small amount of pollution which did penetrate the vehicle will be retained in the cabin for a considerable length of time.

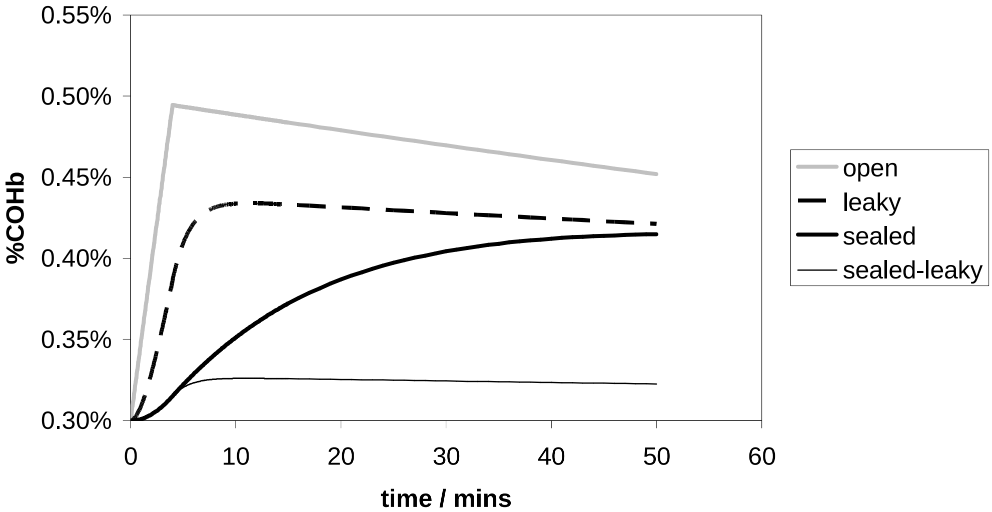

Figure 9.4: predicted %carboxyhaemoglobin response resulting from passage through a hypothetical tunnel in three cabins: sealed, leaky and open (no cabin, see text for further explanation).

9.7 Summary and conclusions

Exposure management for road tunnels is more technically demanding and subject to more uncertainties than a more conventional ‘hazard management’ approach. Implementation of exposure management must, therefore, offer significant advantages over hazard management to justify its adoption. A key advantage is potential energy savings from reduced ventilation, the opportunities for which are greatest in shorter tunnels and those with a higher minimum speed.

We recommend air quality guidelines based upon 15-minute averaging times. Whereas hazard management applies an air quality guideline to the tunnel air, exposure management applies any guideline to individuals using the tunnel, who, in most cases, will be exposed for less than 15 minutes. The pollutant concentrations they are exposed to will usually differ from the concentrations measured by tunnel sensors due to local spatial variability in concentrations and the partial protection provided by vehicle cabins.

Exposure management is based primarily on the recognition that exposure to a higher concentration of carbon monoxide for a shorter period can have an equivalent effect on the body (represented by blood carboxyhaemoglobin levels) as exposure to a lower concentration for a longer period. The same assumption has been applied to exposure to other pollutants, although this is based on weaker scientific evidence. This means that a given air quality guideline applied as a personal exposure limit may translate to a higher sensor limit (i.e. permissible concentration in the tunnel air) if it is possible to confidently specify a maximum probable exposure duration (i.e. journey time through the tunnel). This may be assisted if a minimum speed limit can be successfully implemented and enforced in the tunnel.

The second principle of exposure management is the recognition that because tunnel sensors cannot be located at the point where air enters vehicle cabins (where pollutant concentrations are generally higher) they generally under-represent concentrations at the most exposure-relevant part of the tunnel. There are limited observational data regarding this issue, but there is a tendency in the reported data towards road-level concentrations being 50 % higher than those observed by a fixed sensor. In the interests of conservatism a value of 100 % could be applied. However, we recommend that tunnel-specific data (quality assured for a high level of accuracy) are gathered to better constrain this factor.

The UK has implemented a national exposure management approach for road tunnels. The approach has been considerably conservative and has not addressed the third principle described below, i.e. infiltration into vehicle cabins.

The restriction on infiltration of air into a vehicle cabin reduces occupant exposure relative to pollutant concentrations as indicated by tunnel sensors. However, tunnel ventilation management systems must be designed to protect the health and safety of all users. Consequently this extra protection cannot be assumed as some tunnel users will not have the protection of a cabin (motorcyclists, occupants of open-top cars, cars with windows fully down, etc.).

When a vehicle with a partially or well-sealed cabin exits a road tunnel it retains some of the tunnel air that penetrated into the cabin. Thus exposure to tunnel air persists for some time after the vehicle exits the tunnel. If the vehicle maintains a constant speed then the total effective exposure of the occupants of a vehicle with a partially or well-sealed cabin is still lower than the case of a totally open vehicle. Consequently, **pollutant retention need not be considered in exposure management in this case as an assumption that all vehicles are open is conservative**.

This assumption may be invalidated if vehicle speeds post-tunnel are lower than in the tunnel itself. A longer initial tunnel transit leads to a proportionally longer period of post-tunnel exposure, and it is vehicle speed which is the strongest determinant of physiological response to CO exposure (assuming a constant speed both within the tunnel and post-tunnel), with a stronger response at slower speeds. This is unlikely to occur as exposure management must be designed on the assumption of the minimum average speed in the tunnel (e.g. 10 km h^{-1} in the case of the UK's DMRB approach), but could be significant if a tunnel with a relatively high minimum speed delivers traffic into a congested link or network.

Another exception could occur in the case of multiple tunnels. If a vehicle enters a second tunnel before the concentrations of pollutants in the cabin has sufficiently decayed following passage through the first tunnel (the decay time is tunnel length and speed dependent and vehicle dependent, but is typically tens of minutes) then exposure in the second tunnel is cumulative on top of the exposure arising from the first tunnel. A third exception could occur when a vehicle makes multiple trips through the same tunnel with the time gap between trips being shorter than the decay time for CO in the cabin.

Nevertheless, simply closing vehicle windows, switching off ventilation fans, and especially setting the vents to ‘recirculate’, is very effective in reducing internal concentrations in a vehicle cabin to a small fraction of the concentration inside the tunnel, and is probably the most effective means of reducing the risks to the health and safety of individuals.

In summary, tunnel ventilation systems should be designed to the worst-case for exposure, which is low speeds and fully-open vehicles. Whether a vehicle cabin is open or sealed, the main determinant of the impact of carbon monoxide on health is the length of time spent in the tunnel. Decreasing the rate of infiltration into a vehicle cabin can greatly reduce concentrations inside the vehicle. Setting air vents to ‘recirculate’ is a highly effective mitigation measure for individuals, but only if the vents are re-opened once the tunnel transit is complete.

10. Implementation and enforcement of a NO_2 limit

10.1 Overview

- Levels of NO_2 in a tunnel, and variation along its length, are very hard to predict because they are influenced by the complex (and poorly understood) influences of ambient ozone, entrainment of ambient air, ventilation rates and the significance of several chemical reactions in tunnel air.

- Prediction of in-tunnel NO_2 levels in response to NO_x emissions is difficult to predict, and probably tunnel-specific. However, our review of available observational data suggests that the NO_2/NO_x ratio recommended by PIARC (2000) is generally conservative. However, we have noted exceptions to this rule, such that NO_2/NO_x may be higher a) when there is a high proportion of heavy duty vehicles, b) in long tunnels with weak airflow, c) during congestion, d) in short tunnels and e) approaching tunnel portals.

- Implementation of a monitoring compliance guideline for NO_2 is especially technically challenging. NO_2 monitors typically demand a very high degree of maintenance, which can only be relaxed at the expense of a considerable loss of accuracy.

- The main alternative to direct monitoring is to monitor it indirectly via an algorithm which relates NO_2 to monitored CO, NO concentrations and/or visibility. This approach is highly dependent upon having credible data to base the algorithm upon, ideally real tunnel-specific observations (our current state of knowledge, and the tunnel-specific nature of the determinants of NO_2 mean that theoretical data alone is highly unlikely to be sufficient). Without tunnel-specific data these ratios can be estimated using emission databases or other relevant sources, although such an approach is likely to have significant uncertainties and should not be relied upon alone.

