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AN INVESTIGATION
INTO MANAGEMENT OPTIONS
TOWARDS REDUCING SUSPENDED
PARTICULATE CONCENTRATIONS IN THE
CHRISTCHURCH AIR SHED

A thesis
Submitted in partial fulfilment
Of the requirements for the Degree of
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at
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by
E. V. Wilton

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Applied Science.

AN INVESTIGATION INTO MANAGEMENT OPTIONS TOWARDS REDUCING SUSPENDED PARTICULATE CONCENTRATIONS IN THE CHRISTCHURCH AIR SHED

By
E. V. Wilton

Wintertime suspended particulate concentrations in Christchurch, a city of 310,000, regularly exceed the Canterbury Regional Council’s air quality guideline of 50 µgm⁻³. Maximum 24-hour average concentrations of up to 280 µgm⁻³ are higher than those measured in much more highly populated cities in the United States and United Kingdom. Epidemiological research suggests that the high concentrations currently measured in Christchurch may be associated with significant adverse health effects. The Resource Management Act (1991) requires the Canterbury Regional Council to avoid, remedy or mitigate any adverse effects, or potential effects, of the discharge of suspended particulate.

A model was developed to test the effectiveness of different potential management options in reducing current suspended particulate concentrations to acceptable levels. Two management options targeting emissions from domestic heating are identified as potentially capable of achieving the required reduction. Of these an option prohibiting the use of coal for domestic heating in 1998, prohibiting the use of open fires in 2001 and requiring the replacement of solid fuel burners 15 years after installation proves to be the most effective and viable choice. The Canterbury Regional Council also selected these measures as the preferred management option.

The modelling, despite inherent errors associated with some input parameters, suggests that a guideline concentration of 50 µgm⁻³ can be reached by about the year 2015 if the proposed management plan is introduced.
Acknowledgements

A special thanks to Bob Ayrey for his technical expertise and support, and to Jeff Bluett for all the encouragement and invaluable review comments.
Glossary of Abbreviations

\( \mu \text{gm}^3 \)  
Micrograms per cubic metre

AAQG  
Ambient air quality guidelines

ANSTO  
Australia Nuclear Science and Technology Organisation

CI  
Confidence Interval

CO  
Carbon monoxide

CRC  
Canterbury Regional Council

ESR  
Environmental Science and Research

g/kg  
Grams of particles per kilogram of fuel burnt

MfE  
Ministry for the Environment

MOT  
Ministry of Transport

NIWA  
National Institute of Water and Atmospheric Research

NO  
Nitrogen Oxide

NO\(_2\)  
Nitrogen dioxide

NOAEL  
No observable adverse effects level

NO\(_x\)  
Oxides of nitrogen

PM\(_{10}\)  
Particles in the air less than 10 microns in diameter

PM\(_{2.5}\)  
Particles in the air less than 2.5 microns in diameter

QUARG  
United Kingdom Quality of the Urban Air Review Group

R\(^2\)  
Coefficient of determination

RAD  
Restricted Activity Day

RMA  
Resource Management Act

RPS  
Regional Policy Statement

SO\(_2\)  
Sulphur dioxide

SO\(_x\)  
Oxides of sulphur

TEOM  
Tapered Elemental Oscillating Microbalance

TSP  
Total suspended particulate

UK  
United Kingdom

US  
United States

USEPA  
United States Environmental Protection Agency

WHO  
World Health Organisation
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CHAPTER ONE: INTRODUCTION

Christchurch, a city of 310,000, is located in the South Island of New Zealand (figure 1). Christchurch's poor wintertime air quality is a well-known and long-standing issue. The pollution can be readily seen and smelt on cold, still, winter nights when temperature inversion conditions prevail across the city. Groups have been actively campaigning against the pollution since the 1930's when the Sunlight League campaigned for the installation of smokeless heating appliances in all new houses. Since the 1960's the Clean Air Society has been active in calling for improvements in air quality, advocating subsidies for households to convert to electricity, improved installation, emission standards for motor vehicles and discouraging the use of open fires.

1.1 Legislative Background

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. Home heating appliances installed prior to these orders can still be used. However, restrictions on the use of wet wood (>25% moisture) and high sulphur coals (>1%) introduced under the Clean Air Zone Orders apply to all domestic home heating appliances.

The Clean Air Zone Orders have succeeded in restricting any worsening of Christchurch's air pollution and should result in a gradual decline in domestic home heating emissions with time. They have also helped to reduce concentrations of sulphur dioxide ($\text{SO}_2$) due to the restrictions on the sulphur content of coal. However, these orders do not adequately address the air quality problem.

The Christchurch City Council attempted to tackle the issue again in 1988 by introducing amendments to the Clean Air Zone Orders. These amendments sought to establish a ban on the use of existing open fires. The Council was unsuccessful when the issue was taken to a public hearing. Opponents described the ban as “draconian” because of its social and economic consequences, particularly for the older sections of the community (de Lisle, 1989).
Chapter one - Introduction

Figure 1.1 Christchurch Air Quality Monitoring Sites

The Resource Management Act (1991) gives regional councils the responsibility for air quality. The purpose of this Act as is stated in section 5 of the Act and, as it relates to air, is as follows:

1. To promote the sustainable management of air;
2. "Sustainable management" means managing the use, development, and protection of air in a way, or at a rate, which enables people and communities to provide for their social, economic and cultural well being and for their health and safety while:

a) sustaining the potential of air to meet the reasonably foreseeable needs of the future generations; and

b) safeguarding the life-supporting capacity of air; and

c) avoiding, remedying, or mitigating any adverse effects of the discharge of contaminants to the air on the environment.

The Resource Management Act (RMA) requires 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 (Ayrey, 1995). One of the methods for achieving the objectives and policies stated in the RPS is through regional plans. The Canterbury Regional Council (CRC) is currently preparing the air quality chapter of the natural resources regional plan to achieve "sustainable management" of air quality.

In managing Christchurch’s air quality the Council needs to consider a number of issues identified in the RPS. These include the health and nuisance effects of emissions in both ambient and localised situations. Examples of these issues in the Christchurch context include:

- high concentrations of contaminants, in particular suspended particulate, under temperature inversion conditions (ambient impact, potential health impacts and nuisance effects);
- odours from nearby industry (localised impact, nuisance effects);
- daytime haze (ambient impact, nuisance effects);
- smoke nuisance from neighbouring domestic chimney (localised, nuisance effects).

A number of issues can be addressed under present legislation e.g., industry is regulated through the resource consent process, and smoke nuisance can be addressed under section 17 of the RMA. Some issues, in particular those relating to ambient effects, will be addressed in the air chapter of the Natural Resources Regional Plan. Because of the frequency and extent of high suspended particulate concentrations, and because research indicates that health impacts may occur at concentrations considerably lower than previously thought (Vedal, 1995), priority has been given to addressing suspended particulate concentrations in Christchurch.
1.2 Air quality in Christchurch

The quality of the air can be assessed by the extent to which concentrations of key indicators\(^1\), in ambient air\(^2\), compare with air quality guidelines. In New Zealand air quality guidelines for health protection were prepared by the Ministry for the Environment (MfE) in 1994. Studies used as a basis for these guidelines are over 10 years old and considerable research in these areas has subsequently been conducted. Consequently the basis for current national guidelines is very dated, particularly with respect to suspended particulate (PM\(_{10}\)). In the absence of a national review of the PM\(_{10}\) guideline the Canterbury Regional Council, in 1996, adopted a guideline of 50 \(\mu\text{g}\text{m}^{-3}\) (24-hour average). The basis for this is discussed in chapter two.

Air quality in Christchurch has been monitored for many years. Christchurch's main air quality monitoring site is located in St Albans between Packe and Madras Streets. This site was established by the former Department of Health in 1988 as representative of the inner city area and is currently operated by Environmental Science and Research (ESR) on behalf of the Ministry of Health. Data from this site is also accessed by the Canterbury Regional Council and is particularly valuable because of its historical nature and central location.

Contaminants measured at the St Albans monitoring site include sulphur dioxide (SO\(_2\)), carbon monoxide (CO), nitrogen dioxide (NO\(_2\)), nitrogen oxide (NO) and suspended particulate (PM\(_{10}\)). Meteorological variables such as wind speed, wind direction, and air temperature at ground level and at 10 metre elevation are also continuously recorded at the site.

Air quality monitoring sites have also been established in a number of other locations across the city. Concentrations of PM\(_{10}\), SO\(_2\), CO and meteorological parameters have been measured at a monitoring site in Hornby since 1995 and, with the exception of CO, at a monitoring site in Opawa. Monitoring of PM\(_{10}\), CO and meteorological parameters was also conducted for a period of 2 years, from June 1995, at a site in Beckenham.

Concentrations of air pollutants in Christchurch have been shown to be high during the winter months (May - August) with peak concentrations typically occurring in June and July (figure 1.2). During the winter, temperature inversions, and low wind speeds occur regularly. The former restricts the vertical and the latter the horizontal, dispersion of contaminants. Together these conditions result in the potential for high pollution. Temperature inversions typically

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\(^1\) Key air quality indicators for New Zealand are detailed in MfE (1994)

\(^2\) Ambient air quality is the air quality in a general area, i.e., away from the influence of a specific contaminant discharge.
appear in the early evening and coincide with a period of high emissions both from the domestic home heating sector and motor vehicles. Concentrations generally increase from 4 p.m. and peak around 11 p.m. The combination of emissions and these meteorological conditions result in high pollution\(^3\) on average on about 30 nights per winter (CRC, 1996a).

![Figure 1.2 Maximum 24-hour average concentrations of PM\(_{10}\), NO\(_2\), SO\(_2\), and CO averaged monthly for the years 1988-1996 at the St Albans monitoring site.](image)

In Christchurch CO and PM\(_{10}\) regularly exceed guideline concentrations when temperature inversions and low wind speeds combine. Variations in the extent and frequency with which these guidelines are breached from year to year are primarily dependent on meteorological conditions (CRC, 1996a). Figure 1.3 illustrates the number of days per year the 24-hour average PM\(_{10}\) guideline of 50 \(\mu\)g m\(^{-3}\) and the 8-hour average CO guideline of 10 mg m\(^{-3}\) were exceeded at the St Albans monitoring site (CRC, 1996a).

\(^3\) The Canterbury Regional Council reports pollution levels as high or low on any given night based on a 24 hour average PM\(_{10}\) concentration measured from 9 am to 9am.
Comparing maximum 24-hour average PM$_{10}$ concentrations measured in Christchurch to concentrations measured in the UK (Bower et al., 1995) and US (Vedal, 1997) shows Christchurch concentrations are higher than most of the cities cited in these references. The frequency that the guideline is exceeded is comparable to London and higher than most other UK cities. The annual mean concentrations are likely to be lower in Christchurch than many of these cities due to low concentrations during the summertime and on winter days with low pollution potential (figure 1.4). However, studies indicate that the short term e.g., 24-hour average, concentrations are of greatest concern because of the potential for adverse health impacts (Foster, 1996). The implications for health and the extent to which current concentrations exceed guidelines are the reasons why the Council has identified PM$_{10}$ as the primary air quality issue in Christchurch (CRC, 1997b).

Figure 1.4 illustrates the maximum, 99 percentile, mean, 1 percentile and minimum 24-hour average PM$_{10}$ concentrations for the years 1988-1996 measured at the St Albans monitoring site. The boxes in figure 1.4 illustrate the lower quartile, median and upper quartile PM$_{10}$ concentration per year. Figure 1.4 shows peak concentrations greater than four times the current guideline and shows the low minimum and 1-percentile concentrations are low in comparison, illustrating the effect of summertime PM$_{10}$. 

Figure 1.3 Number of days 24-hour average PM$_{10}$ and 8-hour average CO concentrations have exceeded guidelines, 1988-1996
It is apparent, from the monitoring conducted, that suspended particulate concentrations in Christchurch during winter are high relative to guideline concentrations. The monitoring results summarised in this section support the Council’s approach in identifying suspended particulate concentrations as a priority air quality issue for Christchurch. The extent to which current concentrations may constitute adverse health effects is discussed in section 2.2.

1.3 Overview of the thesis

The sustainable management of the air over Christchurch requires consideration of the social, economic and technical issues relating to adverse effects of discharges to the environment. The purpose of this investigation is to consider the technical aspects of achieving sustainable management of the Christchurch air shed with respect to suspended particulate concentrations. This will be achieved by the development, validation and application of a model that assesses the effectiveness of different management options in reducing PM$_{10}$ concentrations in Christchurch.

1.3.1 Hypothesis of the thesis

The research contained in this thesis aims to explore the hypothesis that suspended particulate concentrations in Christchurch can be reduced to acceptable levels through the implementation of management options targeting primary sources of the contaminant.
Chapter one - Introduction

1.3.2 Objectives

The objective of this thesis is to identify methods that will reduce ambient PM$_{10}$ concentrations in Christchurch to acceptable levels.

1.3.3 Thesis structure

The introduction to this thesis details the legislative requirements that govern the management of air quality in New Zealand. Ambient air quality in Christchurch is considered and current suspended particulate concentrations identified as an issue requiring management intervention. This leads to the identification of a hypothesis, the thesis objectives and details of thesis structure.

Chapter two of the thesis comprises a literature review detailing research into the adverse health effects associated with suspended particulate. This health effects data indicates the potential impact of current PM$_{10}$ concentrations in Christchurch and the basis for air quality targets for PM$_{10}$. Issues relating the measurement and size distribution of PM$_{10}$ are investigated in chapter 3. These issues are important as they have implications for health and future assessment of compliance with guideline concentrations.

Chapter four develops a framework for assessing future PM$_{10}$ concentrations in Christchurch relative to current levels, assuming no additional regulatory intervention. This includes a breakdown of estimated current emissions and predicted future emissions from the domestic heating, transport and industrial sectors. This section also identifies the reduction required to meet air quality targets.

Additional components are added to the framework detailed in chapter four to establish a model that can be used to assess the effectiveness of different management options in reducing PM$_{10}$ concentrations to acceptable levels. This model and the validation of the model are described in chapter five.

Chapter six applies the model to a number of management options for domestic heating to assess their effectiveness in reducing suspended particulate concentrations to acceptable levels. The error and sensitivity of the analysis are considered as is the coincidental reductions in emissions of other contaminants associated with potential management options for reducing PM$_{10}$. Chapter seven summarises the results of the investigation and identifies areas requiring further research.
CHAPTER TWO: LITERATURE REVIEW

This chapter reviews the literature on the properties and epidemiology of suspended particulate. Properties of suspended particulate require consideration because they assist in the understanding of the basis for potential health impacts and determining sources of particles. The following describes properties of suspended particulate and adverse health effects that may be associated with suspended particulate concentrations. The literature reviewed in this chapter indicates that current PM$_{10}$ concentrations in Christchurch need reducing.

2.1 Properties of suspended particulate

Particulate matter is the total mass of airborne particles, irrespective of their chemical properties. Total suspended particulate (TSP) measures particulate in the air irrespective of size and generally captures particles less than 20 microns in diameter. The larger of these particles settle in the nose and mouth. Suspended particulate (PM$_{10}$) refers to particles less than 10 microns in diameter. Particles less than 2.5 microns in diameter are known as fine particles. The size of the particulate is particularly important in terms of its ability to penetrate into the lungs and cause adverse health effects (figure 2.1).

![Figure 2.1: Schematic diagram of penetration of particles into the respiratory system](image)

Section 2.1.2 considers sources and composition of PM$_{10}$. These aspects are important when considering strategies to reduce the potential health impact of particulates.
2.1.1 Classification of particulate

Particulate matter can be classified according to fine and coarse mode, which separates particles on the basis of source of the material. Primary fine-mode particles result from the condensation of molecules typically from combustion processes while secondary fine mode particles result through the reaction of gases, such as SO₂ with NH₄⁺, in the atmosphere (United Kingdom Department of the Environment (QUARG), 1996). These fine-mode particles are formed through the nucleation⁴ of such species and grow by coagulation⁵ and the condensation of other gases on the nuclei. Fine mode particles can be classified as either nucleation mode (new particles in ultrafine or nuclei mode) or accumulation mode (particles grown through coagulation and condensation)(Chow, 1995; USEPA, 1996; QUARG, 1996).

Nucleation mode particles are extremely small with a particle size range of approximately 1nm to 1μm. While the greatest number of particles of total suspended particulate are typically present in the nucleation mode the small size of these particles means only a small contribution to the total mass concentration (Chow, 1995). Particles in the accumulation mode are roughly in the size range 0.05μm to 2μm and can constitute a significant portion of particulate mass concentration (QUARG, 1996). These particles are long lived in the atmosphere, as removal mechanisms are least efficient in this size range (USEPA, 1996).

Coarse-mode particles are formed by crushing, grinding and abrasion of surfaces during which larger pieces of material are broken down to smaller pieces. Fungal spores, pollen, and plant and insect fragments are examples of natural bioaerosol which may form part of suspended coarse-mode particles (USEPA, 1996).

---

⁴ Molecules of complimentary substances combine to form a condensation nucleus
⁵ The combination of existing particles
The terms fine and coarse mode particles were originally intended to refer to each of the modes of distribution (nucleation and accumulation, and coarse) illustrated in figure 2.2 (USEPA, 1996). However, fine particulate is now generally associated with the PM$_{2.5}$ fraction while coarse particulate is often used to refer to the PM$_{10-2.5}$ fraction. In addition to fine-mode particulate the fine (less than PM$_{2.5}$) fraction particles will also contain the smaller of the coarse-mode particulate which can be as small as 1 μm diameter (QUARG, 1996). Similarly under high humidity conditions particles in the accumulation mode (the larger of the fine-mode) may extend into the PM$_{10-2.5}$ coarse fraction (QUARG, 1996).

2.1.2 Sources and composition of particulate

Suspended particulate matter can come from a wide variety of sources. In urban areas combustion processes are likely to be a significant source of suspended particulate. The relative significance of different combustion types e.g., domestic heating, transport or industrial combustion, will depend on location specific factors (Allen, et al.1997).

Wind blown dust is another source of particulate. This can be a significant source overseas where sand and dust storms occur (QUARG, 1996). More locally wind blown dusts can be observed from the Port Hills in the more rural areas towards the mountains on windy days. While these particles are typically larger than 10 μm in diameter, smaller particles can be
suspended in such conditions. Although rare, there is the potential for guideline exceedences in Christchurch to occur primarily as a result of wind blown dust. Another non-combustion source of PM₁₀ are roadways where deposition of particles occurs due to soil on vehicles, erosion of the road itself, and degradation of parts of the vehicle particularly the tyres (QUARG, 1996). Because these particles lie on a surface which readily dries and are subject to turbulence induced by passing vehicles, the particles can easily be re-suspended.

Breaking of waves on the sea cause the ejection of many tiny droplets of seawater into the atmosphere. These droplets dry by evaporation leaving seasalt particles suspended in the air. The majority of particulate from this source is course in size but some are small enough to have an appreciable atmospheric lifetime (QUARG, 1996).

Suspended particulate can also result from secondary reactions in the lower atmosphere (USEPA, 1996). Where these reactions occur sources of oxides of nitrogen (NOₓ), and oxides of sulphur (SOₓ) will also result in additional contributions to ambient PM₁₀. USEPA (1996) indicate that the reactions by which NOₓ and SOₓ contribute to ambient PM₁₀ are as follows:

1. NO reacts with ozone to produce NO₂ under certain conditions. SO₂ and NO₂ react with hydroxyl radical (H) during the daytime to form sulphuric and nitric acid. Particulate may be formed if these acids react with ammonia to produce ammonium sulphates and nitrates;
2. During the night-time NO₂ reacts with ozone (O₃) and forms nitric acid. Particulate may be formed if the nitric acid react with ammonia to produce ammonium nitrate;
3. SO₂ also dissolves in cloud and fog droplets where it may react with dissolved O₃, H₂O₂ or if catalysed by certain metals, with O₂, yielding sulphuric acid or sulphates, that lead to suspended particulate when the droplet evaporates.

Overseas, investigations have been carried out to identify the composition and sources of PM₁₀. The following composition and sources were identified in San Jose, California (Chow et al., 1995) for PM₁₀:

- 45% of PM₁₀ mass came from residential wood combustion;
- Motor vehicles, re-suspended road dust, and secondary particulate in the form of nitrates each contributed 15-20%;

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6 On 16th March 1989 a 24 hour average PM₁₀ concentration of 52 μg m⁻³ was measured at the St Albans monitoring site under wind speeds of 8-10 ms⁻¹
• Secondary particulate in the form of ammonium sulphate and primary marine aerosol were detectable but contributed less than 5% to the average PM$_{10}$.

Generally results from Western US indicate that fugitive dust, motor vehicles and wood smoke are the major contributors to ambient particulate concentrations with secondary particulate primarily comprising of nitrates and organic carbon (USEPA 1996). In the Eastern US sites stationary combustion and fugitive dust are major contributors to ambient particulate with sulphate and organic carbon being the main secondary components (USEPA, 1996).

Using ANSTO technology the Energy Research and Development Corporation of Australia analysed the composition of PM$_{2.5}$ particles in New South Wales in 1992 and 1993. The average composition of PM$_{2.5}$ for 1992 was:

- 23% organic matter;
- 23% ammonium sulphate;
- 22% elemental carbon;
- 6% salt;
- 1% lead;
- nitrates, trace elements, and water vapour made up the remainder (Cohen, 1995).

