Health and Air Pollution in New Zealand:

Christchurch Pilot Study

Authors

G. Fisher[1, 10], T. Kjellstrom[2, 3, 9], A. Woodward[3,11], S. Hales[3], I. Town[4], A. Sturman[5], S. Kingham[5], D. O'Dea[3], E. Wilton[7], C. O'Fallon[8], A. Scoggins[1], R. Shrestha[2], P. Zawar-Rewa[5], M. Epton[4], J. Pearce[5], J. Sturman[5], R. Spronken-Smith[5], J. Wilson[5], S. McLeod[4], R. Dawson[4], L. Tremblay[6], L. Brown[6], K. Trout[6], C. Eason[6], P. Donnelly.

1. National Institute of Water and Atmospheric Research. 2. Australian National University. 3. Wellington School of Medicine and Health Sciences. 4. Christchurch School of Medicine and Health Sciences. 5. University of Canterbury. 6. Landcare Research. 7. Environet. 8. Pinnacle Research. 9. Health and Environment International Trust. 10. Endpoint. 11. Auckland University.

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(NB Results from this study are specific to Christchurch and should not be used for other areas, where there are different emissions and different effects occur.)

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List of Abbreviations

ARC	Auckland Regional Council
CAU	Census Area Unit
CO	carbon monoxide
CO_2	carbon dioxide
ECan	Environment Canterbury
GIS	Geographic Information Systems
HRC	Health Research Council of New Zealand
ICD	International Classification of Diseases
MfE	Ministry for the Environment
MoH	Ministry of Health
MoT	Ministry of Transport
NIWA	National Institute of Water & Atmospheric Research Ltd
NO	nitric oxide
NO_2	nitrogen dioxide
NO_X	nitrogen oxides
NZDep	New Zealand Deprivation Index
O_3	ozone
PAHs	polycyclic aromatic hydrocarbons
PM_{10}	particulate matter with aerodynamic diameter of 10 micrometers or less
$PM_{2.5}$	particulate matter with aerodynamic diameter of 2.5 micrometers or less
RMA	Resource Management Act 1991
SO_2	sulphur dioxide
SO_X	sulphur oxides
TSP	total suspended particulates
US	United States of America
US EPA	United States Environmental Protection Agency
VOC	volatile organic compounds
WHO	World Health Organisation

Executive Summary

Overview

The people of New Zealand are exposed to a wide range of health risks through various activities. Many of these are unavoidable, and many are due to personal choice, however some are due to exposures to contaminants in the environment that can be reduced through community policies. This three-year long study is concerned with identifying and quantifying the health risks due to peoples' exposure to air pollution. For many places, and for much of the time, New Zealand's air quality cannot be considered poor by international standards, yet there are still defined health effects, and there are locations and instances where air quality is poor enough to be of concern.

Measures to reduce air pollution, and its effect on public health, have costs. Effective management and policy needs detailed information on exactly what these effects are and their costs to individuals and society, and the costs to society of mitigation measures. The aim of this "Health and Air Pollution in New Zealand (HAPiNZ) Study" is to explicitly identify the effects of air pollution, throughout New Zealand, to link these effects to the various sources of air pollution, to examine the costs of the effects, and to formulate cost effective policy options that will lead to real and measurable improvements on the health of New Zealanders.

The study is funded under a joint initiative from the Health Research Council, the Ministry for the Environment and the Ministry of Transport, with substantial in-kind contributions from Regional Council air quality monitoring programmes, in particular Environment Canterbury for the pilot study in Christchurch. The work is carried out by a large collaborative group, comprising several organisations and over 20 of New Zealand's leading researchers in air pollution, epidemiology, toxicology, environmental management, economics, and public health policy

Study Methodology

This report covers the first phase of the project, which is a pilot study on Christchurch. The methodology and scope of the project is large and complex. There are many different sources of air pollution, which is transported around the atmosphere by the weather in very complex ways. The resulting human exposures may lead to a wide range of health effects on the population. Effective policy analysis needs to use accurate information on just which sources have particular effects, on particular sectors of the population, in some cases over many years.

Before the study methodology is applied to the whole of New Zealand, this pilot study on Christchurch will be reviewed, evaluated and circulated for comment. Christchurch was chosen for the pilot study for two main reasons (a) it has already been identified as the New Zealand city with the most significant air quality problem – due mainly to winter-time home heating using wood and coal burning, and (b) considerable amounts of data were available from previous studies and the Environment Canterbury monitoring programme.

The study methodology has evolved through this pilot study, and will evolve further during the final phase of the project. New data appears almost daily, and new assessment and calculation methods are continually appearing, many worthy of merit. Revisions to the methodology invariably result in different quantification of the health effects, however where these are potentially significant they are identified. For this reason, considerable emphasis has been placed on comparing the results obtained here, with those obtained previously in this and other studies, and attempting to fully explain any differences.

General Results

The results obtained from the study are wide-ranging and detailed. All the major sources are included, all the common air pollutants are included, the major effects have been quantified, economic costs have been assessed, and some possible policy options have been analysed. The following general results are clear: -

The results of previous studies on the effects of air pollution in Christchurch are broadly confirmed, but with greater detail on the location and scale of these effects.

There are defined health effects, in some portions of the population, down to relatively modest air pollution levels.

The findings are reasonably consistent with similar studies conducted overseas, and previous New Zealand studies, although some aspects of the Christchurch study are more advanced than other studies.

The greatest effect – and greatest cost – occurs due to pre-mature mortality caused by long-term exposure to fine particulates from combustion sources. However many other pollutants have effects that can, and should, be avoided.

Other pollutants such as carbon monoxide, nitrogen dioxide and sulphur dioxide have quantifiable adverse health effects including pre-mature mortality, respiratory disease, and hospital admissions. The number of cases attributable to these other pollutants appears to be fewer than those attributable to particulate matter. However these contaminants may contribute to some of the effect attributed to PM_{10}.

There are also adverse effects from air pollution that may not have direct and obvious public health implications, but nevertheless have costs to society. These include restricted activity days, which can affect large portions of the population on bad air pollution days, and extra medical costs associated with treating respiratory problems such as asthma.

The most sensitive portions of the population are (a) older people, particularly over-65s, (b) infants, particularly under-1s, (c) asthmatics and people with bronchitis, (d) people with other respiratory problems, (e) people with other chronic diseases, such as heart disease.

The effects due to various sources have been examined. These are largely as expected - home heating combustion, industry and motor vehicle emissions sources. Some attempt has been made to attribute the specific effects to specific sources, but this is not always feasible, as the state of the science is still evolving.

Mitigation options are available – for new policies and actions at the government, industry and community level.

The results contained in this Pilot Report are relevant to Christchurch only. The absolute and relative effects associated with air pollution in other areas will be different.

Specific Results

Exposure

A large amount of work has been carried out refining the air pollution exposure estimates for the population of Christchurch. The results obtained are very detailed, identifying exposures to PM_{10} and a number of other pollutants on high resolution time and space scales. An example is shown in Figure 1.

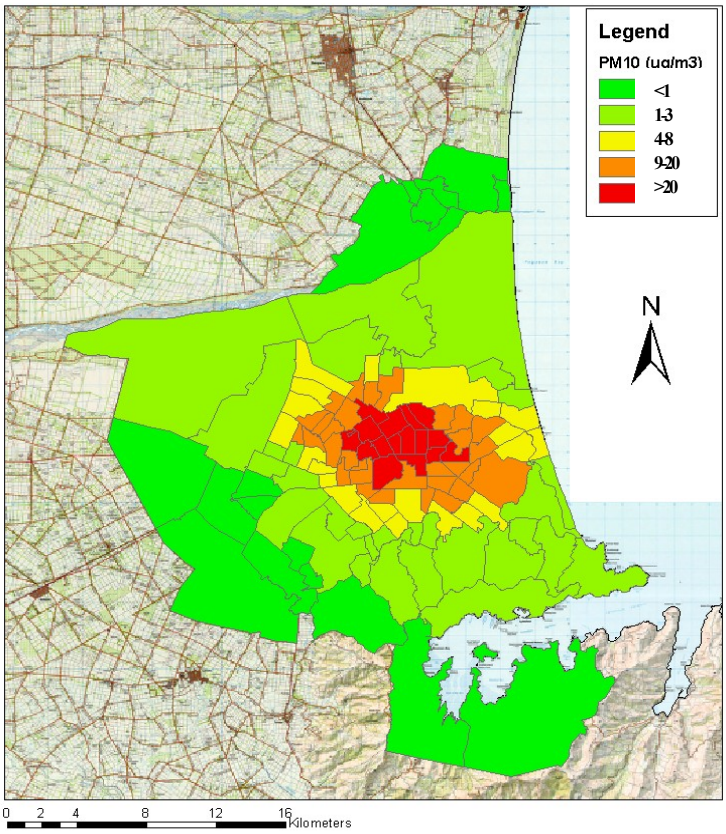

Figure 1: Annual PM_{10} exposure map. (Darkest areas in the central city are those above the 20 µg m^{-3} national annual guideline.)

Particulate measurements have been obtained using the data from the Environment Canterbury monitoring network, complemented by a number of special measurement campaigns by the University of Canterbury and others, and extended using advanced airshed modelling techniques.

Health effects - Mortality

Table 1 shows an estimate the pre-mature mortality in the population of over thirty year olds associated with emissions of air pollution from domestic home heating using solid fuels (mainly wood, with some coal), industrial, and vehicle sources. These are the most likely figures. The numbers will have an uncertainty associated with the input numbers, such as the uncertainty of the exposure estimates, the uncertainty of where people spend all of their time, and the statistical uncertainty inherent in the dose-response calculations.

These results are based primarily on long term exposure to PM_{10}, however other effects occur and these are discussed in the main report. These results are also very specific to Christchurch, and are dominated by winter time solid fuel burning for home heating. The proportions will be different for other cities.

Effect	Domestic	Industrial	Vehicle	Total
Mortality	124	18	16	158

Table 1: Premature deaths in Christchurch due to air pollution, by source.

Vehicle effects can be further categorised by estimating the effects due to petrol, diesel and related sources. Most of the PM_{10} is sourced from diesel vehicles, with at least 70% estimated to come from heavy duty vehicles (those over 3.5 tonnes tare weight). There is also a fraction estimated to come from non-tailpipe sources, such as brake and tyre wear and road dust. This has been estimated at 15% of the total for Christchurch, but is the subject of further research, and may be a higher fraction.

A previous study (Fisher, G. et al. 2002. Health effects due to motor vehicle pollution in New Zealand: Report to the Ministry of Transport) estimated 182 premature deaths attributable to PM_{10} from all sources for Christchurch, based on a methodology used extensively in Europe (Kunzli, N.; et al. 2000. Public-health impact of outdoor and traffic-related air pollution: a European assessment. Lancet 356(9232): 795-801.). This current pilot study using the same methodology estimates slightly fewer premature deaths attributable to PM_{10} at 158. This difference is most likely because this new study estimates exposure to PM_{10} concentrations based on a finer spatial resolution. The 2002 study used estimates from monitoring sites, which tend to be located in areas of high exposure, and extrapolated these across the city. The current work uses more advanced airshed modelling which gives a better exposure estimate. This situation is not necessarily translated to other New Zealand cities. The 2002 study had overestimated exposure in Christchurch however the exposure estimates for many urban centres were underestimated in the 2002 study as no monitoring data were available. Since 2002, many cities have new monitoring programmes which indicate higher than anticipated concentrations of PM_{10}.

There is a significant additional component to the research that is not quantified in this pilot study. This is the very specific nature of the air pollution profile in Christchurch, and the possibility of unaccounted for effects. These are discussed in the report, but relate to two complications: (1) The dominance of winter-time domestic heating emissions of PM_{10}, and (2) the difficulty in explicitly identifying effects due to other pollutants such as CO, NO_2 and SO_2 that are known to have specific effects.

On the first point, the calculations and resulting effects are strongly dominated by winter-time exposures, and exposures during summer-time – to effects that are more influenced by industrial and vehicle emissions – are not well identified. This is being addressed in the next phase of the research

On the second point, effects due to pollutants other than PM_{10} - CO, NO_2 and SO_2 are non-trivial, but cannot at this stage be listed separately as the state of epidemiological knowledge is not sufficiently advanced. For instance, even if by some method the PM_{10} concentrations and effects could be eliminated, air pollution health effects would remain due to the other pollutants. They will likely be less, but not zero. This presents a particular difficulty in analysing policy responses, as a focus on the main source of domestic home heating, as implied by the results here, may not fully address the issue. This point is also being addressed in the next phase of the research.

Health effects - Other effects

The major effects of air pollution are premature mortality however there are a wide range of other effects.

Health impact assessments were carried out for PM_{10}, carbon monoxide, nitrogen dioxide, and sulphur dioxide. However, to preclude double counting of adverse health effects related to air pollution usually only one pollutant is chosen to quantify a specific health outcome. PM_{10} is usually considered as the single pollutant but the health effects due to CO, NO_2, and SO_2 are not necessarily independent. There is considerable evidence, from both this study and numerous others, that particulate effects dominate the total health effects, however recent studies are providing evidence that independent effects for other pollutants – particularly CO, but also NO_2 and SO_2 – may also be significant. There is some rationale for this in growing knowledge about the biological mechanisms which may be different for different types of exposure.

Carcinogenic effects are quite different. One type of cancer from one source is quantified: leukaemia due to benzene exposure. Benzene is considered a Group A (known human carcinogen) human carcinogen by the U.S. EPA, and there is no threshold for effects (i.e. no safe level can be recommended).

Restricted activity days are defined as day where people in the community are affected enough by air pollution to take time off work, or not engage in activities they might have otherwise done.

Hospital admissions are extracted from health statistics.

Using dose-response relationships published internationally, and modified results from a previous study on restricted activity days, the additional effects due to PM_{10}, benzene, restricted activity days, annual number of chronic bronchitis cases (and related diseases) and hospital admissions for all age groups, and minor hospital costs are estimated for Christchurch in Table 2.

Health effect	Annual effect attributed to air pollution
Chronic bronchitis (and related diseases)	52 cases
Acute cardiac admissions	53 cases
Acute respiratory admissions	194 cases
Doctors visits, medicine	Not assessed
Leukaemia cancer cases due to benzene	1.6 case
Minor direct hospital costs	$200,000
Restricted activity days	285,000 days

Table 2: Other health effects in Christchurch due to air pollution.

It is not possible at this stage to ascribe these effects to sources, but a crude break down is approximately:-

- Domestic sources: 76%
- Industrial source: 13%
- Vehicle sources: 11% (approximately - petrol 0.3%, light duty diesel 2.5%, heavy duty diesel 8.2%)

Table 2 indicates another potential effect that has not been quantified – that is the additional visits to doctors and prescription of medicines (particularly for asthma) that might be associated with air pollution. Data on these quantities is not available, but the effects and costs are likely to be a significant fraction of the total.

Economic Impact Assessment

The costs of air pollution effects on Christchurch are estimated using the new statistics from this study, previous research in NZ, and results from overseas studies adjusted for NZ conditions. Table 4 gives a summary of the specific health effects used, and their cost per case. These are not personal costs, but costs to the NZ health system and economy – the external costs of air pollution in Christchurch.

Effect	Cost per case
Mortality	$750,000
Cancer	$750,000
Chronic bronchitis	$75,000
Admission (cardio-vascular)	$3,675
Admission (respiratory)	$2,700
Restricted activity day	$150

Table 3: Estimated costs of specific health effects used in the analysis.

The total costs of health effects of PM_{10} pollution in Christchurch can be estimated from the health effect and the cost per case of that effect, and shown in Table 4. The largest component of the 'health burden' is the loss of life-years as a result of pre-mature mortality, followed by restricted activity days, and then chronic bronchitis.

Effect	Domestic	Industrial	Vehicle	Total
Mortality	$93.0M	$13.5M	$12.0M	$118.5M
Cancer	$0.8M	$0.2M	$0.2M	$1.2M
Chronic bronchitis	$2.7M	$0.7M	$0.6M	$4.0M
Admission - cardio-vascular	$0.1M	$0.05M	$0.05M	$0.2M
Admission - respiratory	$0.4M	$0.1M	$0.1M	$0.6M
Restricted activity days	$30.0M	$7.0M	$6.0M	$43.0M
Minor hospital costs	$0.15M	$0.03M	$0.02M	$0.2M
Total	**$127M**	**$22M**	**$19M**	**$168M**

Table 4: Estimated annual costs of air pollution in Christchurch by source and by effect.

The external health costs due to air pollution in Christchurch, for 2001, are estimated here to be of the order of $168M, or $532 per person per year. Of this cost, some $127M (or 76%) is due to domestic emissions mainly caused by home heating, some $22M (or 13%) is due to emissions from industries, and some $19M (or 11%) is due to vehicle emissions mainly from exhausts of diesel vehicles.

These costs are likely to be underestimates as they do not include possible additional costs due to unidentified effects of pollutants other than PM_{10}, and do not include costs associated with extra doctors' visits due to air pollution. Neither do they include costs associated with indoor air pollution, or costs of effects due to workplace or in-vehicle exposures, which are non-negligible, but beyond the scope of this study.

Policy Options

With the costs of air pollution in Christchurch identified, as well as the major contributing sources, it is possible to investigate some policy options to reduce pollution and reduce costs to the economy and the community.

Three main source types are analysed, each falling within slightly different policy and legislative regimes – domestic emissions, in this case mainly wood and coal burning for home heating (covered by Regional Rules and National Standards), vehicle emissions (covered by National Regulations), and industrial emissions (covered by Regional Rules and National Standards).

The recent implementation of new National Environmental Standards for Air Quality affects all three categories, and places additional emphasis on both local and central government to reduce emissions.

Domestic

Domestic emissions cost Christchurch approximately $127M a year in health effects. The major domestic policy options leading to reduced health effects are methods to reduce (or eliminate) emissions from solid fuel heating.

The major policy options leading to reduced health effects are a ban on the use of open fires, and a prohibition on the installation of burners in houses not currently using solid fuel burners. Capping the allowable useful life of a burner would also result in greater certainty of achieving air quality and health targets. It should be noted that as of September 1, 2005 National Standards require that any newly installed freestanding wood burners must meet an emission limit of 1.5 grams of fine particles per kilogram of wood burnt, and a thermal efficiency rating of 65 percent.

The impact of these policy options relative to the status quo (that is the measures in place as of mid-2004, before any proposed Plan changes) was an estimated reduction in premature mortality of around 39 deaths per year by 2020 (from the current annual total of 158, of which 124 are due to domestic heating emissions).

Social impacts of the policy options include:

- Reduced heating choices for households unable to afford the price of a lower emission solid fuel burner.
- The potential for increased heating costs for households unable to install solid fuel burners.
- Increased heating costs for households previously using open fires and self collected wood.
- Potential impacts on burner manufacturers and retailers if fewer households install solid fuel burners.
- Cost implications to manufacturers including research and design costs, testing costs and inability to supply burners that complies with the lower emission standard.
- Loss of ambience associated with the use of open fires for domestic home heating.

Industry

Industrial emissions cost Christchurch approximately $22M a year in health effects. The major policy options leading to reduced health effects here are on (a) reducing emissions limits, especially for particulates, (b) ensuring compliance with current conditions, (c) adoption of best practice technology for industrial boilers, and (d) avoiding where practicable locating new industries close to residential areas.

The introduction of National Environmental Standards for Air Quality will drive some of these policies, but may require the introduction of new economic instruments and perhaps emissions trading schemes.

In this study, the following policy options were evaluated:

- All new large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge limit for total suspended particulate of 250 mg m^{-3}.
- From 2015 all existing large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge limit for total suspended particulate of 250 mg m^{-3}.

The impact of these modest policy options relative to current regulatory measures for industry was an estimated reduction in premature mortality of around 6 deaths per year by 2020 (from the current annual total of 158, of which 18 are due to industry emissions).

Vehicles

Vehicle emissions cost Christchurch approximately $19M a year in health effects. The major vehicle policy options leading to reduced health effects are those that result in (a) less traffic, (b) better vehicle maintenance, (c) a newer vehicle fleet, (d) less congestion, (e) emissions controls on vehicles, and (f) cleaner fuels. The bulk of the effect is from particulates emitted from the exhausts of diesel fuelled vehicles, and any policy to reduce health effect should focus on these sources.

A detailed policy analysis has not been conducted at this stage as insufficient information is available.

Implications

The HAPiNZ study is an ambitious one. It aims to explicitly identify the causes, effects and costs of air pollution throughout NZ and assess the effectiveness of specific policy options to reduce the effects. The implications of the results, and some caveats on the methodology, assumptions and uncertainties are discussed in the key outcome areas.

Health

Health effects are the subject of considerable international research, and there has been considerable scientific progress even since this project commenced. One of the areas of development has been on the specific effects of particulates, and the growing evidence that not all sources of PM_{10} have the same heath effect. The approach in this study has been to attempt to recognise the latest developments, as much as is reasonable. There will no doubt be further developments.

Another area of considerable uncertainty is the effects of pollution on such factors as restricted activity days, and some of the more subtle health effects. These are all discussed in the main report, but may need to be revised in the near future.

Costs

The calculation of the costs is – and probably will always remain – problematic. A number of assumptions need to be made, which can be debated. The costs are mainly external – that is they are not borne by individuals, but by society as a whole. These are always going to be difficult to quantify since there is no real market for either the costs or benefits. However, the evidence gained from numerous other studies does provide some international consistency of approach. This shows NZ in relation to other countries affected by air pollution, and the results are reasonably consistent.

Even within the last year there have been a large number of studies completed in many countries around the world. These use a wide variety of specific health costs in compiling national aggregated cost

estimates. And these have a wide range – perhaps by a factor of five from the highest estimates to the lowest. While some country specific features will make a difference (for example hospital costs tend to be higher in the USA than many other countries), there is still no consistency of approach. The costs values used in this study are conservative – that is they are at the lower end of the range used internationally.

Despite the difficulties, it is a vital component of the work to assign costs to the effects, as accurately as possible. A full cost/benefit analysis has not been attempted (except in the limited area of reducing domestic heating emissions where previous studies have been discussed). The full costs of implementing some of the policy options need to be further examined.

Policy

This project has attempted – somewhat ambitiously – to assess the effectiveness of some policy options. This extends the scope of the outcomes substantially. Policy is developed in relation to very wide range of factors in the community – not just the effects, or costs. It has been difficult to incorporate secondary implications – such as issues of social equity, or differing community norms in different areas.

Thus the policy analysis components here must be regarded as examples only. Specific policies, in specific areas need to take account of many other inputs. The ideas and calculations shown here, though, should provide a valuable steer to policy development.

Next Stage

The project is currently moving towards applying a detailed analysis to the whole of NZ. This will not be possible in quite the detailed fashion that as been conducted for Christchurch – mainly because the exposure data for other areas is not as detailed.

However the Christchurch case has provided the launching point for new techniques to assess air pollution effects and costs, and the Christchurch case certainly has some very detailed policy options to consider. A review of the methodology has highlighted a number of refined approaches that are being adopted for the next stage.

For the next stage, all of the cities in NZ will have an analysis conducted, with the end result of a fuller picture of air pollution and health for the majority of NZ citizens.

NB. This is a "Pilot Study". Although a great deal of detailed information is provided, it is subject to revision. One of the main purposes of this approach – of using a pilot study – is to allow for a review of the methodology and results. Once this is completed, the main study covering the whole of NZ may adopt a different methodology, and may use different assumptions.

1 Introduction

1.1 Overview

All over the world concern is growing over the health effects of emissions from transport, industry, domestic and other human activities. In New Zealand a recent study commissioned by the Ministry of Transport (MoT) estimated that approximately 400 people aged 30 and over die prematurely each year from exposure to PM_{10} particulates from motor vehicle emissions. It also estimated that an additional 570 people of the same group die prematurely each year from air pollution derived from other sources (Fisher et al 2002). As a result of these preliminary findings the Health Research Council of New Zealand (HRC), Ministry for the Environment (MfE), and MoT commissioned a national ambient air quality management project – Health and Air Pollution in New Zealand (HAPiNZ) – to better define the problem for New Zealanders. The major purpose of the project is to determine the environmental, health, social and economic costs of air pollution from all sources in New Zealand. The study is supported by the Ministry of Health (MoH), the Auckland Regional Council (ARC) and Environment Canterbury (ECan) (Figure 1-1). Other Regional Councils, authorities and communities of interest are also involved.

This report presents the results of a pilot study conducted in Christchurch, New Zealand. In the next phase a similar assessment will be carried out for the rest of New Zealand.

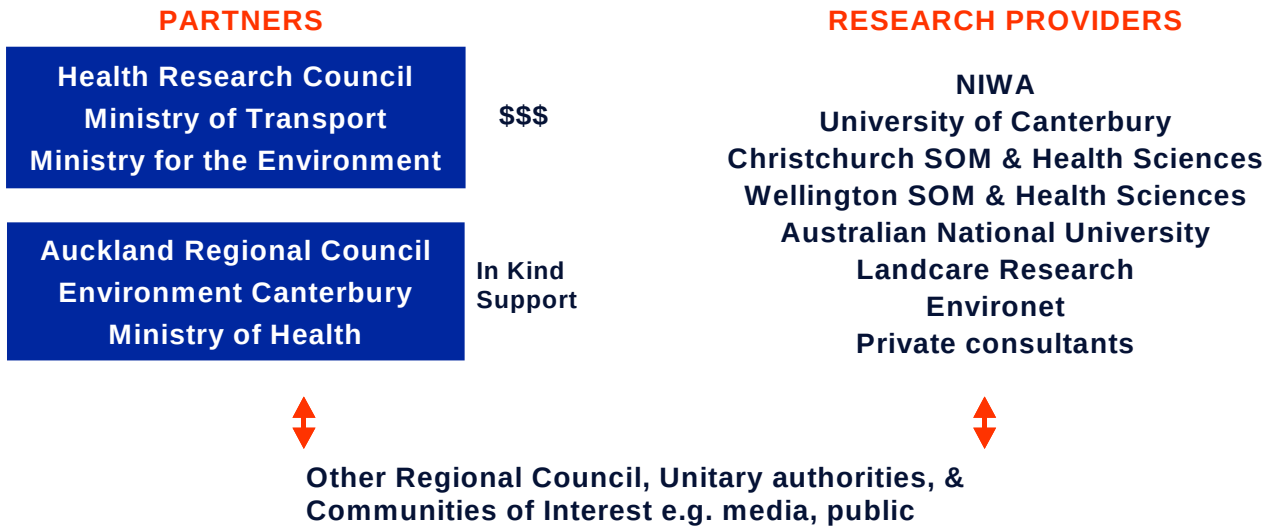

Figure 1-1. HAPiNZ collaboration.

1.2 Scope

The research has evaluated the effects of specific source categories of emissions from vehicles (including private petrol cars, diesel cars, and diesel trucks), industry, domestic and total sources in Christchurch. Source contributions for the following pollutants have been characterised for each defined area: PM_{10}, $PM_{2.5}$, NO_2, O_3, SO_2, CO, and benzene. Creating effective evidence based policy requires linking health and economic end points back to the sources, so preventative policy options and recommendations can be tailored to address emissions sources in order to their contribution to the air pollution problem. The research has five interconnected components:

- Air quality, meteorology and emissions data analysis
- Air pollution exposure assessment

- Health impact assessment
- Economic impact assessment
- Preventative policy assessment

This report presents the results of a one-year pilot study conducted in Christchurch, New Zealand. During the next phase of the project a similar assessment will be carried out for the rest of New Zealand.

NB. The results presented here are relevant only to Christchurch, and are not necessarily indicative of the situation in any other area.

1.3 Uncertainty

This study uses, and relies on, data from a number of sources in deriving its final results. Any study of this type will thus have a number of sources of uncertainty. These include:-

a) The uncertainty in determining the air pollution concentrations at all locations (this relies on monitoring which cannot be carried out at every place, and modelling which has its own inherent uncertainties due to imperfect emissions and weather information.)

b) The uncertainty in knowing what people are exposed to (this relies on knowing where people have spent all their time – an impossible feat, that has to be handled using realistic assumptions on the population as whole.)

c) The uncertainty in knowing the exact dose-response relationships. Although these have been studied extensively on large populations, all of these studies quote statistical uncertainties associated with population sampling techniques.

d) The uncertainty surrounding confounding effects. That is when the response seen might not be entirely due to the pollutant studied, or even to air pollution at all. Various techniques are used to overcome this, but they are still subject to statistical uncertainty.

e) In the economic assessment, the costs to the effects identified will have a range of values, depending on the assumptions made.

f) Finally in assessing policy responses for mitigating effects, a wide range of other factors come into play, such as peoples' social response to a policy. These are very difficult to assess, and can result in large uncertainties in mitigation effectiveness.

The details on each of these uncertainties, where they are known are covered in the report in the relevant chapters. For a robust scientific analysis, each of these uncertainties should be carried forward into the final result, as a range of figures, or a plus/minus percentage. Unfortunately if this technique were done it too simple a fashion it would produce a result with a very high range of uncertainties and render the whole study less useful.

This study has been carried out on the basis that each stage produces a 'most likely result', erring on the side of conservatism where choices appear. The outcome of any one of the stages can be quantified in terms of uncertainty, but a robust and reliable technique for accumulating the uncertainty from one stage to the next has not been developed.

A very simple example illustrates this. Say the exposure to PM_{10} is known to +/- 15% (not unreasonable), say the dose-response is known to +/- 25% (for many pollutants it is worse than this), say the economic assessment is calculated to +/- 50% (and even this might be optimistic), then the simple calculation of overall uncertainty can be +216% to –32%. This is a wide range of uncertainty to work with in policy responses, and for the purposes of this study the 'most likely' values are quoted in the results.

2 Background

2.1 Scope

The purpose of this chapter is to provide a brief background on the relationship between air pollution and health concerns, and in broad terms the nature of the health effects.

2.2 Sources and effects of air pollutants

Urban air pollution is increasingly the result of the combustion of fossil fuels (coal, oil, petrol, diesel, natural gas) for transport, power generation and other human activities. Biomass (wood, agricultural waste) is another important source of air pollution. The infamous London Smog episode during December 1952, caused by intensive coal burning in stagnant weather conditions over several days, led to sulphur dioxide and smoke reaching several thousands of micrograms per cubic metre ($\mu g\ m^{-3}$). This resulted in an estimated excess death toll of over 4,000 people. This event and others led to effective actions during the 1950 to 1970s to reduce air pollution such as banning coal use and requiring emissions controls on cars, starting as early as 1966. Even so, epidemiological studies through the 1980s till now are finding effects at very low levels of exposure or at levels below existing health guidelines.

2.2.1 National Environmental Standards: Air Quality

On 1 September 2005 new national environmental standards come into effect aimed at improving air quality and controlling source emissions. Table 2-1 shows the maximum level for the amount of fine particles, carbon monoxide, nitrogen dioxide, sulphur dioxide and ozone as determined by the air quality standards.

Contaminant	Threshold Concentration	Averaging period	Permissible excess
Carbon monoxide	10 mg m^{-3}	8 hour running mean	One 8 hour period in any 12 month period
Nitrogen dioxide	200 µg m^{-3}	1 hour mean	9 hours in any 12 month period
Ozone	150 µg m^{-3}	1 hour mean	Not to be exceeded
PM_{10}	50 µg m^{-3}	24 hour mean	One 24 hour period in any 12 month period
Sulphur dioxide	350 µg m^{-3}	1 hour mean	9 hours in any 12 month period
	570 µg m^{-3}	1 hour mean	Not to be exceeded

Table 2-1. National environmental standards maximum pollution concentrations (2005).

The major sources that emit pollutants include domestic (e.g. wood and coal burning heating), industrial (coal or heavy oil power stations), mobile (petrol and diesel cars and trucks), and biogenic or natural (e.g. sea salt, wild fires) sources. Although particles (such as PM_{10} and $PM_{2.5}$) are most commonly associated with health effects, other studies link effects to carbon monoxide, nitrogen dioxide, sulphur dioxide, ozone and hydrocarbons. The types and concentrations of pollutants in the ambient air vary greatly from location to location, with time of day, and with season. As these pollutants occur together and are often closely correlated, it has been difficult to clearly identify effects of single pollutants, or potentially damaging combinations of them.

The major pollutants that can produce health effects are the gases carbon monoxide, nitrogen oxides, volatile organic compounds, and sulphur dioxide, as well as solid particulate matter (now commonly referred to as particles). Additionally, other gases (such as ozone) and particles (sulphates and nitrates) can form in the atmosphere from reactions involving some of those primary emissions. The health effects of carbon monoxide, nitrogen dioxide, ozone, particles and sulphur dioxide are reported elsewhere (Denison 2000a) and the following is a brief summary of that information. It should be pointed out that air pollution caused by combustion of fossil fuels or biomass always contains a mixture of several of the pollutants mentioned here. Epidemiological studies should be interpreted with this in mind, particularly when considering unspecific effects, such as increased mortality.

2.2.2 Carbon monoxide

Carbon monoxide is a colourless, odourless gas formed as a result of incomplete combustion of carbon-containing fuels, including coal, wood, petrol and diesel. Carbon monoxide is readily absorbed from the lungs into the blood stream, which then reacts with haemoglobin molecules in the blood to form carboxyhaemoglobin. This reduces the oxygen carrying capacity of blood, which in turn impairs oxygen release into tissue and adversely affects sensitive organs such as the brain and heart (Bascom et al. 1996). Motor vehicles are the predominant sources of carbon monoxide in most urban areas, although domestic burning and industrial sources also contribute. In general there has been a decline in carbon monoxide concentrations, which reflects the efficacy of emissions control systems on newer vehicles.

Long-standing international (and New Zealand) air quality guidelines/standards for carbon monoxide are based on keeping the carboxyhaemoglobin concentration in blood below a level of 2.5%, in order to protect people from an increased risk due to heart attacks. However, there is emerging research that indicates adverse health effects at carboxyhaemoglobin levels less than 2.5%, for example (Morris & Naumova 1998). This new information is especially relevant to New Zealand, because of the relatively high urban air concentrations of carbon monoxide.

2.2.3 Nitrogen dioxide

Nitrogen oxides (primarily nitric oxide, NO, and lesser quantities of nitrogen dioxide, NO_2) are gases formed by oxidation of nitrogen in air at high combustion temperatures. Nitric oxide is oxidised to nitrogen dioxide in ambient air, which has a major role in atmospheric reactions that are associated with the formation of photochemical oxidants (such as ozone) and particles (such as nitrates). Nitrogen dioxide is also a serious air pollutant in its own right. It contributes both to morbidity and mortality, especially in susceptible groups such as young children, asthmatics, and those with chronic bronchitis and related conditions, (Morris & Naumova 1998). One study has also shown that exposure during very early childhood can increase the risk of development of asthma induced by other allergens in the home (Ponsonby et al. 2000). Nitrogen dioxide appears to exert its effects directly on the lung, leading to an inflammatory reaction in the smaller airways and associated lung function changes (Streeton 1997). Increased lung cancer incidence has also been reported in a large case-control study (Nyberg et al. 2000) but the role of co-pollutants ("toxics") needs to be assessed. Motor vehicles are usually the major sources of nitrogen oxides in urban areas. In addition the combustion of fossil fuels from stationary sources (heating, power generation) can also be major sources of anthropogenic emissions of nitric oxide. Indoor concentrations of nitrogen oxides typically exceed those existing outside when unvented combustion appliances are used for cooking and heating (e.g. unflued gas heaters).

Air quality guidelines/standards for nitrogen dioxide are set to minimise the occurrence of changes in lung function in susceptible groups. The lowest observed effect level in asthmatics for short-term exposures to nitrogen dioxide is about 400 $\mu g\ m^{-3}$. Although less data are available, there is increasing evidence that longer-term exposure to about 80 $\mu g\ m^{-3}$ during early and middle childhood can lead to the development of recurrent upper and lower respiratory tract symptoms. A safety factor of 2 is usually applied to those lowest observed effect levels, giving air quality standards for nitrogen dioxide of 200 $\mu g\ m^{-3}$, 1-hour average.

2.2.4 Hydrocarbons

Volatile organic compounds (VOCs) are a range of hydrocarbons, the most important of which are benzene, toluene, and xylene, 1,3-butadiene, polycyclic aromatic hydrocarbons (PAHs), formaldehyde

and acetaldehyde. These compounds have a number of sources, including natural emissions from vegetation, and have very complex pathways in the environment. They can cause various direct health effects, but can also be responsible for odours and are key components in atmospheric chemical reactions leading to photochemical smog.

The potential health impacts of VOCs include carcinogenic and non-carcinogenic effects. Benzene and PAHs are definitely carcinogenic, 1,3-butadiene and formaldehyde are probably carcinogenic, and acetaldehyde is possibly carcinogenic. Non-carcinogenic effects of toluene and xylene include damage to the central nervous system and skin irritation. Heavier volatile organic compounds are also responsible for much of the odour associated with diesel exhaust emissions.

Motor vehicles are a source of volatile organic compounds in urban areas. Benzene, toluene, xylene, and 1,3-butadiene are all largely associated with petrol vehicle emissions. The first three result from the benzene and aromatics contents of petrol, and 1,3-butadiene results from the olefins content. Evaporative emissions, as well as exhaust emissions, can also be significant, especially for benzene. Motor vehicles are major sources of formaldehyde and acetaldehyde. These compounds are very reactive and are important in atmospheric reactions, being products of most photochemical reactions. PAHs arise from the incomplete combustion of fuels, including diesel.

Of the volatile organic compounds, the most important in the New Zealand context is benzene. The maximum allowable level of benzene in petrol is currently 4% by volume; however this is set to be reduced to 1% by 2006. Health effects data and guidelines/standards for hazardous air pollutants have been reported elsewhere (Chiodo & Rolfe 2000), and include recommended air quality guidelines for benzene of 10 $\mu g\ m^{-3}$ (now) and 3.6 $\mu g\ m^{-3}$ by 2010 (when the benzene content of petrol is reduced), both guidelines being annual average concentrations (Ministry for the Environment & Ministry of Health 2002). Cancer risks are assessed using inhalation unit risk (IUR). Inhalation unit risks are defined as the individual lifetime excess risk due to a chronic lifetime exposure to one unit of pollutant concentration. Estimates generally assume a no threshold, low-dose linearity. Based on the WHO air quality guidelines for Europe (2000), the excess lifetime risk of cancer (leukaemia) at an air concentration of 1 $\mu g\ m^{-3}$ is 6×10^{-6}. This was derived from the geometric mean of the range of the calculated unit risk per $\mu g\ m^{-3}$ based on updated results from the Pliofilm cohort study, 4.4×10^{-6} to 7.5×10^{-6} (WHO 2000). The 1987 WHO Air Quality guideline for Europe gave the unit risk as 4 x 10-6. The US EPA estimates a range for unit risks from 2.2×10^{-6} to 7.8×10^{-6} for an increase in the lifetime risk of an individual who is exposed to 1 $\mu g\ m^{-3}$ of benzene air.

Benzene and other toxic compounds in the air have been measured previously in New Zealand in a number of studies commissioned by the Ministry of Health (Stevenson & Narsey 1998). The most recent study surveyed benzene and other toxic organic compounds in air from July 1996 to May 1999 in Auckland, Hamilton, Christchurch and Dunedin. Results suggested an additional lifetime leukaemia risks for New Zealand benzene exposures (excluding exposures from active smoking and from evaporative vehicle emissions from indoor garages) are in the range 6×10^{-6} to 6×10^{-5}. If the whole population were exposed to these levels of risk, they would correspond to between 0.3 and 3 additional leukaemia deaths every year (Stevenson & Narsey 1999b).

2.2.5 Sulphur dioxide

Sulphur oxides (primarily sulphur dioxide and lesser quantities of sulphur trioxide) are gases formed by the oxidation of sulphur contaminants in fuel on combustion. Sulphur dioxide is a potent respiratory irritant, and has been associated with increased hospital admissions for respiratory and cardiovascular disease (Bascom et al. 1996), as well as mortality (Katsouyanni et al. 1997). Asthmatics are a particularly susceptible group. Although sulphur dioxide concentrations in New Zealand are relatively low, and motor vehicles are minor contributors to ambient sulphur dioxide, the measured levels in Auckland (for example) have increased in recent years, after many years of decline, as a result of the increasing number of diesel vehicles (and the relatively high sulphur content of diesel in New Zealand). Sulphur in diesel is being reduced, with new fuel standards set by the Ministry for Economic Development to achieve 500 ppm in 2004 and 50 ppm by January 2006. This is considerably lower than the 2000-3000 ppm contents that prevailed during the 1990's.

There appears to be a threshold concentration for adverse effects in asthmatics from short-term exposures to sulphur dioxide at a concentration of 570 $\mu g\ m^{-3}$, for 15 minutes (Streeton 1997). Ambient

air standards are based on this figure, for example the standard for New Zealand are 350 μg m^{-3} for the 1-hour average, and a guideline applies of 120 μg m^{-3} for the 24-hour average.

Sulphur oxides from fuel combustion are further oxidised to solid sulphates, to a certain extent within the engine and completely in the atmosphere. The former inhibits the performance of exhaust emission control equipment for nitrogen oxides and particles, and this is a major reason why the sulphur contents of petrol and diesel are being reduced internationally. New Zealand reduced the sulphur content of diesel from levels as high as 3000 ppm in 2001 to 500 ppm as of August 2004. The sulphur content in diesel will again be lowered in 2006 to 50 ppm. Many countries are moving to "sulphur-free" petrol and diesel (less than 10 ppm). It is an unfortunate reality that unless the sulphur content of diesel is less than about 120 ppm, vehicles with advanced emission control systems are actually net producers of additional fine particles, because of oxidation of the sulphur oxides to sulphates.

2.2.6 Particulate matter

Evidence from epidemiological studies consistently points to associations between exposure to ambient particulate matter and adverse health effects, even at low levels commonly encountered in urban areas of most countries. Primary particles are emitted directly from sources such as motor vehicles or wood smoke and secondary particles are formed through atmospheric reactions of sulphur dioxide, nitrogen oxides, and certain organic compounds. Most studies of health endpoints have used ambient PM_{10} mass concentrations and/or other ambient particulate indicators such as black smoke (BS), coefficient of haze (COH),backscattering (bsp), total suspended particulates or pre-cursors of secondary particle production through chemical reactions in the atmosphere – sulphates and nitrates. A distinction is made between PM_{10} ("thoracic" particles smaller than 10 μm in diameter that can penetrate into the lower respiratory system), $PM_{2.5}$ ("respirable" particles smaller than 2.5 μm that can penetrate into the gas-exchange alveolar region of the lung), and ultrafine particles smaller than 100 nm which contribute little to particle mass but which are most abundant in terms of numbers and offer a very large surface area and increased lung penetration (Brunekreef. & Holgate 2002). The largest particles (coarse fraction) are mechanically produced by attrition of larger particles. Small particles (<1 μm) are largely formed from gases, and the smallest (<0.1 μm, ultrafine) are formed by nucleation resulting from condensation or chemical reactions that form new particles.

Some recent studies indicate that, in general, $PM_{2.5}$ can be more closely associated with health effects than PM_{10} and it is possible that finer particles cause greater effects, owing to their ability to accumulate and reach the lower region of the respiratory systems. Evidence is also emerging that individual constituents of PM_{10} and $PM_{2.5}$, such as sulphates and strongly acidic particles, are sometimes more closely associated with health effects than the simple particle mass. Many studies have related day-to-day variations in particle mass (especially that indicated by PM_{10}) to day-to-day variations in health parameters. Acute effects have included increased daily mortality, increased rates of hospital admissions for exacerbation of respiratory disease, fluctuations in the prevalence of bronchodilator use and peak flow reductions.

There is no apparent threshold concentration below which no effects occur. As a result the World Health Organization (WHO) has decided not to recommend a specific concentration as the air quality guideline for particles, but has instead expressed the guideline in the form of dose-response relationships for different health effects (WHO 2000). This implies that the "acceptable" level of health impact and a "safe" level of the air pollutant needs to be established by each jurisdiction using the WHO Air Quality Guidelines. Most countries (including New Zealand) have used this advice and other input to set national or local guidelines (typically 50 μg m^{-3} for PM_{10}, 24-hour average) aimed at minimising the occurrence of health effects.

The 24-hour PM_{10} standard has been set in New Zealand as the primary standard for reducing concentrations and subsequent health effects. Other countries have set annual standards, but in many circumstances these do not adequately protect against short term high concentrations and the health effects of these. In general, if a realistic 24-hour standard is being met, then a realistic annual average concentration will also be met.

Evidence is also emerging that long-term exposure to low concentrations of PM_{10} in air can be associated with mortality and chronic effects, such as increased rates of bronchitis and reduced lung function. There have been two published US cohort studies that have analysed the associations between

longer-term (annual) average PM_{10} levels and longer-term mortality (Dockery et al. 1993; Pope III et al. 1995b). More recently, an update of the second study showed that long-term exposure to combustion related fine particulate pollution ($PM_{2.5}$) was an important environmental risk factor for cardiopulmonary and lung cancer mortality, after smoking and other potential confounders were adjusted for (Pope III et al. 2002). These studies of long-term exposure have shown a greater increase of mortality than most of the short-term exposure studies. This difference may reflect the non-linear character of air pollution-mortality relationships. Variation in the exposure-response relationships among the large number of short-term mortality studies could be due to differences in air pollution sources, different vulnerability of the population or different climatic conditions. Sensitive groups that appear to be greater risk of particulate air pollution include the elderly, those with pre-existing respiratory conditions and cardio-pulmonary diseases such as asthma, smokers, children and infants. Particulate matter, especially fine particulate matter, is the primary contributor to a variety of adverse health effects associated with air pollution. However there are difficult technical issues in separating the effects of fine and course particles and in separating particle effects from possible effects of gaseous co-pollutants.

2.2.7 Ozone

Ozone is a secondary air pollutant formed by reactions of nitrogen oxides and volatile organic compounds in the presence of sunlight. These primary emissions arise mainly from motor vehicles. Concentrations in city centres tend to be lower than those in suburbs, mainly as a result of the scavenging of ozone by nitric oxide originating from traffic (Brunekreef & Holgate 2002). Ozone is only one of a group of chemicals called photochemical oxidants (commonly called photochemical smog), but it is the predominant one. Also present in photochemical smog are formaldehyde, other aldehydes, and peroxyacetyl nitrate. Ozone is another air pollutant that has respiratory tract impacts (Woodward et al. 1995). Its toxicity occurs in a continuum in which higher concentrations, longer exposure, and greater activity levels during exposure causes greater effects. It contributes both to morbidity and mortality, especially in susceptible groups such as those with asthma and chronic lung disease, healthy young adults undertaking active outdoor exercise over extended periods, and the elderly, especially those with cardiovascular disease. Substantial acute effects occur during exercise with one hour exposures to ozone concentrations of 500 $\mu g\ m^{-3}$ or higher. Ozone, like particles, is an air pollutant for which there is no indication of a threshold concentration for health effects (Streeton 1997). However, unlike particles, the WHO has established a specific air quality guideline concentration for ozone. More than any other air pollutant, there is considerable variation in air quality guidelines/standards for ozone, because of complexities involved in reducing ambient concentrations of it. In New Zealand a relatively "pure" approach has been taken, and air quality standards for ozone of 150 $\mu g\ m^{-3}$, 1-hour average, and 100 $\mu g\ m^{-3}$, 8-hour average have been established.

3 Relevance to Policy

3.1 Background

Like many countries around the world, New Zealand has already enforced legislation or actions to reduce ambient air pollution levels and the associated health effects. However there is ongoing pressure to devise and enforce policies given studies are still finding acute and longer-term adverse health effects of exposure to air pollutants even at low ambient concentrations or at levels below existing health guidelines. Also of importance is that these effects are generally found to fall disproportionably on vulnerable groups in society, such as older people, those with pre-existing diseases, and possibly poorer people. In New Zealand socio-economic status is unequally distributed by ethnicity. Maori and Pacific peoples have lower socio-economic status than the average (Ministry of Health 2000b). Lower socio-economic groups may suffer greater health effects from air pollution exposure due to the combination of greater exposure and susceptibility (Ministry of Health 2000b, O'Neill et al. 2003). Home heating and quality of insulation is essential for a healthy indoor climate (Howden-Chapman et al. 2003), but the cost of heating can lead some families to heat inadequately or use inefficient and polluting heating systems.

The need to reduce greenhouse gas emissions from fossil fuel burning has also given motivation for developing policies to reduce vehicle emissions and improve efficiency of home heating systems, especially in anticipation of population growth. An alternative to fossil fuel burning for heating purposes is wood, as it is essentially "greenhouse neutral" (unless its use involves major energy use in planting, harvesting and transporting the fire wood). However wood burning emits particulates, CO, and a variety of organic compounds including PAHs.

Christchurch experiences on average 30 days each year when the 24-hour average PM_{10} concentrations exceed the air quality standard of 50 $\mu g\ m^{-3}$. The peak 24-hour levels are above 200 $\mu g\ m^{-3}$. Emissions inventory and modelling show that home heating with wood is the main source of ambient PM in the winter (90%), May to August, whilst motor vehicles are the main source of NO_2. CO comes from both sources. During the rest of the year (8 months) motor vehicles and industries are the sources of almost all air pollution. The CO standard (10 mg m^{-3}, 8 hour average) is exceeded occasionally. It is important to establish the health risks of these exposures in order to set appropriate air quality standards and other regulations for domestic home heating, industry and vehicle emissions.

A major objective of this research is to establish the evidence-base that will enable development of effective policies for addressing air pollution in New Zealand. Creating effective evidence based policy requires linking pollutants and the associated health and economic endpoints back to specific source categories. In this way, the major sources can be targeted and the biggest improvements achieved. Given limited resources, prioritising policy options or actions (at a local or national level) in terms of reducing both pollutant levels and source emissions (as well as understanding the implication of doing nothing) is required. This includes discussing the potential costs and benefits of the implementation of policies to reduce the health and economic effects identified.

There has been increasing pressure to design and implement environment, health and transport policies that improve overall air quality in New Zealand that will contribute to sustainable development. An on-road remote sensing of vehicle emissions from 40,000 vehicles at 16 sites during April 2003 in the Auckland region also gave results that showed the most polluting 10% of vehicles or "gross emitters" are responsible for 53% of the total CO emissions (Fisher et al. 2003). This has recently been confirmed with a second campaign in May 2005 (unpublished). Public awareness campaigns such as the Auckland Regional Council's 0800-Smokey programme, and visibility degradation in cities has also raised the public profile of air pollution as an important environmental health concern. A comprehensive report has recently quantified the monitoring of PM_{10} throughout New Zealand (Ministry for the Environment 2003).

3.2 Regulatory framework

Government authorities throughout New Zealand are under increasing pressure to design and implement policies that improve the overall quality of the environment. This chapter outlines the regulatory framework governing air quality in New Zealand.

3.2.1 Central government ministries

Central government ministries including the Ministry for the Environment, and Ministries of Health, Transport, and Economic Development are involved in advising government, releasing national policy statements, and developing policies, regulations or legislation that influence discharges to air, particularly when national solutions are required. The ministries work closely with other agencies to develop and implement national strategies for improving air quality.

Ministry for the Environment

The Ministry for the Environment (MfE) advises the Government on New Zealand's environmental laws, policies, standards, and guidelines, monitor how they are working in practice, and take actions needed to improve them. The Ministry has duties under various roles, such as the Resource Management Act (RMA), which include developing national tools to achieve sustainable air quality management.

The Ministry recently introduced national environmental standards for air quality, some of which came into force in 2004 while others will do so in 2005 and 2006 (Ministry for the Environment 2003). The standards aim to create a level playing field across New Zealand, provide certainty and consistency, guarantee a similar level of protection for the health of all New Zealanders, and drive effective regional and national policies to improve air quality. The standards are largely based on the Ambient Air Quality Guidelines (Ministry for the Environment & Ministry of Health 2002), with the addition of two compliance criteria for most pollutants - a specified number of times that the standard limit can be exceeded per year, and an upper maximum limit that the above exceedences cannot exceed even once. Prior to developing the standards, the Ministry evaluated the objectives and policies of the standards produced numerous cost benefit analyses. The studies concluded that the proposed standards were the most appropriate, effective and efficient means of meeting the Minister for the Environment's objectives for air quality management.

Ministry of Transport

The Ministry of Transport (MoT) is the Government's principal transport policy adviser. It leads and develops national policy within the framework of the New Zealand Transport Strategy, which was adopted by the Government in 2002. The Ministry also assists the Minister of Transport in advancing legislation through Parliament, drafting regulations and rules in association with the transport Crown entities, and representing New Zealand's transport interests internationally.

The Ministry of Transport works closely with other agencies within the transport sector including crown entities such as Transit New Zealand (management of state highways), Land Transport New Zealand, and the New Zealand Police. Recent policy developments intended to reduce harmful emissions from vehicles include a vehicle exhaust rule, phased in from January 2004 that ensures that all vehicles entering New Zealand are manufactured to internationally recognised emission standards from the United States, European Union, Japan, or Australia. In June 2005 the government announced a series of further measures to tackle pollution from vehicles. These included a visible smoke check at vehicle Warrant/ Certificate of Fitness inspection, tightening controls on imported vehicles and prohibiting the removal, or tampering with emissions control. In addition the MoT will carry out education of vehicle users on the need for, and benefits of, regular vehicle maintenance and repair.

The MoT will also promote: the use of biofuels; improved awareness of vehicle fuel efficiency; increasing transport funding to tackle severe traffic congestion in key urban areas including road developments; public transport initiatives and walking and cycling promotion. Further research on the health impacts of vehicle emissions will be encouraged. The MoT supports the Ministry of Economic Development led work on revising the standards for fuel specifications. From 1 January 2006, revised diesel fuel specifications will lead to the reduction of the sulphur content in diesel fuels to 50 parts per million, making New Zealand diesel cleaner and bring it into line with European standards. The fuel specifications for benzene will also be lowered from 3% to 1% from 1 January 2006.

Ministry of Health

The Ministry of Health (MoH) aims "to improve the health and independence of New Zealanders and reduce the inequalities in health status between all New Zealanders including Maori and Pacific peoples". As outlined in the New Zealand Health Strategy, there are broader determinants of health, which lie outside the health sector, including general socio-economic conditions and environmental conditions (including air quality), social, and community influences, and working and living conditions, which can affect public health and well being (Ministry of Health 2000a). Many of these factors influencing health status act primarily at the level of whole communities and population groups, rather than individuals. Gains in health status will only be achieved through the co-ordinated action of policy makers in many sectors.

The whole of government initiative

The regulatory framework and its implementation is also influenced by the recent "whole of government initiative that aims at reducing duplication in actions between departments and better collaboration to achieve common goals. This could, for instance, bring the Accident Compensation Corporation (ACC) or the Energy Efficiency and Conservation Authority (EECA) together with local government in as partners in air pollution control through investments in injury prevention that also reduces air pollution. An example would be support for public transport promotion that would reduce both traffic crash injuries and motor vehicle air pollution. Another example would be efforts to improve home insulation for energy conservation purposes, which would reduce the amount of firewood needed to heat a house.

3.2.2 Local government - Regional and City Councils

At a regional and local level, under the RMA (sections 5 to 8) Regional Councils and unitary local authorities (City and District Councils) have a responsibility for managing and controlling discharges into the air and therefore managing the quality of the air people breathe or exposed to. The purpose of the RMA is to promote the sustainable management of natural and physical resources, including air. Sections 5 provides that the purpose of the Act is to promote the sustainable management of natural physical resources including safeguarding the life supporting capacity of the air, while section 6 to 8 describe other matters (including the Treaty of Waitangi) which must be considered when making decisions. In particular section 7(f) states persons exercising powers under the Act must have particular regard to "maintenance and enhancement of the quality of the environment". Regional Councils are responsible for gathering sufficient information about the state of the environment (e.g. ambient air quality monitoring) to enable them to carry out their functions (section 35). To manage the environment, Councils must produce regional policy statements and optional regional plans specifying objectives, policies and rules to address issues of concern (sections 63 to 70). Regional Councils also set up education programmes such as 0800-smokey aimed at reducing pollution. Other responsibilities include regional transport (including public transport), and water and river management, biosecurity and environmental regulation.

Under the RMA City and District Councils are required to sustainably manage the city or districts natural and physical environment, including land, water, soil, resources and the coast. City and District Council's primary functions include resource management (urban planning), for example issuing land use permits and controlling subdivision of land. They are also responsible for city services such as water, sewerage and refuse disposal, local roads, libraries, parks and reserves and community development. Overall they have a relatively limited regulatory capacity in terms of controlling of discharges of contaminants into air, which is the Regional Council's function. Some land-use issues managed by local and District Councils are relevant to air quality. For example parking policies can manage congestion and parking pricing can encourage alternative modes of transport. There is often a high degree of co-operation between Regional and City Councils as both have complementary roles.

The RMA requires that environmental policies be subjected to cost benefit analysis before they are adopted, which is one way to choose between competing policies. However environmental policy evaluation is often complicated by the fact that while the costs of implementing a policy are often relatively easy to identify, policy benefits are generally less tangible and nearly always difficult to quantify in monetary terms.

3.3 International agencies

International agencies including WHO and the European Union, as well as major national agencies, such as the US EPA recommend and update air quality standards, dose-response relationships, and set research agendas, as new scientific data in the field of air pollution toxicology and epidemiology and new developments in risk assessment methodology become available. These agencies are concerned with the public health and environmental effects of air pollution, and ways to protect public health and reduce air pollution. The norms and standards recommended by these agencies are useful to governments and environmental health authorities that attempt to protect people from the harmful effects of environmental air pollution in their own country. There are initiatives from within government departments both in New Zealand and Australia to achieve greater collaboration of relevant research topics and better harmonisation of standards, guidelines, and evaluation methodologies. During 2005, Australia commenced a review of its National Environmental Performance Measures.

International agreements that New Zealand is party to, such as the Kyoto Protocol, which aims to reduce the net emissions of certain greenhouse gas emissions (primarily CO_2), also influence air quality levels in New Zealand.

3.4 Private sector

Private or local industries can influence the level of air quality in various ways. For example some businesses contribute to poor air quality through the pollutants emitted from their operations. Conversely, new technologies integrated into products such as solar heating, more efficient wood heaters, and new engine type vehicles (electric, fuel cells, and hybrids) may reduce emissions. In addition private businesses can indirectly influence the level of air quality.

The major cause of elevated PM_{10} concentrations in many towns – including Christchurch - is emissions from solid fuel home heating. Many appliances for heating are designed and manufactured in New Zealand. The Home Heating Association has a central role in improving the performance of such appliances, and has worked to do so with Councils and the Ministry for the Environment. Fuel suppliers can also have an influence, and one major coal supplier has undertaken to reduce its sales of coal in Christchurch.

The role of private sector organisations, (even relatively pro active bodies such as the New Zealand Business Council for Sustainable Development) can be vital in effecting change, but this role is not discussed in this report, except as aids to implementing policy.

Individual behaviour

Individual and community behaviours influence levels of ambient air pollution. One of the major factors for Christchurch is the choice to use a solid fuel burner for heating, and the choice of fuel for that burner. The difference between using hard dry wood, as opposed to soft or wet wood can make a large difference in the amount of pollution emitted. There is wide range of solid fuel heaters on the market, and the variations in individual operating techniques also have a significant effect on emissions (Scott, 2005). Similar choices made in transport choice can have an effect. For example people's time-activity patterns and mode of transport influence the levels of transport emissions. There are various measures for mitigating vehicle emissions through influencing individual behaviour including car pooling, compressed work week, tele-working, school travel management tools such as school buses. These are discussed further under the policy sections of this report.

3.5 Summary

Sustainable air quality requires an integrated approach that encompasses coordination and consensus building across all sectors and levels, identification of technically feasible abatement options, and introduction of policies and instruments to support implementation.

4 Overseas Research

4.1 Scope

The purpose of this chapter is to summarise some of the overseas research conducted on health effects of air pollution, and present a brief review of the results obtained. This review has not yet been able to incorporate all new published research in 2004 and 2005. This will be carried out for the final HAPiNZ report.

4.2 Air pollution and health effect linkages

4.2.1 Air pollution episodes

The history of documenting health effects of air pollution goes back as early as 1930s. During early days, studies were more focussed on severe air pollution episodes. One of the early episodes documented occurred in the Meuse Valley of Belgium in December 1930 (Roholm 1937). Several hundreds of people fell sick mostly in respiratory related illnesses and more than 60 people died during a few days episode. In October 1948, another severe air pollution episode occurred in a small industrial town Donora with a population of about 14,000 situated in a valley in Pennsylvania, USA. Twenty deaths were reported during and immediately following the episode (Ciocco & Thompson 1961). A large increase in various degrees of illness mostly respiratory morbidity and irritation of eyes, nose, breathlessness, headaches, vomiting, nausea, and sore throat was also reported. The most severe episode in the history of air pollution episodes which has been well documented, was the "London Fog", which occurred in London, England during four days from 5th-8th in December 1952 (UKMoH 1954). The air concentrations were 10-100 times higher than what is now recorded in New Zealand centres, and the mortality was dramatically increased, by at least 100% during the "Fog" period. Infants and the elderly were particularly vulnerable.

4.2.2 Epidemiological studies

The most widely used study design in air pollution epidemiological studies is the time series mortality study. Such studies analyse the temporal distribution of deaths and air pollution at current levels. Time series studies describe the short-term relationship between air pollution and mortality (and/or morbidity) by comparing daily mortality or daily cases of morbidity with daily air pollutant levels. A number of time series epidemiological studies during the last 15 years have documented associations between urban air pollution and daily morbidity and/or mortality. Respiratory and heart disease are the main effects. These relationships were shown in a number of cities for small air particles (PM_{10} or $PM_{2.5}$) and gaseous air pollutants (CO, NO_2, SO_2, Ozone) (Burnett et al. 1998; Katsouyanni, K. et al, 1997; Michelozzi et al. 1998; Moolgavkar et al. 1995; Samet et al. 2000; Schwartz & Dockery 1992a; Schwartz & Dockery 1992b; Xu et al. 1994). Similar relationships have also been reported from Sydney (Morgan et al. 1998), Brisbane (Simpson et al. 1997), Melbourne (Simpson et al. 2000) and Perth. The results ranged from 0.7% to 1.6% increase in daily mortality with a weighted mean of about 1% increase for every 10 $\mu g\ m^{-3}$ increase in PM_{10} (Pope III et al. 1995b).

The air pollution epidemiological studies have been critically reviewed in many publications (Dockery & Pope 1994, Pope et al. 1995a, Schwartz 1994). A review of reviews, which critically assessed some 15 reviews of the published studies studying the short-term relationship of air pollution on mortality and morbidity reached the conclusion that the short-term relations reported by many studies were valid and causal (Dab et al. 2001).

A wide range of modelling techniques has been used in time series studies to control for confounding variables like season and weather variables. Earlier studies mainly used Generalised Linear Models (GLM) with parametric smoothers (Katsouyanni et al. 1997; Touloumi et al. 1996; Zmirou et al. 1996). With methodology development in epidemiological studies, the Generalised Additive Models (GAM) have become more and more common in time series studies as it allows to fit models with non-linear

functions of confounders to adjust them in analysis (Anderson et al. 2001; Hoek 2003; Samet et al. 2000). It was reported that the use of default convergence parameters in the GAM functions of the most commonly used statistical software SPlus could lead to an overestimate of the actual effect (Dominici et al. 2002). This prompted other researchers to reanalyse their data using GAM with more stringent convergence criteria and by using the Generalized Linear Models. Such reanalysis of a number of earlier studies did not show any major differences from the earlier findings of the short-term association between air pollution and mortality (Fairley 2003; Hoek 2003).

4.2.3 The relation between exposure duration and effects

Health effects of air pollution can be categorised according to whether exposure is short term or long term. The use of concepts and terms in this area is far from clear. Table 4-1 shows the types of exposure and health effect situations that are relevant for air pollution in New Zealand.

	Duration of effect measurement		
	Short term	**Medium term**	**Long term**
	Hours - 1 day	**2 – 60 days**	**One year or longer**
Duration of exposure			
Hours to 1 day	True short term effects (daily mortality in CVD)	Cumulated effects of daily exposure (distributed lag models)	
Season		Cumulated effects of long term exposure (distributed lag models)	
Years			True long term effects (e.g. COPD, cancer)

Table 4-1. Types of exposure and health effect situations.

In terms of the public health impact the long term relationships are of greatest interest.

Daily time series studies can only demonstrate the short-term relationship between exposure to air pollution and mortality/morbidity. These studies do not provide evidence of mortality due to long-term exposure to air pollution (McMichael et al. 1998). However, time-series studies may indicate differences in health risk between daily air pollution, climate conditions, and different age/sex groups that are relevant to the long term relationships. The relationships between long-term exposure and mortality/morbidity have been evaluated using ecological studies and more recently prospective cohort studies. The ecological studies have found that on average the mortality and morbidity rates tend to be higher in areas with higher air pollution than in areas with lower air pollution (Chappie & Lave 1982; Lipfert 1984; Ozkaynak & Thurston 1987). However, these population-based studies could not control for individual characteristics and have been largely discounted due to the concern that the observed associations could be due to confounding (Pope III et al. 1995b).