10.2 Nitrogen chemistry in tunnels

10.2.1 Sources and emission control of NO_2

NO and NO_2 (together considered as NO_x) are derived from combination of the two major components of air (N and O) in the high temperatures of internal combustion engines. Most NO_x is emitted as nitric oxide (NO), which then indirectly forms nitrogen dioxide (NO_2) through reaction with ozone on a timescale of seconds:

$$NO + O_3 \rightarrow NO_2 + O_2$$

NO can also be oxidised to NO_2 through the free radical-catalysed oxidation of volatile organic compounds (VOCs):

$$NO + XO_2^{\cdot} \rightarrow NO_2 + XO^{\cdot}$$

These indirect links between emission of NO and concentrations of NO_2 are crucial as the formation of most of the NO_2 from vehicle exhausts requires an oxidant – either ozone or free radicals in the presence of VOCs. Ozone entering a tunnel will be rapidly depleted. There is no shortage of VOCs in vehicle exhaust, but free radicals are largely produced by photochemical mechanisms and have a very short atmospheric lifetime. Thus they are unlikely to be entrained very deeply into a tunnel, nor generated photochemically within the tunnel. Alternative mechanisms exist, but the presence and activity of free radicals in road tunnels is unknown. The consequence is that the level of NO_2 in a tunnel is limited by the availability of oxidants from the outside air. Tunnel length and ventilation scheme become the crucial variables.

10.2.2 Impact of ventilation on nitrogen chemistry

The dependence of the transformation of NO to NO_2 upon the presence of ozone (or oxygen in extreme cases – see section 10.2.6 below) implies that the rate of ventilation plays an additional and complex role other than just the dilution of pollutants. Shortly beyond the entrance to the tunnel ozone is entrained from the external air into the tunnel and provides for an initial ozonation of NO. In the case of a longitudinal tunnel this ozone will be rapidly depleted by this process so that the ozonation reaction becomes progressively less active along the tunnel length. Thus a decreasing proportion of the emitted NO is converted into NO_2 by this reaction. [16]We may expect NO_2 to still rise along the tunnel's length due to its direct emission, and a reduced rate of NO conversion due to the remaining ozone, but the NO_2 / NO_x ratio will fall along the tunnel's length (evidence is provided below). If the tunnel is long enough for oxidants to be largely depleted then, in the absence of any NO_2 production mechanism, the NO_2/NO_x ratio should tend towards the average emission ratio in the vehicle fleet.

In the case of a semi-transverse or transverse tunnel fresh air and fresh ozone is being injected along the length of the tunnel. In this case new oxidants may be entering the tunnel at all points and will not be so rapidly depleted. In principle this allows a greater rate of ozonation leading to potentially higher NO_2 concentrations and a higher NO_2 / NO_x ratio. In practice, however, the variation of NO, NO_2 and O_3 along the tunnel length is hard to predict due to

1. variations in external levels of ozone,

2. variations in ventilation rates,

3. varying degrees of entrainment of air through the entrance,

4. other reactions and processes, such as reactions with free radicals.

10.2.3 Transect observations of NO_2 in a simple urban tunnel

Transects of NO_2, NO_x and CO were conducted (3 each way) in winter 1993 and summer 1994 in the Söderledstunnel, Stockholm, a 1.5 km long tunnel which is naturally ventilated in most cases, and has moderate – high traffic flow (SEHA, 1994, 1995). Data from the 1993 study are reproduced in figure 10.1. Peak concentrations of NO_2 and CO were observed towards the tunnel exit. There was a fairly steady rise in NO_2 concentration with depth that was not significantly different from the profile of CO or NO_x. These data indicate that the rate of oxidation did not significantly vary along the tunnel length.

[16] However, the tunnel contains high concentrations of VOCs and this provides a potential alternative conversion route from NO to NO_2 via free-radical catalysed VOC oxidation. Studies have shown that HO_x can be produced in vehicle exhausts due to a thermal reaction between NO_2 and conjugated dienes which may be present in exhaust (Shi & Harrison, 1997). The reaction is relatively slow and we are not aware of any studies of the action and strength of this reaction, or the presence of free radicals, in road tunnels.

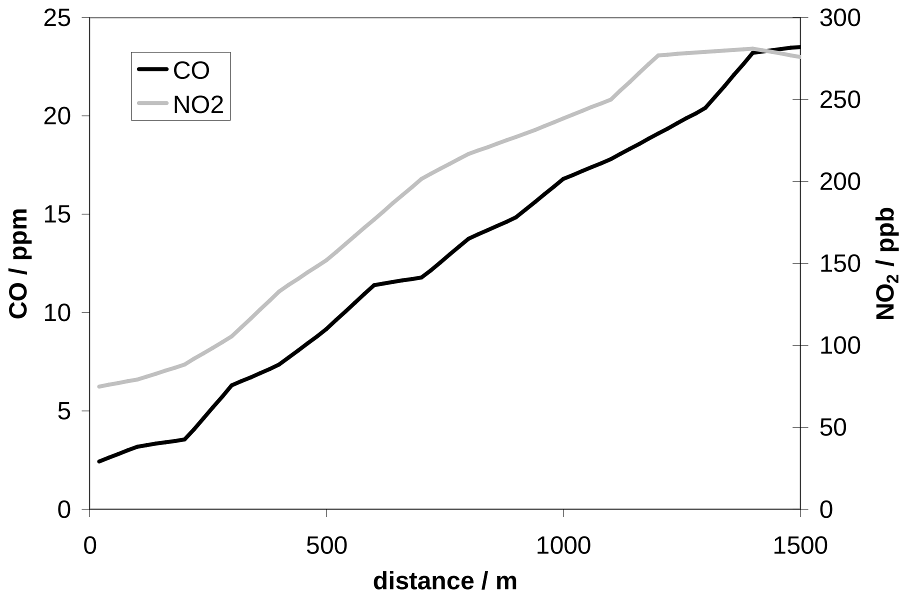

Figure 10.1: Average variation (over three journeys) in CO and NO_2 concentrations measured from a moving vehicle in the 1500 m long Soderledstunnel, Stockholm (SEHA, 1994).

10.2.4 Detailed observations of nitrogen chemistry from 2 long tunnels

Five transects of NO, NO_2 and O_3 were measured in the Tai Lam (northbound and southbound) and Tate's Cairn tunnels in Hong Kong between September 2004 and May 2005 (Yao *et al.*, 2005). Both tunnels are nearly 4 km long, but the Tai Lam tunnel has lower traffic overall, but a larger proportion of HDVs including vehicles originating or fuelled in mainland China.

In each transect concentration of NO peaked in the second half of the tunnel with maximum concentrations of around 1 ppm in the Tai Lam northbound, and approaching 3 ppm in the other tunnels, which have higher emissions (figure 10.2, left). Ozone was rapidly depleted with depth in all tunnels, although never reaching zero (figure 10.2, right). The Tai Lam tunnel is semi-transverse so it is possible that fresh ozone-laden air is injected along the tunnel's length. The data appear to show lower concentrations of ozone in the first half of the Tai Lam tunnel compared to the Tate's Cairn, although concentrations increase over the second half. As well as fresh ozone input from the ventilation system the increase in O_3 towards the end of the tunnel may also be due to entrainment. The concentration of ozone in the Tate's Cairn tunnel did not recover in this way, except in the final 800 m or so approaching the exit.

As a result, NO_2 was found to initially fall due to dilution of entrained air. Beyond about 1500 m depth direct emission and transformation of NO overcame the dilution and NO_2 rose. In the mid-sections NO and NO_2 rose rapidly even though O_3 was a minimum. These rises can be attributed to accumulation of emissions and a low rate of oxidation. However, in the latter section of the tunnels NO fell while NO_2 continued to rise, suggesting that conversion was progressing faster than accumulation. This is partly explained by the slight rise in ozone in this section, but the amount of ozone was insufficient to explain the size of the rise in NO_2. A secondary process must have been acting to increase NO_2 to this degree. Free-radical catalysed VOC oxidation is possible (but no data were collected to indicate this), but the high concentrations of NO, especially in the Tate's Cairn and Tai Lam southbound tunnels, does not rule out that direct reaction of NO with oxygen (see section 9.1.6 below) may have been become significant in the final kilometre of these tunnels.

The mean and maximum observed concentrations (averaged over 5 trips) are presented in Table 10.1.