An emission inventory investigating sources of PM$_{10}$ and other contaminants in Christchurch was conducted during 1995 and 1996 by the Canterbury Regional Council and the National Institute of Water and Atmospheric Research (NIWA). Emissions from domestic home heating, transport (tailpipe emissions only) and industry were examined. Emissions from natural sources, re-suspended road material, and secondary particulate were not included in the assessment. Emissions from back yard burning were not included as present legislation prohibits this activity during the winter’s months when the high PM$_{10}$ concentrations are experienced in Christchurch. Of the sources considered in the inventory domestic home heating was found to be the main source of PM$_{10}$ in Christchurch (Kushel & Foster, 1996). The results of the emission inventory are discussed in more detail in section 4.2.

2.2 Health effects

Significant increases in sickness and death associated with air pollution occurred in Belgium in 1930, Pennsylvania in 1948, and in several episodes in London in the 1950s (Dockery & Pope, 1994). While the severity of these episodes was extreme and the health effects severe,
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epidemiological studies conducted in the 1990's indicate that increases in human mortality\(^7\) and morbidity\(^8\) due to particulate pollution can occur at concentrations less than current guidelines (Dockery & Pope, 1994).

The Canterbury Regional Council carried out a review of the epidemiological studies in 1995 (Foster, 1996). As a result the Council opted to reduce its monitoring and reporting guideline for PM\(_{10}\) from 120 \(\mu\text{g m}^{-3}\) (MfE, 1994) to 50 \(\mu\text{g m}^{-3}\) (24-hour average). A summary of this and details of more recent information is contained in the following sections.

2.2.1 Particulate and general mortality

Numerous studies have investigated the relationship between mortality and air pollution. These have included both longitudinal and cross sectional studies. Longitudinal time series studies, which compare daily particulate concentrations and daily counts of deaths, are common. Many such studies have been criticised for the lack of control of potential confounding factors such as smoking and occupational exposure (Dockery & Schwartz, 1995). By comparison the longitudinal cohort studies provide a more robust assessment of potential confounding factors\(^9\) (Vedal, 1997). In cohort studies a sample population is identified and exposure of this population over the study time frame is measured and estimated\(^10\). Subjects are then observed for the period of study for the occurrence of adverse health effects.

One of the most significant studies was conducted by Dockery et al., (1993) using a longitudinal cohort design. This incorporated greater than 8000 subjects and data from 6 cities. This research showed statistically significant and robust associations between air pollution and mortality after adjusting for smoking. Associations with air pollution were stronger among subjects with occupational exposure to dust, gases, or fumes. Although positive associations between mortality and air pollution levels were observed in all subgroups (defined by occupational exposure and sex), differences between subgroups were not statistically significant. Air pollution was positively associated with death from lung cancer and cardiopulmonary disease, but not with death from other causes considered together.

Pope et al.(1995) also conducted an extensive cohort study. This linked air pollution data from 151 US metropolitan areas with individual risk factors on 550 000 adults residing in these areas.

\(^7\) Loss of life, death rate.
\(^8\) Symptoms or health problems related to pollutants.
\(^9\) Confounding factors (or confounders) are other variables correlated with both the contaminant and the adverse health impact.
\(^10\) Exposure is likely to differ to measured concentrations depending on factors such as the location of the monitoring site.
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This study found significant associations between both fine particulate pollution and sulphates and cardiopulmonary mortality and mortality from all causes. Sulphate pollution was also significantly associated with lung cancer mortality. Associations between air pollution and mortality were statistically significant after adjusting for age, sex, race, various types of smoking and exposure to passive cigarette smoke, occupational exposure, education, body mass index and alcohol use.

Vedal (1997, page 559) comments that these two cohort studies have additional implications relating to the extent of effect of PM$_{10}$ on mortality. “If death is only advanced by a few days, a comparison of mortality rates across cities that experience different particle concentrations would detect no differences in mortality associated with differences in particle concentration. However, for sample populations followed for years, there is a possibility that the cohort study would detect this advance in death date as a difference in mortality rates across the cities. The fact that differences in mortality related to particle concentrations were observed in both of the cohort mortality studies supports the notion that particle exposure does not merely result in advancing the date of death by only a few days.”

Studies have also been carried out in European cities. Touloumi et al. (1994) investigated the relationship between air pollution and daily all-cause mortality in Athens using 24-hour values of sulphur dioxide, smoke, and carbon monoxide. This investigation showed that sulphur dioxide and smoke were both independent predictors of daily mortality though to a lesser extent than temperature and relative humidity. Associations between air pollution and mortality were considered in Amsterdam (population approximately 713,000) (Verhoef, et al., 1996). They concluded that black smoke and PM$_{10}$ were both positively associated with an increased risk of mortality of 1.06% per 100 μg m$^{-3}$ increase in PM$_{10}$. They also noted that the relative risk for individuals over 64 years of age was higher. Associations between particulate pollution and mortality carried out in the US compared with other countries for a 95% CI is detailed in appendix 1. This shows that in general European studies have not been as consistent in the association between suspended particulate concentrations and mortality as studies conducted in the US.

Attempts have been made to evaluate the relative importance of the physical and chemical characteristics of particulate matter on premature mortality. Dockery et al. (1992) found PM$_{10}$ to be most significantly associated with total mortality. A more recent analysis (Schwartz et al., 1996) specifically considers the effect of the fine (PM$_{2.5}$) verses the coarse (PM$_{10}$-PM$_{2.5}$) fraction. In this study PM$_{10}$, PM$_{2.5}$, PM$_{10}$-PM$_{2.5}$ and sulphate concentrations from the "six
cities" study were analysed. PM$_{10}$, PM$_{2.5}$ and sulphate particles were all significantly associated with increased daily mortality. No association was found with the coarse (PM$_{10}$ - PM$_{2.5}$) fraction. The most significant association was found with PM$_{2.5}$ with a 1.5% increase in daily mortality per 10 $\mu$gm$^{-3}$ increase in the two day mean PM$_{2.5}$ concentration (95% CI, 1.1% to 1.9%).

Dockery and Pope (1994) reviewed 8 mortality /air pollution studies in an attempt to estimate the average impact of short-term increases in inhalable concentrations on mortality. They estimated that general mortality increases by 1% in the days following each 10 $\mu$gm$^{-3}$ increase in PM$_{10}$ above 20 $\mu$gm$^{-3}$. In each study the dose-response relationship was apparently linear with increases in mortality observed above a PM$_{10}$ concentration of 20 $\mu$gm$^{-3}$. Estimates ranged from 0.7-1.6 % in the 8 studies reviewed. Results of this study are considered further in section 2.2.8.

2.2.2 Cardiovascular and respiratory mortality
Four of the studies used in Dockery and Pope's review also provided a breakdown of mortality by broad cause of death categories. Deaths due to cardiovascular causes had effect estimates ranging from 0.8% and 1.8% (95% CI) for each 10 $\mu$gm$^{-3}$ increase in PM$_{10}$ concentrations above 20 $\mu$gm$^{-3}$. Respiratory deaths had effect estimates between 1.5% and 3.7% increase for each 10 $\mu$gm$^{-3}$ increase in PM$_{10}$. In all four studies no associations were found with cancer mortality or with other causes.

The increased estimates of effects for respiratory and cardiac causes of death would imply that populations sensitive to the effects of increases in PM$_{10}$ concentrations would include those with pre-existing lung and heart diseases (Vedal, 1995). Schwartz and Dockery (1992a) identified individuals over the age of 65 as a sensitive subgroup.

While there appears to be a correlation between cardiovascular death and PM$_{10}$ concentrations the cause of this has been unclear.

Bates (1992) suggested three possible explanations for the associations between cardiopulmonary disease and particulate pollution:
1. lung disease might be incorrectly diagnosed as pulmonary oedema;
2. fine particles might increase lung permeability and precipitate pulmonary oedema in subjects with limited cardiac reserve;
3. small damage to the lung airways or the lungs themselves might precipitate heart failure in patients with pre-existing borderline heart function.
Seaton et al. (1995) proposed the hypothesis that the association between cardiovascular deaths and air pollution is due to ultra fine particulate. He suggested these particulates provoke alveolar inflammation causing hypercoagulability that results in acute coronary events. This hypothesis was investigated by Peters et al. (1997) in an experimental study measuring plasma viscosity in 3256 men and women during pollution episodes in Augsburg, Germany. During the high pollution episode an increased risk of extreme values of plasma viscosity was observed in both men and women. Peters et al. (1997, page 349) interpretation of the results was that “altered blood rheology due to inflammatory processes in the lung that induce an acute-phase reaction might therefore be part of the pathological mechanisms linking air pollution to mortality”.

2.2.3 Lung cancer
Associations between air pollution and cancer are relatively inconclusive. Each of four studies reviewed by Dockery and Pope (1994) revealed no association between PM$_{10}$ and cancer. In comparison the Dockery et al. (1993) six cities study found a positive relationship between air pollution and mortality. Another cohort study using American Lung Cancer Society data with 550,000 subjects (Pope et al., 1995) estimated a 4% increase in lung cancer mortality for a 10 µgm$^{-3}$ increase in PM$_{10}$.

2.2.4 Respiratory hospital admissions
A number of studies have assessed the effect of pollution on morbidity by considering increases in hospital admissions. Of particular significance are studies conducted in locations in which the environment itself reduces the potential for confounding by other variables.

Schwartz (1995) investigated the relationship between PM$_{10}$ and respiratory hospital admissions in an environment where SO$_2$ concentrations were low and where there was a poor correlation between temperature and PM$_{10}$. Increased risk of respiratory hospital admissions was found to be associated with both PM$_{10}$ concentrations and ozone concentrations.

In a more recent study Schwartz (1997) investigated the relationship between PM$_{10}$ and carbon monoxide (CO) and cardiovascular hospital admissions in Tuscon. Tuscon was chosen because of its low SO$_2$ concentrations and because the PM$_{10}$ concentrations peak during the winter when ozone concentrations are lowest. Both PM$_{10}$ and CO were associated with increased risk of cardiovascular hospital admissions for the study group, which consisted of persons less than 65 years old. During the period of study a total of 14,665 cardiovascular admissions occurred in this age group. Admissions increased by 2.75% per 23 µgm$^{-3}$ increase in PM$_{10}$ (95% CI). A similar study (Moolgavkar et al., 1997) found little evidence of an association between air
pollution and hospital admissions for respiratory causes in Birmingham, Alabama. In the same study ozone, PM\textsubscript{10}, SO\textsubscript{2}, and NO\textsubscript{2} were associated with increases in hospital admissions in Minneapolis-St Paul (Moolgavkar \textit{et al}, 1997).

Pope, (1992) analysed the frequencies of respiratory hospital admissions, including admissions due to pneumonia, bronchitis, and asthma, on a monthly basis against monthly PM\textsubscript{10} concentrations. A statistically significant relationship was found between respiratory hospital admissions and ambient particulate matter concentrations in two valleys in Utah between 1985 and 1989.

An alternative longitudinal study design, which has value in minimising the potential for confounding, is when a significant change in particle concentration occurs through extraordinary means. A case in point occurred in Utah Valley where the major source of emissions is a steel mill. When the mill closed for a year, due to labour strike, concentrations of particulate fell due to reasons unrelated to the normal mechanisms that influence concentration changes. Therefore potential confounding other variables could largely be ignored when comparing periods of operation to the year of closure. Hospital admissions for respiratory conditions were increased in years when the mill was operating relative to the period when it was closed (as reported in Vedal, 1997).

\textbf{2.2.5 Asthma}

An epidemiological study of diagnosed asthmatics in which subjects reported daily asthma symptoms during the duration of the study, found significant associations between the probability of moderate to severe asthma symptoms and sulphate particulate levels (Ostro \textit{et al}, 1991). The study was conducted in Denver, which has been likened to Christchurch in terms of its pollution problems (Ayrey, 1995) and only considered the winter months. Controls were made for temperature, day of study, previous-day illness, and use of a gas stove. Ozone concentrations were low and did not create a confounding influence.

Other studies have been conducted on the relationship between asthma symptoms and particulate concentrations. Schwartz \textit{et al} (1993) found a significant relationship between PM\textsubscript{10} exposure on the previous day and emergency room visits for asthma from 8 hospitals in the Seattle area for persons under 65 years. The mean PM\textsubscript{10} concentration during the study was 29.6 \textmu g m\textsuperscript{-3} and the maximum was 103 \textmu g m\textsuperscript{-3}. The authors found a 12\% increase in the likelihood of someone with asthma needing emergency treatment with every 30 \textmu g m\textsuperscript{-3} increase in the 4-day average PM\textsubscript{10}. No noticeable association was found between PM\textsubscript{10} and asthma emergency visits for the elderly.
Other studies of air pollution and respiratory effects indicate significant relationships between air pollution and asthmatic symptoms. Roemer et al. (1993) studied 73 children with chronic respiratory symptoms, all between the ages of 6 and 12 years old, from two small non-industrial towns in the Netherlands. During the study period there were 6 days when 24-hour PM$_{10}$ concentrations exceeded 110 $\mu$gm$^{-3}$. There was a small, but statistically significant, negative association between PM$_{10}$, black smoke, and sulphur dioxide with both morning and evening peak expiratory flow. However, there was a consistent positive association between these contaminants and the prevalence of wheeze and bronchodilator use.

### 2.2.6 Restricted activity days (RAD)

Ostro (1987) defined all-cause restricted activity days (RAD) as days spent in bed, days missed from work, and days when activities are partially restricted due to illness. He examined the relationship between restricted activity days per fortnight and PM$_{2.5}$ concentrations for 49 areas in the US. For the six years studied (1976-1981) he found statistically significant correlations between RAD and PM$_{2.5}$. His analysis for the six years indicates approximately 91,200 restricted activity days occur each year, per 1 million population, for every 1 $\mu$gm$^{-3}$ increase in annual average PM$_{2.5}$ (American Lung Association, 1995).

Pope et al. (1992) using records from a primary school and a district school examined the relationship between school absenteeism and PM$_{10}$ concentrations in Utah. Concentrations of PM$_{10}$ over the study period (school years of 1985-1990) averaged 50 $\mu$gm$^{-3}$ with the maximum 24-hour concentration equal to 365 $\mu$gm$^{-3}$. Correlations between absenteeism and PM$_{10}$ concentrations were positive, statistically significant and robust. Regression results indicated that an increase of 100 $\mu$gm$^{-3}$ PM$_{10}$ in the 28-day moving average was associated with an increase in the absence rate equal to approximately two percentage points or an increase in overall absences equal to approximately 40% (Pope et al., 1992).

### 2.2.7 Studies on respiratory effects in children

Dockery et al. (1989) provided evidence that rates of respiratory illnesses and symptoms are elevated for children living in cities with high particulate pollution. In their study, rates of chronic cough, bronchitis, and chest illness during the 1980-1981 school year were positively associated with all measures of particulate pollution (TSP, PM$_{15}$, PM$_{2.5}$, and fine fraction aerosol sulphate). They were also positively, but less strongly, associated with concentrations of the two gases SO$_2$ and NO$_2$. Frequency of earache was also positively associated with particulate concentrations but no association was found with asthma, persistent wheeze, hay fever, or non-respiratory illness. Results from this study also suggested that increased rates of illness were not
associated with permanent loss of pulmonary function. However, further data is needed on the long-term effects of air pollution on lung function and the later development of respiratory disorders.

2.2.8 Quantification of health effects of PM$_{10}$

Dockery and Pope (1994) reviewed a number of studies with quantified health effects associated with increases in PM$_{10}$. These indicated increases in daily mortality ranging from 0.5 to 1.5% (weighted average 1%) per 10 µgm$^{-3}$ increase in daily PM$_{10}$ concentrations greater than 20 µgm$^{-3}$. As indicated earlier the presence of a threshold below which there are no observable adverse effects is not conclusive with many studies concluding that no such lower limit has been adequately determined across a range of studies. Notwithstanding this the Dockery and Pope (1994) review has been used to assess the potential health effects associated with particulate pollution based on a no effects threshold of 20 µgm$^{-3}$. Vedal (1997) also suggests that while no threshold level appears to exist there is only meagre evidence to support occurrences of effects below a PM$_{10}$ concentration of 20 µgm$^{-3}$.

Dockery & Pope (1994) also reviewed the percentage of deaths attributed to cardiopulmonary and respiratory causes. Increases in cardiopulmonary and respiratory deaths associated with a 10 µgm$^{-3}$ increase in PM$_{10}$ ranged from 0.8 to 1.8 % (weighted average 1.4%) and 1.5-3.7% (weighted average 3.4%) respectively.

Studies reviewed by Dockery and Pope (1994) quantifying increases in other health effects indicated the following weighted averages for health effects per 10 µgm$^{-3}$ increase in daily PM$_{10}$ concentrations greater than 20 µgm$^{-3}$:

- 0.8% increase in hospitalisations for respiratory illnesses;
- 1% increase in emergency room visits for respiratory illnesses;
- 3.4% increase in emergency room visits for asthma$^{11}$;
- 1.9% increase in hospitalisations for asthma.

The American Lung Association (1995) used a selection of quantitative studies of the health impacts relative to particulate concentrations to assess the cost to society associated with suspended particulate concentrations. Their report indicated that nearly $11 billion US would be saved annually if nation-wide suspended particulate pollution was reduced to meet a standard

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$^{11}$ One study only.
of 50 μg/m³ averaged over 24-hours and annual average of 30 μg/m³ (American Lung Association, 1995).

2.2.10 Estimating health effects in Christchurch from increases in PM$_{10}$ concentrations

Few studies investigating the relationship between PM$_{10}$ and health effects have been conducted in Christchurch. One study conducted by Dawson et al. (1983) was flawed in terms of study design and statistical powers and no inferences can reasonably be drawn from them (Town, Christchurch School of Medicine, pers. comm., 1997). A more recent study of the impact of air pollution on 40 subjects aged 55-83 with chronic obstructive pulmonary disease (COPD) found both PM$_{10}$ and NO$_2$ were significantly associated with increased symptoms and medication use (Harre, et al., 1997).

Study design for an area such as Christchurch is an important factor. Christchurch has a population of approximately 310,000. This is considerably less than sample sizes used in overseas investigations. Detection of significant relationships between mortality and air pollution in Christchurch using longitudinal time series studies will be limited by the relatively small sample size. Historically these studies have also been limited by the classification system for hospital admissions and cause of death. This has recently been addressed and an investigation of associations between particulate pollution and hospital admissions is due to be completed in 1998 (Town, Christchurch School of Medicine, pers. comm., 1997).

In the absence of epidemiological studies for Christchurch estimates of potential health effects have been made based on associations from overseas studies (Foster, 1996). It should be noted that these sorts of analysis provide estimates of potential effect only do not constitute actual observed health effects. However, as discussed later in this section, potential effects do have significance under the RMA.

In Foster's (1996) analysis statistics on health effects were collected for the population of urban Christchurch living in areas that are likely to be subject to elevated pollution concentrations. Mortality and morbidity data were obtained from the Ministry of Health. Morbidity statistics were limited to public hospital admissions.

Foster (1996) uses the following assumptions in the calculation of potential health effects of PM$_{10}$ for Christchurch:

- a linear dose-response relationship (as indicated in Dockery & Pope, 1994);
• an incremental increase in mortality for each 10 $\mu g m^{-3}$ increase in PM$_{10}$ concentration above 20 $\mu g m^{-3}$ (Dockery & Pope, 1994);
• uniform exposure throughout the city urban areas (excluding the hill suburbs) to PM$_{10}$ concentrations measured at the St Albans monitoring site$^{12}$.

Foster's (1996) analysis is based on a methodology used in British Columbia where the potential health effects of particulate matter were determined by calculating daily increments of PM$_{10}$ for each 10 $\mu g m^{-3}$ increase above a concentration$^{13}$ of 20 $\mu g m^{-3}$. The health impacts associated with PM$_{10}$ were estimated in British Columbia by multiplying the daily mortality (or other measured health effect) by the percentage increase in health effect (e.g., 1% for mortality) per 10 $\mu g m^{-3}$ increase in PM$_{10}$ and by the total number of PM$_{10}$ increments per year (Vedal, 1995). The results of these calculations for mortality indicated that 82 deaths per year in British Columbia might have been associated with PM$_{10}$ concentrations (Vedal, 1995).

Foster (1996) calculates increments in PM$_{10}$ concentrations measured in Christchurch for the years 1992 and 1993, using the above methodology. This gives increments of 416 and 298 respectively. These represent a "high pollution" year (1992) and a "low pollution" year (1993). Foster (1996) estimates the potential for 21 - 29 deaths per year associated with PM$_{10}$ concentrations in Christchurch. Details of these calculations are contained in appendix 2.

Table 2.1 contains Foster's (1996) estimations of hospitalisations that may be associated with PM$_{10}$ concentrations in years equivalent to 1992 and 1993 and the total hospitalisations observed for the study area during these years. Details of these calculations are contained in appendix three.

Table 2.1: Estimates of hospital admissions that may be associated with PM$_{10}$ concentrations in Christchurch (Foster 1996)

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<tbody>
<tr>
<td>Asthma</td>
<td>17</td>
<td>13</td>
<td>810</td>
<td>846</td>
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<tr>
<td>Cardiac</td>
<td>17</td>
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<tr>
<td>Respiratory</td>
<td>11</td>
<td>8</td>
<td>1190</td>
<td>1237</td>
</tr>
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$^{12}$ Concentrations of PM$_{10}$ are likely to vary across the city. It is still to be determined how representative the St Albans data is of PM$_{10}$ concentrations in other Christchurch areas.