Prospective cohort studies of mortality/morbidity associated with long-term exposure to air pollution using individual data on smoking habits, occupation and other potential confounding factors provide better evidence of reduced life expectancy due to air pollution exposure. Relatively few of these studies have been completed due to time and cost involved. The Harvard Six-Cities study (Dockery et al. 1993), the American Cancer Society study (ACS) (Pope III et al. 1995b; 2002) and the Adventist Health Study of Smog (AHSMOG) (Abbey et al. 1999) are the most widely quoted. The Harvard Six-Cities study compared populations in cities with different annual average air pollution (PM) levels and found an increase of annual mortality. The ACS study adjusted for individual smoking habit data, and found that an increasing annual average PM level in a city increased annual mortality for people from that city. An independent assessment of the results of both the Harvard Six-Cities and the ACS studies confirmed the original findings of an association between mortality and particulates (Krewski et al. 2000). The ACS study has been extended with an addition of eight years of follow up data (Pope III et al. 2002). The new results supported the original findings and showed clearer associations between annual average air

pollution ($PM_{2.5}$) and heart- and lung-disease as well as lung cancer, after smoking habits and several other potential confounders had been taken into account. Individual exposure was assumed to be identical to the air pollution monitoring results of the city a person had lived in. These studies of long-term exposure have shown more substantial mortality increase (about 4% per 10 μg m^{-3} PM_{10} increase of annual average) than most of the short-term exposure studies (about 1% per 10 μg m^{-3} of daily average). Relatively high mortality increase was also found in the study by Scoggins et al (2004) of the effect of annual NO_2 exposure in Auckland (see Appendix 1).

The modelling method that includes same day PM_{10} concentrations and lags of PM_{10} levels as independent variables ahs been called the distributed lag model (Schwartz, 2000: Zanobetti et al. 2000). The overall effect of a unit increase in PM_{10} is the increase in mortality due to a unit increase in PM_{10} level on the same day plus the increases in mortality due to PM_{10} on previous days.

Schwartz (2000) used a quadratic polynomial distributed lag model with 5 lags to study the effect of PM_{10} on daily mortality of persons 65 years of age and over in 10 US cities. He compared the results with the results from the unconstrained distributed lag models, and the models with the same day PM_{10} levels and the 2 days moving average of PM_{10} levels (lag 0 and lag 1) as air pollution exposures and found that the overall effects from the distributed lag models were higher than the effects using one or 2 days moving average of PM_{10} levels. The study reported an increase of 1.41% in daily deaths for a 10 ug/m3 increase in daily PM_{10} (Schwartz, 2000).

Another study in ten European cities (Zanobetti et al. 2002) estimated the combined effects of PM_{10} on the same day and the lagged effects up to 40 days using 3rd and 4th degree polynomial distributed lag models. It has found that the estimated effect of PM_{10} was more than doubled in many cities when lagged effects were considered as compared to the 2 days moving average (same day and the day before) of PM_{10} levels. The results were consistent for all distributed lag models including unconstrained distributed lag models (Zanobetti et al. 2002).

Distributed lag modelling has also been used to study the extended effects of cold temperature and air pollution simultaneously. Goodman et al (2004) analysed Dublin data from April 1980 to December 1996 to assess the cumulative net effects of daily minimum temperature and Black Smoke (BS) particulate air pollution exposure over the following 40 days using polynomial distributed lag models. As in other studies, this study also reported higher effects of the extended exposure to air pollution on mortality. The study estimated a 1.1% increase in total non-trauma mortality associated with an increase of 10 ug/m3 in daily mean BS over the succeeding 40 days whereas the effect of each 10 ug/m3 increase in three days mean BS was 0.4% increase in total non-trauma mortality (Goodman et al. 2004).

Results from the studies analysing cumulative exposure of air pollution showed that the effects of air pollution persist for more than a few days. This is also supported by the findings from the London smog episode study, which showed that the peak deaths generally lagged behind the peak exposure (UKMoH, 1954). The higher relative risk associated with cumulative exposure to PM_{10} compared to the results using one or two days moving averages of PM_{10} showed that the time series studies using a single day exposure level underestimate the real risk of PM_{10}. The risk of longer-term exposure to PM_{10} will generally be higher than the risks that have been reported by the time series studies using a single day PM_{10} level or two/three days moving average of PM_{10} levels as exposure variables. It is consistent with the findings of higher risk estimates in cohort studies than in the time series studies (Dockery et al. 1993; Pope et al. 1995; Pope et al. 2002).

Both short-term and long-term associations between air pollution and mortality/ morbidity have been established using ecological studies, time series, and prospective cohort studies. But the biological mechanisms of these associations are not clear. Recent research has looked at the association between pollutants and initial symptoms such as heart rate variability (Park et al. 2003), increase in blood fibrinogen (Hoeppe et al. 2003), increased blood pressure and blood pressure reactivity (Liao et al. 2003) that may lead to serious health conditions. Further research is needed to understand why there is an effect of air pollution on mortality and morbidity.

A key concern in interpreting the short-term association between air pollution and mortality as demonstrated by time series studies is that it is unclear whether the association is just due to the early deaths of people who are going to die in a few days or weeks regardless of air pollution exposure. If deaths are being brought forward by just a couple of days, the public health impact of air pollution

would be much less than if life expectancy is being reduced by months or years (Brunekreef & Holgate 2002). Recent studies studying the "harvesting" of deaths (mortality displacement) suggest that the association is not due to mortality displacement (Dominici et al. 2003; Schwartz 2001; Zeger et al. 1999; Morgan et al 2003). Instead they have found the effect estimates from the existing time series studies are smaller than the effect when the air pollution exposure variable in the analysis takes the medium term into account (see Table 4.1). Thus, the cumulated effect is greater than the "true short term" effect. The effects estimated from the prospective cohort studies are even greater (Table 4-2).

Duration of effect measurement	**Dose response relationships (**% increase in daily mortality for 10 μg m^{-3} increase in PM_{10}	**Reference**
Short term	1	APHEA2 study (Katsouyanni et al 2001), European 10 cities (Zanobetti et al 2000), European 17 cities (Medina et al 2004)
Medium term	1.6	APHEA2 study (Katsouyanni et al 2001)
Long term	8.4	Harvard 6 cities study (Pope et al 1996)
Long term	4.2	American Cancer Society study, first stage (Krewski et al 1998, Pope et al 1995)
Long term	6	American Cancer Society study, second stage (Pope et al 2000).

Table 4-2. Comparison of dose-response relationships for different study types.

4.3 Summary

A large number of epidemiological studies have been carried out worldwide that have shown associations between ambient air pollution levels and adverse health effects, including increased mortality. The long-term mortality increase associated with long-term exposure is substantially higher than the short-term increase. The exact biological mechanisms by which air pollution causes increased morbidity and mortality remain to be determined. It would seem that inflammation of the airways is a common pathway for several air pollutants. It is also apparent that there are groups within the population that are particularly susceptible to the effects of air pollution, including the elderly, people with existing respiratory and cardiovascular disease, asthmatics, infants and children. Another issue that has not yet been resolved is whether particulate air pollution from different sources causes different levels of health risk.

5 New Zealand Situation

5.1 Scope

The purpose of this chapter is to examine the specific elements of the New Zealand situation. It includes a discussion of the applicability of overseas results in New Zealand and previous New Zealand studies linking air quality and health effects. Factors that may influence the extent to which overseas studies can be applied here include the types of air pollution that are associated with significant human exposures, the geographic setting and the climatic situation, and the socio-economic status and general health status of the exposed population. There is reason to believe that most of the studies from Europe or North America can be used to interpret what may happen in New Zealand, taking into account the particular pollutant mixtures and climate factors.

5.2 Applicability of overseas research

One measure of the applicability of overseas research is to consider the results of studies in New Zealand. A national risk assessment commissioned by the Ministry of Transport, confirmed that similar levels of health effects occur here, even though there is generally a good air quality relative to many other parts of the world. The MoT study estimated that the number of people above 30 years of age who experience "premature mortality" in New Zealand due to exposure to PM_{10} emissions from vehicles is 399 per year (with a 95% confidence interval of 241-566 people). This compares with 571 people above 30 experiencing premature mortality due to PM_{10} from other air pollution sources (mainly home heating), and with 502 people dying from road accidents (all ages) in 1996 (Fisher et al. 2002). This result, based on a methodology used in Europe (Kunzli et al. 2000), suggests mortality due to vehicle related air pollution is similar to the accident road toll. The ratio between traffic air pollution-related deaths and fatal motor vehicles accidents in New Zealand (0.8) is also consistent with overseas findings (Table 5-1). The ratio in Europe at around 2-3 is higher than that in New Zealand, no doubt due to higher levels of air pollution. Because of its relative geographic isolation and the prevailing relatively strong wind patterns, New Zealand has lower urban air pollution levels per km travelled than European countries. In addition the general traffic safety record in New Zealand (as judged by age-standardized traffic crash mortality rates) is worse than in European countries. Thus, the ratios in Table 5-1 make sense.

Other relevant studies to date are those carried out in Christchurch. These show an association between 24-hour concentrations of PM_{10} and mortality (1-day lag) and hospital admissions. A 10 $\mu g\ m^{-3}$ increase in 24-hour PM_{10} is associated with a 1% increase in all causes mortality and a 4% increase in respiratory mortality (Hales et al. 2000b), and a 3% increase in respiratory hospital admissions of adults and children and a 1% increase in cardiac hospital admissions of adults (McGowan et al. 2002). The results of these studies are consistent with studies elsewhere in the world, especially those for which the major sources of PM_{10} are solid fuel combustion processes.

Country	Population (million)	Traffic accident deaths (A)	Mortality due to traffic air pollution (B)	Ratio B/A
France	58.3	8,919	17,629	2.0
Austria	8.1	963	2,411	2.5
Switzerland	7.1	597	1,762	3.0
New Zealand	3.7	502	399	0.8

Table 5-1. Air pollution mortality (for adults ≥ 30 years) and the road toll (1996).

The Christchurch studies are related to the winter-time particles problem caused by wood and coal combustion for domestic heating. They may not be relevant to PM_{10} concentrations associated with motor vehicles. For instance, New Zealand in 2001 had 430,000 registered diesel vehicles, and this had grown to 574,500 by 2004, about 18% of the fleet (Land Transport NZ, 2004). Also, as mentioned in previous chapters of this report, the sulphur content of New Zealand diesel prior to 2001 measured as high as 3000 ppm. As of 2004 the maximum allowable sulphur content in New Zealand was lowered to 500 ppm and in 2006 it will again be lowered to 50 ppm. These reductions bring New Zealand in-line with European nations where the maximum sulphur content of diesel is 350 ppm, reducing to 50 ppm in 2005, and in several urban areas it is already less than 10 ppm. Therefore in any studies conducted in New Zealand prior to 2001, it is likely that the PM_{10} associated with motor vehicles may be relatively high in sulphates.

Although the available evidence is limited, the linear dose-response relationships for the health outcomes of mortality and hospital admissions in the WHO Air Quality Guidelines (WHO 2000) show a steeper relationship (that is, a larger relative risk) for sulphates than for either total PM_{10} or other particulate size fractions. There is still considerable uncertainty as to whether SO_2 is the pollutant responsible for the observed adverse health effects, or rather, a surrogate for ultrafine particles or some other correlated substance.

A major point of difference between New Zealand urban areas and most cities in developed countries overseas is the relatively high concentrations of carbon monoxide (see e.g. WHO 2000, Dirks et al, 2003). The biological mechanism by which carbon monoxide affects health is quite specific. It reduces the oxygen transport capability of haemoglobin causing "chemical suffocation". It is worth considering what impact the impaired oxygen release to tissue, and the consequent effects on such sensitive organs as the brain and heart, has on the ability to be able to cope with exposures to other air pollutants, such as PM_{10}, which can cause inflammation of airways.

Another air pollutant of importance is nitrogen dioxide. There have been some relatively high concentrations of nitrogen oxides measured at inner city sites in Auckland and Christchurch close to major roads and busy intersections (Auckland Regional Council, 2003 – plus regular web site updates www.arc.govt.nz, Environment Canterbury, 2002, plus regular web site updates www.ecan.govt.nz), Generally elevated concentrations of NO_2 in Christchurch and other urban areas are also due to emissions from other combustion sources – mainly domestic heating and industry. Again, the impact of exposures to nitrogen dioxide, which affects the surface of the lungs, on the ability to cope with concentrations of PM_{10} (for example) is an area of research well worth considering in the New Zealand context. A spatial analysis of long-term effects of annual exposure to NO_2 and mortality in Auckland found a similar number of pre-mature deaths attributable to air pollution as the Ministry of Transport study results for Auckland (Scoggins et al. 2004) (see Appendix 1).

Overseas studies that are also relevant and applicable to New Zealand are those that estimate the cancer risk associated with atmospheric exposures to benzene. As mentioned in previous chapters of this report, prior to 2004 New Zealand petrol had a high benzene content, especially the "premium" grade, which exceeded 4% by volume (as of 2004 maximum allowable benzene content was set at 3%, reducing to 1% by volume by 2006), so the health effects of exposures to benzene are worthy of study. Unfortunately benzene exposure data is limited. A comprehensive national survey was carried out 10 years ago (Ministry of Health, 1999), but this has not been updated and there is no more recent information on benzene in air in the New Zealand environment. The cancer risk (leukaemia) can be estimated using the geometric mean of the World Health Organization unit risk (that is, for 1 $\mu g\ m^{-3}$ exposures) of 6.0 per million of the population (WHO 2000).

5.3 New Zealand studies on air quality and health

Research undertaken so far in New Zealand has not been sufficient to fully quantify the health effects associated with air pollution or to identify the best measures to reduce them (Scoggins 2004). Therefore, the HAPiNZ project includes new epidemiological studies of selected research questions that are crucial for valid health impact assessments in New Zealand.

5.3.1 Epidemiological and clinical studies

An ecological cross-sectional study was undertaken in the Auckland region for the period 1996-1999 which integrated the use of urban air shed models with GIS to establish the relationship between mortality and long-term annual exposure to air pollution. After controlling for age, sex, ethnicity, socio-economic status, smoking, and occupation there was a 1% (95%CI: 0.7-1.3) increase in non-external causes mortality per 1 μg m^{-3} increase in annual average NO_2. Results also suggested the effect of air pollution on mortality may be modified by socio-economic status (as measured by NZDep96) (Scoggins et al. 2004).

Prior to these studies epidemiological research on air pollution in New Zealand has focussed on particulate pollution in Christchurch. A previous New Zealand study (Hales et al. 2000b) that analysed the mortality effect of PM_{10} indicated that an increased total and respiratory mortality could indeed be measured. This study was designed to investigate the relationship between the daily number of deaths, weather and ambient air pollution. This involved using daily data for the city of Christchurch (population 300,000) from June 1988 to December 1993. Poisson regression models were used controlled for season using a parametric method. The results showed that above the third quartile (20.5 degrees C) of maximum temperature, an increase of 1 degree C was associated with a 1% (95% CI: 0.4 to 2.1%) increase in all-causes mortality and a 3% (0.1 to 6.0%) increase in respiratory mortality. An increase in PM_{10} of 10 μg m^{-3} was associated (after a lag of one day) with a 1% (0.5 to 2.2%) increase in all-causes mortality and a 4% (1.5 to 5.9%) increase in respiratory mortality. No evidence was found of interaction between the effects of temperature and particulate air pollution. The overall conclusion was that unusually high temperatures (or conversely unusually low temperatures) and particulate air pollution are independently associated with increased daily mortality in Christchurch. The fact that these results are consistent with those of similar studies in other countries strengthens the argument that the associations are likely to be causal.

A further study undertook an analysis of mortality among census areas in Christchurch (Hales et al. 2000a). The number of deaths following days with high particulate air pollution (defined as 24 hour average PM_{10} > 50 μg m^{-3}) was compared with deaths on matched unpolluted days (defined as PM_{10} < 50 μg m^{-3}). The possible role of population age structure, relative deprivation (estimated using NZDep96) and local exposure to outdoor air pollution from household fires (estimated using a chimney density index) was explored. There was a statistically significant association between mortality and air pollution. Substantial variation in pollution-related mortality among census area units was found. Relative deprivation (but not the proportion of elderly people or the proportion of households using solid fuel heating: “chimney density”) was found to be a statistically significant predictor of mortality patterns. There was also a positive association between solid fuel heating proportion and relative deprivation. These findings suggest that relative deprivation may increase vulnerability to the effects of particulate air pollution on daily mortality, independently of the effects of age and local variation in exposure.

A Christchurch study of hospital admissions for cardio-respiratory disease showed an increase of daily admissions the day after high air pollution days (McGowan et al. 2002). Daily data was analysed for the period June 1988 to December 1998 using a time-series approach that controlled for weather variables. For all age groups combined there was a 3.37 % (95% CI 2.34-4.40) increase in respiratory admissions (2 day lag), and 1.26 % (95%CI 0.31-2.21) increase in cardiac hospitalisations (no lag), for each interquartile rise in PM_{10} (value 14.8 μg m^{-3}).

A few other studies of health effects of air pollution in New Zealand have been published. Dawson et al (1983) studied the relationship between hospital attendance for acute asthma attacks and air pollution levels in Christchurch during the winter of 1981 and found a negative correlation. No explanation for this unexpected result was found, but the relatively small study size would have limited the statistical power of the study. Another study of asthma in Christchurch children (Wilkie et al. 1995) focussed on potential air pollution during the summer of 1993 around a fertilizer plant. No increase of asthma was found compared to a control group of children from the whole of Christchurch. The pollution situation was quite different from the winter smoke of major concern. The only other study is a panel study of 40 subjects with chronic obstructive pulmonary disease (Harre et al. 1997), in which their reported prevalence of night time chest symptoms increased during the day after a 24-hour period when the PM_{10}

levels increased by 35 μg m^{-3} or more. Again, the small study size makes it difficult to draw definite conclusions.

5.3.2 Health impact and risk assessments

The first risk assessment, based on daily dose-response relationships and current air pollution levels in Christchurch concluded that each year the days of high air pollution (due to all sources) are associated with 29 extra deaths and 40 extra hospital admissions (Foster 1996). In addition, it was estimated that air pollution is associated with 82,000 days of 'restricted activity' per year, such as absence from school or work due to respiratory symptoms, per year (Canterbury Regional Council 1997). These calculations were revised in 1999 following a more detailed study and an adoption of the 'no threshold' criterion to 40-70 deaths, around 75-100 hospitalisations per year, and 300,000 to 600,000 restricted activity days (Wilton 1999). A slightly more realistic assumption (and the one adopted in this study) might be to use the threshold approach, but with a lower value than the one used in the first (1996) study, but higher than used in the second (1999) study. A figure that is representative of the 'background' PM_{10} – judged for Christchurch as 3.8 μg m^{-3} – would be appropriate (see discussion in the Appendix at A2.2.3). This gives the result of 220,000 to 350,000 restricted activity days per year for greater Christchurch.

This approach is not unrealistic given the overall emphasis of this study to formulate mitigation policies. There is little that can be done about these 'background' levels, as they are almost entirely due to sources beyond human control (e.g. sea spray).

The method used for overall assessment was similar to that used by the British Columbia Ministry of Environment, Lands and Parks to calculate the health impact of particulate air pollution in the province (BCMELP 1995). For each 10 μg m^{-3} "increment" of 24-hour particulate air pollution above 20 μg m^{-3} a certain percentage increase of mortality or morbidity is assumed to occur. For instance, in Christchurch a 1% increase of total daily mortality was assumed in the first assessment to occur for each "increment". These calculations have been widely debated in Christchurch and some critics believe that the lack of local data supporting this risk assessment puts in question the regional air quality management policy.

It should be pointed out that 29 (or 40-70) extra deaths may seem small, as it is only 1% of all deaths in Christchurch during a year. However, these deaths are related to conditions during the 30 worst polluted days. Thus, 29 deaths are about 10% of the deaths during those days. In addition, not all deaths are truly preventable. People still die of 'old age' and many of the deaths during the worst polluted days have nothing to do with air pollution. The 29 extra deaths may therefore be a much larger proportion of the 'preventable' deaths during these days.

Another risk assessment of the health effects of air pollution has been produced for the Land Transport Pricing Study of the Ministry of Transport (Ministry of Transport 1996). The aim was to estimate the cost of health damage due to air pollution and other environmental impacts from motor vehicles on roads. Based on a review of a number of epidemiological studies it was concluded that lifetime exposure to 10 μg m^{-3} particle air pollution would increase total mortality by 1.6% and that lifetime exposure to 1 μg m^{-3} benzene would increase cancer mortality by 4 per million. The estimates were eventually expressed as the estimated cost in dollars per kilometre of road and the cost of particulate air pollution health damage were about 20 times greater than the cost of benzene health damage. These calculations are likely to be very approximate, but they indicate the importance of particle air pollution when indicators are established to monitor health effects of air pollution.

5.3.3 Earlier health impact assessments

A national risk assessment, commissioned by the Ministry of Transport, estimated that the number of people above 30 years of age who experience premature mortality in New Zealand due to exposure to emissions of PM_{10} particulates from vehicles (Fisher et al. 2002) (Table 5-2). Results suggested that pre-mature mortality due to vehicle related air pollution is similar to the accident road toll, with 404 people dying from road accidents (all ages) in 2002. However while the car crash death rate consistently peaks in the 15-24 year age group (Connor 2001), the average age of people dying attributable to air pollution is more likely to occur in the very young (less than 5 years old) or elderly (greater than 65 years old) (WHO 2000). This is due to the general frailty of people at the earlier and later stages of life, where they become more susceptible to a number to illnesses and health effects (Hales et al. 2000?).

	Due to Total PM_{10}	Due to Vehicle related PM_{10}
Auckland	436 (264-619)	253 (153-359)
Wellington	79 (48-112)	56 (34-80)
Christchurch	182 (110-259)	41 (25-58)
Dunedin	48 (29-69)	6 (3-8)
Rest of North Island**	133 (81-189)	21 (13-30)
Rest of South Island**	80 (48-114)	19 (12-27)
All of New Zealand**	970 (586-1376)	399 (241-566)

*Threshold PM_{10} for mortality effect 7.5 µg m^{-3}
**Places with more than 5,000 people

Table 5-2. 2002 estimated attributable number of deaths (95%CI) (over 30 years old) per annum due to PM_{10} emissions.

Fisher et al (2002) calculated mortality effects using the number of people exposed at different annual PM_{10} levels following the methodology used by Kunzli et al (2000). A hockey stick (where 'hockey stick' refers to the shape of the curve – reminiscent of a hockey stick lying down, with a long flat are and a steep rise near one end) dose-response relationship was applied above the annual average PM_{10} threshold (which was assumed to be 7.5 µg m^{-3}), and a linear increase of mortality was assumed at 4.3% above the background mortality rate for New Zealand for each 10 µg m^{-3} annual average increase of PM_{10}. Fisher et al (2002) and Kunzli et al (2000) calculated the number of deaths based on exposure-response relationship from two long-term studies in the US (Dockery et al. 1993; Pope III et al. 1995b). This was a preliminary study that was subject to many assumptions and uncertainties. The estimates were based on the best available information at the time and should be considered the first attempt in New Zealand to quantify health effects due to vehicle-related air pollution. Overall results suggested that air quality is a significant public health problem for New Zealanders – even though it is not a problem on the same scale as in many other parts of the world.

The Wellington Regional Council recently replicated the MoT health impact assessment (Fisher et al. 2002) method for the Wellington region, taking into account the following differences from the NIWA study: variations in the monitored ambient PM_{10} levels, increased geographical area of interest leading to increased population size, lower base mortality, and changes to the contribution of vehicles to ambient PM_{10} levels (O'Reilly 2003). The revised annual number of pre-mature deaths from vehicle related air pollution was 48 (95% CI: 24-66), which was similar to the results of the MoT health impact assessment, 56 deaths (Fisher et al. 2002) (Table 5-2).

These previous studies were focused on vehicle emissions effects for specific reasons at the time. There have also been New Zealand studies on effects associated with other sources, particularly PM_{10} in Christchurch. The most comprehensive of these (Hales et al 2000b) has independently confirmed that the dose-response relationships for pre-mature mortality in Christchurch closely match those found in overseas studies.

Similarly, and independent study undertaken for NO_2 exposure in Auckland (Scoggins et al, 2004) indicated that the overseas dose-response relationships were similar to those found in Auckland.

There have been no other specific New Zealand studies on health effects of air pollution due to other sources, for instance industrial sources, or for specific effects in other locations.

Finally, during the process of developing and setting the new air quality standards, throughout 2003 and 2004, the Ministry for the Environment conducted a number of studies and assessments of the health effects of air pollution. These studies were not formally published, but were used in the consultation process and are available at the Ministry's web site www.mfe.govt.nz.

5.4 Summary

A preliminary health impact assessment of the effects of air pollution in New Zealand was carried out for the MoT. However there has been no detailed quantitative assessment of the range of health effects including mortality, morbidity, and restricted activity days. With the exception of the work done for the Land Transport Pricing Strategy, New Zealand studies of the economic consequences (including the health impacts of air pollution) are lacking. Linking exposure, health and economic end-points back to sources define the nature of the "air pollution problem" for New Zealand.

6 Confounding Factors in Epidemiological Studies

6.1 What is a confounding factor?

The type of health effects that have been associated with air pollution exposure are generally not unique, but can also be caused by a variety of other factors, such as heat and cold, tobacco smoking, allergens, and occupational exposures. In addition, indoor air pollution not related to the outdoor air quality can cause such effects. It is also well established that the health effects of concern are associated with age, sex, ethnic group and socio-economic status, the latter possible a proxy for a combination of exposure to the other factors (e.g. tobacco smoking and occupational exposures). Confounding occurs when a factor is associated with the health effects and at the same time associated with the air pollution studied. But confounding does not occur just because the factor is a potential risk factor for the health effect being studied. It can be an effect modifier though, meaning that the health effect of air pollution is more (or less) severe because of the other exposure.

For instance, if the spatial distribution of air pollution in Christchurch is correlated with smoking habits, so that high air pollution areas also have high levels of smoking, then smoking will be a confounder for any health outcome that is caused by smoking. However, if the smoking habits are totally unrelated to the air pollution distribution confounding does not occur, but smoking can be an effect modifier so that smokers have greater health effects than non-smokers.

The issue of confounding is primarily of importance in epidemiological studies where a quantitative assignment of the causal role of different exposure variables is made. If confounding is ignored, misleading conclusions about causes and effects may be drawn. This would only occur if assessments of the impact of interventions are made and the interventions are influencing not only the air pollution exposure but also a confounding factor. For instance, an intervention that aims at reducing wood smoke exposure by making wood burning more expensive could have the effect that low income families reduce their heating and end up living in colder houses. The lowered indoor temperature may increase respiratory diseases, while the lower wood smoke concentrations would decrease respiratory diseases. Thus, the net effect of an intervention of this type could be no change in morbidity even though air pollution is reduced. This would be confounding in the implementation of an intervention.

6.2 Confounding factors

6.2.1 Climate factors

It is well known that days with low minimum temperatures and days with high maximum temperatures are associated with an increase of mortality (Howden-Chapman & Carroll 2003). This relationship has also been studied in this project, as there is an association between air pollution levels and temperature in Christchurch.

The non-linear relationship of daily temperature with daily mortality/morbidity is studied using two hockey stick approaches one for low temperatures and another for high temperatures. Daily maximum temperatures have been used to study the effects of high temperatures on mortality whereas daily minimum temperatures have been used to study the effects of low temperatures on mortality. In this approach, it is assumed that a positive linear relationship between mortality and maximum temperature starts after a certain maximum temperature (threshold maximum temperature) and there is no effect of temperature on mortality below this threshold temperature. Similarly, there is a negative linear relationship between mortality and minimum temperature below the threshold minimum temperature and no relationship above this minimum temperature.

Another factor of interest is the changing climate in Christchurch. The maximum temperatures at St Albans have gone up during the last 40 years, and the rate of change has been greatest during the last 10 years (leading to warming of about 2ºC) (Zhang, Kjellstrom, and Shrestha, to be published).

As the climate variables, temperature, relative humidity and wind speed, are routinely recorded in the air pollution monitoring by Environment Canterbury, and can be taken into account in the epidemiological analysis.

6.2.2 Influenza

Influenza and other respiratory infection epidemics generally occur during the cold months, the same months when both the number of deaths and air pollution levels in Christchurch are higher than the rest of the year. During a respiratory disease epidemic, the number of deaths is likely to increase. This increase in deaths may not be the effect of air pollution, but the increased mortality may be wrongly linked to air pollution if the occurrence of influenza is not fully accounted for in the analysis. Some epidemiological studies of the short-term association between air pollution and mortality have controlled for influenza epidemics. Some studies tested the delayed influenza effects for up to 15 days.

Table 6-1 shows the monthly distribution of influenza deaths (ICD 9 code: 487) by year in Christchurch. Most influenza deaths occur during cold months. A higher number of influenza deaths are likely to indicate an epidemic in those years. However, the number of deaths is very small and there was no death recorded as influenza after 1996 due to a change in the coding methodology. This makes it difficult to identify epidemics based on the number of reported influenza deaths. (Note that the total monthly mortality in Christchurch is approximately 200, so the numbers in Table 6-1 are small in comparison).

Instead of defining an epidemic based on the number of reported influenza deaths, the weekly number of reported consultations and laboratory "isolations" for influenza cases can be used for this purpose. These numbers for the years 1994 to 1999 were obtained from Dr Michael Baker (ESR, unpublished data). This data is collected only from May to September each year.

Table 6-2 shows summary statistics of weekly number of consultations and laboratory confirmations (isolations) for influenza 1994 to 1999. Figure 6.1 shows the distribution of weekly number of GP consultation cases. The distribution was skewed towards low values.

Based on the distribution, it was assumed that any week with the number of consultation greater than the median i.e. 12 was an influenza epidemic week. (Based on the distribution in Figure 6-1 a threshold of 20 could be chosen for an epidemic). All days in epidemic weeks were considered to be having epidemics and all days in non-epidemic weeks were considered not to have epidemics. A dummy variable that represents epidemic (more than 12 GP consultations/week) (0) or non-epidemic (1) was created to use in the model to analyse the association between daily air pollution and daily mortality.

Modelling with the method used for the daily time series mortality analysis (covered elsewhere in this report) showed that addition of the influenza epidemic variable did not change the association between air pollution and mortality in the time-series analysis. Consequently, influenza epidemics are not a confounding influence in this study.

Year	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1988	1	1	3	6		1				
1989					3	1	2			1
1990				2	2	5			1	
1991				6	1	1		1		
1992			1	1						
1993				1	1	1		1		
1994			1		2	4	2			
1995				1	2	1		1		
1996		1		3	8	1				

Table 6-1. Monthly distribution of influenza deaths (ICD 9 code: 487) by year in Christchurch. (In 1996 the coding methodology changed and this analysis is now not possible).

	Number of weeks	Mean	Median	Min	10th Percentile	90th Percentile	Max
Consultation	917	18.52	12	0	2	46	98
Isolation	917	5.73	5	0	0	13	26

Table 6-2. Summary statistics of the weekly number of general consultations and laboratory confirmations (isolations) for influenza cases for May to September from 1994 to 1999.

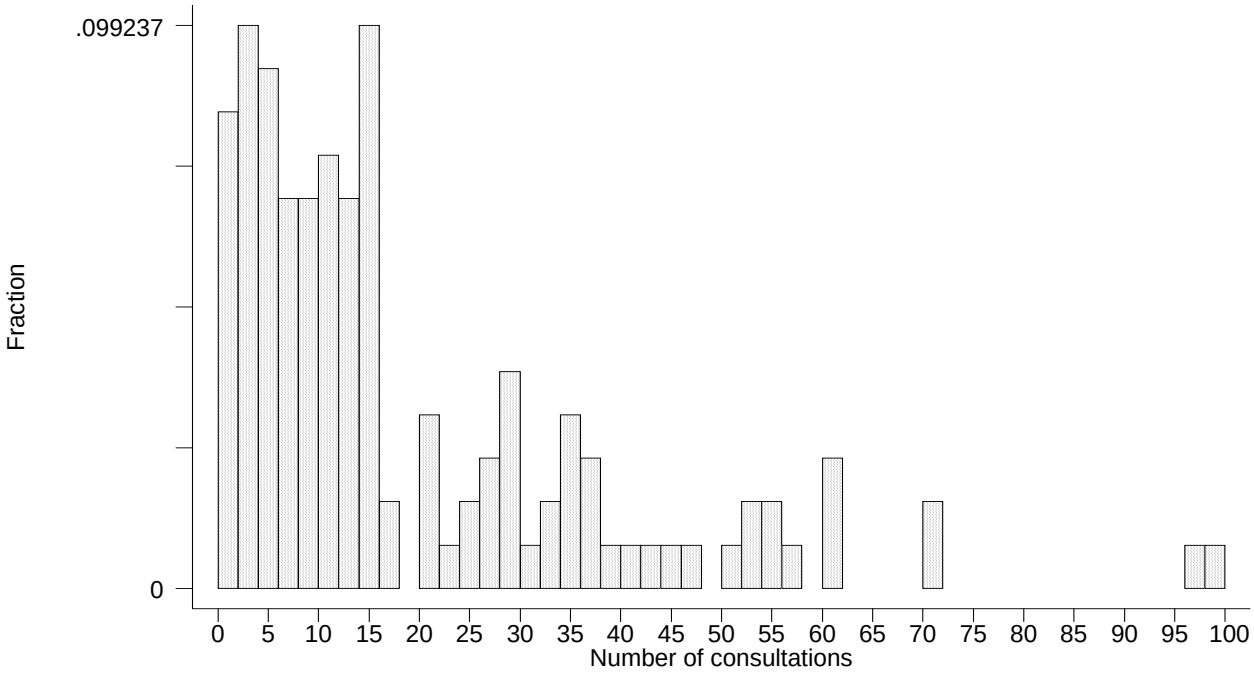

Figure 6-1. Frequency distribution of weekly number of GP consultations for influenza 1994-1999 (May to September).

6.3 Adjustments

Age, sex and ethnic group

It is well known that age, sex and ethnic group are important determinants of mortality and morbidity. Confounding may occur if different time periods or different geographic areas studied have different population composition.

In the time-series analysis the mortality for different causes of death was calculated for the whole population and for selected subsets by age and sex. The key analysis is comparing day-to-day changes in air pollution and health variables. During such short time periods of comparison it is not likely that any significant change in population structure takes place. It is usually assumed that the population size and age/sex/ethnic composition does not change much during the period of study (in this case, 12 years), but a long-term mortality change variable in the multiple regression analysis adjusts for any background trends due to population changes.

Thus, the time-series analysis took age and sex into account. Because of the relatively small (7%) proportion of Maori and other ethnic groups (i.e. Pacific 2% and Asian 5%) in the Christchurch population it was not possible to do a meaningful analysis of the daily number of deaths for these groups.

In the GIS-based ecologic study in Auckland, age (0-14, 15-64, 65+ years), sex and ethnicity (European, NZ Maori, Pacific Peoples, Asian + Other) were all accounted for.

6.4 Summary

Confounding in epidemiological studies and health risk assessments means that an extraneous variable influences the results so that effects of air pollution are exaggerated or underestimated. This result can only occur when the air pollution exposure is associated with another exposure or condition in the target population (e.g. tobacco smoking, socio-economic deprivation) and when this other factor is also associated with the health effect of concern.

For the air pollution situation in Christchurch population characteristics that may be confounding factors would be age, sex, ethnic group and socio-economic status. Other exposures that may be confounding factors include climate factors, tobacco smoking, and occupational exposures. All of these factors have been taken into account, when relevant, in the analysis of data in this project.

7 Epidemiology and Dose-Response Relationships

The epidemiological approach used in this study follows closely a methodology that has been used elsewhere and has become a de-facto standard for such studies (Kunzli et al 2000). The details are summarised here. As the Künzli (2000) study has an important role in this analysis, it is described it in some detail below. The dose-response relationship used in the Künzli study was derived from two long-term studies in the USA, and these will also be described in some detail in order to understand the basis for the calculations.

7.1 The Künzli study

This study was published in September 2000 in the well-respected medical science journal The Lancet. It presented the results of an international collaborative study, which was funded by the National Science Foundation (USA), the Austrian Federal Ministry of Environment, Youth and Family Affairs (and other Austrian government agencies), the Agency for Environment and Energy Management, France, and the Federal Department of Environment, Transport, Energy and Communications, Switzerland.

A detailed methodological report was prepared for the 3rd WHO Ministerial Conference of Environment and Health, London, 1999 (Künzli et al. 1999). The study analysed the public health impact of outdoor and traffic-related air pollution in three countries: Austria, France and Switzerland. The conclusion was that outdoor air pollution caused 6% of total mortality, and half of this was related to air pollution from motorized traffic. The estimated number of traffic air pollution related deaths was about twice the number of traffic crash deaths, so the term "hidden road toll" seems apt for this type of air pollution public health impact.

The annual average outdoor air pollution exposure levels in 1 km^2 grid squares covering each of the countries was estimated using GIS methodology and a combination of air monitoring and emission inventory data. As the air concentrations of these pollutants are often closely correlated it was decided to use PM_{10} as a proxy for total air pollution and a "useful indicator of several sources of outdoor air pollution such as fossil-fuel combustion". The contribution by traffic to the estimated air pollution levels was calculated from Swiss emission-dispersion models. The traffic share of total outdoor PM_{10} varied according to the PM_{10} level. For concentrations of $PM_{10} < 15$ $\mu g\ m^{-3}$ the share was 28%, increasing to 58% for $PM_{10} > 40$ $\mu g\ m^{-3}$. (This relationship may be the opposite in Christchurch, as high PM_{10} is due to wood and coal smoke from home fires). Population data for the 1 km^2 grid squares was used to estimate the exposed population at different annual PM_{10} levels. The PM_{10} data by grid square were categorized into groups in 5 $\mu g\ m^{-3}$ steps (0-5, >5-10, >10-15, etc. $\mu g\ m^{-3}$ annual mean). Thus, for each country the population living in areas with different PM_{10} exposure categories could be calculated.

In order to calculate the air pollution associated deaths, dose-response relationships reported from two long-term exposure epidemiology studies form the USA (Dockery et al. 1993; Pope et al. 1995) were used. It was assumed to have a "hockey-stick" shape, with no additional mortality risk below 10 $\mu g\ m^{-3}$ (none below 7.5 $\mu g\ m^{-3}$ for the average of the category 5-10 $\mu g\ m^{-3}$). Above this threshold the mortality increased according to a linear function with a slope of 4.3% increase of mortality for each 10 $\mu g\ m^{-3}$ increase of annual PM_{10}. This estimate was derived from the two studies described below. By accumulating the annual air pollution attributable deaths for all of the grid squares, the total air pollution impact and the impact from traffic air pollution was calculated. Uncertainties of the estimates were quantified and 95% confidence intervals reported, but the exact method for quantification of confidence intervals was not reported.

7.2 Studies providing the dose-response relationship

A large number of studies have demonstrated associations between daily PM_{10} levels and daily mortality, including one study in Christchurch (Hales et al. 2000b).

These studies have been reviewed in a number of reports (e.g. NRC, 1998) and will not be dealt with in detail here.

The so-called "six city study" examining non-external mortality (Dockery et al. 1993) was a prospective cohort study of 8111 white adults (aged 25 - 74 years) in six US cities (Portage, Topeka, Watertown, Harriman, St Louis and Steubenville) where the long-term average "inhalable particle" levels (PM_{10}) were 18, 24, 26, 31, 33 and 47 $\mu g\ m^{-3}$. Individual data on age, sex, weight, height, education level, complete smoking history, occupational history and medical history was available. Spirometric test results were also available. Much of the air pollution would be due to traffic. Power stations, industry and home heating could be other sources, but these were not identified. It was assumed that the people from each city were exposed to the average level of PM_{10} in that city.

It was found that when all the potentially confounding variables in the study were taken into account, there was a significant increase of total mortality and cardio-pulmonary mortality when comparing the worst polluted and the least polluted city. The rate ratios (RR) were 1.26 (95% confidence interval = 1.08-1.47) and 1.37 (1.11 – 1.68) respectively. Lung cancer also had a tendency for increase (RR = 1.37), but it was not statistically significant. The combined mortality of all other causes of death was not increased (RR = 1.01). The difference of the mean long-term PM_{10} levels was about 60 $\mu g\ m^{-3}$. As the total mortality was increased 26% due to this difference the conclusion would be that the mortality increase per 10 $\mu g\ m^{-3}$ PM_{10} is 4.3%. This is the logic behind the RR of 1.043 used in the Künzli report. The Dockery et al. (1993) report also provides data on "fine particle" exposures ($PM_{2.5}$). The difference between the worst and least polluted cities was 30 $\mu g\ m^{-3}$ for $PM_{2.5}$. Thus, a dose-response relationship for $PM_{2.5}$ would use an RR function of 8.3% increase per 10 $\mu g\ m^{-3}$ instead of the 4.3% for PM_{10}.

The Dockery et al. (1993) study also quantified the combined effect of smoking and air pollution. Non-smokers in the study had a non-significant RR of 1.19 for total mortality comparing the worst and least polluted cities, while current and former smokers had an RR of 1.33 (1.03 – 1.70). The other long-term study quoted is the study by Pope et al. (1995). This study covered 552,138 volunteer adults included in a cancer prevention study. Individual information was collected about age, sex, weight, height, race (4.1% were black), smoking history, alcohol use, occupational exposures, and other characteristics. The participants' mortality was monitored over 7 years. Their air pollution exposure was based on the address at the time of entry into the study, and data on air pollution monitoring in the cities where they lived. $PM_{2.5}$ one-year average levels varied between 9 and 33.5 $\mu g\ m^{-3}$ (difference = 24.5 $\mu g\ m^{-3}$).

Multiple regression analysis showed an increase of all-cause mortality associated with air pollution level. The RR for the fine particle range of 24.5 $\mu g\ m^{-3}$ was 1.17 (1.09-1.26), equivalent to a 6.9% increase of mortality per 10 $\mu g\ m^{-3}$ $PM_{2.5}$. When converted to a risk function for PM_{10}, this would mean 3.5% increase of mortality per 10 $\mu g\ m^{-3}$, very similar to the findings in the Dockery et al. (1993) study. Again, cardio-pulmonary mortality was especially increased. However, the study by Pope et al. (1995) did not find a higher mortality increase among smokers.

A large number of time-series studies of the association of daily mortality and daily average PM_{10} levels, including studies in Sydney (Morgan et al. 1998) and Christchurch (Hales et al. 2000b). These studies generally find an increase of total mortality of about 1% per 10 $\mu g\ m^{-3}$ PM_{10}. Based on these studies the WHO Air Quality Guidelines (WHO, 2000) recommended a linear dose-response function of 0.74% (0.62-0.86%) increase of mortality per 10 $\mu g\ m^{-3}$ PM_{10} for calculations of daily mortality increases. WHO also recommends a linear dose-response function for long-term exposure and long-term increases of mortality at 10% (3-18%) per 10 $\mu g\ m^{-3}$ PM_{10} (WHO, 2000). This guideline dose-response coefficient is similar to (but higher than) the results found in the two studies quoted above (Dockery et al. 1993; Pope et al. 1995). The use here of the Künzli study coefficient of 4.3% increase/PM_{10} can be considered "conservative" as the use of the WHO guideline coefficient of 10% would more than double the estimates calculated. It should also be pointed out that the WHO guideline values assume no threshold for the start of the mortality effect, which can significantly increase the estimated number of deaths from air pollution when using these guideline values.

8 The Christchurch Pilot Study

8.1 Study objectives

This component of the study, described in the following sections, focuses on the urban area of Christchurch. It is formulated as a pilot to the main study, in order to highlight issues and assumptions before the assessment is made on the whole of New Zealand.

8.2 Study methods

The study has five interconnected components: air quality, meteorology and emissions data analysis, air pollution exposure assessment, health impact assessment, economic impact assessment, and preventative policy assessment.

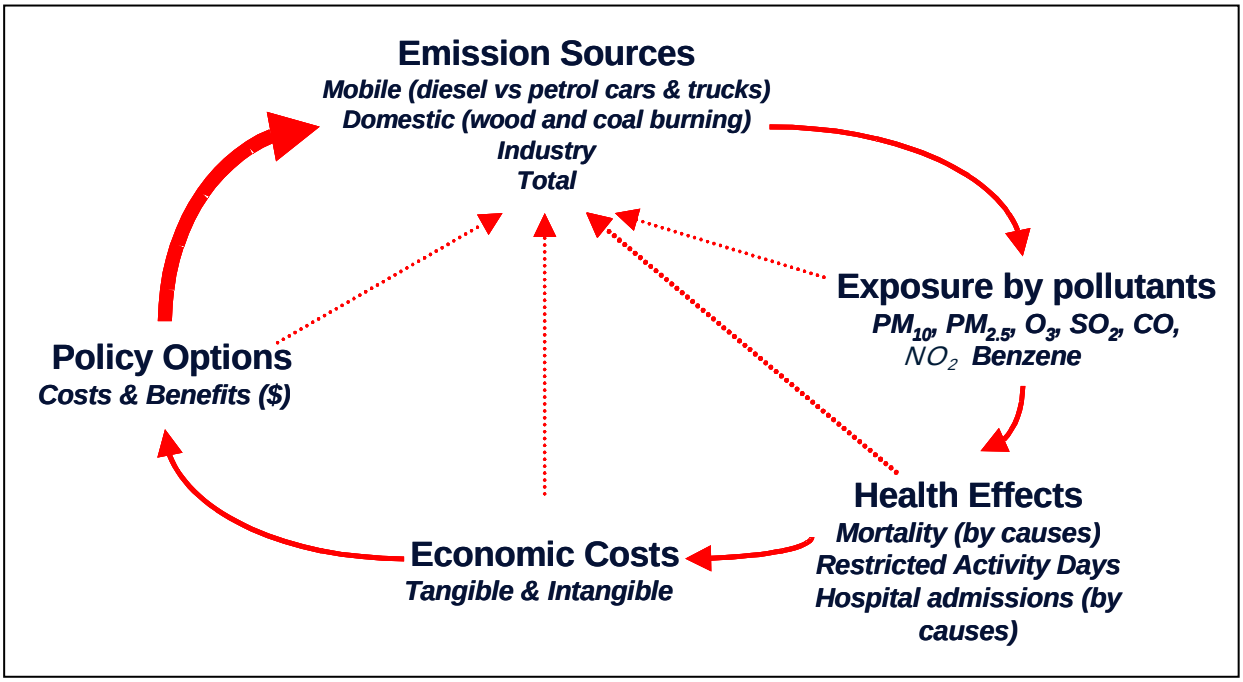

Figure 8-1. HAPiNZ methodology overview.

The air quality, meteorology and emissions data analysis involves collecting a range of data sources in a GIS framework including census derived variables such as chimney density, vehicle ownership, journey to work matrix, emissions inventory, vehicle fleet emissions, pollution monitoring, and urban airshed modelling. The data is used to assess pollution exposure at a Census Area Unit (CAU) level and builds upon the methods used in the MOT risk assessment (Fisher et al. 2002) to produce more accurate measures of exposure. The source proportion for each pollutant is estimated at the CAU level. The seasonal variation in emissions, exposures and effects also needs to be considered.

The epidemiological analysis and health effects assessment combines exposure-response relationships for New Zealand conditions, and evidence from overseas. These and estimates of the size of the population exposed to different levels of air pollution will be used to quantify the number affected in each population group, and each defined geographic area (for each health-end point). The health end-points examined include pre-mature ("hastened") deaths, hospital admissions, and restricted activity days. Age, sex, and ethnic group distribution of effects by diagnostic category (ICD code) will be assessed, to the extent possible.

To validate exposure and to confirm that biological and clinical effects occur, a panel study of bio-marker effects, symptoms, lung function and restricted activity days will be carried out in Christchurch.

The economic impact assessment attempts to calculate the tangible (such as health care costs associated with pollution-related illness) and intangible costs (such as lost years of life and lost quality of life due to illness) using a "willingness-to-pay" approach.

The policy assessment includes identifying policy options for domestic, vehicle and industrial emissions, projecting emissions for year 2005, 2010 and 2020 from status quo (year 2001), and assessing the impact of policy options on emissions within a cost/benefit framework.

8.3 Study area

Christchurch, a coastal city on the Canterbury Plains of New Zealand, has the third largest population in New Zealand and the largest of the South Island. Approximately 316,000 (2001) persons live in Christchurch city TLA. Eighty-two percent of the population is of white European descent and seven percent of the population is of NZ Maori decent. The local tangata whenua is Ngai Tahu iwi. Figure 8-2 shows the age distribution of the Christchurch population in 2001. Nineteen percent is under 15 years old, 67% of the population is between 15 and 64 years old, and 14% is greater than or equal to 65 years old.

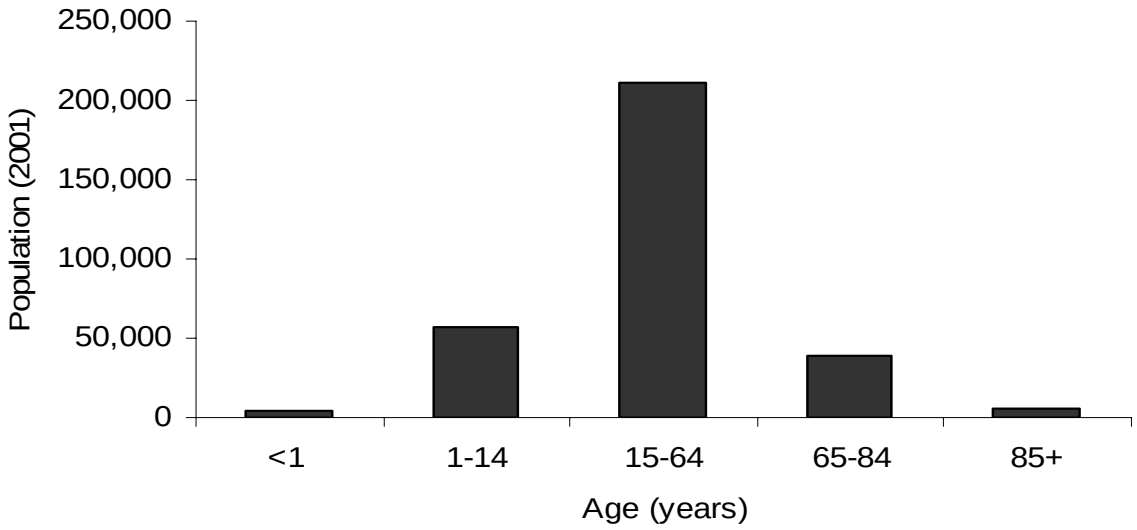

Figure 8-2. Age distribution of the Christchurch study population.

The city is surrounded by the Pacific Ocean in the east, the Canterbury plains extending as far as the Southern Alps in the north and west, and the Port Hills in the south. Apart from the hillside suburbs in the south, the Christchurch urban area is totally flat, gently sloping downwards from the airport in the west to the Pacific Ocean in the east. Figure 8-3 shows a map of Christchurch with the boundaries of CAUs. These CAUs are the basis for the resident location of routinely collected health data.

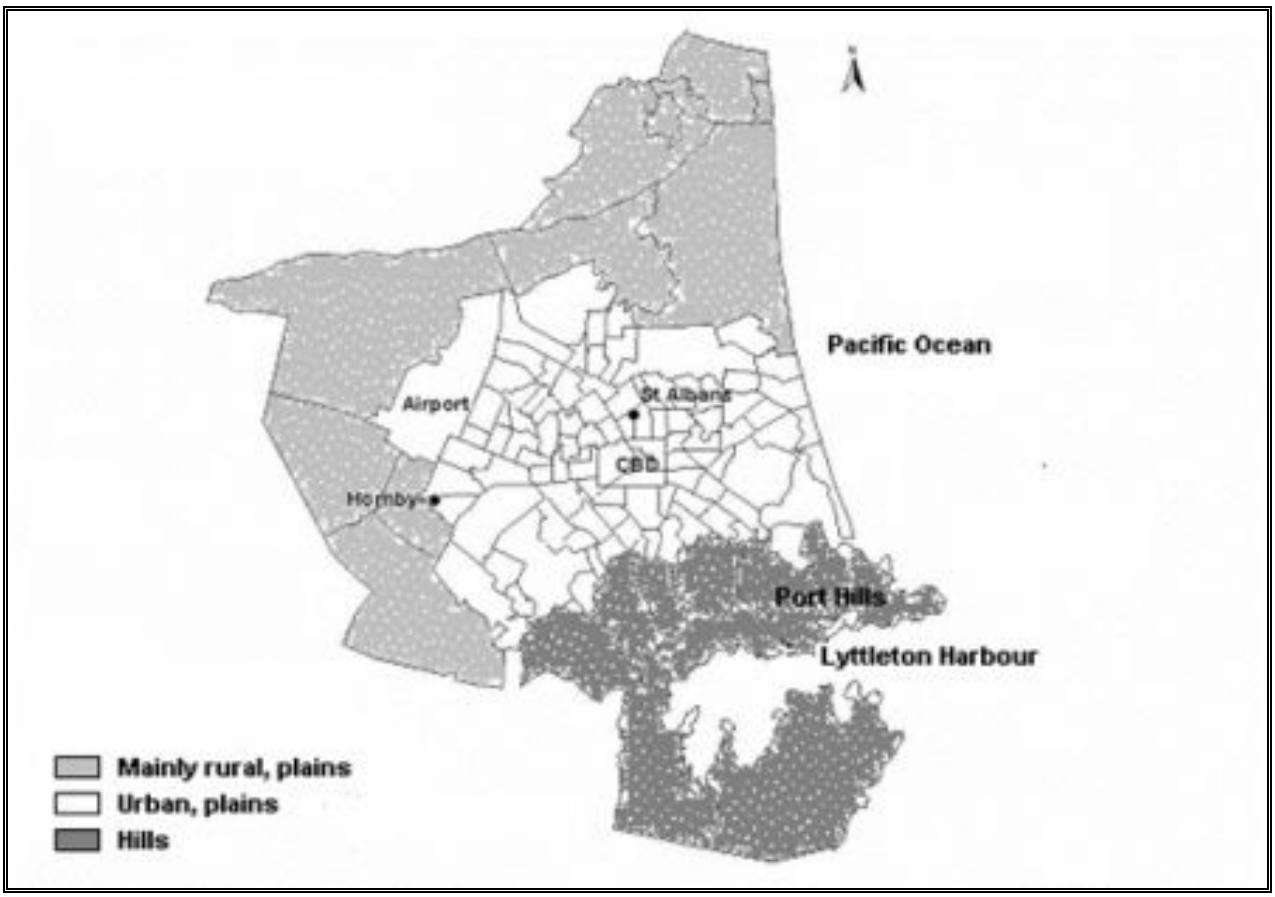

Figure 8-3. Map of Christchurch with boundaries of CAUs and monitoring sites.

The presence of the Southern Alps, the Port Hills, the Banks Peninsula and the Pacific Ocean in the vicinity of Christchurch largely influence its climate. The Southern Alps forms a massive barrier to the westerly air streams. As a result, Christchurch has relatively low rainfall. Christchurch has two main climatic zones the Port Hills and the Canterbury Plains. The Port Hills have higher humidity and greater seasonal variability than the Canterbury Plains, which are drier and have a more evenly distributed rainfall during the year. For several decades, air pollution from the burning of coal and wood in the winter has been a major concern in Christchurch. The meteorology and topography is conductive to lengthy calm periods and frequent low level inversions that severely limit the vertical and horizontal (due to low wind speed) dispersion of pollutants. Burning of wood and coal is a common method of home heating that creates significant particulate air pollution problems. Recently in addition to wood smoke, air pollution from the other sources, particularly vehicle emissions, is being assessed in greater detail.

9 Air Pollution Exposure Assessment

9.1 Objectives

The aim of this part of the project is to accurately estimate annual concentrations of air pollution contaminants, initially PM_{10}, for Christchurch. The previous work of Fisher et al (2002) used a crude measure of exposure and used two values to represent the whole of the city. The new estimations of exposure will produce values with greater spatial resolution down to census area units (CAUs).

9.2 What is exposure?

The concept of pollution exposure is fundamental to studies looking at impacts of the environment and health. Consequently methods of exposure assessment have been discussed in the literature (e.g. Hawkins et al (1992); Rappaport and Smith (1991); Ashmore (1995); Colls and Micallef (1997); Lebret (1995)). A variety of definitions have been given, but one that explains it well is that given by Nurminen et al (2000) who define exposure as "both to the concentration of an agent at the boundary between an individual and the environment and to the duration of contact between the two" (Nurminen et al. 2000).

Research into relationships between pollution and health has adopted two contrasting approaches, each based on different premises. Time-series studies of the acute effects of pollution have usually assessed pollution exposure based on measured data from one or, at best, a few monitoring stations within a city (Pope III et al. 1991; Schwartz 1991; Schwartz 1993). This assumes limited spatial variations in pollution exposures, and that single, daily average estimates can be applied to the whole study population. In contrast, geographical studies, which have generally focused on chronic effects of exposure, have typically used measures such as distance from a source such as a road (Brunekreef et al. 1997; Edwards et al. 1994; Livingstone et al. 1996; Murakami et al. 1990; Nitta et al. 1993), local traffic density (Weiland et al. 1994; Wjst et al. 1993), or modelled concentrations (Briggs et al. 1997; Briggs et al. 2000; Elliott & Briggs 1998; Oosterlee et al. 1996; Pershagen et al. 1995) as indicators of exposure to pollution. Such studies assume that spatial variations in pollution occur across areas and related to distance from source, and the indicators used subsequently represent these.

9.3 Exposure analysis

A large amount of work has been carried out refining the air pollution exposure estimates for the population of Christchurch. The results obtained are very detailed, identifying exposures to PM_{10} and a number of other pollutants on high resolution time and space scales.

Particulate measurements using Tapered Elemental Oscillating Microbalance (TEOM) have been obtained using the data from the Environment Canterbury monitoring network, complemented by a number of special measurement campaigns by the University of Canterbury and others, and extended using advanced airshed modelling techniques. It appears that the TEOM data show systematically lower values than the mini-vol equipment used by the University of Canterbury. Recent modifications of the measurement methods in Christchurch have confirmed that the previous TEOM values were too low. The model outputs are correlated with the outputs of the mini-vol results. In addition, some problems have been experienced in applying the complex and new methodologies involving advanced computer modelling, translating the very fine scale spatial data into GIS formats, and conducting detailed exposure assessments at the CAU level. A number of revisions have occurred in the results, and these may continue as more is learnt about the process through the research.

This comprised a substantial portion of the research work, and the full details are covered in Appendix 2.

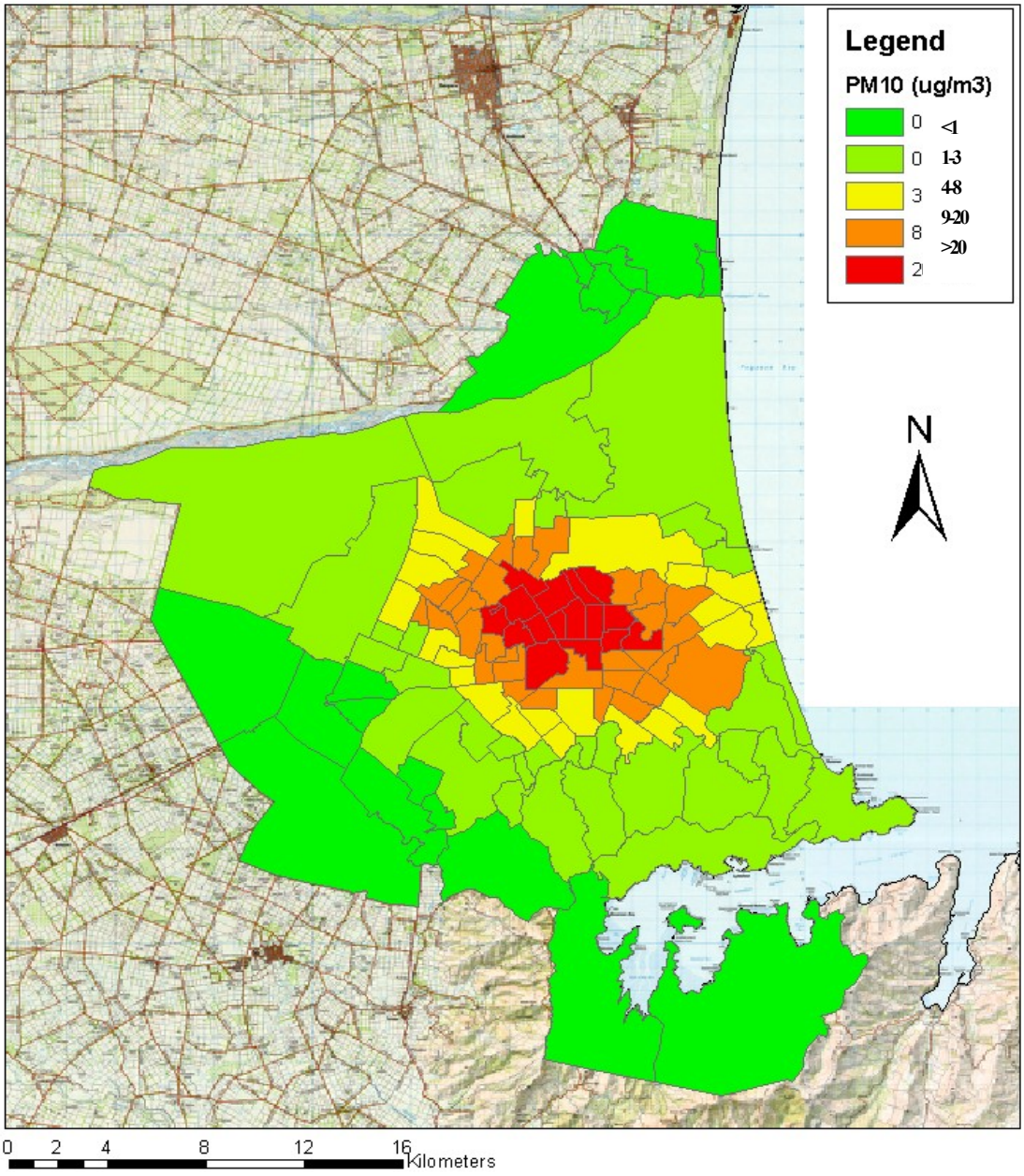

Figure 9-1. Annual PM_{10} exposure map. (Darkest areas in the central city are those above the 20 µg m^{-3} national annual guideline.)

10 Christchurch Health Impact Assessment

10.1 Scope

The new data from this study and previously published reports were used to assess exposure-response relationships (with confidence intervals) for Christchurch conditions. These and estimates of the size of the population exposed to different levels of air pollution are used here to quantify the number of cases for various health end-points (mortality, hospital admissions, and restricted activity days) by source contribution, including wood smoke, vehicles, or industry.

Since the time of this part of the study – conducted in mid to late 2004 – there have been a number of new studies published. In addition, the research conducted here has provided new results that can be used in future health effects assessments. These studies have further quantified air pollution effects on children, and the elderly, and have included effects such as asthma. These have not been covered in this pilot study health effects analysis, but will be examined further in the full study due for release early in 2006. These results are covered in Appendix 3.

Caveat

As noted earlier, the calculation of health effects have been conducted using known and justifiable dose response associations – mainly those for PM_{10}, and also benzene. This is the approach taken by many other studies. However the actual effects are due to a complex mix of pollutants, of which PM_{10} is a good indicator. Other pollutants by themselves – such as CO, NO_2 and SO_2, are known to have defined health effects, but at this stage of the research it has not been possible to explicitly identify the proportions for Christchurch. The extent of what these contributions might be is calculated, and presented in Appendix 3. It is not appropriate to 'add' these effects to the PM_{10} effect, neither is it appropriate to 'subtract' them – and thus try to identify the effect strictly due to PM_{10}. The sate of knowledge of the epidemiology is not yet advanced enough to attempt this.

Whilst this may be a less than desirable outcome for the policy analysis, it is difficult to overcome. The key point remains – air pollution is associated with health effects. Those associated with ambient annual PM_{10} concentrations are well established, but the independent associations with other pollutants are not known at this time.

10.2 Methodology

10.2.1 Health impacts of PM_{10} on mortality

All previous research indicates that the primary air pollutant that affects health is particulate matter (PM_{10} or $PM_{2.5}$). The basic calculation method used by Kunzli et al (2000), for Austria, France and Switzerland and in the preliminary analysis for New Zealand (Fisher et al. 2002) will be used for PM_{10} exposure estimates in this report.