Table 10.1 Mean and maximum concentrations observed in the southbound tube of the Tai Lam tunnel (Hong Kong) as observed by Yao *et al*. (2005)

	NO_2 / ppb	NO_x / ppb
Mean	52	1331
Maximum	82	2720

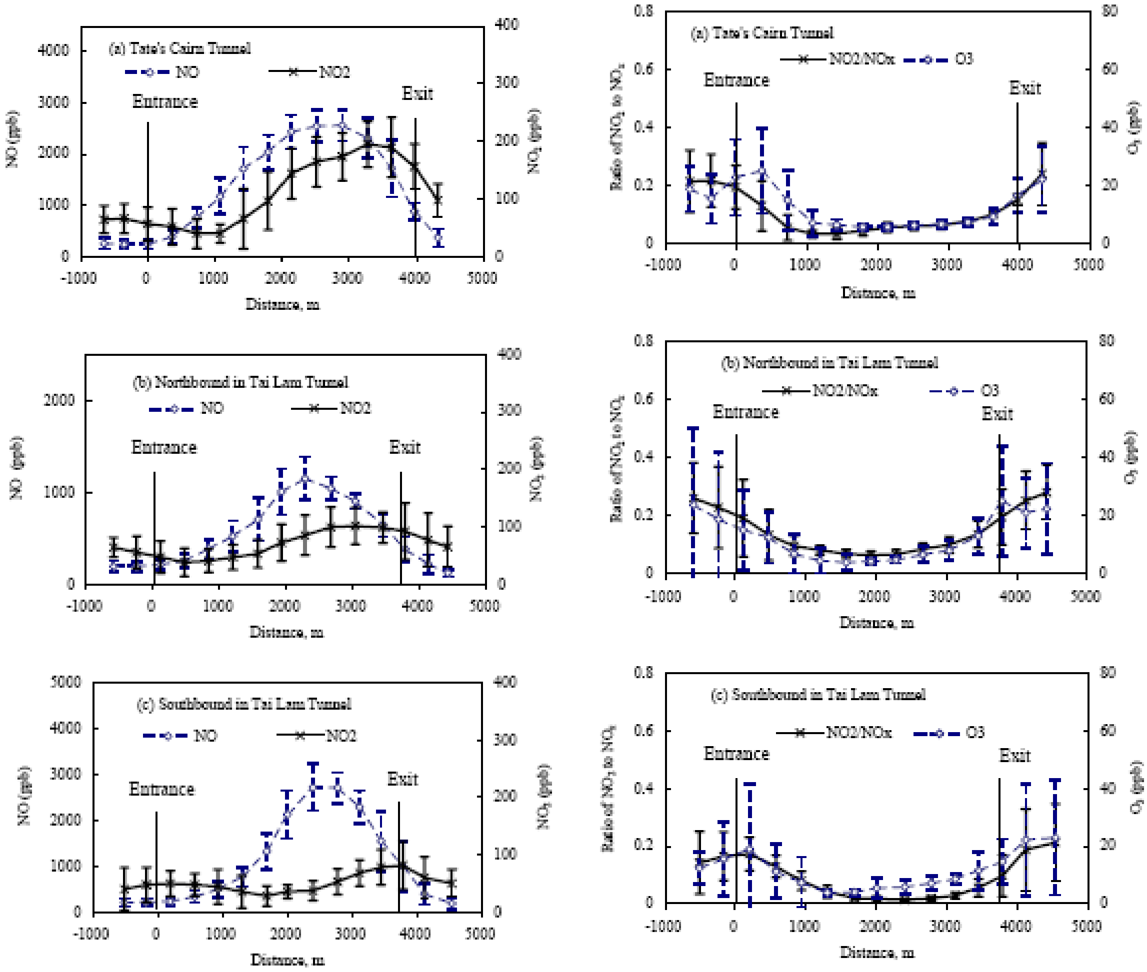

Figure 10.2: transect profiles of NO and NO_2 (left), NO_2 / NO_x and O_3 (right) from the Tate's Cairn and Tai Lam tunnels, Hong Kong (from Yao *et al.*, 2005).

10.2.5 Transect observations of NO and NO_2 in the more complex M5 East tunnel

At 4 km long Sydney's M5 East tunnel is similar in length to the Tai Lam and Tate's Cairn tunnels but carries a greater volume of traffic. A simple linear relationship with depth along the tunnel is not expected in the M5 East due to its unusual ventilation layout in which fresh air is injected and vitiated air removed near the mid-point of both tubes. Tunnel air is not released at any of the portals as air is transferred from near the exit of one tube to near the entrance of the other at both ends. The consequence is that one may expect a 'sawtooth' transect profile, with maximum concentrations recorded near the tunnel mid-point just before the air exhaust point.

160 transects were performed (80 in each direction), recording vehicle-exterior concentrations of NO and NO_2 (but not ozone), in an intensive six day study in the M5 East tunnel in Sydney in March/April 2004 (Holmes Air Sciences, 2005). Transects were restricted to three daytime periods (6am – 9am, 11am – noon and 3pm – 6pm). Concentrations were reported as 30 second averages. With an average transit time of 5.2 minutes this typically provided on the order of 10 data points per transect.

As expected, maximum concentrations of NO and NO_2 were recorded on approach to the exhaust points. Average concentrations of NO at the exhaust points were in the range 4 – 5 ppm (westbound) and 4 to just over 6 ppm (eastbound) – significantly higher than in the Tai Lam and Tate's Cairn tunnels (see above). Average concentrations of NO_2 at the exhaust points were in the range 0.2 – 0.3 ppm (westbound) and 0.2 – 0.35 ppm (eastbound). Although there was significant random variation in profiles between transects on different days, on average the expected 'sawtooth' profile was observed in all three time periods in both eastbound and westbound tubes, although the pattern was slightly clearer in NO_2 than NO.

34 individual profiles were presented in the Report. In general NO frequently exceeded 4 ppm at one or several points along the transect, but rarely exceeded 8 ppm. NO_2 reached ~0.4 ppm at some point of the transect on 13 out of the 34 presented profiles, and significantly exceeded 0.4 ppm on 3 profiles. A maximum of ~ 0.8 ppm was recorded.

10.2.6 Potential for excess chemical production of NO_2

If concentrations of NO_x are high enough (~ 2 ppm) then a secondary (termolecular) reaction with oxygen may become significant:

$$2\,NO + O_2 \rightarrow 2\,NO_2$$

This reaction is second order with respect to NO, and there is no shortage of oxygen in any tunnel, and thus it could lead to rapid NO_2 production when NO is high. In theory, this might occur in longer tunnels or when air flow in the tunnel is insufficient to dilute NO emissions, and is another reason why NO_2 is likely to be higher near the tunnel exit. In many tunnels ventilation is provided by the movement of the vehicles themselves, so in congested conditions NO_x emission is higher and the air flow is reduced leading to a 'worst case' scenario for NO_2.

The activation of the termolecular oxygenation of NO has been cited to explain severe NO_2 episodes in ambient air (e.g. the London smog episode of December 1991, Bower *et al.*, 1994), but its occurrence in road tunnels may be rare, and probably limited to very long or poorly ventilated tunnels. However, its true prevalence is unknown due to the fact that NO_2 is rarely monitored in tunnels.

One study that may have observed this reaction occurring and its consequence for air quality management is that of Indrehus & Vassbotn (2001). NO_2 and NO_x (as well as CO) were measured in the 7.5 km Hoyanger tunnel in Norway for 20 days in spring 1994 and 25 days in spring 1995. Traffic flows are relatively low in this tunnel (means of 28.9 h^{-1} in 1994 and 18.9 h^{-1} in 1995) and this meant that a bi-directional design was more economical. Ventilation here is nominally driven naturally by the pressure differences associated with the portals being at different altitudes. The piston effect is negligible due to the low traffic travelling in opposing directions in the same tube. A longitudinal system is installed to be triggered by CO concentrations exceeding a certain level. The Norwegian Public Roads Administration decrees that road tunnels should be closed if the concentration of CO at the tunnels mid-length exceeds 100 ppm for longer than 15 minutes (NPRA, 2004). This rarely occurred in the Hoyanger tunnel, yet users regularly complained of poor visibility and foul odours. Unlike most relevant authorities worldwide the NRPA has also set an in-tunnel pollution limit for NO_2, of 0.75 ppm at the tunnel midpoint and 1.5 ppm at the tunnel ends. Like CO, if this limit is exceeded for more than 15 minutes the tunnel should be closed. However, NO_2 was not routinely monitored as CO monitoring is more reliable (see chapter 8) and, to quote Indrehus & Vassbotn:

"…the CO concentration has been assumed to be the main source of poor air quality."

Their study set out to investigate if the NO_2 guideline was being exceeded, and why, by installing monitors in the tunnel 2 km from one end.