$^{13}$ A number of studies (Dockery & Pope, 1994; Dockery et al., 1992; Pope et al., 1992; Schwartz, 1993) showed increases in adverse health effects could be detected for every 10 $\mu g m^{-3}$ increase in PM$_{10}$ above a concentration of 20 $\mu g m^{-3}$.
Table 2.1 shows that estimates of the number of hospital admissions per year for asthma, cardiac and respiratory causes that may be associated with PM$_{10}$ concentrations is not large compared the total admissions.

The effects of increases in PM$_{10}$ concentrations on restricted activity days have also been assessed in various countries. In a study by Ostro (1987), the mean estimated coefficient for PM$_{2.5}$ across six years indicated approximately 91,200 restricted activity days (RAD) each year per 1 million population for every 1 $\mu$g m$^{-3}$ increase in annual average PM$_{2.5}$ (American Lung Association, 1995). Estimates of RAD for Christchurch made in 1996 (Foster, 1996) based on Ostro’s (1987) statistics indicated that up to 82,000 RAD per year might be associated with particulate pollution.

The calculation by Foster (1996) provides an indication of the potential impact of particulate pollution on the wider population. However, two assumptions underlying the model need to be reconsidered. The calculations were based on a relationship between PM$_{2.5}$ and PM$_{10}$ of 60%. Results of an investigation into the relationship between PM$_{10}$ and PM$_{2.5}$ in Christchurch indicate that the relationship is approximately 90% (see chapter 3). Another assumption in the calculation was a no effects threshold of 30 $\mu$g m$^{-3}$ based on studies which suggest little evidence for a threshold below 20 $\mu$g m$^{-3}$ and which consider effects in terms of 10 $\mu$g m$^{-3}$ increases. However, this assumption relates to comparisons of daily health effects and PM$_{10}$ concentrations. The extent to which this threshold applies to annual average data is unclear. Estimates of effect of RAD’s in the US (American Lung Association, 1995) based on Ostro (1987) do not consider a no effects threshold. Both these assumptions would indicate that the estimate of potential impact is conservative but also highlight the uncertainties of this calculation.

Much of the overseas data quantifying the potential effects of suspended particulate on health comes from eastern cities in the United States. Concern has been raised as to the validity of extrapolating effects observed in overseas environments to determine what the implications might be in Christchurch (Fahey, Deputy Mayor, pers. comm., 1997). There appears to be no reason to suspect that suspended particulate generated in Christchurch is less harmful that that produced elsewhere. This concern appears to be based on the assumption that the composition of the particles, which will differ from location to location, is a significant factor in determining health effects associated with suspended particulate. Veda (1997) examines the relative roles of particle composition versus particle size in determining health effects and concludes that the
evidence favouring a role for particle composition is not as convincing as the evidence favouring a role for particle size.

Notwithstanding this, Foster’s (1996) estimates of potential effects of suspended particulate based on extrapolation of overseas data are a theoretical exercise based on a number of assumptions. At best these provide an indicator of what the potential impact of suspended particulate concentrations in Christchurch may be on health.

2.2.11 Critique of health studies

As in all epidemiological studies, studies of the association between particulate and health effects are subject to the possibility that observed associations may be due to an unknown or uncontrolled factor (confounder) correlated with both exposure and disease. This could be some other pollutant or variable correlated with \( PM_{10} \) or with a specific component of the mixture that makes up \( PM_{10} \). Many studies attempt to control for potential confounding by other contaminants, both statistically and through selecting an environment with low concentrations of particular contaminants. However, the ability to control for variables such as meteorology is not as straightforward.

A number of health researchers appear to believe that the health studies do not provide strong evidence of a causal relationship between ambient particulate concentrations and health. Vedal (1997) suggests that a case against causality could be based on the following observations:

1. Similar associations between increases in particle concentration and adverse health effects are found at any range of particle concentration studied;
2. Similar associations are observed across settings where the source and chemical composition of the particles differ;
3. The health outcomes associated with particle concentration increases are not very specific, given the associations observed for cardiovascular and other outcomes;
4. There must be significant misclassification of exposure that results from using central monitors of ambient particle concentration for populations that spend the vast majority of time indoors.

Observations indicated in arguments 1 and 2 above are plausible. However it is also possible that another factor is confounding the particle-health relationship in all settings. Variability in particle concentration occurs as a result of changes in particulate emission rates and meteorological conditions. Because of the real potential for confounding by the latter, in particular the effect of temperature, investigators have attempted to be rigorous in controlling for
the potential effects of meteorology (Vedal, 1997). The ability of the models to control for potential effects of meteorology is limited by to the extent to which the appropriate meteorological conditions can be specified.

Non respiratory health outcomes (argument 3 above) have been considered by a number of researchers. As discussed earlier alternative mechanistic hypothesis (e.g., Seaton et al, 1995, Bates, 1992) have been proposed regarding the biological plausibility of the cardiovascular health outcomes.

Some misclassification of exposure, as indicated in 4 above, is inevitable and may have some impact on the analysis. In addition to problems with the location of the monitoring equipment relative to location of the subjects, differences in measurement methods may also influence the analysis. The penetration of outdoor particles to the indoors (Larsen, 1989; Schwartz et al., 1993) appear to account for exposures in indoor settings.

The association between particulate pollution and adverse health effects has been considered in terms of the Bradford Hill criteria for assessing causality (Vedal, 1997). Vedal (1997) notes that not all of these criteria need to be met for the relationship to be considered causal and concludes that the relationship between particulate pollution and mortality seems to satisfy many, although not all, of the Bradford Hill criteria.

Despite some inconsistencies in the relationship between particulate pollution and adverse health effects (as shown in appendix 1) a substantial number of studies show associations between suspended particulate and adverse health effects. These have been carried out in a variety of locations in attempts to exclude potential confounding factors. The study of the steel mill closure in Utah, reported in Vedal (1997), would seem particularly significant for the control of potential confounding by meteorology and other variables. Overall evidence favours a causal relationship between particulate pollution and adverse health effects.

As discussed previously the legislation governing the management of air quality in New Zealand is the Resource Management Act. Section 3 of this Act defines "effects" as including any potential effect of high probability and any potential effect of low probability, which has a high potential impact. The health effects literature indicates that elevated PM$_{10}$ concentrations may be associated with several adverse health impacts including mortality. While these epidemiological studies are not conclusive in establishing cause and effect, the weight of evidence supports setting policies to reduce the potential for adverse health effects as indicated
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by these studies. The following section details the guidelines for particulate chosen by authorities for health protection.

2.3 Guidelines for PM$_{10}$

Health authorities in a number of countries have used health effects research as the basis for reviewing air quality guidelines for suspended particulate. Details of revised guidelines for PM$_{10}$ for a number of countries follow and are summarised in table 2.2.

In 1995 the United Kingdom Expert Panel on Air Quality Standards recommended an air quality target standard of 50 [$\mu$gm$^{-3}$] (24-hour average) based on a 99-percentile concentration. The UK government has accepted the judgement of the Expert Panel on Air Quality and is developing a strategy to achieve air quality standards and objectives. The draft strategy sets a target of the year 2005 for achievement of its objectives (United Kingdom Department of the Environment, 1996).

In 1996 WHO released a draft of new ambient air quality guidelines (AAQG) for a number of contaminants. These acknowledged that there did not appear to be a no observable adverse effect level (NOAEL) for PM$_{10}$ and include a guideline concentration of 0 [$\mu$gm$^{-3}$]. WHO suggest an approach based on quantification of potential adverse health effects.

The Canadian authorities are in the process of reviewing their National Ambient Air Quality Objectives (NAAQO). At this stage they appear to be considering a two-tier approach which consists of:

1) A reference level - a level above which there are demonstrable effects on human health and the environment. This reference level provides the basis for establishing goals for long term air quality management.

2) An air quality objective - A level selected based upon consideration of scientific, social, economic and technological factors. It provides a basis for air quality management and is intended to provide protection for the general population and environment.

Reference levels of 25 [$\mu$gm$^{-3}$] for PM$_{10}$ and 15 [$\mu$gm$^{-3}$] for PM$_{2.5}$ (24-hour average) are suggested (Bhattacharyya, Senior Scientist, Greater Vancouver Regional District, pers. comm., 1997).

The Province of British Columbia, in the absence of any NAAQO, set an air quality objective of 50 [$\mu$gm$^{-3}$] in 1995 (Bhattacharyya, Senior Scientist, Greater Vancouver Regional District, pers. comm., 1997).
Chapter two - Literature Review

The United States Environmental Protection Agency (USEPA) set their National Ambient Air Quality Standards (NAAQS) for PM$_{10}$ of 150 $\mu$g m$^{-3}$ (24-hour average), based on particulate matter health effects as determined in 1986. In the 1980s the state of California reviewed essentially the same health effects literature and set state ambient air quality standards of 50 $\mu$g m$^{-3}$ PM$_{10}$ averaged over 24-hours (American Lung Association, 1995). In the 1990s the USEPA were required, as a result of an American Lung Association lawsuit, to review the PM$_{10}$ standards.

The USEPA decided on an additional standard for PM$_{2.5}$ rather than a review of the PM$_{10}$ standard. This decision was based on observations indicating that the fine size fraction appears more pathogenic than the coarse fraction. The introduction of a new indicator would seem less stringent, in the US situation, than a revision of the PM$_{10}$ guideline as several years monitoring is required to determine non-compliance. The EPA staff initially recommended a PM$_{2.5}$ standard in the range of 18-65 $\mu$g m$^{-3}$ 24-hour average and had difficulty reaching a consensus on the exact level, on the averaging time or on the form of the PM$_{2.5}$ standard.

In the US the Clean Air Act requires that the National Ambient Air Quality Standards be reviewed every 5 years. The next review is scheduled to begin during the year 2000. A review at this time will have access to an additional 5 years of research targeting current information gaps and areas of concern regarding the establishment of a new standard. Consequently the USEPA conclude "...there does not appear to be any compelling reason to set a restrictive PM$_{2.5}$ standard at this time" (Wolff, 1996). In 1997 the USEPA set a PM$_{2.5}$ standard of 65 $\mu$g m$^{-3}$, 24-hour average.

Australia's National Environment Protection Council discussion document (1997) proposes national air quality standards for PM$_{10}$ and PM$_{2.5}$ of 50 $\mu$g m$^{-3}$ and 25$\mu$g m$^{-3}$ respectively. These standards are based on an allowable exceedence of 1 day per year (Environment Australia, 1997).

In 1997 the Canterbury Regional Council adopted a guideline for PM$_{10}$ of 50 $\mu$g m$^{-3}$, 24-hour average.
### Table 2.2: Summary of guideline concentrations for suspended particulate

<table>
<thead>
<tr>
<th>Country</th>
<th>24-hour average PM_{10} concentration</th>
<th>24-hour average PM_{2.5} concentration</th>
<th>Allowed exceedences (days per year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>United Kingdom</td>
<td>50 µg/m³</td>
<td>-</td>
<td>3</td>
</tr>
<tr>
<td>WHO</td>
<td>0 µg/m³</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>United States</td>
<td>150 µg/m³ (not revised)</td>
<td>65 µg/m³ (new guideline)</td>
<td></td>
</tr>
<tr>
<td>Australia</td>
<td>50 µg/m³</td>
<td>25 µg/m³</td>
<td>1</td>
</tr>
<tr>
<td>Canada</td>
<td>Under review</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>British Colombia</td>
<td>50 µg/m³</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>Canterbury Regional Council</td>
<td>50 µg/m³</td>
<td>-</td>
<td>1</td>
</tr>
</tbody>
</table>

### 2.5 Summary

The literature review indicates that sources of PM\textsubscript{10} vary between locations. The proportion of primary versus secondary particulate depends on the concentrations of chemicals such as O\textsubscript{3}, \textit{O}H, and H\textsubscript{2}O\textsubscript{2} and climatic conditions in the lower atmosphere. No studies were found that indicated the contribution of secondary particulate to measured PM\textsubscript{10} concentrations in Christchurch. The emission inventory for Christchurch does not consider secondary particulate formation and therefore may not accurately predict the relative amounts of particulate from each source. It is likely, however, that secondary particulate formation in wintertime\textsuperscript{14} is not as significant in Christchurch as in many overseas cities due to low wintertime temperatures and low ground level ozone concentrations (Ayrey, Canterbury Regional Council, pers. comm., 1998).

Investigations into adverse health effects and particulate pollution repeatedly show significant relationships for a variety of health effects such as mortality (both cardiovascular and respiratory) and respiratory disorders. These studies provide the basis for the review of air quality guidelines for PM\textsubscript{10} in a number of countries. Air quality guidelines established by the CRC in 1996 are consistent with those used in or proposed for the UK, Australia and British Colombia.

\textsuperscript{14} PM\textsubscript{10} concentrations in Christchurch exceed guidelines during the winter months only.
Chapter two - Literature Review

The literature review, combined with data from section 1.2 (current PM$_{10}$ concentrations in Christchurch) indicates that current suspended particulate concentrations are a concern in Christchurch and that these concentrations may result in a range of adverse health effects.
CHAPTER THREE: MEASUREMENT METHODS TO MONITOR PM\textsubscript{10} CONCENTRATIONS IN CHRISTCHURCH

3.1 Background

An important aspect of setting guidelines or air quality target is specifying the measurement method for assessing compliance. Chow (1995) indicates that the methods of measuring compliance should be specified in the development of guidelines and management strategies. This is because of variances observed between different measurement techniques in a number of overseas studies.

A number of different methods have been used at the St Albans monitoring site to determine concentrations of PM\textsubscript{10}. This chapter investigates several different methods of measurement and the issues related to the measurement of suspended particulate concentrations in Christchurch.

The purpose of the monitoring is one consideration in determining an appropriate measurement method. Air quality monitoring data in Christchurch serves a variety of purposes. These include:

1. assessing compliance with air quality guidelines;
2. studying health effects epidemiology;
3. comparing concentrations in different areas of the city and region;
4. assessing trends in air pollution;
5. air quality investigations e.g., concentrations of SO\textsubscript{2}, PM\textsubscript{10} and NO\textsubscript{2} used in an investigation into daytime haze in Christchurch;
6. temporal variations in air quality, particularly over a 24-hour period, assist in the understanding of air quality in Christchurch.

To achieve the outcomes in 1-6 above accurate measurement for averaging periods of 1 hour or less are of primary importance. Automated methods are also preferred in the Canterbury Region due to the extensive monitoring network. Because of this continuous analysers (TEOM (tapered elemental oscillating microbalance) and beta gauge methods) are the predominant measurement methods used to record PM\textsubscript{10} concentrations at Christchurch's St Albans monitoring site and other monitoring sites in the Canterbury Region. These methods can provide data for hourly and 10 minute (TEOM only) intervals and are less labour intensive than gravimetric methods.

The gravimetric method is the specified reference method for the USEPA. This method uses gravimetric analysis of particulate captured through an airflow of 1133 litres/min by a 25cm x
20cm typically glass-fibre filter. USEPA reference method and equivalency status criteria are described in appendix 4.

The TEOM and beta gauge methods meet the USEPA equivalency status. The operational principals for the TEOM and beta gauge methods are detailed in Chow, (1995).

Investigations in many overseas cities, detailed in Chow (1995), show differences in monitoring results for two or more different collocated PM$_{10}$ measurement methods. Many of these differences were attributed to quality assurance procedures and differences in size selective inlets. However, differences observed between TEOM analysers and other methods have been shown to occur due to volatilisation of compounds such as ammonium nitrate and organics abundant in wood smoke (Meyer, 1996; Chow, 1995). This occurs because the tapered element of the TEOM requires a constant temperature and has a default setting of 50°C to minimise effects of thermal expansion and contraction of the PM$_{10}$ on the filter.

The advantage of the TEOM over other methods of PM$_{10}$ measurement is its sensitivity to changes in mass concentration. This results in it being able to provide precise measurements for time periods of less than one hour. Because the TEOM is a direct measurement of mass concentration some inaccuracies associated with other measurement methods are avoided. The TEOM is the Canterbury Regional Council’s preferred method of measurement and is being widely adopted in New Zealand.

Suspended particulate concentrations in Christchurch have been measured using a TEOM with an inlet temperature of 50°C since 1994. Because wood smoke is a predominant source of PM$_{10}$ in Christchurch it is likely that these measurements are lower than measurements made using alternative techniques. Although the TEOM is a good method for providing the information that the CRC requires from its monitoring network, there are some problems that arise from employing this technology. The following section addresses these problems. In addition to these monitoring issues, health effects data, discussed in chapter two, indicates that PM$_{2.5}$ may have greater implications for health that the coarse particles. This section also attempts to identify the proportion of the PM$_{10}$ that is PM$_{2.5}$.

The purpose of these investigations is to determine:

- The extent to which TEOM analysers operating at an inlet temperature of 50°C record lower concentrations than other measurement methods;
- The impact of reducing the TEOM inlet temperature to 30°C;
• The relationship between PM$_{10}$ and PM$_{2.5}$ concentrations in Christchurch.

**3.2 Method**

To explore the above issues the following methods were used.

1) Measurement of PM$_{10}$ at Christchurch’s St Albans monitoring site during winter 1996 using:
   - Rupprecht & Patashnick Tapered Elemental Oscillating Microbalance (TEOM);
   - Wedding and Associates Beta Gauge;
   - Wedding and Associates High Volume;

2) Measurement of PM$_{10}$ at Christchurch’s St Albans monitoring site during winter 1997 using:
   - Rupprecht & Patashnick Tapered Elemental Oscillating Microbalance (TEOM), operated with an inlet temperature of 50 °C, measuring PM$_{10}$;
   - Rupprecht & Patashnick Tapered Elemental Oscillating Microbalance (TEOM), operated with an inlet temperature of 30 °C, fitted with a size selective inlet measuring PM$_{10}$;
   - Rupprecht & Patashnick Tapered Elemental Oscillating Microbalance (TEOM), operated with an inlet temperature of 30 °C, fitted with a size selective inlet measuring PM$_{2.5}$;
   - Wedding and Associates Beta Gauge measuring PM$_{10}$.

Relationships between 24-hour average paired observations were evaluated by regression analysis and in terms of the percent difference between the observations (e.g., (TEOM-hi-voL)/Hi-voL). Relationships were also considered for concentrations greater than 20 μgm$^{-3}$ and 50 μgm$^{-3}$. The latter are investigated to compare the correlation for higher concentrations. Concentrations greater than 20 μgm$^{-3}$ are considered, as measurement inaccuracies are common for some measurement methods concentrations lower than this.

**3.3 Results**

**3.3.1 Different instruments**

The comparison between 24-hour average concentrations for the TEOM, beta gauge, and high volume samplers are shown in tables 3.1 and 3.2.
Table 3.1: Results of regression analysis on 1996 collocated analysers

<table>
<thead>
<tr>
<th>Analyser</th>
<th>Dependent variable</th>
<th>$r^2$</th>
<th>Equation</th>
</tr>
</thead>
<tbody>
<tr>
<td>TEOM vs High volume</td>
<td>TEOM</td>
<td>0.97</td>
<td>$y = 0.61x + 2.7$</td>
</tr>
<tr>
<td>Beta gauge vs High volume</td>
<td>Beta gauge</td>
<td>0.98</td>
<td>$y = 0.93x + 1.9$</td>
</tr>
<tr>
<td>TEOM vs Beta gauge</td>
<td>TEOM</td>
<td>0.98</td>
<td>$y = 0.66x + 0.8$</td>
</tr>
</tbody>
</table>

Table 3.2: Mean percentage difference in 24-hour average concentrations during 1996

<table>
<thead>
<tr>
<th>Analyser</th>
<th>Mean % difference</th>
<th>Mean % difference for concentrations &gt;20 $\mu$g m$^{-3}$</th>
<th>Mean % difference for concentrations &gt;50 $\mu$g m$^{-3}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>TEOM vs High volume</td>
<td>-28%</td>
<td>-23%</td>
<td>-37%</td>
</tr>
<tr>
<td>Beta gauge vs High volume</td>
<td>4%</td>
<td>2%</td>
<td>-3%</td>
</tr>
<tr>
<td>TEOM vs Beta gauge</td>
<td>-32%</td>
<td>-30%</td>
<td>-33%</td>
</tr>
</tbody>
</table>

These results show beta gauge and high volume 24-hour average concentrations measured during 1996 are highly correlated ($r^2 = 0.98$) with little variation in measurements (mean percent difference of 2% for concentrations greater than 20 $\mu$g m$^{-3}$). The 1996 TEOM concentrations are also highly correlated with both the beta gauge and the high volume sampler ($r^2 = 0.97$ and 0.98). However, measured concentrations are significantly lower ($p = \text{less than } 0.01$), with mean percentage differences of -30% and -23% for the beta gauge and high volume samplers respectively.