The formula (Kunzli, et al. 1999) for calculation of mortality is:-

$$P_o = \frac{P_e}{1 + [(RR - 1)(E - B) / 10]}$$

Where:

- P_o = baseline mortality per 1000 people, after deducting the air pollution effect (this will depend on the other variables).
- P_e = the crude mortality rate per 1000 people for the selected age group.
- E = PM_{10} exposure level in the area of interest.
- B = threshold PM_{10} exposure level for mortality effect (3.8 $\mu g\ m^{-3}$ was used for Christchurch).

RR = the epidemiologically derived relative risk for a 10 μg m^{-3} increment of PM_{10}, assuming a liner dose-response relationship above the threshold (B).

The increased mortality is then calculated:

$$D_{10} = P_o * (RR - 1)$$

Where:

D_{10} = the number of additional deaths per 1000 people onto the baseline mortality for a 10 μg m^{-3} increase in PM_{10}.

P_o = baseline mortality per 1000 people, after deducting the air pollution effect (this will depend on the other variables).

RR = the epidemiologically derived relative risk for a 10 μg m^{-3} increment of PM_{10}, assuming a liner dose-response relationship above the threshold (B).

And then:

$$N_c = D_{10} * P_c * (X_c - B) / 10.$$

Where:

N_c = the number of deaths due to PM_{10}.

D_{10} = the number of additional deaths per 1000 people onto the baseline mortality for a 10 μg m^{-3} increase in PM_{10}.

P_c = the population ('000s)

X_c = the PM_{10} exposure level.

RR = the epidemiologically derived relative risk for a 10 μg m^{-3} increment of PM_{10}, assuming a liner dose-response relationship above the threshold (B).

The above method is applied to each CAU in Christchurch using the specific population size and air pollution data by source. The exposure data used was modelled concentrations, validated by monitoring as described in an earlier chapter. The number of deaths due to PM_{10} by source in whole Christchurch is then calculated by adding the number of deaths in each CAUs. The climate variables and the crude mortality rates are assumed to be the same in each CAU in this pilot study analysis.

Air quality data and measurements

These have been produced by the model described earlier, and the basic calculation unit is annual average PM_{10} concentration by CAU.

Mortality data

The mortality rate of 2.8 per 1000 in the population of age over 30 years for non-external deaths is used as the base annual mortality rate (P_o). This is based on the 1996 Census and mortality data. All CAUs in Christchurch are assumed to have the same baseline annual mortality rate.

Dose-response relationships

The dose-response relationship used is the same as Kunzli et al (2000); that is an increase in mortality of 4.3% per 10 μg m^{-3} increase in PM_{10}

.

10.2.2 Health impacts of PM_{10} on morbidity

Non-mortality effects of PM_{10} are covered under two categories - chronic obstructive pulmonary diseases (COPD), and respiratory admissions to hospital.

Chronic obstructive pulmonary disease

The incidence of PM_{10} pollution also affects a number of chronic obstructive pulmonary diseases and allied conditions. These include: -

- Bronchitis

- Chronic bronchitis
- Emphysema
- Bronchiectasis
- Extrinsic allegoric alveolitis
- Chronic airways obstruction

These are identified by codes ICD9: 490-496 and occur in Christchurch at the rate of 2.06 per 1000 per year population for all ages, and 3.19 per 1000 per year for over 30 year olds.

The calculation process was similar to the one used for mortality. The annual dose-response rate used is as per Kunzli et al (1999), for 30 years and over, of 9.8% (95% CI: 0.9-19.4) per 10 μg m^{-3} of PM_{10}.

Daily respiratory admissions

The dose-response relationships from the Christchurch daily hospital admission and daily particulate air pollution study were used (McGowan et al. 2002) to estimate the daily number of hospital admissions due to PM_{10}. These were then added up to produce annual numbers of admissions. The percentage increases in daily hospital admission associated with 10 μg m^{-3} increase in daily PM_{10} from the McGowan study are 0.85% (95% CI: 0.21 – 1.49) for total cardiac admission and 2.28% (95% CI: 1.58 – 2.97) for total respiratory admission for all age groups. The dose-response functions were applied to daily hospital admissions and daily monitored PM_{10} levels (in all of Christchurch) for 1998. The annual number of hospital admissions attributed to PM_{10} for 1998 was calculated as follows:

$$N_r = (DR/100) \times H \times \text{Sum}[(Ei-B)]$$

Where:

N_r = Annual number of respiratory hospital admissions attributed to PM_{10}

DR is the percentage increase in daily hospital admission per 1 μg m^{-3} increase in PM_{10},

H is the baseline average number of hospital admissions per day,

Ei is the daily PM_{10} level and B is the threshold PM_{10} level for its effects on hospital admission.

'Sum' is the summation of each of the 365 days into an annual number.

Asthma (ISD9: 493) has not been covered here, but will be analysed further in the final report.

10.2.3 Cancer risk from benzene

Cancer risks are assessed using inhalation unit risk (IUR). Inhalation unit risks are defined as the individual lifetime excess risk due to a chronic lifetime exposure to one unit of pollutant concentration. That is, the probability that a person contracts leukaemia when exposed to a 1 μg m^{-3} benzene for the average lifetime. In New Zealand the average life expectancy is currently estimated at 79 years. (Statistics New Zealand).

The estimated number of cancer (leukaemia) cases due to benzene exposure in Christchurch per annum is:

$$N_c = Pop \times (R * E) / L$$

Where:

Nc = Cancer cases per year
Pop is the number of people exposed
R is the lifetime risk estimate associated with a 1 μg m^{-3} increase in benzene. Risk estimate was taken to be 6×10^{-6}, with a 95% confidence interval = 4.4×10^{-6} – 7.5×10^{-6}.
E is the estimated long term exposure
L is the average life expectancy (years)

It is valid to add any benzene effects to the PM_{10} effects as the health effects are different and are associated with different illnesses.

Air quality data and measurements

There has been limited benzene monitoring in Christchurch. Health impact assessment was carried out for a high (maximum annual average – 6.6 μg m^{-3}) and a low (minimum annual average – 3.0 μg m^{-3}) exposure scenario (Table 10-1).

Pollutant	Time period	Monitoring site
Benzene annual	1998 (low exposure scenario)	Hoon Hay
	1997 (high exposure scenario)	Stanmore Rd/Haast St

Table 10-1, Benzene exposure alternatives for health impact assessment.

10.3 Restricted activity days

Restricted activity days were calculated based on the same method as used by Wilton (2001b). In this pilot study a dose-response relationship was applied to the estimated annual average $PM_{2.5}$ levels in the whole city with an estimated range of 12 μg m^{-3} to 16 μg m^{-3}. A threshold for effect at 3.8 μg m^{-3} was applied.

10.4 Results

The results are presented for PM_{10} and benzene, as a total for the exposure assessed, and also broken source by source, for the three major sources – domestic, vehicle and industrial.

The source apportionment is based on either explicit modelling results (for PM_{10}), or from the ratio of sources in the emissions inventory. These are summarised in Table 10-2.

10.4.1 Source apportionment of other pollutants

The source apportionment is based on either explicit modelling results (for PM_{10}), or from the ratio of sources in the emissions inventory. Relative contributions to pollutant emissions in Christchurch were obtained from the latest inventory of emissions to air (Scott & Gunatilaka 2004) (Table 10-3).

Pollutant	Domestic home heating	Industry	Motor vehicles
PM_{10}	70%	14%	16%
Benzene	34%	2%	64%

Table 10-2. Inventory source apportionment of PM_{10} and benzene in Christchurch (Adapted from Scott & Gunatilaka 2004). The PM_{10} ratios are confirmed in the modelling.

These figures are a crude proportion of effects associated with each source, although in the analysis following more precise figures were used from the outputs of the exposure modelling that take into account the distribution of sources and population across the city.

10.4.2 Mortality – annual PM_{10}

The Kunzli derived calculation was carried out on the annual exposure modelling results for Christchurch as described above. The effects were calculated separately for PM_{10} due to each of the three major sources, with a zero threshold approach. The results are shown in Table 10-3 based on the

CAU-specific modelling of exposures for domestic (mainly wood smoke), industrial and vehicle sources. Background, or natural, sources and effects are excluded.

Number of deaths in the population of over 30 years old			
Domestic emissions	**Industrial emissions**	**Vehicle emissions**	**Total**
124	18	16	158
Compared with 2002 MoT study		*41*	*182*

Table 10-3. Effects of PM_{10} on mortality by source.

These results are slightly lower than those estimated in the MoT study. The main difference is the more detailed exposure modelling used here producing a gradient of exposures from higher in the centre of the city to lower at the city fringes. This indicates slightly lower exposures on average than those estimated for the MoT study, which used a cruder estimate for exposure across the city.

The effects of vehicle emissions can be further categorised by estimating the effects due to petrol, diesel and related sources. The vehicle fleet is categorised in the Environment Canterbury emissions inventory into diesel vs. petrol, and also estimates the fraction from heavy duty vs. light duty, and from non-tailpipe sources (Wilton 2001a). Most of the PM_{10} is sourced from diesel vehicles, with at least 70% estimated to come from heavy duty vehicles (those over 3.5 tonnes tare weight). There is also a fraction estimated to come from non-tailpipe sources, such as break and tyre wear and road dust. This has been estimated at 15% of the total for Christchurch, but is the subject of further research, and may be a higher fraction.

10.4.3 Mortality – benzene

Benzene has a reasonably well defined cancer risk, on a zero threshold basis. The benzene exposure for the region has not been explicitly modelled, nor has its relationship to other pollutants been examined. The two scenarios presented represent exposure results from monitoring conducted in two parts of the city (high and low measurements). The average exposure for the city is likely to lie between these two values, but for the sake of further analysis the more conservative high value is adopted.

Health Effects	**Exposure**	**Health effects by source**			
		Vehicles	**Domestic**	**Industry**	**Total**
Cancer (leukaemia)	0 (low exposure scenario)[1]	0.46 (0.34 – 0.58)	0.24 (0.18 – 0.29)	0.014 (0.011 – 0.018)	0.72 (0.53 – 0.90)
Cancer (leukaemia)	0 (high exposure scenario)[1]	1.02 (0.74 – 1.28)	0.54 (0.39 – 0.67)	0.032 (0.023 – 0.040)	1.59 (1.16 – 1.98)

1 See explanation in the text

Table 10-4. Number of deaths by source associated with benzene exposure.

The estimated number of cases (leukaemia) cases associated with benzene exposure in Christchurch per year is 0.72 (low exposure scenario) and 1.59 (high exposure scenario), with about 95% probability the number of cases is in the range of 0.5 to 2 per year.

10.4.4 Morbidity

Chronic obstructive pulmonary disease

Table 10-5 shows the number of cases of chronic obstructive pulmonary disease (COPD) associated with PM_{10} in Christchurch for 1999.

Health effect	Annual number attributed to PM_{10} (95% CI)
COPD (Bronchitis, chronic bronchitis, emphysema, bronchiectasis, extrinsic allegoric alveolitis, chronic airways obstruction	52.1 (5.5 – 91.2)

Table 10-5. Number of bronchitis cases associated with PM_{10} exposure.

Daily respiratory admissions

Table 10-6 shows the number of acute hospital admissions associated with PM_{10} in Christchurch for 1998. These figures are being updated with a dose-response relationship from an updated hospital admission analysis.

Health effect	Annual number attributed to PM_{10} for all age groups (95% CI)
Acute cardiac admissions which include ischaemic heart disease, heart failure and dysrhythmia	53 (13 – 92)
Acute respiratory admissions which include acute respiratory infections, pneumonia and influenza, chronic airways disease, asthma and other diseases of upper tract	194 (135 – 253)

Table 10-6. Number of hospital admissions associated with PM_{10} exposure.

Restricted activity days

The estimated range of restricted activity days in Christchurch annually is between 220,000 and 350,000. This estimate is lower than the previous estimate by Wilton (2001b) of between 300,000 to 600,000 restricted activity days in Christchurch. Previous estimates were based on annual average concentrations of PM_{10} measured from 1990 to 2000, which ranged from 19 µg m^3 to 36 µg m^3 and assumed a 60% $PM_{2.5}/PM_{10}$ ratio and no threshold concentration, whilst here the $PM_{2.5}$ is estimated at 12-16 µg m^3 and applied a threshold at 3.8 µg m^3.

10.5 Summary

The health impact assessments detailed above are based on exposures derived from modelling and validated against monitoring, and published dose-response relationships. The results are used in the economic impact and policy analyses. Further refinements of the methodology will be tested in the next stage of the project. Table 10-7 shows a summary of the health effects used in further analyses in this pilot study.

Effect	Domestic	Industrial	Vehicle	Total
Mortality	124	18	16	158
Bronchitis	36	9	7	52
Cardiac admissions	37	9	7	53
Respiratory admissions	136	32	27	194
Cancer	1.1	0.3	0.2	1.6
Restricted activity days	200,000	45,000	40,000	285,000

Table 10-7. Annual effects of air pollution in Christchurch by source (The confidence intervals are omitted for clarity – see text and other tables).

11 Economic Impact Assessment

11.1 Scope

The data on health outcomes, including dose-response relationships, reported earlier is used to calculate:

- Quantitative measures of 'health outcomes' or 'health gains'. For instance numbers of cases of various pollution-related conditions, deaths, life-years lost, QALYs or DALYs lost, restricted activity days, etc.
- Financial values of these outcomes – for example calculating a likely range of NZ values for a QALY and applying these to the quantitative measures of QALY effects.

An example of the kind of material produced is given in Table 11.1 from a USA report. It shows expected benefits from a tightening of the Clean Air Act. The first two numerical columns show the expected number of avoided cases, and their estimated monetised value. (The third column – the estimated money value per case – is not in the original source, but derived from it.)

Some caution is of course needed in using material from another country, written in a different context (the source is a review of methods of estimating public health benefits of air pollution regulations). The table does suggest, however, that by far the most important economic benefits are likely to be from preventing pre-mature mortality, and preventing the development of chronic bronchitis, with additional contributions from the prevention of days of work-loss and restricted activity. Health-care cost savings are relatively small. That does not mean, though, that they should not be estimated here.

It is useful here to set out a complete list of the expected benefits of lessened air pollution.

'Tangible' or 'Resource Cost Saving' benefits

- 'Averted' health-care costs
  - Hospital costs
  - Primary health-care, including medicines, inhalers, etc.
- Averted lost production
  - Days off work, school or usual activities limited
  - Production lost due to premature death
  - Work productivity impaired

'Intangible' gains, or gains in well-being

- Health gains
  - Prolongation of life. Life years gained.
  - Improved quality of life. Morbidity reduced. QALYs gained.
- Other 'well-being' gains, say from more pleasant environment.

***NB** All dollar figures quoted in the remainder of this sections are in NZ$ unless otherwise indicated – converted using foreign exchanges rates applicable as of mid-2004.*

11.2 Methodology

Earlier chapters have provided a range of estimates of the health effects for Christchurch of current pollution levels. The objective of this chapter is to put a present-day dollar value on these effects. Some of these values are direct resource costs – for instance the costs of hospital health-care. Others are estimates of society's 'willingness-to-pay' (WTP) to avoid undesirable health outcomes, such as premature death (see for example Bicknell (2001), for further discussion).

The effects tabulated elsewhere in this report are mainly the consequence of PM_{10} pollution. Some effects are also given, however, for NO_2, SO_2, CO, and benzene pollution. The effects of the first three

of these probably overlap to a considerable extent with those of PM_{10} and are not considered further in this summary report (but will be in the full report). The effects of benzene are separate, but are small compared with those of PM_{10}, so also are not further considered here.

PM-related outcomes	Avoided Cases	Monetised values (US$M 1990)	Derived value per case (US$1990)
Mortality	3,300-15,600	1,800-75,100	500,000-4,800,000
Chronic bronchitis	45,000-75,000	11,700-19,400	260,000
Hospital admissions			
All respiratory (all ages)	3,600-5,700	42-72	2,000
CHF	1,200-2,100	30-35	20,000
Ischemic heart disease	1,200-2,400	30-49	20,000
Acute bronchitis	2,000-20,000	1	50
Lower respiratory symptoms	179,000-299,000	2 to 4	2.5
Upper respiratory symptoms	36,000-60,000	1	20
Work-loss days	1,900,000-3,148,000	156-261	82.5
Minor restricted-activity days	15,697,000–26,28,000	600-1,000	38.25

Table 11-1. Annual benefits of proposed (Ozone and) $PM_{2.5}$ standards for partial attainment scenario in 2010 (Source: Adapted from National Research Council, 2002).

The main health effects given earlier are:

- Premature deaths as a consequence of pollution. This measure can be refined further to take account of the reduction in life expectancy from this mortality. That is, the number of life-years lost, so taking into account the stage of life at which premature deaths occur. For the calculations here it has been assumed that each death represents the loss of 5 years of life. This is an average for the whole Christchurch population. Because the average number of years of life lost is uncertain, the calculated results also show results for the loss of 3.75 years or 2.5 years of life. Subsequent research as part of this programme may provide evidence of the loss of life expectancy in exposed populations under New Zealand conditions, in which case the calculations will be updated to reflect these findings.
- Acute hospital admissions for either a respiratory condition, or for cardio-vascular disease.

There are also some effects that have not been tabulated earlier, including

- Chronic bronchitis (incorporates, COPD, or emphysema). This has been calculated from Christchurch admissions data at 52 cases per year due to PM_{10}, and community WTP to avoid a case at 10 percent of the WTP to avoid a premature death.
- Primary health-care costs – GP consultations, pharmaceuticals, etc - of illness associated with pollution is assumed to be 25 percent of secondary-care costs for acute hospital admissions.
- Restricted activity days. Wilton (2001b) increased earlier estimates for Christchurch to a range of 300,000 to 600,000 days annually. That is, an average 1 to 2 days per inhabitant. They cover days spent in bed, days missed from work and school and days when activities are partially restricted due to illness. For the calculations here, 10 percent have been assumed to be 'Work Loss' days or the equivalent in seriousness, and the remaining 90 percent 'Minor Restriction' days. This is a subjective estimate that will need to be refined in the final analysis.

11.2.1 Data sources

The results already reported previously give a good part of what is required, in terms of extra mortality for given increases in PM_{10} in winter and summer separately. Ideally more fine detail would be useful, in terms of age-group of deaths, and also ethnic and socio-economic status.

The principal other data sources are:

- Health-care costs. For hospital care principally from DRG (Diagnostic Related Conditions) values used by the Ministry of Health in the allocation of funding. For primary care and pharmaceuticals, from primary health surveys, and Pharmac expenditure data.
- Value of averted 'work loss' and 'restricted activity days'. From 'time cost' data as for example used in Transfund manuals for transport sector project analyses.
- QALY (Quality-adjusted Life-Year) data – from DALY (Disability-adjusted Life Year) estimates by disease, calculated by the Ministry of Health.
- Value of Life. Land Transport Safety Authority (LTSA) estimates of the 'Value of Statistical Life'. These are further discussed below.
- Value of a Life-Year, or QALY, or DALY. Derived from the 'Value of Life', as discussed below.

11.2.2 Calculation methods

The intention is to use 'life year saved' outcome measures (including QALYs and DALYs) as the principal benefit criteria. This is in preference to using 'lives saved', which was the approach taken in the European tri-nations study. Given the availability of such measures, the next step is valuing them in dollar terms. The discussion below looks at the issues involved.

Valuing Lives, Life-Years, and QALYs.

Much work has been done in the transport sector in recent decades on 'willingness-to-pay' measures of the 'value of statistical life'. In New Zealand this research has been led by the Land Transport Safety Authority (LTSA – now Land Transport New Zealand). Miller and Guria (1991) on the basis of their research recommended adopting a value of $2 million per life saved in 1990 prices. This was in place of the 'human capital' based value of $235,000 used previously. Their recommendation was accepted and has been used since in land transport investment appraisals. By June 2002 the value had been adjusted upwards for inflation to an amount of the order of $2.6 million.

Further work was carried out in the late 1990s (Guria et. al. 1999). This resulted in a recommendation of an increase in the VoSL to $4 million. This recommendation has not yet been accepted.

Both the 1991 and 1999 LTSA estimates were largely based on sample surveys of what New Zealanders were 'willing to pay' to buy road safety for their families. People were asked what they were prepared to pay in dollars or time saved for small reductions in fatality risks. In the 1998/99 study there were questions on living in different localities, trading off different living costs and different rates of exposure to death by road accident.

Life-Years and QALYs

If one assumes the Value of Statistical Life (VoSL) for an individual should be a function only of that individual's remaining life expectancy L (not all accept that assumption), then one could put an age-related $ value on a Life-Year ($LY) or QALY of

$$\$LY = \$QALY = VoSL/L$$

This, however does not take account of time preference. Assuming a real discount rate of d% per annum, one obtains from standard compound interest formulae

$$\$LY = \$QALY = VoSL/d * [1-(1+d)-L]$$

Suppose it is assumed that respondents to the LTSA's surveys were thinking about the risk to an 'average' person of 'average' age. For the present population (2001 estimates), the average age ranges

from approximately 34.7 years (males) to 36.5 years (females). Remaining life expectancies at those ages are 43.0 and 45.4 years respectively.

Using these numbers and the formula above gives the following values. Note that the higher the discount rate, the higher the estimated Life-Year values. This ensures that the sum of the discounted life-years equals the LTSA's VoSL.

Discount Rate	Males	Females	Total
0%	60,445	57,239	58,777
3%	108,415	106,041	107,198
5%	148,183	146,281	147,202
7%	192,493	191,099	191,770
10%	264,389	263,617	263,984

Table 11-2. Value per life-year ($).

Note that these values are higher than those generally implied in health sector allocation decisions – for instance a new drug costing more than, say, $30,000 per QALY gained, would probably have difficulty in making it on to the list of subsidised pharmaceuticals. In part this may reflect budget constraints.

Is NZ value of VoSL reasonable by international standards?

An article in The Economist of December 4, 1993 tabulated for a number of countries the value of life given in research publications of that date relating to the economic cost of transport accidents. A selection of these were as follows–

USA	$3,940,000	Sweden	$1,870,000
New Zealand	$1,750,000	Britain	$1,670,000

The estimates for these four countries were all derived on a 'willingness-to-pay' basis, replacing the inferior 'human-capital' approach used in earlier times.

The richer an economy, the more its citizens would be expected to be prepared to pay for life-saving safety measures. Allowing for this, the numbers in the Economist article suggest that the New Zealand value (converted from the NZ$2 million LTSA estimate in 1991) is very much in line with estimates for countries of similar income per head.

There has of course been much subsequent work on this issue. Some examples of values for specific countries or in specific settings are –

Australia

An Access Economics report for the Australian Society for Medical Research calculated the Australian 'value of a statistical life' as in the range A$4.5M to A$10.5M, with the midpoint at A$7.5M. However this is a straight exchange rate conversion from USA values, and is undoubtedly too high. For a real discount rate of 3%, the mid-value of a life-year in Australia was A$150,000.

A more conservative value in the Australian literature is contained in an Applied Economics report to the Department of Health and Ageing. A value of life of A$1 million is assumed, and at a discount rate of 5% the resulting value for a healthy life year is A$60,000. The A$1 million is derived from Bureau of Transport Economics estimates, partly based on a 'human capital' approach, which would now be rejected as inappropriate, and too low, by virtually anyone working in this field.

OECD and Europe

More sophisticated estimates, and more directly related to this study, are available elsewhere.

The OECD report on Chemical Risk Management Decision Making (OECD 2002) has a particularly wide-ranging and useful discussion of the issues involved (pages 148+).

Of particular relevance is the tri-nation European health and economic evaluation reported in Sommer et al (1999 and Kunzli et al. (2000). In that study, the preferred 'value of preventing a statistical fatality' is given a 'basic value of EU1.4 million'. A NZ$ is currently about 0.55 of a Euro, so this equates to about NZ$2.5 million. The authors describe their estimate as 'rather conservative' (page 30).

Rather than using this value to derive a 'life year value', the authors instead adjust for most casualties of air pollution being at older ages by using a lower value of EU0.9 million (NZ$ 1.64 million).

Elsewhere

Krupnick et al (2000) find for Canada a range from C$1.2 to C$3.8 million, falling to C$0.6 million for those aged 70+.

Summary

Overall, the work by New Zealand's LTSA on valuing 'statistical life' appears to be of as high a standard as most of the best international work, and of considerably higher standard than that in Australia, for one example.

Might the VoSL be expected to vary with kind of death?

There is considerable discussion on this topic. The OECD report is particularly cogent. (Table 3.5. p 165). Two factors mentioned are that people may dread a lingering death more than a sudden death; and that involuntary risks over which they have no control, or risks which are someone else's responsibility and novel risks, are regarded as worse than others. But pesticide residue control was found in one study to be preferred to automobile exhaust controls.

On such arguments, Sommer et al (1999) consider that "most likely the aversion against air pollution related risk is considerably higher than the aversion against the risk of fatal road accidents."

Equity issues

Should greater weight be given to interventions which reduce differences in life expectancies of different population groups? And should years of life or QALYs gained vary with age?

Survey work, including some conducted by the Wellington School and Medicine and Health Sciences, suggests a wide range of opinion in answer to the first. However, the inclusion of an ethical judgement in the calculation of years of life saved – the assumption of equal life expectancies – in itself has some impact on equity. On the second, the community does appear to prefer saving 'youthful years'.

11.3 Discussion

Value of Statistical Life and Value of a Statistical Life Year

The Land Transport Safety Authority's (LTSA) estimate of the value of a 'Statistical Life' is based on good quality research, and appears generally comparable with estimates in other countries. The original April 1991 value of NZ$2 million has been adjusted for changes in average ordinary-time earnings to an estimated value for June 2004 of NZ$2.725 million. Assuming this is for a citizen of current average age of about 35 with life expectancy at that age of about a further 44 years, this gives the following range of VoSLYs, for discount rates ranging up to 10 percent per annum.

Derived values of 'statistical life-years' vary with discount rate. The 'preferred value' of the discount rate in health economics is of the order of 3 to 5 or 6 % per annum; but it is likely official decision-makers will require a 10% discount rate. Because reductions in air pollution probably most benefit the elderly, lower values of VoSL and of life-years should also be used in sensitivity testing, to reflect the lower value the community appears to put on 'older' years saved.

For the assumption of a loss of five years of life for each pollution-related death, these values are multiplied by the sum of (discounted) life-years 5 years into the future from the present day. The results are in the first column of Table 11-4. The second and third columns show the effect of varying the average years of life lost.

Note that the Sommer et al (1999) study used the VoSL rather than VoSLY concept, but reducing it by about 35 percent to allow for deaths from pollution occurring at older ages. The reductions in the table above from the New Zealand VoSL are more substantial.

The working value used for the major analyses later in this study assumes a zero harvest and a 6% discount rate - $750,000 per case. As noted in Table 11.4 above this value can range from a low of $156,000 (if loss of 2.5 years of life assumed and 0% discount rate used), to a high of $1,048,000 (if loss of 5 years of life assumed and 10% discount rate used).

This compares with a rate of $765,000 that is used in the European Union ExternE study (which assumed a 3% discount rate).

Discount Rate	Males	Females	Total
0%	$63,351	$59,990	$61,603
3%	$113,627	$111,140	$12,351
5%	$155,307	$153,313	$154,279
6%	$178,033	$176,309	$177,142
7%	$201,748	$200,286	$200,990
10%	$277,100	$276,290	$276,675

Table 11-2. Value per statistical life-year.

	years lost per early death		
Discount Rate	5	3.75	2.5
0%	$308,016	$232,295	$156,575
3%	$514,536	$388,243	$261,949
5%	$667,949	$504,176	$340,403
6%	$746,184	$563,329	$380,473
7%	$824,097	$622,260	$420,423
10%	$1,048,817	$792,377	$535,937

Table 11-3. Value of lost years of life.

A discount rate of 6% was selected, giving a cost per pre-mortality case of $746,184 ($750,000) assuming 5 years of life lost

Chronic obstructive pulmonary diseases

The value adopted here is 10% of the value attributed to mortality due to PM_{10} – or $75,000 per case. This reasonably consistent with the values adopted in the US (shown in Table 11.1). Not in absolute terms, since in the US costs are higher reflected in the value of $170,000 (US$260,000), but in the relative terms, where the value ranges from 52% - 6% of the value of a life, compared to the 10% adopted here.

Acute hospital admissions

The values used here are $2,700 for respiratory admissions, and $3,675 for cardio-vascular admissions. These are from NZHIS published 2000/01 average DRG (Diagnostic-related Group) 'prices', excluding GST, and adjusted for subsequent inflation to mid-2004. (This compares to slightly higher values of $3,000 and $4,000 used in Bicknell, 2001, and $6,880 used in ExternE.)

Cancer

A rate of $750,000 per case has been used, equivalent to the PM_{10} pre-mature mortality cost. This figure is likely to be lower than actual, and subject to future revision.

Minor hospital costs

In addition to the direct costs, an allowance has been made for some extra costs imposed on the hospitals due to non-admission activities associated with air pollution – such as emergency room visits for respiratory problem that are not severe enough to result in admission. These are estimated at $1,500 per day in winter months, and $250 per day in summer months – or $200,000 per year.

Restricted activity days

Valued at $160 for a 'Work Loss' day, and $40 for a 'Minor Restriction' day. Taking these at 90 and 10 percent respectively gives a weighted average of $150 per restricted activity day. There is a wide range of values used, with the European ExternE rate as high as $175 for European countries. In contrast, the rate used in Middle Eastern countries applying the ExternE methodology (e.g. Jordan) is lower at $44. These figures reflect the economic circumstances of the country.

Costs of doctors' visits that may arise are not considered.

11.4 Results

11.4.1 Health effects used

The current annual economic effects discussed above are calculated for the results shown in Table 11.5 (repeated here from the previous chapter).

Effect	Domestic	Industrial	Vehicle	Total
Mortality	124	18	16	158
Cancer	1.1	0.3	0.2	1.6
Chronic bronchitis	36	9	7	52
Admissions - cardiac	37	9	7	53
Admissions - respiratory	136	32	27	194
Restricted activity days	200,000	45,000	40,000	285,000

Table 11-5. Summary results of the health effects analysis (repeated from previous chapter).

11.4.2 Economic costs used

For the sake of conducting further analysis without carrying forward such a wide range of options, the following choices are recommended. (These can of course be re-visited if further information becomes available): -

* **Dose response:** The original Kunzli figures are most widely accepted for policy purposes and have been used here.
* **Discount rate:** A 6% discount rate has been selected. This is a reasonable compromise between a 'low' rate and the 'high' rate often used for public policy analysis (10%).
* **Years of life lost:** A value of 5 years is used.

From the above analysis and assumptions, the key costs in Table 11-6 were used.

Effect	Cost per case
Mortality	$750,000
Cancer	$750,000
Chronic bronchitis	$75,000
Admission - cardio-vascular	$3,675
Admission - respiratory	$2,700
Restricted activity day	$150

Table 11-6. Summary of costs of events used in the analysis.

Each of these has a range of values, as detailed in the discussions in the text and in earlier tables.

11.4.3 Analysis

The data from health effects (Table 11-5) and costs (Table 11-6) are combined to arrive at the summary of the costs of air pollution, by source, in Christchurch (Table 11-7).

Effect	Domestic	Industrial	Vehicle	Total
Mortality	$93.0M	$13.5M	$12.0M	$118.5M
Cancer	$0.8M	$0.2M	$0.2M	$1.2M
Chronic bronchitis	$2.7M	$0.7M	$0.6M	$4.0M
Admission - cardio-vascular	$0.1M	$0.05M	$0.05M	$0.2M
Admission - respiratory	$0.4M	$0.1M	$0.1M	$0.6M
Restricted activity days	$30.0M	$7.0M	$6.0M	$43.0M
Minor hospital costs	$0.15M	$0.03M	$0.02M	$0.2M
Total	**$127M**	**$22M**	**$19M**	**$168M**

Table 11-7. Summary valuation of health effects of PM_{10} pollution in Christchurch.

11.4.4 Source apportionment

In this analysis, the treatment of natural sources of PM_{10} has been problematic. As discussed elsewhere, the natural sources are assumed to have less of an effect as anthropogenic sources. This is accounted for informally by others (such as Kunzli) by using a threshold. The natural source contribution is not negligible in terms of total PM_{10} in the airshed, but it may be a relatively minor contributor to the health effects, although research results on this aspect of the problem are not available. Since one of the major purposes of this current study is to examine policy options for mitigating effects, the natural background PM_{10} is excluded. The calculation is based only on the explicit effects due to each of the main three source categories.

11.4.5 Pollutants other than PM_{10}

One of the objectives of this study was to explicitly identify the effects of different types of pollutants on public health. This may be important in assessing mitigation policies, especially if there is some sort of trade off involved. For instance if a policy can reduce CO emissions, but increases PM_{10} or NO_x emissions – is there a net benefit or cost? (This is not an unreasonable example, since a shift from petrol to diesel vehicles may result in a decrease on CO emitted per kilometre travelled, but an increase in PM_{10} emissions per kilometer. Another example might be a solid fuel heater that produces less smoke – e.g. fewer PM_{10} particles, may by doing so produce more $PM_{2.5}$ and NO_x but seem acceptably less smoky).

Under ideal circumstances this would be possible, but unfortunately the state of the epidemiological research is such that it is not yet possible. This situation is covered by assuming that the health responses across the range of pollutants are integrated. In other words the effects due to NO_2, SO_2 and CO are subsumed in the effects due to PM_{10} – which are much larger. This has been done to avoid 'double counting' of the effects. This is not a perfect arrangement, but is not too unrealistic since urban air pollution is a mix of the various pollutants and in most cases is dominated by PM_{10}. Therefore dose response studies based on PM_{10} (which the majority are) will in most cases satisfactorily account for effects due to other pollutants. Finally, the effects other than mortality, due to each of the source categories are calculated using the same ratio as the explicit effects calculated for mortality. This is a rather crude assumption, not fully supported by research results, but is carried out here as a way to apportion costs for the policy analysis.

11.5 Summary

These estimates will receive further work and refinement. They do show that the largest component of the 'health burden' is the loss of life-years as a result of premature mortality, followed probably by 'restricted activity days', and then 'chronic bronchitis'. The direct health-care costs are a relatively minor part of the total.

11.5.1 Analysis

A range of results has been presented, covering various choices in the following key parameters: -

A. Discount rate for assessing value of life lost (0% to 10%)
B. Years of life lost (5, 3.75, 2.5)

These result in a range of annual costs for pre-mature mortality in Christchurch from $59.3M to $118.5M (used here - $118.5M), a range of chronic bronchitis costs from $0.4M to $6.9M (used here- $4.0M), restricted activity day costs of $43.0M and another $2.0M in cancer and other hospital admissions.

The total costs of PM_{10} pollution in Christchurch are thus of the order of $168M per year, of which nearly two-thirds is due to home heating, and a fifth due to vehicle emissions. The study area comprises a population of 316,000, giving the cost per person of $532 per annum.

11.5.2 Discussion

The results obtained above will have important implications, and are used in the policy analysis in this study. Whilst it is not possible to independently validate these results, it can be instructive to compare them with results from other studies, if only to assess their consistency.

Two comparisons are made. The first is with a preliminary cost analysis conducted for assessing the effects of air pollution in Auckland (Fisher, 2002). This showed that the cost per person for the Auckland region is of the order of $353 per person per annum. This analysis was based on the earlier pre-mature mortality estimates (Fisher et al, 2001). This is not an unreasonable level of agreement, since although wintertime air pollution levels in Christchurch are much higher than in Auckland, and the annual average is slightly higher.

The second comparison is with a large economic assessment carried out for Canada (Canadian Royal Society, 2001). Although the methodology was not precisely the same as that used here, the final outcome was a cost per person per annum of $390 ($NZ). This is lower than the NZ estimates as the Canadian study did not include the costs of restricted activity days, which are estimated to be relatively high.

In summary, the consistency between these different results, arrived at independently, is remarkably good – the external health costs due to PM_{10} air pollution in Christchurch, for 2001, are of the order of $168M, or $532 per person. Given that some health costs have not been fully assessed (asthma, doctor's visits, effects on children, secondary social costs), it is likely that this is an underestimate.

12 Preventive Policy Assessment

12.1 Scope

The purpose of the preventive policy assessment for Christchurch is to examine some policy options for managing and reducing the costs and effects of air pollution. This includes examining the effect of different policy options in reducing air pollution, at different spatial distributions as well as the costs and benefits associated with the measures. This analysis is conducted to show methodologies, rather than produce defined policy results.

Caveats

This is a pilot study, and much of the analysis attempted in this chapter is exploratory. There are three major caveats that must be applied.

1. **The policy options chosen are by no means comprehensive, and since this analysis was undertaken a number of new options have become apparent, with the development of the air plan, the introduction of standards, and the vehicle emissions rules.**
2. **The process of associating a particular effect to a particular source is incomplete. This is discussed fully in an earlier chapter. This dilemma applies particularly to vehicle emissions. The primary indicator for air pollution health effects is PM_{10} – however, as discussed in Appendix 3, there are effects associated with other pollutants such as CO, NO_2 and SO_2. These are not explicitly identified as being separate from PM_{10} although it is clear that they are associated with part of the effect. For instance a complete removal of PM_{10} will not completely remove the health effect if the other pollutants are still present. This is a very difficult analysis with the current level of knowledge, and the current results should be taken as indicative only for developing policy responses.**
3. **The analysis is relevant to Christchurch only. The absolute and relative effects, especially the spilt between domestic, industrial and vehicle sources, will be different for other areas.**

12.2 National policy

The national policy context for the bulk of air quality management, and by implications the effects of poor air quality, has been previously determined by the Ministry for the Environment through its National Ambient Air Quality Guidelines (1994, and revised in 2002), recently supplemented by the National Environmental Standards.

The guidelines have been used as the quantitative assessment tool for policy setting, Regional Council Plans and Resource Management Act processes. Specific health risk assessments have been undertaken in many cases – either for particular discharges, or for detailing Council objectives (in particular Environment Canterbury has undertaken a number of studies – see below). These health risk assessments have been undertaken as extensions to national policy, since prior to 2004 no binding national policy on air pollution and health had been available, outside of the implicit provisions in the air quality guidelines and guidance from the Ministry for the Environment.

With the introduction of National Air Quality Standards in September 2005, the national policy setting has altered, with implications for health risk management. The quantitative values for acceptable ambient air quality do not change significantly, but there is substantially increased onus on Regional Councils and Territorial Local Authorities to comply. The implementation of Standards to the extent that all of New Zealand must be in compliance by 2013, has become a focus for many stakeholders. The health effects analysis will help to justify some of the policy measures needed.

12.3 Policy in Christchurch

Prior to the Resource Management Act (1991) air quality in Christchurch City was managed under the Clean Air Act (1972) by the Christchurch City Council and the former Department of Health. Clean Air Zone orders (1974 & 1984) established under this Act focused on domestic heating as this was considered a significant source of air pollution. These orders prohibited the installation of open fires and restricted the installation of wood burners to models meeting the specifications of the Council in an area designated the "Clean Air Zone" of Christchurch. Home heating appliances installed prior to these orders were permitted to be used. However, restrictions on the use of wet wood (>25% moisture) and high sulphur coals (>1%) introduced under the Clean Air Zone Orders applied to all domestic home heating appliances.

The Resource Management Act (RMA) required Regional Councils to prepare a Regional Policy Statement (RPS). The purpose of the RPS is to achieve the purpose of the RMA by providing an overview of the resource management issues of the region, and policies and methods to achieve integrated management of the natural and physical resources of the whole of the region. One of the methods for achieving the objectives and policies stated in the RPS is through regional plans.

Environment Canterbury has prepared the air quality chapter of the proposed Canterbury Natural Resources Regional Plan to achieve "sustainable management" of air quality (Environment Canterbury 2002). This was publicly notified on the 1 June 2002 and is referred to as the "Air Plan". Submissions to the proposed Air Plan (Chapter 3 of the Proposed NRRP) were heard in May to July 2004, and depending on whether or not it is referred to the Environment Court, the plan may not by operative until 2008. Nevertheless, the first of the proposed rules on new solid fuel burners and the replacement of existing burners with lower emissions models, came into effect early in 2003, and were applied to Canterbury from 1 January 2004 (Environment Canterbury 2003).

12.4 Air plan provisions

The air plan contains objectives, policies methods and rules for the improvement of air quality in the region and has particular emphasis on improving concentrations of PM_{10} in Christchurch. The air plan includes measures relating to domestic heating, motor vehicles, outdoor burning, agrichemical spraying and industrial emissions.

12.4.1 Air plan provisions - Domestic emissions

The primary target for reductions in PM_{10} concentrations in Christchurch is domestic home heating, as studies (e.g. Wilton 2001a) indicate that this source contributes around 90% of existing PM_{10} concentrations during the winter months. The air plan rules relating to domestic home heating differ for two clean air zones within the city, which are shown in Figure 12-1.

In Clean Air Zone 1:

- From 1 January 2003, the installation of a solid fuel burner in new houses or existing houses that do not have an open fire or solid fuel burner is a discretionary activity requiring resource consent for a discharge to air.
- Any solid fuel burners installed before January 2004 must meet the 1.5 g/kg emission criterion
- Any solid fuel burner installed from January 2004 should meet the 1.0 g/kg emission criterion. However, a burner with an emission rating of between 1.0 and 1.5 g/kg may be installed but may require a retrospective resource consent when the air plan becomes operative and there is no guarantee that this would be granted.
- From January 2008, the use of solid fuel burners installed prior to 1993 is prohibited.
- From 2008 the use of all solid fuel burners not meeting the emission criterion of 1.0 g/kg is prohibited once the burner has been installed for 15 years.
- The use of open fires is prohibited from 1 January 2006

In Clean Air Zone 2

- The installation of open fires was prohibited from 1 January 2002 (note - the installation of open fires has been prohibited in the Clean Air Zone 1 since 1984)
- Up to 1 January 2004, solid fuel burner meeting any emission criterion can be installed in any home, although any solid fuel burner installed from 1 January 2003 must meet emission stack criterion.
- Any solid fuel burner installed to replace an existing burner or open fire from January 2004 should meet the 1.0 g/kg emission criterion. However, a burner with an emission rating of between 1.0 and 1.5 g/kg may be installed but may require a retrospective resource consent when the air plan becomes operative and there is no guarantee that this would be granted.
- From 1 January 2004, any solid fuel burner meeting the 1.0 g/kg emission criterion can be installed (subject to the existing building consent requirements) in new dwellings or existing dwellings not currently using a solid fuel burner. However, burners not meeting the 1.0 g/kg criterion can only be installed if resource consent for a discharge to air is granted.

In both Zones:

- Any solid fuel burner installed from 1 January 2003 must meet emission stack criterion outlined in rule AQL2 (http://www.ecan.govt.nz).
- Outdoor burning is prohibited in urban areas of Christchurch and in all residential areas of Canterbury.

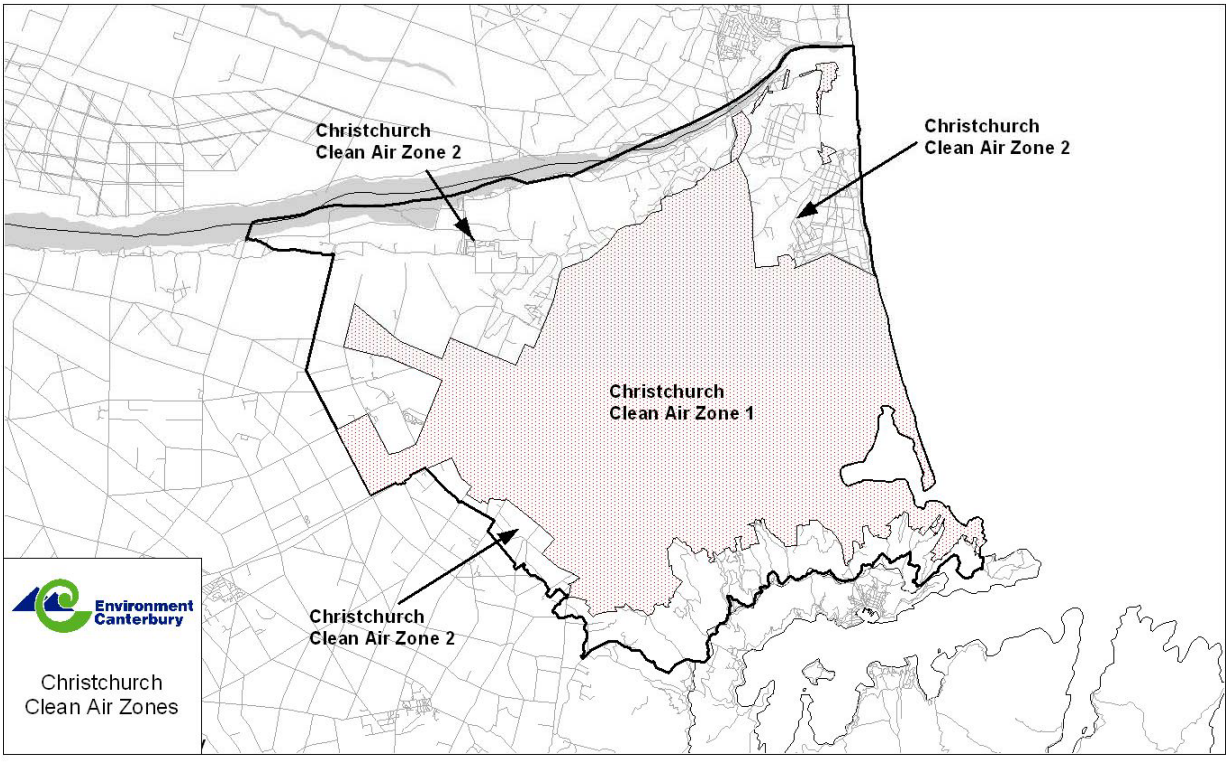

Figure 12-1. Clean air zone boundaries for Christchurch.

12.4.2 Air plan provisions - Industrial emissions

The air plan also includes regulations relating to emissions from industrial processes. These classify different activities as permitted, controlled, discretionary or prohibited as per the Resource Management Act (RMA, 1991). Prior to the air plan, industrial activities in Christchurch were reasonably heavily regulated under the Clean Air Act and the transitional provisions of the RMA. The primary differences in the air plan, provisions that impact on contaminant concentrations in Christchurch include:

- All new large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge for total suspended particulate of 250 mg m^{-3}.
- From 2015 all existing large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge for total suspended particulate of 250 mg m^{-3}.

The impact of the new standards on industrial provisions in the air plan has yet to be determined, but will likely result in some revisions within the timeframe for standards implementation.

12.4.3 Air plan provisions - Vehicle emissions

Methods for reducing emissions from motor vehicles specified in the air plan include:

- Promote traffic management to minimize localized air quality problems associated with vehicle exhaust emissions
- Promote initiatives to reduce smoky vehicle exhaust emissions
- Promote a nationally coordinated initiative to reduce the adverse effects of motor vehicle exhaust emissions.

These focus on advocacy and education, as Regional Councils are limited in their legal ability to reduce emissions from motor vehicles. Some methods for influencing motor vehicle emissions are available, e.g. providing public transport services, promoting and encouraging the use of "environmentally-friendly modes" (EFMs) such as walking, cycling and public transport; monitoring emissions concentrations and trends; and developing the Regional Land Transport Strategy, which can identify future transport needs for the region, the appropriate role for various forms of transport, and the means to achieve these. District Councils manage the local roading infrastructure, including pedestrian footpaths and cycle lanes, and have the ability to implement traffic management control measures to reduce traffic congestions. Through the district planning process, District Councils are able to implement zoning, subdivision design, parking requirements, density provisions that encourage the use of EFMs.

The primary capability to address the impact of motor vehicle emissions on air quality lies with central government. The principal strategies driving their activities are the New Zealand Transport Strategy (NZTS), in particular, the objectives "protecting and promoting public health" and "ensuring environmental sustainability," and the National Energy Efficiency and Conservation Strategy (NEECS). NEECS has a target to achieve at least a 20% improvement in energy efficiency by 2020. Three of the 6 goals are relevant to transport emissions: reduce CO_2 emissions, reduce local environmental impacts, and improve health and welfare. The Action Plan: Transport Programme outlines several "objectives" (also called "measures") to achieve the goals and target, broadly categorized as follows:

- Reduce fuel consumption: introduce eco-efficient vehicles, investigate vehicle efficiency standards and vehicle fuel consumption labelling
- Improve traffic flow: investigate and support road pricing initiatives, traffic and speed management, demand reduction and traffic demand management, improve effectiveness of "alternatives to roading" funding
- Encourage use of low energy transport modes: encourage personal energy efficient transport behaviour, reduce the need for travel, encourage use of alternative freight transport methods (i.e. shipping or rail), encourage energy efficient vehicle fleet management

Other strategies have some relevance to transport and vehicle emissions, including the Vehicle Fleet Emission Control Strategy (VFECS), the Government's Climate Change programme, the Sustainable Development Programme of Action and the New Zealand Health Strategy.

As a result of these strategies, a number of measures are either under investigation or in the process of implementation, such as:

- Adoption of the Land Transport Vehicle Exhaust Emission Rule 2003 (33001) requiring all vehicles, from 2004 (2006 for heavy vehicles), entering New Zealand to be manufactured to an approved emissions standard from the United States, Europe, Japan or Australia
- Education of vehicle users on the need for, and benefits of, vehicle maintenance.

- Adoption, through the Traffic Regulations 1976, of a "10-second rule" for visible emissions from motor vehicles
- Introduction of visible smoke check as part of Warrant/Certificate of Fitness checks
- Tightening controls on imported vehicles to ensure that they continue to meet the standards to which they were built at their time of arrival in New Zealand
- Prohibition on tampering with emission control equipment
- Revision of the New Zealand fuel specifications to reduce the benzene (4.2% to 1%), total aromatics, olefins and sulphur (from 3000 ppm to 50 ppm) content of the fuel
- Introduction of new fuel specification to allow up to 10% ethanol in petrol
- Investigation into the potential use of congestion pricing for roads in major urban areas
- Investigation of more sustainable settlement designs / forms
- Developing fuel consumption / efficiency labelling for cars entering New Zealand

In addition, the Government has increased its financial support of initiatives to encourage walking, cycling and public transport use. Examples of the types of programmes implemented include personal, business and school travel plans, walking school buses, new public transport services, installation of cycle lanes, education and information programmes, etc. The Government has also stated, in the NZTS, that it anticipates that this HAPiNZ project will "give the government information to help it target and develop further new initiatives, including those that target at risk groups". This implies that the consideration of policy tools to mitigate significant vehicle emissions may not be limited to those identified above, but could include other options.

12.4.4 Other issues

The entire policy discussion above has focussed on provisions that are already being considered (or implemented) in the Environment Canterbury Air Plan.

With the introduction of new National Environmental Standards, and likely further developments in health effects research, there are wider policy options that may need to be assessed in the next stage of this study. These include:-

- The development of "airsheds" as management areas under the Standard Regulations
- The development of "straight line paths" under the Regulations
- Amendments to the Regulations (July 2005)
- Air toxics (other than benzene), especially PAHs and 1,3 butadiene.
- Reviews of PM_{10} management methods
- Possible inclusion of $PM_{2.5}$ (and other new pollutants)
- The role of any national vehicle emissions standards
- The effects of other programmes (such as the Ministry for the Environment's Warm Homes Project).

12.5 Methodology

The policy measures proposed by Environment Canterbury for managing air pollution in Christchurch are focused on reducing concentrations of PM_{10}, as this has been identified as the main contaminant of concern in the region. While PM_{10} is a key contaminant in this study, the implications of other contaminants are also of interest. Additional consideration is therefore given to motor vehicle emissions as a significant contributor to CO, NO_2 and benzene effects. Effects of sulphur dioxide are also considered, although these are relatively minor.

The policy options assessment was broken down into three categories as per the emission inventory breakdown. These were domestic emissions, industrial emissions and vehicle emissions. The impact of each policy option on emissions was considered individually and then policy options combined and the impact on concentrations evaluated. The integration of the policy options assessment with other components of this research is shown in Figure 12-2. This suggests the need to identify potential policy options upfront to allow an assessment of changes in emissions and concentrations for each policy option.

Figure 12-3 shows a more detailed picture of the method of integrating the policy options assessment for each source with the emissions and concentrations evaluations.

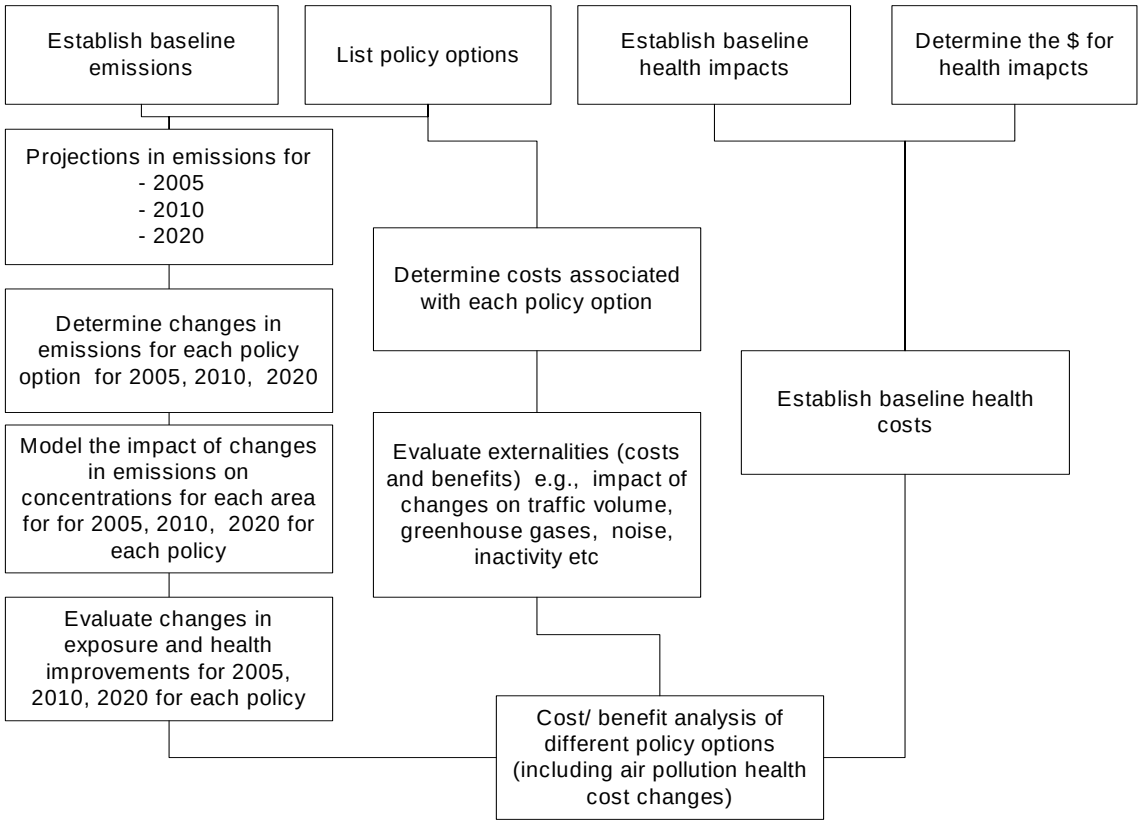

Figure 12-2. Process for assessing the costs and benefits of managing air pollution.

12.5.1 Methodology - Domestic emissions

The analysis of the effectiveness of policy options in reducing domestic emissions was based on the approach described in "Estimates of the effects of management options for reducing PM_{10} concentrations in Christchurch" (Wilton 2001a). The method of relating emissions to concentrations using the box model, however, was not required as more detailed modelling using the method described earlier was used. This study also examined the impact of policy options at a much higher spatial resolution than that carried out by Environment Canterbury. The policy options examined in this study were similar to those examined by Environment Canterbury and included:

1. A new standard for solid fuel burner installations (mandatory) of 1.5 g/kg.
2. A ban on the use of open fires plus a new standard of 1.5 g/kg for new burner installations.
3. A ban on the use of open fires plus a new standard of 1.5 g/kg for new burner installations and allow no new installations of solid fuel burners in houses not currently using them.

4. A ban on solid fuel burning for domestic home heating (Assumes burners are replaced 15 years after installation - may need to be regulatory).

The integration of the 1 g/kg standard into the regulations for Christchurch, as proposed by Environment Canterbury was not specifically included in the assessment as the impact of this rule, over and above achieving a 1.5 g/kg standard is unknown. The new National Environmental Standards for Air Quality impose a requirement for 1.5 kg/kg wood burners nationally.

A number of other policy approaches were considered for reducing emissions from domestic heating. These included setting limits at the point of discharge rather than "approving burners", and relying wholly on technology rather than a combination of technology and restrictions in burner numbers to achieve reductions.

Setting a limit on chimney emissions has significant advantages in that it is more closely aligned with the objective of the policy than "approving burners". It is also ideal in that it does not prescribe how the emissions should be reduced. However, the practicalities and cost of enforcing such an option would be prohibitive.

The option of relying wholly on technology, rather than a combination of technology and restrictions on burner numbers could involve specifying a real life emission limit for solid fuel burners that allows for ongoing installations of new burners. One limitation to this approach is the current understanding on the relationship between appliance emissions under test conditions and what occurs in real life.

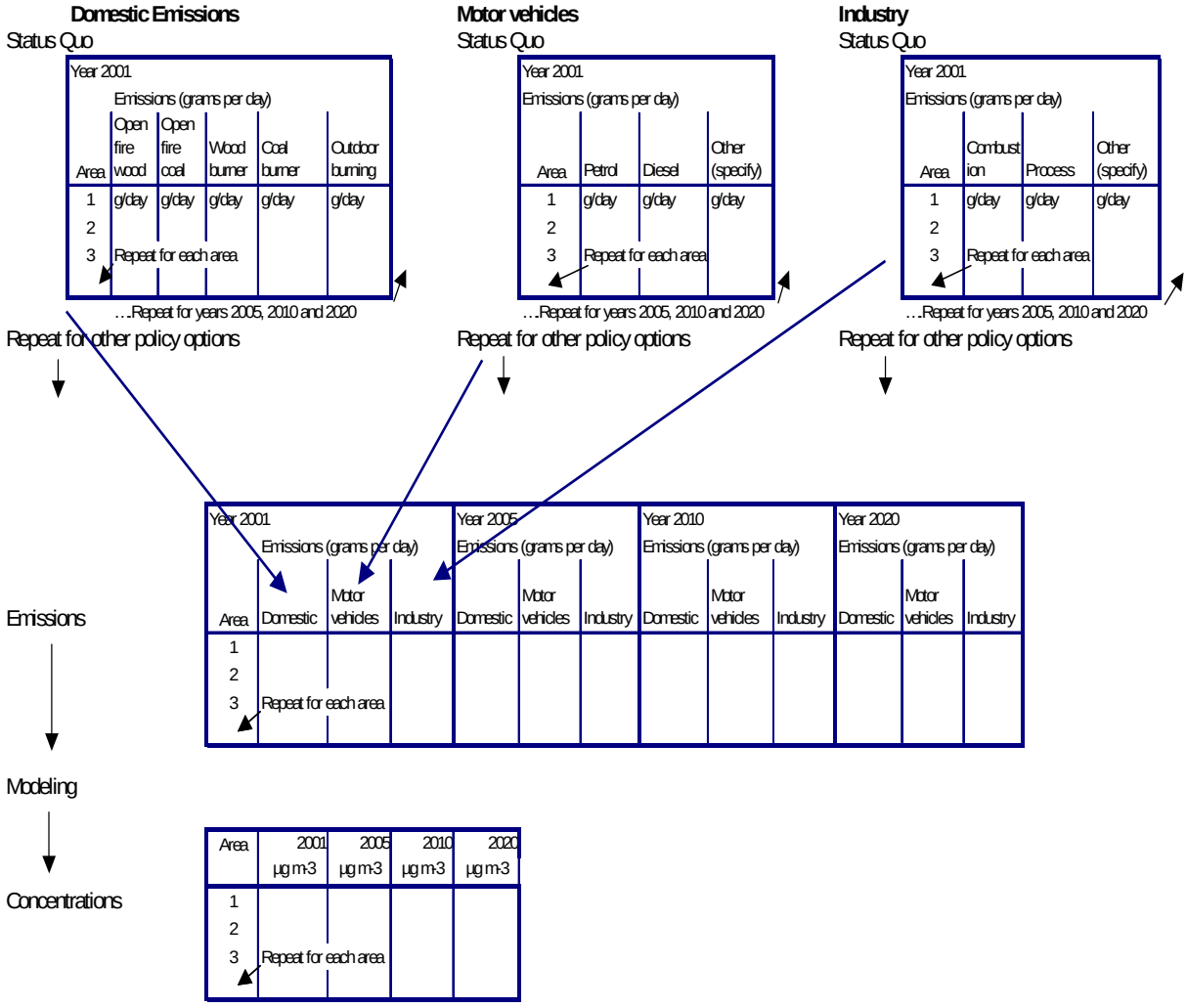

Figure 12-3. Process for integrating the policy options assessment with air pollution data.

12.5.2 Methodology - Industrial emissions

The policy option evaluated for industry was the requirement that all large-scale combustion processes meet an emission limit of 250 mg m^{-3}. The approach taken here was to rely on the original analysis for this option, which was carried out by Environment Canterbury and to extend this to include an assessment of the spatial impact of variations in emissions from this source. This involved a spatial disaggregation of industrial emissions by process type to identify areas where reductions were likely to occur (this option was developed before the introduction of standards - others will be considered in the next stage of the work). The 250 mg m^{-3} limit is modest in terms of Best Practice Option (BPO). A more realistic option, being adopted by many Councils, is 50 mg m^{-3}.

12.5.3 Methodology - Vehicle emissions

The analysis of the motor vehicle emissions policy options has not been quite so straightforward when compared with the domestic emissions analysis. Generally there are two overarching inputs:

- The motor vehicle emissions of concern in the Christchurch setting (by pollutant, fuel type, vehicle type).
- "Typical" areas where exposure to these emissions needed mitigation.

For each relevant pollutant:

- Temporal factors associated with exposure (time of day (i.e. the one or two hours where the daily maximum occurs), day of week, seasonal variation).
- The "external" factors affecting the exposure (Geographical / topographical conditions, climate (wind, rain, temperature, etc) and the relative importance between them (i.e. do policies have to vary for different geographical areas, or for different climatic conditions or both?).
- The relevant traffic conditions affecting the exposure (stop/start, free flow, speed, idling, etc).
- The possibility of "extreme" days – usually a combination of vehicle emissions, traffic conditions and weather to create the extreme – the frequency of such occurrences and the severity compared to "normal" occurrences (i.e. to identify policy tools that may need to be in place to address these "marginal events").

A full policy analysis has not been undertaken for this Pilot Study. The options and methodologies are discussed here, but the analysis awaits more detailed information and will be undertaken for the final phase of the study.

12.6 Policy options

Policy options are assessed under the same general categories previously identified – domestic, vehicle and industrial.

12.6.1 Policy options - Domestic emissions

The policy options evaluated for domestic emissions include:

- A new standard for solid fuel burner installations (mandatory) of 1.5 g/kg (Assumes burners are replaced 15 years after installation - may need to be regulatory).
- A ban on the use of open fires plus a new standard of 1.5 g/kg for new burner installations.
- A ban on the use of open fires plus a new standard of 1.5 g/kg for new burner installations and allow no new installations of solid fuel burners in houses not currently using them.
- A complete ban on solid fuel burning for domestic home heating.

People affected by the domestic heating policies include:

- Homeowners with solid fuel burners that require replacement.
- Homeowners that use open fires for domestic home heating.

- Tenants renting homes with open fires or solid fuel burners.
- Homeowners with existing dwellings using other heating methods that may want to install a solid fuel burner.
- Those building new dwellings that may want to install a solid fuel burner.

A number of social and other issues may arise as a result of different policy options for domestic heating. These relate to benefits of introducing the measures such as improved health and reduced healthcare costs associated with reductions in contaminant concentrations as well as the costs associated with the introduction of policy. An analysis of the costs and benefits associated with different policy options is shown below.

Social impacts associated with different options for domestic home heating include:

- Reduced heating choices for households unable to afford the price of a lower emission solid fuel burner
- The potential for increased heating costs for households unable to install solid fuel burners.
- Increased heating costs for households previously using open fires and self collected wood.
- Potential impacts on burner manufacturers and retailers if fewer households install solid fuel burners.
- Cost implications to manufacturers including research and design costs, testing costs and inability to supply burners that complies with the lower emission standard.
- Loss of ambience associated with the use of open fires for domestic home heating.

It is possible that households e.g. some of those previously using open fires, may choose not to heat their dwellings as a result of the increased costs of alternative heating methods. This may have negative impacts on health and lifestyle and may result in increased medical costs. There is also the potential for increased health impacts associated with the use of non-flued gas heaters for solid fuel burning.

A detailed social impacts assessment was carried out for a range of management options for domestic home heating for Christchurch in 2001 (Taylor, 2001). This indicates that less than one third of households earning less than $15,000 per year use solid fuel burning for home heating compared to over a half of households with an annual income of greater than $30,000.