The monitoring revealed that the 1.5 ppm NO_2 limit was exceeded 17 % of the time in 1994 and 1.3 % of the time in 1995. The difference may in part be due to the significant reduction in traffic in 1995 due to the construction of a new road, and will otherwise be due to random variation in ventilation caused by meteorology. The wind speed within the tunnel varied between 0 and 2 m s^{-1} in both directions (which may be compared with the values of 2 – 6 m s^{-1} in one consistent direction typical in busy uni-directional urban tunnels). The highest NO_x values were all associated with the lowest wind speeds. A reduced conversion of NO to NO_2 in an ozone-limited atmosphere was observed with increasing NO_x up to a value of ~ 1 ppm. Oxidation was at a minimum at 2 ppm of NO_x. When air flow was generally less than 1 m s^{-1} NO_x often rose above 5 ppm despite the relatively low traffic flow, due to the lack of ventilation. Above this level there was an indication of an extra source of NO_2 consistent with the activation of oxidation with oxygen. These high values had not been observed by the control system, which only monitored CO. CO remained below its 100 ppm limit throughout (the maximum observed value was 58 ppm), so the tunnel had remained open despite NO_2 rising to a maximum of 5.87 ppm, far above its limit value. In this study a rough calculation showed that at an air flow of 0.5 m s^{-1}, with 6 ppm of NO and an initial 1.5 ppm of NO_2, the air would remain in the tunnel for 4.1 hours before exiting, in which time oxidation of NO with oxygen would produce a further 1.3 ppm of NO_2, i.e. nearly doubling the NO_2 concentration.

Whether this is common around the world, or whether it occurs in New Zealand's tunnels is unknown due to lack of data. Nevertheless, the data from Indrehus & Vassbotn (2001) indicate the potential for large increases in NO_2 in a tunnel where NO exceeds a threshold which cannot be specified with any certainty at this point, but is in the region of 2 – 5 ppm. This could correspond to an NO_2 concentration of around 0.1 – 1 ppm (assuming a NO_2/NO_x ratio between 5 and 20 %, see section 10.3). We have noted above how such concentrations have also been observed in busy tunnels with good airflow (Tai Lam, Tate's Cairn and M5 East) as well as in the low-traffic, low-airflow Hoyanger tunnel. Pending any more detailed observations, which we highly recommend, we suggest that the unpredictable and potentially harmful effects of the termolecular reaction could be minimised by adoption of an in-tunnel NO_2 limit of 0.4 ppm or lower so as to prevent the reaction from leading to excess NO_2 production.

10.3 Predicting NO_2 concentrations from NO_x data

10.3.1 Overview

The most direct way to predict NO_2 concentrations in a tunnel is to combine a NO_2/NO_x ratio with NO_x concentration data, if available, or NO_x emissions data. NO_x concentrations within the tunnel can be estimated if the NO_x emissions are known, along with the air flow rate and the background (external) concentrations (which, as a first approximation, can be estimated). The value of the NO_2/NO_x ratio varies between tunnels, varies with traffic flow and fleet mix, and varies along the tunnel length. The more accurately all of these variabilities can be quantified the more accurate the prediction of NO_2 concentrations will be. However, this is very demanding in terms of monitoring and NO_2 in particular is difficult to monitor accurately inside tunnels (see below). The simplest approach is to use a single conservative NO_2/NO_x ratio. PIARC (2000) reviewed the small amount of international data available at the time and recommended a value of 0.1, or 10 % be adopted. Below we briefly review more recent data and consider whether this is appropriate, and how real NO_2/NO_x ratios differ.

10.3.2 NO_2/NO_x ratio in simple tunnels

Variation of NO_2/NO_x with depth can be constructed from the transect studies in the Söderledstunnel (Stockholm), reported above (SEHA, 1994, 1995). In both winter 1993 and summer 1994 surveys the NO_2/NO_x ratio fell from over 10 % externally to approximately 6 % along most of the length of the tunnel. When corrected for background contributions this corresponds to an approximate emission ratio of 5 % inside the tunnel. This indicates that the rate of oxidation did not significantly vary along the tunnel length.

10.3.3 NO_2/NO_x ratio in long or complex tunnels

Very low NO_2 / NO_x ratios (a minimum of 2 %) were observed deep inside the 4 km long Tai Lam tunnel (Hong Kong) in the transect study of Yao *et al*. (2005) (figure 10.2). The low rate of NO oxidation mid-tunnel suggests that the NO_2 / NO_x ratio in this section should resemble the ratio of NO_2 to NO_x in the vehicle exhaust. NO_x emissions were higher in the Tai Lam southbound as use of high-sulphur fuel is believed to be higher and emission standards generally lower on average in this tube. This will lead to higher direct NO emissions per vehicle, but not NO_2. In the Tai Lam northbound and the Tate’s Cairn the NO_2 / NO_x ratio in the mid-section was considerably higher (~ 6 %), so that both tunnel-mean and maximum NO_2 concentrations were higher (maxima of the order of 100 ppb in the Tai Lam northbound and 200 ppb in the higher-traffic Tate’s Cairn tunnel.).

During the intensive six day study in the busy 4 km long M5 East tunnel in Sydney (Holmes Air Sciences, 2005) most of the transect data presented were indicative of a NO_2/NO_x ratio of 5 – 6 %, with slightly higher values near the fresh air entry points of the tunnel where concentrations are lower. The highest NO_2 concentration observed lasted for over a minute, peaking at above 0.8 ppm. This was towards the end of a westbound transit (31st March 2004) at around 16:17. Congestion was indicated by the long transit time in this case (in the 10 – 15 minutes band) and corresponded to a peak in NO (~ 12 ppm). The values here indicate a NO_2/NO_x ratio slightly raised above the typical daytime value for this tunnel of 5 – 6 %. This may be attributable to an additional source of NO_2 becoming activated in these conditions, but there is insufficient evidence to assess this proposition. As the NO_2/NO_x ratio deep inside the tunnel is expected to be an indicator of the NO_2/NO_x emission ratio, then temporary or localised changes in this concentration ratio may reflect changes in the emission ratio (see chapter 7). Thus, NO_2 concentrations may increase beyond what may predicted by assuming a constant NO_2/NO_x ratio in the case of congested conditions.

In an accompanying part of this study measurements of NO and NO_2 were also made using an open path UV monitor (OPSIS R600 DOAS) located near in-tunnel monitor CP2. This monitor is located near the westbound exit portal, which was considered likely to record the highest concentrations of NO_2 in the tunnel due to the gradient of the exit ramp. Clear and consistent diurnal patterns were observed matching traffic flows. Weekday concentrations were higher than weekends despite similar traffic volumes, however heavy duty vehicle flows were higher on weekdays. The daytime NO_2/NO ratio derived from this instrument was typically ~ 5 % (most values within 3 – 7 %). At night some higher values were found, consistent with lower emissions and a lower likelihood of oxidant depletion. In total a maximum of 2 % of values were above 10 % and all values were below 20 %. The 2 % of values in which the ratio was above 10 % occurred late at night at low NO concentrations, and thus it was concluded that a ratio of 10 % would be appropriate and conservative for this tunnel.

In the study in the 7.5 km long Hoyanger tunnel (Indrehus & Vassbotn, 2005) the NO_2/NO_x ratio decreased with increasing NO_x up to a value of ~ 1 ppm consistent with a reduced conversion of NO to NO_2 in an ozone-limited atmosphere (figure 10.3). The minimum ratio was approximately 10 %, albeit with scatter in the range 2 – 20 %. Above this level there was a clear increasing trend in NO_2/NO_x indicating an extra source of NO_2 consistent with the activation of oxidation with oxygen. At 5ppm NO_x, the ratio was approximately 12 – 15 %.

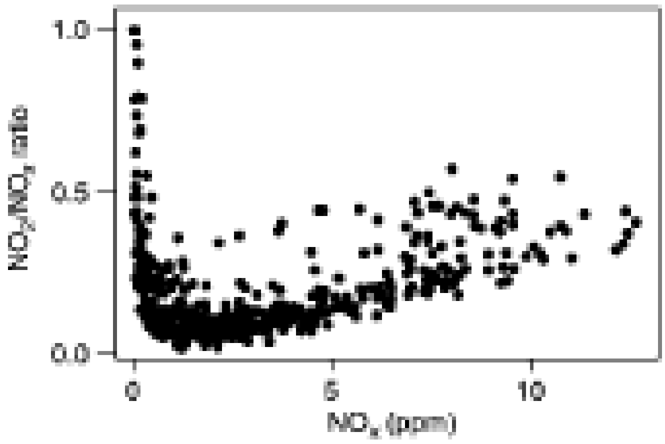

Figure 10.3: NO_2/NO_x ratio as a function of NO_x concentration in the Hoyanger tunnel, Norway, as observed by Indrehus & Vassbotn (2001).

In summary, we find that the 10 % ratio recommended by PIARC (2000) is generally conservative. However, we have noted exceptions to this rule, such that NO_2/NO_x may be higher in the following cases:

- Where the NO_2/NO_x emission ratio is high due to penetration of new technology vehicles or a high proportion of HDVs (see chapter 7),
- In long tunnels (few km) with weak airflow,
- During congestion,
- In short tunnels (< 500 m),
- Approaching tunnel portals.