3.3.2 Different inlet temperatures and size fractions

Tables 3.3 and 3.4 compare 24-hour average PM$_{10}$ and PM$_{2.5}$ concentrations for the TEOM at different inlet temperatures and for a beta gauge.
Table 3.3: Results of regression analysis on 1997 collocated analysers

<table>
<thead>
<tr>
<th>Analyser</th>
<th>Dependent variable</th>
<th>$r^2$</th>
<th>Equation</th>
</tr>
</thead>
<tbody>
<tr>
<td>PM$_{2.5}$ TEOM vs TEOM @ 30°C</td>
<td>PM$_{2.5}$ TEOM</td>
<td>0.99</td>
<td>$y = 0.96x - 5.2$</td>
</tr>
<tr>
<td>PM$_{2.5}$ TEOM vs TEOM @ 50°C</td>
<td>TEOM @ 50°C</td>
<td>0.98</td>
<td>$y = 0.76x + 7.3$</td>
</tr>
<tr>
<td>TEOM @ 50°C vs TEOM @ 30°C</td>
<td>TEOM @ 50°C</td>
<td>0.99</td>
<td>$y = 0.75x + 1.7$</td>
</tr>
<tr>
<td>Beta gauge vs TEOM @ 50°C</td>
<td>TEOM @ 50°C</td>
<td>0.97</td>
<td>$y = 0.63x + 0.3$</td>
</tr>
<tr>
<td>Beta gauge vs TEOM @ 30°C</td>
<td>TEOM @ 30°C</td>
<td>0.98</td>
<td>$y = 0.84x - 3.1$</td>
</tr>
</tbody>
</table>

Table 3.4: Mean percentage difference in 24-hour average concentrations during 1997

<table>
<thead>
<tr>
<th>Analyser</th>
<th>Mean % difference</th>
<th>Mean % difference for concentrations $&gt;20$ $\mu$g.m$^{-3}$</th>
<th>Mean % difference for concentrations $&gt;50$ $\mu$g.m$^{-3}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>PM$_{2.5}$ TEOM vs TEOM @ 30°C</td>
<td>-19%</td>
<td>-11%</td>
<td>-8%</td>
</tr>
<tr>
<td>TEOM @ 50°C vs TEOM @ 30°C</td>
<td>-19%</td>
<td>-23%</td>
<td>-24%</td>
</tr>
<tr>
<td>Beta gauge vs TEOM @ 50°C</td>
<td>-34%</td>
<td>-32%</td>
<td>-38%</td>
</tr>
<tr>
<td>Beta gauge vs TEOM @ 30°C</td>
<td>-22%</td>
<td>-18%</td>
<td>-20%</td>
</tr>
</tbody>
</table>

These results show high correlations ($r^2 = 0.97 - 0.99$) between 24-hour average concentrations measured by all particulate analysers during the 1997 study. TEOM measurements were lower than those measured using the beta gauge at both the higher (50°C) and lower (30°C) inlet temperature. The relationship and mean percentage differences were similar to the 1996 study for the TEOM operating at an inlet temperature of 50°C. The mean percentage difference between the beta gauge and the TEOM operating at an inlet temperature of 30°C is lower (-22%) than for the TEOM operating at the higher inlet temperature (-34%).

3.4 Discussion

At the lower inlet temperature of 30°C TEOM analysers still record 24-hour average PM$_{10}$ concentrations lower than those measured using a beta gauge, and presumably the hi-volume sampler$^{15}$. However, the average difference is reduced from approximately 34% for the inlet

$^{15}$ The Wedding beta-gauge measured 24 hour average PM$_{10}$ concentrations comparable to those measured using a Wedding high-volume sampler ($r^2 = 0.98$, $y=0.93x+1.9$). The beta-gauge is initially calibrated against a hi volume sampler.
operating at 50°C to approximately 22% for the inlet temperature of 30°C. The implications of this are interesting, particularly because, while there are a number of different standards specifying procedures for different methods of measurement, at present there is no standard that specifies the conditions of measurement of PM\textsubscript{10}. Meyer (1996) considers that it is unlikely that any filter based or non-filter-based method can give an absolute measure of all ambient particulate matter species present. Meyer (1996) suggests that in the absence of standardising key parameters such as sampling temperature, filter face velocity, equilibration temperature and relative humidity measurements of PM\textsubscript{10} mass can vary by as much as a factor of two. This suggests that the gravimetric method is also not an ideal method of measurement and that in the absence of standardisation of measurement conditions there is no "right" concentration. Notwithstanding this, methods that minimise bias that may occur as a result of variations in the particle size to mass ratio or through the measurement of atmospheric moisture are preferable. The TEOM method of measurement is currently the only continuous method that is a direct measure of mass concentration.

Guidelines or standards for PM\textsubscript{10} are based on adverse health effects. It could be suggested that conditions of measurement should replicate those that impact on health. It would follow then that a temperature of 37°C and 100% relative humidity might be the best conditions under which PM\textsubscript{10} should be measured. However, in areas where volatilisation of organics is experienced the inlet temperature is typically reduced to 30°C. In the absence of any national or international standards specifying the conditions of measurement for PM\textsubscript{10} continued use of the TEOM operating at an inlet temperature of 30°C is an appropriate monitoring method for the Canterbury Region.

The relationship between the PM\textsubscript{2.5} and the PM\textsubscript{10} both at 30°C indicates that, particularly at higher concentrations, a large proportion (approximately 90%) of the PM\textsubscript{10} is PM\textsubscript{2.5}. This is higher than relationships observed in the United States where PM\textsubscript{2.5} constitutes approximately 60% of PM\textsubscript{10}, on average (Dockery & Pope, 1994). Crustal particulate (PM\textsubscript{10} - PM\textsubscript{2.5}) as fugitive dust is a significant contributor to PM\textsubscript{10} concentrations in both Eastern and Western US cities. The higher proportion of PM\textsubscript{2.5} may have implications for health. A large proportion of the quantified epidemiological research is from cities in the United States. If, as suggested by Schwartz et al. (1996), it is the smaller PM\textsubscript{2.5} particles that are associated with adverse health effects the relationship between health effects and PM\textsubscript{10} may be greater in an environment with a larger PM\textsubscript{2.5} portion. These results suggest that this may be the case in Christchurch.
3.5 Summary
This chapter compares PM$_{10}$ concentrations using different techniques and instrument settings. Monitoring results using the TEOM method were lower than other methods. Differences were reduced when the TEOM inlet temperature was decreased from 50°C to 30 °C. The TEOM operating at an inlet temperature of 30°C is a good measurement option for the Canterbury Region because the method is sensitive to changes in mass concentration for periods of less than one hour and potential bias associated with other methods is avoided.
CHAPTER FOUR: DEVELOPMENT OF A FRAMEWORK FOR ASSESSING METHODS TO REDUCE PM$_{10}$

Chapter four uses the results of a number of investigations commissioned by the CRC. These include:

- an air shed model (Gimson & Fisher, 1997) which models the relationship between PM$_{10}$ emissions and measured concentrations;
- projected home heating methods (Wood, 1997a), industrial growth (Barber, 1997), and transport emissions (Wood, 1997b);
- an emission inventory investigating sources of PM$_{10}$ emissions in Christchurch (CRC, 1997a);
- an allocation mechanisms report (Brady & Keller, 1997) which considers various ways of allocating a reduction in emissions across sectors$^{16}$.

In this chapter the results of these studies are used to estimate future PM$_{10}$ concentrations in Christchurch relative to current levels. This is then used as the basis for a model, described in chapter five, that will be used to assess the effectiveness of different management options in chapter six. This stage will be constructed using the results from three experiments:

a) the reductions required in PM$_{10}$ concentrations
b) the relative contribution of different sectors to PM$_{10}$ concentrations
c) estimating future PM$_{10}$ emissions from major sectors

Results from b and c above are used in conjunction with earlier work (Brady & Keller, 1997) to allocate reductions between primary sources.

4.1 Reductions required in suspended particulate concentrations

4.1.1 PM$_{10}$ concentrations in Christchurch

The St Albans site is Christchurch’s primary air quality monitoring site for the monitoring and reporting of air quality. This is largely because of its central location as well as the historical significance of the site. This site was selected by the former Department of Health to be “representative” of Christchurch air quality. However, it has been shown from monitoring carried out in other locations that there are variations in PM$_{10}$ concentrations across the city on a typical winter’s night (CRC, 1996a).
The air quality monitoring sites located in Hornby and Beckenham both measure PM$_{10}$ concentrations. The concentrations recorded at these sites are usually lower than those recorded at St Albans. These sites report considerably fewer days when the PM$_{10}$ guideline is exceeded than St Albans. However a monitoring site in Opawa measures similar concentrations to St Albans. Although the numbers of days in which the PM$_{10}$ guideline is exceeded is fewer (CRC, 1996b). Smoke monitoring carried out during 1993 found concentrations were higher in Linwood and Beckenham than St Albans while concentrations in Burnside were slightly less than St Albans (CRC, 1993).

The recently released emission inventory for Christchurch (CRC, 1997a) provides an indication of variations in emission rates across the city. The St Albans monitoring site, located near the corner of Purchase and Madras Streets, borders the emission inventory’s St Albans survey area and Central Business District survey area. Compared with other inner city survey areas these are high and low emission density areas respectively. On an emission density basis the site is likely to represent average emissions for the inner city area. Emissions of PM$_{10}$ outside of the inner Christchurch area, on average, are lower than those for the inner city area. However, some locations such as Bryndwr indicate higher emissions than those estimated for the inner city areas (CRC, 1997a). It is important to note, however, that the quantity of PM$_{10}$ emissions in any area is not an indication of PM$_{10}$ concentrations due to the impact of meteorological conditions.

Based on the research presented here it is likely that a reduction in PM$_{10}$ concentrations based on the St Albans monitoring data will achieve air quality goals for most if not all of the city.

### 4.1.2 Achieving the air quality target

The overall level of reduction required has been based on an air quality target of 50 $\mu$g m$^{-3}$ 24-hour average (section 2.3). This goal is based on the 99.7 percentile of concentrations measured from the years 1988-1996. The 99.7 percentile concentration can be calculated non-parametrically by taking the average of the maximum 24-hour average PM$_{10}$ concentration for each year. The 99.7 percentile is the minimum acceptable period of exceedence being an average of one day per year (Brady & Keller, 1997). Appendix 7 contains concentrations of

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16 For the purpose of this report sectors are limited to the domestic heating, transport and industrial sectors.
PM\textsubscript{10} associated with various percentiles\textsuperscript{17} and the corresponding reduction required to get to 50 μgm\textsuperscript{-3}, calculated using equation E4.1. The 99.7 percentile concentration of the 24-hour average PM\textsubscript{10} concentrations measured from 1988-1996 is 189 μgm\textsuperscript{-3}.

\begin{equation}
E4.1 \quad R = 100(1 - \frac{t}{c})
\end{equation}

where
- \( R \) = the percentage reduction
- \( t \) = the air quality target (50 μgm\textsuperscript{-3})
- \( c \) = the percentile concentration

*Equation E4.1: Percentage reduction to reach air quality target*

The reduction required in 1996 emissions to reduce PM\textsubscript{10} concentrations to meet a target of 50 μgm\textsuperscript{-3} (24-hour average), allowing on average one exceedence per year, was calculated to be 74% (i.e., \( 100(1 - \frac{50\mu gm^{-3}}{189\mu gm^{-3}}) \)). However, a recent publication by the Ministry for the Environment (MfE, 1997) indicates that air quality guidelines should be regarded as a minimum requirement to maintain air quality. They propose categories to describe air quality to assist in setting goals for air quality management. The Ministry for the Environment’s (1997) proposed categories and the percentage ranges of the guideline value are shown in Table 4.1. To achieve the MfE “acceptable” category for PM\textsubscript{10} concentrations in Christchurch reductions of more than 74% would be required.

\textsuperscript{17} Calculated using the CRC air quality database for the years 1988-1996 for the St Albans monitoring site.
Table 4.1: Proposed air quality categories (MtE, 1997)

<table>
<thead>
<tr>
<th>Category</th>
<th>Maximum measured value</th>
<th>Comment</th>
</tr>
</thead>
<tbody>
<tr>
<td>Action</td>
<td>exceeds the guideline</td>
<td>completely unacceptable, by national and international standards.</td>
</tr>
<tr>
<td>Alert</td>
<td>between 66% and 100% of the guideline</td>
<td>a warning level, which can lead to guidelines being exceeded if trends are not curbed.</td>
</tr>
<tr>
<td>Acceptable</td>
<td>between 33% and 66% of the guideline</td>
<td>a broad category, where maximum values might be of concern in some sensitive locations, but are generally at a level which does not warrant dramatic action.</td>
</tr>
<tr>
<td>Good</td>
<td>between 10% and 33% of the guideline</td>
<td>peak measurements in this range are unlikely to affect air quality.</td>
</tr>
<tr>
<td>Excellent</td>
<td>less than 10% of the guideline</td>
<td>of little concern. If maximum values are less than a tenth of the guideline, average values are likely to be much less.</td>
</tr>
<tr>
<td>Not Assessed</td>
<td></td>
<td>insufficient monitoring data to assess the category.</td>
</tr>
</tbody>
</table>

The calculation of the reduction required uses monitoring data from the St Albans monitoring site for the years 1988-1996. The three years from 1994-1996\(^{18}\) include data measured using a TEOM with an inlet temperature of 50°C whereas a beta gauge was used prior to this. Differences in \(PM_{10}\) measurements made using the different methods are discussed in chapter three. It is recommended in that chapter that future monitoring be conducted using a TEOM operating with an inlet temperature of 30°C. However, chapter three shows that this method measures \(PM_{10}\) concentrations approximately 20% lower than historical methods.

The implications of these differences for the air plan, however, are not that significant. Because none of the methods used are “wrong” it is not appropriate to adjust the historical data for consistency with current methods. If the historical data were adjusted the reduction required would be slightly less (69% as opposed to 74%).

\(^{18}\) A TEOM was used from 17 August 1994.
Chapter four - Development of a framework for assessing methods to reduce PM$_{10}$

4.2 Sources of ambient PM$_{10}$ concentrations

4.2.1 Sources of PM$_{10}$ emissions
The relative contribution of the industrial, transport and home heating sectors to total PM$_{10}$ emissions was investigated through an emissions inventory conducted during 1995 and 1996 (CRC, 1997a). An attempt to validate the emission inventory has been made using domestic heating surveys (Lamb, 1997; Masilamani, 1997) fuel and energy use data (CRC, 1997, Brady and Pullen, 1985) and an assessment of PM$_{10}$ from motor vehicle emissions in Christchurch using a different methodology (Wood, 1997b). These comparisons are discussed later in this section.

As a part of this emission inventory domestic home heating emissions were assessed through home heating survey of approximately 800 households that was conducted across Christchurch. This survey aimed to determine:

- The type of home heating methods used on a typical winter’s night;
- The type and amount of fuels used on different appliances;
- Diurnal variation in fuel use.

From this data the amount of emissions were derived$^{19}$.

Motor vehicle emissions were investigated by examining, from a road network model, the number of vehicle kilometres travelled in Christchurch for different vehicle classes and fuel types used. Emissions were then applied per kilometre travelled on the basis of vehicle class, fuel and the speed of travel in different locations and for different times of the day.

Industrial emissions were assessed based on information contained in applications for resource consents and historical licensing data.

Emissions were collated on an urban-wide basis and an inner-city basis. Emissions from the inner city area were used in the derivation of an emissions model relating emissions to concentrations measured at the St Albans monitoring site$^{20}$. Table 4.2 shows results of the emissions inventory data for the urban Christchurch area and for the inner city area. It also shows the temporal variations in emissions.

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$^{19}$ Chapter 5.0 details on how the quantities of emissions are derived from this data.

$^{20}$ The St Albans monitoring site is located in the inner city. Emissions from the inner city are most likely to impact on concentrations measured at this site.
Table 4.2: Wintertime PM$_{10}$ emissions for different sectors from urban and inner Christchurch by time of day (CRC, 1997a)

<table>
<thead>
<tr>
<th>Time of Day</th>
<th>Home heating PM$_{10}$ emissions g/ha</th>
<th>Transport PM$_{10}$ emissions g/ha</th>
<th>Industrial PM$_{10}$ emissions g/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Inner city Urban Chch</td>
<td>Inner city Urban Chch</td>
<td>Inner city Urban Chch</td>
</tr>
<tr>
<td>6 a.m. - 10 a.m.</td>
<td>64 44</td>
<td>27 17</td>
<td>14 11</td>
</tr>
<tr>
<td>10 a.m. - 4 p.m.</td>
<td>125 91</td>
<td>56 34</td>
<td>26 20</td>
</tr>
<tr>
<td>4 p.m. - 10 p.m.</td>
<td>652 406</td>
<td>36 22</td>
<td>18 12</td>
</tr>
<tr>
<td>10 p.m. - 6 a.m.</td>
<td>88 77</td>
<td>6 4</td>
<td>18 11</td>
</tr>
<tr>
<td>Total</td>
<td>929 618</td>
<td>125 77</td>
<td>76 54</td>
</tr>
<tr>
<td>Percentage of total</td>
<td>82.2 82.5</td>
<td>11.0 10.2</td>
<td>7.0 7.0</td>
</tr>
</tbody>
</table>

These estimated emissions are used to define a relative contribution of the different sectors to PM$_{10}$ emissions for Christchurch as illustrated in figure 4.1. The relative contributions of sources within the inner and the wider urban Christchurch areas are very similar.

Figure 4.1: Relative contribution of emissions from different sectors (based on 1996 emissions data)

The basic emission inventory methodology used in the Christchurch emissions inventory is based on standard international practise. However, the specific methodology used for the Christchurch inventory is more detailed than typical inventories conducted overseas (pers. comm., Fisher, NIWA, 1997).
Chapter four - Development of a framework for assessing methods to reduce PM$_{10}$

Validation of domestic heating surveys
Results of the home heating survey in terms of households using different home heating methods can be compared to similar surveys such as Lamb (1997) and Masilamani (1997). These surveys are typically different in the questions asked. However, a comparison of the results, from the two examples with the authors indicates that they are not inconsistent with the emission inventory (pers. comm., Masilamani, 1997; Lamb, 1997).

Validation of domestic heating emissions
The amount of fuel used for domestic heating as determined by the emission inventory was compared to the Regional Energy Survey (CRC, 1996b). The use of wood based on the emission inventory was higher than predicted in the Regional Energy Survey$^{21}$. However, the latter survey estimated the proportion of self-collected wood to be 20%. Survey results for the emission inventory showed that approximately 50% of wood used in Christchurch was self-collected. The estimate used for the fuel use survey is therefore likely to be low, accounting for a differences in wood use between the two studies (Ayrey pers. comm., 1998).

The emission inventory also shows a greater amount of coal is burnt in Christchurch than is estimated using the Regional Energy Survey (1996b). Differences may be because of predictions that a large amount of coal is privately transported from the West Coast for domestic use in Christchurch (John Dyer, Coal Research, pers. comm., 1997).

The emission factors applied are estimates based on overseas research and have not been validated by stack testing in Christchurch. The basis for these emission factors is discussed in appendix 5. The extent to which the emission factors used adequately account for the burning of wet wood is uncertain. Emission factors used are based largely on Todd's (1994) estimates for dry wood. However, these are reasonably consistent with USEPA emission factors, which are based on actual operating conditions. The burning of wet wood has been identified as one variable that can impact on emissions. Although the impact is thought to be small compared to factors such as fuel loading (Quraishi, 1989).

In Christchurch the burning of wet wood (>25%) is not permitted. However, enforcement of this regulation is difficult and it is expected that the use of wet wood is not uncommon. If emission factors used in the Christchurch emission inventory and subsequently in this analysis do not

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$^{21}$ The Regional Energy Survey (CRC, 1996) estimates annual fuel use based on annual sales for the domestic market. Because it is done on an annual basis comparisons with the emission inventory require the extrapolation of daily fuel use to annual fuel use.
adequately account for the effect of burning wet wood the overall contribution to total PM$_{10}$ from the domestic sector may be greater than 90%. Further investigations into measures to reduce the use of wet wood in domestic burners would be useful.

Overall the relative proportions of PM$_{10}$ emission arising from each sector is generally consistent with a previous estimate made by Brady & Pullen (1985) based on fuel use data.

**Validation of transport emissions estimates**

Estimated PM$_{10}$ emissions from the transport sector were compared to similar estimations made by Wood (1997b) using the methodology detailed in Watkins (1991). This estimated PM$_{10}$ emissions of approximately 1.4 tonnes per day, compared to approximately 1.1 tonnes per day estimated in the emission inventory. This indicates variations of about 25% for the two investigations.

**Validation of industrial emission estimates**

Industrial emission estimates from the emission inventory were compared to a study done by Coal Research Limited (Damiano and Campbell, 1997) and to a survey of emissions in the Hornby area (Bluett, 1997). The PM$_{10}$ emissions estimated in the Coal Research study were approximately one third lower than those estimated in the emission inventory. The primary cause of the differences is the emission factors, which are lower in the Coal Research investigation. A closer correlation was observed when compared with SO$_{2}$ emissions estimated by Bluett (1997) for the Hornby area. The latter study used the same emissions factors but a more detailed methodology for determining fuel use. The fuel use estimates compare reasonably well with Bluett’s (1997) research.

The differences observed in the comparative studies are considered and used to estimate the standard error for each variable in an error analysis detailed in Chapter Six.

**4.2.2 The relationship between estimated PM$_{10}$ emissions and measured concentrations in the air shed**

Scientists at the National Institute of Water and Atmospheric Research (NIWA) conducted an investigation into the relationship between estimated emissions and measured concentrations. A box model$^{22}$, was established to quantify the effect of meteorological conditions on emissions at different times of the day. This model indicated that emissions are related to 24-hour concentrations by the equation detailed in appendix 10.

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$^{22}$ Model that treats the air shed as a box in which contaminants are evenly dispersed and move into and out of at rates determined by meteorological conditions.
Chapter four - Development of a framework for assessing methods to reduce PM$_{10}$

Validation of the model against observed 24-hour average concentrations using actual meteorological measurements for those days gives ratios of observed PM$_{10}$ to modelled PM$_{10}$ between 0.68 and 1.51. The mean and median ratios were both 0.99 and the standard deviation 0.19. Although modelled and observed 24-hour average concentrations correspond well; modelled results for 1-hour concentrations to measured hourly concentrations do not correlate as well.