An economic incentives programme has been provided by Environment Canterbury to minimise negative social and economic impacts that may occur as a result of its proposed policy measures. The incentives programme offers both full and partial assistance in the replacement of open fires and older solid fuel burners with cleaner heating methods. To be eligible for the full assistance programme the homeowner must hold a community services card or receive a specific government benefit. Table 12-1 outlines the incentives offered to households eligible for partial assistance. The "Clean Heat" programme will provide assistance for home owners and landlords to upgrade their heating methods, with a target of 30,000 households.

Stage	Measure	Incentive
Initiation	Assessment	FREE
	Sealing Open Fire/Removal of old burner	$100
Appliance Installed	Heat Pump	$500
	Night Storage - Electric	$300
	Fixed-Flued - LPG/Diesel Oil	$500
	Solid Fuel Burner	$500
Efficiency Measures	Ceiling Insulation	$3.50/$m^2$
	Under floor Insulation	$3.50/$m^2$
	Dampness Barrier - (if necessary)	$2.00/$m^2$
	Draft Stopping - Various Items	30% up to $100

Table 12-1. Environment Canterbury's 'Clean Heat Programme' – Incentives.

12.6.2 Policy options - Industrial emissions

The policy options evaluated for industrial emissions include:

- All large-scale industrial combustion processes meet a PM_{10} emission discharge concentration of 250 mg m^{-3}.

Those affected by the policy for reducing PM_{10} emissions from industry include:

- Any new industry / commercial activity proposing to burn wood or coal with a heat output of greater than 40 MW.
- Any industry / commercial activity burning wood or coal with a heat output of greater than 40 MW from 2015.

Many of the larger-scale industrial combustion processes will require additional emissions reduction equipment to achieve the proposed policy options. Costs associated with the emissions equipment upgrade could be in the order of around $25,000 to $30,000 per year for some of the larger-scale burners in Christchurch. An assessment of the costs relative to improvements in air quality carried out by Coal Research Limited (CRL, 1997) indicates cost benefit ratios ranging from $7002 per tonne of PM_{10} reduced to $111,725 per tonne. Sites with the best (lowest) cost to benefits ratio were typically moderate sized boilers in the 1-5 MW range.

Social impacts include the potential for loss of jobs if the additional costs of compliance are prohibitive for some industry. Other impacts include the location of new industry outside the Christchurch area.

12.6.3 Policy options - Vehicle emissions

As noted earlier, an analysis of vehicle emissions policy options has not been undertaken at this stage. The detailed information required was not available at the time of this pilot analysis. This is not seen as a major shortcoming, given the relatively minor contribution of vehicle emissions to the total air pollution problem in Christchurch. This situation may be quite different in other centres, particularly in Auckland where vehicle emissions are known to be the predominant contributor to air pollution.

12.7 Assessment of policy options

12.7.1 Assessment - Domestic emissions

The starting point for this analysis is the estimate of the costs of domestic emissions in Christchurch for the year 2001 (Table 12-2). The mortality estimates shown are based on the dose-response relationship of 4.3% increase per 10 µg m^{-3} in 24-hour average PM_{10}.

Effect for domestic sources	Number	Cost
Mortality	124	$93.0M
Cancer cases	1.1	$0.8M
Bronchitis cases	36	$2.7M
Cardiac cases	37	$0.1M
Respiratory cases	136	$0.4M
Restricted activity days	200,000	$30.0M
Minor hospital costs	total	$0.15M
Total		**$127M**

Table 12-2. Effects of domestic air pollution emissions in Christchurch (repeated).

In Christchurch the status quo option includes a low emission limit for new burner installations of 1.5 g/kg. The health benefits of additional policy options are the difference between the status quo health impacts and the additional policy impacts. Figure 12-4 shows a reduction in premature mortality of around 60 by 2020 (to 64) for the status quo option (includes the effect of 1.5 g/kg standard), by 99 by 2020 (to 25) for a ban on open fires along with the uptake of the new woodburner standards, by 113 (to 11) for an open fire ban and restrictions on new burner installations, and by 124 (to 0) for a complete ban on solid fuel burning.

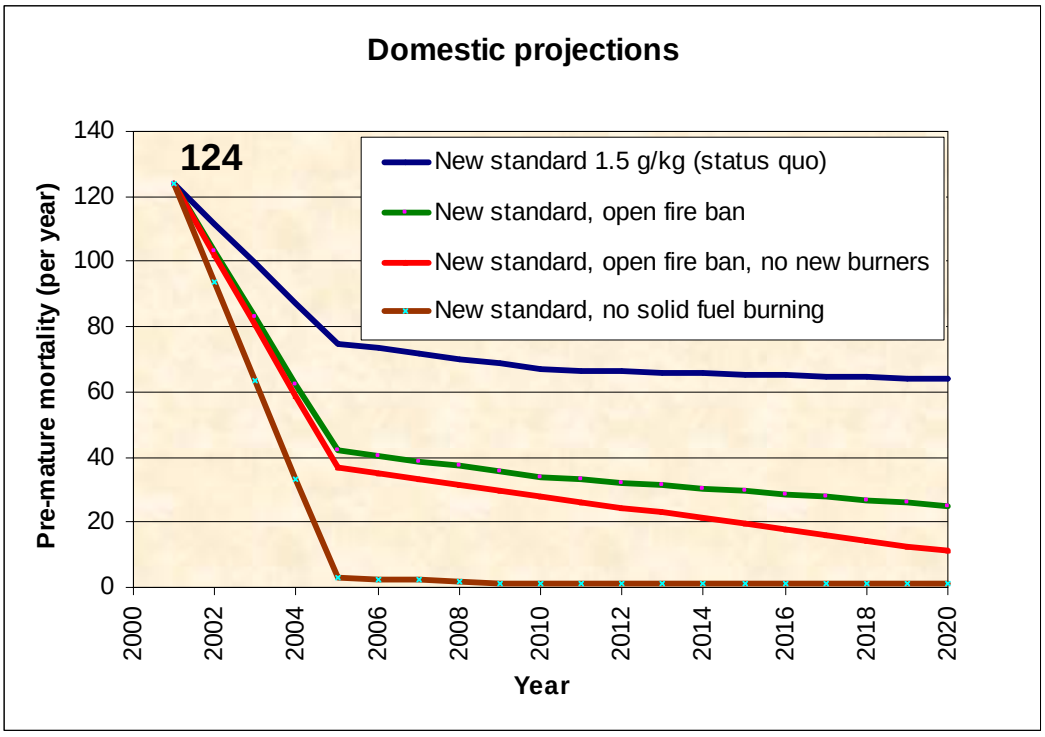

Figure 12-4 Projections in premature mortality for policy options for domestic heating.

12.7.2 Assessment - Industrial emissions

The starting point for this analysis is the estimate of the costs of industrial emissions in Christchurch (Table 12-3). The mortality estimate is based on a dose-response relationship of 4.3% per 10 µg m^{-3} increase in PM_{10} as discussed earlier.

Effect for industrial sources	Number	Cost
Mortality	18	$13.5M
Cancer cases	0.3	$0.2M
Bronchitis cases	9	$0.7M
Cardiac cases	9	$0.05M
Respiratory cases	32	$0.1M
Restricted activity days	45,000	$7.0M
Minor hospital costs	total	$0.03M
Total		**$22M**

Table 12-3. Effects of industrial air pollution emissions in Christchurch.

The health impacts of requiring that all large-scale combustion processes meet an emission limit of 250 mg m^{-3} by 2015 is shown in Figure 12-5.

The reduction in premature mortality (currently 18), compared with the option of not implementing the policy (24) is around 6 premature deaths per year by 2020 (to 18).

The costs of implementing the policy option of requiring all large-scale industrial combustion processes meet a PM_{10} emission discharge rate of 250 mg m^{-3} has not been assessed.

Although the existing number of large-scale boilers affected by the option is small, at around 30, costs could vary significantly from site to site depending on the existing technology.

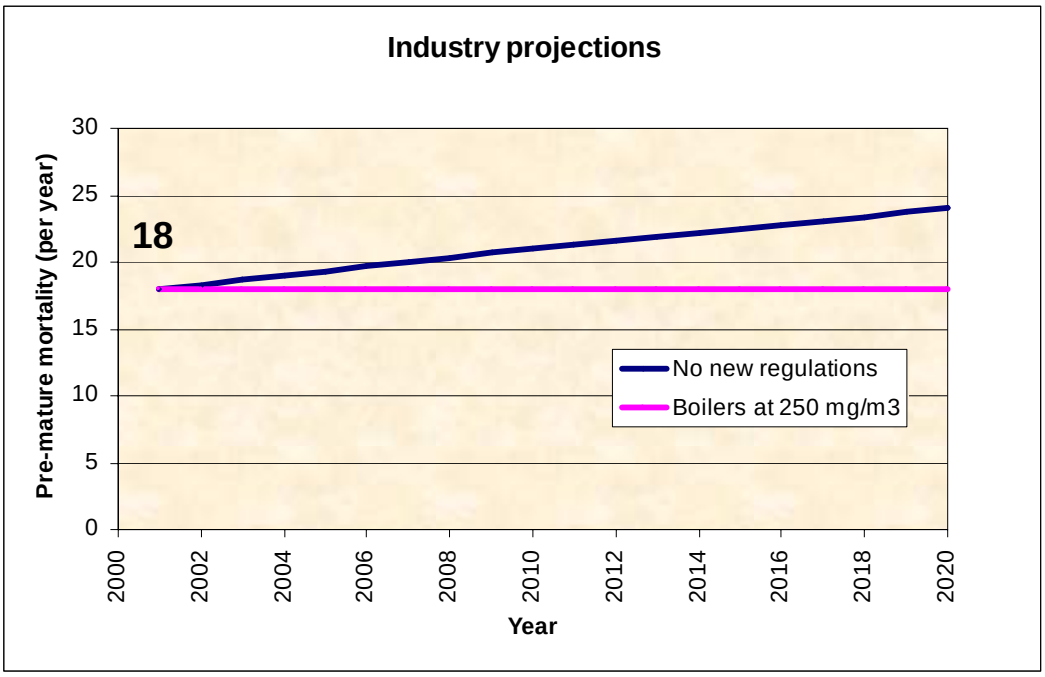

Figure 12-5. Projections in premature mortality for policy options for industry.

12.7.3 Assessment - Vehicle emissions

This analysis estimates the current (2001) costs of vehicle emissions in Christchurch (Table 12-4). The mortality estimate is based on a dose-response relationship of 4.3% per 10 µg m^{-3} increase in PM_{10} as discussed earlier.

Effect for vehicle sources	Number	Cost
Mortality	16	$12.0M
Cancer cases	0.2	$0.2M
Bronchitis cases	7	$0.6M
Cardiac cases	7	$0.05M
Respiratory cases	27	$0.1M
Restricted activity days	40,000	$6.0M
Minor hospital costs	total	$0.02M
Total		**$19M**

Table 12-4. Effects of vehicle air pollution emissions in Christchurch.

A projection for various policy options has not been undertaken, as sufficient information is not available at this stage.

12.8 Policy example analysis

Domestic: The following policy options will have significant impacts in reducing the health impacts associated with domestic emissions:

- A ban on the use of open fires
- No new installations of solid fuel burners in houses not currently using them

In addition, capping the useful life of a burner at 15 years will ensure the continual upgrading of burners with the most recent technology burners.

Industrial: Policy recommendations for industrial emissions include: -

- All new large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge for total suspended particulate of 250 mg m^{-3}.
- From 2015 all existing large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge for total suspended particulate of 250 mg m^{-3}.

National environmental standards for air quality may also impact on the granting of resource consents for discharges to air in Christchurch. The 250 mg m^{-3} limit is relatively modest, with 50 mg m^{-3} being adopted by many Councils now. The implications of the 2004 Regulations and the 2005 Amendments are still be analysed, and will be included more fully in the next stage of this work.

Vehicles: Policy options for vehicle emissions have not been analysed, but will be undertaken in the next stage.

12.8.1 Justification for options

Recommendations for policy options for domestic heating and industry are the same as those selected by Environment Canterbury. The main differences between this assessment and that carried out by Environment Canterbury are that this assessment includes a more detailed evaluation of exposure based on modelling, and that this assessment assumes different dose-response relationships for particles from domestic home heating, industry and motor vehicles. Including more detail on the spatial distribution of concentrations and exposure allows for the selection of policy options that have the most impact upon

the exposure of the largest number of people. As a result, it provides a better estimate of the relative effectiveness of different policy options in improving health. Results showed the same cost-effectiveness rankings for the policy options as the Environment Canterbury assessment.

The more stringent policy option of prohibiting the use of solid fuel burning in Christchurch was also considered. These benefits need to be considered in the context of the costs, the social impacts and the objectives of the policy.

Evaluation of policy options for industry was limited as this source is already regulated in Christchurch. The options considered were those included in the draft air plan for Christchurch. These measures will help curb increases in industrial PM_{10} emissions, which are predicted to increase in time in the absence of further controls.

12.9 Discussion

What are the options and issues for implementing these recommendations? In the absence of policy options for air quality, a reduction in premature mortality of around 56 deaths per year is estimated to occur (to 102 from the current 158). Implementing the full policy option recommendations could result in an additional reduction in premature mortality of around another 66 deaths per year, from domestic heating (60), and industry (6) by 2020, to a total of 36 overall. This figure will be reduced further with vehicle emissions reduction occurring, and extra mitigation in place

The policy impact analysis is dependent on a number of assumptions. Of particular significance to the calculations of health benefits is the dose-response relationships used to estimate the benefits of reducing PM_{10} concentrations. Policy options targeting domestic heating will result in significant reductions in PM_{10} concentrations. Other assumptions that are significant in establishing policy impacts include the impact of predicted changes in tailpipe emissions from the New Zealand vehicle fleet.

12.9.1 Implementation and cost issues

Options for domestic heating and industry are being implemented through the air chapter of the natural resources regional plan (NRRP) for Canterbury. Additional requirements for industry may be required as a result of the NES for air quality.

To address issues of cost associated with policies relating to domestic heating, Environment Canterbury has implemented a financial incentives package. The purpose of the package is to assist with the costs of converting from non-complying heating methods to cleaner options such as electricity, gas, low emission burners and for energy efficiency measures.

It is possible that the cost of upgrading some existing industrial boilers will impact on the economic feasibility of their activity. The stricter requirements may also discourage new industries from locating within the city. Compliance monitoring and enforcement of the industrial policies can be carried out effectively through the existing compliance monitoring regime. Additional measures will need to be implemented to monitor compliance with domestic heating and traffic related policies.

Costs for implementing any new vehicle emissions controls have not been evaluated by Environment Canterbury, and have not been fully analysed by the Ministry of Transport. However they are not insignificant, either in dollar terms or in terms of social equity.

12.9.2 Potential barriers

Some households may choose to ignore the regulations and continue to use prohibited appliances. The provision of assistance in the form of financial incentives may reduce the incidence of deliberate non-compliance. Ongoing non-compliance may be minimised by the use of fines and the publicising of prosecutions.

12.10 Summary

The policy impacts analysis indicates a reduction in premature mortality of around 56 deaths per year may occur in the absence of any additional measures to improve air quality in Christchurch. An additional 66 premature deaths per year may be avoided by implementing the following policy options:

- A ban on the use of open fires.
- No new installations of solid fuel burners in houses not currently using them.
- Requiring that non-complying solid fuel burners be replaced 15 years after installation.
- Requiring all new large scale solid fuel burners with a heat output of greater than 40 kW meet an emission discharge for total suspended particulate of 250 mg m^{-3}.
- From 2015 all existing large scale solid fuel burners with a heat output of greater than 40 kW are required to meet an emission discharge for total suspended particulate of 250 mg m^{-3}.

The costs associated with implementing options for domestic home heating are significant.

(NB. A number of other policy options could have been assessed – such as effects of the new air quality standards, tighter controls on industrial dischargers through consents – particularly using a 50 mg m^{-3} discharge limit, vehicle emissions control, fuel specifications, congestion pricing etc. These will be considered, following advice from the Ministry of Transport and Ministry for the Environment in the next phase of the project).

13 Next Phase

This Pilot Study is the first phase of the HAPiNZ project. It has been undertaken in great detail to assess the options and methodologies for undertaking an analysis for the whole of New Zealand. The format and scope of this Pilot Report are such that it has allowed a great deal of review and refinement of the approach for the final part of the study.

Since the work was commenced, and the exposure assessment for Christchurch was carried out, a great deal more information has become available. This includes new monitoring data, better modelling, new results on health effects, and new policy within New Zealand. As many of these aspects as are feasible will be incorporated into the final report.

These include:-

- Updated exposure estimates (using 2004, instead of the 2001 year used here)
- Updated economic costs
- Update the literature background, especially with new material from the USA, UK and Europe
- Provide a wider range of health effects information to assess policy options
- Include the effects on asthma
- Include a wider range of effects on children
- Explore further the effects of fine particles

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Appendix 1 – New Dose-Response Relationships

The epidemiological approach used in this study follows closely a methodology that has been used elsewhere and has become a de-facto standard for such studies (Kunzli et al 2000). The details are summarised in chapter 7.

However, since an excellent data set was available for Christchurch, this was used to examine some new and finer scale aspect of the dose-response relationship. This research is described in this Appendix, but results are preliminary, and subject to further confirmation before being applied more widely.

A1.1 Scope of new studies

It was established by Hales et al (2000b) that an increase of daily PM_{10} levels in Christchurch is associated with an increase in daily mortality. This finding made it of interest to quantify the dose-response relationships for different health effects of air pollution under New Zealand conditions. New exploratory epidemiological studies have been started within the HAPiNZ project on mortality and morbidity in relation to air pollution exposures. This includes a renewed daily time-series analysis for Christchurch, including distributed lag analysis, and seasonal and annual spatial analysis of the association between air pollution and health effects in both Christchurch and Auckland. Preliminary results will be presented here.

In order to take proper account of exposure variations on a seasonal and diurnal basis detailed analysis was carried out on Christchurch air monitoring data over a 12-year period. The variation of dose-response relationships over time, by age and sex, and by pollution sources has also been studied.

The epidemiological work is ongoing and future results will be relevant for the whole of New Zealand analysis in the final report. It is intended to widen the scope of this work and compare assessments with Australian cities. The formalised collaboration with the Australian National University allows for this. The latest international knowledge from the literature, and collaboration with ongoing research programmes in USA and UK, has and will be used. This will lean on very recent data emerging on the causes and effects of particulate toxicology, as well as established literature. The aims of this component of the research are:

- To estimate past exposures to particulate matter and other air pollutants in Christchurch.
- To investigate dose-response relationships for mortality and morbidity effects in Christchurch, relevant for health risk assessments, combining the new local epidemiological results and evidence from overseas.
- To prepare methods for estimation of health impacts from air pollution in Christchurch and other urban centres of New Zealand.

A1.1.1 Daily time-series analysis

This methodology has been widely used to establish that daily air pollution exposures in urban areas are associated with increased mortality and morbidity. It is a powerful tool for establishing that acute effects take place and by using so-called distributed lag models the cumulated health effects of short-term high air pollution exposures can be measured. A key feature of this methodology is that any seasonal variations in "background" mortality or morbidity (due to other factors than air pollution) is deducted from the daily variation of deaths potentially associated with air pollution. Much discussion has ensued about which method for background adjustment is the best. A method developed for a European collaborative research project (APHEA and APHEIS) was used, which included cities and populations with air pollution exposures and other characteristics relatively similar to New Zealand's conditions.

A major drawback of these studies is that the seasonal background adjustment of daily mortality or morbidity data may in reality adjust away the long-term effects of air pollution that are of greatest importance to public health. If the reason for the higher respiratory and cardiovascular mortality and

morbidity in the winter is the medium term seasonal variations of air pollution, these effects may be underestimated by such time-series methods.

In this study a combination of different methods will be used in order to better describe the air pollution related health effects in Christchurch. However, the daily time-series analysis results may be used to give an indication of the relative importance of different air pollution sources and the other exposures of interest, such as climate factors.

A1.1.2 Spatial variation of annual air pollution and health effects

A study was carried out using GIS methods and airshed modelling to estimate annual average exposure for each CAU in Auckland (Scoggins et al. 2004) (see chapter A1.3). This study provides important long-term dose-response relationship estimates under New Zealand conditions and it will therefore be described and the results used in this pilot study report of Christchurch. A study of the long-term dose-response relationships for the Christchurch population is in progress as a part of a PhD project by one of the HAPiNZ team members (Rupendra Shrestha, ANU).

A1.2 Daily time-series / dose-response estimates

A1.2.1 Methodology

The Poisson regression protocol by APHEA (Air Pollution and Health: A European Approach) (Katsouyanni et al. 1996) and APHEIS (Air Pollution and Health: a European Information System) (APHEIS, 2001) was used as a starting point for the methodology, taking into account more recent updating of this protocol (Katsouyanni et al. 2001). This approach enables comparison of dose-response relationships with studies in different countries. Mortality (all causes) and hospital admission (respiratory and heart disease) data for all population groups in Christchurch for the period 1988 to 1999 were acquired from the New Zealand Health Information Service. The population characteristics data used include age, sex, address (census area unit), ethnicity and 4-digit ICD-9 code. Air pollution exposure variables, as well as meteorological data were included as independent determinants in the regression modelling (data from ECan and NIWA). Missing data were accounted for by modelling based on meteorological data, day of week and season (Shrestha & Kjellstrom 2003). Time-series analysis was carried out for the whole city population. Age-specific analysis was also carried out. Each air pollutant and combinations of them were assessed, in order to identify the characteristics of the exposure that have the greatest impact on daily mortality (analysis of morbidity effects, hospital admissions, are in progress).

Long-term trends of mortality and seasonal patterns were modelled using mathematical functions. In Christchurch there are more deaths in the winter than in the summer in the age groups at 45 years and above. The winter/summer ratio (Ratio) of mortality increases with age (approximate figures): age 45-64 Ratio = 1.2; age 65-79 Ratio = 1.3: age 80+ Ratio = 1.5. The daily time-series modelling excludes any effects of climate and air pollution on these seasonal time trends. Calendar effects were modelled with dummy variables for day of the week. Influenza incidence was recorded from general practitioner surveys and laboratory records. Influenza was initially considered a potential confounding factor, but as described earlier there was no evidence of confounding from this variable.

Poisson regression using the method of maximum likelihood in the Stata software (Stata 1997) was applied. In order to assess the shape of the dose response functions for different exposure variables, scatter-plots with best fit curves from the lowess-smoothing method (Locally Weighted Estimated Smooth Surface) comparing single variable and multi-variable models were used. Seasonal effect modifications were analysed by separate winter (May-August) and non-winter (September-April; here labelled "summer") models. The relationship between winter particles (most of them from wood smoke) and summer particulate (with virtually no wood smoke) was examined. Different lag-periods for effect (half-day, 1-day, 2-day, etc.) were tested. The lag is the distance in days between the day of air pollution used in the model and the day of mortality. The peak impact of air pollution does not necessarily occur on the same day as the peak air pollution. A somewhat delayed effect is likely. The

analysis of different lags describes these delays. Interactions in multi-pollutant models were assessed to identify any problem of multi-collinearity between air pollution variables.

Based on the mathematical functions for the dose-response relationship between exposure to air pollution, climate factors and health effects, estimates of the attributable risk for the different factors were made for each season, winter and summer

The study covers 12 years, whilst an initial publication on Christchurch (Hales et al. 2000b) covered only 6 years and found a statistically significant effect of PM_{10} on respiratory and total mortality (1% increase of total mortality per 10 μg m^{-3} PM_{10} increase). With the increased number of days to study, the analysis has sufficient power to detect an effect of at least this magnitude. To the extent possible, specific effects on different mortality diagnoses, and effects on different age and gender groups were analysed.

Exposure variables

These were the concentrations of the air pollutants PM_{10}, CO, and NO_2 expressed as daily average levels or yearly levels. The details are given below. Daily values used came from hourly monitoring data and were calculated as midnight to midnight values. This is different from Environment Canterbury reported averages which are on a 9am to 9am basis.

Potential confounding variables

These were the climate variables ground temperature, relative humidity and wind speed. The temperature difference between ground and 10 meters height and wind direction are important variables behind the actual PM levels, but were not considered likely to affect mortality. Other potential mortality confounders, such as age, sex, and ethnic group, are taken into account in daily time-series analysis by the assumption that the composition of the population changes little from day to day.

Effect variables

The health-endpoints were daily mortality (total, non-external, cardiovascular, and respiratory). Most of the results reported below are for non-external mortality.

Analytical methods

Poisson regression modelling was used to investigate the association between daily mortality/ morbidity with daily air pollution levels, while controlling for other variables like temperature, relative humidity, long-term trend, seasonal variation, day of the week. The dependent variable in the model is the daily number of deaths/hospital admissions and independent variables are daily air pollution level, daily weather variables and other confounders. The model used in this analysis is

$$E(Y_t) = \exp(\beta_0 + \beta_1 * PM_{10,\,t-l} + \text{Sum}[\beta_i * X_i])$$

Where $E(Y_t)$ is the number of deaths on day "t", $PM_{10,t-l}$ is PM_{10} level on day "t-l" where the letter "l" is the lag l (number of days) and X_i's are the daily climate confounding variables. The β_i's are the fitted coefficients that represent the response function (β_0 may be non zero in this model, but the only reason it would be other than zero is if some confounding factors had not been explicitly accounted for. It is expected to be small, otherwise the variance in the model due to unknown factors would render the model unusable). Most of the confounding variables have non-linear relationship with the dependent variable. In order to adjust for these non-linear confounding effects of seasonality and weather variables, some studies have used natural cubic splines smoothing while fitting Generalised Linear Models (GLM) and other studies have fitted Generalised Additive Models (GAM) with non-parametric splines smoothing, for example lowess smoothing.

Following the APHEA protocol, a linear term for year to control for long-term trends in mortality and a series of sine and cosine terms in the model to control for seasonal variations was used. To control for non-linear relationship of temperature with the outcome variable, new variables based on daily maximum temperature and daily minimum temperatures were created and used the new variables in the models rather than observed temperatures. For this, a threshold maximum temperature above which the daily maximum temperature had a positive linear effect on daily mortality was estimated. A variable HOT was created such that its value was set to 0 if the maximum temperature was below the threshold

temperature otherwise maximum temperature minus threshold temperature. Similarly, a threshold minimum temperature below which the daily minimum temperature had a negative linear effect on daily mortality was estimated. A variable COLD was created such that its value was set to 0 if the minimum temperature was above the threshold temperature otherwise minimum temperature minus threshold temperature. The threshold temperatures were identified by minimizing the Akaike Information Criteria (AIC). The new variables HOT and COLD were used in the models to control for non-linear temperature effects.

To investigate the effect modification by season, a dummy variable for season (1 for the winter months May –August and 0 for the non-winter months September – April) and the interaction between season and pollution level were added in the models.

A1.2.2 Features influencing health risks

In order to better understand the exposure conditions in Christchurch a number of different analyses of the air quality monitoring data has been carried out by Kjellstrom, Shrestha, Scoggins, Exeter, and Dirks. Several conference reports have been presented and journal publications based on these analyses are under preparation.

There are a number of questions that the research could answer, including:

- To what extent does 24 hour average air monitoring data obscure major variations in exposure as reflected by hourly monitoring results?
- Can the hour of the day when the peak occurs influence the time lag seen in epidemiological analysis? (Lag 0 or lag 1)?
- Can the hourly variation of the different pollutant levels during different seasons and during different days of the week improve the understanding of the source apportionment (vehicles or wood-smoke)?
- Can the date of beginning of winter wood burning be identified from meteorological and air monitoring data?
- Does an Air Quality Index based on a combination of monitoring results for PM_{10}, CO and NO_2 provide a useful estimate of exposure of relevance to the mortality effects?
- Can the "chimney density index" based on census data be used as a proxy for the spatial emission distribution of wood-smoke?

Each of these questions will be explored further. This pilot study highlights some aspects, including the hourly variation of PM_{10}, CO and NO_2 during winter and summer days and the relationship of exposure levels to different air pollution sources.

The analysis here will use the St Albans (Packe St) monitoring data to represent exposure levels in the whole Christchurch metropolitan area, as the CAU-based spatial distribution of air pollution exposure levels was not available at the time of this epidemiological analysis. This approach may assign too high exposure levels to people living on the outer suburbs of Christchurch, which would tend to underestimate any dose-response relationships documented. (This is similar to the approach taken earlier by Hales et al (2000b) that confirmed the dose-response relationship for Christchurch – based on St Albans data – was similar to that being used elsewhere, for instance in Kunzli et al, 2000.)

Harmonising and completing the air monitoring data set

The study period for the epidemiological analysis is 12 years (1988-1999) during which time the air quality monitoring methods for PM_{10} changed several times. In order to make the recorded levels comparable for the whole period a harmonisation method was applied, which is described in detail by Shrestha & Kjellstrom (2003). Fortuitously, periods of changeover from one monitoring method to another included some time when both methods were used. This made it possible to directly compare two methods for the same hours of measurement. Thus, all hourly PM_{10} values were converted to the method used in 2003, TEOM at 40 degrees C.

Another problem was that for approximately 10% of all the hours over the 12-year period, no recordings were made due to equipment breakdown, power-cuts etc. Using hourly data on climate variables, hour of the day, day of the week, season, etc. a multiple regression model was fitted that could estimate the

hourly PM_{10} level (Shrestha et al. 2002). Similar models were established for CO and NO_2. Using these models all the missing hourly PM_{10}, CO and NO_2 data were estimated to complete the data set of hourly air quality.

Temporal variation in exposure

Figures A1-4 to A1-6 show the average hourly levels for calm days (average wind-speed < 2 m/s) in winter (an example for May to August 1997) and summer (September to April) for the three pollutants at St Albans, Christchurch. There is a clear pattern of low levels for all pollutants in the afternoons, most likely caused by some breeze blowing during these hours, even though the average for the day is calm.

In the summer there are two distinct peaks reflecting meteorological conditions and emission variations due to driving patterns and variations in industrial emissions. There is a morning peak between 8.00 and 10.00 and a longer evening peak between 18.00 and 23.00. Traffic count data shows that the driving pattern has two distinct peaks especially on weekdays; one in the morning between 8 to 10 and another in the evening between 17 to 18 indicating a higher number of vehicles on the road during office and school commuting hours. The winter patterns are similar, but the morning and evening peaks are higher and longer. This reflects the temperature inversions that occur during cold nights, the regular changes in the direction of the slow wind movement during these calm days (at 9 am and 5 pm), and the variations in emissions from wood burning, industry and traffic. The daily patterns with high air pollution concentrations in the evenings make it intuitively logical that a lag effect on mortality would occur.

The ratio between winter and summer values is greatest for PM_{10} and lowest for NO_2. The three pollutants are closely correlated, especially PM_{10} and CO, due to the dominating influence of meteorology on daily variations. However, differences within a geographic area will occur depending on the emissions from different sources.

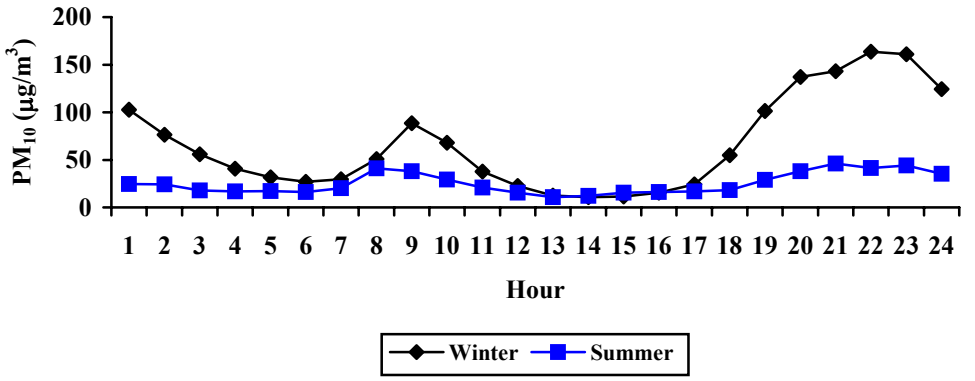


Figure A1-1. Hourly PM_{10} concentration by season for calm weekdays in 1997.

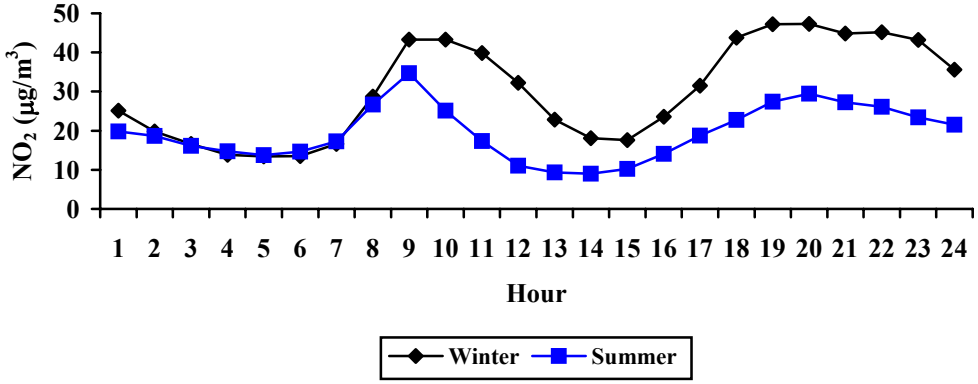


Figure A1-2. Hourly NO_2 concentration by season for calm weekdays in 1997.

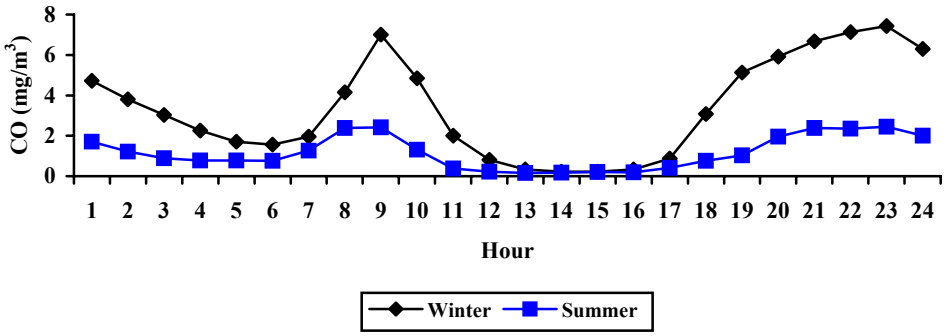


Figure A1-3. Hourly CO concentration by season for calm weekdays in 1997.

In Figures A1-1 to A1-3 the diurnal patterns of PM_{10} (Figure A1-1) and CO (Figure A1-3) are highly correlated, with troughs and peaks occurring at the same times. This reflects the similar sources (home heating and vehicles) and the daily usage patterns for heating and vehicles. However the pattern for NO_2 is slightly different, with higher night-time background values and broader peaks and troughs (Figure A1-2). This is due to the different behaviour of NO_2 in the atmosphere. It forms principally from the conversion of the NO released by the heating and vehicle sources, and the time and rate of this depends on factors such as the availability of oxidants, the temperature and the behaviour of sink mechanisms. Hence the diurnal patterns are not as well correlated to the source activities, although these are the same – home heating and vehicles.

Factors of potential importance to variations in human exposure

The discussion above deals only with outdoor monitoring data, while the people studied are exposed indoors as well (as most people spend most of their day indoors). New Zealand dwellings are generally not very well sealed and outdoor air pollution easily enters the indoors even when windows are closed. A small scale student project at Canterbury University (Simon Kingham supervisor) demonstrated that there was little difference between indoor and outdoor air pollution levels, except that houses with open fires had higher indoor levels of PM_{10} than the outdoor levels (at the same time and site) due to smoke from the fireplace. Higher indoor levels of NO_2 can also be expected when unflued gas heaters are used (e.g. Hahn et al. 2003).

Another factor of importance is whether people spend their time at home or in other parts of town. The most vulnerable groups for air pollution health effects are children and the elderly, and it is likely that they spend on average more time indoors at home, or near the home, than the adult age groups. As seen in the figures above, the hours of highest exposure are generally in the evening when most people are likely to be at home. In the winter, higher than summer values occur in the hours before and after midnight when it is most likely that people will be at home.

A1.2.2 Results - mortality

Dose-response for different air pollutants

The time-series analysis using Poisson multiple regression initially included daily data for all three air pollutants and all climate variables available. It was clear that there was a strong correlation between the measured concentrations of the three air pollutants, which can be seen in the three figures of hourly variations of the pollutant levels above.

Such multi-collinearity between different exposure variables obscures the analysis. When all three air pollutants variables are used together in the model, the effects of CO and NO_2 on daily non-external mortality were found to be statistically non-significant (lag = 0 days). Separate model fittings with each pollutant showed that the association between daily NO_2 and daily mortality was not statistically significant and between CO and mortality, the association was significant only at 10% level. The strongest association between daily air pollution level and daily mortality was found for PM_{10}.

The results from the model fittings are expressed in the form of the coefficients in the equation indicating the increase of mortality per μg m^{-3} of pollutant; e.g. a coefficient of 0.001 means a 0.1% increase of mortality for every μg m^{-3}, or a 1% increase for every 10 μg m^{-3}, which is how these results are usually expressed.

Response 65+ 1988-1999		
Variable	**Coefficient**	**(95% CI)**
Hot	**0.0202**	**(0.0054 - 0.0350)**
Cold	0.0026	(-0.0023 - 0.0076)
Rh	0.0010	(-0.0001 - 0.0021)
PM_{10}	**0.0014**	**(0.0002 - 0.0025)**
CO	-0.0009	(-0.0171 - 0.0154)
NO_2	-0.0008	(-0.0020 - 0.0004)

Table A1-1. Dose-response calculations for the 12-year period 1988-1999 with all variables (bold numbers are statistically significant, p< 0.05).

It was suspected that a combined effect of the different pollutants may exist and that an "air quality index", based on a combination of the measurements of the three pollutants PM_{10}, CO and NO_2 (eg Kjellstrom et al. 2002), may be a better indicator of hazardous air pollution exposure. However, none of the different combined measures of the pollutants was more strongly associated with mortality than PM_{10} on its own. Due to a lack of significant association of daily CO and NO_2 levels with daily mortality in this population, their relationship with mortality was not further pursued in this Christchurch analysis, and the epidemiological studies have only focussed on the relationship of PM_{10} with mortality.

The time-series analysis of the whole study period has given evidence of a relationship between daily PM_{10} and daily mortality similar to what has been reported from other cities. This relationship is statistically significant and if expressed as a linear function (1.4% increase of total daily mortality for each 10 μg m^{-3} increase of daily average PM_{10}.; confidence interval 0.2 – 2.5%) it is very similar to what has been reported in overseas studies (usually a 1% increase; see chapter 4.2.2). The relationship is displayed in Figure A1.4, which shows it may not be linear across the full range of exposures. There may be some suggestion of a steeper rate of mortality increase at lower PM_{10} levels (below 50 μg m^{-3}) than at higher PM_{10} levels. Almost all the days in the higher category (> 50 μg m^{-3}) are winter days.

In this presentation of the data the effect of climate variables is already adjusted for and the specific effect of PM_{10} is expressed.

Results, dose-response for season and climate factors

In Christchurch there are now sufficient data points to make the analysis more precise by using additional variables for season etc.

Using the same method as was employed in the 1988-1993 analysis by Hales et al (2000b), the dose-response coefficients have been calculated for PM_{10}, Relative Humidity, Hot (maximum temperature above 26.9 degrees C), and Cold (minimum temperature below 14.3 degrees C). In order to focus on the high-risk group, only the data for the age group above 65 years of age are shown here. The following section explores the age specific health risks in more detail showing that below age 65 there was no record of any measurable increase of mortality associated with air pollution exposure. The air pollution data used are from the new set corrected for the analytical method and with missing values imputed from a model, which provides more extensive observations than were available to Hales et al. Summer and winter coefficients have been created by including a season variable in the Poisson multiple regression analysis.

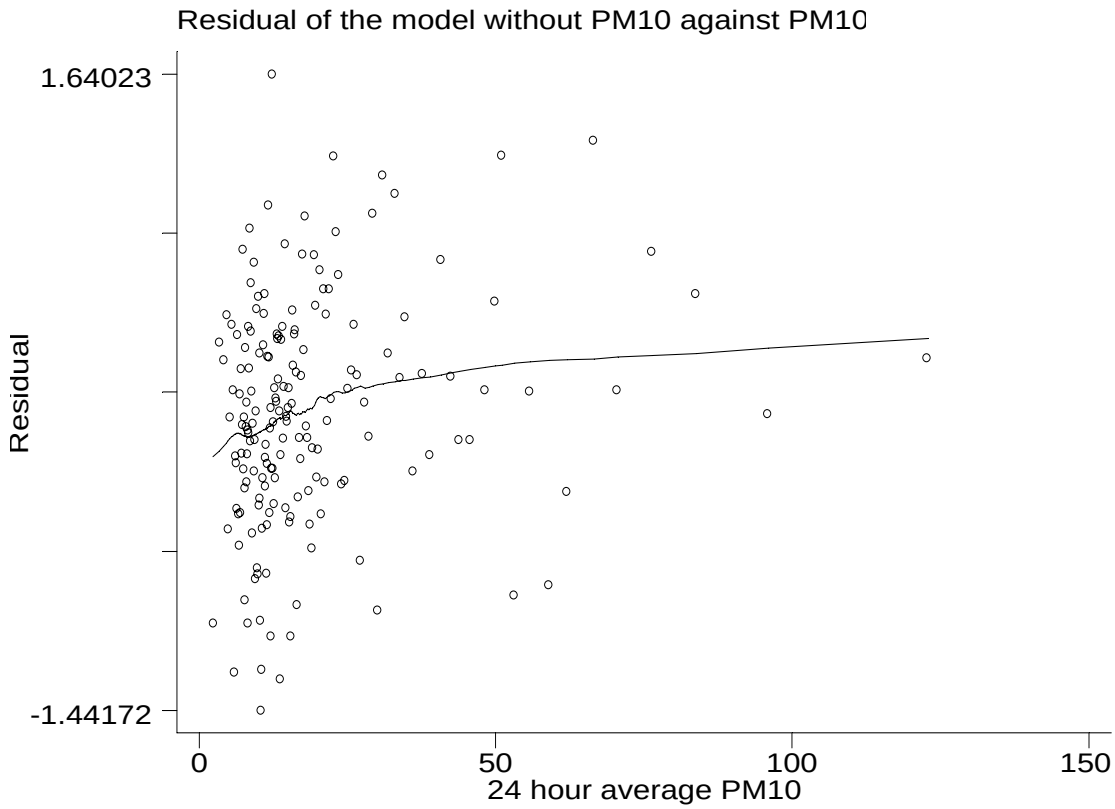


Figure A1-4. LOWESS curve for daily mortality in relation to daily PM_{10}

Tables A1-4 to A1-6 show the results for the full 12-year period and two six-year periods. The first 12-year table (Table A1-4) shows a 0.9% increase of mortality per 10 μg m^{-3} PM_{10} at lag 0 in the winter, and a 3.9% increase in the summer at lag 0. Effects are also seen at lag 1 and lag 2. The HOT variable also has a significant effect of 2% increased mortality per degree C above 27 degrees. Relative Humidity has an effect in the summer only, which is logical, as the combination of high temperature and high humidity creates additional heat stress.

Table A1-5 showing the first 6-year period still shows effects of Hot and the strongest winter PM_{10} effect at lag 1, which is similar to the data in Hales et al (2000b). However, the much stronger effect in the summer was not identified by Hales et al (2000b). Table A1-6 with 1994 to 1999 data shows only a consistent effect of relative humidity and limited effects of PM_{10}.

Notes for Tables A1-4 to A1-6:

1. Figures are given in four decimal places as the results are interpreted in % changes with 10 unit changes in the variable.
2. Bold figures are significant at 5% level of significance.
3. Italic figures are significant at 10% level of significance.
4. All Hot, Cold and Rh were used in the model in order to make comparison possible even though those variables were not significant.
5. Hot and Cold variables were created based on 'all years' model.
6. Lag x means only the lag effect of PM_{10}. Weather data were of the same day for all models.
7. Models do not account for population change over the time period in Christchurch.

Response 65+ 1988-1999			
Variable	Lag0	Lag1	Lag2
Hot	**0.0214**	**0.0227**	**0.0231**
Cold	0.0025	*0.0038*	**0.0047**
Rh (W)	-0.0003	-0.0003	-0.0002
Rh (S)	**0.0018**	**0.0017**	**0.0017**
PM_{10} (W)	**0.0009**	**0.0007**	**0.0009**
PM_{10} (S)	**0.0039**	**0.0041**	**0.0044**
	Interaction effect	Interaction effect	Interaction effect

Table A1-2. Dose-response calculations for the 12-year period 1988-1999.

Response 65+ 1988-1993			
Variable	Lag0	Lag1	Lag2
Hot	**0.0414**	**0.0403**	**0.0409**
Cold	0.0044	0.0040	*0.0058*
Rh (W)	-0.0001	0.0000	0.0001
Rh (S)	0.0006	0.0005	0.0005
PM_{10} (W)	0.0006	**0.0016**	**0.0013**
PM_{10} (S)	**0.0057**	**0.0043**	**0.0053**
	Interaction effect	No interaction	Interaction at 10%

Table A1-3. Dose-response calculations for the 6-year period 1988-1993.

Response 65+ 1994-1999			
Variable	Lag0	Lag1	Lag2
Hot	0.0129	0.0144	0.0147
Cold	0.0010	0.0036	0.0036
Rh (W)	-0.0006	-0.0008	-0.0007
Rh (S)	**0.0026**	**0.0026**	**0.0025**
PM_{10} (W)	*0.0009*	0.0000	0.0005
PM_{10} (S)	0.0025	**0.0033**	*0.0030*
	No interaction	Interaction at 10%	No interaction

Table A1-4. Dose-response calculations for the 6-year period 1994-1999.

Preliminary analysis (see chapter 9) shows that PM_{10} and CO have not decreased over time, while there is a decreasing tendency for SO_2 and an increasing tendency for NO_2. Other factors, such as decreasing trends in tobacco smoking and changes in health services access would be worth exploring, even though 6 years is a short time period for there to be sufficient changes to have a noticeable impact on the size of the air pollution effect.

An important observation from the Christchurch pilot study is that the summer PM_{10} appears to have greater association with daily mortality than the winter PM_{10} for the same increment of PM_{10}. If this is confirmed by further studies, it could be explained by different dose-response relationships for different particle sources.

Age, sex, ethnic group, socio-economic status specific response rates

To get a better picture of the importance of age, a table and some figures were prepared showing the age specific distribution of mortality in Christchurch. Table A1-7 shows total number of deaths by age group from 1988-1997. About 45% of deaths occur above age 80, 40% between age 65 and 79, and the remaining 15% below age 65. The number of deaths among the younger age groups is so small that an air pollution effect analysis has very limited statistical power.

As mentioned earlier in the older age groups the most pronounced seasonal mortality variations are seen, with higher mortality in the winter (Figure A1-5).

Age group	Total mortality	Age-specific mortality rate per 1,000 of the total population
<1	350	7.6
1-14	90	0.14
15-44	768	0.46
45-64	3869	5.5
65-79	11530	32.5
80+	12473	97.8
Total	12473	144.0

Table A1-5. Number of deaths in different age groups.

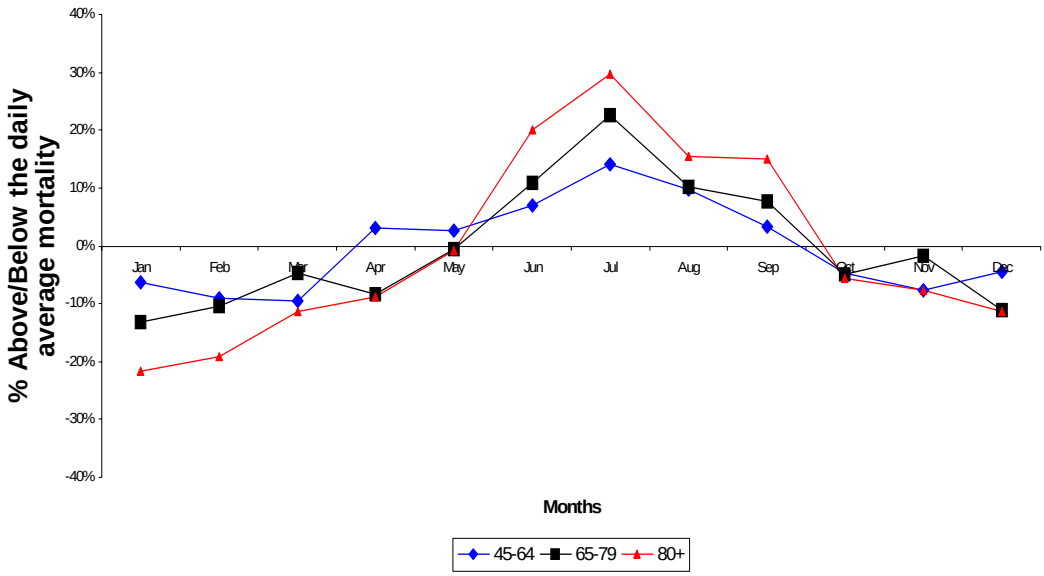


Figure A1-5. Seasonal variation of daily mortality for age groups above 45.

Age –related effects among the elderly

Multiple regression time-series analysis according to the APHEA protocol was used to analyse the associations between daily mortality, climate and air pollution, for different age groups (0-1, 2-64, 65-79, and 80+) and two seasons ("winter" = May – August; "summer" = September – April). There were no statistically significant effects in the age groups < 65 years. Only the results of the age groups >= 65 years are shown in detail here. The environment variables that were significantly associated with non-external mortality in at least some of the older age groups were minimum and maximum temperature, relative humidity and PM_{10}. (Table A1-7) In the winter the correlation between minimum temperature

and PM_{10} was strong and multiple regression analysis with both variables included negated the effect of the other variables.

Variable	Age > 65	65-79	> 80	All ages
Hot	**2.1***	**2.8***	1.4	**2.3***
Cold	- 0.2	0.2	**- 0.8***	0.4
Rh (w)	- 0.03	0.02	**- 0.05***	-0.02
Rh (s)	**0.18***	0.19	0.17	**0.13***
PM_{10} (w)	**0.86***	**1.3***	0.37	0.63
PM_{10} (s)	**3.9***	2.9	**4.7***	**3.5***
*Significant at 5% level				

Table A1-6. Dose-response coefficients for different age groups.

Table A1-8 shows the percentage increase in daily non-external mortality for 10 μg m^{-3} in PM_{10}, 1°C increase in daily maximum temperature above 26.9°C (variable Hot), 1°C decrease in daily minimum temperature below 14.3°C (variable Cold) and 1% increase in relative humidity for different age groups. There is a 0.29% increase of mortality per 10 μg m^{-3} PM_{10} in the younger elderly age group 65-79 whereas it is 4.7% increase in the age group 80+ in the summer. The effects on mortality of minimum temperature or PM_{10} in the winter is higher in the age group 65+ than in the whole population, and the effect is particularly high in the age group 80+. However, the summer mortality effect of heat is the strongest in the younger elderly (age 65-79), while the effect of PM_{10} is the highest in the 80+ group. This finding has important consequences for the calculation of "burden of disease" associated with air pollution and heat. There are also differences in the effects on cardiovascular and respiratory mortality, Table A1-9. Only the combined age group >65 years is shown due to the limited number of deaths in each group rendering all the risk coefficients non-significant.

Variable	> 65, circulatory	> 65, respiratory
Hot	**2.3***	- 2.1
Cold	**- 0.75***	- 0.10
PM_{10} (w)	0.44	1.3
PM_{10} (s)	0.71	**13***
*Significant at 5% level		

Table A1-7. Dose-response coefficients for mortality in two disease types.

A1.2.3 Summary of findings

The new daily time-series analysis of mortality and morbidity in Christchurch has documented that health effects due to PM_{10} indeed occur in Christchurch after climate variables (those that are likely to affect mortality) and seasonality are taken into account. Probably, because of the relatively small population (compared with overseas studies) some of the apparent increases of mortality or hospital admissions were not statistically significant, but enough of the results are significant to conclude that serious health effects indeed occur.

The effects can be documented only for people over age 64 and there are apparent differences in the effects in the age group 65-79 compared with the age group 80 and over. The younger elderly appear more affected by heat than by PM_{10}, while for the older elderly it is the opposite.

The dose-response relationship for non-external mortality increase for the whole year and the whole population is very similar to what has been reported from other studies (approximately 1% increase of mortality per 10 μg m^{-3} PM_{10}). By the introduction of a season variable in the analysis a much steeper dose-response relationship in the summer period than in the winter period was shown, even though the winter PM_{10} levels are much higher. Further research and analysis of any differences in dose-response relationships for different PM sources is needed to clarify this issue.

A1.3 Spatial variation (Auckland case study)

A1.3.1 Scope of the studies

In order to establish dose-response relationships for "long-term" exposures, different approaches than the one described above are needed. It became clear when the research started that one method that could be applied to both Christchurch and Auckland was a spatial epidemiology approach using CAU-based air pollution and mortality data to calculate any associations between these variables accounting for any confounding by age, sex, ethnic group, socio-economic status, etc. at a CAU level. Data for this type of analysis was available from Auckland and the preliminary results are included here. A similar analysis for Christchurch will be carried out during the project period.

Estimating exposure to air pollution requires simplifications and assumptions and therefore has limitations. Measurements of ambient pollution are taken at a small number of sites and are seldom recorded continuously and moreover, urban populations are highly mobile, therefore assumptions have to be made to estimate personal exposures. There is also variation in pollutants over time and space due to factors such as meteorology, topography, and varying emission sources. Early ecological studies in the USA relied on pollutants measured for biweekly (Lave & Seskin 1972) or 3-monthly periods (Ozkaynak & Thurston 1987). Measuring instruments were often changed over time and across cities, and some had little reliability. It was necessary to assume the data for a single point was representative of a vast geographical area. For example no single monitoring site is representative of the Auckland region due to complex coastal environments, topography, and variations in climatic factors over relatively small areas (Scoggins et al. 2004). Recently, detailed understanding of complex urban air quality processes has been aided by the application of urban airshed models. These models account for spatial and temporal variations as well as differences in the reactivity of air pollutants and therefore can provide a detailed spatial picture of pollutant levels. Coupled with GIS techniques, these models may greatly improve exposure measurements to link with health (Cicero-Fernandez et al. 2001; English et al. 1999; Hoek, G. et al. 2001). In addition, GIS-based exposure maps, which may be used to quantify the number of people exposed to air pollution, can identify high exposure areas for policy developers and planners in a simple and realistic manner.

This section carries out a spatial analysis of long-term ambient air pollution levels and mortality within the Auckland region for the period 1996-1999 (Scoggins et al. 2004). Urban airshed modelling and GIS-based techniques to quantify long-term annual exposure to ambient air pollution levels and associated mortality.

The data analysis was undertaken at a census area unit (CAU) level. Census area units are areas defined for statistical purposes by Statistics New Zealand. CAUs are variable in size and population. A CAU typically contains 3,000 people and a maximum of approximately 8,000 people. The average size of CAUs is approximately 14 km^2 (in central urban areas the average size is approximately 2 km^2). There are 320 CAUs in the Auckland region (excluding offshore islands). Routinely collected health data is provided at the CAU scale. The surface areas of CAUs reflect the size of urban airshed model output grids (9 km^2).

A1.3.2 Methodology

Exposure variables

One hourly average NO_2 concentrations in micrograms per cubic metre ($\mu g\ m^{-3}$) for 1999 were produced by the CALGRID urban airshed model, used to simulate the urban air quality of Auckland. The model results are an extension of those presented by Gimson (2000). CALGRID is a photochemical air quality model, which simulates the transport, diffusion, deposition and chemistry of O_3 and its precursors (e.g. NO_x). CALGRID uses a diagnostic pre-processor, CALMET to produce surface and upper air meteorological fields. CALMET is driven solely by local meteorological data (such as wind speed & direction, temperature, humidity, cloud cover, and rainfall) from the Auckland region, which is obtained from the New Zealand National Climate Database (CLIDB). CALGRID is driven by meteorological output from CALMET, and emissions from the ARC emissions database (ARC 1998; Joynt et al. 2002).

The models were run on a 3 km grid that covered almost the entire Auckland region. The final grid had a total of 1296 grid cells (36 rows by 36 columns, grid cell size 9 km^2). All model output was in the standard NZMG coordinate system and was compared with corresponding concentrations at air quality monitoring sites around the Auckland region (Ministry for the Environment 2000) (Table A1-8).

Monitoring site	Type of Site	Correlation	IOA[1]
Khyber Pass		0.18 (n=8296)	0.502
Mt. Eden	Residential	0.65 (n=8636)	0.766
Musick Point	Remote	0.77 (n=6333)	0.803
Penrose	Industrial	0.60 (n=7988)	0.752

[1] Index of Agreement $IOA = 1 - \frac{\sum_{i=1}^{N}(P_i - O_i)^2}{\sum_{i=1}^{N}(|P_i - O_{mean}| + |O_i - O_{mean}|)^2}$ O = observed P = predicted or modelled

Table A1-8. Comparison of hourly NO_2 observed and modelled values at air quality monitoring sites in Auckland.

The urban airshed modelling carried out in Auckland gave an Index of Agreement (IOA) above 0.75 for NO_2 at most sites. This shows good model performance. Model results are averaged over 3km by 3km grid-cells, and are comparable with point observations only if the monitoring site is representative of the surrounding area. This is the case for Mount Eden, Musick Point and Penrose, but not for the Khyber Pass site where the local hourly values at the road side vary strongly due to the traffic flow variations and the local value does not reflect the average value in the grid square.

The NO_2 modelled concentrations were averaged over the whole year and annual average NO_2 was used as a long-term air pollution exposure indicator. Annual average NO_2 modelling concentrations were converted from point-based (X, Y coordinates) grid coverage into 3 km by 3 km polygon grid coverage in ARC INFO User 7.2.1 and ArcView GIS 3.2. Polygon grid coverage concentrations were converted to census area unit (CAU) concentrations. A CAU totally within a grid square was assigned the grid square value and for CAUs that overlapped more than one grid square an area-weighted average concentration was calculated (Scoggins et al. 2004).

A1.3.3 Potential confounding variables

Age, Sex, Ethnicity

The 1996 Census provided information by CAU for the Auckland region on resident population, sex, age (0-14, 15-64, 65+ years), and ethnicity (European, NZ Maori, Pacific peoples, Asian + Other).

Socio-economic status

The New Zealand Deprivation Index 1996 (NZDep96) was used as a summary measure of socio-economic status (Crampton et al. 2000). NZDep96 combines nine variables from the 1996 Census of Population and Dwellings and is a composite indicator of relative social and economic deprivation. NZDep96 was expressed in quintiles whereby 1 represents the least deprived 20% of CAUs, and 5 represents the most deprived 20% of areas (Spatial Analysis Facility 2001). Because NZDep96 is an area-based measure, it reflects the average socio-economic position of individuals in the area. But an advantage of this index is that it captures contextual or neighbourhood effects that may be difficult to separate from effects of individual-level socio-economic variables (Blakely & Pearce 2002).

Occupation

The New Zealand Statistical Standards for Occupation 2002 was obtained from Statistics New Zealand. The fraction of trades workers (Major Group 7) and plant and machine operators (Major Group 8) was calculated as a percentage of the adult population in each CAU to indicate potential exposure to pollutants within workplaces.

Smoking

The fraction of smokers (defined as one or more cigarettes per day) and ex-smokers was calculated for each CAU. This information was obtained from the 1996 Census.

Urban/Rural domicile

CAUs were classified into urban and rural domicile using the 1996 classification of urban and rural CAUs that was obtained from Statistics New Zealand. All rural CAUs (n=18 or 5.6%) were excluded from the analysis presented here.

Effect variables

The New Zealand Health Information Service provided mortality data, for the years 1996 to 1999. Cause of death was categorised according to the International Classification of Disease Version 9 (ICD-9 code), and is defined for non-external (ICD 1-799) and circulatory and respiratory (ICD 390-519) broad causes of deaths. External causes of mortality (ICD 800-999) were included only for the calculation in Table 7-9. The domicile code, which represents the deceased's usual residence address, is the same as the CAU code and was used to map mortality data to place of residence. It was also assumed ambient air pollution and mortality can be measured at place of residence, which follows overseas methodology (Aunan 1996; Huang & Batterman 2000).

While some studies suggest place of residence may not be closely associated with spatial and temporal patterns of exposure (Hoek et al. 2001), others conclude residence is one of the major personal determinants of exposure to outdoor air pollution (Kunzli & Tager 2000). Residence location is a particularly useful measure of exposure for long-term mean concentrations, whereas the short-term within and between personal variability in exposure may be substantial due to short-term variation in time-activity patters across diverse environments (Kunzli & Tager 2000).

A1.3.4 Analytical methods

Logistic regression was used to investigate how air pollution (annual NO_2) influences the probability of dying, while controlling for age (0-14, 15-64, 65+ years), sex, ethnicity (European, New Zealand Maori, Pacific peoples, Asian + Other), NZDep96 (quintiles, 1-5), smoking, and occupation. As mentioned before the rural CAUs were excluded. The dependent variable was the adjusted mortality rates in the population. A binomial model was applied because of the very small denominator populations in most cells. Separate logistic regression models were run in SAS 8.2 for non-external cause mortality and circulatory and respiratory mortality:

The same model was fitted for deaths due to external causes (ICD 800-999). This was done to check whether an association between air pollution and circulatory and respiratory mortality (and non-external cause mortality) was possibly due to uncontrolled confounding.

A1.3.5 Results

Key results are summarised in Tables A1-9 and A1-10. The lack of association between NO_2 and external cause mortality (Table A1-9) indicates that it is unlikely that some general residual confounding of mortality occurs.

Mortality	Unadjusted odds ratio (95% CI)	P-value
Non-external causes	1.016 (1.016-1.020)	<0.0001
Circulatory and respiratory	1.022 (1.019-1.025)	<0.0001
External causes	1.002 (0.994-1.010)	0.5653

Table A1-9. Unadjusted odds ratios in non-external, circulatory and respiratory, and external mortality associated per 1 µg m^{-3} increase in annual average NO_2.

Variables	Unadjusted odds ratio (95% CI)	P-value
Age, sex, ethnic, NZDep only	1.013 (1.011-1.015)	<0.0001
Age, sex, ethnic, NZDep, smoking, occupation	1.010 (1.007-1.013)	<0.0001

Table A1-10. Adjusted odds ratios (95% Confidence Intervals in non-external mortality associated per 1 µg m^{-3} increase in annual average NO_2.

These two tables indicate that annual air pollution levels in Auckland (here represented by NO_2 levels) increase non-external mortality, even when all the available potentially confounding variables have been adjusted for. The estimated dose-response relationship for mortality at 10% (confidence interval 7 – 13%) per 10 ug/m^3 NO_2 appears higher than other long-term exposure risk estimates, and needs confirmation in further research. The annual PM_{10} and NO_2 levels in Auckland are similar.

A1.4 Census analysis

This study is still in progress and will be completed by the end of 2005 and no results can be included in this report. It will provide a new way of linking individual air pollution exposure estimates with individual death records. The record linkage of census and mortality data makes it possible to assign estimates of annual air quality estimates for each CAU (in the whole of New Zealand) to the people dying in that CAU. In addition, the census data contain information about whether the person dying was a smoker or non-smoker and to what income or NZDep category the person belonged. This makes it possible to account for these potential confounding factors in the epidemiological analysis at an individual level. The study has great importance for analysis in areas where no air quality monitoring has taken place but air pollution exposure estimates are given by the air quality model. Results will be provided in the final report.

A1.5 Dose response functions for different seasons

The results presented above for Christchurch show that the increase of daily mortality over the whole year is about 1.4% per 10 µg m^{-3} PM_{10} (confidence interval 0.2%-2.5%) (Table A1-1). This is similar to what has been reported in previous research in Christchurch and in studies from other cities. However, when the analysis was repeated by season (winter and summer), the effect of a certain level of PM_{10} appears much less in the winter (0.9% per 10 µg m^{-3}) than in the summer (3.9%) (Table A1-2). The confidence intervals are large and there is uncertainty about this difference. A possible reason for any difference is that PM from different sources could have different effects on acute mortality. This could be a consequence of the composition of the PM_{10} (since size and chemical composition varies depending on the source), or may reflect the effects of other pollutants that are present in the emissions. It could also be due to an interaction between season and PM_{10} exposure due to some yet unspecified factor. A stronger dose-response relationship in the summer has been reported recently also from an analysis of 100 cities in the USA (Peng et al. 2004).