10.4 NO and NO_2 monitoring

Several studies have noted the particular difficulties involved in the reliable and accurate monitoring of NO and/or NO_2 in road tunnels. Jacques & Possoz (1996) noted that the chemi-luminescence method, used routinely in ambient monitoring stations around the world, is accurate only up to approximately 500 ppb.

> " …its implementation as a practical measurement technique results in a very sophisticated and expensive instrument that must be handled with care. To our knowledge no such devices have ever been specially designed to be installed in the adverse environmental conditions that exist in a tunnel… The full scale measurement range is programmed to be 1 ppm."

Indrehus & Aralt (2005) note that chemi-luminescence monitors demand a very high degree of maintenance due to their exposure to high aerosol levels, representing a high cost that needs to be justified. An alternative is to use electrochemical sensors, but these have very low accuracy (of the same order as the NO_2 in-tunnel limit values). A low-maintenance open-path optical technique (DOAS, or differential optical absorption spectroscopy) is employed to monitor NO at 2 locations per tube in the M5 East tunnel (Sydney), but this also has a low accuracy with respect to the job it is required to do, especially in optically turbid conditions.

10.5 Managing NO_2 and CO via CO monitoring

One option for implementing a NO_2 guideline but avoiding problems with NO_2 monitors is develop an algorithm which relates NO_2 to CO concentrations. If this can be shown to be reliable then NO_2 concentrations can be estimated from CO measurements. The NO_2 guideline value can also then be translated to a corresponding CO limit. This will be relatively simpler to achieve in a shorter tunnel in which the conversion of NO to NO_2 is less likely to be limited and in a tunnel with a more predictable traffic flow (e.g. busy, but carrying mostly a local fleet of regular users).

As a simple illustrative example, assume that the fleet-averaged emission factors for a given tunnel are 5 g km^{-1} for CO and 1.2 g km^{-1} for NO_x, giving a NO_x/CO ratio of 0.24. Background concentrations are assumed to be negligible. Furthermore we assume a NO_2/NO_x ratio of 0.1. If a NO_2 guideline of 0.4 ppm (0.75 mg m^{-3}) applies, then the CO concentration in the tunnel corresponding to the NO_2 limit being reached would be given by

$$\left(\frac{NO_{2\max}}{NO_2 / NO_x}\right) / \left(NO_x / CO\right), \qquad = \left(\frac{0.75}{0.1}\right) / \left(0.24\right)$$

= 27.2 ppm (showing in this hypothetical example that the NO_2 limit is three times more demanding than a 87 ppm CO limit)

This approach is highly dependent upon the accurate representation of the NO_2/NO_x and NO_x/CO ratios. Without tunnel-specific data these ratios must be estimated using emission databases or other relevant sources. Multiple sources are recommended due to the large uncertainties involved. The simplest approach is to assume that NO_2/NO_x and NO_x/CO are both constant, but this could lead to large errors. In practice, these ratios will be sensitive to the traffic fleet mix and any changes in flow, such as speed changes and any incidents of congestion. In the case of predictable patterns (such as diurnal patterns) diurnally varying values of the NO_2/NO_x and NO_x/CO ratios should be applied. The factors used should be chosen so as to represent the actual traffic fleet mix and flow in the tunnel, including predictions into the future. We recommend that this emission modelling include alternative scenarios such as to establish the range of likely values.

In the case of an opened and operating tunnel we strongly recommend that such approximations are not relied upon. These estimates should be verified or adjusted on the basis of observed values obtained from an extensive monitoring campaign, which includes simultaneous measurements of external background concentrations.

10.6 Managing NO_2 and CO via CO, NO and visibility monitoring

Due to the technical limitations in measuring NO_2 or NO in tunnels, several studies have considered the practicality of modelling NO and NO_2 concentrations on the basis of CO and/or aerosol monitoring. For example, optical aerosol monitors have been installed in some Norwegian tunnels for ventilation control. The Bomlafjord tunnel in Norway is a 7.9 km long sub-sea tunnel. It has a single bi-directional tube and carries approximately 2 500 vehicles per day. Despite the low level of emission, such a tunnel has significant ventilation demand due its long length, significant gradients (max 8.5 % for the Bomlafjord), the inability to vent mid-length and the lack of piston effect. In the Bomlafjord tunnel a longitudinal system was installed with axial fans triggered initially by CO and NO measurements monitoring. In the first year of operation the operational cost of the ventilation system was deemed to be very high and multiple complaints of poor visibility were made, despite limit values not being exceeded. In the light of long-term CO emission reductions, and the difficulties in measurement of

NO and NO_2 in tunnels a new control system was implemented using optical particle monitors. The system included four steps of increased ventilation triggered at 75, 150, 225 and 300 $\mu g\ m^{-3}$.

A model was devised to predict NO concentrations in the Bomlafjord tunnel, based on data from both CO and aerosol monitors (Indrehus & Aralt, 2005). The resulting predictions were compared with 6 weeks worth of monitored NO data at four locations in the tunnel. A correlation of 0.807 was found between modelled and measured concentrations, reducing to 0.771 if only the aerosol data were used. The model was found to under-predict at high concentrations. It was concluded that CO monitoring was required in addition to aerosol monitoring in order to predict NO and NO_2 concentrations.

The effectiveness of this arrangement to protect users from high concentrations of CO and NO_2 was analysed by Indrehus & Aralt (2005). They found that the optimisation brought about a 15 % decrease in electrical consumption. During this period the NPRA limit for aerosol (0.75 mg m^{-3} at the mid-point) was not breached (mean at the two monitors nearest to the mid-point were 66.5 – 99.4 $\mu g\ m^{-3}$ with a maximum of 673 $\mu g\ m^{-3}$). The CO concentrations were far below the limit of 100 ppm at the midpoint (mean at the two monitors nearest to the mid-point were 3.6 – 12.6 ppm with a maximum of 47.4 ppm). The success of the ventilation scheme in protecting users from NO_2 was assessed by converting the limit value for NO_2 (1.5 ppm, or 0.75 ppm at the mid-point) into an equivalent limit for NO based on the assumption of a NO_2/NO_x ratio of 0.1. There is some uncertainty on what this ratio should be as it is variable between tunnels due to the influence of length, ventilation, background oxidants and emissions. Nevertheless, with a ratio of 0.1 it was found that the NO (and by implication NO_2) limit values were not breached, but by a smaller margin than for CO.

11. Feedback and control

11.1 Overview

- Monitoring data can be used in a feedback loop to fully or partly control ventilation. In such a system desirable (and cost-effective) dynamic stability is best achieved with an element of 'prediction' rather than just 'reaction'.

- In some sensitive locations venting of road tunnel air into the external environment may compromise local air quality. This impact will be very dependent on local topography, existing air quality and the nature of the built environment. In such sensitive environments it may be necessary to monitor or model the local impact. These data can then be used in a feedback loop as another variable to be considered in ventilation control.

- Further reduction of the air quality impacts of a tunnel may be achieved through traffic management. Generally, congestion is a worst-case scenario within a tunnel due to high emissions, an increased number of vehicles in the tunnel volume, a reduced traffic-induced airflow and the potential for the initiation of a chemical reaction which produces excess nitrogen dioxide. A number of traffic management options have been listed which can contribute to air quality management.

11.2 Feedback to ventilation control

Once the ventilation system has been designed to prevent a certain concentration limit being breached, and the tunnel is built, there are limited opportunities to make changes. The second use of the limit values is in the management of the ventilation system.

The existence of limit values for CO and visibility means that measurements of either or both can be used to control ventilation. For example, the Central Artery tunnel in Boston uses only CO monitors to meet both CO and visibility guidelines (Betchel/Parkers Brinkerhoff, 2006). This may be due to the relatively low contribution of diesel-engined vehicles and heavy goods vehicles on this route.

However, measurements of visibility are directly used in ventilation control in many tunnels, including the Kingsway tunnel, Liverpool (Imhof *et al*., 2006) and the Tauerntunnel, Austria (Schmid *et al*., 2001).

In simple terms, if the limit value is breached or likely to be breached, additional forced ventilation can be activated. However, this is not as simple as it sounds. A 'live' system will continuously monitor CO and/or visibility in the tunnel (hopefully at multiple locations). Once a decision to activate extra ventilation is made there can be a significant time delay before the extra fans reach full speed. A simple system that switches the additional thrust on and off in response to threshold levels is liable to be unstable, rapidly switching fans on and off in way that is uneconomical. Changes in ventilation induce pressure changes in the tunnel that can have unpredictable effects on air flow and concentrations in other parts of the tunnel, especially in complex tunnels with curves, changes in gradient and branches. Experience in ventilating complex mineshafts has shown how a stable system is preferable to an optimal one (Jacques & Possoz, 1996). Achieving a stable system can require complex air flow monitoring and computer modelling.