The model indicates that a linear relationship exists between estimated emissions and measured concentrations for 24-hour average concentrations. This implies that a specified reduction in emissions (e.g., 74%) will most likely result in the same percentage reduction in PM$_{10}$ concentrations. The derivation of the equation and further details regarding the assumptions and limitations of the model are detailed in CRC Report Number U97/67 (Gimson & Fisher, 1997).

Variations in emissions for different periods of the day and hourly average PM$_{10}$ concentrations for a day of typical worst case meteorological conditions are illustrated in figure 4.2. Because information on daily variations in emissions is limited to four periods of 4-8 hours (table 4.1), hourly emissions could be estimated by dividing the total over the period by the number of hours (figure 4.2a). However, the actual distribution of the emissions is more likely to be as illustrated in figure 4.2b, which shows an estimated distribution of the same total emissions.

![Figures 4.2a (averaged) & 4.2b (estimated): Variations in PM$_{10}$ emissions over a 24-hour period relative to changes in PM$_{10}$ concentration](image)

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23 Comparisons were made with days of the 17 highest 24 hour average PM$_{10}$ concentration measured at the St Albans monitoring site. Only the highest concentrations were compared as the model was developed based on the worst case meteorological conditions.

24 The average PM$_{10}$ concentration for each hour of the day for all days when the 24 hour average concentration exceeded 50 $\mu$g/m$^3$ from 1988-1996.
Low wind speeds coupled with temperature inversion conditions restrict the dispersion of contaminants released into the air shed, resulting in the potential for high pollution episodes. On a typical high pollution night, these conditions occur from around 4 p.m. and can last up until 10 a.m. the following morning. Emissions during this period will have a greater impact on the 24-hour average concentration than emissions that occur during the daytime, when dispersion is typically not as restricted due to higher wind speeds and lack of inversion. Consequently, the time of day in which emissions occur is important in assessing the relative contribution of different sources to 24-hour average PM$_{10}$ concentrations.

Emissions inventory data for a typical winter’s night indicate relative contributions to PM$_{10}$ emissions of 82%, 10% and 8% from domestic heating, transport and industry, respectively. The relative contributions of these sectors to PM$_{10}$ concentrations will be dependent on variations in emissions relative to meteorological conditions. The airshed box model has been used to calculate the relative contribution to PM$_{10}$ concentrations of different sectors. Figure 4.3 illustrates the relative contribution to PM$_{10}$ concentrations from the different sectors for the Christchurch area based on the above modelling.

![Figure 4.3: Relative contribution to 24-hour average PM$_{10}$ concentrations of sectors for Christchurch for a typical winter’s night (based on 1996 emissions data)](image)

Figure 4.3 indicates that the domestic heating sector contributes 90% of the 24-hour average PM$_{10}$ concentrations on a typical winter’s night.
Chapter four - Development of a framework for assessing methods to reduce \( PM_{10} \)

4.3 Projected changes in \( PM_{10} \) emissions with time

The following sections attempt to quantify changes in emissions of \( PM_{10} \) from the different sources in the future.

4.3.1 Domestic heating

Existing controls on home heating methods in the Christchurch Clean Air Zones prohibit the installation of open fires and restrict solid fuel burner installations to those meeting specific emission criteria. As a result of these regulations and other factors, such as changes in availability, cost of alternative methods of home heating and lifestyle factors, significant changes in home heating patterns and hence emissions of \( PM_{10} \) are predicted to occur over the next 25 years.

Figure 4.4 (Ball, 1997) illustrates trends in principal means of home heating for the years 1961-1996 and projections for the years 1997-2021. The historical trends illustrated are based largely on census data obtained from Statistics New Zealand and an energy use survey by Moody (1987). Later figures are based on the 1996 emissions inventory (CRC, 1997a) and a 1997 public opinion survey on peoples’ attitudes towards air quality (Lamb, 1997).

![Figure 4.4: Historical and predicted principal methods of home heating for Christchurch households (from Ball, 1997)](image-url)
Figure 4.4 shows a predicted decline in the use of open fires and electricity. Gas and oil use for domestic home heating is predicted to increase. The use of solid fuel burners is estimated to continue to increase until approximately 2001.

Future changes in home heating methods, particularly regarding open fires and older wood burners are discussed in more detail in Wood (1997a). Wood (1997a) considers the following issues:

- The historical decline in open fires;
- The growth in sales and installation of solid fuel burners;
- The proportion of new burners replacing the existing stock of burners;
- The proportion of burners going into new houses;
- The proportion of burners going into existing houses to replace open fires;
- The number of houses with open fires which are being demolished or relocated;
- The proportion of solid fuel burners going into houses which previously had no burner or open fire.

While the validation of projections such as those made by Wood (1997a) is difficult, these predictions show the same trends as estimated in figure 4.4 (Ball, 1997). Wood’s (1997a) analysis was undertaken using all available information including consulting with experts in the appropriate fields and taking into consideration factors such as predicted increases in electricity prices. Notwithstanding this, establishing numbers relating to future home heating methods must still include a large degree of uncertainty.

Wood’s (1997a) analysis includes the assumption that 85% of solid fuel burners requiring replacement due to age are replaced with new wood burners and that 65% of the open fires that are replaced by other methods of home heating each year, will be replaced with a wood burner. These predictions were made prior to the introduction of a test criteria of 1.5 grams per kilogram (g/kg) for new solid fuel burners. The introduction of a 1.5 g/kg burner is estimated to result in temporary price increases in solid fuel burners of about $200-$300 (pers. comm., Ball, 1997). This may impact on the predictions in Wood (1997) regarding future household home heating choices. The following estimates of the potential effect of this increase were made in

\[^{25}\text{This equates to approximately 60\% of the open fires which are replaced with other forms of heating which chose solid fuel. A proportion are not replaced due to the demolition of older houses.}\]
consultation with Ball (CRC, Energy and Planner) and McChesney (CRC, Manager Energy and Transport) in 1997

- A reduction of 5% in households choosing solid fuel methods to replace expired wood burners, i.e., 80% replace solid fuel methods with new burners;
- A decrease of 5% in the number of open fires being replaced with wood burners, i.e., 60% replace open fires with solid fuel burners;
- A decrease of 10% in the number of households with existing, non-solid fuel methods, changing to solid fuel burning.

Projections in emissions from domestic home heating (figure 4.5) were estimated by applying the emission and fuel use factors to the projected changes in domestic home heating methods (as detailed in appendix 8). Emission factors, based on year of installation, were assumed to apply for the whole life of an appliance and the projected changes in home heating methods were based on the conclusions in Wood (1997a).

![Graph showing projected PM10 emissions from domestic heating](image)

**Figure 4.5: Projected PM10 emissions from domestic heating**

Because of present legislation prohibiting the installation of open fires in Christchurch city and criteria adopted in 1997 by the CRC applying to the approval of new solid fuel burning appliances, a decrease in current emission levels is predicted as older (more polluting) appliances and open fires are replaced. This trend is shown in figure 4.5.
4.3.2 Transport

Wood (1997b) investigated future PM$_{10}$ emissions from the transport sector (figure 4.6). Future emission estimates were based on population and household predictions made by Statistics New Zealand, changes in economic fortunes based on National Bank Economic Forecasts and urban development aspirations included in the Christchurch City Plan. An assessment of increases in emissions was made by pairing household and population predictions with a vehicle allocation model and employment data to estimate present and future vehicle trips made. This included the impact of tailpipe emissions resulting from changes in traffic flow associated with projected increases in traffic congestion.

As discussed in section 4.2.1 emission levels at 1996 were validated against estimates made for the emission inventory, which used a different methodology.

![Figure 4.6: Estimated increases in PM$_{10}$ emissions from the transport sector (from Wood, 1997)](image)

Figure 4.6 shows that PM$_{10}$ emissions from the transport sector are predicted to increase by 0.3 tonnes per day by 2021.

4.3.3 Industry

Barber (1996) investigated trends and projections for the quantity and types of fuel used for industrial and trade activities in Christchurch. This analysis indicated a 50% increase in energy requirements for this sector by the year 2021. Projections of the energy requirements in terms of fuel type were not attempted. For the purpose of this analysis a 50% increase in emissions of PM$_{10}$ from the industrial sector has been assumed (figure 4.7). Although the relative contribution of this increase is minor compared with current emissions from the domestic sector.
it becomes significant if emissions from other sectors are reduced so that overall emissions are reduced to acceptable levels.

![Graph showing estimated increases in PM$_{10}$ emissions from the industrial sector.](image)

**Figure 4.6: Estimated increases in PM$_{10}$ emissions from the industrial sector**

### 4.4 Methodology for assessing the effectiveness of strategies to reduce PM$_{10}$ concentrations

A methodology for assessing reduction in PM$_{10}$ concentrations needs to consider:

- Changes in emission with time;
- The effect of variations in emissions from each source over a 24-hour period relative to meteorological conditions;
- The reduction required in PM$_{10}$ concentrations.

Section 4.3 estimates PM$_{10}$ emissions for the years 1996-2021 for the domestic heating, transport and industrial sectors. Projected emissions from each sector are weighted for the effect of meteorological conditions on variations in emissions over a 24-hour period. The details of this weighting system are contained in appendix 6. Total emissions are then calculated for each year and expressed as a percentage of the 1996 total (figure 4.8). A dashed line (at 26%) indicates the percentage reduction required (74%) to meet the air quality target.
Figure 4.8 shows predicted PM$_{10}$ emissions in the future decrease as a result of an estimated decline in home heating emissions. As a result of a decline in the domestic heating sector and an increase in emissions from other sectors the proportion of the total coming from this sector in the future is estimated to decrease.

### 4.5 Allocating reductions in PM$_{10}$ concentrations between sectors

Sections 4.1 and 4.2\textsuperscript{26} indicate that reduction in 24-hour average PM$_{10}$ concentrations required to meet a target of 50 $\mu$g/m$^3$ is 74% of 1996 emission levels.

A number of possible scenarios exist for allocating the required reduction in PM$_{10}$ emissions between the main contributing sectors. These scenarios are summarised as follows:

1. Equal percentage reduction in emissions from each sector;
2. Reduction by each sector in proportion to its relative contribution to PM$_{10}$ concentrations;
3. Reduction by the domestic sector only;
4. Reductions using Best Practicable Option (BPO) or Best Available Control Technology (BACT);
5. Reductions based on energy efficiency only e.g., decrease in energy requirements due to increased insulation;

\textsuperscript{26} This section indicates that the reduction required in PM$_{10}$ emissions is likely to be the same percentage as that required to reduce PM$_{10}$ concentrations to acceptable levels.
6. Percentage reductions based on location and contribution to ambient concentrations.

These scenarios have been examined for the Canterbury Regional Council (Brady & Keller, 1997) in terms of:

- The ability to achieve required reductions;
- The costs associated with achieving the reduction;
- The technical and legal feasibility;
- Consistency with CRC and City Council policy;
- The practicality associated with enforcing the measures;
- Equity issues;
- Social impacts.

On the basis of these criteria, the most appropriate scenarios were judged to be reductions proportional to relative contribution and reductions by the domestic sector only (Brady & Keller, 1997). Brady and Keller (1997) analysis projected fuel use and in the absence of an emissions model, used the relationship between concentrations and emissions to account for the effects of meteorology at different times of the day.

The assessment of the reduction required from each sector under the two allocation scenarios has been recalculated, as a part of this analysis, using the emissions inventory data and the emissions model. This calculation shows that to reduce emissions from the domestic sector alone, a reduction in domestic PM$_{10}$ emissions of 82% would be required with no reduction required from transport or industry$^{27}$. Reducing emissions in proportion to their relative contribution would also require an 82% reduction from domestic home heating, with a 4% reduction from transport and a 5% reduction in PM$_{10}$ emissions from industry$^{28}$.

This indicates less than 1% difference in the reduction that would be required from domestic home heating under the two scenarios. This is because with current emission levels, a 4 or 5% reduction from industry or transport equates to less than 1% reduction in overall emissions. However, the relative contribution of emissions from these sectors is predicted to increase as emissions from the domestic sector decrease. If management measures target domestic home heating alone the reduction required is likely to be greater than 82% with time. For example, an 87% reduction would be needed by 2025, due to projected additional emissions from industry

$^{27}$ This compares with a 84% reduction in Brady & Keller. (1997).
$^{28}$ This compares with a 81% reduction in Brady & Keller. (1997).
and transport. Further investigations are being conducted into the cost and feasibility of obtaining reductions in emissions from the transport and industrial sectors.

One option to restrict the predicted increases in industrial emissions would be to not allow any industrial growth in the City. An alternative approach would be to introduce management options to reduce emissions from existing processes and require new processes to meet restricted emission criteria. An option which restricts TSP emissions to 250 mg m\(^{-3}\) has been suggested by Coal Research Limited (CRL, 1997) for boilers greater than 1 MW.

Coal Research Limited (1997) indicate that this option would result in a 23% reduction in PM\(_{10}\) emissions from the industrial sector. If these controls were introduced in the year 2001, by 2021 emissions in the industrial sector would have increased by 15% over 1996 levels. This compares with a predicted 50% increase in emissions over the same period in the absence of controls.

At this stage, Council's ability to control emissions from motor vehicles is limited. However, the Ministry of Transport (MOT) is at present investigating vehicle emissions control on a national basis. Until these investigations are completed and decisions are made by Central Government it is very difficult to quantify the potential effects of management options.

Because of the small effect of any reduction in industry and motor vehicles on PM\(_{10}\) concentrations in ambient air, further sections consider management options to achieve the required reduction in emissions from the domestic sector.

### 4.6 Summary

This chapter estimates the future PM\(_{10}\) concentrations in Christchurch relative to current levels and desired levels. This assessment is the basis for the model detailed in Chapter Five.

This chapter also identifies domestic heating as the primary source of PM\(_{10}\) emissions and the sector that management options should target. Management options for this domestic heating are discussed further in chapter six.
CHAPTER FIVE: DEVELOPMENT AND VALIDATION OF THE MODEL

This chapter describes the model used to assess the effectiveness of the management options for domestic heating. This model has been developed for this thesis and has not been used to assess emissions outside of Christchurch. The model attempts to assess the effectiveness of different management options for domestic heating, taking into account all possible influencing factors such as installation standards for solid fuel burners and fuel switching.

Development of the model is described in section 5.1 and validation in section 5.2. Chapter four introduced and described the input elements of the model including current emissions, projected emissions and weighting factors. Two other input components of the model are the implementation of management options and the impact of the options on installations of new solid fuel burners. All of these components are integrated to provide a fully developed model that will be used to assess the effectiveness of different management options in reducing PM$_{10}$ concentrations in Christchurch.

5.1 Development of the model

Future PM$_{10}$ emissions for a status quo option are calculated by combining estimated PM$_{10}$ emissions for the years 1996 to 2021 from the domestic heating, transport and industrial sectors, as described in chapter four. Chapter four also identifies the domestic heating sectors as the primary target for management measures. Consequently the PM$_{10}$ emissions from the transport and industrial sectors predicted for the status quo option change as predicted for all management options considered in chapter six.

In contrast estimated emissions from domestic heating will vary for each management option. Estimated PM$_{10}$ emissions on a typical winters day$^{29}$ are calculated by multiplying the number of households with a particular home heating method by the emission factor (g/kg) and fuel use factor (kg) for that method (see appendix five), then adding together the emissions from each heating method. Total predicted PM$_{10}$ emissions vary from year to year depending on the number of households using each of the different home heating methods.

For simplification, the factors influencing the number of households using different heating methods at any year are defined by these four stages in order:

$^{29}$ Calculations are for typical wintertime emissions for a 24 hour period because guidelines are based on a 24 hour average.
1. The number of households using a particular method of domestic heating in 1995;
2. Projected changes in the number of households using particular methods of domestic heating;
3. The effect of the implementation of each management option for domestic heating;
4. The effect of stage 3 on the installation of new solid fuel burners i.e., as replacement methods.

Stages 1) and 2) are described in chapter four. The effect of introducing different management options (stage 3) is assessed by manipulating the numbers of households using specific methods. For example, if the use of open fires is prohibited it reduces the number of households using open fires by 100%\(^30\). The stage 4 component is then introduced by estimating the proportion of the households that were using open fires that convert to solid fuel burners. Total emissions are then reduced by the total amount of emissions from open fires (stage 3) minus any additional emissions occurring as a result of the emissions due to conversions (stage 4).

A simplification of the integration of all the components of the model is shown in figure 5.1

The effectiveness of each management option in reducing emissions is assessed by comparing total emissions for each management option, expressed as a percentage of 1996 emission levels, to an air quality target level set at 26%\(^31\) of the 1996 emission level (as detailed in section 2). Prior to this comparison emissions from each sector are weighted to account for variations in the time of day emissions from each sector occur relative to meteorological conditions (appendix 3).

### 5.2 Key Assumptions

Key assumptions associated with the model are as follows.

- The average amount of fuel used on appliances will remain constant.
- There will be no variation in the time of day that emissions from different sources occur.
- There will be no decline in the performance of solid fuel burners with time.
- New installations of wood burners will be a mixture of burners meeting different emission criteria. Estimates of the number installed that meet each criterion is based on the relative proportion of wood burners meeting each emission criteria.

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\(^{30}\) Assuming 100% compliance.

\(^{31}\) A 74% reduction in 1996 emissions is required which equates to an air quality target set at 26% of 1996 emissions.
Chapter five - Development of and validation of the model

- That the proportion of households that burn wood versus coal will not change (other than specified in management options).
- 100% compliance with proposed management options.

Assumptions relating to the application of specific scenarios are detailed in chapter 6.
Contribution of domestic heating to 24-hour average PM$_{10}$ concentrations

Contribution of transport to 24-hour average PM$_{10}$ concentrations

Contribution of industry to 24-hour average PM$_{10}$ concentrations

Stage 1 & 2

No. of households using different solid fuel domestic heating methods

Effect of implementation of management option on installation of new solid fuel burners

Stage 3

Total PM$_{10}$ emissions from all domestic solid fuel burning appliances

Emission factor & Average quantity of fuel 24-hours

Weighting factor

Contribution of domestic heating to 24-hour average PM$_{10}$ concentrations

Estimated PM$_{10}$ emissions from transport sector

Weighting factor

Contribution of transport to 24-hour average PM$_{10}$ concentrations

Estimated PM$_{10}$ emissions from industrial sector

Weighting factor

Contribution of industry to 24-hour average PM$_{10}$ concentrations

Percentage of 1996 emission levels

Divided by the value for year = 1996

Total emissions (weighted)

Figure 5.1: Diagram illustrating the primary components and integration of the model
5.3 Model formulae

Equation E5.1 is the formula used to describe the model.

\[
E_n = \frac{\eta f g^2 + l f s g^3 + m f s g^2 + o f s g^2 + p f s g^2 + \eta f g^2 + q f s g^2 + \eta f g^2}{(k f g^2 + l g^2 + m f s g^2 + o f s g^2 + p f s g^2 + \eta f g^2 + q f s g^2 + \eta f g^2)}
\]

*Equation E5.1: Description of the model*

where \( E_n \) = emissions in year \( n \) as a percentage of 1996 emission levels, for \( n = 1997 \) to 2021 and \( E_{1996} = 100\% \).

Table 5.1 details the variables in equation E5.1 for a management option described as option 7 in chapter six. For other management options the formulae and variables described in stages 3 and 4 will vary in accordance with assumptions described in the text in chapter six.