As pointed out earlier of the hundreds of studies of air pollution and mortality none has specifically compared the effect of motor vehicle smoke, industrial smoke and wood smoke. The Santa Clara County study in California (Fairley, 1999) compared different seasons. The autumn and winter dose-response coefficients were lower than in the summer, but the difference between seasons was not statistically significant. However, the proportion of winter smoke emanating from wood burning was much lower than in Christchurch. Another report apportioned the contribution to $PM_{2.5}$ from different sources in six US cities could identify motor vehicles, coal burning, and soil dust as sources, but no attempt was made to identify wood smoke sources (Laden et al. 2000). A daily time-series analysis estimated that the increase of daily mortality caused by a 10 µg m^{-3} increase of $PM_{2.5}$ was 3.4% for

motor vehicle sourced particles and 1.1% for coal burning sources. This indicates that vehicle sourced particles may create a higher dose-response coefficient than other particulate. Unfortunately wood smoke sources were not identified. The only study found where daily mortality effects of wood burning and other sources were compared (Mar et al, 2003) did not produce statistically significant differences. As the policy implications of different dose-response relationships for PM_{10} from vehicles and wood burning are great, this issue is important to study further in Christchurch and other places where both sources are significant, but at this stage the prudent approach is to use the same relationship in the current health impact assessment.

An updated analysis (Pope et al. 2002) of one of the US studies provides important additional information on the long-term dose-response relationship for particles and mortality (Table A1-11). This study followed up 500,000 US adults for 16 years and gave the most detailed estimates available so far.

Mortality cause	1979-1983	1999-2000	Average (combined)
All causes	1.04 (1.01 – 1.08)	1.06 (1.02 – 1.10)	1.06 (1.02 – 1.11)
Cardio-pulmonary	1.06 (1.02 – 1.20)	1.08 (1.02 – 1.14)	1.09 (1.03 – 1.16)
Lung cancer	1.08 (1.01 – 1.16)	1.13 (1.04 – 1.22)	1.14 (1.04 – 1.23)
All other causes	1.01 (0.97 – 1.05)	1.01 (0.97 – 1.06)	1.01 (0.95 – 1.06)

Table A1-9. Relative risk for mortality associated with a 10 μg m^{-3} increase of annual $PM_{2.5}$, adjusted for age, sex, race and a number of other factors (including smoking). (95% confidence interval in brackets) (Pope et al. 2002).

Assuming that the $PM_{2.5}$ to PM_{10} ratio is approximately 0.8 in Christchurch (based on co-incident measurements made in Christchurch by Environment Canterbury) and using the figures in Table A1-11 the best estimate for all cause mortality, cardio-pulmonary mortality and lung cancer mortality increase would be 4.8%, 7.2% and 11% respectively per 10 μg m^{-3} PM_{10} (confidence intervals are +- ¾ of these values). The numbers for all cause mortality appears higher than that used by Kunzli et al. (2000), but the confidence interval is large (2 – 11 %; Table A1-11). The analysis of the dose-response relationship for annual NO_2 and non-external mortality in Auckland (see chapter A1.3) showed a confidence interval in the same range (7 – 13%) as the one presented in Table A1-11.

A1.6 Summary

It appears from the data presented in previous chapters and the discussion above that the 4.3% increase of mortality for people over age 30 used by Kunzli et al (2000) for all sources of PM_{10} is not necessarily the best available estimate of the dose-response relationship for the purposes of health risk assessments in New Zealand. Taking the recent study by Pope et al. (2002) and the HAPiNZ study by Scoggins et al. (2004) into account indicates that the true figure for non-external mortality increase could be in the range 4 – 7 % for each 10 μg m^{-3} increase of PM_{10}. However, more evidence is needed before modifications of the dose-response relationships are applied in the overall Health Impact Analysis. The results of the time-series study (Appendix A1.2) and the GIS study (Appendix A1.3) indicate that the overseas dose-response relationships appear valid also in New Zealand.

Appendix 2 – Christchurch exposure analysis

This appendix covers all the details on the Christchurch air pollution exposure analysis summarised in chapter 12 of the report.

A2.1 Methods

A2.1.1 Emissions

Environment Canterbury emissions inventory

Note: Much of the information following is taken from Environment Canterbury documents.

Internationally, regulatory authorities use emission inventories as an air quality management tool. An inventory identifies the major sources discharging contaminants to the air, provides estimates of the quantities emitted and determines relative contributions to total emissions. Emissions to the air in Christchurch are reassessed on a regular basis to monitor trends over time and to determine changes in the relative contribution of sources to emissions (Scott & Gunatilaka 2003). Previous inventories completed for Christchurch include, NIWA (1998), Wilton (2001a), and Scott & Gunatilaka (2003) compiled the most recent inventory for Environment Canterbury. The inventories present emissions to the air for a "typical winter's day" for an area within the Christchurch territorial boundary.

The emissions inventories contain raw emissions data in kilograms or tonnes. The contaminants investigated include PM_{10}, $PM_{2.5}$, CO, CO_2, NO_x, and SO_x. The emissions are divided into five time periods: 6am-10am, 10am-4pm, 4pm-10pm, 10pm-6am and 24hr total emissions. The inventories focus mainly on emissions from domestic home heating, vehicle and industrial emissions however, other sources such as aviation, rail and lawn mowing were included in the 2002 inventory. Total emissions from all sources (except natural sources) for each time period are also given.

A few changes have been made to the study areas for the three inventory years to enable inter-comparisons and for trends to be established. The main study area, Metropolitan Christchurch, encompassed the entire Christchurch territorial boundary area and comprised the main portion of the Christchurch airshed, as defined by Sturman et al. (2002). Metropolitan Christchurch was separated into three different sub-areas. These were Inner Christchurch, Suburban Christchurch and Outer Christchurch as illustrated in Figure A2.1. Inner Christchurch was identical to the Inner Suburb area as defined in the 1999 inventory and includes 44 of the 2001 census area units. Similarly, Suburban Christchurch was the same as the previously defined 25 Suburb area. It includes all suburbs within Inner Christchurch and an additional 42 census area units. Outer Christchurch includes all census area units outside Suburban Christchurch and within the Christchurch territorial boundary (Scott & Gunatilaka 2003).

Domestic home heating

For the 2002 inventory domestic home heating emissions were estimated by applying emission factors to home heating activity data. The activity data collected included number, type and age of appliance, and type and quantity of fuel consumed. Home heating activity and fuel use data were primarily collected by household survey (Lamb 2003). The survey, conducted by telephone, compiled information on methods of home heating, volume and types of fuel used, fuel sources and frequency of appliance operation. Householders who used solid fuel heating methods were also invited to participate in a diary panel. Burning behaviour was monitored by participants on a day-to-day basis for a two-week period. The 2002 inventory integrated all quantifiable appliance types and relied on householders to provide accurate burner type and age data. Wood burners were classified by installation date into the pre-1992, 1992-2000, 2001-2002 burner categories. Burners were grouped into these different age categories to take into account changes in allowable emission limits over time. In addition, Christchurch City Council permit information was used to collect pellet burner installation data. Emissions were calculated based on the fuel used by each appliance group and the corresponding emission factor. Emission factors were applied to the activity data to estimate the quantity of emissions from each source

(Equation A2.1). Emission factors are representative values, which provide a measure of contaminant discharge for a specific type of activity and fuel consumption. Factors are expressed as the weight of pollutant divided by a unit weight, volume, distance or duration of activity emitting the pollutant (USEPA 1995) (Table A2-1) (Scott & Gunatilaka 2003).

$E = F \times EF/1000$ Equation A2.1.

Where: E = contaminant emission (kg)
F = fuel used in 24 hours (kg)
EF = emission factor for a particular contaminant, activity and fuel (g/kg)

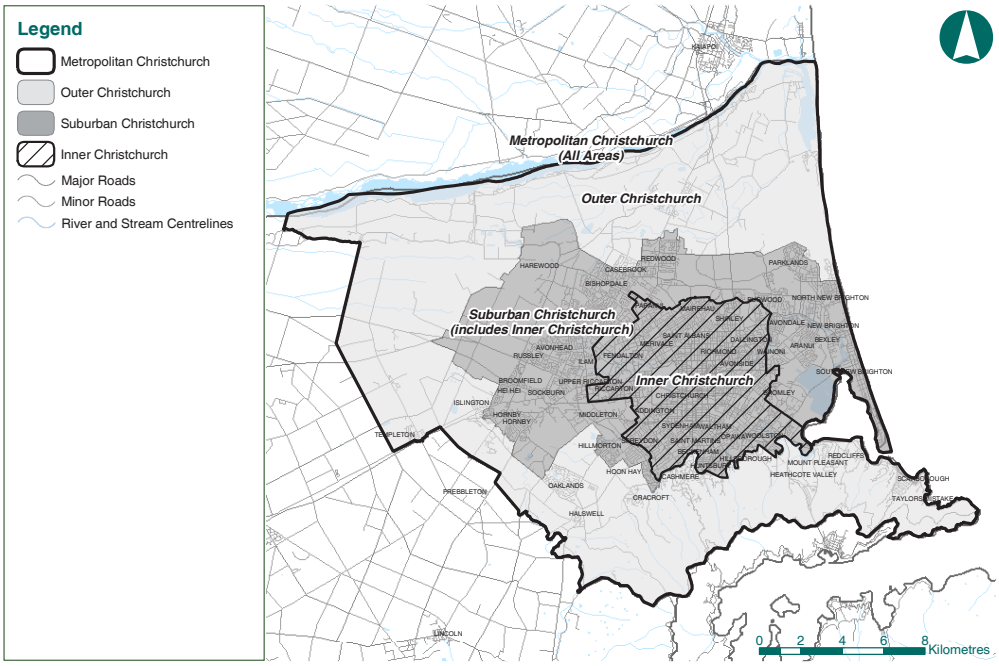


Figure A2-1. Christchurch emission inventory study area 2002 (Scott & Gunatilaka 2003).

Appliance	PM_{10} (g/kg)	$PM_{2.5}$ (g/kg)	CO (g/kg)	NO_X (g/kg)	SO_2 (g/kg)	CO_2 (g/kg)
Open fire (wood)	9	8	68	0.4	0.2	1717
Open fire (coal)	21	20	70	4.1	5.1	2721
Pre-1992 wood burner	12	11	98	1	0.2	1726
1992-2000 wood burner	5	5	63	0.5	0.2	1802
2001-2002 wood burner	3	3	34	0.5	0.2	1855
Multi-fuel burner (wood)	12	11	98	1	0.2	1726
Multi-fuel burner (coal)	19	17	110	1.6	1.1	2170
Pellet burner	1.6	1.5	15	5.2	0.2	1476
Oil burner	0.3	0.2	0.5	2.0	4.0	3200
Gas burner	0.3	0.3	0.2	1.5	0.0	2500

Table A2-1. Domestic home heating emission factors (Scott & Gunatilaka 2003).

Motor vehicles

Emissions from the motor vehicle sector were estimated using activity data in the form of vehicle kilometres travelled (VKT) for various congestion conditions or levels of service. Fleet-weighted emission factors obtained from the New Zealand Traffic Emission Rates (NZTER) database (Ministry of Transport 1998) were applied to the VKT data obtained from the Christchurch Transport Study (CTS) model to calculate emissions (Equation A2.2). Calculated emissions include exhaust emissions from on-road motor vehicles and exclude off-road vehicles, paved road dust, and brake and tyre wear. The

Christchurch Transport Study (CTS) model, developed by the Canterbury Regional Council estimated VKT for the various study areas. The CTS model, a "vehicle driver" trip model, represents trips made by motorised vehicles in Metropolitan Christchurch within defined timeframes. It is acknowledged that the 1998 NZTER may need updating to be used in the studies such as that undertaken here, and updated emissions data from the Ministry of Transport will be used as soon as it is available.

Trip generation and distribution patterns were established through household interview surveys. These were applied to land-use data for the 1996, 1999 and 2002 inventory years to determine travel demand. Travel demand, included in the CTS model as a series of origin-destination trip matrices, indicate where people and goods come from and go to in Christchurch. The data are assigned to the road network using shortest path calculations to estimate the most likely routes people will use when travelling from origin to destination. The CTS model provided traffic flows representative of five different time periods: morning peak (8am-9am), inter-peak (12pm-1pm), afternoon peak (5pm-6pm) and evening peak (10pm-11pm).

The final result was a "loaded" road network which classified each road as either central urban or suburban and included information about the length of each link, the volume of traffic on each link, the resultant link travel speed and the delays incurred at each controlled intersection. VKTs for each road link were then calculated by multiplying the peak hourly volume of traffic with the link distance giving VKTs for the morning peak (AM), inter-peak (IP), afternoon peak (PM), and evening peak (EV) categories. The data were aggregated to represent the time periods selected for the inventory and validated by using hourly traffic counts. Prior to applying emission factors to the VKT data a level of service (LoS) value/driving condition category was allocated to each VKT estimate. The driving condition groupings used were:

- Free flow: Little or no vehicle impedance with warm running, LoS level A/B, volume/capacity<35%
- Interrupted: Moderate vehicle interaction with warm running, LoS level C/D, volume/capacity >35% and <70%
- Congested: Severe vehicle interaction with warm running, LoS level E/F, volume/capacity>70%

The emission factors used in the assessment were obtained from the NZ-TER database. The emission rates were developed through two vehicle emission testing programs designed to provide real measures of emission performance for New Zealand vehicles under New Zealand conditions (Ministry of Transport 1998). The testing was chassis dyno-based to provide a good indication of emissions produced under actual driving conditions. Emission rates were provided in several forms including single vehicle emission rates, corridor flow composite emission rates and drive cycle composite emission rates. Corridor flow composite emission rates from the NZ-TER were used to determine motor vehicle emissions in Christchurch. The New Zealand national motor vehicle fleet data based on VKT splits were input into the database. Two different road classifications were used in the assessment. The road types selected, as classified in the VFEM, were:

- Central urban: Routes with a combination of one-block links, essentially all with signalised intersections, an approximate minimum of six per kilometre; a maximum speed limit of 50 km/h.
- Suburban: A mixed route representing daily commuter/local traffic (outside the CBD) with a high frequency of intersections, mainly uncontrolled and relying upon queuing and gap acceptance to change or join traffic flows; a maximum speed limit of 50 km/h.

Tables A2-2 and A2-3 outline the fleet weighted average emission factors derived for the defined fleet composition, driving conditions (LoS – level of service) and road types (A through to F as per Land Transport New Zealand definitions) in Christchurch (Scott & Gunatilaka 2003).

$$E = EF \times VKT/1000 \qquad \text{Equation A2.2}$$

Where: E = contaminant emission (kg)
VKT = vehicle kilometres travelled (km)
EF = emission factor for each contaminant per LoS and road type (g/km)

The NZ-TER database only provided particle emission factors in the form of total particulate matter (PM). Particle emissions may be separated into the PM_{10} and $PM_{2.5}$ size fractions when the proportion of diesel and petrol derived emissions are known. This study did not provide data regarding the

emissions from diesel and petrol vehicles, and accurate fractionation could not be obtained. The PM results have been presented in this report both as PM_{10} and $PM_{2.5}$. This assumes a worst-case scenario (from a health perspective) where all PM is in the $PM_{2.5}$ (and thus PM_{10}) size fractions.

LoS	PM	CO	NO_x	SO_x	CO_2
E-F	0.29	30.93	3.70	0.40	732.47
C-D	0.22	20.47	3.19	0.32	561.60
A-B	0.18	15.65	2.59	0.26	412.93

Table A2-2. Emission factors for Central Urban in g/km (Scott & Gunatilaka 2003).

LoS	PM	CO	NO_x	SO_x	CO_2
E-F	0.21	20.41	2.71	0.28	478.63
C-D	0.17	16.03	2.59	0.24	409.28
A-B	0.15	12.62	2.41	0.22	368.22

Table A2-3. Emission factors for Suburban g/km (Scott & Gunatilaka 2003).

Industry

Emissions from the industrial and commercial sector were estimated by applying emission factors to point source-based activity data. The 2002 assessment includes all industrial and commercial activities currently consented to discharge contaminants (PM_{10}, $PM_{2.5}$, CO, CO_2, NO_x, and SO_x) to air in Metropolitan Christchurch. Industrial activities that discharge volatile organic compounds were excluded. The approach adopted previously where activity data were collated on the basis of process rather than by primary industrial type (Wilton 2001a), was also implemented in 2002.

Three methods were employed to collect activity data for the 2002 emission inventory. The primary data collection method comprised a survey of all 686 industrial and commercial activities consented to discharge contaminants to air. Consent and compliance information contained in resource consent files held by Environment Canterbury were used to collect activity data for industries that did not respond to the survey, and were not included in the previous inventory assessment. For the remaining premises, i.e. those that did not respond to the survey but were included in the previous inventory assessment, information from the previous inventory was used. Industrial survey data were collected for industrial and commercial activities on a monthly basis to establish temporal variations. Details provided in the survey regarding hours and days of operation allowed daily wintertime emissions to be assessed. Emissions at other times of the year are not presented in this report.

The methodology adopted for the previous inventory assessment is outlined in Wilton (2001a). In general, emissions were based on activity data gleaned from resource consent files. Fuel quantity data were not always available and estimates were sometimes required. These estimates were based on maximum allowable fuel or material use data and assumptions were required regarding the actual proportion of fuel used in practice. A higher level of uncertainty is associated with these data as actual activity rates are unknown and the values used are dated.

The emission factors used in the 2002 inventory are identical to those used by Wilton (2001c). Although preference was given to emission testing data obtained locally, emission factors sourced from the USEPA's AP-42 database were used in most instances due to the lack of robust local data. Overseas emission factors, however, do not necessarily provide a good representation of emissions in New Zealand. Process, fuel, control systems and technology used in New Zealand may vary significantly from that used in the United States. The emission factors indicated in Wilton (2001c) are adequate however and are listed in Table A2-4 (Scott & Gunatilaka 2003).

$E = F\,x$ Equation A2.3

Where: E = contaminant emission (kg)
F = fuel used in 24 hours (tonne)
EF = emission factor for a particular contaminant, activity and fuel (kg/t)

Industrial Process	Unit (U)	PM_{10}	$PM_{2.5}$	CO	NO_x	SO_x	CO_2
		kg/U	kg/U	kg/U	kg/U	kg/U	kg/U
Abrasive blasting - fabric filters	Tonne	0.69					
Sand blasting	Tonne	13	1.3				
Coal fired boilers							
Chain grate	Tonne	1.8	0.666	3	3.75	18	2400
Spreader and Vekos	Tonne	3.1	1.085	2.5	5	18	2400
Overfeed stokers	Tonne	3.1	1.147	3	3.8	18	2400
Underfeed stokers	Tonne	3.1	1.891	5.5	4.75	13.5	2400
Low Ram	Tonne	1.8	0.63	3	5	18	2400
Diesel boiler	Tonne	0.47	0.112	0.67	3.24	10.5	3193
Foundry - Aluminium only	Tonne	0.95					
Foundry - non ferrous	Tonne	89.6	79.744				
Foundry - steel - open hearth	Tonne	8.75	6.33				
Foundry - steel - electric induction	Tonne	0.05	0.0445				
Incinerators	Tonne	1.5145	0.999	1.48	1.78	1.09	
Light fuel oil (LFO) boilers	Tonne	1.1	0.715	0.67	6.3	3.87	3193.
LPG boilers	Tonne	0.058	0.033	0.705	2.6	0.007	2885
Meat smoking	Tonne	26.5	26.5				
Pneumatic conveying	m3	0.0011					
Quarrying	Tonne	0.00064	0.00018				
Seed cleaning	Tonne	0.0094	0.0034				
Seed handling	Tonne	0.0039	0.00026				
Waste oil burners	Tonne	1.4		0.7	2.7	0.02	3105
Wood fired burners	Tonne	3.25	2.73	6.8	0.75	0.75	1069.
Diesel Generator (kW)							
40	kg/hr	0		0.1	0.6	0.1	22.4
140	kg/hr	0.1		0.5	2.1	0.3	78.3
170	kg/hr	0.2		0.6	2.6	0.3	95.1
220	kg/hr	0.2		0.7	3.3	0.4	123.1
300	kg/hr	0.3		1	4.5	0.6	167.8
340	kg/hr	0.4		1.1	5.1	0.7	190.2
1000	kg/hr	0.28		2.68	11.67	1.38	564
150kVa=120kW	kg/hr	0.1		0.4	1.8	0.2	67.1
175kVa=140kW	kg/hr	0.1		0.5	2.1	0.3	78.3
220kVa=174kW	kg/hr	0.2		0.6	2.6	0.3	97.3
640	kg/hr	0.18		1.71	7.47	0.88	361.1
Aggregate handling and storage							
sand	kg/t	7.92E-05	2.4E-05				
cement	kg/t	0.000462	0.00014				
coal	kg/t	0.000159	5E-05				

Table A2-4. Industrial and commercial emission factors (Scott & Gunatilaka 2003).

Notes on Table A2-4:

- The AP-42 contains emission factors for filterable and condensable particulate matter. For example, combustion sources such as furnaces and boilers discharge both filterable and condensable particulate. This is incorporated into their emission factors**.** Fugitive dust from storage piles and unpaved roads on the other hand only consists of filterable particulate. The emission factors for these activities reflect this. The emission factors do not, however, allow for the secondary formation of particulate, that is, particulate formed during atmospheric reactions of gaseous SO_2 and NO_x.
- The emission factor for pneumatic conveying is 0.11kg of PM_{10} per 100 m^3 conveyed, that is, 0.0011kg per 1m^3 of material conveyed.

- For the aggregate handling and storage emission factor the following were assumed:
    - Wind speed=2 m/s (based on Christchurch meteorological data)
    - Moisture content for sand=7.4%, cement=2.1% and coal=4.5% (USEPA 1995)
- Ash content of waste oil is 20%
- Emissions from cement handling with use of baghouse filters are not detectable (in accordance with AP-42)
- Asphalt plant – hot mix with venturi scrubber – no significant discharge according to AP-42
- Foundry discharges for melting of bronze and brass are from a rotary furnace with uncontrolled emissions
- Textile printing emission factors based on roller method
- The following industries/activities were excluded owing to lack of available emission factors and/or suitable activity data:
    - Debonding brake shoes (three industries)
    - Composting (two locations)
    - Burwood

A2.1.2 Complications with yearly comparisons

The 1996 (NIWA 1998) and 1999 (Wilton 2001a) inventories were not directly comparable to 2002 in their published forms due to differences in emission factors, fuel data and methodologies. The data were recalculated (where possible) for the 2002 inventory to enable comparisons and trends to be established from 1996 to 2002. Methodological differences such as the impact of telephone surveys versus door-to-door surveys could not be taken into account. Industrial emissions, however, have not been recalculated and only original data are presented. The methods used to collect industrial activity data between inventory years were substantially different and the emissions could not reliably be back-cast. Caution is advised when comparing industrial and commercial emissions from one year to the next on the following basis:

- The quality of the fuel use data varies depending on the data collection method used.
- Total fuel use and calculated emissions are dependent on the number and type of industries included from one year to the next. Inclusion of an industry in one inventory year and not in another will impact on trends in emissions over time.
- The quality of activity classification, and subsequent emission calculation, can vary depending on the data collection method used. Surveys can provide more direct information and allow more refined classification. This ensures that more appropriate emission factors are selected and can significantly change emissions calculated for that source from one inventory to the next. For example, foundries classified as “non-ferrous” (emission factor = 89.6 kg/U) in one year may be reclassified as “aluminium only” (emission factor = 0.95 kg/U) in another year when more detailed information becomes available.

Table A2-5 outlines key characteristics of the various inventories and identifies the main differences. It also states how the back-cast estimates were calculated.

Inventory	1996	1999	2002	1996 – 2002 Comparison
Study area	Suburb by suburb data. Also summary data for the equivalent Inner and Suburban Christchurch areas. Did not include an equivalent Metropolitan or Outer Christchurch area.	Identical Inner and Suburban Christchurch and Metropolitan Christchurch areas (Inner-City/Suburb, 25 Suburb areas and the Christchurch Territorial Boundary), some individual suburbs (e.g. Sumner) and properties over/under 4 hectares.	Inner, Suburban, Outer and Metropolitan Christchurch. Data were also collected for 10 alternative sub-areas. These are not presented in this report.	Comparisons could be made directly with the 1996 and 1999 Inventories for Inner and Suburban Christchurch and with Metropolitan Christchurch in 1999.
Domestic home heating Survey method and focus	Telephone and Door-to-Door surveys	Door-to-door survey	Telephone survey, home heating diary	
	Main living area	Main living area	Main living area	
	Typical winter's day	Typical winter's night (same as day)	Typical winter's day and/or night	
Domestic home heating Number of households surveyed and associated errors	800 (3.5%)	1701 (2.4% error)	1467 (2.5% error)	
Domestic Home Heating Burner categories	Pre-1989, 1989-1992 and 93+ wood burner categories were selected to coincide with changes in emission criteria rules.	Burner categories were: >10 years (Pre-1989) 5<>10 years (mid-1989 to mid-1994) <5 years (mid-1994 to mid-1999) These were adjusted to correspond with the pre-1989, 1989-1992 and 93+ categories.	Burner categories were: >10 years (pre-1992) <10 years (1992-2000) 1-2 years (2000-2001)	Burner numbers in 1996 and 1999 were adjusted to correspond with the 2002 categories. The pre-1989 and 1989-1992 burner categories were added together. The number of burners installed in 1992 were subtracted from the total and added into the 1992-2000 category.

Inventory	1996	1999	2002	1996 – 2002 Comparison
	Included pot bellies, incinerators and enclosed coal burners	Included a multi-fuel burner category - wood burners using coal, potbellies, incinerators and enclosed coal burners.	The multi-fuel burner category includes the same types of burners as those in 1999.	Potbellies, incinerators, enclosed coal burners and wood burners using coal in 1996 were added together to obtain numbers comparable to the 1999 and 2001 multi-fuel burner categories.
Domestic home heating Fuel characteristics[1]	Wood Log = 1.6 kg	Wood log = 1.4 kg	Wood log = 1.9 kg	A log weight of 1.9 kg was assumed for 2002 and the average fuel use per appliance was calculated. A weighted average across all study areas was derived based on population. The weighted averages were applied to appliance data in 1996, 1999 and 2002. This enabled trends to be established purely on the basis of burner numbers and their associated emissions, rather than fuel variations from one year to the next.
	Coal bucket = 10 kg	Coal bucket = 9kg	Coal bucket = 9kg	As above.
Motor vehicles	VKT data from Environment Canterbury	NZTER emission rates, VKT using the TRIPS transport model	NZTER emission rates, VKT using the CTS/TRACKS transport model	The CTS/TRACKS model was used to estimate VKT, based on land-use data for the appropriate years, in 1996 and 1999. The revised data were directly comparable to 2002.
Industry	Industry – Part A, B and C Resource consent files and questionnaires for select industries	Resource consent files	Questionnaires were distributed to all industries with consents. Resource consent files and previous inventory data were used to supplement this information.	
Emission factors	Domestic heating, motor vehicles and industry – a variety of sources including the USEPA	Domestic heating emission factors were revised to take into account more recent and local emission data. Motor vehicle emission factors revised as part of the vehicle fleet emission control strategy. Industrial emission factors included more detailed data on emissions from different types of coal boilers.	Domestic heating emission factors revised to take into account more recent and local emission data. Motor vehicle emission factors were different. This was due to the updated motor vehicle fleet. Industrial emission factors were the same as those used in 1999 with the exception of foundries and abrasive blasting. Improved classification of activities, however, resulted in different emission factors being used in 2002 for the same industry in	The emission factors revised in 2002 were applied to 1996 and 1999 data to re-estimate domestic heating emissions. The motor vehicle emission factors were applied to the revised VKT data. It was not possible to re-run 1996 industrial data using the 1999 and 2002 emission factors as Ecan did not have access to the original 1996 datasets. Further, individual industries were not reclassified to directly correspond with the 2002 inventory classifications. The industry data, therefore, are not directly comparable from one inventory to the next.

[1] Householders were asked how many logs were used and an assumed wood weight was applied to the data.

Inventory	1996	1999	2002	1996 – 2002 Comparison
			1999.	

Table A2-5. Characteristics of the 1996, 1999 and 2002 Christchurch emission inventories (Scott & Gunatilaka 2003).

A2.1.3 Other indicators

Census

Not all towns and cities in New Zealand have emissions inventories therefore, for this exposure assessment to be applied to cities and towns across the country other indicators of pollution emissions must be explored. The New Zealand census is a huge data source containing such indicators and methods employing census data could potentially be derived to estimate air pollution emissions. The census contains data regarding the fuel type used in home heating, the number of motor vehicles owned per household and a journey to work matrix all of which could be used to estimate emissions. The only limitation of the census is that it doesn't contain any information concerning indicators of industrial emissions. As yet no methods have been derived employing solely census data to derive emissions of air pollutants. Future work will investigate the possibility of incorporating this data source in further investigations.

Journey to work analysis

One of the research directions was focused upon a novel set of techniques to estimate the volume of traffic flow in Christchurch using the journey to work data from the 2001 census. The dataset was provided as a matrix by Statistics New Zealand and identifies the CAU in which each individual lives and works. The matrix was used to model the Christchurch working population's journeys to and from work through the Christchurch road network using the CAU centroids, which are connected to the nearest road link, as the origin and destination. Speed limit impedances were assigned to each road link to increase the accuracy of the model output. Major roads were assigned higher speed limits and therefore were more likely to be used than minor roads with lower speed limits. An "All-or-Nothing" method was used to estimate the most likely route people would use when travelling from origin to destination. Using this method traffic flow is assigned without considering whether or not there is adequate capacity or heavy congestion and travel time is a fixed input and does not vary depending on the congestion on a link. The model output was a revised road network containing road links with road link length and number of trips data attached. By multiplying the number of trips by the link length a value for VKTs was estimated for each road link. These VKT estimates were split into two categories those belonging to road links located within CAUs and those belonging to road links running along the boundaries of CAUs. Using GIS software the VKTs for road links located within CAUs were totalled and the VKTs for boundary road links were halved and the values assigned to both adjacent CAUs. The VKTs were then summarised for each CAU giving VKT estimates for every CAU in Christchurch (see Figures A2-2 and A2-3).

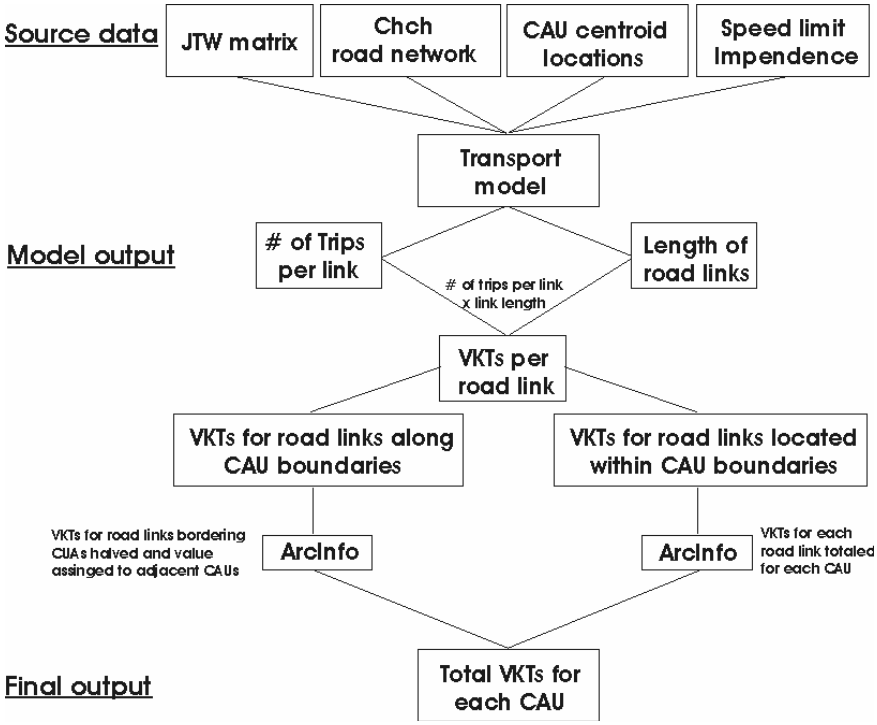


Figure A2-2. Flow diagram outlining the journey to work matrix analysis.

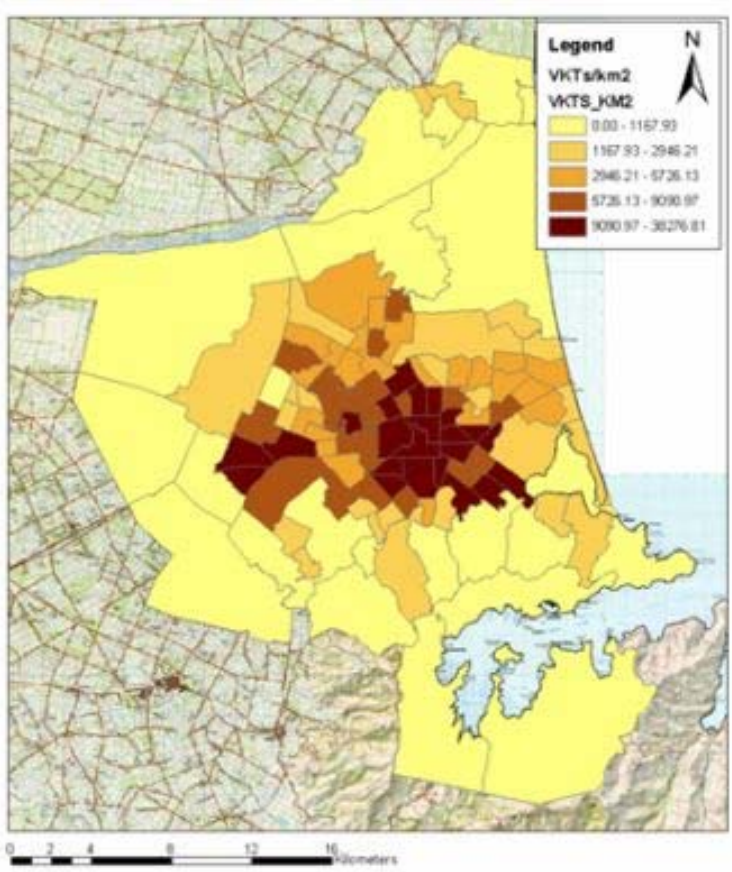


Figure A2-3. CAU map of journey to work VKT output.

Census Area Unit (CAU) emissions

The aim of this exercise was to break down the 1999 emissions from the three larger study areas to the 117 CAUs of the Christchurch territorial authority (TA) for the TAPM air pollution modelling. A copy of the 2002 emissions inventory along with revised 1999 emissions inventory data were obtained from Environment Canterbury. In addition to this 10-area data for both industrial and vehicle emissions were also provided (see Figure A2-4). Unfortunately, the 10-area data for domestic emissions was unavailable, since Environment Canterbury has constructed its emissions inventory on a 'suburb' basis (as illustrated in Figure A2-5).

The emissions inventory report outlined which CAUs were within the three large study areas. As outlined in the emissions inventory methodology, areas "inner Christchurch" and "suburban Christchurch" overlap therefore the emissions for the CAUs located outside of inner Christchurch, but within suburban Christchurch were determined by subtracting the total emissions of the various pollutants for inner Christchurch from those of suburban Christchurch.

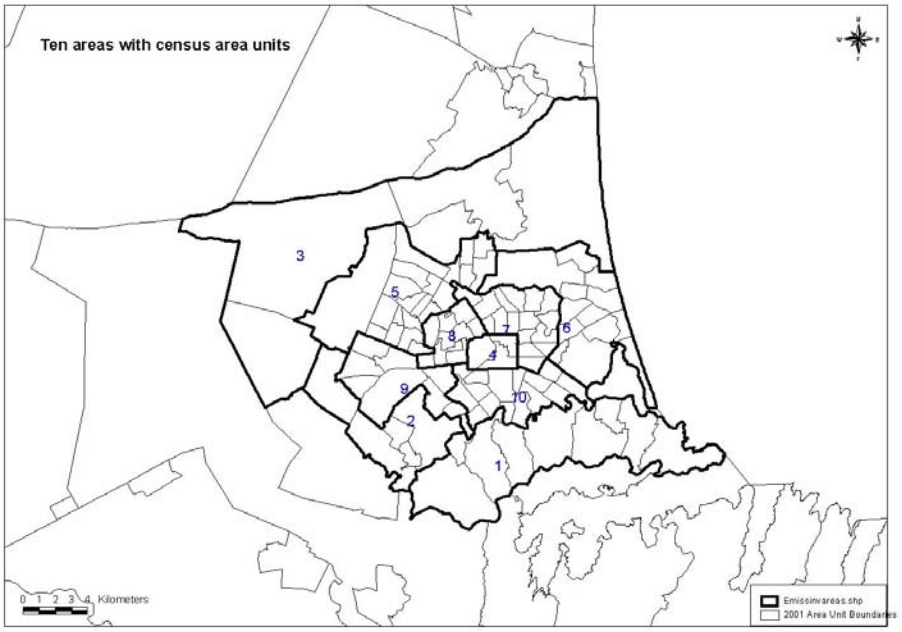


Figure A2-4. Map displaying the 10 areas with the CAUs (provided by Environment Canterbury).

Using supplementary data sources such as the 2001 census and vehicle modelling data, various weighted approaches were employed to break down the emissions inventory data to the 117 CAUs. The 2001 census was selected on the basis that it was within two years of 1999 opposed to the 1996 census, which was three and therefore a less accurate representation of the spatial variation in census data for 1999.

Relevant data were selected from the 2001 dwelling module of the census. These data included information regarding the "fuel type used to heat dwelling" for the 117 census area units comprising the Christchurch TA. The data were tabulated in a format that was compatible with the GIS software ArcView 3.2. Using ArcView 3.2 the census data were added to the corresponding year's census area unit map located in the Geography Department's GIS database. This allowed for the production of chloropleth maps at census area unit level displaying the spatial characteristics of the census data (Figure A2-6).

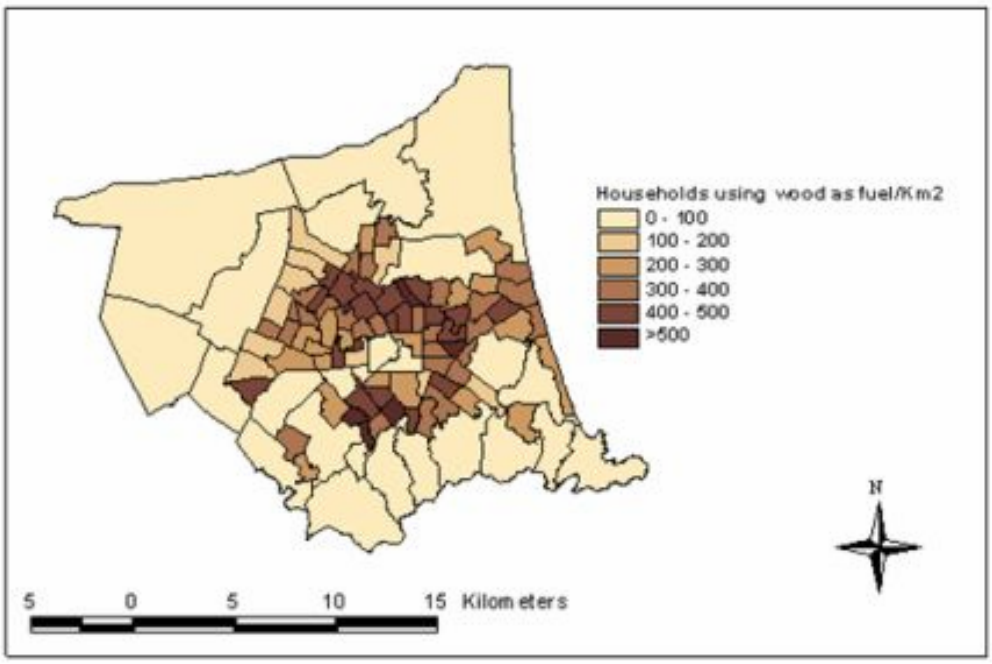


Figure A2-5. CAU choropleth map showing the number of households that burn wood to heat their homes per km^2.

Domestic emissions for 1999 by CAU break down

Revised 1999 domestic emissions and fuel use data were taken from the 2002 emissions inventory. Unfortunately, the inventory only contained revised total daily emissions data and data for the 4 daily time periods was required in order to conduct the TAPM modelling. To divide the total daily emissions into the 4 time periods the distributions of the original fuel use data over the 4 time periods, taken from the 1999 home heating survey, were used. The resulting fuel use values were then combined with emission factors taken from the 2002 inventory to calculate emissions in kilograms for the three large study areas (equation 4).

The "fuel type used to heat dwelling" data taken from the census were then employed to break down the revised 1999 domestic emissions to CAUs. Previous research has shown that the majority of domestic emissions come from the burning of fossil fuels, in particular wood and coal. Therefore, data were extracted from the census concerning the number of dwellings per CAU that burn wood and coal to heat their homes. From this, each CAU was assigned a percentage based on the number of dwellings per CAU that use these fuels out of the total number of dwellings per larger emissions inventory region. The percentages were then used to break down the raw emissions data taken from the 2002 inventory for the three large regions. Box 1 shows an example of this. Firstly, the total number of dwellings in the Bryndwr CAU that used wood and coal to heat their homes (519) were divided by the total number of dwellings in the larger emissions inventory region using the fuels (23778). The following proportion or percentage was then applied to each pollutant category's emission value for the larger area where the Bryndwr CAU is located. In this example the 24 hr PM_{10} total was used. The resulting value is the 1999 PM_{10} 24 hr total for the Bryndwr CAU.

Example of domestic emissions calculation for CAU
Bryndwr households burning coal = 75
Bryndwr households burning wood = 444
Total households burning solid fuels = 519
Total households burning solid fuels in larger area = 23778
Bryndwr's proportion of emissions = (519/23778)*100 = 2.2%
Bryndwr's 1999 daily total PM_{10} emissions = 2.2% x 6216.3kg = 135.7kg

This methodology makes various assumptions these include, that the 2001 census data are representative for 1999, that every CAU in Christchurch has the same distribution of heating

appliance types and fuel use and that coal and wood emit the same volume of contaminants per kg when burnt. Unfortunately, home heating survey results are unavailable for individual CAUs making it impossible to assess the differences in home heating appliance types and fuel use across all CAUs however, future approaches could incorporate emissions factors from the emissions inventory to attempt to allow for differences in the amount of contaminants emitted by the various solid fuel types.

Vehicle emissions for 1999 by CAU

The revised 1999 vehicle emissions data were broken down from the 10 areas to the 25 emissions inventory suburbs using the distribution of emissions from the original 1999 25 inventory suburb vehicle emissions data.

Example of vehicle emission calculation for 25 suburbs

Revised Area-7 estimated PM_{10} daily total = 162.6kg
Area-7 contained three of the original 25 suburbs, Linwood, Shirley and St Albans:
Original 1999 Linwood daily total PM_{10} emissions = 61.8kg
Original 1999 Shirley daily total PM_{10} emissions = 19.0kg
Original 1999 ST Albans daily total PM_{10} emissions = 69.3kg
Total for all suburbs = 150.1kgs. Converted into proportions these are:
Linwood = 41%
Shirley = 13%
St Albans = 46%
New revised daily total 25 suburb PM_{10} emission:
Linwood = 162.6 x 41% = 66.66kg
Shirley = 162.6 x 13% = 21.14kg
St Albans = 162.6 x 46% = 74.8kg

The newly calculated 1999 25 suburb emissions where then broken down to census area units using 2001 vehicle kilometres travelled (VKT) data derived from the NZTER database and then summarised for every census area unit. From the census area unit VKT data, VKT totals were derived for each of the larger 25 suburbs. Each CAU was then assigned a percentage calculated by dividing the total number of VKTs per CAU by the total number of VKTs per larger area. This percentage was then used to break down the 1999 25 suburb vehicle emissions data. Box 3 gives an example demonstrating this procedure.

Example of vehicle emission calculation for CAUs

E.g. Shirley revised daily total PM_{10} emissions = 21.14kg
Shirley contained 4 CAUs, Dallington, Shirley East, North Richmond, and Burwood
Dallington VKTs 18350
Shirley East VKTs 27022
North Richmond VKTs 34327
Burwood VKTs 44207
Total VKTs 123906
Converted into proportions these are:
Dallington = 14.8%
Shirley East = 21.8%
North Richmond = 27.7%
Burwood = 35.7%
Therefore the CAU emissions are:
Dallington = 21.14 x 14.8% = 3.1kg
Shirley East = 21.14 x 21.8% = 4.6kg
North Richmond = 21.14 x 27.7% = 5.9kg
Burwood = 21.14 x 35.7% = 7.5kg

This methodology assumes that the vehicle fleet composition for Christchurch is identical for every CAU. In reality this is not the case however, higher resolution vehicle fleet composition data for each CAU are not available. If these data do become available in the future the information could be used to more accurately analyse the vehicle emissions.

Industrial emissions for 1999 consent data

1999 industrial emissions consent data was accessed through the Department of Geography GIS databases. The data was originally prepared by Emily Wilton for the 1999 emissions inventory. Using ArcView 3.2 the point source 1999 industrial emissions consent data were joined to the 2001 census area unit polygon map. This identified the census area unit each industry was located within. Using Microsoft Excel, total emissions for each CAU were calculated by summing up emission estimates from each individual source.

The industrial consent data did not separate the particulate pollution into both the PM_{10} and $PM_{2.5}$ size fractions. Therefore, as outlined in the 2002 inventory methodology under vehicle emissions, a worst-case scenario was adopted where all particulate matter is in the $PM_{2.5}$ (and thus PM_{10}) size fractions. This may overestimate industrial emissions somewhat, but no information exists which can refine this methodology.

Lastly, the raw CAU data in kg for each source were normalised to obtain the emissions in g/s/km2 and organised in a format suitable for use in ArcView 3.2. Using ArcView 3.2, the emissions data were joined with the corresponding year's CAU map facilitating the production of chloropleth emission map themes (see Figure A2-6). Following this, the chloropleth emission map themes were converted to shapefiles. The shapefiles were then converted to arcinfo coverages using the shapefile to coverage command found in the arc toolbox. The coverages were subsequently converted to $1km^2$ grids using an arc macro language (AML) file created by for previous emissions inventory work. The AML file made use of the arc command polygrid, which assigns a code to each cell in the grid according to the "polygons" or CAUs, it overlays. If a cell contained more than one polygon the code of the polygon with the greatest area in the cell was used. The resulting output encompassed thirty grids for each of the three major air pollution sources, domestic home heating, vehicle and industry. Grids were also produced for total emissions. Five grids for each contaminant were created, one for each of the five time periods. In total one hundred and twenty grids were produced. The grids were produced in both New Zealand map grid and Geographic (lat-long) map projections. The grids were then converted into ASCII using another AML file for airshed modelling work using TAPM.

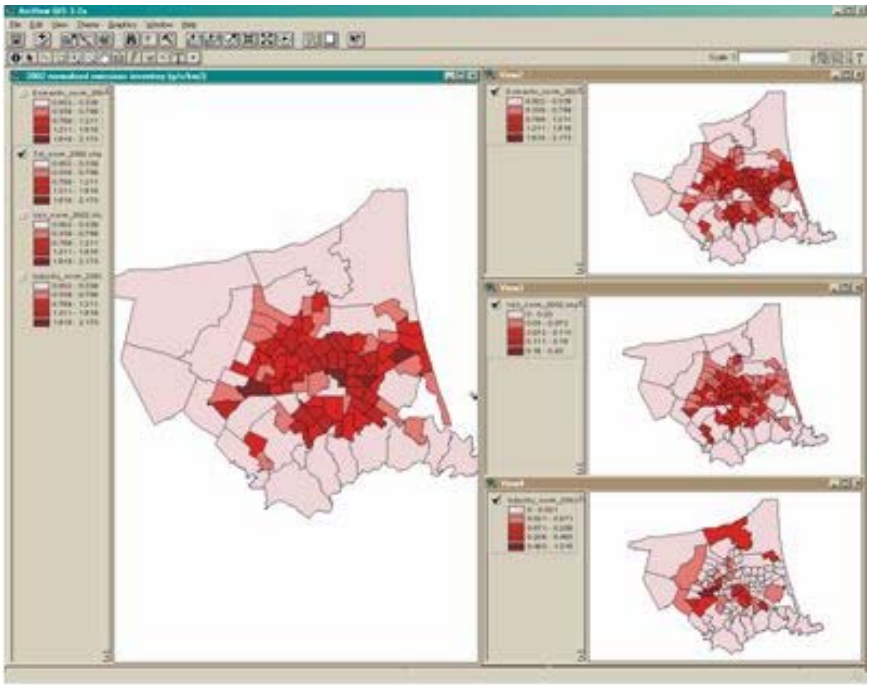


Figure A2-6. Screen capture of ArcView 3.2 showing examples of the chloropleth emission maps.

A2.1.4 Air quality monitoring in Christchurch

Air quality monitoring data can be employed to assess and validate air pollution modelling results and these data play an important role with regard to the air pollution exposure investigations. The Regional Council (now known as Ecan) has monitored air quality in Christchurch since 1988 with the original monitoring site located in Packe St, St Albans. This site was established by the former Department of Health. The Ministry of Health continued to operate this monitoring site as part of its contribution to an international air quality monitoring program. Due to land redevelopment at the original St Albans monitoring site in 1998 Environment Canterbury established a similar site in Coles Pl approximately one kilometre away. In 1999 the Ministry for the Environment took over this responsibility (Aberkane 2001). Additional air pollution monitoring in Christchurch has been undertaken by the Geography Department at Canterbury University for other projects. This information is included here as it adds to this study.

A2.1.5 Environment Canterbury monitoring

Note: Much of the information following is taken from Environment Canterbury documents.

In addition to the monitoring undertaken in St Albans, monitoring in a number of other suburbs in Christchurch has also been carried out in previous years by Environment Canterbury. A monitoring site was established in 1995 in a residential area of Hornby (at the South Hornby School), with a number of industries in the vicinity. Between June 1995 and March 1997 monitoring was carried out in the residential area of Beckenham (at the Christchurch City Council Service Centre). In July 1996 a monitoring site was established at Opawa in a residential area (Mary McLean Pl), adjacent to a large industrial area. Monitoring at this site continued until June 1999. An investigation into air pollution in Sumner was undertaken between July 1999 and January 2000. The locations of these sites are shown in Figure A2-7 (Wilton, Aberkane, and Harvey 2002).

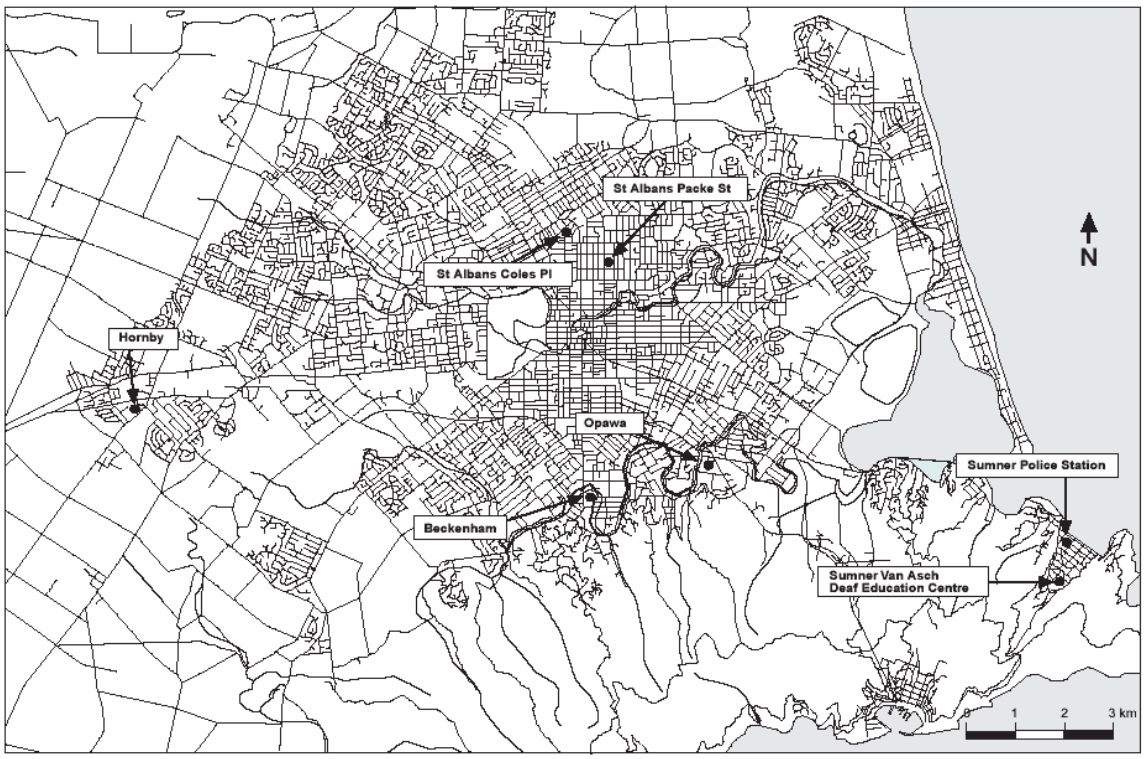


Figure A2-7. Location of past and present Christchurch air quality monitoring sites (Wilton, et al, 2002).

Suspended particulate (PM_{10} and $PM_{2.5}$)

PM_{10} is currently monitored continuously at all sites using the Rupprecht and Patashnick Co., Inc. Tapered Elemental Oscillating Microbalance (TEOM) particulate monitors. However, a number of different measurement methods have been used at MoH's air quality monitoring site in St. Albans. These include a MPSI beta gauge for the period from May 1988 until 13 June 1994 and a high volume sampler for the period 18 June to 7 August 1994. An Environment Canterbury owned TEOM has been operational at this site since 17 August 1994. In May 1996 a Wedding & Associates (W&A) beta gauge was installed. A beta gauge (W&A) was also used at the Opawa monitoring site until October 1996. Analysis of the various methods during the 1996 winter monitoring concluded that the TEOM was the most suitable continuous sampling method for use in Christchurch (Aberkane 2001).

In 1999 the operating temperature of the TEOMs was set to 40° C as this reduces the loss of particulate through particle volatilisation while avoiding complications when running at temperatures close to that of the ambient air temperature. Environment Canterbury also monitored $PM_{2.5}$ in 1997 using the Rupprecht and Patashnick Co., Inc. TEOM fitted with a $PM_{2.5}$ inlet (Aberkane 2001).

Sulphur dioxide (SO_2)

Environment Canterbury measures SO_2 using the technique of fluorescence. This technique detects SO_2 concentrations by the measurement of light emitted by SO_2 after it has been irradiated with light of a different wavelength. The measurement can be specific to SO_2 and the limit of detection is typically less than 1 ppb (Fisher et al 1995). Other techniques used in NZ include wet-chemical and flame photometric detection. Wet chemical gives a measure of total acidity and therefore, is not specific to SO_2. The detection limit is approximately 10 ppb and samples are collected over 24 hours. Because the MfE guidelines (2002) exist for 1 hour and 10 minute averaging periods for SO_2 this technique is not recommended for air quality monitoring in the Canterbury Region. Flame photometric detection for SO_2 is an acceptable technique. The detection limit is typically 1-5 ppb. This equipment is no longer commonly available from suppliers and is unlikely to be used in the Canterbury Region (Aberkane 2001).

Environment Canterbury has a number of SO_2 analysers. These include Thermo Environmental Instruments Inc. model 43C pulsed fluorescence and Advance Pollution Instrumentation (API) 100A SO_2 analysers.

Carbon monoxide (CO)

Carbon monoxide is monitored in the Canterbury Region using infrared spectrophotometry. A broad band of radiation is passed through a gas cell containing the air sample. The amount of light absorbed by the carbon monoxide is compared to a reference beam. High sensitivities (around 0.05 ppm) can be achieved by the use of multiple pass cells (Fisher et al 1995). There are 2 instrumental methods using this principle, the non-dispersive method and the gas filter correlation method. The difference between these methods is the reference beam that the sample is compared to. In the non-dispersive method the sample is split and the reference part has all CO removed by a catalytic converter. This method is prone to interference by water vapour and CO_2. In the preferred method of gas filter correlation a rotating filter is used containing CO on one side and N_2 on the other. The CO side of the filter produces a reference beam. Environment Canterbury currently owns Monitor Labs ML9830 and API 300 instruments, which use the gas filter correlation method.

In addition to ambient monitoring of CO, short-term monitoring at street level has also been conducted. Because of the short-term nature of this monitoring CO analysers have generally been hired for this purpose (Aberkane 2001).

Nitrogen dioxide (NO_2)

Nitrogen dioxide is calculated based on the measurement of nitrogen oxides using ozone chemiluminescence. Monitoring of NO_2 at the St. Albans monitoring site is conducted via single channel measurement. This requires an airflow that is alternated between two processes. One passes the air through a catalytic converter reducing nitrogen dioxide to nitric oxide. This nitric oxide is then additional to the nitric oxide already present in the sample. This allows for a measurement of NO_x. The other bypasses the converter allowing for measurements of NO. Nitrogen dioxide is calculated

based on the difference between measurements of total oxides of nitrogen (NO_x) and nitric oxide measurements. The nitrogen oxide monitor at the St. Albans monitoring site is an API 200A (Aberkane 2001).

Ozone (O_3)

Ozone in ambient air is not currently monitored continuously in the Canterbury Region. Short-term monitoring was conducted from January to April 1998. Because of the duration of this monitoring, equipment was hired for this period. The most common method of ozone measurement is ultraviolet photometry. A sample of air passes through an absorption cell where ultraviolet light at 254nm wavelength is also passed. Ozone present in the sample will absorb some of the energy and the difference in energy levels in the sample from that in a reference measurement are related to the ozone concentration (Fisher et al 1995). Another method is ethylene chemiluminescence in which the light produced from a reaction between ozone and ethylene gas is detected by a photomultiplier. This method is not proposed for use in the Canterbury region because of the flammable nature of ethylene gas (Aberkane 2001).

In addition to the data collected by Environment Canterbury, data from monitoring equipment at the St. Albans Packe St site, which is owned by the Ministry for the Environment and is operated and maintained by Watercare Services Lt, is available (Aberkane 2001).

Visibility

Visibility is measured at the St. Albans monitoring site using nephelometry, which measures the light scattering of a sample of air. As light can be scattered by fine particulate and/or water droplets in the air, a heated inlet is used to ensure humidity remains below 60% and so only fine particulate is measured. At times two nephelometers have been in use at St. Albans comparing monitoring results from heated and unheated inlets. Other methods of measuring visibility include visual observation, photographic systems and long-path systems. Fine particles measured using a TEOM fitted with a 2.5 mm inlet may also provide an indication of visibility (Aberkane 2001).

Meteorological data

Meteorological conditions strongly influence the concentrations of contaminants in ambient air. Meteorological conditions typically measured at an air quality monitoring site include wind speed, wind direction and temperature. However, other meteorological parameters can provide useful information from an air quality perspective. These include temperature measurements at 2 heights, which can provide an indication of the presence of a temperature inversion, and solar radiation, which indicates atmospheric stability (Aberkane 2001). Other parameters such as relative humidity, rainfall measurements, wind speed at more than one height, and multispectral radiation measurements can provide information for atmospheric dispersion modelling (Fisher, Graham, and Bell. 1995).

Meteorological data is logged every 10 minutes to provide detailed data for comparison to air quality data. Temperature and relative humidity are measured at some sites. All external temperature sensors must be shielded from sources of heat, such as footpaths and buildings, and out of direct sunlight. The accuracy and heights of measurement for the different meteorological parameters are detailed in Table A2-6 (Aberkane 2001).

	Height of measurement	Method	Accuracy
Wind speed	6-10 metres	Pulse Output Anemometer	<10 ms^{-1} = ± 0.5 ms^{-1}
Wind direction	6-10 metres	Potentiometer Windvane	3°
Temperature / Humidity	0-10 metres	Thermistor	1°C / 2% RH
Solar radiation	0 metres	Various methods	10 Wm^{-2}

Table A2-6. Measurements of meteorological parameters and required accuracy (Aberkane 2001).

A2.1.6 University of Canterbury monitoring

Winter 2002

Particulate monitoring was carried out in winter 2002 using MiniVol particulate samplers. These samplers draw air (and PM_{10}) through a filter and the particulate deposition on the filter is weighed using a high-resolution electronic balance. Each filter collected PM_{10} for 24 hours. For the mesoscale or city-wide study, nine MiniVol particulate samplers were placed on lamp-posts, at a height of approximately 3 m in quiet back streets in residential neighbourhoods scattered throughout Christchurch, as well as at a rural site to the west of the city. Measurements were taken from July 29 to August 8, 2002. The local-scale study used the same MiniVol sensors, but this time they were located in a more focussed area around the St Alban's neighbourhood. Monitoring at this scale occurred from August 9-19, 2002. These MiniVol data were corrected for temperature and pressure (see equations in the 2003 monitoring section below) and then analysed using ARC/View version 3.2.

Winter 2003 and summer 2004

A method was devised in order to validate the spatial and temporal nature of the model developed for predicting personal exposure (see section on TAPM). This portion of the report describes the nature of the monitoring in terms of study area, equipment utilized, site selection criteria, temporal and spatial considerations, and possible pollution confounders. This phase of the research is classified as 'validation' of the models developed in earlier phases and was undertaken as part of a PhD study.

The study area for the winter monitoring was the Christchurch Territorial Local Authority (TLA). Ten local primary schools and an Environment Canterbury control site were chosen as site locations for this study. The schools provided a secure environment and were the most conducive locations for locating the instruments a sufficient distance away from other point contaminant sources (i.e. domestic burners). An added benefit of conducting the monitoring at local schools was the promotion of awareness about pollution and health among the students and staff.

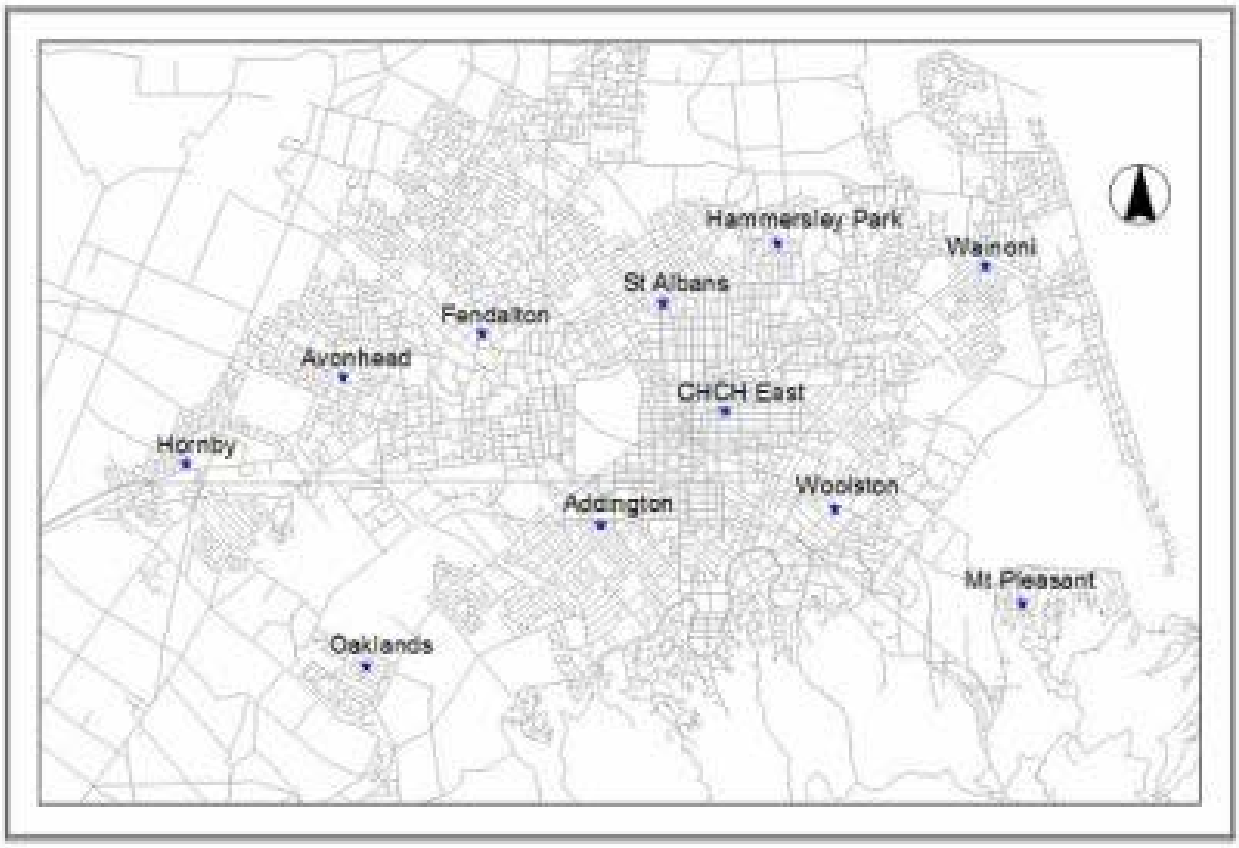


Figure A2-8. Distribution of the monitoring stations across the study area (for winter 2003).