A system that reacts to changes in concentrations in the tunnel has a number of disadvantages. As it reacts to past events (changes in emission or air flow due to localised congestion, for instance, leading to localised rises in concentration) it is always trying to 'catch-up' and this time lag can lead to inherent dynamic instability. Such a system is also dependent upon monitor data which is prone to inaccuracy (see above). An alternative approach is not to react, but to anticipate. Such a system does not rely (entirely) on monitoring concentrations, but instead models those concentrations as or preferably before they happen, based on traffic data. At a basic level these traffic data can be average traffic counts (preferably long-term to establish variability). A more sophisticated system will make live observations of traffic flow. The predictive ability of the system is improved if traffic flow data from upstream of the tunnel are available, giving vital extra minutes to calculate the expected emissions and concentrations in the tunnel and to bring fans up to speed if necessary in advance of the predicted limit breach. Such a system requires a significant cost in terms of design and test. However, in the case of a relatively polluted tunnel in which extra ventilation is regularly required this is compensated by a lower operational cost in terms of energy saved from unnecessary fan operation.

We have found one example of a tunnel with an optional forced ventilation system in which busy traffic leads to quite high concentrations of NO_2 without the forced ventilation being activated (Indrehus & Vassbotn, 2001). This has been due to CO remaining below the locally specified limit value and the absence of NO_2 monitoring as part of the ventilation control system. In some cases the fans are automatically operated during peak periods as a preventative measure (e.g. Craeybeckx tunnel, Antwerp; Shing Mun, Hong Kong; Klaratunnel, Stockholm).

11.3 Air filtration and treatment

Filtration or other treatment of tunnel air to remove or reduce pollutants is not widely implemented, but this may change due to recent technological improvements. Use of electrostatic precipitation (ESP) to remove particulates has been applied most widely in Japan. Norway is the only other country with significant application of ESP, where road dust emission from studded tyre use is a major cause of reduced visibility, but it is not regularly operational (Child & Associates, 2004). Filtration is expensive to install and operate, and especially to retrofit to existing tunnels, and is therefore only likely to be cost-effective for severely polluted tunnels or operation at peak traffic periods. Technology to reduce NO_2 in tunnel air is at a relatively earlier stage of development and adoption. Two rival systems will be trialled in the ventilation stacks of the 11 km Central Circular Shinjuku Tunnel in Tokyo due to open in 2008.

11.4 Consequences for external air quality

A small amount of the particulates emitted from vehicles inside a road tunnel will be deposited to surfaces. However, the majority of the particles, along with the trace gases, that are emitted by vehicles will eventually be vented into the external atmosphere (in the absence of in-tunnel filtration or denoxification) via the tunnel portals and by any exhaust stacks which may be fitted as part of a ventilation system. Once vented the vitiated air will rapidly mix with ‘fresh’ air and dilute and disperse leading to a rapid drop in concentrations relative to inside the tunnel. This is particularly the case in which stacks are used in which polluted air is released above the height of most exposed individuals, above local obstacles which may provide shelter from winds or induce downdraughts, and at a height at which winds will normally be stronger leading to more effective dispersion.

However, stack and especially portal emissions can lead to highly localised ‘hotspots’ of increased concentrations. It is quite possible that road tunnel emissions can lead to localised breaches of the National Environmental Standards for PM_{10} and NO_2 around stacks and portals, as well as exceedences of Regional Air Quality Guidelines or

Targets. This is critically important if these locations coincide with residences, businesses or any other land-use in which people are likely to be exposed. Tunnels commonly pass under hills, and those hills can create local airflow patterns, including sheltering, induced downdraughts and valley-inversions which locally inhibit dispersion and further lead to local air quality problems which would be exacerbated by a road tunnel emission. And in some locations, dense buildings, and especially tall buildings may further worsen dispersion characteristics whilst increasing exposure. In such cases the local built or natural topography may reduce the effectiveness of a tall stack.

In such locations where venting of internal tunnel air may compromise external air quality in an exposure-sensitive environment great care needs to be taken over the rate of tunnel ventilation. In these locations we recommend that dispersion modelling is first employed to identify potential problems and locations of possible 'hotspots'. We also recommend that post-opening, air quality monitoring should be conducted in the identified locations for at least a year. After that period sufficient data should have been collated to a) identify and implement any necessary mitigation options, b) validate (and adjust if necessary) the dispersion modelling such that modelling can henceforth replace the monitoring, or c) satisfy all parties that there is no external air quality issue related to the tunnel.

In particularly sensitive or difficult locations a ventilation control system could be employed which would use live data from these external monitors to adjust and control the tunnel fan system to prevent breaches of standards outside the tunnel.

Sturm *et al*. (2004) and PIARC (2008) report on an example of such a system in Austria (Kalvariengürtel, or City Tunnel Graz-North, Graz). The city of Graz is in a deep alpine valley and is prone to very low wind speeds and consequent PM_{10} and NO_2 air quality problems. The location of this urban tunnel exit near tall residential buildings prevented the use of stacks. The tunnel is relatively short (600 m) with not especially high traffic. In such a case in-tunnel concentrations are relatively low, so they key air quality issue is the venting of polluted air, and especially NO_x, at the ground-level portals in a built-up area with existing high NO_2 levels. In this case a system was deployed that was based on CO and visibility inside the tunnel, but would discontinue powered ventilation - reducing portal emissions - if NO_2 measured immediately outside the tunnel portals rose above 150 $\mu g\ m^{-3}$ as a 30 minute average. However, if the internal NO_2 concentration rose above 800 $\mu g\ m^{-3}$ then the ventilation would enforce an airflow of 5 $m\ s^{-1}$.

11.5 Consequences for traffic management

Alternative approaches to controlling tunnel air quality involve limiting emissions by controlling the traffic flow. Variables that are amenable to control include

a) Minimum or maximum speed limits (gas emissions will generally be reduced at higher speed, although emissions of NO_x and resuspension of particles may be increased),

b) vehicle emission control (e.g. vehicles not meeting specified emission standards can be barred from the tunnel),

c) fleet mix (e.g. HGVs could be barred from the tunnel),

d) traffic flow (could be regulated by traffic signals on entry),

e) flow reduction (on-ramps or individual lanes can be closed to reduce the number of vehicles in the tunnel),

f) redistribution of emissions in time (for example HGVs can be restricted to certain times of the day when their impact will be lessened, e.g. night-time or outside peak traffic periods).

Many road tunnels, especially in Australia, are tolled, or form part of toll roads and networks. Several of the variables listed above can be influenced by differential and variable tolling. This needs to be carefully reviewed so that it is a) effective, and b) made clear whether it is used primarily as an air quality or traffic management tool or a revenue-generation tool so as to identify potential conflicts (i.e. high emission vehicles could generate increased toll revenue). This may be offset by the savings made in ventilation costs if in-tunnel concentrations are reduced. Where electronic tags are used new or renewed tags could be made conditional upon vehicles meeting emission standards, which could include testing. This could be applied selectively to those parts of the fleet contributing disproportionately to poor air quality.

Traffic management can be a primary tool for managing air quality in a tunnel, especially in the case of congestion, supported by ventilation responses. Low traffic speeds, perhaps as a result of congestion, generally lead to higher emissions per kilometre and a reduced piston effect, thus giving rise to a worst case scenario for in-tunnel air quality.

For example, the Incident Response Plan of the M5 East tunnel (Sydney) notes that if traffic speeds fall below 20 km h^{-1} only one lane should remain open, so as to limit the number of vehicles in the tunnel and hence reduce emissions. Although this should be effective in maintaining CO concentrations below the appropriate guideline, it can have the undesirable effect of worsening congestion further upstream. The Plan was later revised in light of the experience of operation following tunnel opening. It was found that the automated system to increase airflow in the event of an incident did not react quickly enough to maintain sufficient airflow.

HDVs have a disproportionately large impact on air quality, especially through PM (resuspension, soot and ultrafine particles) and NO_2. Concentrations of these pollutants are sensitively dependent on the number, speed and variation in emissions of HDVs. This is becoming increasingly so as emission controls penetrate the car fleet. Therefore any controls that focus on HDVs are likely to have a large impact.

More commonly systems are installed to improve safety in tunnels by reducing the causes of accidents and fires. These are often caused by congested traffic, so measures to reduce congestion will also lead to improvements in air quality. An increasingly common example is the use of variable signs at some tunnels. These allow the posting of variable speed limits, plus advance warnings of lane closures. Both measures allow a smoothing of traffic flow near capacity-saturation conditions and prevent the development of congestion. In some cases these systems have been retro-fitted to existing tunnels. Unfortunately, as they have not been installed specifically for the purposes of air quality, we are not aware of any assessment study of their effectiveness in reducing pollutant concentrations.