**Table 5.1 Variables and additional formulae associated with equation E5.1 for the preferred management option**

<table>
<thead>
<tr>
<th>Stage one: Establishing the number of households using different heating methods in 1996</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>( k ) = no. of houses using oil in 1995</td>
<td>Emission Inventory plus projections data for year 1 (Wood, 1997)</td>
</tr>
<tr>
<td>( l_{1996} ) = ( b - \alpha_{1996} ) where ( b ) = no. of houses with open fires in 1995</td>
<td></td>
</tr>
<tr>
<td>( m_{1996} ) = ( \theta - \omega_{1996} ) where ( \theta ) = no. of houses with incinerators in 1995</td>
<td></td>
</tr>
<tr>
<td>( r_{1996} ) = ( \rho - x_{1996} ) where ( \rho ) = no. of houses with potbellies in 1995</td>
<td></td>
</tr>
<tr>
<td>( o_{1996} ) = ( \sigma - \omega_{1996} ) where ( \sigma ) = no. of houses with coal burners in 1995</td>
<td></td>
</tr>
<tr>
<td>( p_{1996} ) = ( \tau - \omega_{1996} ) where ( \tau ) = no. of houses with pre 1989 wood burners in 1995</td>
<td></td>
</tr>
<tr>
<td>( q_{n} ) = no. of houses with 1989-1992 wood burners in 1995 for ( n = 1996 ) to 2005</td>
<td></td>
</tr>
<tr>
<td>( q_{2006} ) = 4200 (from the status quo option ( q_{2005} (a - (m_{2005} + r_{2005} + o_{2005})) ) ( - (m_{2006} + r_{2006} + o_{2006})) ))</td>
<td></td>
</tr>
<tr>
<td>( t_{1996} ) = no. of houses with post 1992 wood burners in 1995</td>
<td></td>
</tr>
</tbody>
</table>

**Stage 2: Projected changes in households using solid fuel burners**

| \( l_{n} \) = \( l_{(n-1)} - \alpha_{n} \) for \( n = 1997 \) to 2001, \( l_{n} = 0 \) for \( n = 2002 \) to 2021 | |
| \( m_{n} \) = \( m_{(n-1)} - \omega_{n} \) for \( n = 1997 \) to 2002, \( m_{n} = 0 \) for \( n = 2003 \) to 2021 | |
| \( r_{n} \) = \( r_{(n-1)} - x_{a} \) for \( n = 1997 \) to 2002, \( r_{n} = 0 \) for \( n = 2003 \) to 2021 | |
| \( o_{n} \) = \( o_{(n-1)} - \omega_{a} \) for \( n = 1997-2012, o_{n} = 0 \) for \( n = 2003 \) to 2021 | |
| \( p_{n} \) = \( p_{(n-1)} \times w_{a} \) for \( n = 1997 \) to 2002, \( p_{n} = 0 \) for \( n = 2003 \) to 2021 | |

\[32\] No increases in the number of households using oil were included in the analysis. Because of the low PM\(_{10}\) emissions from oil burners increases in the use of oil are not significant.
Chapter five – Development and validation of the model

Stage 2: Increases in new solid fuel burners (apportioning the increase among burners meeting different test criteria).

| φn | the no. of households installing wood burners at year n |
| ζn | the no. households installing wood burners to replace older burners |
| Nb(subset of φn) | (the number of households with wood burners in the ≥3.0 g/kg ≤ 5.5 g/kg category) where βn= % of Clean Air Approved wood burners with emission test results ≥3.0 g/kg ≤ 5.5 g/kg at year n33 |
| τ1997=+φnβn | (the number of households with wood burners in the ≥3.0 g/kg ≥ 1.5 g/kg category) where χn= % of Clean Air Approved wood burners with emission test results ≤3.0 g/kg ≥ 1.5 g/kg at year n |
| τ2001=+φnδn | (the number of households with wood burners in the ≤1.5 g/kg or equivalent category) where δn= % of Clean Air Approved wood burners with emission test results ≤1.5 g/kg or equivalent34 at year n |

<table>
<thead>
<tr>
<th>Source</th>
<th>Table 5.2</th>
</tr>
</thead>
</table>

Table 5.2

Stage 2: Decreases in older burners (Apportioning the decrease between older appliances)

| an | the no. of wood burners being removed at year n |
| u | % of pre 1989 solid fuel burners which are incinerators35 |
| x | % of pre 1989 solid fuel burners which are potbellies |
| w | % of pre 1989 solid fuel burners which are wood burners |
| z | % of pre 1989 solid fuel burners which are coal burners |

<table>
<thead>
<tr>
<th>Source</th>
<th>Emission inventory</th>
</tr>
</thead>
</table>

Stage 3: Implementation of management options

| αn | the number of open fires no longer used for year n for n= 1996 and 1997 |
| α1998 | 0.111997 for years n= 1998 to 2001 (accelerated decline in open fires as detailed in chapter 6) |
| α2001 | l2000 (reduces the no. of open fires to zero as per management option) |

33 the proportion of burners in each category that are installed in the years 1997-2001. Proportions are described in table 3.1. They are based on the number of approved burners in each category from 1997 to 2001 using current expiry dates and predicted increases in the number of wood burners meeting the 1.5 g/kg criteria. Assumes the number of installations in each category will be proportional to the number of available models in each category. 

34 In establishing the criteria Council deemed burners with an average emission test result of 2 g/kg or less combined with a low emission test result of 1.5 g/kg or less as meeting the required emission level. 

35 Used to allocate the decline in the use of pre 1989 solid fuel burners between the different types of pre 1989 appliances. Coal burners, potbellies and incinerators decline at a slower rate, on average, than wood burners.
Chapter five – Development and validation of the model

### Stage 4: Impact of management option on installation of new solid fuel burners

<table>
<thead>
<tr>
<th>Year</th>
<th>Formula</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>1998-2001</td>
<td>( \psi_n = \phi_n + 0.8\alpha_n )</td>
<td>As per assumptions described in table 6.2</td>
</tr>
<tr>
<td>2002</td>
<td>( \psi_{2002} = \phi_n + 0.2\alpha_n )</td>
<td></td>
</tr>
<tr>
<td>2003</td>
<td>( \psi_{2003} = \phi_n - \zeta_{2003} + 0.7(m_{2001} + r_{2001} + o_{2001} + p_{2001}) )</td>
<td></td>
</tr>
<tr>
<td>2004-2005</td>
<td>( \psi_n = \phi_n - \zeta_n )</td>
<td></td>
</tr>
<tr>
<td>2006</td>
<td>( \psi_{2006} = \phi_n + (a_{2000} - (q_{2005} - q_{2006})) )</td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>( \psi_{2007} = \phi_n - \zeta_{2007} + 0.7q_{2006} )</td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>( \psi_{2008} = \phi_n - \zeta_{2008} )</td>
<td></td>
</tr>
<tr>
<td>2009-2014</td>
<td>( \psi_n = \phi_n )</td>
<td></td>
</tr>
<tr>
<td>2015</td>
<td>( \psi_{2015} = \phi_n + a_n - h_n )</td>
<td></td>
</tr>
<tr>
<td>2016-2021</td>
<td>( \psi_n = \phi_n + a_n )</td>
<td></td>
</tr>
</tbody>
</table>

### Stage 4 + Stage 2: New installations of solid fuel burners

<table>
<thead>
<tr>
<th>Year</th>
<th>Formula</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>1996-2008</td>
<td>( \gamma_n = \gamma_{(n-1)} + \psi_n \beta_n )</td>
<td>Emission inventory (also detailed in appendix 5)</td>
</tr>
<tr>
<td>2009-2013</td>
<td>( \gamma_n = \gamma_{(n-1)} - (a_n - q_{(n-1)}) )</td>
<td></td>
</tr>
<tr>
<td>2014-2021</td>
<td>( \gamma_{2015} = \gamma_{2014} - (a_{2015} - q_{2014}), \gamma_n = 0 )</td>
<td></td>
</tr>
<tr>
<td>2014-2021</td>
<td>( \eta_n = \eta_{(n-1)} + \psi_n \chi_n )</td>
<td></td>
</tr>
<tr>
<td>2015-2021</td>
<td>( \eta_{2015} = \eta_{2014} - (a_{2015} - q_{2014}), \eta_n = 0 )</td>
<td></td>
</tr>
<tr>
<td>2014-2021</td>
<td>( \eta_n = \eta_{(n-1)} + \psi_n \chi_n )</td>
<td></td>
</tr>
<tr>
<td>2014-2021</td>
<td>( \eta_{2015} = \eta_{2014} - (a_{2015} - q_{2014}), \eta_n = 0 )</td>
<td></td>
</tr>
</tbody>
</table>

### Determining the PM\(_{10}\) emissions from the different appliances on a typical winter’s night

- \( f_i \): emission factors for the appliances \( i \)
- \( g_i \): average amount of fuel used on a typical winter’s day for appliances \( i \)

<table>
<thead>
<tr>
<th>Appliance</th>
<th>Emission Factors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wood burners</td>
<td>1.5 g/kg</td>
</tr>
<tr>
<td>Oil burners</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>Open fires - wood</td>
<td>1.5 g/kg</td>
</tr>
<tr>
<td>Potbelly - wood</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>Incinerators - wood</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>Coal burners - wood</td>
<td>1.5 g/kg</td>
</tr>
<tr>
<td>Pre 1989 wood burners</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>1989-1992 wood burners</td>
<td>1.5 g/kg</td>
</tr>
<tr>
<td>Post 1992 wood burners</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>Open fires - coal</td>
<td>1.5 g/kg</td>
</tr>
<tr>
<td>Potbelly - coal</td>
<td>3.0 g/kg</td>
</tr>
<tr>
<td>Coal burner - coal</td>
<td>1.5 g/kg</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Type</th>
<th>Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>Incinerator</td>
<td>( v_1 )</td>
</tr>
<tr>
<td>Potbelly</td>
<td>( v_2 )</td>
</tr>
<tr>
<td>Coal range</td>
<td>( v_3 )</td>
</tr>
<tr>
<td>Households</td>
<td>( o_1 )</td>
</tr>
<tr>
<td>Incinerator</td>
<td>( o_2 )</td>
</tr>
<tr>
<td>Potbelly</td>
<td>( o_3 )</td>
</tr>
<tr>
<td>Coal range</td>
<td>( o_4 )</td>
</tr>
</tbody>
</table>
Chapter five – Development and validation of the model

\[ \pi = \% \text{ of houses that use an open fire that burn wood} \]
\[ s = \% \text{ of houses that use an open fires that burn coal} \]
\[ \mu_1 = f_3g_3V_1 + f_1g_{13}O_1 \text{ (emissions from an incinerator on a typical winters night)} \]
\[ \mu_2 = f_3g_5V_2 + f_1g_{14}O_2 \text{ (emissions from a potbelly on a typical winters night)} \]
\[ \mu_3 = f_3g_6V_3 + f_1g_{15}O_3 \text{ (emissions from coal burner on a typical winters night)} \]

<table>
<thead>
<tr>
<th>Industrial and transport emissions</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>( i_n ) = projected industrial emissions at year n</td>
<td>As described in chapter 4</td>
</tr>
<tr>
<td>( j_n ) = projected transport emissions at year n</td>
<td>Emission inventory</td>
</tr>
<tr>
<td>( i ) = industrial PM_{10} emissions in 1995</td>
<td></td>
</tr>
<tr>
<td>( j ) = transport PM_{10} emissions in 1995</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Weighting factors</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>( d_j ) = weighting factors for the relationship between emissions and concentrations</td>
<td>Detailed in Appendix 6</td>
</tr>
<tr>
<td>where ( j = 1 ) = domestic heating, ( 2 ) = industry, ( 3 ) = transport (appendix 6)</td>
<td></td>
</tr>
</tbody>
</table>

Table 5.2 contains estimates of the proportion of solid fuel burners meeting the different emission test criteria. These estimates are based on the number of appliances that meet the different criteria at present and data on the expiry of the approvals for installation. This data is used in the formula described in table 5.2, stage 2.

**Table 5.2: The proportion of authorised appliances meeting each emission category**

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>≥3.0g/kg ≤ 5.5g/kg</td>
<td>0.67</td>
<td>0.54</td>
<td>0.45</td>
<td>0.38</td>
<td>0.18</td>
<td>0</td>
</tr>
<tr>
<td>≤3.0g/kg ≥ 1.5g/kg</td>
<td>0.24</td>
<td>0.34</td>
<td>0.34</td>
<td>0.19</td>
<td>0.09</td>
<td>0</td>
</tr>
<tr>
<td>≤1.5g/kg or equivalent</td>
<td>0.09</td>
<td>0.11</td>
<td>0.21</td>
<td>0.43</td>
<td>0.73</td>
<td>1</td>
</tr>
</tbody>
</table>

**5.4 Validation of the model**

This section investigates the stability of the model derived to assess the effectiveness of different management options for domestic heating. The validation of several of the model input parameters are discussed in chapter four. Because of the uncertainties identified in that chapter an estimate of the error associated with the use of the model is considered in chapter six. This section considers only the stability of the model itself i.e., is it logical and does it do sensible things.
The robustness of the model itself can be investigated by considering the output for a selection of effects that can be easily predicted. For example, the emission inventory shows that open fires contribute 48% of the PM$_{10}$ emissions from domestic heating. Therefore if open fires are banned domestic heating emissions could decrease by 48% minus the increase associated with a proportion of these households converting to solid fuel burners. The domestic heating sector is responsible for 90% of the concentrations. Therefore a 48% reduction in domestic emissions equates to an overall reduction of 43% of the total PM$_{10}$. If we assume that all households using open fires convert to electricity in 1998 then the model should show a reduction of at least$^{36}$ 43% of the 1996 emission levels (figure 5.1).

Figure 5.1: Modelled emissions if open fires are prohibited in 1998 and all users convert to non-solid fuel alternatives.

Figure 5.1 illustrates the effect of prohibiting the use of open fires in 1998, assuming that all users convert to non-polluting alternatives. As expected the reduction shown is slightly greater than 43% (at 47%) due to the additional reductions associated with the phase out of older burners during 1996, 1997 and 1998.

This test shows how the model responds to the implementation of a management option (stage 2) but does not consider how the model responds if additional burners are installed to replace open fires (stage 4). If we assume instead that all users of open fires convert to new solid fuel burners then the reduction achieved will be less. The year in which open fires are prohibited

$^{36}$ The reduction should in fact be slightly greater than 50% due to the replacement of older wood
also affects the predicted emissions. For example, if open fires were to be prohibited in 1998 the reduction observed should be less than if they were prohibited in 2003. This is because of changes in the emission standards for solid fuel burners during this period. At present solid fuel burners meeting a criteria of 5.5 g/kg can be installed until September 2001. After this time only solid fuel burners meeting the 1.5 g/kg criteria may be installed\(^\text{37}\). If a ban on open fires were imposed in 1998 the fires could be replaced with the higher emitting 5.5 g/kg solid fuel burners and, therefore, the overall reduction achieved would be less.

Small differences are also expected in the estimated concentrations from 2016 onwards. This is because the predicted decrease in open fires from 1998-2002 prior to the regulatory intervention in 2003 are assumed to have a higher replacement rate (60%)(section 4.3.1). This is lower than the replacement rate of 100% assumed for the prohibition. Hence the total number of solid fuel burners used by 2016 is less for the prohibition of open fires in the 2003 option. This scenario is illustrated in figure 5.2.

Figure 5.2: Modelled emissions if open fires are prohibited in 1998 and 2003 assuming all convert to solid fuel burners

Figure 5.2 shows that the model is sensitive and responds as expected to factors such as changes in the test criteria for solid fuel burners.
5.5 Validation of the input parameters

Validation of many of the input parameters is discussed in chapter four. The two additional model input parameters not discussed in chapter four are the implementation of management options (stage 3) and the effect of these on the installation of solid fuel burners (stage 4).

The primary uncertainty surrounding the implementation of management options is the assumption of 100% compliance. Enforcement issues are beyond the scope of this investigation and have not been formally addressed by the Canterbury Regional Council at this stage. The extent to which households affected by the implementation of management options convert to new solid fuel burners is uncertain. For each management option this proportion has been estimated (see table 6.2).

It is apparent that there is much subjectivity surrounding the values of some model-input parameters, particularly those influencing the number of households using different home heating methods. Emission and fuel use factors, the reduction required and weighting factors are other subjective factors. Given the uncertainties it is likely that the uncertainties associated with the calculations will be large. However, these extrapolations and assumptions are necessary to make the required assessment. The estimated standard error associated with the analysis is considered further in section 6.2.

5.6 Summary

This chapter describes the model used to assess the effectiveness of management options to reduce PM$_{10}$ concentrations in Christchurch. The validity of the model was tested by observing the output for the effect of banning open fires for a variety of time frames and assumptions. The model output is consistent with the predictions in all cases. However, examination of the model input parameters indicates that these is uncertainty associated with model input parameters. This is examined further in chapter six.
CHAPTER SIX: ASSESSING THE EFFECTIVENESS OF DIFFERENT MANAGEMENT OPTIONS

This chapter investigates the effectiveness of management options aimed at reducing PM$_{10}$ emissions by applying the model to a series of management options for domestic heating. Results are presented graphically and the effectiveness of the option in terms of the likelihood of meeting the air quality target is discussed.

Uncertainties associated with the analysis are considered in section 6.2 and a methodology for assessing the effect of management options on exceedences is detailed in section 6.3. The effect of management options on other contaminants is considered briefly in section 6.4. This discussion is limited to the coincidental reduction in emissions associated with management options aimed at reducing PM$_{10}$ concentrations.

6.1 Management Options

A number of management options for reducing emissions from solid fuel domestic home heating have been identified and discussed in an issues and options document regarding air quality management in Canterbury (CRC, 1993). Table 6.1 details a selection of potential management options for domestic home heating.

Table 6.1: Potential management options to reduce PM$_{10}$ emissions from domestic home heating

<table>
<thead>
<tr>
<th>Management options</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>1  Status quo</td>
<td></td>
</tr>
<tr>
<td>2  Education</td>
<td></td>
</tr>
<tr>
<td>3  Prohibit the use of coal</td>
<td></td>
</tr>
<tr>
<td>4  Prohibit the use of open fires</td>
<td></td>
</tr>
<tr>
<td>5  Require the replacement of solid fuel burners not meeting a specified criteria after a limited useful life</td>
<td></td>
</tr>
<tr>
<td>6  Smokeless fuels</td>
<td></td>
</tr>
<tr>
<td>7  Prohibit the use of all solid fuel domestic heating methods that do not meet a specific criteria (includes open fires)</td>
<td></td>
</tr>
<tr>
<td>8  Prohibit the use of all solid fuel domestic heating methods</td>
<td></td>
</tr>
<tr>
<td>9  Prohibit the use of solid fuel domestic home heating methods on days of high pollution potential</td>
<td></td>
</tr>
</tbody>
</table>
Subsequent sections assess the effectiveness of these management options in achieving the air quality target. Each analysis is based on a number of assumptions. Table 6.2 details assumptions made for the different management options illustrated in figures 6.1-6.9. In addition to these all calculations are based on the assumption of 100% compliance with proposed measures. The specific assumptions that are used for each management option are specified in the text prior to the illustration of the effectiveness of that option.

Table 6-2: Assumptions associated with management options aimed at reducing PM$_{10}$ concentrations

<table>
<thead>
<tr>
<th>No.</th>
<th>Assumption</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>new open fires will not be permitted and existing open fires will continue to decline at projected rate</td>
</tr>
<tr>
<td>2.</td>
<td>installations of solid fuel burners currently approved to the 5.5 g/kg criterion will continue until their expiry date (current maximum expiry date is September 2001)</td>
</tr>
<tr>
<td>3.</td>
<td>solid fuel burners approved from June 1997 which meet the 3 g/kg criterion will continue to be installed until January 2000</td>
</tr>
<tr>
<td>4.</td>
<td>the criterion for all solid fuel burning appliances installed after January 2000 will be 1.5 g/kg of fuel burnt</td>
</tr>
<tr>
<td>5.</td>
<td>Council will not approve applications for installation of solid fuel burners not meeting the required criteria at the specified dates</td>
</tr>
<tr>
<td>6.</td>
<td>all wood burners not meeting the 1.5 g/kg criterion will be replaced after 15 years useful life (Wood, 1997)</td>
</tr>
<tr>
<td>7.</td>
<td>an average useful life for coal ranges, incinerators and potbellies of 19 years</td>
</tr>
<tr>
<td>8.</td>
<td>emissions from transport and industry will continue to increase as predicted (figures 4.6 &amp; 4.7)</td>
</tr>
<tr>
<td>9.</td>
<td>use of the same emission factors, fuel use factors and operation factors, as detailed in appendix 5, for the period of assessment</td>
</tr>
<tr>
<td>10.</td>
<td>the percentage of incorrectly operated wood burners will be reduced by 25, 50, or 100% (appendix 5)</td>
</tr>
<tr>
<td>11.</td>
<td>households burning coal on open fires, potbellies, and incinerators will convert to burning wood</td>
</tr>
<tr>
<td>12.</td>
<td>30% of households burning coal on a coal burning appliance are assumed to convert to burning wood on these appliances</td>
</tr>
</tbody>
</table>

38 The extent to which households using coal burning appliances will convert to burning wood if coal is prohibited is uncertain. However, the design of most of these appliances is restrictive in
Chapter six - Assessing the effectiveness of different management options

13. the phase out of 50% of the open fires during the years 1998 to 2002 with use of the remaining 50% ceasing in 2003

14. of the open fires phased out from 1998 to 2002 80% are replaced with solid fuel burners

15. of the remaining open fires only 20% are replaced with solid fuel burners with the other 80% choosing options with less capital outlay (40% gas/40% electricity)

16. as a result of the phase out of solid fuel burners after a 15 year useful life, the replacement of old solid fuel burners with new burners is assumed to be lower (70%), due to the regulatory impact, than the same assumption in the status quo (80%)

17. the phase out of 40% of the open fires during the years 1998 to 2001 with use of the remaining 60% ceasing by 2002

18. of the open fires phased out from 1998 to 2001, 80% are replaced with wood burners. Of the open fires replaced after 2001, only 30% are replaced with wood burners with the other 70% choosing options with less capital outlay (e.g., gas or electricity)

19. as a result of an awareness campaign/incentives programme the replacement rate of wood burners is 50% as opposed to 70% as illustrated in figure 6.7

20. the phase out of 40% of the open fires during the years 1998 to 2001 with use of the remaining 60% ceasing by 2002

21. the overall replacement of open fires with solid fuel burners is 30% compared with 50% as illustrated in figure 6.7

22. a reduction of 50% in the number of new houses installing wood burners

Option 1: Status quo

The Clean Air Zone (Christchurch) order (1977) and the Clean Air Zone (Canterbury Region) Order (1984) preclude the installation of open fires and solid fuel burners not meeting the Clean Air Council (1987) criteria or criteria deemed as equivalent. In 1992 the joint New Zealand/
Australia standard, NZS7403 (Domestic solid fuel burning appliances – Method for determination of flue gas emissions), was deemed an equivalent standard. This specified an emission criterion of 5.5 grams of particulate per kg of fuel burnt (g/kg). In June 1997 this criterion was revised downwards by the CRC from 5.5 g/kg to 3 g/kg effective immediately and 1.5 g/kg from the year 2000. Figure 6.1 shows estimates of emissions for the current status quo option and what emission levels are estimated to have been had the 5.5 g/kg criterion remained.