To collect these air pollution data, MiniVols, low cost portable particulate samplers produced by AirMetrics, were used. They are able to sample for PM_{10} over periods of a few hours (depending on the quantity of pollution in the air) up to 24-hours. One issue with some particulate measuring instruments is that they heat the inflow of sampled air, chemically altering the sample (Greenwell et al. 2002). MiniVols do not heat the sample and, thus, avoid this problem. They are fairly quiet and did not disturb the schools they are placed in. It should be noted that all FRM methods are substantially more expensive, not portable, not suitable for indoor use, or in most cases, a combination of all three. The manufacturer, AirMetrics of Eugene, Oregon, USA, calibrates the MiniVols. In addition to the MiniVol samplers, HOBO temperature and humidity sensors, were installed at the monitoring sites. The temperature data collected by each HOBO was used during the calculation of particulate concentrations at actual temperature and pressure (see equations below).

MiniVols contain silicon filters that capture particulates in the size range desired, in this case particles 10 micrometers and smaller. Before filters are placed into the MiniVol, they were desiccated for 48 hours to remove moisture and weighed using a gravimetric scale.

The scale used was a Sartorius ME215-P with accuracy to the one-hundred thousandth of a gram. The scale was successfully calibrated in late May of 2003, one month before fieldwork commenced. The filters are then labelled, placed in the appropriate MiniVol at its respective location, and then collected 24 hours later. The used filters are taken back to the lab, desiccated for 48 hours, and then gravimetrically weighed on a scale in the laboratory. The difference between the end weight and the start weight yield the primary factor in determining the concentration in micrograms per cubic meter for the site. For an indicated rotameter flow rate (Qind) the flow rate at actual sampling conditions (Qact in l/min) is given by the following equation (Equation A2.4):

$$Q_{act} = (m_{vol} Q_{ind} + b_{vol}) \times \sqrt{\frac{P_{std}}{P_{act}} \times \frac{T_{act}}{T_{std}}} \qquad \text{Equation A2.4}$$

Where m_{vol} = Slope of sampler calibration graph
b_{vol} = Intercept of sampler calibration graph
P_{std} = standard atmospheric pressure, 760 mm Hg
T_{std} = standard temperature, 298 Kº
P_{act} = actual ambient pressure, mm Hg
T_{act} = actual ambient temperature, Kº

The volume of air that passed through the filter during the sampling period at actual ambient conditions (V_{act} in m^3) is derived from Equation A2.5.

$$V_{act} = \frac{60_{min/hr} \times Q_{act} \times t_{hr}}{1000_{l/m3}} \qquad \text{Equation A2.5}$$

Where t_{hr} = sampling period, in hours

Lastly, to derive the PM_{10} concentration at actual ambient conditions ($[PM]_{act}$) the net mass gain of the filter was divided by the volume of air that passed through the filter (Equation A2.6).

$$[PM]_{act} = \frac{M_{PM}}{V_{act}} \qquad \text{Equation A2.6}$$

Where $[PM]_{act}$ = PM concentration, in μg m^{-3} (actual)
M_{PM} = Mass of particulate matter collected on the filter, in μg

The monitoring was undertaken during the month of July 2003 and an additional week of monitoring was carried out in February 2004 in order to assess PM_{10} concentrations during the summer season and to establish a baseline concentration from which to compare the winter results.

Monitoring sites were selected based on the following source criteria (which are broadly consistent with the standard monitoring criteria suggested by the Ministry for the Environment (MfE 2003):

- Not within 30 meters of traffic throughways,
- Not within 50 meters of domestic heating stacks,
- Not within 30 meters of industrial stacks.
- No major obstructions (trees, buildings) for 360° at three meters or higher within ten meters of the device.

Mounting specifications (MiniVol):

- Mounted at a height of three meters from ground level.
- The device is to be mounted on a cylinder of a diameter no greater than 300cm.
- The device should be mounted on the western side (270° +/- 45°) of the mounting cylinder.

Each monitoring site contained a data logger for recording temperature and humidity readings. The loggers at each site were mounted on the same cylinder as the MiniVol, but at 180° opposite Figure A2-9) its respective MiniVol in azimuth and at a height no more than 300 cm above its respective MiniVol.

In addition to the monitoring undertaken during July 2003, monitoring was also carried out from February the 20th to the 27th, 2004. This monitoring was conducted in order to compare the winter results with summer and to establish baseline concentrations of PM_{10}.

Figure A2-9. MiniVol monitoring site at Wainoni, Christchurch, July 2003 (Wilson).

GIS grid interpolation of the monitoring data

The July 2003 monitoring data was plotted in ARC/map using each site's latitude and longitude coordinates. By employing the interpolate grid tool under the surface extension in ArcView 3.2, 500 m grids were derived for every day in July including a monthly average based upon the July monitoring point data. The value of each cell within the grid was generated using inverse distance weighting (IDW), which assumes that each input point has a local influence that diminishes with distance. The points closer to the processing cell are weighted greater than those farther away. A radius of 4000m around each monitoring site was set as a barrier inside which cell values were created. Outside of this radius no value was given to grid cells. Maps of the resulting grids were constructed using ARCMap. Following this, average July concentrations were calculated for each census area unit (CAU) in the Christchurch Territorial Local Authority. Firstly, the 2001 CAU map was overlaid onto the gridded monthly average pollution exposure map created using monitoring data for each monitoring station and the grid cells were allocated to their respective CAU. An average exposure value for each CAU was then determined by taking the mean of all cells contained in each CAU. Minimum and maximum cell values were also calculated for each CAU. To avoid extrapolation anomalies around the edge of the city, CAUs were selected in the following manner:

- CAUs lying between monitoring sites
- CAUs in contact with the CAU in which the monitoring site is located

This was done using a macro designed, created and operated by John Thyne in the Department of Geography. Essentially the macro selected each CAU polygon sequentially, overlaid it onto the grid data and then calculated the min, max and mean for the grid cells lying within the polygon. The resulting data were then joined with the 2001 CAU polygon map in ArcView 3.2. Maps of the resulting CAU polygon coverage were then constructed using ARCMap.

A2.1.7 Air pollution modelling

To obtain high-resolution (1 km^2) average values for ground level concentration of PM_{10} for the urban area of Christchurch, high-resolution simulation of meteorology and dispersion is required. This is a particular strength of modelling due to the limited nature and extent of measured data. Models generate spatially (i.e. down to 100 m^2) and temporally (i.e. hourly) dense datasets that can augment observed datasets. Validation of modelled data is through direct comparison with the observed data with statistical measures. If the agreement between modelled and measured data is within an acceptable range, it is assumed that modelled data are also valid for areas without measurements. To generate such a dataset, The Air Pollution Model (TAPM) is employed for this research project.

TAPM is a three-dimensional incompressible, non-hydrostatic, primitive equations model, which uses a terrain-following coordinate system (Hurley 2002). The meteorological component of the model is supplied with a database derived from the Limited Area Prediction System (LAPS) analysis data from the Australian Bureau of Meteorology (Puri et al. 1998). The sea surface temperature is derived from Rand's global long-term means at a resolution of 100 km. Explicit cloud micro-physics option uses the scheme of (Katzfey & Ryan 1997) for warm rain (ice processes are ignored). Short-wave and long-wave fluxes are calculated using Mahrer and Pielke (1977). TAPM is not suitable for representing deep atmospheric circulations due to the assumption of incompressibility and the fact that non-hydrostatic effects only go up to 5000 m. Above this level, the meteorological variables are smoothed out in order to minimize wave reflections from the model top. TAPM uses a gradient diffusion approach with a counter-gradient correction for turbulence closure. The diffusivity (K) is calculated with the standard E-ε scheme.

For computational efficiency, the model can be used in a one-way interacting, telescoping grid-nesting configuration. The simulations presented here use four grids with grid spacing of 18, 9, 3, and 1 km respectively with each grid having 45 zonal and meridional grid nodes. The largest grid covers most of the South Island of New Zealand (Figure A2-10); while the extent of the highest resolution grid (grid 4) is presented by Figure A2-11. To simulate the dispersion of PM_{10}, the air pollution module of TAPM was used in a tracer mode (with no chemistry). The tracer was allocated to represent emissions due to home heating with solid fuels. Gridded emission inventories were prepared using GIS software; these were derived from Environment Canterbury's 2002 emissions inventory (refer to CAU emissions derivation above). Unfortunately, the inventory has poor spatial and temporal resolution; in addition, it does not recognize differences in emission patterns between weekdays and weekends that can potentially be a shortcoming.

The boundary layer adjacent to the ground is parameterized with the Monin-Obukhov surface similarity. Surface properties are accounted for with an energy balance scheme using the land-use and soil type as input. The scheme uses a single-layer canopy and therefore does not resolve the flow within the canopy or the roughness sub-layer. Urban surface albedo (α_U = 0.15), emissivity (ε_U = 0.95), and anthropogenic heat flux (A_U = 30 Wm^{-3}) are specified according to Oke (1987). An urban roughness length of 1 m is used to calculate the surface layer scaling variables.

A2.1.8 Exposure variations

A number of previous studies in other countries have noted that different population groups are exposed to different levels of ambient air pollution with the socio-economically disadvantaged having the highest exposures (Jerrett et al. 2001; Mitchell & Dorling 2003). This disparity in exposure between different social groups has received limited explicit attention in New Zealand. This is despite the numerous studies that have noted ethnic and social differentials in health outcomes among New Zealanders such as hospital admissions, asthma and most cancers (e.g. Ellison-Loschmann *et al* (2004)). This section considers whether different social and ethnic groups within Christchurch are exposed to differential levels of particulate air pollution. More specifically people in different age, ethnic and deprivation groups are exposed to different levels of particulate air pollution using the estimates that have been calculated for CAUs across the city.

For each CAU in Christchurch the percentage of the population aged 65 and above was calculated using the 2001 census. The results were sorted into ascending order for each age group and placed

into quintiles, so that the upper quintiles are characterised by the greatest proportion of elderly people. The mean total pollution estimates was then calculated for each age group quintile.

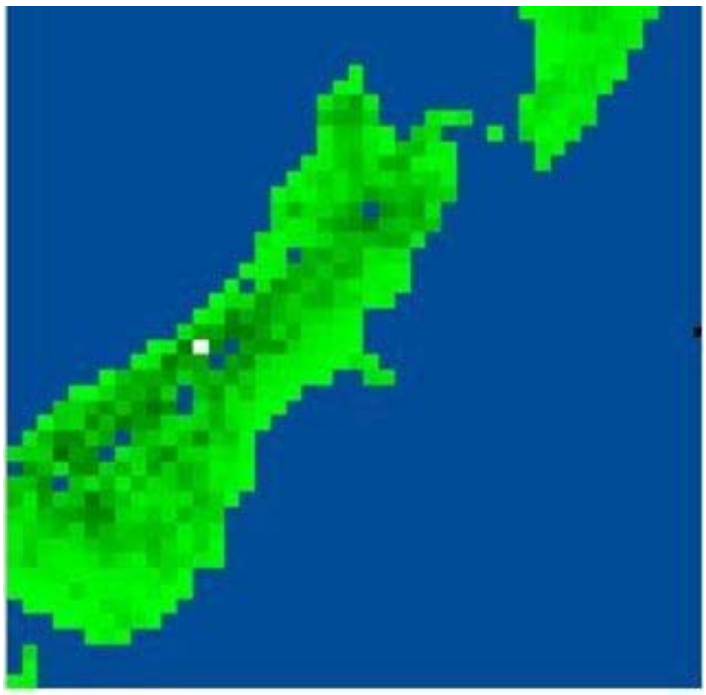


Figure A2-10. Geographical extent of grid 1. Each pixel is 18 km^2.

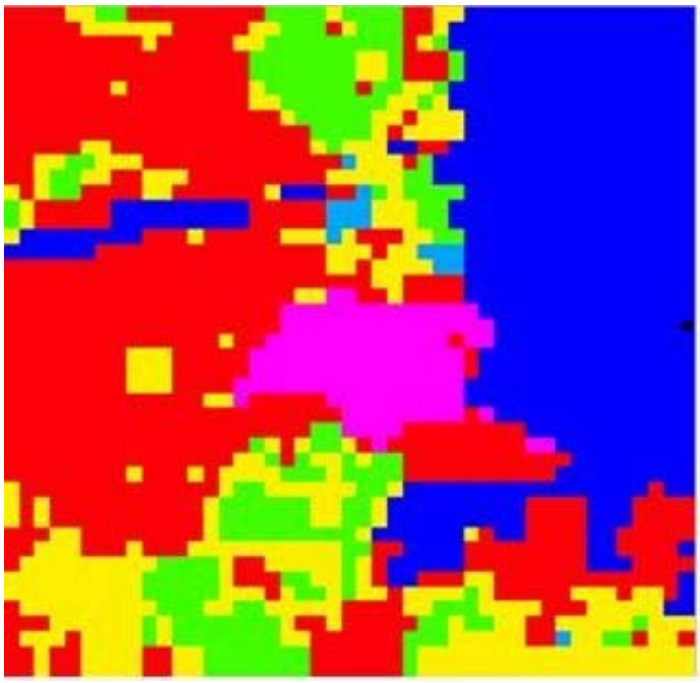


Figure A2-11. Geographical extent of grid 4. Each pixel is 1 km^2. Pink pixels show the urban area of Christchurch.

For each CAU in Christchurch the percentage of the population aged 65 and above was calculated using the 2001 census. The results were sorted into ascending order for each age group and placed into quintiles, so that the upper quintiles are characterised by the greatest proportion of people of elderly people. The mean total pollution estimates was then calculated for each age group quintile.

The distribution of particulate pollution relative to the distribution of ethnic groups was investigated by calculating the mean pollution level for CAUs divided into quintiles according to the proportion of European, Maori, Pacific Island and Asian.

Previous studies have noted that pollution levels tend to be highest in areas of high social deprivation (Mitchell & Dorling 2003). For example, Friends of the Earth Pollution Injustice Campaign found that 662 of the largest factories in the United Kingdom were located in areas with an annual average income of less than £15,000, with only six factories in areas where average annual income levels were greater than £30,000.

Given the apparent social gradient in air pollution exposure noted elsewhere, the Christchurch pollution estimates were compared to the NZDep 2001 score in order to identify whether there is a social gradient in air pollution exposure in the city. The distribution of particulate pollution relative to material deprivation was investigated by comparing the mean pollution levels (domestic, vehicle, industrial and total) in deprivation quintiles calculated from the NZDep 2001 score (the upper quintile representing the 20% most deprived CAUs in New Zealand).

A2.2 Results

A2.2.1 Emissions

Figure A2-12 and Table A2-7 contain summary results from the 2002 emissions inventory. They summarise contaminant discharges from each source during the winter of 2002 by study area. Table A2-8 provides a summary of contaminant discharges for each inventory year, excluding the other sources category and $PM_{2.5}$ (as these were not included in previous inventories).

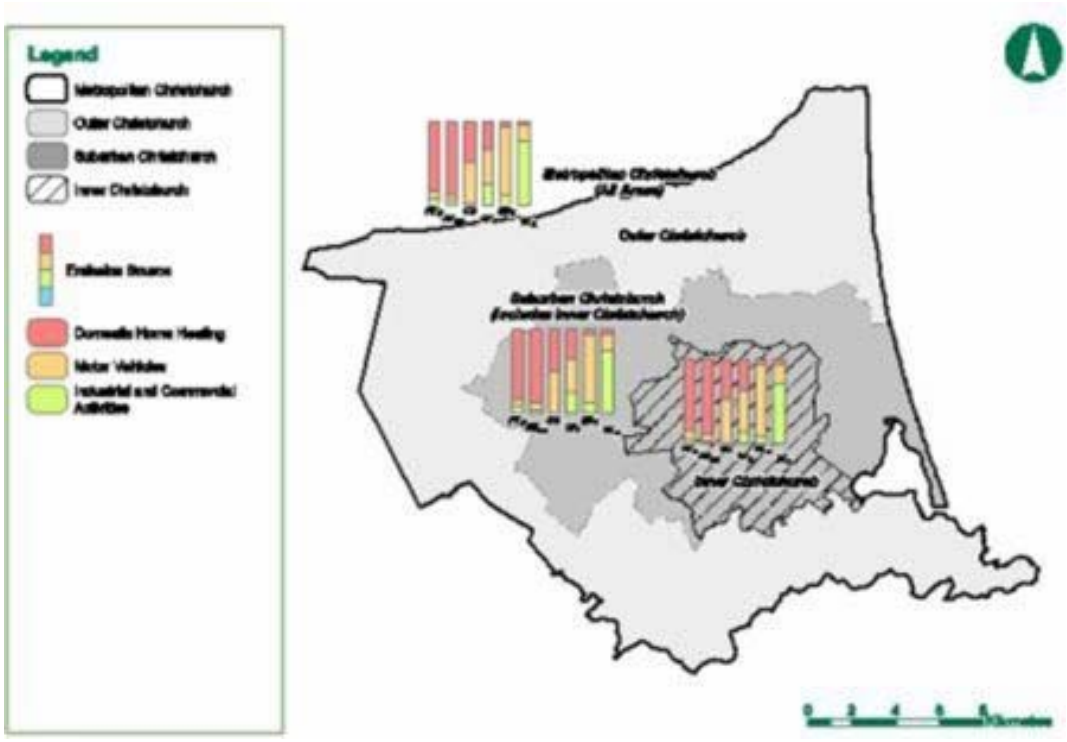


Figure A2-12. Relative contribution of sources to contaminant emissions during winter months (Scott & Gunatilaka 2003)

Inner Christchurch						
PM_{10}	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.41	0.77	3.30	0.56	5.04	86%
Industry/commercial	0.07	0.07	0.05	0.09	0.28	5%
Motor vehicles	0.13	0.21	0.17	0.02	0.54	9%
Total	0.61	1.06	3.52	0.67	5.86	
$PM_{2.5}$	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.38	0.72	3.07	0.52	4.70	88%
Industry/commercial	0.03	0.03	0.02	0.03	0.11	2%
Motor vehicles	0.13	0.21	0.17	0.02	0.54	10%
Total	0.54	0.97	3.26	0.58	5.35	
CO	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	3.58	6.86	27.64	4.86	42.95	48%
Industry/commercial	0.11	0.11	0.07	0.13	0.42	0.5%
Motor vehicles	11.47	18.51	15.07	1.99	47.04	52%
Total	15.16	25.48	42.79	6.98	90.40	
CO_2	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	99.47	186.84	772.74	112.72	1171.77	38%
Industry/commercial	97.19	119.15	76.41	126.06	418.81	13%
Motor vehicles	340.11	685.62	436.21	56.51	1518.45	49%
Total	536.77	991.60	1285.36	295.29	3109.03	
NO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.05	0.09	0.44	0.06	0.65	7%
Industry/commercial	0.18	0.20	0.13	0.21	0.72	8%
Motor vehicles	1.95	3.23	2.54	0.37	8.08	86%
Total	2.18	3.52	3.11	0.64	9.45	
SO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.02	0.03	0.15	0.02	0.22	7%
Industry/commercial	0.52	0.60	0.42	0.76	2.31	70%
Motor vehicles	0.18	0.30	0.24	0.03	0.75	23%
Total	0.72	0.93	0.81	0.81	3.28	

Suburban Christchurch						
PM_{10}	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.86	1.53	6.00	1.17	9.56	83%
Industry/commercial	0.29	0.33	0.15	0.18	0.95	8%
Motor vehicles	0.23	0.37	0.30	0.04	0.94	8%
Total	1.37	2.24	6.45	1.39	11.45	
$PM_{2.5}$	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.80	1.44	5.59	1.10	8.92	86%
Industry/commercial	0.16	0.17	0.08	0.08	0.49	5%
Motor vehicles	0.23	0.37	0.30	0.04	0.94	9%
Total	1.19	1.98	5.97	1.22	10.36	
CO	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	7.62	13.52	50.72	10.47	82.32	49%
Industry/commercial	0.48	0.52	0.26	0.30	1.55	1%
Motor vehicles	19.91	32.15	26.45	3.61	82.12	49%
Total	28.01	46.19	77.43	14.38	165.99	
CO_2	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	220.42	368.67	1415.46	260.82	2265.36	35%
Industry/commercial	377.27	465.44	307.83	390.18	1540.73	24%
Motor vehicles	598.05	1219.36	772.90	103.77	2694.08	41%
Total	1195.75	2053.46	2496.19	754.77	6500.17	
NO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.11	0.19	0.80	0.13	1.23	7%
Industry/commercial	0.63	0.73	0.45	0.56	2.37	13%
Motor vehicles	3.44	5.70	4.53	0.68	14.36	80%
Total	4.18	6.62	5.79	1.36	17.95	
SO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.03	0.06	0.27	0.04	0.41	5%
Industry/commercial	1.81	2.11	1.36	1.79	7.07	80%
Motor vehicles	0.32	0.52	0.42	0.06	1.32	15%
Total	2.16	2.70	2.06	1.89	8.80	

Outer Christchurch						
PM_{10}	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.16	0.28	0.91	0.18	1.52	73%
Industry/commercial	0.07	0.11	0.08	0.08	0.34	16%
Motor vehicles	0.05	0.08	0.07	0.01	0.22	10%
Total	0.28	0.47	1.06	0.27	2.08	
$PM_{2.5}$	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.15	0.26	0.86	0.16	1.43	77%
Industry/commercial	0.04	0.06	0.05	0.06	0.21	11%
Motor vehicles	0.05	0.08	0.07	0.01	0.22	12%
Total	0.24	0.40	0.98	0.24	1.86	
CO	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	1.46	2.52	8.38	1.51	13.87	42%
Industry/commercial	0.12	0.18	0.14	0.16	0.60	2%
Motor vehicles	4.63	6.85	6.11	0.92	18.50	56%
Total	6.21	9.54	14.63	2.59	32.97	
CO_2	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	41.45	64.53	226.03	38.28	370.29	25%
Industry/commercial	103.77	160.65	55.08	45.11	364.61	25%
Motor vehicles	154.36	346.26	195.09	26.71	722.42	50%
Total	299.58	571.45	476.19	110.10	1457.32	
NO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.02	0.03	0.11	0.02	0.18	4%
Industry/commercial	0.12	0.21	0.08	0.06	0.47	12%
Motor vehicles	0.83	1.27	1.10	0.17	3.38	84%
Total	0.98	1.51	1.28	0.25	4.03	
SO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.00	0.01	0.03	0.01	0.05	3%
Industry/commercial	0.41	0.60	0.27	0.15	1.43	80%
Motor vehicles	0.08	0.11	0.10	0.02	0.31	17%
Total	0.49	0.73	0.39	0.17	1.79	

Metropolitan Christchurch						
PM_{10}	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	1.02	1.81	6.89	1.36	11.08	82%
Industry/commercial	0.35	0.44	0.23	0.26	1.29	10%
Motor vehicles	0.28	0.45	0.37	0.05	1.16	9%
Total	1.66	2.70	7.49	1.68	13.53	
$PM_{2.5}$	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.95	1.69	6.42	1.28	10.35	85%
Industry/commercial	0.20	0.23	0.13	0.14	0.70	6%
Motor vehicles	0.28	0.45	0.37	0.05	1.16	10%
Total	1.43	2.38	6.93	1.48	12.21	
CO	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	9.15	16.03	58.92	12.11	96.20	48%
Industry/commercial	0.60	0.69	0.40	0.46	2.15	1%
Motor vehicles	24.54	39.00	32.56	4.52	100.63	51%
Total	34.29	55.72	91.88	17.09	198.97	
CO_2	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	263.37	433.00	1637.32	301.96	2635.65	33%
Industry/commercial	481.05	626.09	362.91	435.29	1905.34	24%
Motor vehicles	752.42	1565.62	967.99	130.48	3416.50	43%
Total	1496.83	2624.71	2968.22	867.73	7957.49	
NO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.13	0.22	0.90	0.15	1.41	6%
Industry/commercial	0.75	0.93	0.53	0.62	2.84	13%
Motor vehicles	4.28	6.98	5.63	0.85	17.73	81%
Total	5.16	8.13	7.07	1.62	21.98	
SO_x	**6am-10am (t)**	**10am-4pm (t)**	**4pm-10pm (t)**	**10pm-6am (t)**	**Total (t)**	**Total (%)**
Domestic home heating	0.03	0.07	0.30	0.05	0.46	4%
Industry/commercial	2.22	2.72	1.63	1.94	8.50	80%
Motor vehicles	0.39	0.64	0.52	0.08	1.63	15%
Total	2.65	3.42	2.45	2.07	10.59	

Table A2-7. Contaminant emissions by study area and time of day, during winter months, 2002 (Scott & Gunatilaka 2003).

Inner Christchurch	Domestic home heating				Motor vehicles				Industrial and commercial*			Total	
	1996	1999	2002	Overall change	1996	1999	2002	Overall change	1996	1999	2002	1996	1999
	t/day	t/day	t/day		t/day	t/day	t/day		t/day	t/day	t/day	t/day	t/day
PM_{10}	5.67	6.24	5.08	-0.10	0.69	0.64	0.54	-0.22	0.51	0.25	0.28	6.87	7.13
CO	42.15	50.58	42.95	0.02	55.67	51.59	47.04	-0.16	0.23	0.35	0.42	98.05	102.52
CO_2	1040.95	1338.24	1171.77	0.13	1434.02	1486.64	1518.45	0.06	371.10	481.00	418.81	2846.07	3305.88
NO_x	0.73	0.83	0.65	-0.12	8.61	8.41	8.08	-0.06	0.82	0.79	0.72	10.16	10.02
SO_x	0.33	0.31	0.22	-0.34	0.69	0.72	0.75	0.09	1.56	1.96	2.31	2.58	2.99

Suburban Christchurch	Domestic home heating				Motor vehicles				Industrial and commercial*			Total	
	1996	1999	2002	Overall change	1996	1999	2002	Overall change	1996	1999	2002	1996	1999
	t/day	t/day	t/day		t/day	t/day	t/day		t/day	t/day	t/day	t/day	t/day
PM_{10}	11.09	11.45	9.62	-0.13	1.20	1.12	0.94	-0.22	1.02	1.03	0.95	13.31	13.59
CO	84.16	95.56	82.32	-0.02	96.65	89.74	82.12	-0.15	0.48	1.07	1.55	181.28	186.37
CO_2	2025.57	2516.25	2265.36	0.12	2557.19	2636.85	2694.08	0.05	688.88	976.79	1540.73	5271.64	6129.89
NO_x	1.36	1.47	1.23	-0.10	15.19	14.88	14.36	-0.06	1.45	1.80	2.37	18.00	18.14
SO_x	0.59	0.52	0.41	-0.31	1.20	1.26	1.32	0.10	3.06	6.61	7.07	4.85	8.39

Metropolitan Christchurch	Domestic home heating			Motor vehicles			Industrial and commercial*		Total	
	1999	2002	Overall change	1999	2002	Overall change	1999	2002	1999	2002
	t/day	t/day		t/day	t/day		t/day	t/day	t/day	t/day
PM_{10}	13.14	11.16	-0.15	1.37	1.16	-0.15	1.18	1.29	15.69	13.61
CO	109.98	96.20	-0.13	109.72	100.63	-0.08	1.35	2.15	221.04	198.97
CO_2	2885.73	2635.65	-0.09	3328.29	3416.50	0.03	1252.00	1905.34	7466.02	7957.49
NO_x	1.68	1.41	-0.16	18.33	17.73	-0.03	2.25	2.84	22.25	21.98
SO_x	0.59	0.46	-0.22	1.54	1.63	0.05	8.32	8.50	10.44	10.59

- Quantities discharged from this sector cannot be compared between inventories because of changes in the method of collecting information. Increases in industrial/commercial activity since 1999 suggest that increases in contaminant loadings may have occurred, although it is also possible that changed in technology may have offset some increases. The 2002 inventory will provide a basis for assessment of temporal trends in the context of future inventories.

Table A2-8. Change in contaminant emissions from 1996 – 2002 (Scott & Gunatilaka 2003).

CAU derivation results

Figures A2-13 to A2-32 are census area unit maps displaying the PM_{10} results from the CAU emissions derivation. The figures show the volume of PM_{10} emitted in grams per second per square kilometre for the five different time periods for the specified source contributors during the winter of 1999. The figures illustrate the spatial distribution of PM_{10} emissions over the city by CAU and time period. Due to the nature of the methodology used and the assumptions made to break down the emissions from the larger emissions areas the spatial distribution seen in the maps for domestic emissions are very similar. The same can be seen in the vehicle emissions maps. However, this is not the case for the industrial emissions as these CAU values were generated from individual point source emissions totaled for every CAU. An assessment of the temporal variation in the emissions can be made from the PM_{10} values indicated in the legends of the CAU maps and it can be seen that the majority of domestic PM_{10} emissions in the winter occur between 4pm and 10pm.

Figures A2-33 to A2-36 are examples of the resulting grids generated from the CAU emissions maps for use in the TAPM modelling. They display 24 hr total emissions of PM_{10} from the three major contaminant sources and a grid for total emissions during the winter.

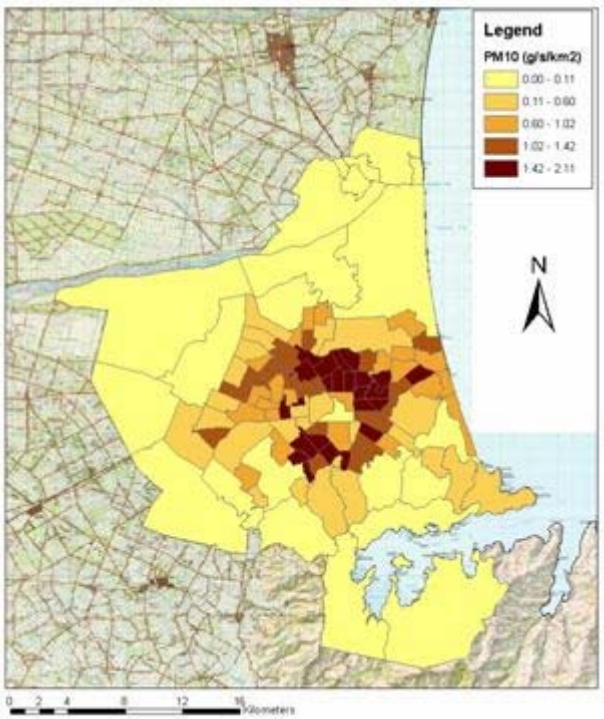


Figure A2-13. CAU map displaying daily domestic PM_{10} emissions for a typical winters day in 1999

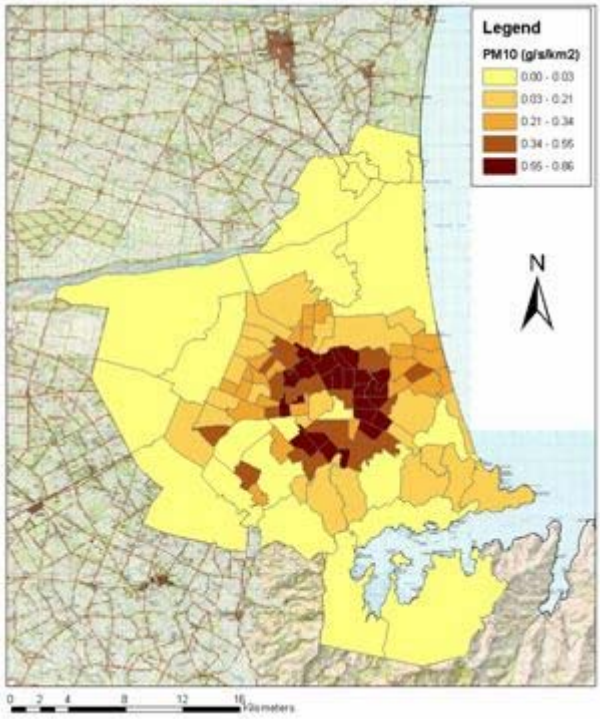


Figure A2-15. CAU map displaying domestic PM_{10} emissions between 10am and 4pm for a typical winters day in 1999

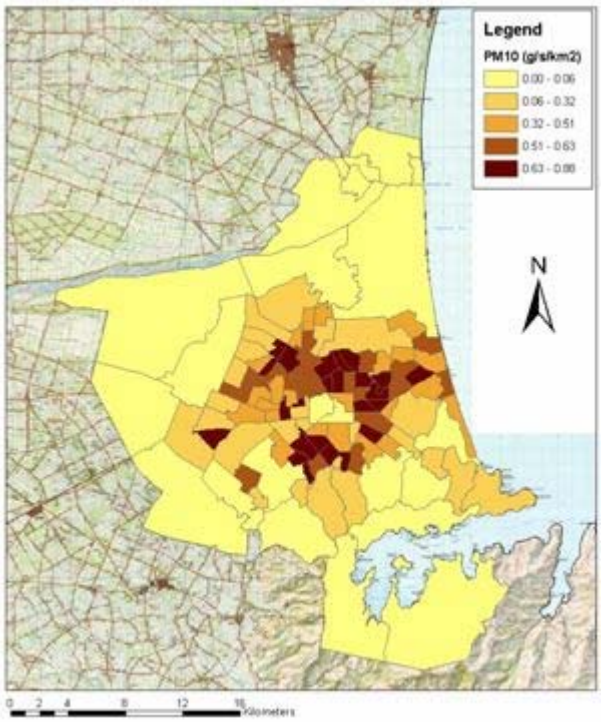


Figure A2-14. CAU map displaying domestic PM_{10} emissions between 6am and 10am for a typical winters day in 1999

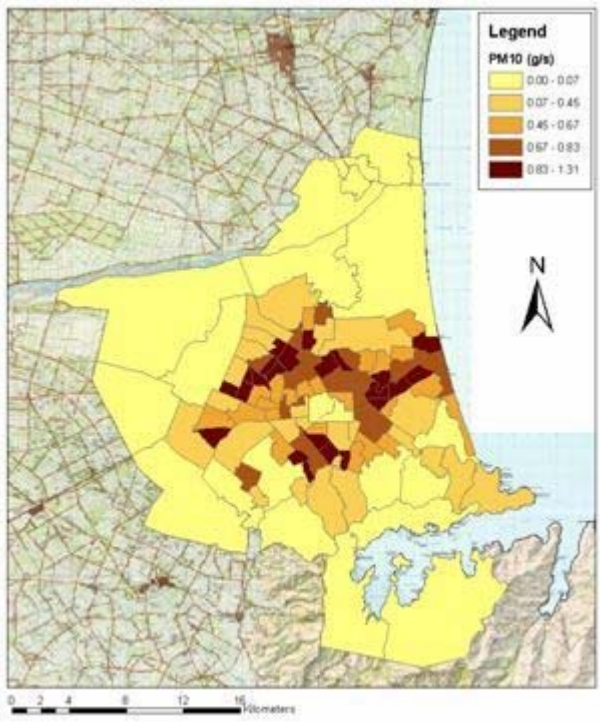


Figure A2-16. CAU map displaying domestic PM_{10} emissions between 4pm and 10pm for a typical winters day in 1999

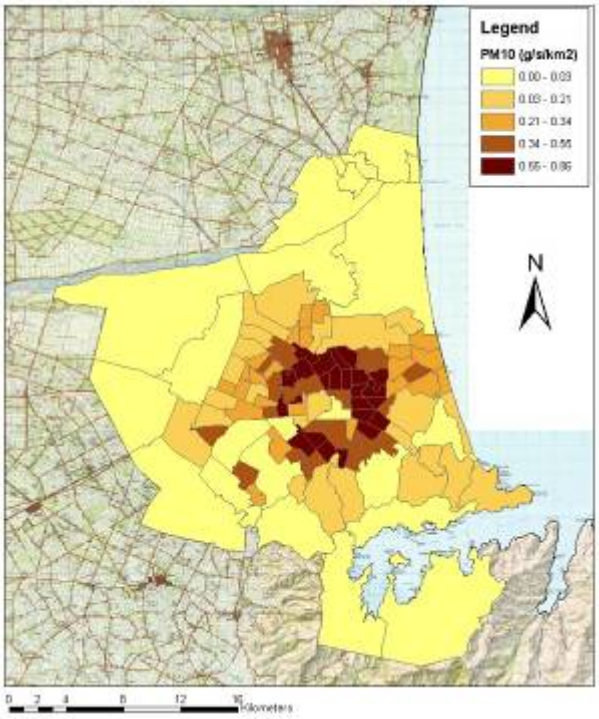


Figure A2-17. CAU map displaying domestic PM_{10} emissions between 10pm and 6am for a typical winters day in 1999

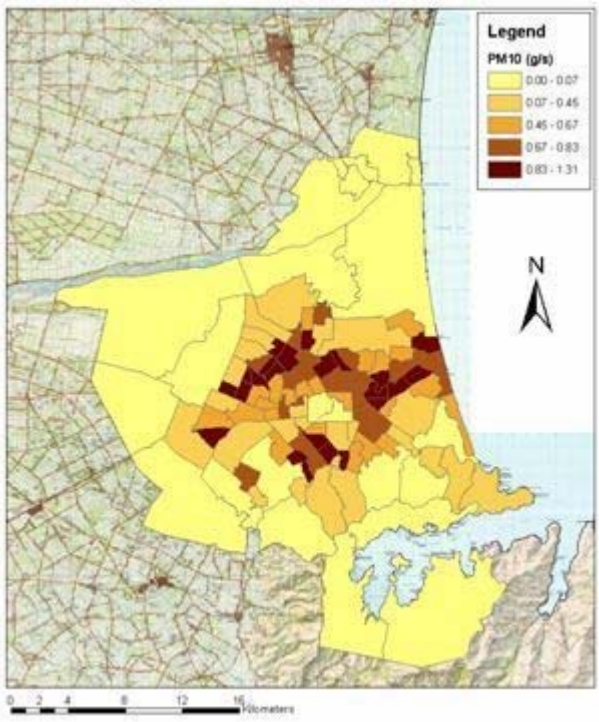


Figure A2-18. CAU map displaying daily vehicle PM_{10} emissions for a typical winters day in 1999

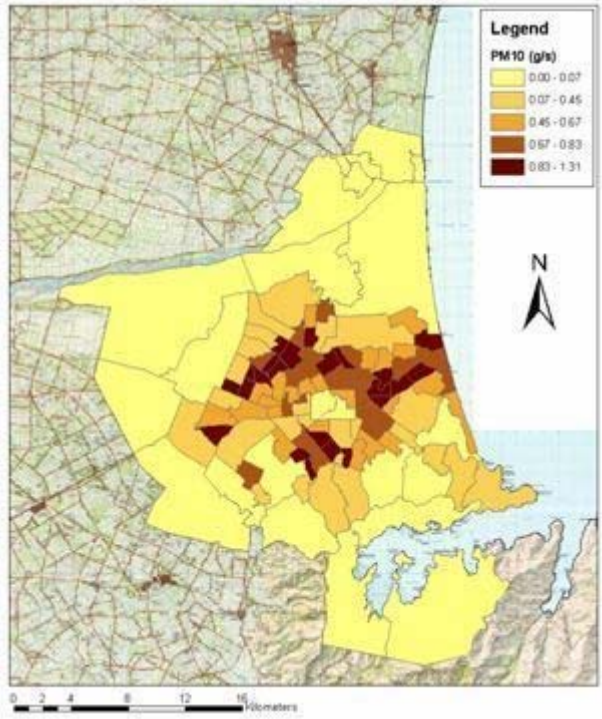


Figure A2-19. CAU map displaying vehicle PM_{10} emissions between 6am and 10am for a typical winters day in 1999

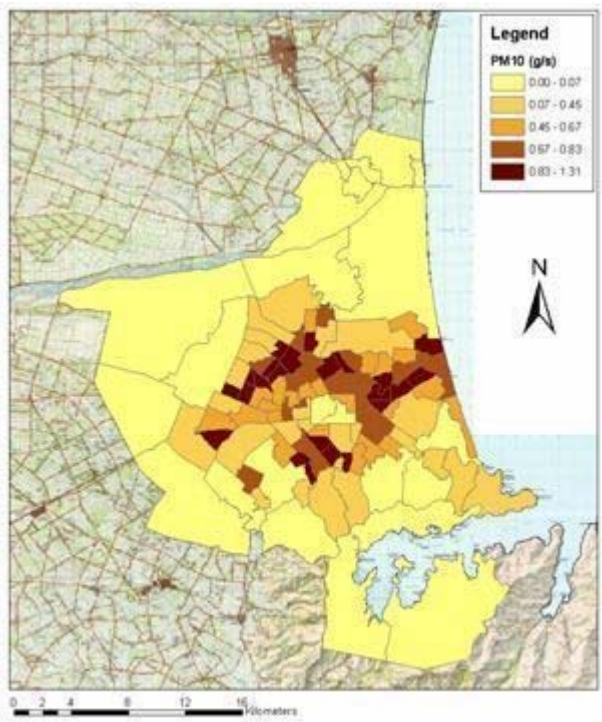


Figure A2-20. CAU map displaying vehicle PM_{10} emissions between 10am and 4pm for a typical winters day in 1999

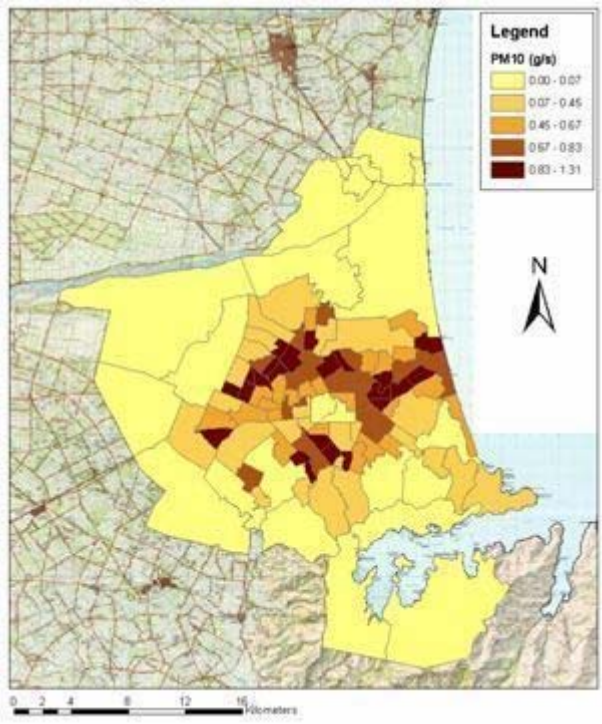


Figure A2-21. CAU map displaying vehicle PM_{10} emissions between 4pm and 10pm for a typical winters day in 1999

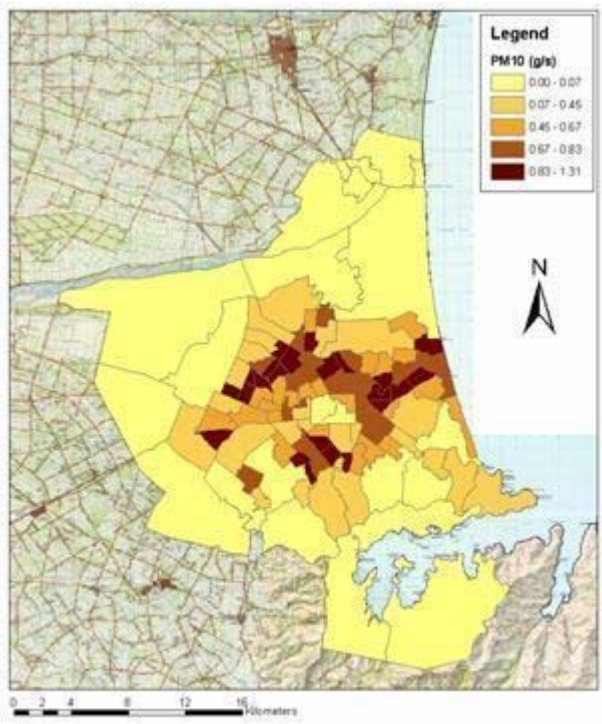


Figure A2-22. CAU map displaying vehicle PM_{10} emissions between 10pm and 6am for a typical winters day in 1999

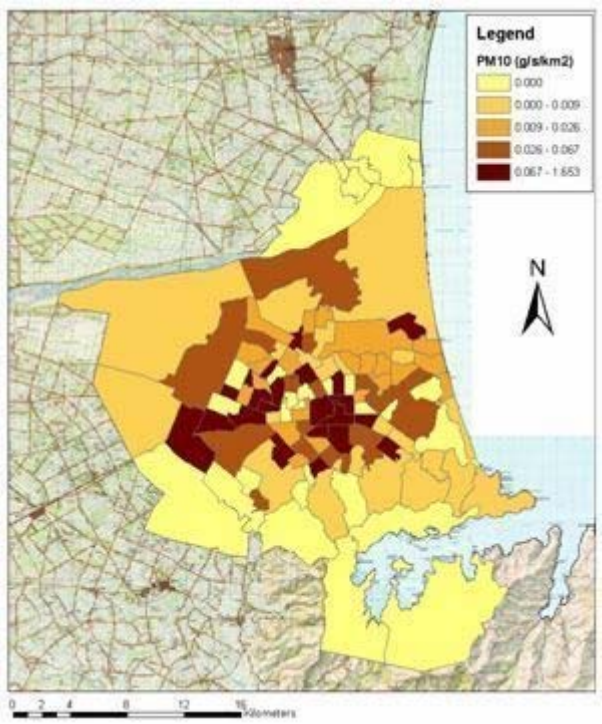


Figure A2-23. CAU map displaying daily industrial PM_{10} emissions for a typical winters day in 1999

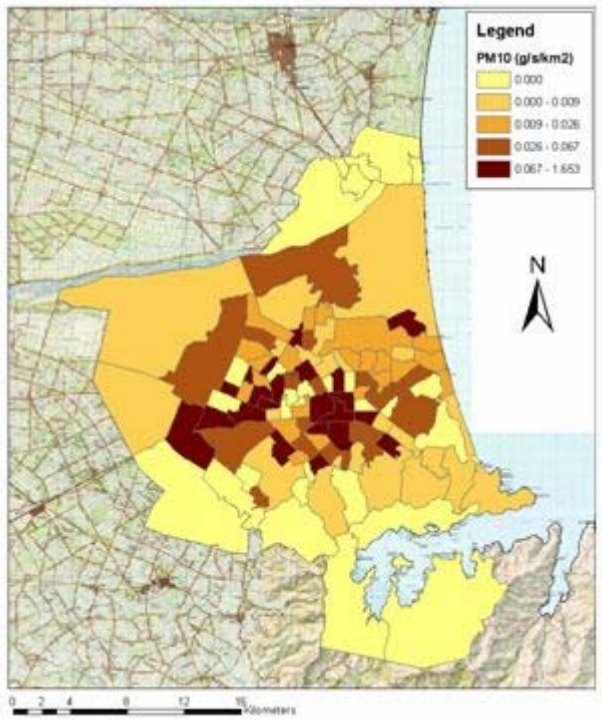


Figure A2-24. CAU map displaying industrial PM_{10} emissions between 6am and 10am for a typical winters day in 1999

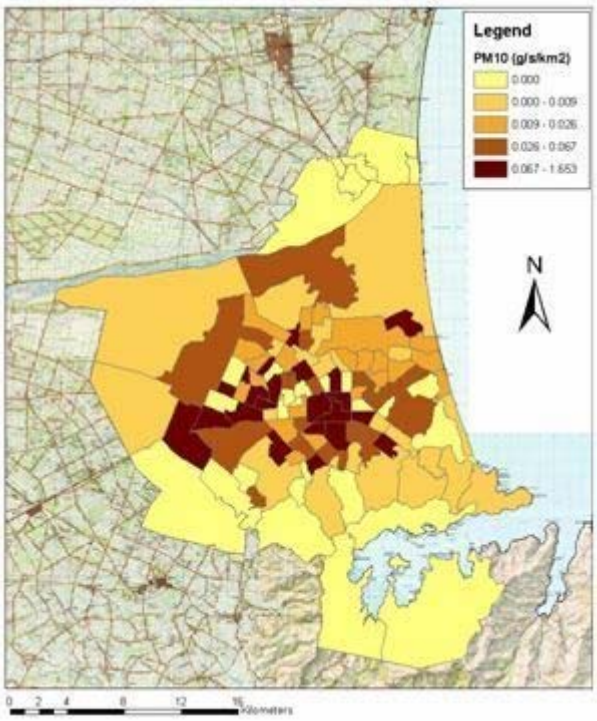


Figure A2-25. CAU map displaying industrial PM_{10} emissions between 10am and 4pm for a typical winters day in 1999

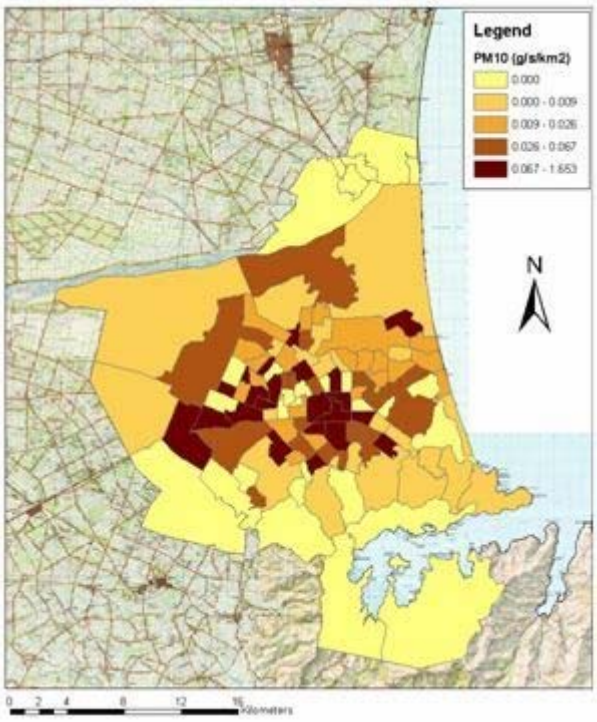


Figure A2-26. CAU map displaying industrial PM_{10} emissions between 4pm and 10pm for a typical winters day in 1999

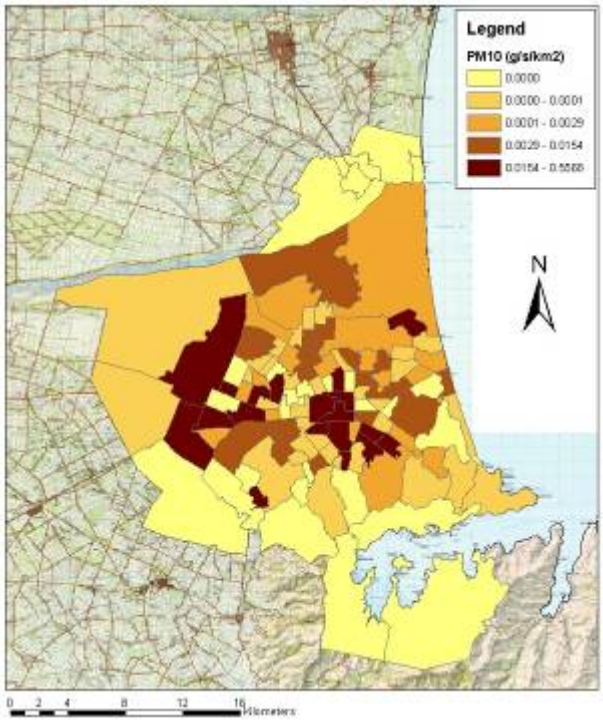


Figure A2-27. CAU map displaying industrial PM_{10} emissions between 10pm and 6am for a typical winters day in 1999

1999 Total emissions

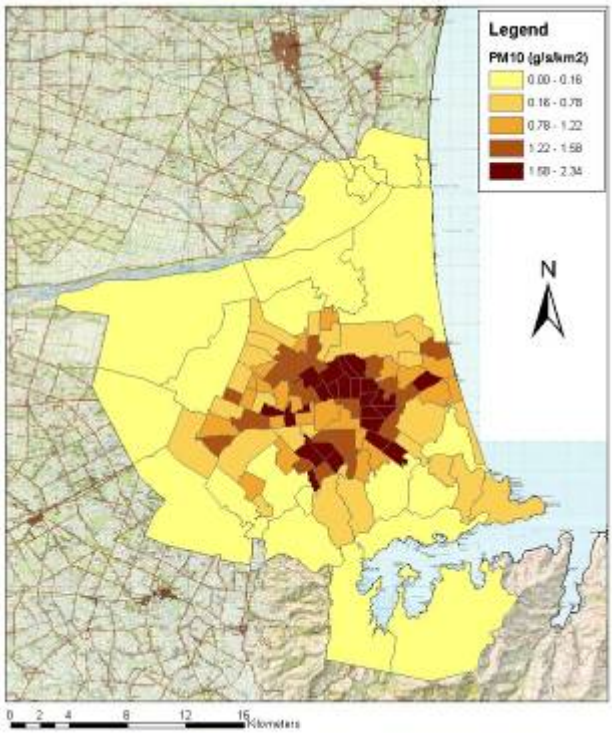


Figure A2-28. CAU map displaying total daily PM_{10} emissions for a typical winters day in 1999

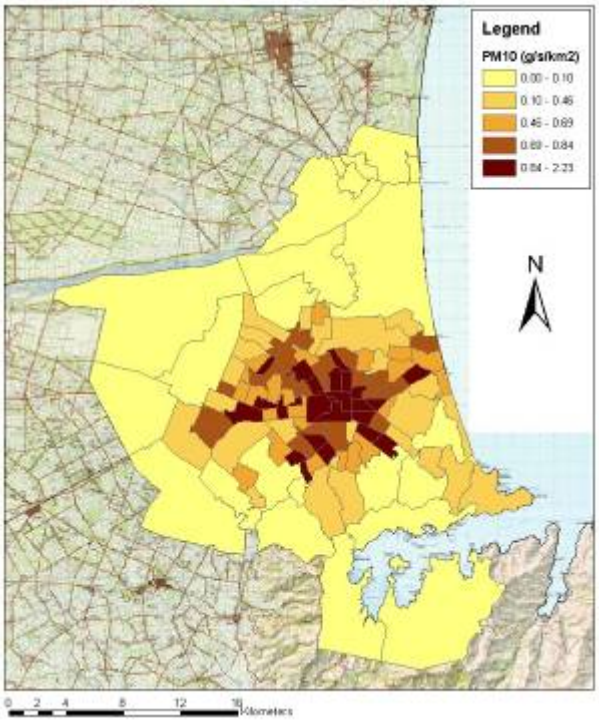


Figure A2-29. CAU map displaying total PM_{10} emissions between 6am and 10am for a typical winters day in 1999

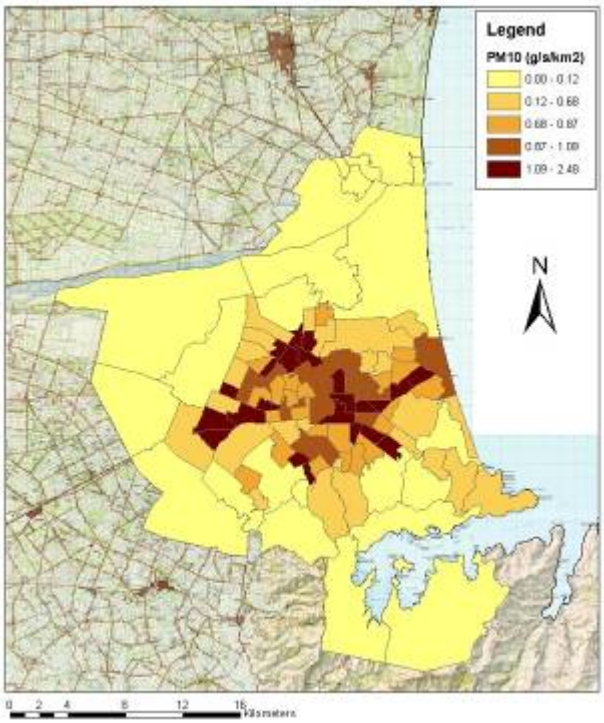


Figure A2-30. CAU map displaying total PM_{10} emissions between 10am and 4pm for a typical winters day in 1999

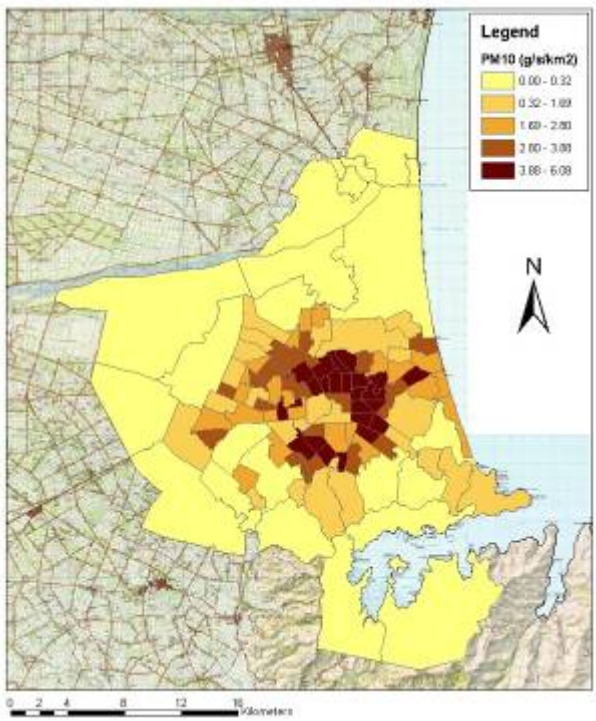


Figure A2-31. CAU map displaying total PM_{10} emissions between 4pm and 10pm for a typical winters day in 1999

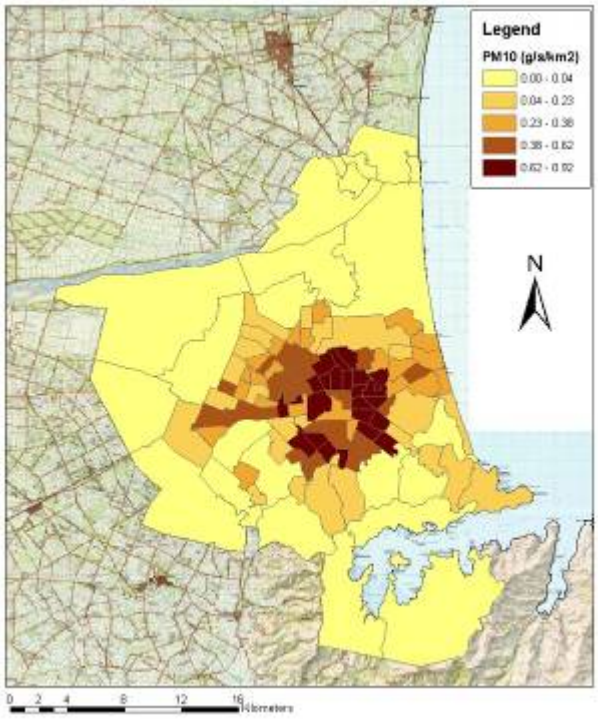


Figure A2-32. CAU map displaying total PM10 emissions between 10pm and 6am for a typical winters day in 1999

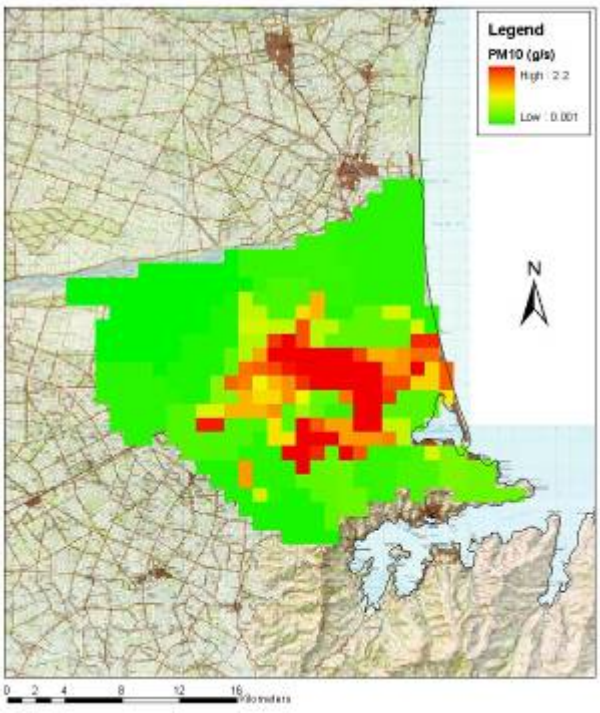


Figure A2-33. Emissions grid of daily domestic PM_{10} emissions

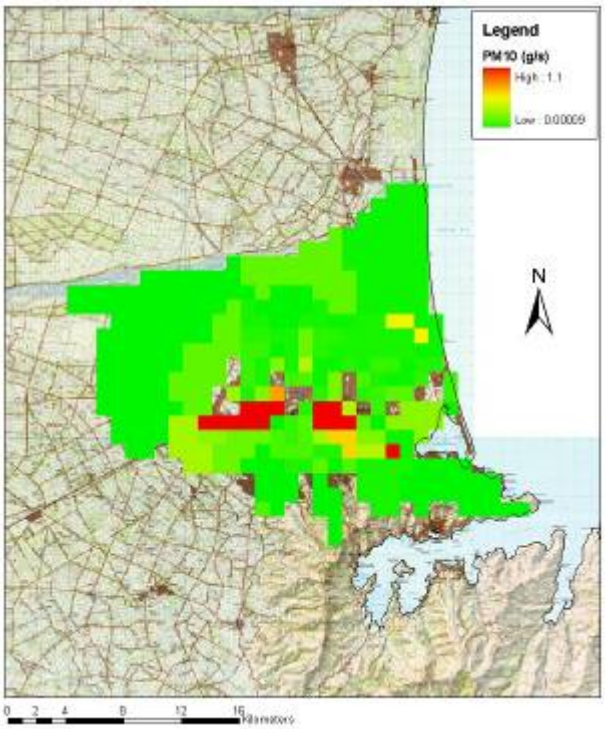


Figure A2-34. Emissions grid of daily industrial PM_{10} emissions

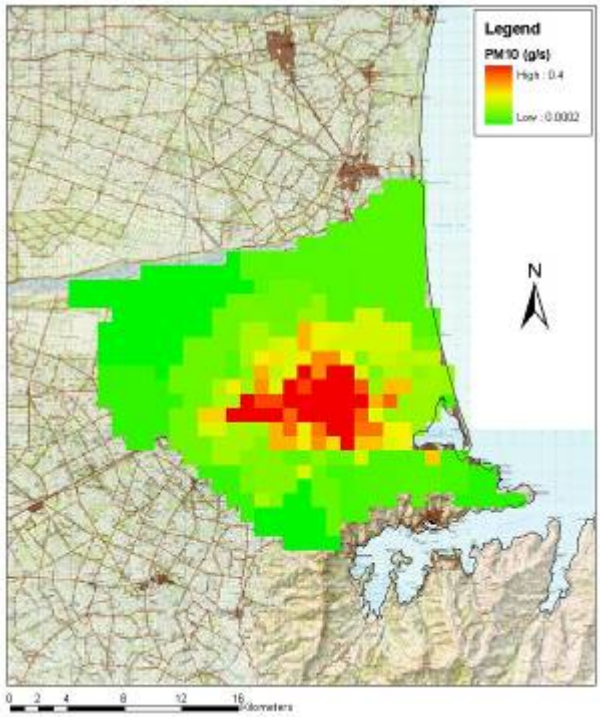


Figure A2-35. Emissions grid of daily vehicle PM_{10} emissions

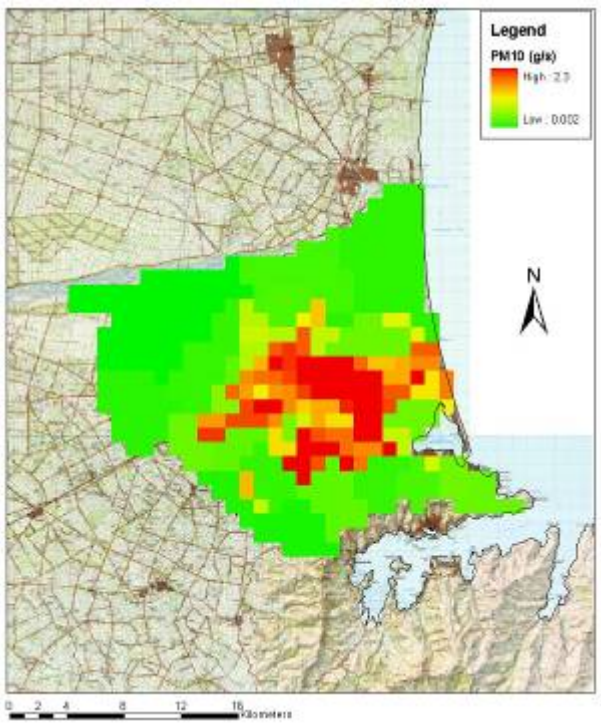


Figure A2-36. Emissions grid for total daily PM_{10} emissions

A2.2.2 Monitoring

Environment Canterbury results

Figures A2-37 and A2-38 show the long-term results of the air quality monitoring undertaken in Christchurch. Figure A2-37 displays the number of days the 50 µg m^{-3} guideline for PM_{10} was exceeded and the maximum recorded value for each year from 1988 to 2002. Figure A2-39 shows the same data for carbon monoxide. Figures A2-40 and A2-41 focus on the year 2002 displaying monthly average, maximum and minimum concentrations at Coles Pl. Figure A2-42 illustrates the winter season (May to August) average hourly concentrations of both PM_{10} and CO.