Several guidelines exist that decree when a tunnel should be **closed** to traffic due to excessive levels of air pollutants. For example, in Norway tunnels should be closed if CO in the tunnel mid-point exceeds 100 ppm for more than 15 minutes. It has been noted (Indrehus & Vassbotn, 2001) that NO_2 levels can reach dangerously high levels in at least one tunnel without triggering a closure – both because 100 ppm of CO was not reached, and because monitoring of NO_2 was not part of the tunnel's control system.

11.6 Feedback and control – summary

Monitoring data can be used in a feedback loop to fully or partly control ventilation. In such a system desirable (and cost-effective) dynamic stability is best achieved with an element of ‘prediction’ rather than just ‘reaction’.

In some sensitive locations venting of road tunnel air into the external environment may compromise local air quality. This impact will be very dependent on local topography, existing air quality and the nature of the built environment. In such sensitive environments it may be necessary to monitor or model the local impact. These data can then be used in a feedback loop as another variable to be considered in ventilation control.

Further reduction of the air quality impacts of a tunnel may be achieved through traffic management. Generally, congestion is a worst-case scenario within a tunnel due to high emissions, an increased number of vehicles in the tunnel volume, a reduced traffic-induced airflow and the potential for the initiation of a chemical reaction which produces excess nitrogen dioxide. A number of traffic management options have been listed which can contribute to air quality management.

12. Conclusions

This Report is intended to assist NZTA in setting guidelines for air quality in road tunnels in New Zealand. We have reviewed the relevant ambient and occupational guidelines for general use, and the guidelines adopted for road tunnels in other countries. The National Environmental Standards for Air Quality in New Zealand quite explicitly apply to all locations outside a tunnel in which people are likely to be exposed. They do not, however, apply inside tunnels.

Table 12.1 below summarises the objectives of in-tunnel air quality guidelines, the type of health risk they relate to, and the pollutant and averaging time that most practically represents that risk.

Table 12.1: Summary of the possible objectives of in-tunnel air quality guidelines, the time-frame of associated health risks, the pollutant or indicator best suited to representing that risk and comments on whether sufficient information exists to recommend a related guideline

Objective (see section 1.6)	Health risk type (see chapter 3)	Pollutant/indicator	comments
Safety	Immediate-acute	CO, visibility	CO guidelines recommended
Health	Delayed-acute	NO_2, particles	NO_2 guidelines recommended, Insufficient data to set particles guidelines
	Chronic	NO_2, particles, benzene	Insufficient data to set long-term tunnel-specific guidelines for NO2 & particles. CO guidelines provide occupational protection for benzene
Wellbeing	Anxiety & stress	Visibility, odour	Unquantifiable at present

We conclude that in setting guidelines NZTA are faced with a number of key choices, which are presented below.

12.1 Carbon monoxide guidelines

Carbon monoxide limits have been set around the world and are the conventional tool for managing road tunnel air quality. Due to this international precedent, the relatively high level of understanding of the effects of carbon monoxide exposure, the relevance of the rapid physiological impact of these effects to exposure times in tunnels and the proven practicality of CO monitoring in tunnels, we strongly recommend that a CO limit be adopted as a guideline.

What that limit should be depends upon the margin of safety required and assumptions about whom the limit is protecting. We recommend that a limit corresponding to the World Health Organisation's ambient air quality guideline of **87 ppm** averaged over **15 minutes** be adopted. This limit is designed to ensure that the level of carboxyhaemoglobin in the blood of non-smoking tunnel users does not exceed 2.5 % as a result of a single passage through a tunnel. This corresponds to the generally accepted lowest observed adverse effect level (LOAEL) and provides protection for the general healthy population, but also more susceptible tunnel users, such as coronary patients and pregnant women. It also allows for an increased rate of COHb production in the body associated with light activity, such as walking, steady running or cycling in adults. Effects of raised carboxyhaemoglobin on children are poorly understood, and we cannot guarantee that children will be offered the same level of protection by this limit. However, the guideline CO value will also keep the carboxyhaemoglobin levels in children riding in vehicles through a road tunnel below 2.5 % also. This limit is similar to the 100 ppm limit recommended by PIARC. The reduction in that limit to 70 ppm appears to be based on achievability rather than any new evidence on the effects of CO on the body. Therefore we cannot recommend it on the basis of scientific evidence alone.

We find no basis to challenge the widely adopted occupational safety guidelines for CO of 200 ppm as a 15-minute average and 30 ppm as an 8-hour average. Although the PIARC recommendation of a reduction in the 8-hour limit to 20 ppm may be laudable, the lack of justification provided by PIARC leaves us unable to determine whether the recommendation is based on medical evidence or other considerations, and we are therefore unable to endorse the recommendation on scientific grounds.

Recent rapid reductions in CO emissions from road vehicles have meant that a CO limit is not as demanding upon ventilation as it used to be. Thus, more stringent CO limits could be adopted than have been hitherto, without necessarily additional ventilation demand. NZTA could consider adopting a more stringent CO guideline than the WHO value. This might be considered appropriate in the case of children having pedestrian or cycling access to a tunnel. However, our review of the current state of knowledge suggests that setting a limit on the basis of this purpose could not be justified scientifically.

12.2 Nitrogen dioxide guidelines

Recent technological changes have reduced CO emissions from road vehicles faster than emissions of oxides of nitrogen (NO_x, consisting of nitric oxide and nitrogen dioxide) such that the ratio of NO_x to CO in road tunnel air has increased over the last decade or more. In the meantime, research has increasingly related a range of adverse health effects to exposure to nitrogen dioxide, especially when mixed with particles, as occurs in road traffic exhaust. A range of different nitrogen dioxide exposure limits exist in the occupational, ambient and road tunnel contexts. The trend in emissions has meant that, if applied to road tunnels, NO_2 limits are gradually becoming more demanding upon ventilation than CO limits.

Much of the research on nitrogen dioxide and particles has focussed on longer exposure times than occur in tunnels, and particularly on long-term chronic exposure amongst people living close to major highways. It is unclear how much of this research relates to exposure during single short passages through most road tunnels, except that we must conclude that a risk to health does exist. The research is certainly relevant, however, where repeated tunnel usage is likely, such as experienced by commuters or professional drivers who use tunnels on a regular basis.

Thus, we conclude that a strong rationale exists for the adoption of a nitrogen dioxide guideline in New Zealand road tunnels to provide for the protection of health of tunnel users.

However, this need for a NO_2 guideline must be considered in the context of the greater difficulty in setting, implementing and enforcing such a guideline compared to CO.

Whereas there is widespread agreement on appropriate CO limits for short-term exposure, the same cannot be said for NO_2, with limits ranging from 0.11 ppm to 8 ppm noted in this Report. This represents the incomplete state of knowledge in terms of the effects of short-term exposure, and repeated short-term exposure to NO_2, and its especially complex interaction with exposure to particles. We must expect that it could be decades before such knowledge gaps are satisfactorily resolved.

Implementation and enforcement of a NO_2 guideline are further complicated by the chemical reactivity of oxides of nitrogen which make concentrations of NO_2 in a tunnel difficult to predict. Monitoring is impeded by technological difficulties in obtaining accurate and reliable measurements in a tunnel environment and by uncertainties regarding the ability of a measurement at a point to represent the concentration variability within the tunnel volume. We have briefly reviewed some approaches which may be adopted to deal with some of these difficulties. In summary, however, implementation and enforcement of a NO_2 limit is more technically demanding and complex than for a CO limit.

Considering the evidence, we recommend that a NO_2 guideline is adopted. This would bring New Zealand in line with many other countries around the world. We recommend that the Permanent International Association of Road Congresses (PIARC) recommendation, i.e. **1 ppm** to be exceeded no more than 2 % of the time, be adopted in New Zealand. This guideline is based upon protection of healthy non-asthmatic subjects. We recognise that experimental evidence exists (including experiments involving human volunteers in busy road tunnels) that asthmatics are susceptible to an observable adverse response to lower concentrations than this. This is reflected in the adoption of more stringent guidelines for ambient exposure (WHO) and the guideline of 0.4 ppm as a 15-minute average as adopted for road tunnels in France. However, this is based on a precautionary approach to evidence based on exposures of 30 minutes or more and, unlike for carbon monoxide, the significance for much shorter duration exposures is currently unknown. Any nitrogen dioxide guideline adopted should be reviewed within a decade due to the rapidly developing health research literature and rapidly evolving vehicle emissions.