Under the status quo option suspended particulate emissions are predicted to decrease with time. This is because the decrease in emissions from older appliances being replaced with newer, less polluting burners is likely to exceed increases in emissions resulting from an overall increase in appliance numbers.

Figure 6.1: Estimated total emissions of PM10 based on the previous test criterion of 5.5 g/kg for new installations of solid fuel burning appliances compared with the current test criteria of 3 g/kg and 1.5 g/kg in 2000

Projected total emissions associated with the status quo option are illustrated in figure 6.1. This assessment is based on the following assumptions numbered 1-7 detailed in table 6.2.

**Option 2: Education**

The effectiveness of an education strategy on suspended particulate emissions is difficult to determine. During the 1996 winter the Canterbury Regional Council undertook a publicity campaign discouraging the use of open fires, coal and pre 1989 wood burners on nights of high pollution potential (CRC, 1996b). Results of a pre-campaign and post-campaign survey indicated that while there was an increased awareness of air quality issues, the publicity had
done little to alter behaviour. Similar results were observed following an extensive air quality campaign in 1997 (Masilamani, 1997).

The extent to which reductions in suspended particulate emissions could occur, in theory, as a result of fewer households incorrectly operating their wood burners can be examined. The emission factors used for the emissions inventory (appendix 5) wood burners allow for incorrect operation. Alterations to the percentage of incorrectly operated wood burners used in emission calculations can give an estimate of the reductions in emissions that could result if different proportions of households operated their wood burners correctly (figure 6.2).

The assessment of emissions as illustrated in figure 6.2 is based on assumptions numbered 1-10 detailed in table 6.2.

![Figure 6.2: Estimated reduction in total emissions as a result of education measures to reduce the percentage of incorrectly operated wood burners](image)

The maximum reduction in emissions achievable from improved operation of wood burners alone is estimated at 8%. An additional reduction of 14% is required to achieve the air quality target by 2021. The amount of educational effort needed to achieve 25%, 50%, or 100% reductions in incorrect operation, as indicated in figure 6.2, is unknown but is likely to be high.

**Option 3: Prohibit the use of coal**

The estimated effect of banning the use of coal for domestic heating is illustrated in figure 6.3. It is assumed that the majority of households burning coal will convert to wood. Therefore the predicted reduction is substantially smaller than what might be expected, given the relative contribution of domestic coal burning to PM$_{10}$ emissions.
The assessment of emissions as illustrated in figure 6.3 is based on assumptions 1-9, 11 & 12 as detailed in table 6.2.

Figure 6.3 illustrates that a reduction of approximately 20% could be expected as a result of a coal ban. In the longer term a coal ban does not appear to have a large additional effect relative to the status quo. This is because the status quo assumes a significant decrease in the use of coal on open fires as well as the natural phase out of high emitting coal burners.

**Option 4: Prohibit the use of open fires**

The effect of prohibiting open fires for a variety of scenarios is considered in chapter 5. Figure 6.4 illustrates the effect of banning open fires based on assumptions 2-9, 13, 14 & 15 as detailed in table 6.2.
Chapter six - Assessing the effectiveness of different management options

**Figure 6.4: Estimated emissions resulting from prohibiting the use of open fires in 2003**

Figure 6.4 indicates that, based on the above, this option comes close to the air quality target around the year 2013. It is important to note, however, that estimated emissions associated with this are dependent on the general assumptions regarding the replacement rate of older wood burners i.e., by the year 2013 all solid fuel burners meet the 1.5 g/kg criterion. The estimated emissions will also be very dependent on the proportion of open fires replaced with solid fuel and non-solid fuel alternatives. The percentages used for the replacement of open fires and old solid fuel burners with new burners, for the purpose of assessing the effectiveness of management options, are estimates based on the current and predicted trends in home heating methods (Wood, 1997a).

**Option 5: Require the replacement of solid fuel burners not meeting a specified criteria after a limited useful life**

The useful life of a wood or coal burning appliance will vary depending on the physical properties of the burner, the fuel burnt, and the frequency of use. An average figure of 15 years is suggested by manufacturers as the useful life of a wood burner40 (Dave Pullen, Home heating association, pers. comm., 1997; Canterbury United Council, 1988). The emission criterion for solid fuel burners effective from the year 2000 is 1.5 grams of particulate per kilogram of fuel burnt. This management option considers the effect of requiring that all solid fuel burners that do not meet this criteria be replaced with a new solid fuel burner or alternative heating method 15 years after the appliance was installed.

The assessment of emissions as illustrated in figure 6.5 is based on assumptions 1-5, 6-9, & 16 as detailed in table 6.2.

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40 The actual life of any particular burner will vary according to the physical properties of the burner, the type and quality of the fuel burnt in it, the frequency of use and other factors including maintenance. Recognising that there is scope for variability among burners, but also recognising the need for an agreed useful life that can be used for planning and projection purposes, an average useful life of 15 years for all burners has been selected. This is consistent with previous estimates (Canterbury United Council, 1988).
Option 6: Smokeless fuels

“Smokeless” fuels have been used successfully overseas to reduce emissions from domestic home heating. These fuels are usually produced from coal with the most common examples being coke and charcoal type products. Previous investigations by the former Department of Science and Industrial Research (now ESR) into the use of “smokeless” fuels have indicated that fuels of this kind would not be a commercially viable option due to the cost of manufacture and the subsequent cost to users (pers. comm., R. B. Ayrey, 1997). This is because the economic production of such fuels is normally a result of by-product from a large-scale manufacturing process such as coal gas production. Recent inquiries into potential industrial sources of such fuels indicate no recent change in viability (pers. comm., L. F. Damiano, Coal Research, 1997). However, a small number of manufacturers have either produced or are currently looking at alternative fuels involving compacted sawdust and other materials.
The potential for development of the latter fuels to reduce particulate emissions has not been established. In their present form the emissions could be expected to be similar to those from the burning of ultra dry firewood (pers. comm., R. B. Ayrey, 1997).

**Option 7 Prohibit the use of all solid fuel burners, including open fires, that do not meet specific test criteria**

Standards require that all solid fuel burners approved for installation in the Christchurch Clean Air Zones from the year 2000 meet a test criterion of 1.5 g/kg. Because existing approvals were not revoked, some appliances meeting the previous 5.5 g/kg and the current 3 g/kg criteria can be installed until the September 2001. If a ban on older wood burners was introduced before 2002, these could be replaced with appliances that do not meet the 1.5 g/kg criterion. Similarly, open fires could be replaced with such burners.

A prohibition of the use of all solid fuel burners that do not meet the test criterion of 1.5 g/kg would include the prohibition of all open fires and solid fuel burners not meeting this criterion. However, because of the current high contribution of domestic coal burning a prohibition on the use of coal is also included as a separate option. This combination of options could be implemented over a variety of time periods.

At this stage this appears to be the option preferred by the Canterbury Regional Council. The timeframe recommended by this Council for inclusion in a draft plan is:

1) prohibit the use of coal on existing appliances by 30 September 1998;
2) prohibit the use of open fires by 30 September 2001;
3) phasing out of solid fuel burners not complying with the 1.5 g/kg criterion after 15 years useful life.

The assessment of emissions for the option described in points 1-3 above is illustrated by the solid line in figure 6.7. This is based on assumptions 2-6, 8, 9, 11, 12, & 17-19 as detailed in table 6.2. An assessment of emissions for the option described in points 2-3 above (i.e., excluding the coal ban) is illustrated by the bold dashed line in figure 6.8. This is based on assumptions 2-6, 8, 9, & 17-19.
Figure 6.7 illustrates estimated emissions for this option both excluding (bold dashed line) and including (solid line) a prohibition on the use of coal by October 1998.

One major uncertainty in estimating the effectiveness of various management options is predicting home heating methods chosen by households when replacing an open fire or solid fuel burner. The estimated standard error (section 6.2) for the predicted number of households using solid fuel burners by the year 2021 for this management option is 30%. However, additional measures such as an awareness campaign encouraging non solid fuel alternatives or an incentives program limited to non solid fuel replacements could considerably reduce the number of households choosing to use solid fuel methods. Figure 6.7 illustrates estimated emissions if a reduced percentage of households replace open fires and solid fuel burners with wood burners.

The assessment of emissions as illustrated in figure 6.8 is based on assumptions 2-6, 8, 9, 11, 12, & 20-23 as detailed in table 6.2.
Chapter six - Assessing the effectiveness of different management options

Figure 6.8: Estimated emissions for the preferred management options combined with an awareness campaign encouraging the use of non-solid fuel alternatives.

These results suggest that to ensure that the preferred management option does result in PM$_{10}$ concentration in Christchurch meeting the air quality target additional measures should be adopted. These could include an awareness campaign encouraging the use of non-solid fuel methods of home heating, the introduction of a management option to reduce industrial emissions, and rules or education to reduce the incorrect operation of appliances. The latter may be able to be achieved by a rule relating to visible smoke to be included in an air plan with the introduction of the proposed prohibitions.

It is also possible that guidelines for particulate will be reduced even further in the future as our understanding of the health impact associated with suspended particulate concentrations improves. It is possible, therefore, that even stricter regulatory control may be required in the future.

Option 8: Prohibit the use of all solid fuel domestic home heating methods

Figure 6.9 illustrates the effect of introducing a ban on all solid fuel burning in the years 2000 and in 2003. Given the extent of the reduction obtained as a result of a ban on solid fuel domestic home heating, additional requirements relating to controlling the predicted increases in emissions from the industrial and transport sectors are not major factors in achieving the air quality target for PM$_{10}$.
While this option clearly achieves the air quality target other factors such as social and economic costs preclude its selection as a preferred management option for Christchurch. In addition to the cost of replacement of heating methods a ban on solid fuel domestic heating is likely to increase electricity costs for all Christchurch households. This is because the increase in peak electricity demand would require additional transmissions and distribution capacity. Ball (1997) concludes that the combination of relatively little extra emission reductions and substantially increased costs makes this option and expensive means of reducing emissions.

Notwithstanding this, this option may need further consideration in future years if wood burner numbers are not adequately reduced or if health effect studies indicate that further reductions in the guideline for PM$_{10}$ are required.

**Option 9: Prohibit the use of solid fuel domestic home heating on days/nights of high air pollution potential**

Pollution potential forecasting has been a component of the Canterbury Regional Council’s air quality campaign for a number of years. Estimates of pollution potential based on predicted meteorological conditions have been made on a daily basis during winters since 1992.

In 1996 the Canterbury Regional Council combined pollution potential forecasts with an education campaign requesting people to use alternatives to open fires, coal and pre 1989 wood
burners on nights when the pollution potential was high. Surveys conducted indicated that there was good awareness of the campaign but that it had not been successful in altering behaviour (CRC, 1997b). A similar campaign was also instituted by the Canterbury United Council in 1987. Awareness of the campaign for voluntary “fire free days” was high (73%) but the success in terms of changing people’s behaviour was low with only 14% of those who heard or saw an announcement complying at least once (CRC, 1997b).

Compulsory fire free days would be more effective in reducing emissions than the voluntary programmes of the past. However, a significant level of uncertainty is associated with forecasting meteorological conditions likely to result in high pollution days. The number of days for which high pollution potential would be predicted is likely to be twice the number of days on which high pollution events would actually occur (pers comm., Aberkane CRC, 1997). Based on data from previous winters, this would range from 40%-60% of winter days. This option would achieve reductions in 1996 emission levels of between 0 and 90% depending on the extent of compliance and the extent to which PM$_{10}$ emissions from the transport and industrial sectors are permitted to increase. Management on this basis would, however, create uncertainty both for householders and enforcers. Thus other options are likely to be preferred.

**Summary**

Section 6.1 applies the model to a variety of management options for domestic heating. Only a few options appear to meet the air quality target. Social and economic considerations are beyond the scope of this investigation. However, these aspects have been considered by the CRC (Ball, 1996; Lamb, 1997). As a result of the technical, social and economic analysis the Council Committee has chosen option 7 as a preferred management option. The timeframe for the implementation is important from a social perspective. The timeframe chosen by the Council for the implementation of the individual components of option 7 are shown in figure 6.7.

**6.2 Error and sensitivity analysis**

Chapter five investigates the validity of the model and the input parameters. While the model responds in a logical way the error surrounding the accuracy of the input parameters is likely to be high. The estimated standard error for each of the input variables is described in appendix 9.

The estimated standard error for the years 1996-2021 for the preferred management option illustrated in figure 6.7 is shown in figure 6.10.
Figure 6.10: Error analysis of calculations of PM$_{10}$ emissions based on a prohibition of the use of open fires by October 2001 and non complying wood burners after 15 years useful life by 2002

The emission value at 2021 for the preferred management option expressed as a percentage of 1996 emission levels is 27%. The estimated standard error of this value (figure 6.10) is ±21% (95%CI) and was calculated by combining the estimated standard errors of the individual variables in equation E5.1 using formulae given in Topping (1971, p82). This result indicates that for this management option 68% of emission levels at the year 2021 will lie in the range of 19 - 35% of 1996 emission levels and 95% would lie in the range 6%-48%.

The sensitivity of the analysis was examined for the preferred management option (figure 6.7). This was done by considering the effect of variances in input parameters on the calculated emissions at the year 2021. The number of houses using solid fuel domestic heating and the emission factors for wood burners meeting the 1.5 g/kg criterion are the variables with the greatest impact on calculated emissions in that order.

6.3 Effect of management options on exceedences

Assessment of the impact, in terms of the number of exceedences of the air quality guidelines for PM$_{10}$, of introducing various management options, as discussed in previous sections, is illustrated in figure 6.11. This shows the mean$^{41}$ number of days the guideline concentration is predicted to be exceeded relative to the emission level as a percentage of 1996 emissions.

$^{41}$ Variations in exceedences will occur from year to year, to date depending on meteorological conditions and PM$_{10}$ emissions. The indicated exceedences refer to the average number that may occur over a nine year period.
Chapter six - Assessing the effectiveness of different management options

The concentrations of PM$_{10}$ used to calculate the data illustrated in figure 6.11 are contained in appendix 7. These have been calculated using equation E4.1 and the methodology detailed in section 4.1.2.

![Figure 6.11: Average number of predicted exceedences of PM$_{10}$ guideline of 50 $\mu$gm$^{-3}$ relative to 1996 emission levels](image)

As an example the preferred option, illustrated in figure 6.7, shows a reduction in emissions of approximately 63% by the year 2002. A reduction of this magnitude would suggest that, on average, the guideline of 50 $\mu$gm$^{-3}$ would be exceeded on 3 days per year. The magnitude of errors contained in the model inputs suggests that the numbers of days the guideline may actually be exceeded may be between 0-6 days (3 ± 3 days).

6.4 Reductions in emissions of other contaminants as a result of management options to reduce PM$_{10}$

While the Canterbury Regional Council has given priority to the management of PM$_{10}$, concentrations of other contaminants in the Christchurch airshed are at times high enough to be of concern. These pollutants include SO$_2$, CO and NO$_2$. Assessment of any reductions required in these contaminants and appropriate management options for these contaminants are beyond the scope of this investigation. Management options that are aimed at reducing PM$_{10}$ will, however, impact on emission of other contaminants.

Overall reductions in emissions of each contaminant depends on the relative contribution of the domestic heating sector to total emissions of that particular contaminant. On a typical winter’s’ night in Christchurch approximately 33% of carbon monoxide (CO) emissions, 37% sulphur
oxides (SO\textsubscript{x}) emissions and 4% of the nitrogen oxide (NO\textsubscript{x}) emissions come from the domestic heating sector (CRC, 1997a).

Reductions in emissions of SO\textsubscript{x}, CO and NO\textsubscript{x} associated with the preferred management option (figure 6.7) are assessed by exchanging emission factors for these contaminants into equations E4.1. The emission factors, obtained from the Christchurch emission inventory are described in appendix 5. The results of these calculations are shown in figure 6.12.

Figure 6.12: Reductions in emissions of CO, NO\textsubscript{x}, and SO\textsubscript{x} from domestic home heating associated with the preferred management option aimed at reducing PM\textsubscript{10} concentrations.

Figure 6.12 shows a reduction in home heating SO\textsubscript{x} emissions of approximately 90% associated with the prohibition of the use of coal for domestic use, open fires and the phase out of older burners. This reduction is primarily attributable to prohibiting coal for domestic use. Oil burners are also a source of domestic SO\textsubscript{x} emissions. While numbers of these burners are currently low (less than 5000 households) future restrictions on solid fuel may result in this becoming a more popular method of home heating. The effect of an increase in the number of households using oil (up to 20,000) is illustrated in figure 6.12 by the gradual increase in SO\textsubscript{x} emissions after 2003. Compared to the reductions achievable through the banning of coal and SO\textsubscript{x} emissions from other sources, an increase in the number of oil burners is unlikely to have significant effects on ambient sulphur dioxide concentrations experienced in Christchurch on a typical winter’s day.

A reduction in home heating CO emissions of approximately 65% is associated with prohibiting the use of open fires and the phasing out older, pre 1989 burners. A further reduction of just over 10% is expected by the phase out of non-complying burners and replacement by burners
with lower emissions. Increases in the use of gas or oil have little impact on CO emissions relative to the reductions achieved through the improved combustion of wood.

Nitrogen oxide home heating emissions are predicted to decrease by approximately 50% with a ban on open fires and pre 1989 burners. Predicted increases in gas as a method of home heating, as illustrated in figure 4.4, will result in increases in emissions from the domestic sector. These increases are unlikely to be significant compared to the predicted reductions (figure 6.12), and compared to the contribution of the transport sector to NOx emissions (approximately 90%).

Table 6.3 shows the effect of the preferred management option on overall emissions of other contaminants.

### Table 6.3: Estimated reductions in emissions of CO, SOx, and NOx associated with banning open fires and wood burners not complying with the 1.5 g/kg test criterion by 2021.

<table>
<thead>
<tr>
<th>Contaminant</th>
<th>Reduction in domestic home heating emissions by 2021 (approximately)</th>
<th>Overall reduction in emissions by 2021 (approximately)</th>
</tr>
</thead>
<tbody>
<tr>
<td>PM10</td>
<td>87 %</td>
<td>74 %</td>
</tr>
<tr>
<td>CO</td>
<td>70 %</td>
<td>23 %</td>
</tr>
<tr>
<td>SOx</td>
<td>80 %</td>
<td>30 %</td>
</tr>
<tr>
<td>NOx</td>
<td>50 %</td>
<td>2 %</td>
</tr>
</tbody>
</table>

The extent to which management options to reduce suspended particulate concentrations will address other air quality issues is yet to be fully determined. It is unlikely that issues such as daytime haze and high street level CO concentrations will be markedly improved as a result of measures to address PM10. Ambient CO concentrations will be reduced to some extent but it is likely that further measures targeting transport management/ emissions will be required to further reduce CO to acceptable concentrations. It is also likely that the further restrictions in the transport sector would be required to address the daytime haze issue.

### 6.5: Summary

This chapter shows the results of applying a newly developed model to a variety of management options to predict the effect of these options on PM10 concentrations. This modelling has provided the technical basis for the selection of a preferred management option by the
Canterbury Regional Council to control ambient PM\(_{10}\) concentrations. Council's decision was also based on social and economic considerations detailed in a report by Ball (1998).

Because of uncertainties associated with the input parameters for the model an error analysis indicating the estimated standard error for this preferred option has been calculated. As predicted the error associated with this calculation is large (approximately 30% of the estimated PM\(_{10}\) concentration at 2021).

The estimated number of days that the PM\(_{10}\) guideline of 50 \(\mu\)gm\(^{-3}\) could be exceeded under different management options is examined in section 6.4. This shows that for the preferred management option the number of exceedences should be reduced from approximately 28 days per year to an average of approximately 3 ±3 days per year by the year 2002\(^{42}\).

The implementation of this management option is also likely to reduce emissions of other contaminants. Section 6.5 shows that minimal reductions in NO\(_x\) concentrations can be expected through the implementation of this option. However, emissions of CO and SO\(_2\) are likely to decrease by 20-30% of 1996 emission levels.

\(^{42}\) assuming 100% compliance
CHAPTER SEVEN: CONCLUSIONS

7.1 Summary
The research contained in this thesis aims to test the hypothesis that “suspended particulate concentrations in Christchurch can be reduced to acceptable concentrations through the implementation of management options targeting primary sources of particulate”.

This hypothesis was tested by:

- An assessment of the adverse health effects associated with suspended particulate and appropriate guideline concentrations to avoid these effects;
- An examination of the current suspended particulate concentrations measured in Christchurch to determine the levels of reduction required reach an acceptable guideline concentration;
- The identification of the primary source of wintertime PM$_{10}$ concentrations in Christchurch;
- The development, validation and application of a model to determine the effectiveness of different management options for reducing suspended particulate concentrations;
- An investigation into an appropriate method of measuring compliance.

The literature reviewed indicates that current PM$_{10}$ concentrations in Christchurch are likely to be associated with adverse health effects. The potential for adverse health effects will be significantly reduced if the 24-hour average PM$_{10}$ concentrations are reduced to meet a guideline of 50 $\mu$gm$^{-3}$.

Based on measured concentrations from 1988-1996 the reduction in PM$_{10}$ concentrations required to meet a 50 $\mu$gm$^{-3}$ guideline is 74%.

The domestic heating sector was identified as the primary source of wintertime PM$_{10}$ concentrations in Christchurch. An analysis of a number of different management measures identified two options that the model indicates are likely to meet the air quality target by 2021. These are:

- prohibiting the use of open fires, and requiring the replacement of wood burners after 15 years useful life;
- prohibiting the use of all solid fuel domestic heating.