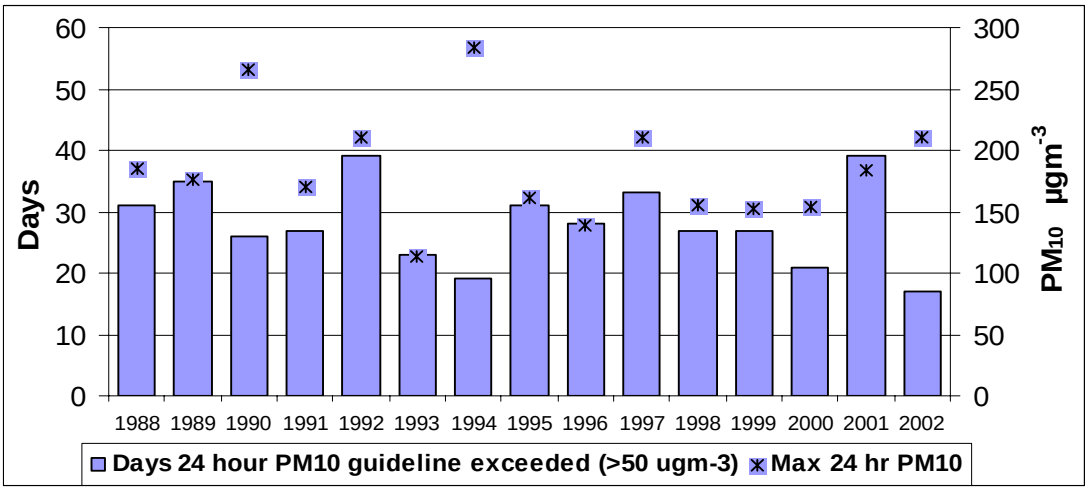


Figure A2-37. Record of 24 hour PM_{10} exceedences in Christchurch from 1988 to 2002 (Environment Canterbury).

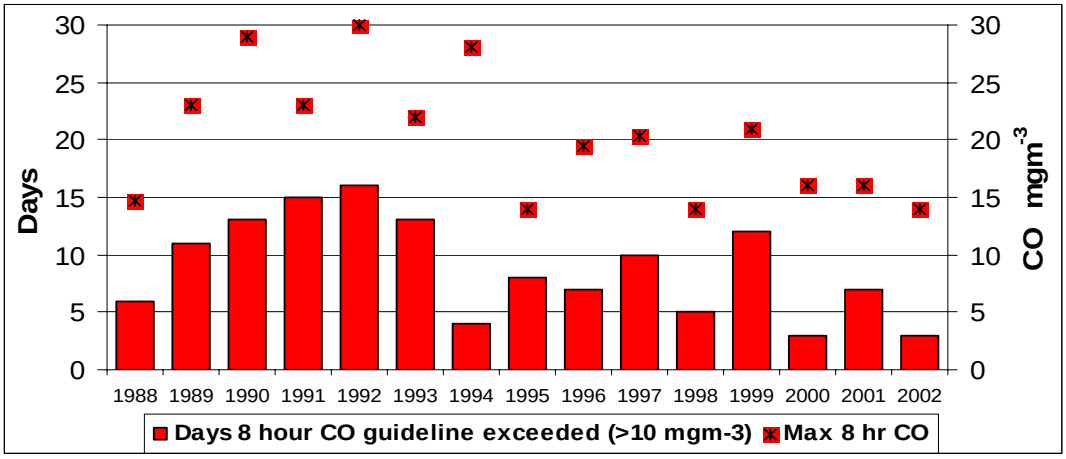


Figure A2-38. Record of hourly CO exceedences in Christchurch from 1988 to 2002 (Environment Canterbury).

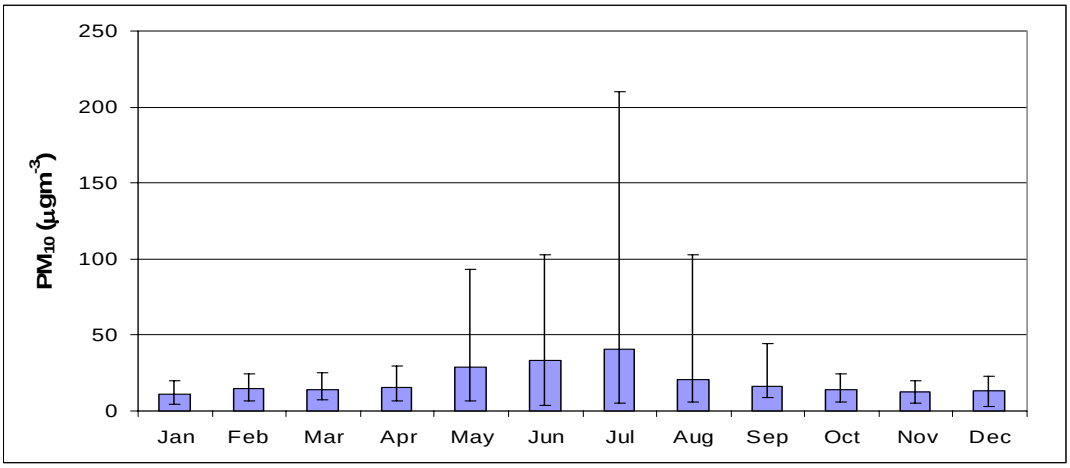


Figure A2-39. Monthly average, maximum and minimum 24-hour PM_{10} concentrations at Coles Pl, 2002.

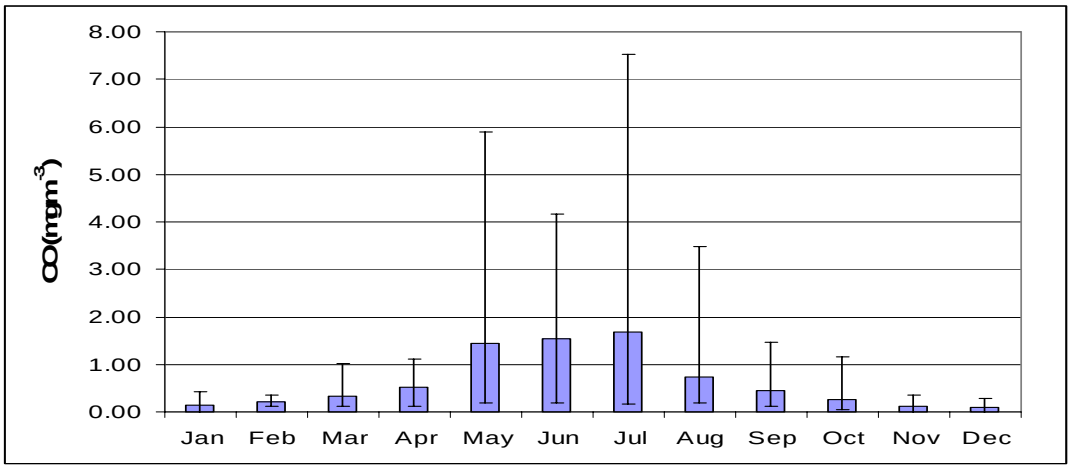


Figure A2-40. Monthly average, maximum and minimum hourly CO concentrations at Coles Pl, 2002.

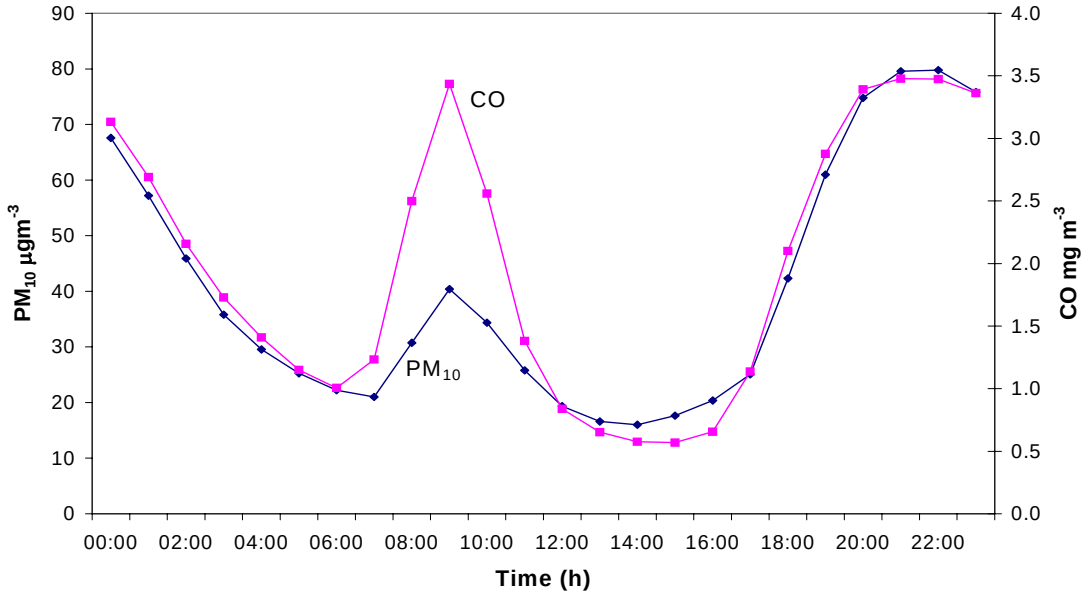


Figure A2-41. May-August hourly average values for PM_{10} and CO at St Albans (Environment Canterbury).

University of Canterbury monitoring results

Results from the 2002 sampling can be seen in Table A2-9. The results from the monitoring show an especially high spatial variation in particulate pollution levels. Figure A2-42 indicates both the city-wide (mesoscale) and neighbourhood level (local scale) concentrations over the period. This preliminary research in 2002 provided evidence that more research in the area of spatial distributions of pollution was necessary in Christchurch and led to the more intensive monitoring detailed in the next section, 2003 Monitoring. Comparison of results from the UC MiniVol at Coles Place and Environment Canterbury's TEOM can be seen in Figures A2-43 and A2-44. The TEOM can be seen to produce lower concentrations values than the MiniVols. This is particularly the case between the 1st and 3rd August, and the results from these days significantly affect any linear relationship between the two. It should be noted that on these dates, all the MiniVols consistently reported high PM_{10} values, thus suggesting that this is not a result of rogue values at the Coles Place MiniVol.

A: Christchurch-wide

	29-Jul		30-Jul		31-Jul		1-Aug		2-Aug		3-Aug		4-Aug		5-Aug		6-Aug		7-Aug		8-Aug	
Location	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**
Cashmere	1.8	42.5	2.2	52.5	2.0	36.7	1.7	43.0	0.7	68.5	1.8	114.4	0.5	14.0	1.0	13.6	0.7	16.9	1.0	14.7	0.6	15.0
Coles Pl	2.5	52.6	3.1	68.2	2.4	58.0	2.2	55.7	0.9	112.0	1.6	121.0	1.0	21.7	1.4	27.3	0.7	12.2	1.2	19.4	0.7	16.1
Colligan	1.9	43.2	3.0	55.7	2.2	26.3	1.8	49.6	1.0	62.1	1.9	122.6	0.7	16.4	1.8	26.1	0.7	10.6	1.3	19.4	1.0	23.4
Linwood	3.1	42.1	4.3	81.0	4.4	78.1	2.8	62.5	0.8	79.6			1.4	19.1	2.6		1.0	20.5	2.0	32.8	1.3	20.0
Marble	2.0	22.8	2.9	59.7	1.5	34.5	2.3	63.5	0.9	108.8	1.2	124.9	0.9	24.3	1.7	31.1	0.5	10.9	0.9	12.6	0.8	18.6
Highland Pl	2.5	34.0	2.5	58.0	2.3	44.9	2.2	59.0	0.8	107.2	2.1	120.5	0.7	24.0	1.8	32.4	0.8	15.9	1.1	13.1	0.6	11.3
Rural	1.0	24.4	1.1	41.7	1.3	27.3	1.2	59.6	0.6	100.2	1.6	206.0	0.7	42.6	1.5	21.0	0.6	3.7	0.6		0.3	4.3
Silver	2.4	47.3	4.0	84.0	1.8	38.7	2.2	56.5	0.7	68.9	1.3	119.7	1.0	28.6	1.2	20.0	0.6	12.9	1.3	18.7	0.9	19.4
Town	2.1	39.4	2.5	53.6	2.2	39.5	2.0	68.1	0.9	103.1	1.4	112.1	0.7	17.0	1.3	11.9	0.9	10.7	1.4	15.3	0.7	15.2
Ecan (Coles)		*26.0*		*35.0*		*39.0*		*20.0*		*15.0*		*24.0*		*16.0*				*7.0*		*8.0*		*16.0*
Mean	2.2	38.7	2.8	61.6	2.2	42.7	2.0	57.5	0.8	90.0	1.6	130.1	0.8	23.1	1.6	22.9	0.7	12.7	1.2	18.3	0.8	15.9
Median	2.1	42.1	2.9	58.0	2.2	38.7	2.2	59.0	0.8	100.2	1.6	120.7	0.7	21.7	1.5	23.5	0.7	12.2	1.2	17.0	0.7	16.1
St. dev.	0.6	10.0	0.9	13.8	0.9	16.3	0.5	7.5	0.1	20.0	0.3	30.9	0.3	8.6	0.5	7.6	0.2	4.8	0.4	6.5	0.3	5.6
Min	1.0	22.8	1.1	41.7	1.3	26.3	1.2	43.0	0.6	62.1	1.2	112.1	0.5	14.0	1.0	11.9	0.5	3.7	0.6	12.6	0.3	4.3
Max	3.1	52.6	4.3	84.0	4.4	78.1	2.8	68.1	1.0	112.0	2.1	206.0	1.4	42.6	2.6	32.4	1.0	20.5	2.0	32.8	1.3	23.4

b: In/around Coles Place

	9-Aug		10-Aug		11-Aug		12-Aug		13-Aug		14-Aug		15-Aug		16-Aug		17-Aug		18-Aug		19-Aug	
Location	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**	**Abs**	**Conc**
Coles Pl	1.4	26.0	6.7	110.3	1.4	64.2	0.3	17.1	0.7	22.5	1.9		0.3	11.2	1.0	28.6	0.4	16.3	0.7	13.2	3.3	52.0
Cornwall	1.8	28.4	7.6	127.8	1.9	25.0	0.4	8.3	1.1	43.8	2.2	289.2	0.5	9.1	1.2	22.7	0.5	18.9	0.8	18.7	4.0	65.0
Frome	1.0	19.5	5.5	82.0	1.2	8.5	0.3	15.7	1.2	25.8	1.2	19.0	0.3	11.6	1.3	28.7	0.5	15.0	0.7	25.8	4.4	55.6
Harold	1.4	24.3	6.5	100.6	1.4	17.3	0.3	15.0	0.9	30.0	1.8	11.6	0.3	47.7	1.0	12.8	0.4	18.9	0.7	21.0	3.3	
Huggins	1.6	25.8	7.6	157.2	2.1	32.4	0.3	23.2	1.1	32.5	2.0	49.4	0.5	10.7	1.5	16.6	0.5	15.5	0.8	6.8	3.4	53.6
Kinnersley	1.3	25.5	6.2	85.1	1.3	15.9	0.3	17.9	0.8	27.7	1.5	15.5	0.4		1.0	8.7	0.5	12.2	0.7	24.1	3.1	34.1
Redwood	1.7	26.3	5.9	414.7	1.1	16.5	0.6	16.1	1.8	31.6	1.4	19.4	0.3	8.1	1.2	22.6	0.5	6.6	0.7	15.0	3.2	49.7
Sheppard	1.6	25.7	6.6	111.8	1.4	23.5	0.4	4.6	0.8	25.9	1.7	19.9	1.0	8.6	1.5	19.0	0.8	11.8	0.8	15.6	3.3	53.4
Trist	1.5	23.9	7.9	132.0	2.3	10.5	0.4	10.5	1.1	29.3	2.3	19.9	0.5	6.1	1.3	16.8	0.5	16.6	0.8	8.1	3.3	54.9
Ecan (Coles)		*9.0*		*103.0*		*14.0*		*9.0*		*27.0*		*20.0*		*6.0*		*9.0*		*8.0*		*11.0*		*43.0*
Mean	1.5	25.0	6.7	146.8	1.6	23.8	0.4	14.3	1.1	29.9	1.8	55.5	0.5	14.1	1.2	19.6	0.5	14.6	0.7	16.5	3.5	52.3
Median	1.5	25.7	6.6	111.8	1.4	17.3	0.3	15.7	1.1	29.3	1.8	19.6	0.4	9.9	1.2	19.0	0.5	15.5	0.7	15.6	3.3	53.5
St. dev.	0.2	2.5	0.8	103.2	0.4	16.9	0.1	5.6	0.3	6.1	0.3	95.1	0.2	13.7	0.2	6.7	0.1	3.9	0.1	6.6	0.4	8.6
Min	1.0	19.5	5.5	82.0	1.1	8.5	0.3	4.6	0.7	22.5	1.2	11.6	0.3	6.1	1.0	8.7	0.4	6.6	0.7	6.8	3.1	34.1
Max	1.8	28.4	7.9	414.7	2.3	64.2	0.6	23.2	1.8	43.8	2.3	289.2	1.0	47.7	1.5	28.7	0.8	18.9	0.8	25.8	4.4	65.0

Table A2-9. PM monitoring results – Christchurch July/Aug 2002 (missing values left blank, and the high degree of small scale variation is due to local sources)

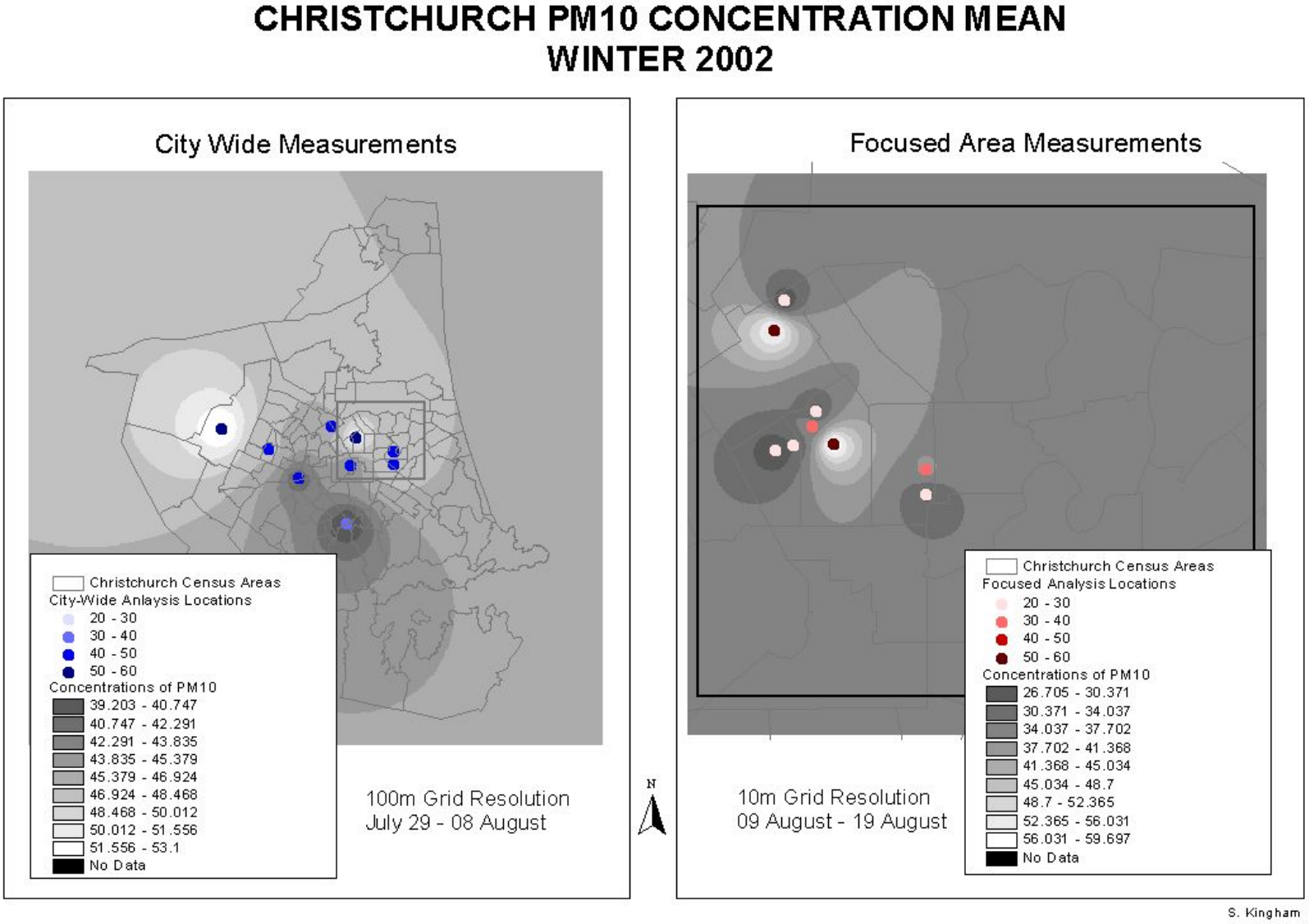


Figure A2-42. Spatial patterns in PM_{10}, Christchurch 2002

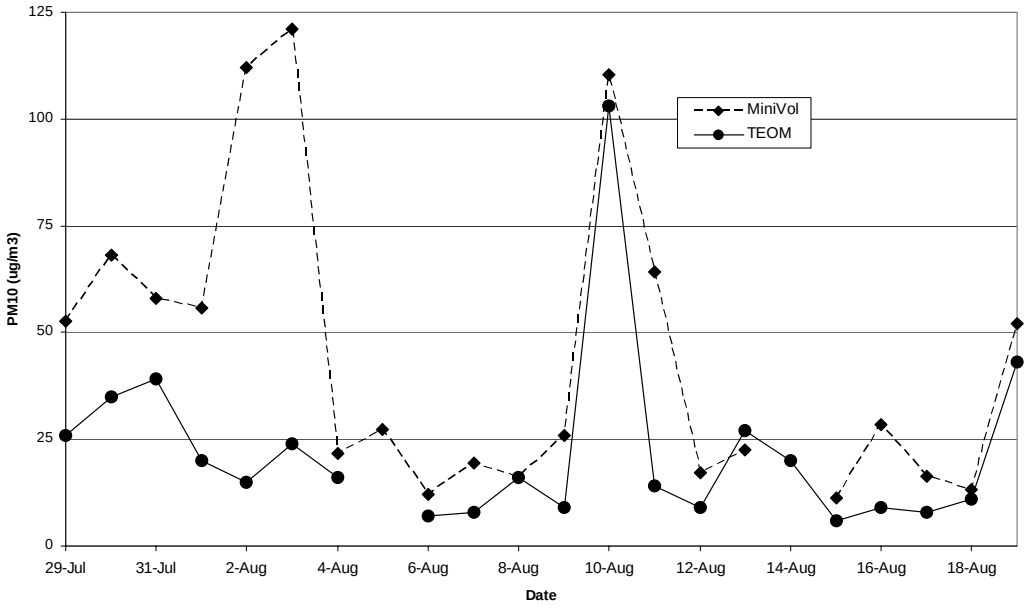


Figure A2-43. Temporal variation in TEOM and MiniVol, Coles Place, July/August 2002.

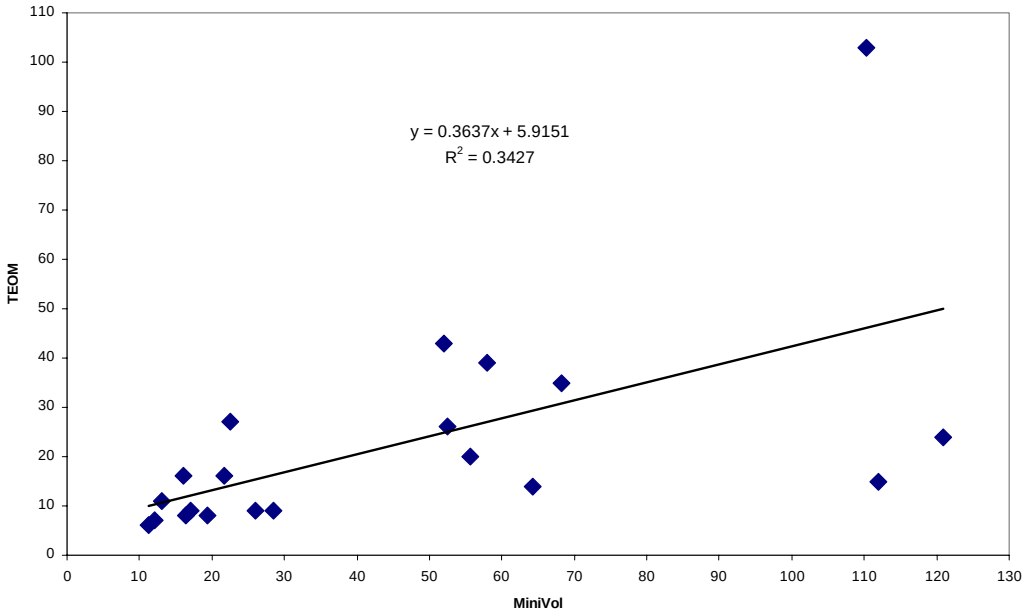


Figure A2-44. Regression of TEOM vs. MiniVol at Coles Place, July-Aug 2002.

Winter 2003 and summer 2004 monitoring results

The results of the July 2003 monitoring can be seen in Figures A2-45 to A2-59. The results show the spatial and temporal variations in PM_{10} across the city. Note the number of occasions where PM_{10} concentrations have exceeded the 50 $\mu g\ m^{-3}$ guideline in Figures A2-45, A2-46, A2-47, A2-48 and Table A2-10. These figures show that during the month of July PM_{10} levels regularly exceeded the guideline and on occasion exceeded 100 $\mu g\ m^{-3}$.

Figure A2-49 compares TEOM data collected by Environment Canterbury with MiniVol measurements at Coles Pl and as outlined in the 2002 results, the TEOM consistently produces lower values. This is also evident in Figure A2-48, which displays the temporal pattern in daily PM_{10} concentrations at Coles Pl for both MiniVol and TEOM.

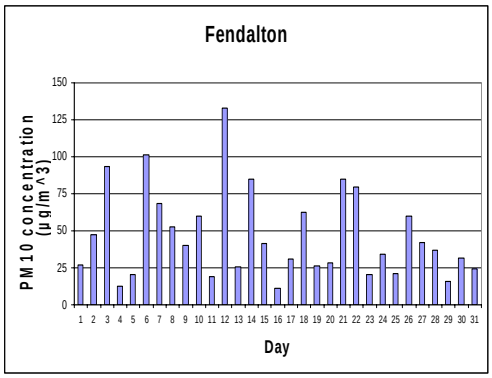
Figure A2-50 and A2-51 show the temporal variation in PM_{10} across all the monitoring sites including the Environment Canterbury TEOM. The figure displays a consistent temporal pattern of PM_{10} pollution across the city with the exception of Mt Pleasant where the MiniVol was located approximately 100m above sea level on the hills and as expected, air pollution levels were the lowest (approximately 10-20 $\mu g\ m^{-3}$). This site should be relocated in subsequent monitoring activities.

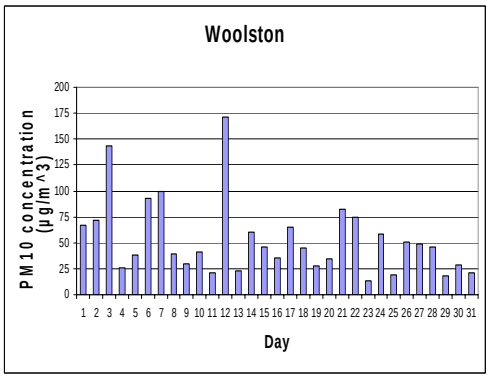
Figure A2-52 displays an interpolated grid derived from the July monthly averages of PM_{10} across all monitoring sites and Figure A2-53 is the CAU map of the same data. The figures give an indication of the spatial pattern of PM_{10} concentrations across the city for the month of July 2003 and show that on average there are higher concentrations of PM_{10} in central Christchurch and near the coast in particular, in St Albans, Woolston and Wainoni.

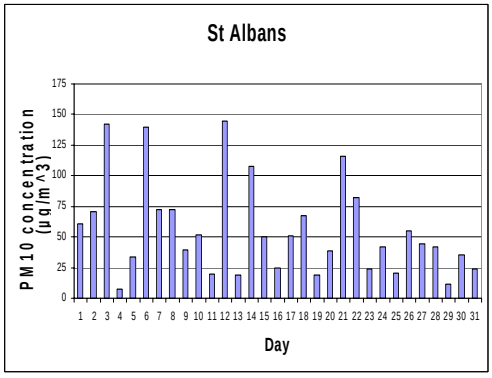
Figures A2-54 to A2-59 are interpolated grids for selected days of varying PM_{10} concentrations during July 2003. The grids range from days where high concentrations of PM_{10} were recorded to days of low PM_{10} levels. These figures illustrate where pollution settles under particular meteorological conditions and the effect meteorology has on PM_{10} concentration and spatial distribution.

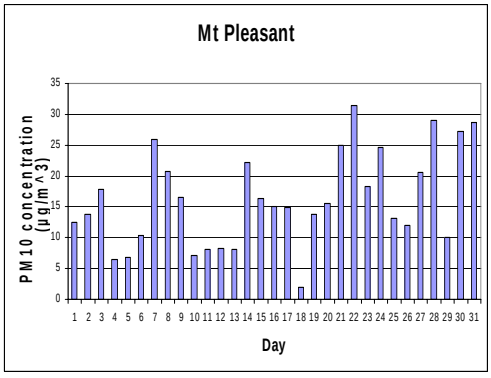
The February 2004 summer monitoring results show that the average values of PM_{10} across all monitoring sites ranged from between 13 $\mu g\ m^{-3}$ to 18 $\mu g\ m^{-3}$ (see Table A2-10). As expected, when compared to the July 2003 monthly averages these values are considerably lower. The February averages represent the baseline ambient levels of PM_{10} pollution in Christchurch city — that is, the levels of PM_{10} experienced in Christchurch without the contribution from home heating, the main seasonally variable source of PM_{10}.

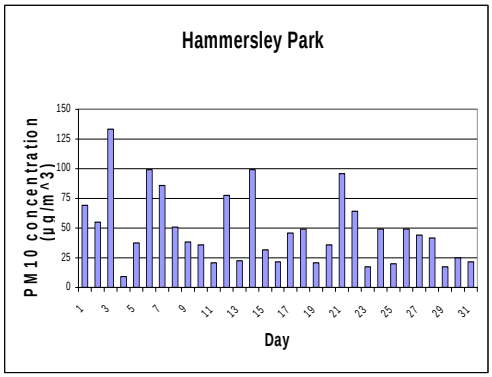
At two monitoring sites during the summer monitoring period concentrations reached as high as 34 $\mu g\ m^{-3}$. These higher levels can be attributed to hot dry days experiencing high wind speeds where levels of dust and pollen in the air can become more concentrated. For example, days of strong northwesterly winds where high levels of dust and pollen are transported across the Canterbury Plains.











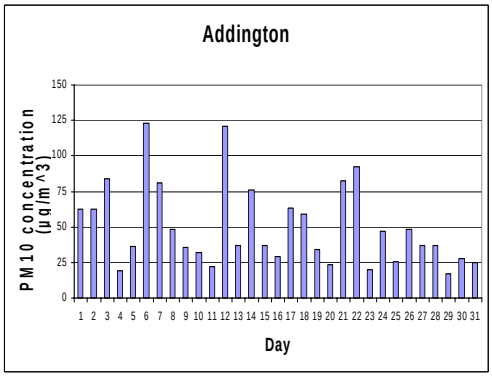
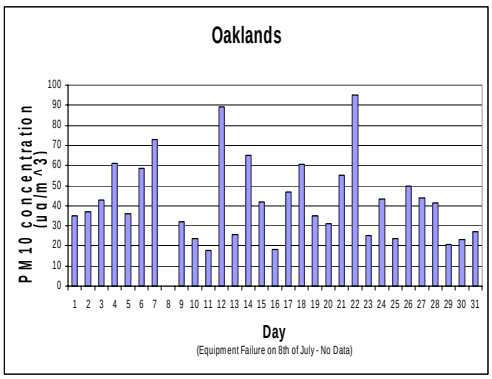
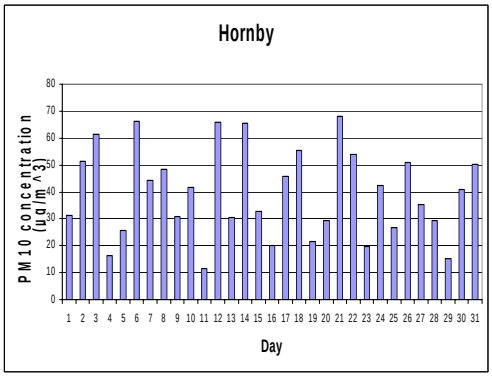


Figure A2-45. Fendalton, St Albans, and Hammersley Park. Daily PM_{10} averages for the month of July 2003

Figure A2-46. Woolston, Mt. Pleasant and Addington. Daily PM_{10} averages for the month of July 2003





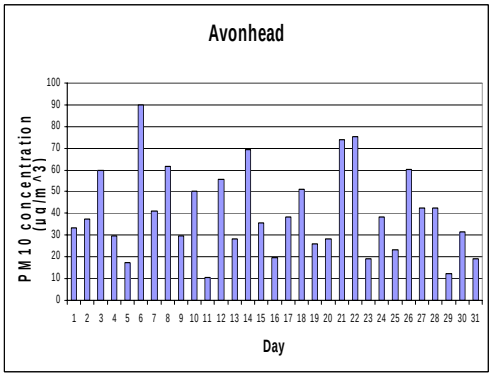


Figure A2-47. Oaklands, Hornby and Avonhead. Daily PM_{10} averages for the month of July 2003

		Mean	*Max*	*Min*	Days over guideline
UoC	**FEND**	*46*	*133*	*10*	11
	COLE	*56*	*144*	*7*	15
	HAMP	*48*	*133*	*8*	10
	WAIN	*51*	*154*	*9*	12
	CHCE	*44*	*149*	*7*	10
	WOOL	*53*	*171*	*14*	12
	MTPL	*16*	*32*	*2*	0
	ADDN	*50*	*123*	*17*	11
	OAKL	*43*	*95*	*18*	9
	HORN	*40*	*68*	*11*	9
	AVON	*40*	*90*	*11*	10
ECAN	**COLE**	*38*	113	*5*	7

Table A2-10. Monitoring sites with mean monthly value for July 2003 and numbers of days over the MfE guideline

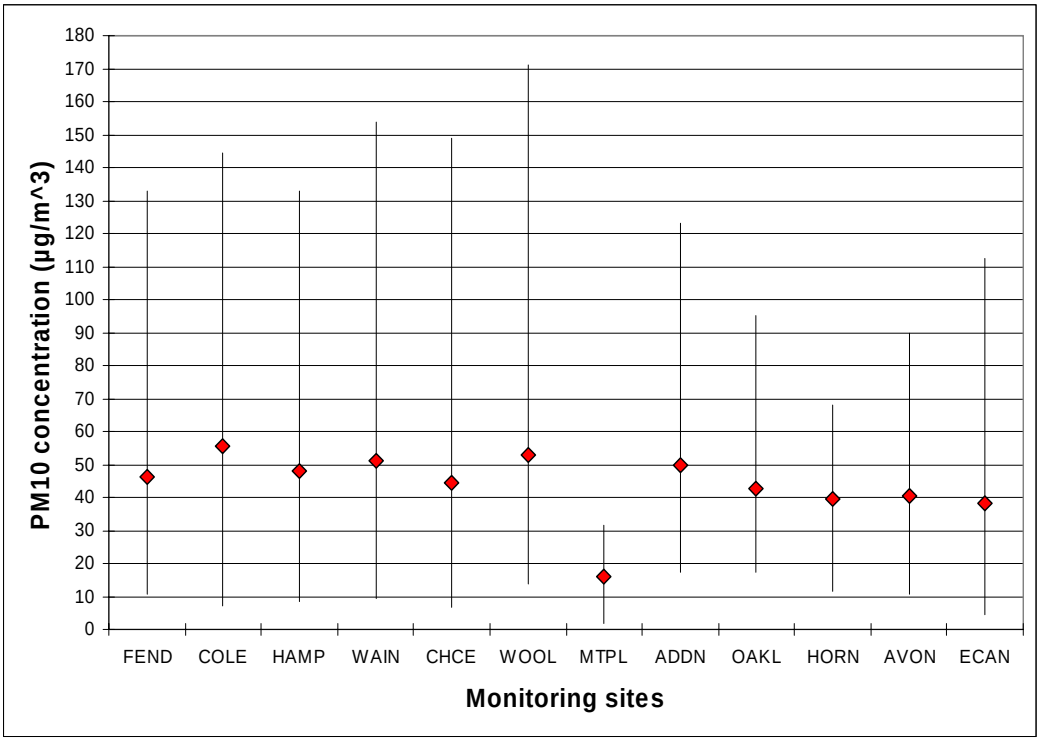


Figure A2-48. Monthly PM_{10} average, with maximum and minimum ranges

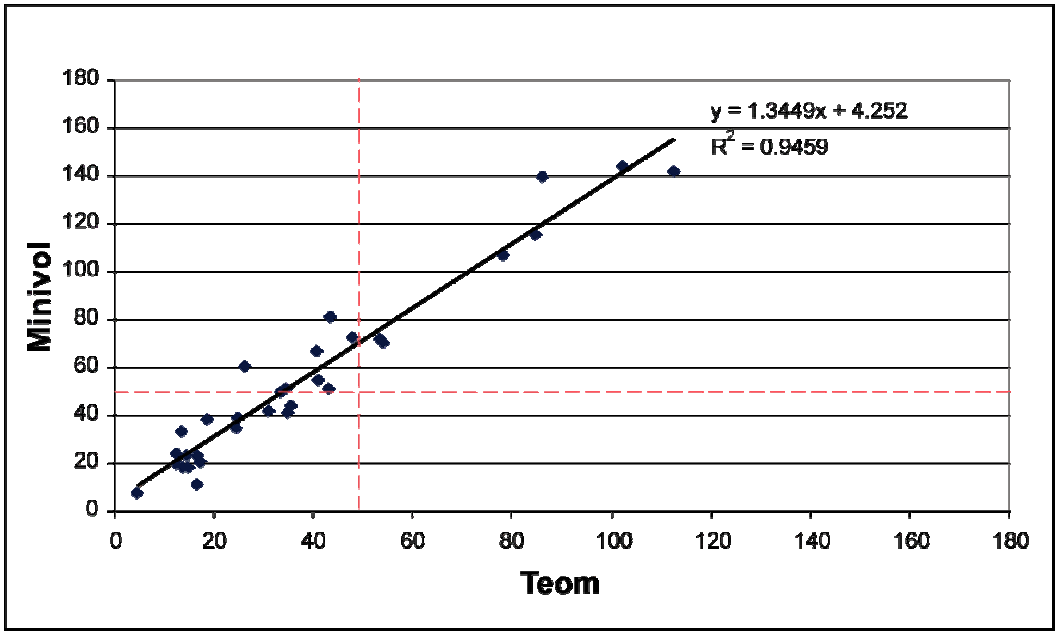


Figure A2-49. Regression of TEOM vs. MiniVol adjusted for temperature and pressure. St Albans (Coles Place), July 2003

Not all of the University of Canterbury, Geography Department's MiniVols were available for the summer monitoring. An extra unit was provided by NIWA and installed at MT Pleasant. It was noticed during the monitoring that the MiniVol was giving higher than expected values (see Mt Pleasant minimum value in Table), and once monitoring had been completed the MiniVol was tested using a Buck flow rate monitor. It was observed that the flow rates were higher than those indicated by the MiniVol and those calculated using the MiniVols calibration equation, therefore suggesting that the results obtained from the instrument are higher than the actual levels of PM_{10}

experienced at the site. Corrections were made using the buck calibration data and a value of 0.5 litres/minute was added to the actual flow rate to compensate for the difference.

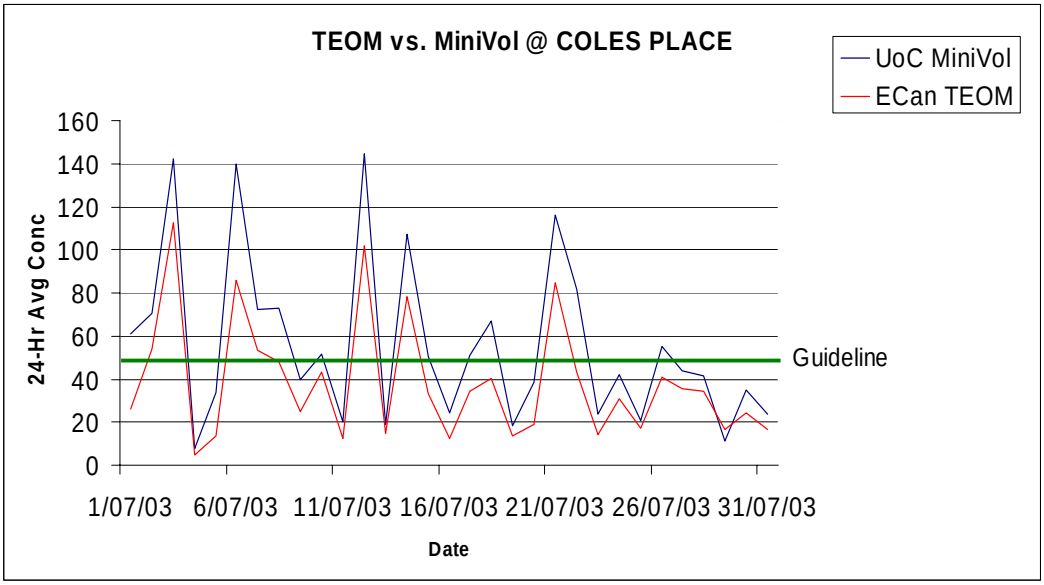


Figure A2-50. Comparison of daily averages from the TEOM and MiniVol. St Albans (Coles Place), July 2003.

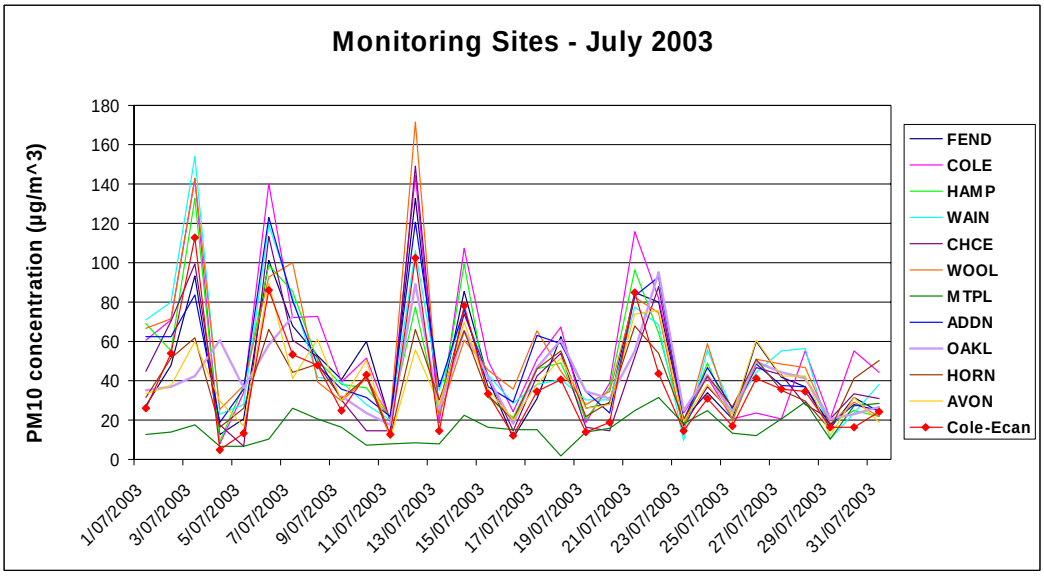


Figure A2-51. Daily running averages for all Monitoring Sites. July 2003.

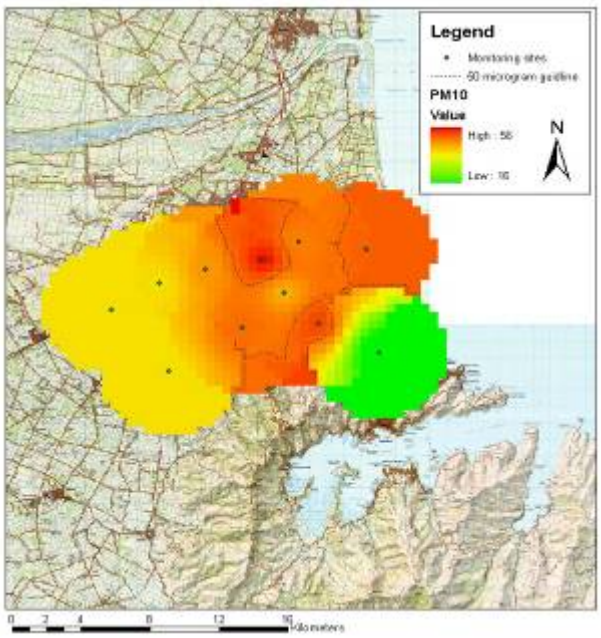


Figure A2-52. 500m grid displaying monthly average PM_{10} in μgm^{-3} for July 2003

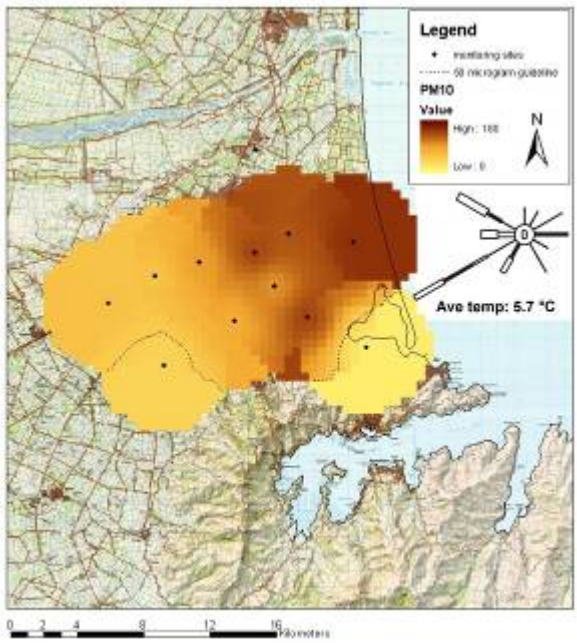


Figure A2-54. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 3rd – 4th July 2003.

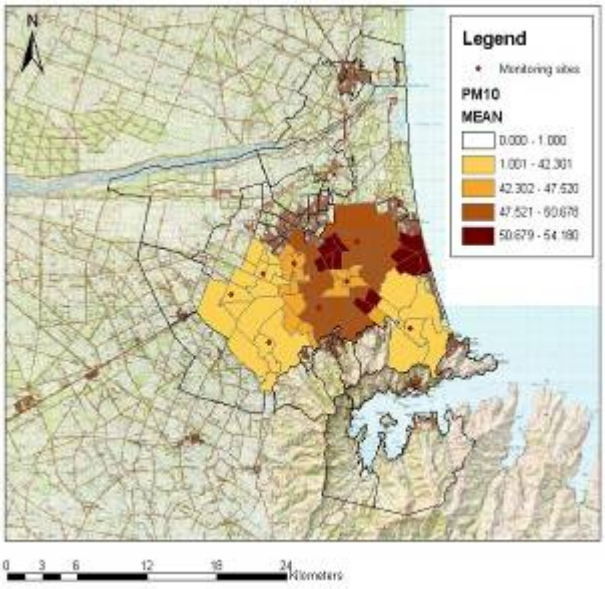


Figure A2-53. Census area unit map displaying monthly average PM_{10} in μgm^{-3} for July 2003 derived from interpolated monitoring data

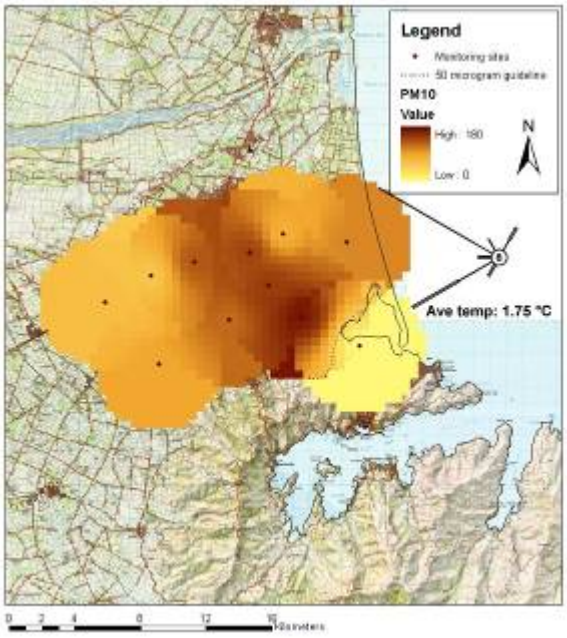


Figure A2-55. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 12th – 13th July 2003

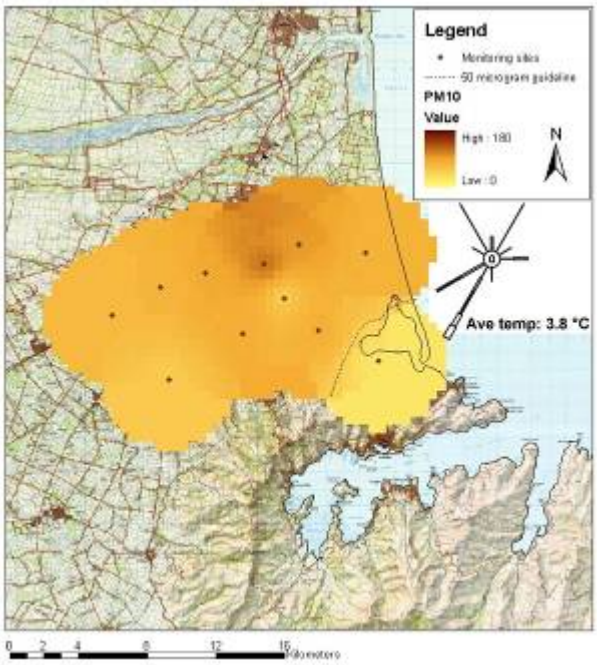


Figure A2-56. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 21st – 22nd July 2003

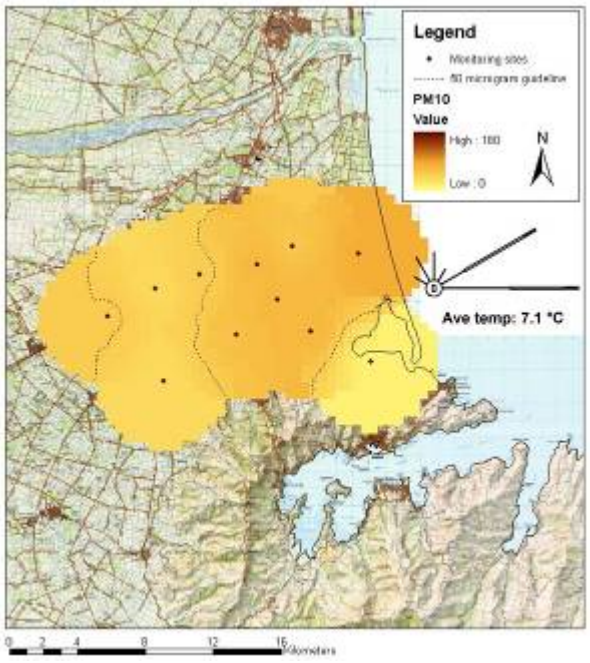


Figure A2-57. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 2nd – 3rd July 2003

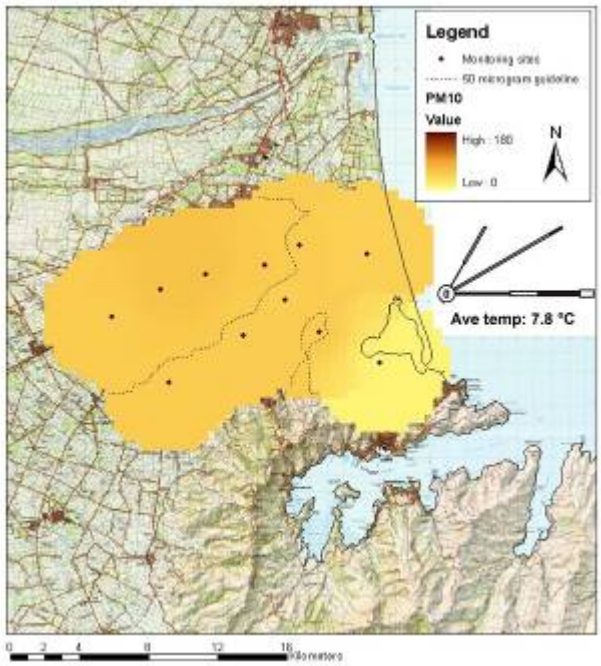


Figure A2-58. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 26th – 27th July 2003

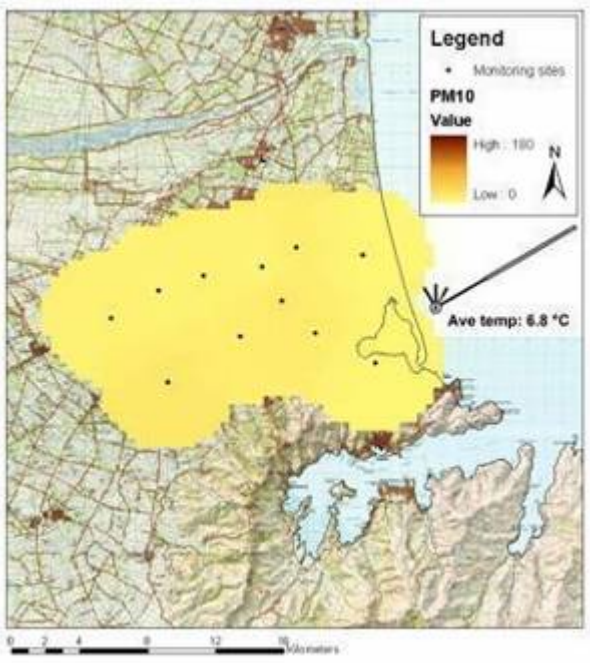


Figure A2-59. 500m grid displaying 24hr average PM_{10} in μgm^{-3} for 29th – 30th July 2003

Site	Max	Min	Mean
FEND	27	6	17
COLE	23	6	13
HAMP	25	4	14
WAIN	26	6	14
CHCE	25	1	15
WOOL	23	6	15
MTPL	21	9	12
ADDN	29	3	17
OAKL	24	6	16
HORN	34	5	18
AVON	35	3	18

Table A2-11. Summary table of summer (February) 2004 monitoring results.

A2.2.3 Airshed modelling

Airshed modelling with TAPM is approached in two ways. The first method looks at simulating meteorology and dispersion for case study periods that were performed for July 2003. The second method looks at simulating a yearly cycle of meteorology and dispersion, which is performed for 1999. The model settings – outlined above – differ slightly between the two methods but should not affect the results very much.

July 2003 TAPM modeling results

Two simulations for July 2003 are performed using two different inventories. The surface wind field is a typical northeasterly flow over the city. Visual inspection of the particulate matter plume shows that, qualitatively, there is very little difference in concentrations produced by each inventory. Figure A2-60 shows the plume shape at 1800 NZST for July 12; the extent and coarse features are almost the same for both simulations except for the middle of the domain in the simulation with the census data where the 130 μg m^{-3} contour line is situated. Since the meteorology is the same, differences arise due to variation in emission rate at each node.

The accepted standard way to determine the quality of modelled meteorological data is through calculating the Index of Agreement between the observed (O) and the predicted (P) values. The IOA is defined as (the bar indicates average values):

$$IOA = 1 - [\overline{(P-O)^2} \,/\, \overline{(|P-\overline{O}|+|O-\overline{O}|)^2}]$$

IOA can range from 0.4 (minimum value achieved between two datasets that are randomly generated), to 1 (for perfect agreement); values above 0.6 are considered to be good and are considered to show good skill by the model. IOA for the simulation of July 2003 meteorology for u-component (east-west component of wind velocity), v-component (north-south component of wind velocity), and wind speed at the St. Albans monitoring station are 0.71, 0.71, and 0.53 respectively. The IOA for temperature and relative humidity are 0.85 and 0.58. It has to be noted that IOA is highly dependent on chosen model parameters and through testing it can be improved.

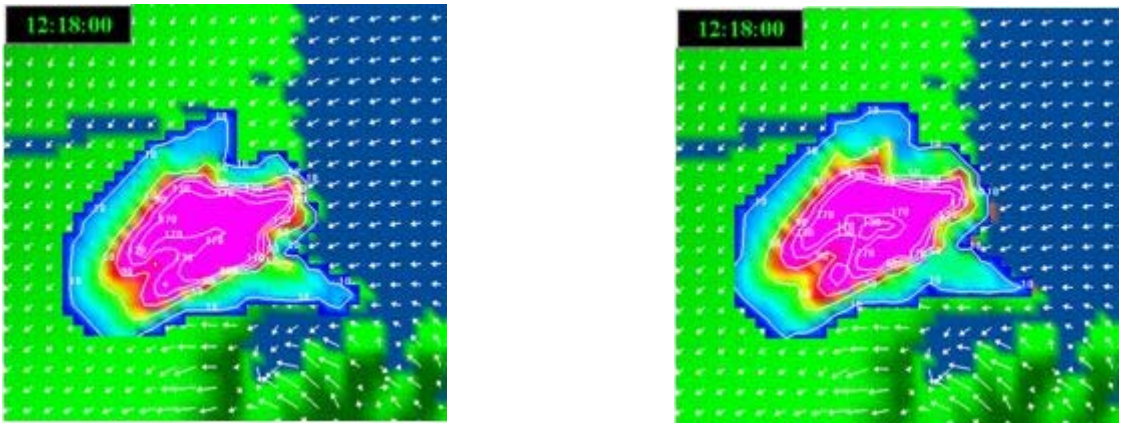


Figure A2-60. Surface wind field (simulated) and PM_{10} concentration contour lines (contour intervals of 40 µg m^{-3}) at 1800NZST on July 12. The panel on the left is with Environment Canterbury emission inventory, and the panel on the right is with census inventory.

Figure A2-61 shows the comparison of 24-hour-averages of measured and modeled data at Coles Place. The model has a tendency to underestimate – by a considerable margin – PM_{10} concentrations for the July 1, 3, and 6. However, the model then tends to generally overestimate concentrations at the end of the month. Putting aside causes such as poor simulation of physical mechanisms that control dispersion – such as temperature inversion strength – and the coarse temporal and spatial resolution of the emission inventory, using paired-in-time average values for inter-comparison of data (especially the 24-hour average) can be misleading. Since averages are highly sensitive to outlier values. It is more acceptable to show results in percentiles, as described below.

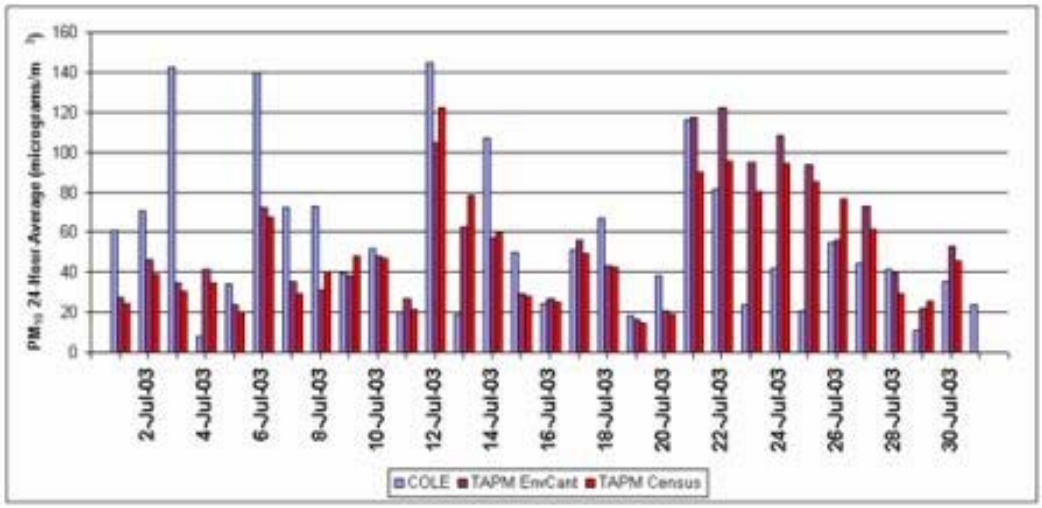


Figure A2-61. Measured and modelled 24-hour-averages of PM_{10} for July 2003 at Coles Place

The results for the ground level concentration of PM_{10} are presented as percentiles for the month of July 2003 in Figure A2-62. To mitigate the undesirable influence of unusual events on the highest observed and predicted values the Robust Highest Concentration (RHC) is also calculated here. RHC is defined as:

$$RHC = C(R) + \left(\overline{C} - C(R)\right)\ln\left((3R-1)/2\right)$$

Where $C(R)$ is the R^{th} highest concentration and $\overline{C}$ the mean of the top $R-1$ concentration. The value of R = 11 is used here so that $\overline{C}$ is the average of the top top-ten concentrations.

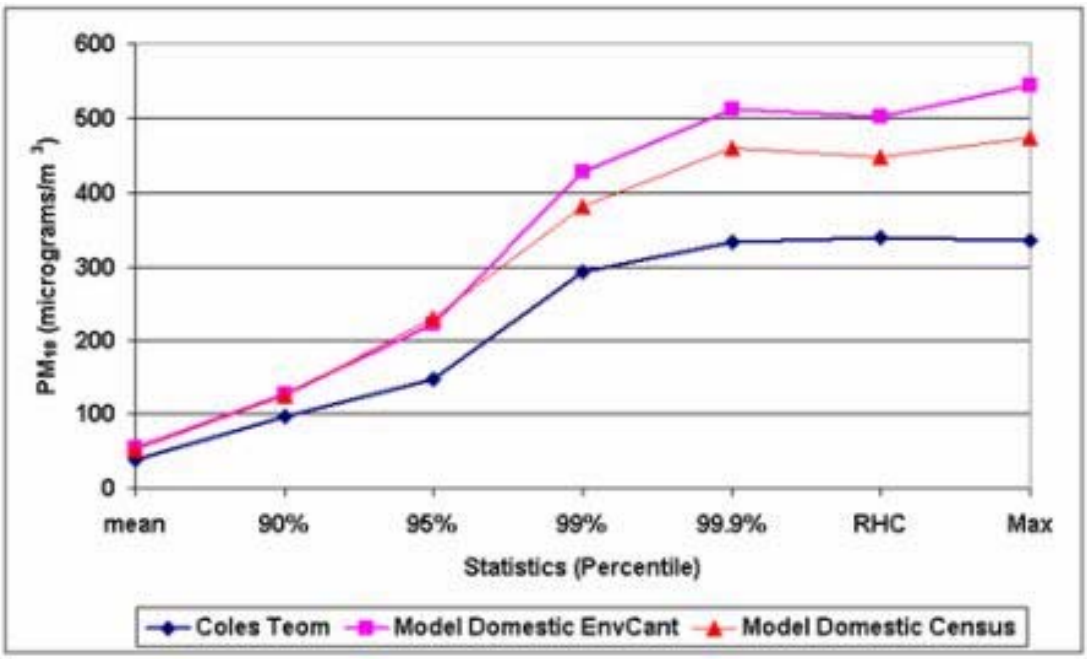


Figure A2-62. Statistics for measured and modelled ground level concentration of PM_{10} (hourly-averages) at the Coles Place. RHC is the Robust Highest Concentration as defined in the text.

Above the 90th percentile, the model has a tendency to overestimate PM_{10} concentrations Figure A2-63. But it is encouraging to see that the modelled and measured means and the 90th percentiles are much closer. It is known that TEOM has a tendency for underestimating PM_{10}. Therefore as far as hourly averages between the mean and the 90th percentile are considered, model results are acceptable.

Comparison is also made with the 24-hourly averaged data gathered with the MiniVols at eleven locations around Christchurch (Figures A2-64 and A2-65). Generally the model underestimates the monthly averages. However, the modelled values should increase once the contribution to emissions from transport and industry are added.