12.3 Implementation

We recommend that implementation of an air quality guideline is supplemented by continuous, permanent monitoring to at least ensure that the objectives of the guidelines are being met. Monitoring is also required where feedback to mitigation measures (ventilation control or traffic management) is installed. We recommend that an experimental study is conducted to determine the spatial representativeness of the monitoring installed, by comparing measured concentrations at exposure-relevant locations through a tunnel to those reported by the permanent monitors.

An exposure management approach (which applies any guideline to individuals using the tunnel taking into account the duration of their exposure, rather than applying solely to the tunnel air) may offer potential energy savings from reduced ventilation, the opportunities for which are greatest in shorter tunnels. Such an approach requires that a minimum speed can be assured. That speed is tunnel-dependent. Exposure management depends upon parameterisations which can be specified with acceptable confidence for CO, but their applicability to NO_2 guidelines is more uncertain.

Pollutant retention in a vehicle post-tunnel need not normally be considered in exposure management in this case as an assumption that all vehicles are open is conservative. This assumption may be invalidated if vehicle speeds post-tunnel are lower than in the tunnel itself. This is unlikely to occur in most cases but could be significant if a tunnel with a relatively high minimum speed delivers traffic into a congested link or network.

Tunnel ventilation systems should be designed to the worst-case for exposure, which is low speeds and fully-open vehicles. Whether a vehicle cabin is open or sealed, the main determinant of the impact of carbon monoxide on health is the length of time spent in the tunnel. Decreasing the rate of infiltration into a vehicle cabin can greatly reduce concentrations inside the vehicle. Setting air vents to ‘recirculate’ is a highly effective mitigation measure for individuals, but only if the vents are re-opened once the tunnel transit is complete.

The complexity of nitrogen chemistry in tunnels, and the uncertainties regarding emissions of oxides of nitrogen (and nitrogen partitioning in particular) lead us to recommend that implementation of a NO_2 guideline should not be undertaken without experimental campaigns to improve our knowledge in the New Zealand context. The design and scope of these campaigns would depend upon the nature of the implementation (e.g. direct monitoring, or using NO, NO_x, CO or visibility as proxies) but would generally consist of detailed, intensive campaigns to establish key relationships between NO_2 and other measures, and the variability in those relationships, and to establish the location and nature of peak concentrations, including studies to establish if the termolecular reaction of NO with oxygen (see chapter 9) is leading to rapid NO_2 production anywhere in the tunnel.

All of the suggested NO_2 guidelines are likely to be more demanding on ventilation than the recommended CO guideline of 87 ppm, although this is specific to the emissions in any given tunnel. The NO_2 guideline could be implemented by adopting a correspondingly stringent CO guideline. This would depend upon the satisfactory development of a robust relationship between CO and NO_2 concentrations which would be specific to a given tunnel. This relationship would be expected to change over the years such that a regular review, involving a detailed monitoring campaign, would be required.

12.4 Particles and other toxic species

The effects on human health of short-term exposure (order of minutes) to airborne particulate matter, and repeated exposure, as would be experienced in repeated passage through a road tunnel, are insufficiently established to propose a guideline. We have found no evidence of such a guideline existing anywhere else in the world. This is also true for other toxic species present in vehicle exhaust, such as PAHs or benzene.

The 8-hour occupational CO guideline recommended provides more than adequate protection against benzene exposure, relative to the New Zealand 8-hour Workplace Exposure Standard which applies in all road tunnels. Lack of vehicle emission factors for formaldehyde, styrene and toluene prevent us from determining if the CO guidelines provide protection with regards to the WHO short-term exposure guidelines for these substances. However, the limited data available suggest that the emission reductions achieved for CO have generally been achieved also for other toxic species. Thus, we have no evidence to suggest that a CO guideline does *not* provide similar protection for these species.

Safety in road tunnels is maintained partly through the implementation of visibility limits, although this is intended to be independent of air quality (i.e. health) impacts associated with visibility-reducing particles. PIARC recommends a set of 5 in-tunnel visibility limits corresponding to 5 traffic conditions. However, *perceptions* of poor air quality may be related to reduced visibility arising from haze and dust, and the appearance of visibly smoky plumes. We have reviewed two examples of Norwegian tunnels in which CO guidelines were not exceeded, yet users regularly complained of poor visibility.

Visibility monitoring (which is installed as part of the permanent ventilation system in many tunnels worldwide) provides the *potential* for an alternative assessment of the air quality and health risk within a tunnel, including an indirect means of assessing levels of NO_2. However, the relationships between visibility, CO and NO_2 are likely to be quite specific to traffic characteristics and tunnel geometry and ventilation, i.e. they are probably quite tunnel-specific and need to be determined empirically through observation.

13. Detailed Recommendations

13.1 Guidelines

- We strongly recommend that a CO limit be adopted as a guideline for all tunnel users.

- We recommend that a carbon monoxide guideline equivalent to the World Health Organisation's ambient air quality guideline of 87 ppm averaged over 15 minutes be adopted.

- We recommend that a separate and additional NO_2 limit is also adopted.

- We recommend that the PIARC recommended nitrogen dioxide limit of 1 ppm not to be exceeded more than 2 % of the time be adopted.

- We recommend that a nitrogen dioxide guideline of 0.4 ppm as a 15-minute average be considered on a precautionary basis to provide extra protection for asthmatics.

- We recommend that the New Zealand Workplace Exposure Standards (WES) for CO (200 ppm as a 15-minute average) be adopted for occupational exposure of staff.

- We recommend that the NIOSH Recommended Exposure Limit (REL) for CO (30 ppm as an 8-hour average) be adopted for occupational exposure of staff.

- We recommend that the NIOSH Recommended Exposure Limit (REL) for NO_2 (1 ppm as a 15-minute average) be adopted for occupational exposure of staff.

- We are unable to recommend any guidelines for particles at this time, due to the international lack of health-based evidence on the risk posed by very brief exposures (seconds to minutes) as apply to road tunnels users.

- This report has not explicitly reviewed visibility guidelines as these are devised and implemented for the purpose of maintaining tunnel user safety, rather than protecting health. However, we have highlighted some instances where visibility monitoring can be used to provide indirect information about in-tunnel air quality in general, and NO_2 concentrations specifically.

- We recommend that observational research be conducted to determine if the rapid and possibly unobserved production of nitrogen dioxide via the direct reaction between nitric oxide and oxygen is a significant issue in New Zealand's tunnels and whether compliance with a specific NO_2 limit would prevent it.

- Due to the substantial remaining uncertainties regarding the health effects of brief exposure to NO_2, the emissions, determinants and levels of NO_2 in State Highway road tunnels, and technical challenges in monitoring NO_2 ,we recommend that an NO_2 guideline be implemented for the purposes of design only (rather than compliance monitoring) at this stage.

- We recommend that any guidelines adopted are reviewed on a decadal basis in the light of new emission trend data, forecasts and new health research.

13.2 Implementation

- We highly recommended that a tunnel's ventilation system's performance is checked by monitoring at least once, and preferably on at least a decadal cycle.

- We recommend that permanent internal monitoring be included as part of any tunnel design. No system should rely on a single monitor.

- Numerical modelling and/or a pre-deployment study is recommended to assist in characterising the representativeness of any permanent monitor.

- We recommend that a research programme be conducted to determine the nature of NO_x emissions and resulting NO_2 concentrations and NO_2/NO_x ratios in New Zealand tunnels to facilitate the implementation of a NO_2 guideline and to better quantify the risks arising from road vehicle emissions, and specify mitigation options with greater certainty and confidence.

- We recommend that the relationship between concentrations of CO, NO, NO_2 and measurements of visibility are investigated in specific New Zealand tunnels to further investigate the viability of using visibility monitoring to inform air quality assessment and better integrate visibility (safety) and air quality management.

- Before such research can be conducted we recommend that a NO_2/NO_x concentration ratio of 0.1 be assumed in tunnels, rising to 0.2 within 200 m of the tunnel ends.

13.3 Other matters

- In the case of a tunnel with a low level of traffic a minimum air flow should be included in the design so as to cope with the transient effects of gross polluting vehicles or tunnel road blockage.

- Long-term changes in traffic and emissions have to be estimated at the design stage of a tunnel ventilation system, or its upgrade. Estimates will be substantially supported by a programme of long-term emissions monitoring in existing (and future) road tunnels, which we strongly recommend. This is enabled by an enhanced approach to in-tunnel monitoring (i.e. incorporating monitoring of airflow, traffic characteristics and ambient air quality). Such a programme would also feedback into national vehicle emission assessment and forecasting.

14. References

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