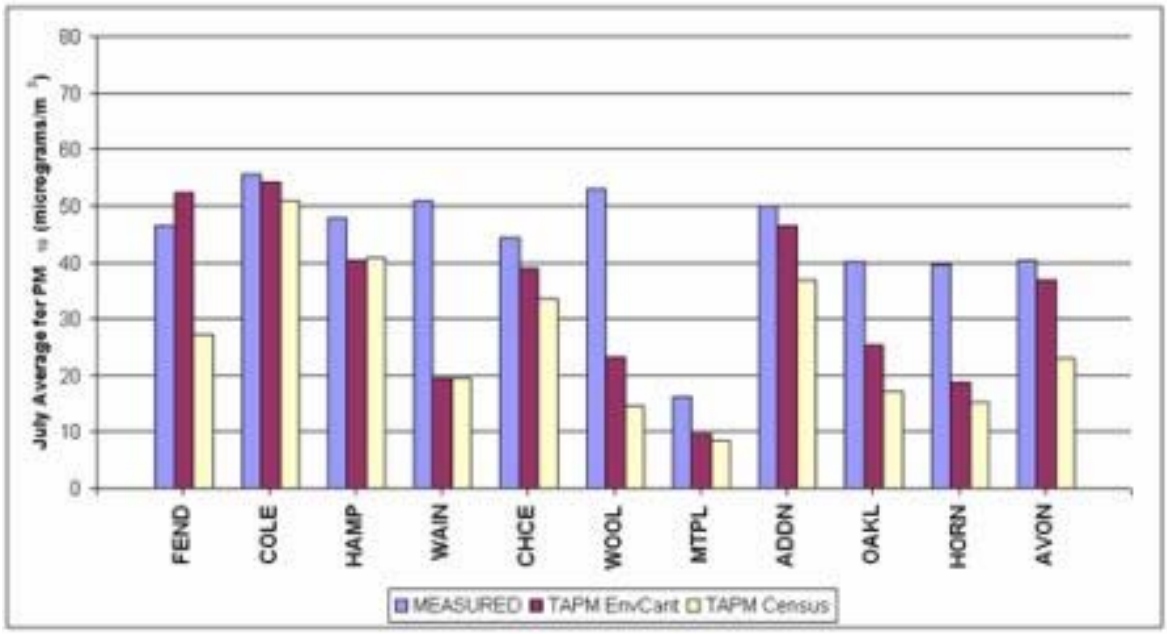


Figure A2-63. Measured and modelled averages for July 2003.

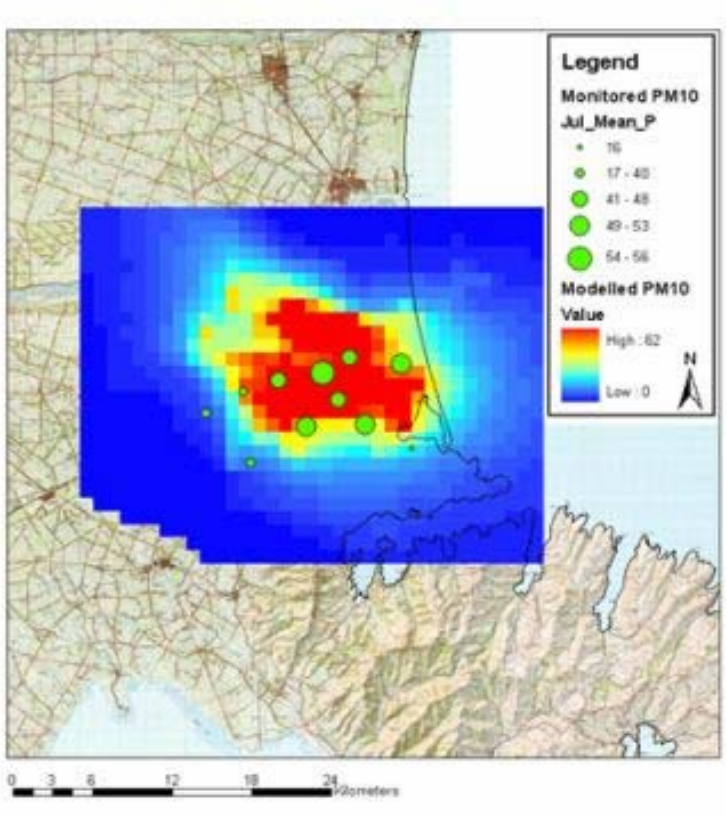


Figure A2-64 Modelled monthly PM_{10} averages for July 2003 from the Environment Canterbury emissions inventory with monitored data for comparison.

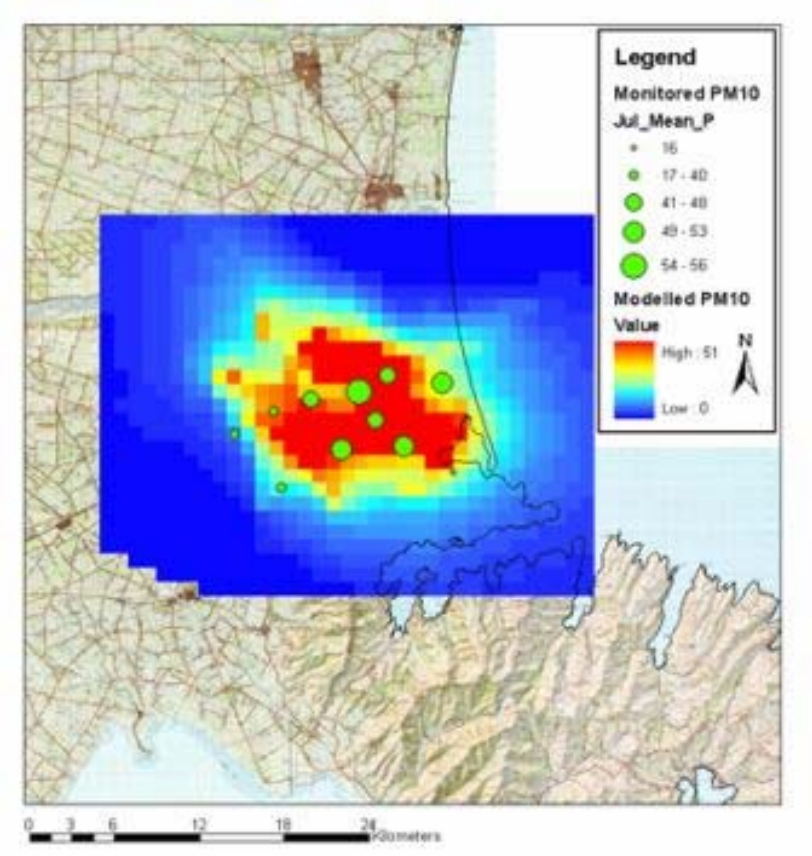


Figure A2-65 Modelled monthly PM_{10} averages for July 2003 from the census regression analysis with monitored data for comparison.

CAU	AVG	MIN	MAX
586001			
586002			
586401			
586402			
586501			
586502			
587301	40	40	40
587302	40	40	40
587400	40	40	40
587500	40	40	40
587701	41	40	41
587702	40	40	43
587811	40	40	40
587812	40	40	40
587820	40	40	40
587830	40	40	40
587841	40	40	40
587843	41	40	46
587844	42	40	45
587902			
587903	40	40	40
588101	48	48	48
588102	52	48	53
588200	54	51	56
588300	48	46	50
588400			
588500	46	46	46
588600	45	43	46
588700	45	44	46
588800	42	40	44
588900	43	43	44
589000	44	44	45
589100	45	44	46
589200	46	46	47
589300	48	48	50
589400	45	45	46
589500	46	46	47
589601	41	40	42
589602	41	41	41
589700	42	41	44
589800	40	40	40
589900	40	40	41
590000	41	40	43
590100	42	41	44
590200	44	43	45
590300	47	44	49
590400			
590500	50	48	53
590601	49	48	51
590602	50	49	51
590700	46	45	48
590800	48	47	49
590900	49	49	50
591101	50	50	51
591102	51	50	53
591200	37	16	53
591300	17	16	27
591500	47	45	49
591600	49	48	50
591700	46	44	49
591800	52	51	56
591900	51	50	52
592000	48	46	49
592100	50	47	53
592200	51	50	52
592300	53	52	55
592401	54	52	55
592402	54	51	56
592500	52	52	53
592600	50	48	53
592701	50	49	51
592702	48	48	49
592811	49	49	50
592812	49	49	50
592820	50	50	51
592900	50	49	51
593000	50	50	51
593100	51	51	51
593200	49	49	49
593300	48	46	48
593400	49	47	50
593501	46	46	47
593502	49	46	51
593600	49	49	50
593700	50	50	51
593800	51	51	51
593900	41	25	52
594010	52	51	53
594020	38	19	51
594100	46	38	52
594200	52	50	53
594300	50	46	53
594400	51	50	52
594500	49	48	50
594600	49	48	50
594700	50	50	50
594800	49	48	50
594900	49	47	50
595000	50	50	50
595100	47	46	48
595200	46	46	46
595300	44	43	47
595400	50	49	51
595500	50	50	51
595600	51	51	51
595700	51	51	51
595800	51	51	51
595900	26	16	51
596000	16	16	17
596101	20	16	37
596102	16	16	16
596200	16	16	16
596400	16	16	16
596502			
596503			
596504			

Table A2-12. Average, minimum and maximum 24 hr PM_{10} values for typical winter's day for Christchurch census area units, in micrograms per cubic meter derived from TAPM modelling. Shaded CAUs contain monitoring stations.

1999 Modelling results

Figure A2-66 shows the comparison between modelled and measured PM_{10} concentrations (hourly averages) for the St. Albans site. The model is able to satisfactorily simulate the dispersion of particulate matter at this site. Particularly it is encouraging to see the good agreement between the modelled and measured annual means and 90th percentile rankings. The mean difference between the modelled values and those measured at St. Albans over 10 minute averaging periods was 3.8 μg m^{-3}. As a result it is assumed that the base value of PM_{10} that cannot be accounted for by emissions from domestic, vehicle

or industrial sources is 3.8 μg m^{-3}. This base value can be attributed to 'natural' sources, such as dust and sea spray.

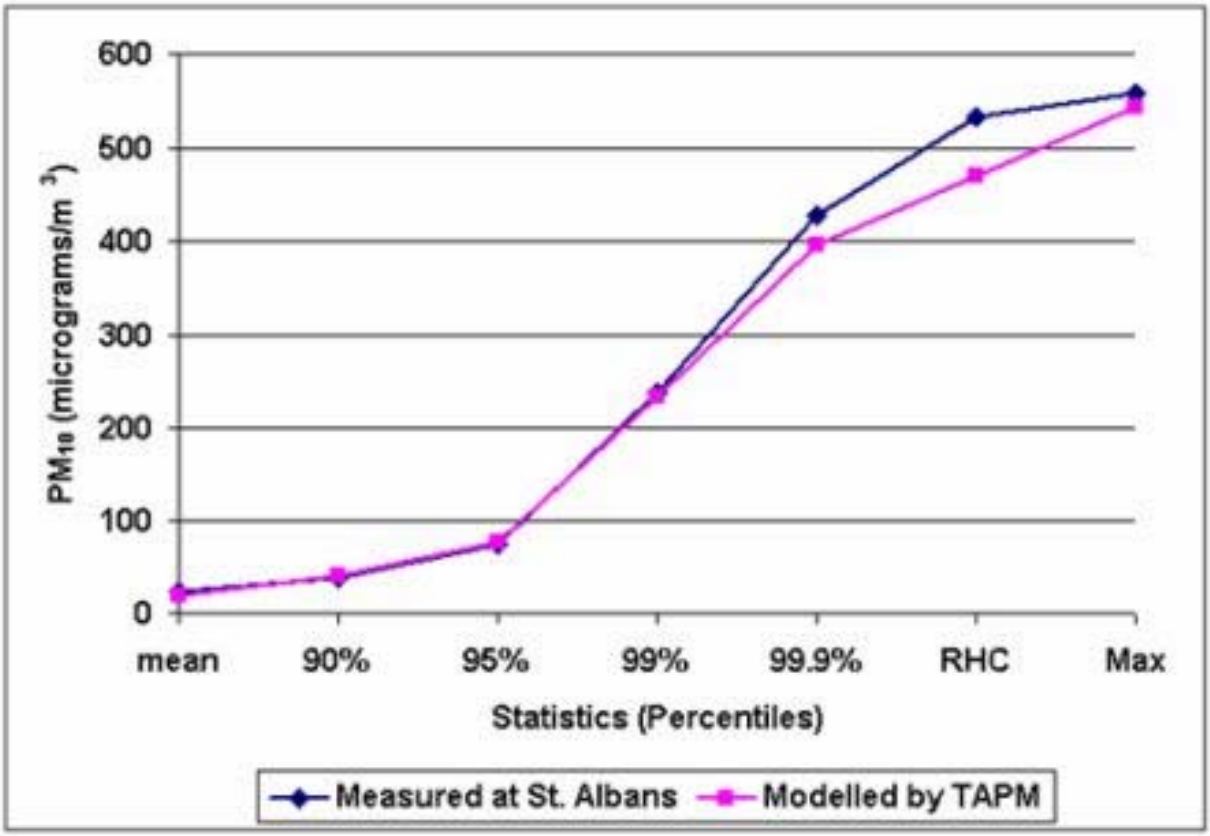


Figure A2-66. Comparison between modelled and observed hourly average PM_{10} for St Albans

Annual PM_{10} exposure

Table A2-13 shows modelled annual PM_{10} exposure values for Christchurch census area units. The contribution of different emission sources is shown with the estimated value of 3.8 μg m^{-3} as the base value.

CAU	BASE	DOM	IND	VEH	TOTAL	CAU	BASE	DOM	IND	VEH	TOTAL	CAU	BASE	DOM	IND	VEH	TOTAL
586001	3.8	0.0	0.0	0.0	3.8	589602	3.8	5.5	1.5	1.0	11.9	593200	3.8	21.5	2.0	2.9	30.2
586002	3.8	0.0	0.0	0.0	3.8	589800	3.8	1.0	0.0	0.0	4.9	589700	3.8	7.7	2.1	1.0	14.6
586401	3.8	0.0	0.0	0.0	3.8	589900	3.8	2.5	0.1	0.1	6.5	593300	3.8	21.0	1.0	2.0	27.8
586402	3.8	0.0	0.0	0.0	3.8	590000	3.8	6.3	0.5	0.5	11.2	593400	3.8	20.2	1.1	1.9	27.0
586501	3.8	0.0	0.0	0.0	3.8	590100	3.8	6.5	0.3	0.3	11.0	593501	3.8	16.4	1.0	1.9	23.1
586502	3.8	0.0	0.0	0.0	3.8	590200	3.8	6.7	0.3	0.3	11.0	593502	3.8	11.4	1.0	1.2	17.4
587301	3.8	0.0	0.0	0.0	3.8	590300	3.8	6.1	0.0	0.2	10.1	593600	3.8	17.2	1.0	1.2	23.1
587302	3.8	0.0	0.0	0.0	3.8	590400	3.8	0.1	0.0	0.0	3.9	593700	3.8	14.7	1.0	1.0	20.5
587400	3.8	0.0	0.0	0.0	3.8	590500	3.8	6.2	0.7	0.4	11.1	593800	3.8	5.3	0.5	0.5	10.1
587500	3.8	0.0	0.0	0.0	3.8	590601	3.8	0.1	0.0	0.0	3.9	593900	3.8	6.9	0.5	0.6	11.9
587701	3.8	1.4	0.0	0.0	5.2	590602	3.8	1.2	0.0	0.0	5.0	594010	3.8	10.6	1.0	1.0	16.4
587702	3.8	0.6	0.0	0.0	4.4	590700	3.8	12.5	0.8	1.1	18.2	594020	3.8	4.8	0.4	0.4	9.3
587811	3.8	0.8	0.3	0.0	4.9	590800	3.8	15.3	0.8	1.5	21.3	594100	3.8	4.3	0.3	0.3	8.7
587812	3.8	0.5	0.0	0.0	4.3	590900	3.8	12.1	0.0	0.9	16.9	594200	3.8	7.8	1.0	1.0	13.6
587820	3.8	0.0	0.0	0.0	3.8	591101	3.8	0.4	0.0	0.0	4.2	594300	3.8	4.8	0.2	0.3	9.1
587830	3.8	0.0	0.0	0.0	3.8	591102	3.8	0.9	0.0	0.0	4.7	594400	3.8	2.9	0.0	0.0	6.7
587841	3.8	0.0	0.0	0.0	3.8	591200	3.8	1.7	0.0	0.0	5.5	594500	3.8	7.3	0.5	1.0	12.6
587842	3.8	0.0	0.0	0.0	3.8	591300	3.8	1.1	0.0	0.0	4.9	594600	3.8	6.4	0.1	0.8	11.0
587843	3.8	0.3	0.0	0.0	4.1	591500	3.8	13.4	0.6	1.4	19.3	594700	3.8	9.2	0.0	0.8	13.7
587844	3.8	0.2	0.0	0.0	4.0	591600	3.8	18.0	1.0	2.0	24.8	594800	3.8	4.8	0.0	0.0	8.6
587902	3.8	0.1	0.0	0.0	3.9	591700	3.8	21.0	1.3	2.3	28.4	594900	3.8	2.1	0.0	0.0	5.9
587903	3.8	0.0	0.0	0.0	3.8	591800	3.8	16.9	1.4	1.4	23.4	595000	3.8	6.5	0.0	0.4	10.8
588101	3.8	1.5	0.0	0.0	5.3	591900	3.8	14.0	3.1	2.9	23.8	595100	3.8	1.9	0.0	0.0	5.7
588102	3.8	10.0	0.8	1.0	15.6	592000	3.8	9.6	2.0	2.4	17.7	595200	3.8	1.4	0.0	0.0	5.2
588200	3.8	5.1	0.1	0.1	9.1	592100	3.8	13.3	4.0	3.1	24.2	595300	3.8	2.3	0.0	0.0	6.1
588300	3.8	12.0	1.0	1.2	17.9	592200	3.8	18.3	4.2	3.3	29.5	595400	3.8	3.6	0.0	0.0	7.4
588400	3.8	1.9	0.0	0.2	5.9	592300	3.8	14.1	5.0	4.5	27.5	595500	3.8	2.8	0.0	0.0	6.6
588500	3.8	4.7	0.3	0.4	9.1	592401	3.8	18.6	4.9	3.7	30.9	595600	3.8	1.9	0.0	0.0	5.7
588600	3.8	4.5	1.0	1.0	10.3	592402	3.8	20.9	3.3	3.7	31.8	595700	3.8	2.8	0.2	0.0	6.9
588700	3.8	6.7	1.0	1.0	12.6	592500	3.8	18.5	3.0	3.5	28.8	595800	3.8	3.3	0.0	0.0	7.1
588800	3.8	5.0	1.9	1.0	11.7	592600	3.8	25.4	2.4	2.7	34.3	595900	3.8	2.5	0.0	0.0	6.3
588900	3.8	8.4	4.3	1.5	17.9	592701	3.8	18.8	2.2	2.4	27.2	596000	3.8	1.9	0.0	0.0	5.7
589000	3.8	9.6	4.2	1.8	19.4	592702	3.8	12.3	2.7	1.7	20.5	596101	3.8	2.8	0.0	0.0	6.6
589100	3.8	10.5	5.3	1.8	21.3	592811	3.8	9.7	1.8	1.6	16.9	596102	3.8	1.0	0.0	0.0	4.8
589200	3.8	14.0	3.9	2.7	24.4	592812	3.8	15.3	1.6	1.7	22.5	596200	3.8	0.7	0.0	0.0	4.5
589300	3.8	18.4	1.5	2.2	25.9	592820	3.8	4.7	1.0	0.3	9.8	596400	3.8	0.1	0.0	0.0	3.9
589400	3.8	11.8	4.4	1.4	21.4	592900	3.8	7.4	1.0	1.0	13.2	596502	3.8	0.0	0.0	0.0	3.8
589500	3.8	15.3	1.1	1.6	21.8	593000	3.8	12.6	1.0	1.0	18.4	596503	3.8	0.0	0.0	0.0	3.8
589601	3.8	4.0	1.9	0.5	10.2	593100	3.8	5.1	0.9	0.7	10.6	596504	3.8	0.0	0.0	0.0	3.8

Table A2-13. TAPM derived annual PM_{10} exposure values for the major sources of emissions for Christchurch census area units. Highlighted CAUs contain values exceeding the 20 µg m^{-3} Ministry for the Environment guideline.

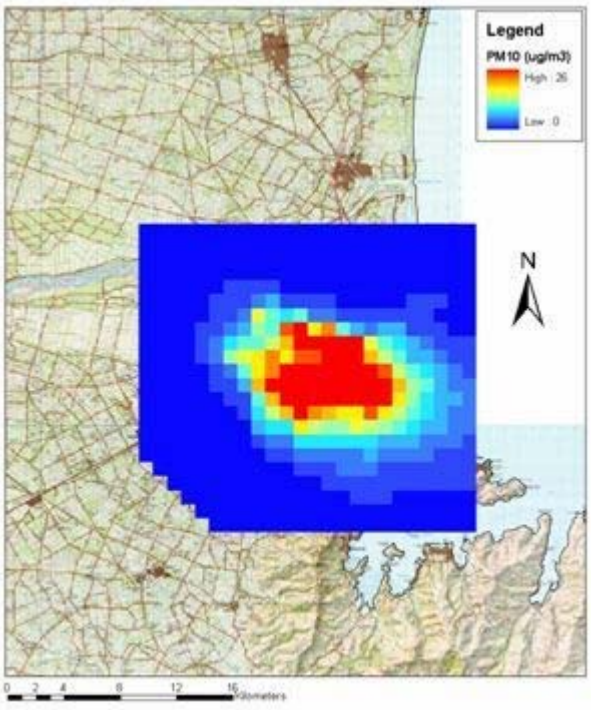


Figure A2-67. Annual domestic PM_{10} exposure grid from TAPM.

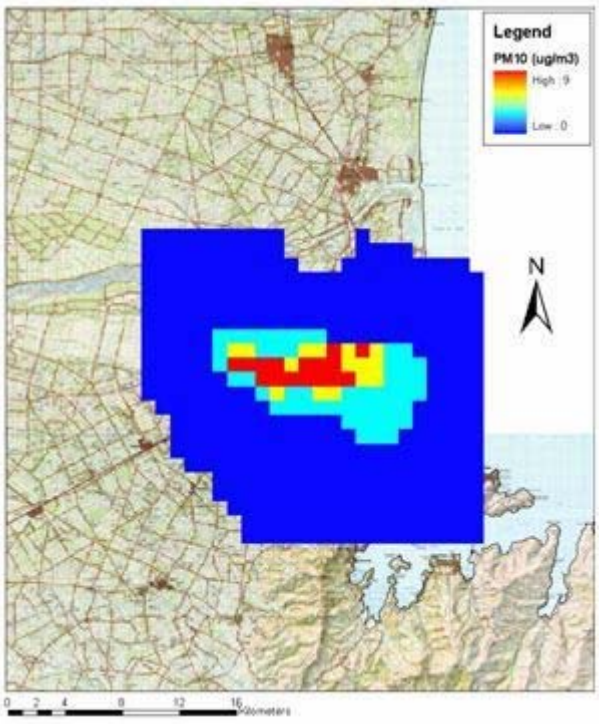


Figure A2-68. Annual industry PM_{10} exposure grid from TAPM.

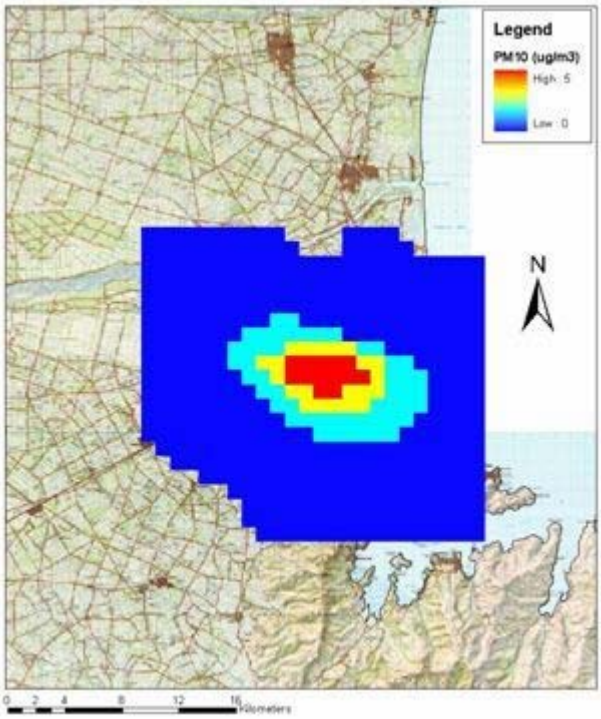


Figure A2-69. Annual vehicle PM_{10} exposure grid from TAPM.

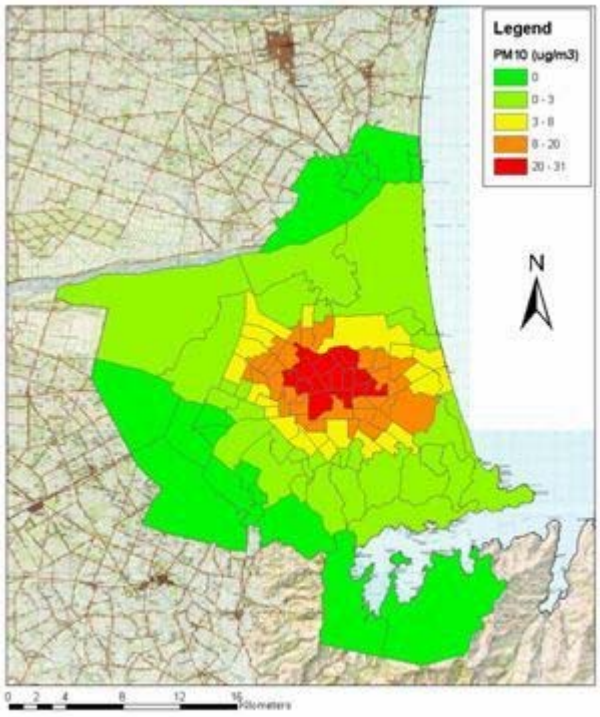


Figure A2-70. Annual PM_{10} census area unit exposure map. Dark CAUs represent areas with values above the 20 μgm^{-3} Ministry for the Environment guideline.

A2.2.4 Exposure variations

Exposure variations for elderly populations

The results show that there is no significant pattern in exposure to particulate pollution among the population aged 65 or over Figure A2-71. Mean pollution exposure is lowest in quintile three and highest in quintiles two and four.

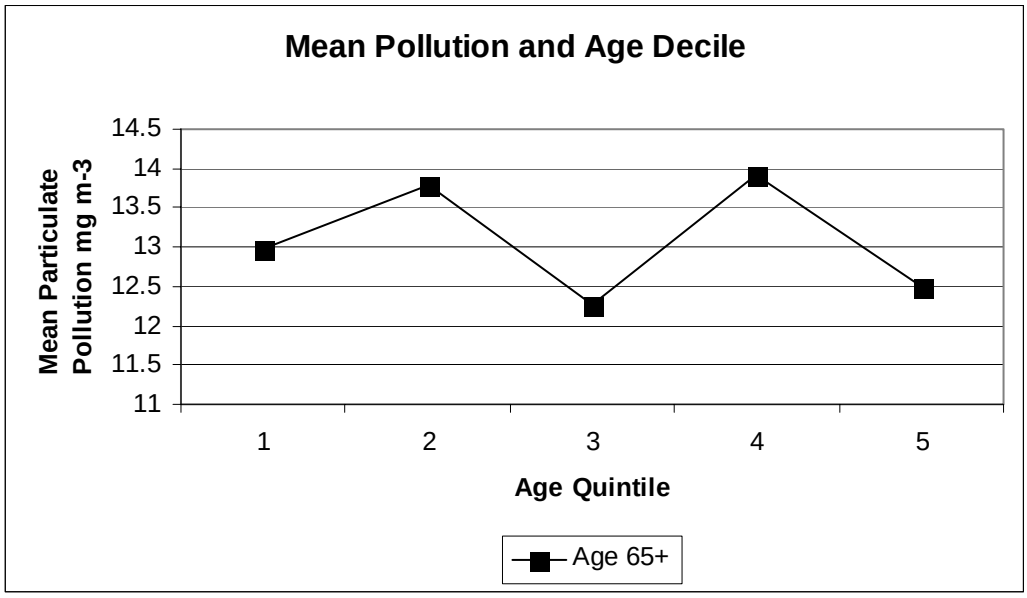


Figure A2-71. Mean annual particulate pollution by age quintile for the proportion aged over 65 in Christchurch. (Quintile is the population evenly divided into 5 categories).

Exposure variations by ethnicity

Figure A2-72 shows that the mean levels of pollution in the two deciles with the highest proportion of European were lower than in the three lowest European quintiles. The pattern among Maori was less clear, with significantly higher mean pollution levels in quintile four (quintile with the second highest proportion of Maori) yet the quintile with the highest proportion of Maori having mean pollution levels that were lower than areas with the lowest proportion of Maori (quintiles 1 and 2). The pattern for the Pacific Island and Asian populations was similarly unclear, although the mean pollution levels were significantly larger in the highest Pacific Island and Asian quintiles than the lowest quintiles.

Despite the non linearity of the results across the ethnicity quintiles, a comparison of quintiles 1 and 5 provides evidence to suggest that there is an ethnic disparity in the pollution estimates. For the European population the pollution levels are lower for quintile 5 (high proportion of European) compared to quintile 1 (low proportion of European). For the Maori, Pacific Island and Asian communities the pattern is reversed with higher levels of pollution in the highest quintile (high proportion of each ethnic group) compared to the lowest quintile (low proportion of each ethnic group). However, the results must be interpreted with some caution as the predominant ethnic group in Christchurch is European with relatively small numbers for the other three ethnic groups, especially Asian (Table A2-14).

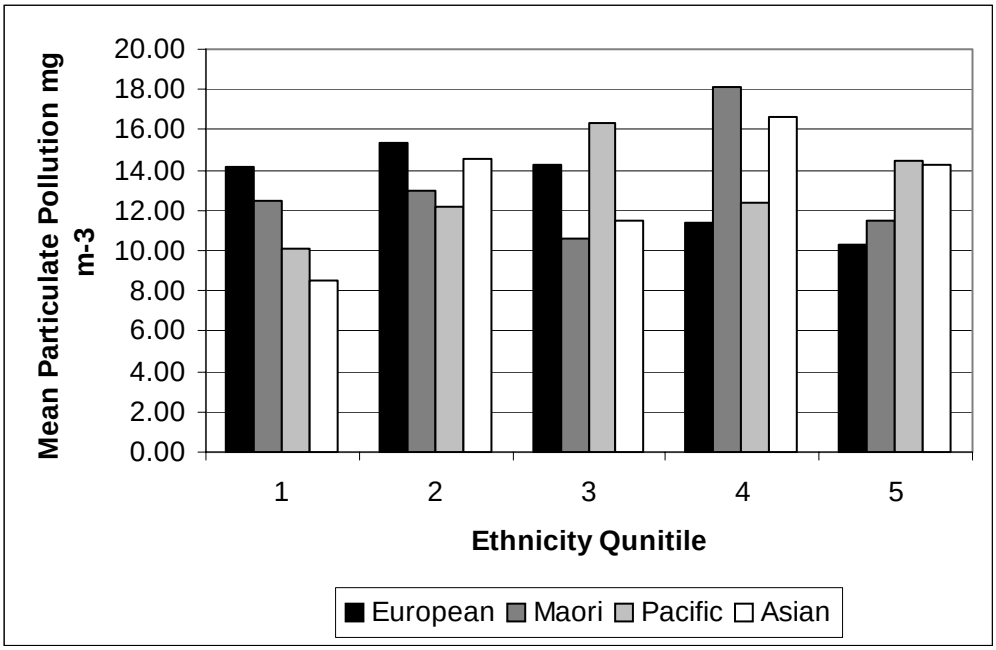


Figure A2-72. Mean annual particulate pollution for specified ethnic groups in Christchurch.

Ethnic Group	Population	Percent of population
European	275,100	82.6
Maori	21,960	6.6
Pacific	7,605	2.3
Asian	17,547	5.3
Other	10,929	3.3
Total	333,141	100

Table A2-14. Proportion of Christchurch residents in each ethnic group.

Exposure variation by social deprivation

The results show that the mean pollution levels are similar in deprivation quintiles one and two (low deprivation) but are significantly higher in deprivation quintiles three, four and five (Figure A2-73). Mean pollution levels vary by deprivation quintile with higher mean pollution levels in areas of high deprivation and lower pollution levels in areas of low deprivation. This suggests that for some of the poorest people in Christchurch there is evidence of environmental inequality, with higher pollution exposures in deprived CAUs than in more affluent CAUs (Pearce & Kingham, 2005). In the case of Christchurch, this is because the older central and southern city suburbs tend to have older housing stack that accommodates those on lower incomes and also tend to relay to a greater extent on solid fuel heating.

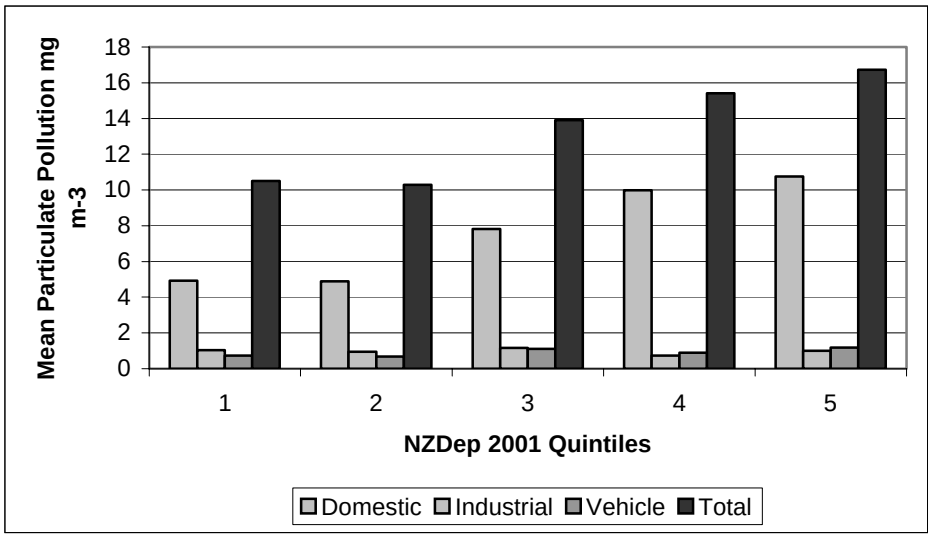


Figure A2-73. Mean annual particulate pollution for deprivation quintiles (NZDep 2001) in Christchurch.

A2.2.5 Benzene exposure

Air pollution research on the health effects of benzene and other air "toxics" has not been as extensive as for the major (criteria) pollutants. The extensive monitoring networks for criteria pollutants have provided relatively comprehensive temporal and spatial data on ambient concentrations that has allowed studies of their health impacts. In contrast measurement of air toxics have been taken inconsistently and in only a few locations, making it difficult to assess the extent of cumulative ambient exposures across all air toxics that could affect human health. Benzene is perhaps the most widely studied (Ministry of Health, 1999). The major concern of benzene is the carcinogenic effects associated with long-term exposure, best represented by annual average concentrations. Benzene is considered a Group A (known human carcinogen) human carcinogen by the U.S. EPA, and there is no threshold for effects (i.e. no safe level can be recommended) (WHO 2000).

Sources of benzene in ambient air include cigarette smoke, combustion and evaporation of benzene-containing petrol, petrochemical industries and other combustion processes. In New Zealand motor vehicles and household fires are the most significant sources of benzene. There are also some industrial activities that use and discharge benzene. Motor vehicle exhaust emissions of benzene derive partly from unburnt benzene in the fuel, and partly from the dealkylation of other aromatic hydrocarbons. Other sources of benzene that may impact locally include oil refining, petrochemical production, and synthetic rubber manufacture (Ministry for the Environment & Ministry of Health 2002). These sources contribute to elevated levels of benzene in the ambient air, which may subsequently be breathed by the public. Inhalation is the dominant pathway for benzene exposure in humans. In the body, benzene is converted to carcinogenic metabolites, which is what causes the health problems. Smoking is a large source of personal exposure, while high short-term exposures can occur during refuelling of motor vehicles.

Differences or similarities between benzene concentrations in ambient air found within a city may be the result of a number of factors including geographical and meteorological situations, traffic densities, emission control technology, proportion of petrol-fuelled vehicles, petrol formulation, traffic control and flow patters, and the extent of non-vehicle benzene emissions.

There has been limited benzene monitoring in Christchurch. Figure A2-74 shows quarterly benzene concentrations measured at selected monitoring locations in Christchurch from October 1996 – May 1999. The annual average guideline value for benzene is 10 µg m^{-3}, which will be reduced to 3.6 µg m^{-3} to be achieved by 2010 (Ministry for the Environment & Ministry of Health 2002). Winter benzene concentrations are at least two or three times higher than summer concentrations. Benzene and other

toxic levels have been measured previously in New Zealand cities (including Christchurch) in a number of studies commissioned by the MoH or MfE (Stevenson & Narsey 1999a; Stevenson & Narsey 1999b).

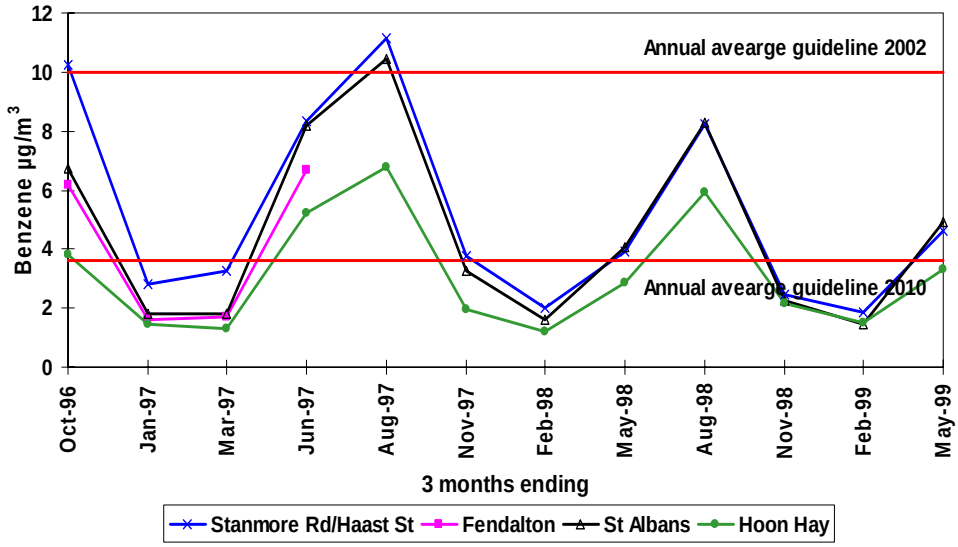


Figure A2-74. Quarterly benzene concentrations at selected monitoring location in Christchurch from October 1996 – May 1999.

Table A2-15 shows annual average concentrations for the monitoring locations. The minimum and maximum concentrations were 3 µg m^{-3} (Hoon Hay) and 6.6 µg m^{-3} (Stanmore Rd/Haast Street), respectively. These sites experienced annual concentrations that are above the proposed UK guideline value of 3.3 µg m^{-3} that is due to come into effect in 2010. There is currently no standard for benzene in New Zealand.

Monitoring sites	Annual average benzene µg m^{-3}	
	1997	1998
Stanmore Rd/Haast St	6.6	4.2
Fendalton	n/a	n/a
St Albans	5.9	4.1
Hoon Hay	3.8	3.0

Table A2-15. Annual average benzene in Christchurch.

A2.2.6 CO exposure

Figure A2-75 shows observed 8-hour average CO concentrations at the Packe Street monitoring site during 1999. The ambient air quality guideline was exceeded 12 times during 1999 during winter months. Figure A2-76 shows the daily one-hour maximum CO levels at the same site during 1999. There were no guideline exceedences.

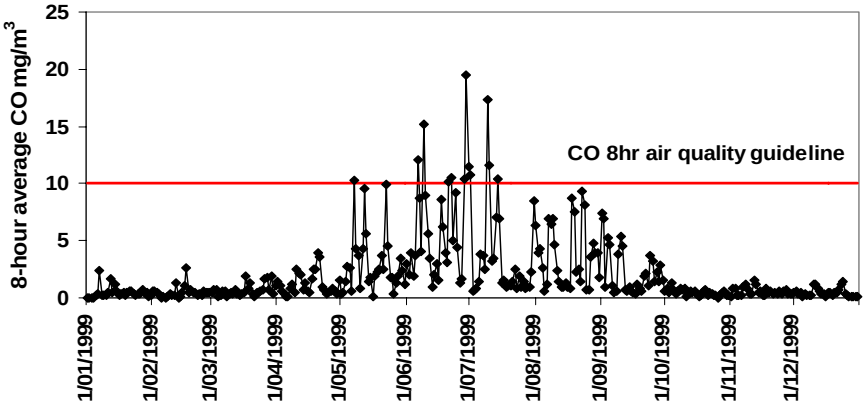


Figure A2-75. CO 8-hour average during 1999 measured at Packe Street St Albans.

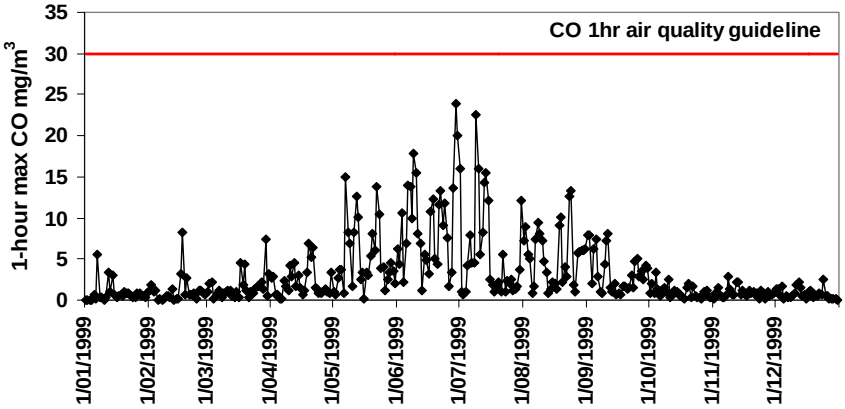


Figure A2-76. CO 1-hour maximum during 1999 measured at Packe Street St Albans.

A2.2.7 NO_2 exposure

Figure A2-77 shows 24-hour average NO_2 concentrations measured at Packe Street, St Albans during 1999. Levels remain relatively low during the year and there were no guideline exceedences.

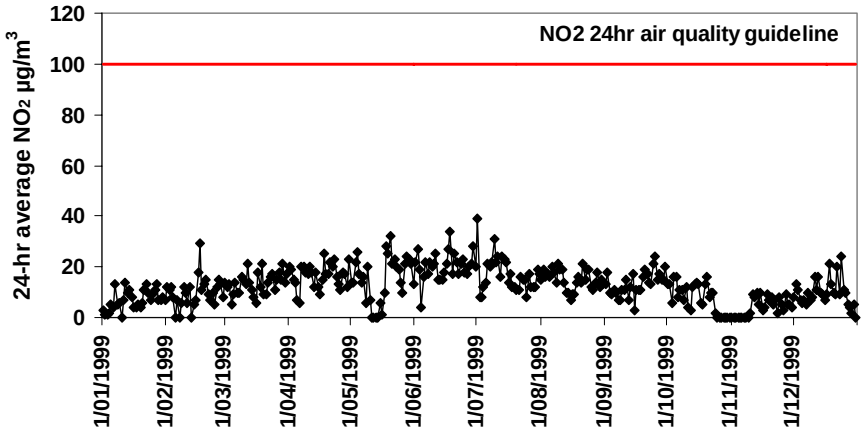


Figure A2-77. NO_2 24-hour average during 1999 measured at Packe Street St Albans.

A2.2.8 SO_2 exposure

Figures A2-78 and A2-79 show 24-hour average concentrations measured at Packe Street, St Albans and Hornby monitoring sites, respectively. Neither site had any guideline exceedences; however levels are higher at Hornby due to nearby industrial sources.

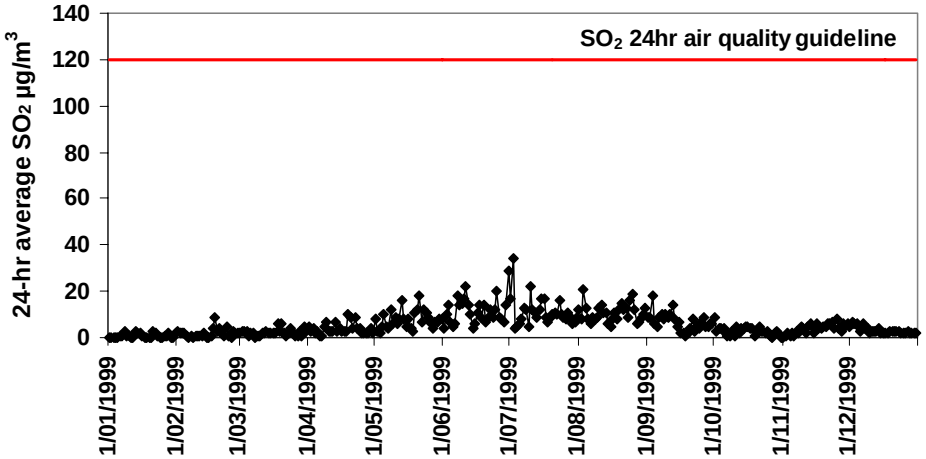


Figure A2-78. SO_2 24-hour average during 1999 measured at Packe Street St Albans.

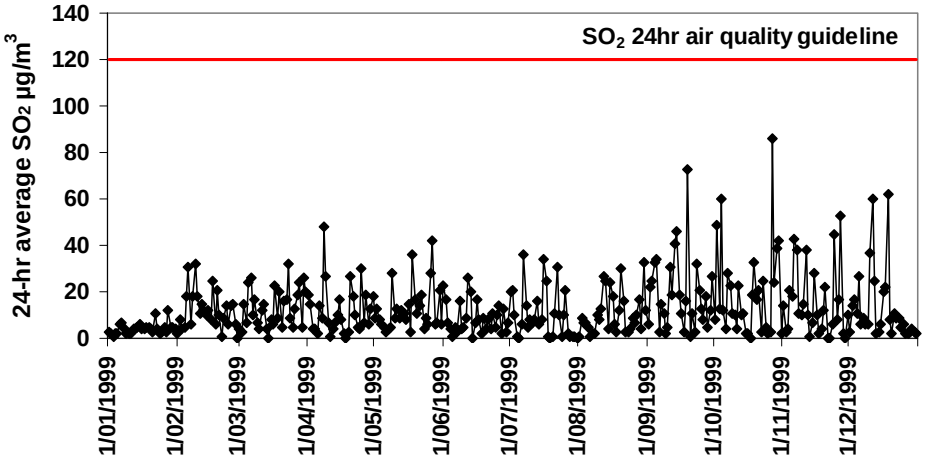


Figure A2-79. SO_2 24-hour average during 1999 measured at Hornby.

A2.3 Discussion

A2.3.1 Emissions

The following was taken from the 2002 emissions inventory (Scott & Gunatilaka 2003).

The major sources of PM_{10} in Metropolitan Christchurch in the winter were domestic home heating (82%), industrial and commercial activities (9%), and motor vehicles (9%). Domestic home heating discharged 11 t of PM_{10} on a winter's day with peak emissions occurring during the evening period (4pm-10pm). Wood burners were the major source of domestic home heating PM_{10} emissions with a contribution of 7 t (62%). Total PM_{10} emissions in Metropolitan Christchurch decreased 13% from 1999 with the greatest reductions occurring in motor vehicles (-15%) and domestic home heating (-15%).

The major sources of $PM_{2.5}$ in Metropolitan Christchurch in the winter were domestic home heating (85%), industrial and commercial activities (6%), and motor vehicles (9%). Domestic home heating discharged 10.4 t of $PM_{2.5}$ on a winter's day with peak emissions occurring during the evening period (4pm-10pm). Wood burners were the major source of domestic home heating $PM_{2.5}$ emissions with a contribution of 6.5 t (62%). While $PM_{2.5}$ was not quantified in previous years it is likely that reductions similar to PM_{10} emissions would have occurred.

The major sources of CO in Metropolitan Christchurch in the winter were motor vehicles (51%), domestic home heating (48%), and industrial and commercial activities (1%). Motor vehicles discharged 101 t of CO on a winter's day followed by 96 t from domestic home heating. Emissions from motor vehicles were evenly distributed throughout the day whereas domestic home heating emissions peaked during the evening period. Wood burners were the major source of domestic home heating CO emissions with a contribution of 68 t (70%). Total CO emissions in Metropolitan Christchurch decreased 10% since 1999 with the greatest reductions occurring in the motor vehicle (-8%) and domestic home heating sectors (-13%). Major sources of NO_x in Metropolitan Christchurch were motor vehicles (81%) and industrial and commercial activities (13%), followed by domestic home heating (6%). Motor vehicles discharged 18 t of NO_x on a winter's day with a 3 t contribution from industrial and commercial activities. Emissions were evenly distributed throughout the day. The most significant industrial source of emissions were coal-fired boilers that contributed 1.5 t of NO_x (52%). Emissions of total NO_x in Metropolitan Christchurch decreased by 1% since 1999. Decreases occurred in the motor vehicle (-3%) and domestic home heating sectors (-16%) with potential increases occurring in the industrial and commercial sector.

The major sources of SO_x in Metropolitan Christchurch were industrial and commercial activities (80%), motor vehicles (15%), and domestic home heating (4%). Industrial and commercial sources discharged 8.5 t of SO_x on a winter's day, followed by 1.6 t from motor vehicles and 0.5 t from domestic home heating. Emissions were greatest during the morning and daytime periods (6am–4pm). The major industrial source of emissions was coal-fired boilers that contributed 5.4 t of SO_x (63%). Emissions of total SO_x in Metropolitan Christchurch increased by 1% between 1999 and 2002. Increases occurred in the motor vehicle (+5%) and industrial sectors (+2%). Domestic home heating SO_x conversely decreased by 22%.

Emissions of all contaminants discharged by the domestic home heating sector decreased from 1999 in response to declining numbers of open fires and older wood burners, and increased electricity and gas use. Emissions of PM_{10}, CO, NO_x from motor vehicles also decreased as greater numbers of overseas motor vehicles (installed with superior emission control equipment) infiltrated the motor vehicle fleet. Motor vehicle CO_2 and SO_x emissions, on the other hand, increased. Increased fuel consumption (related to higher VKT) and greater numbers of diesel vehicles are likely causes. While trends were difficult to establish for industrial and commercial activities, it is possible that emissions of all contaminants have increased however, the contribution of emissions from the industrial sector is low compared to the other major sources.

Uncertainty

The quality of the estimated emission values provided in the 2002 inventory depends on the level of uncertainty associated with the data. Data users need to be aware of the limitations associated with the data; changes in the method of data collection, as inventory capability is developed, should have the effect of reducing uncertainty.

For the 2002 emission inventory a qualitative rating system approach was adopted to determine uncertainty. The areas of uncertainty were identified and scored in accordance with the system outlined in the inventory.

The rating system was based on the premise that data reliability is determined by the quality of the activity data and the emission factors used. In general, data have a greater level of certainty if the activity data are locally derived and recent, and if the emission factors are also locally derived and comprehensive. Considerations such as temporal and spatial representation are also important as both spatial and temporal data are presented in the inventory. The lower the score the greater the level of confidence or certainty for the estimate. Based on the scoring system, the domestic home heating, motor vehicle and industrial sectors had a mid-low level of uncertainty. Overall, it was suggested that the

estimates were of above average quality with a reasonably high level of confidence and reliability. For more detailed information regarding the uncertainties associated with the emissions data refer to the 2002 emissions inventory Scott and Gunatilaka (2003).

CAU emissions derivation

With regard to the use of the emissions data in airshed modelling it is important to understand that the values are estimates and not the true volume of air pollutant emissions. The levels of uncertainty and error in the data must also be considered. Hourly emissions data for 1 km^2 grid cells across Canterbury would be ideal for input into TAPM (the air pollution model). However, the 2002 inventory data is for four time periods across three large areas of Christchurch. To achieve a better spatial resolution of the emissions data the data taken from the 2002 inventory were manipulated and broken down to census area units. The methods employed can add additional errors increasing the level of uncertainty.

A2.3.2 Monitoring

Environment Canterbury

Over a year most contaminants at Canterbury sites show a seasonal variation with maximum concentrations measured in the winter, and minimum concentrations in the summer. During winter meteorological conditions are such that dispersion of contaminants is frequently poor compared to other seasons. Emissions from the domestic home heating sector also increase during this period. The combined effect of the additional emissions and poor dispersion is an increase in contaminant concentrations during the winter (see Figures A2.39 and A2.40). The frequency and extent of high pollution episodes during the winter are largely dependent on weather patterns during these months. Elevated pollution concentrations are generally associated with still and frosty nights as the temperature inversion, which occurs under these conditions, restricts the dispersion of contaminants. In addition to the seasonal variations daily patterns of pollution have also been documented. It has been seen that contaminant levels become highest in the evenings with an additional smaller peak in pollution concentrations during the morning. This pattern corresponds to domestic heating and vehicle emissions patterns during the day (see Figure A2.41).

A2.3.3 Airshed modelling

Evidence so far from the modelling study with TAPM shows that the model is more suitable for deriving long-term (annual) statistics for the particulate matter concentrations. The success of comparison between modelled and measured data at St. Albans for 1999 places great confidence in modelled values where monitoring data is not available. However, more work has to be performed on simulating the July 2003 case study month.

A2.3.4 Exposure variations

There are variations in personal exposure. It has been shown that levels of pollution are highest at certain times of day (morning and evening), at certain times of year (winter), and in certain parts of the city. It has also been shown there may be some difference in population exposure by ethnicity and socio-economic status (deprivation). In addition, there is evidence from other research that pollution exposure will vary according to variety of individual behaviours, including such things such as smoking, time spent indoor, migration and personal mobility (Boudet et al. 2001; Cotterill & Kingham 1997; Kingham et al. 1998; Koistinen et al. 2001; Kousa et al. 2001). Quantifying these and producing exposure assessments that account for this is beyond the scope of this study, but it is important to be aware of this.

Appendix 3 – Effects due to specific pollutants

The health effects analysis carried out in chapter 10 were based on PM_{10} (and benzene), as the most reliable total indicator of the health effects of air pollution. However there are defined health effects associated with a number of other pollutants, and these are calculated here.

The effects due to CO, NO_2, and SO_2 in Christchurch are assessed. These cannot be 'added' to the effects associated with PM_{10}, as the overall effect associated with PM_{10} almost certainly contains some of the effects associated with these other pollutants. However the proportion cannot be determined at this time, due to a lack of epidemiological knowledge.

A3.1 Health impacts of CO, NO_2, SO_2

Health impact assessment was carried out for carbon monoxide, nitrogen dioxide, and sulphur dioxide. Dose-response functions from local and overseas studies were used to estimate the annual number of deaths and hospital admissions attributable to each pollutant. There is uncertainty over the degree to which a single pollutant serves as a surrogate measure for the complex mix of particles and gases that results from fuel combustion from vehicles, industry and domestic sources. To preclude double counting of adverse health effects related to air pollution usually only one pollutant is chosen to quantify health outcomes (Lipfert 1997; Lipfert & Wyzga 1995). Particulate matter is usually considered the single pollutant, hence the health effects due to CO, NO_2, and SO_2 are not necessarily independent. There is considerable evidence, from both this study and numerous others, that particulate effects dominate the total health effects, accounting for up to 85% of the total health costs.

A3.1.1 Short-term (daily) mortality/morbidity

The daily exposure-response functions were applied to daily hospital admissions and daily mortality and pollutant levels for 1999. The annual number of hospital admissions or deaths attributed to the pollutants (N) for 1999 was calculated as follows:

$$N_s = (DR/100) \times H \times \text{Sum}\,[(Ei-B)]$$

Where:

N_s = Annual number of respiratory hospital admissions attributed to PM_{10}

DR is the percentage increase in daily hospital admissions or daily mortality per 1 μg m^{-3} increase in pollutant,

H is the baseline average number of hospital admissions or deaths per day (in 1998),

Ei is the pollutant level (Christchurch average for different days in 1998),

B is the threshold pollutant level for its effects on hospital admission or mortality,

'Sum' is the summation of each of the 365 days into an annual number.

A3.1.2 Long-term (annual) mortality

The annual average number of deaths attributable to annual average nitrogen dioxide (the only non-PM pollutant for which an annual dose-response coefficient was available) (N) was calculated as follows:

$$N_l = (DR/100) \times (A - B) \times (C)$$

Where:

N_l = Annual number of respiratory hospital admissions attributed to PM_{10}

DR is the percentage increase in annual mortality per 1 μg m^{-3} increase in annual average NO_2,

A is the annual average NO_2 concentration.

B is the annual average NO_2 threshold exposure level for mortality effect,

C is the baseline annual average number of deaths.

A3.1.3 Air quality data and measurement

Basic air quality data

Environment Canterbury air quality monitoring data for a relevant time period was used to carry out health risk assessment, using data from the site at Packe St., St Albans see Table A3-1). No spatial modelling data for the non-PM_{10} pollutants were available at this stage.

Pollutant	Time period	Monitoring site
CO 1 hr average maximum	1999	St Albans (Packe St)
NO_2 annual	Average 1998-2001	St Albans (Packe St)
NO_2 24 hr average	1999	St Albans (Packe St)
SO_2 24 hr average	1999	St Albans (Packe St)

Table A3-1. Air quality monitoring data used for health impact assessment.

Source apportionment

Some information on the proportion of contribution from the three main source groups can be gained from the Environment Canterbury emissions inventory (Table A3-2).

Pollutant	Domestic home heating	Motor vehicles	Industry
CO	48%	51%	1%
SO_2	5%	15%	80%
NO_2	6%	81%	13%

Table A3-2. Source apportionment of CO, SO_2, and NO_2 in Christchurch.

Health data

Mortality data for the period 1990 to 1998, and acute hospital admissions data for the period 1994-98 was obtained from New Zealand Health Information Services (NZHIS), MoH.

Dose-response relationships

Relationships for mortality and morbidity used are summarised in Tables A3-3 and A3-4.

A3.1.4 Threshold concept

The effects of some pollutants may not be significant until some threshold has been released. This is not the case with PM_{10}, which has been determined to have a 'zero' threshold – that is effects can be seen down to concentrations as low as 1 μg m^{-3}.

This is also true for CO and SO_2, but has not been confirmed for NO_2 (see discussion in chapter 10). Thus the calculations made below show the effects for various threshold values of NO_2.

Pollutant	Health outcome	Percent increase in *daily* mortality associated with 1 μg m^{-3} increase in pollutant	Reference
CO 1 hr average maximum	Non-external cause mortality	0.00058 (95%CI 0.0001 – 0.0011)	(Denison et al. 2000b)
SO_2 24 hr average	Non-external cause mortality	0.06 (0.03 – 0.15)	(Norwegian Institute for Air Research & WHO 1996)
		Percent increase in *annual* mortality associated with 1 μg m^{-3} increase in pollutant	
NO_2 Annual average	Non-external cause mortality	0.013 (0.011–0.015)	(Scoggins et al. 2004)
NO_2 Annual average	Circulatory & respiratory mortality	0.018 (0.015–0.021)	(Scoggins et al. 2004)

Table A3-3. Dose-response relationships: mortality.

Pollutant	Health outcome	Percent increase in *daily* health outcome associated with 1 μg m^{-3} increase in pollutant	Reference
CO 1 hr average maximum	Cardiovascular admissions (0-64 yrs)	0.001 (0.00018 – 0.00182)	(Denison et al. 2001)
SO_2 24 hr average	Circulatory & respiratory admissions	Not available	
NO_2 24 hr average	Respiratory admissions (65+ yrs)	0.30 (0.02 – 0.58)	(Codde et al. 2003)
NO_2 24 hr average	Cardiovascular admissions	0.15 (0.01 – 0.29)	(Codde et al. 2003)

Table A3-4. Dose-response relationships: morbidity (acute hospital admissions only).

A3.2 Results

The health effects are calculated using the exposure data for Christchurch according to the methodologies described above.

A3.2.1 CO effects

Other pollutants were not modelled explicitly in the same manner as PM_{10}. A separate source apportionment methodology using emissions inventory information has been conducted.

Tables A3-5 and A3-6 below show the number of deaths and hospital admissions associated with various sources for CO. There were relatively fewer deaths associated with CO in Christchurch, which is not surprising given the pollution levels are relatively low.

Health Effects	Threshold (mg m^{-3})	Health effects by source (95% confidence intervals)			
		Vehicles	Domestic	Industry	Total
Non-external cause mortality	0	12 (2 - 24)	12 (2 - 25)	0.6 (0.1 - 0.9)	25 (4 - 44)

Table A3-5. Number of deaths by source associated with CO exposure.

Health Effects	Threshold (mg m^{-3})	Health effects by source (95% confidence intervals)			
		Vehicles	Domestic	Industry	Total
Cardiovascular admissions (0-64 yrs)	0	60 (11 - 109)	56 (10 - 103)	1 (0.2 – 2)	117 (21 - 214)

Table A3-6. Number of hospital admissions by source associated with CO exposure.

A3.2.2 SO_2 effects

Table A3-7 below shows the number of deaths associated with various sources for SO_2. There were very few deaths associated with SO_2 in Christchurch.

Health Effects	Threshold (µg m^{-3})	Health effects by source (95% confidence intervals)			
		Vehicles	Domestic	Industry	Total
Non-external causes mortality	0	1 (0.8 - 4)	0.4 (0.2 - 0.9)	7 (4 - 18)	8 (5 - 23)

Table A3-7. Number of deaths by source associated with SO_2 exposure.

The equivalent calculation for morbidity effects had not been made as a suitable dose-response function could not be found (although as of 2005 these are now available).

A3.2.3 NO_2 effects

Tables A3-8 and A3-9 below show the number of deaths and hospital admissions associated with various sources for NO_2. These are shown for a number of different threshold values. There is some evidence that a threshold of 13 μg m^{-3} may be appropriate for mortality (Scoggins et al 2004).

Health Effects	Threshold[1] (μg m^{-3})	Health effects by source (95% confidence intervals)			
		Vehicles	Domestic	Industry	Total
Non-external causes mortality	0	373 (316 - 431)	28 (23 – 32)	60 (51 – 69)	461 (390-532)
	7.5	176 (149 – 203)	13 (11 – 15)	28 (24 – 33)	217 (184-251)
	13	32 (27 – 36)	2 (2 - 3)	5 (4 – 6)	39 (33-45)
	15	-	-	-	Nil
Circulatory & respiratory mortality	0	302 (252 – 352)	22 (19 – 26)	48 (40 - 57)	372 (311 - 435)
	7.5	143 (119 – 166)	11 (9 – 2)	23 (19 - 27)	177 (147 – 205)
	13	26 (21 - 30)	2 (1.6 - 2.2)	4 (3 - 5)	32 (26 - 37)
	15	-	-	-	Nil

[1] Threshold levels based on Scoggins et al (2004)

Table A3-8. Number of deaths by source associated with NO_2 exposure.

The number of deaths associated with NO_2 vehicle sources (32 deaths) (assuming a threshold of effects of 13 μg m^{-3}), was relatively similar to the estimated number associated with vehicle related PM_{10} in the earlier study (41 deaths) (Fisher et al 2002). The health effects associated with NO_2 should not be treated as in addition to the estimates provided for PM_{10} mortality effects.

Health Effects	Threshold[1] (μg m^{-3})	Health effects by source (95% confidence intervals)			
		Vehicles	Domestic	Industry	Total
Respiratory admissions (65+ yrs)	0	41 (2 - 79)	3 (0.2 - 6)	7 (0.4 – 13)	51 (3 - 98)
	10	15 (0.8 - 28)	1 (0.1 - 2)	2 (0.1 – 5)	18 (1 - 35)
	33	-	-	-	Nil
Cardiovascular admissions	0	66 (4 - 26)	5 (0.3 - 9)	11 (0.7 – 20)	82 (5 - 156)
	10	23 (2 - 43)	2 (0.1 - 3)	4 (0.3 - 7)	29 (2 - 53)
	33	-	-	-	Nil

[1] Threshold levels based on EPI programme (0%, 10%, and 33% of the NO_2 24hr guideline 100 μg m^{-3}

Table A3-9. Number of hospital admissions by source associated with NO_2 exposure.

The morbidity effects in Table A3-9 show if NO_2 is assumed to have an effect down to zero concentration, the respiratory and cardiovascular admission rates can be significant. This table also shows the much greater portion of effect associated with vehicle emissions (as noted in Table A3-2).