The option of prohibiting the use of coal, open fires and a mandatory phase out of wood burners after 15 years useful life is the option preferred by the Canterbury Regional Council.
The extent to which this option will meet the air quality target is largely dependent on the number of households that choose to use solid fuel burners in the future. To ensure that the air quality target is met additional measures may be required. These include:

- an awareness campaign to encourage the use of non-solid fuel alternatives;
- management options to reduce increases in industrial emissions;
- education or regulatory measures to reduce the incorrect operation of solid fuel burners.
- The implementation of a rule in the air plan relating to visible smoke from a domestic dwelling to reduce the number of households incorrectly operating their burners.

### 7.2 Further Research Issues

This investigation has identified the following issues as requiring further research:

- The extent to which emissions of PM$_{10}$ from the industrial and transport sectors are allowed to increase and subsequent management options that may be required;
- The extent to which reducing PM$_{10}$ concentrations based on monitoring results for the St Albans site will result in acceptable PM$_{10}$ concentrations in other areas of Christchurch;
- Methods for enforcing the proposed management options;
- Methods for reducing the use of wet wood;
- The extent to which concentrations of other contaminants in Christchurch may need to be reduced;
- The need to review the PM$_{10}$ guideline or to introduce a PM$_{2.5}$ guideline on the basis of future health effects research;
- Ways and data to further validate the reduction scenario model developed.

The results of this investigation show that through management of PM$_{10}$ emissions it is possible to reduce wintertime PM$_{10}$ concentrations to CRC guideline levels by the year 2021. Thus the hypothesis that this thesis tests is shown to be correct.
References


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References


References


References


Todd, J. J. & Singline, R. 1989, The impact of wood heaters on Air Quality in Australia, fuelwood report no. 2; Centre for Environmental Studies, University of Tasmania, Hobart.


References


Wood, D. R. 1997a, Domestic home heating projections and the demise of the open fire. Canterbury Regional Council U97/68.

Appendix 1: Epidemiological studies for PM$_{10}$ and mortality

Table A.1: Longitudinal time series studies: mortality (from Vedal, 1997)

<table>
<thead>
<tr>
<th>Location and period of study</th>
<th>Particle measure</th>
<th>PM$_{10}$ daily mean (range)</th>
<th>% change in mortality for each 10 $\mu$g/m$^3$ increase in PM$_{10}$ (95%CI)</th>
<th>reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>St Louis, MO (1985-86)</td>
<td>PM$_{10}$</td>
<td>28 (1-97)</td>
<td>1.5% (0.1, 2.9)</td>
<td>Dockery et al., (1992)</td>
</tr>
<tr>
<td>Kingston, TN (1985-86)</td>
<td>PM$_{10}$</td>
<td>30 (4-67)</td>
<td>1.6% (-1.3, 4.6)</td>
<td>Dockery et al., (1992)</td>
</tr>
<tr>
<td>Santa Clara, CA (1980-82, 84-86)</td>
<td>CoH$^{43}$</td>
<td>35 (N/A)</td>
<td>0.8% (0.2, 1.5) total 3.5% (1.5, 5.6) respiratory 0.8% (0.1, 1.6) cardiac</td>
<td>Fairley et al., (1990)</td>
</tr>
<tr>
<td>Philadelphia, PA (1973-1980)</td>
<td>TSP</td>
<td>40 (26-70)**</td>
<td>1.2% (0.7, 1.7) total 3.3% (0.1, 6.6) respiratory 1.7% (1.0, 2.4) cardiac</td>
<td>Schwartz &amp; Dockery, (1992a)</td>
</tr>
<tr>
<td>Utah Valley, UT (1985-1989)</td>
<td>PM$_{10}$</td>
<td>47 (1-365)</td>
<td>1.5% (0.9, 2.1) total 3.7% (0.7, 6.7) respiratory 1.8% (-0.5, 3.7) cardiac</td>
<td>Pope et al., (1992)</td>
</tr>
<tr>
<td>Birmingham, AL (1985-1988)</td>
<td>PM$_{10}$</td>
<td>48 (21-80)**</td>
<td>1.0% (0.2, 1.9) total 1.5% (-5.8, 9.4) respiratory 1.6% (-0.5, 1.6) cardiac</td>
<td>Schwartz et al., (1993)</td>
</tr>
</tbody>
</table>

$^{43}$ CoH = Coefficient of haze
<table>
<thead>
<tr>
<th>Location and period of study</th>
<th>Particle measure</th>
<th>PM$_{10}$ daily mean (range)</th>
<th>% change in mortality for each 10 µg m$^{-3}$ increase in PM$_{10}$ (95% CI)</th>
<th>reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Detroit, MI (1973 - 1982)</td>
<td>TSP</td>
<td>48 (32-73)**</td>
<td>1.0% (0.5, 1.6)</td>
<td>Schwartz et al., (1991)</td>
</tr>
<tr>
<td>Steubenville, OH (1978 - 1984)</td>
<td>TSP</td>
<td>61 (22-125)*****</td>
<td>0.7% (0.4, 1.0)</td>
<td>Schwartz &amp; Dockery, (1992b)</td>
</tr>
<tr>
<td>Los Angeles, CA</td>
<td>KM</td>
<td>N/a</td>
<td>Particles pollution associated with total mortality but not cardiac or respiratory mortality 1% (0.5, 1.6)</td>
<td>Kinney, 1991</td>
</tr>
<tr>
<td>Cinicinnati, OH</td>
<td>TSP</td>
<td>62 (21-117)**</td>
<td>0.3% (-0.1, 0.8)</td>
<td>Moolgavkar, 1995</td>
</tr>
<tr>
<td>Philadelphia, PA (1973-88)</td>
<td>TSP</td>
<td>37 (8-186)</td>
<td>0.3% (-0.1, 0.7)</td>
<td>Moolgavkar, 1995</td>
</tr>
<tr>
<td>Cook County, IL (1985-92)</td>
<td>PM$_{10}$</td>
<td>37* (4-365)</td>
<td>0.5% (0.1, 0.9)</td>
<td>Styler, 1995</td>
</tr>
<tr>
<td>Salt Lake City, UT (1985-1990)</td>
<td>PM$_{10}$</td>
<td>48* (9-194)</td>
<td>-0.2, (-1.1, 0.7)</td>
<td>Styler, 1995</td>
</tr>
<tr>
<td>Efurt, East Germany (1988-89)</td>
<td>TSP</td>
<td>58* (6-358)</td>
<td>0.7% (n/a, p=0.04)</td>
<td>Spix, 1993</td>
</tr>
<tr>
<td>Athens, Greece (1984-88)</td>
<td>BS$^{44}$</td>
<td>83 (n/a)</td>
<td>0.4% (0.1, 0.8)</td>
<td>Touloumi, 1994</td>
</tr>
<tr>
<td>Beijing (1989)</td>
<td>TSP</td>
<td>206 (85-552)**</td>
<td>0.9% (0.0, 1.8) only in summer</td>
<td>Xu, 1994</td>
</tr>
<tr>
<td>Sao Paulo, Brazil (1990-91)</td>
<td>PM$_{10}$</td>
<td>82</td>
<td>1.2% (0.6, 1.7)</td>
<td>Saldiva, 1995</td>
</tr>
<tr>
<td>Bratislava, Slovak Republic (1987-91)</td>
<td>TSP</td>
<td>49-0-396)</td>
<td>0.0%(-0.7, 0.7)</td>
<td>Bacharova, 1996</td>
</tr>
<tr>
<td>Paris France (1987-92)</td>
<td>PM$_{13}$</td>
<td>51 (19-137)*</td>
<td>1.5% (0.4, 2.7)</td>
<td>Dab, 1996</td>
</tr>
</tbody>
</table>

$^{44}$ BS = British Smoke
<table>
<thead>
<tr>
<th>Location and period of study</th>
<th>Particle measure</th>
<th>PM$_{10}$ daily mean (range)</th>
<th>% change in mortality for each 10 ( \mu \text{g} \cdot \text{m}^{-3} ) increase in PM$_{10}$ (95%CI)</th>
<th>reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mexico City, Mexico</td>
<td>TSP</td>
<td>112* (36-251)</td>
<td>0.3% (0.2, 0.4)</td>
<td>Borja-Aburto, 1996 (1990-92)</td>
</tr>
<tr>
<td>Santiago, Chile (1989, 91)</td>
<td>PM$_{10}$</td>
<td>115 (32-367)</td>
<td>0.7% (0.5, 1.0) total</td>
<td>Ostro, 1996</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>1.2% (0.9, 1.8) respiratory</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.8% (0.3, 1.1) cardiac</td>
<td></td>
</tr>
<tr>
<td>Køln, Germany (1975-85)</td>
<td>TSP</td>
<td>37* (n/a-167)</td>
<td>0.3% (-0.2, 0.7)</td>
<td>Spix, 1996</td>
</tr>
<tr>
<td>Barcelona, Spain (1985-91)</td>
<td>BS</td>
<td>n/a (11-126)</td>
<td>0.5% (0.3, 1.1) total</td>
<td>Sunyer, 1996</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.9% (-0.1, 2.0) respiratory</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.9% (0.4, 1.4) cardiac</td>
<td></td>
</tr>
<tr>
<td>Athens, Greece (1987-89)</td>
<td>BS</td>
<td>84 (9-333)</td>
<td>0.5% (0.3, 0.7)</td>
<td>Touloumi, 1996</td>
</tr>
<tr>
<td>Amsterdam, Netherlands (1986-92)</td>
<td>PM$_{10}$</td>
<td>38 (n/a-163)</td>
<td>0.6% (-0.1, 1.4)</td>
<td>Verhoef, 1996</td>
</tr>
<tr>
<td>Milan, Italy (1980-89)</td>
<td>TSP</td>
<td>76 (2-291)</td>
<td>0.7% (0.1, 1.6)</td>
<td>Vigotti, 1996</td>
</tr>
<tr>
<td>Wracow, Poland (1979-89)</td>
<td>BS</td>
<td>54* (26-247)***</td>
<td>0.1% (-0.2, 0.3)</td>
<td>Wojtyniak, 1996</td>
</tr>
<tr>
<td>Poznan, Poland (1983-90)</td>
<td>BS</td>
<td>34* (-92)***</td>
<td>0.1% (-0.1, 0.3)</td>
<td>Wojtyniak, 1996</td>
</tr>
<tr>
<td>Cracow, Poland (1977-89)</td>
<td>BS</td>
<td>73* (26-247)***</td>
<td>0.2% (0.0, 0.4)</td>
<td>Wojtyniak, 1996</td>
</tr>
<tr>
<td>Lodz, Poland (1977-90)</td>
<td>BS</td>
<td>57* (20-151)***</td>
<td>0.2% (0.1, 0.4)</td>
<td>Wojtyniak, 1996</td>
</tr>
<tr>
<td>Lyon, France (1985-90)</td>
<td>PM$_{10}$</td>
<td>38 (3-180)</td>
<td>0.2% (-0.6, 1.0) total</td>
<td>Zmirou, 1996</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.8% (0.0, 1.7) respiratory</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.8% (-0.2, 1.9) cardiac</td>
<td></td>
</tr>
</tbody>
</table>

*Median, ** (5th to the 95th percentile), *** (10th to the 90th percentile), n/a not available.
Appendix 2: Calculation of potential mortality effects for Christchurch (from Foster, 1996)

Mortality data were selected for Christchurch areas that are likely to be susceptible to PM$_{10}$ concentrations. Data were available on the basis of areas divided for census purposes. Suspended particulate data from the St Albans monitoring site were used. A home heating survey conducted in Christchurch in 1995 indicated that there are areas with both greater and lesser PM$_{10}$ emissions than St Albans (NIWA, 1995). Variations in air quality across Christchurch City are largely dependent on topography and meteorology as well as emissions. As such variations between different areas are likely. However, monitoring data from the St Albans site have been used to represent concentrations of PM$_{10}$ in Christchurch.

The ICD codes used respiratory deaths were: 480-486, 490-493, 496 -507.
The ICD codes used for cardiopulmonary deaths were: 402, 410-414, 426-429.

Assumptions:
- For each 10 $\mu$g$m^{-3}$ increase in daily PM$_{10}$
  - total mortality increases 1.0%$^{45}$
  - respiratory mortality increases 3.4%
  - cardiac mortality increases 1.4%

Baseline mortality data for Christchurch for 1992 were$^{46}$:
- total mortality: 2519 deaths per year
- respiratory: 264 deaths per year
- cardiac: 721 deaths per year

Baseline mortality data for Christchurch for 1993 were:
- total mortality: 2506 deaths per year
- respiratory: 227 deaths per year
- cardiac: 737 deaths per year

Baseline deaths per day for Christchurch were:

<table>
<thead>
<tr>
<th>Year</th>
<th>Total Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>1992</td>
<td>6.9</td>
</tr>
<tr>
<td>1993</td>
<td>6.9</td>
</tr>
</tbody>
</table>

Calculations and Impact

Extra deaths associated with increases in PM$_{10}$ each year:

<table>
<thead>
<tr>
<th>Year</th>
<th>Total Mortality</th>
<th>Respiratory Mortality</th>
<th>Cardiac Mortality</th>
</tr>
</thead>
<tbody>
<tr>
<td>1992</td>
<td>total mortality 6.9 x 0.01 x 416 = 29</td>
<td>respiratory mortality 0.70 x 0.034 416 = 10</td>
<td>cardiac mortality 2.0 x 0.014 x 416 = 12</td>
</tr>
<tr>
<td>1993</td>
<td>total mortality 6.9 x 0.01 x 298 = 21</td>
<td>respiratory mortality 0.62 x 0.034 x 298 = 6</td>
<td>cardiac mortality 2.0 x 0.014 x 298 = 8</td>
</tr>
</tbody>
</table>

Each 10 $\mu$g$m^{-3}$ increase in daily PM$_{10}$ above 20 $\mu$g$m^{-3}$ for the Christchurch population in 1992 is associated with$^{47}$:
- (2519 $\div$ 365 days) x 0.010 = 0.069 extra total deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.
- (264 $\div$ 365 days) x 0.034 = 0.025 extra respiratory deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.
- (721 $\div$ 365 days) x 0.014 = 0.028 extra cardiac deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.

Each 10 $\mu$g$m^{-3}$ increase in daily PM$_{10}$ above 20 $\mu$g$m^{-3}$ for the Christchurch population in 1993 is associated with$^{47}$:
- (2506 $\div$ 365 days) x 0.010 = 0.069 extra total deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.
- (227 $\div$ 365 days) x 0.034 = 0.021 extra respiratory deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.
- (737 $\div$ 365 days) x 0.014 = 0.028 extra cardiac deaths per 10 $\mu$g$m^{-3}$ increase in PM$_{10}$.

Total number of daily 10 $\mu$g$m^{-3}$ increments above 20 $\mu$g$m^{-3}$ each year were:

- 1992: 416
- 1993: 298

$^{45}$ Averages derived by Dockery et al. (1994).
$^{46}$ PM$_{10}$ concentrations on a year of relatively high pollution (1992) and relatively low pollution (1993) were chosen to represent a range in possible deaths per year.

$^{47}$ Each 10 $\mu$g$m^{-3}$ increase in daily PM$_{10}$ above 20 $\mu$g$m^{-3}$ is equivalent to no effects below a concentrations of 30$\mu$g$m^{-3}$.
Appendix 3: Calculation of potential morbidity for Christchurch (From Foster, 1996)

Health statistics and PM$_{10}$ concentrations are based on the information in appendix two. Hospitalisation statistics were available only for public hospitals. Results are therefore likely to underestimate potential effects.

The ICD codes used respiratory diseases were: 480-486, 490-493, 496-507. The ICD codes used for cardiopulmonary diseases were: 402, 410-414, 426-429. The ICD code used for asthma was: 493

Assumptions:

For each 10 µg/m$^3$ increase in daily PM$_{10}$:
- respiratory hospitalisations increase 0.8%
- cardiac hospitalisations increase 0.6%
- asthma hospitalisations increase 1.9%

Baseline hospitalisations in Christchurch for 1992 and 1993 were:

<table>
<thead>
<tr>
<th>Year</th>
<th>Respiratory</th>
<th>Cardiac</th>
<th>Asthma</th>
</tr>
</thead>
<tbody>
<tr>
<td>1992</td>
<td>1190</td>
<td>2534</td>
<td>810</td>
</tr>
<tr>
<td>1993</td>
<td>1237</td>
<td>2845</td>
<td>846</td>
</tr>
</tbody>
</table>

Baseline hospitalisations per day for 1992 and 1993 were:

<table>
<thead>
<tr>
<th>Year</th>
<th>Respiratory</th>
<th>Cardiac</th>
<th>Asthma</th>
</tr>
</thead>
<tbody>
<tr>
<td>1992</td>
<td>3.3</td>
<td>6.9</td>
<td>2.2</td>
</tr>
<tr>
<td>1993</td>
<td>3.4</td>
<td>7.8</td>
<td>2.3</td>
</tr>
</tbody>
</table>

Average number of 10µg/m$^3$ increments above 20 µg/m$^3$ each year were:

<table>
<thead>
<tr>
<th>Year</th>
<th>1992</th>
<th>1993</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>416</td>
<td>298</td>
</tr>
</tbody>
</table>

Calculations and Impact

Potential increases in hospitalisations associated with increases in PM$_{10}$ each year:

### 1992
- respiratory admissions - 3.3 x 0.008 x 416 = 11
- cardiac admissions - 6.9 x 0.006 x 416 = 17
- asthma admissions - 2.2 x 0.019 x 416 = 17

### 1993
- respiratory admissions - 3.4 x 0.008 x 298 = 8
- cardiac admissions - 7.8 x 0.006 x 298 = 14
- asthma admissions - 2.3 x 0.019 x 298 = 13

Each 10 µg/m$^3$ increase in daily PM$_{10}$ above 20 µg/m$^3$ for the Christchurch population in 1992 is associated with:

- (1190 ÷ 365 days) x 0.008 = 0.026 extra respiratory admissions per 10 µg/m$^3$ increase PM$_{10}$.
- (2534 ÷ 365 days) x 0.006 = 0.042 extra cardiac admissions per 10 µg/m$^3$ increase PM$_{10}$.
- (810 ÷ 365 days) x 0.019 = 0.042 extra asthma admissions per 10 µg/m$^3$ increase PM$_{10}$.

Each 10 µg/m$^3$ increase in daily PM$_{10}$ above 20 µg/m$^3$ for the Christchurch population in 1993 is associated with:

- (1237 ÷ 365 days) x 0.008 = 0.027 extra respiratory admissions per 10 µg/m$^3$ increase PM$_{10}$.
- (2845 ÷ 365 days) x 0.006 = 0.047 extra cardiac admissions per 10 µg/m$^3$ increase PM$_{10}$.
- (846 ÷ 365 days) x 0.019 = 0.044 extra asthma admissions per 10 µg/m$^3$ increase PM$_{10}$.


PM$_{10}$ concentrations on a year of relatively high pollution (1992) and relatively low pollution (1993) were chosen to represent a range in possible effects per year.

Each 10 µg/m$^3$ increase in daily PM$_{10}$ above 20 µg/m$^3$ is equivalent to no effect below a concentration of 30 µg/m$^3$. 

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Appendix 4: Performance standards for USEPA reference method for determining PM$_{10}$

The USEPA reference method for measuring PM$_{10}$ concentrations is a gravimetric method determining suspended particles selected by inertial separation. The particles undergo filtration and the remaining PM$_{10}$ mass on the filter substrate weighed. The performance standard specifications for this method for a reference PM$_{10}$ method are contained in table A2.

**Table A2: Performance standards for USEPA reference method for measuring PM$_{10}$ (from Chow, 1995)**

| Sampling Inlet | • A sampling inlet with a cut-point of 10 ± 0.5 μm aerodynamic diameter. The cut point shall be determined in a wind tunnel using liquid particles with aerodynamic diameters ranging from 3-25 μm and wind speeds of 2, 8, and 24 km per hour.  
  • No more than 5% of solid particles with aerodynamic diameters of 25 μm should penetrate the inlet at the 8 and 24 km/hr wind speeds.  
  • When the “sampling effectiveness” curve determined from these measurements is applied to an idealised size distribution$^{51}$ the mass of particles should be within ± 10 % of the mass collected by the ideal sampler.  
  • The flow rate over a 24-hour period shall remain within ± 5% of the initial reading for the average flow regardless of filter loading and within ± 10% of the initial flow rate for any instantaneous flow measurement. Sample volumes are adjusted to mean sea level pressure and 25°C.  
  • Measurement precision, determined by repeated collocated sampling, should be within ± 5 μg m$^{-3}$ for concentrations less than 80 μg m$^{-3}$ or ±7% of measured PM$_{10}$ for concentrations exceeding 80 μg m$^{-3}$ for a 24-hour period.  
  • The filter media should collect more than 99% of 0.3 μm particles, |

$^{51}$ Fine and course log-normal distributions with modes at 0.5 μm and 14 μm, respectively, geometric standard deviations of 2.0 in both modes, a coarse/ fine mass ratio of 3, and a total mass of 300 μg m$^{-3}$.
have an alkalinity of less than 25 microequivalents/gram, and should not gain or lose weight equivalent to more than 5 µg m\(^{-3}\), estimated from the nominal volume sampled over a 24-hour period.

- Prior to weighing filters should be equilibrated at a constant temperature, within ±3% between 15 °C and 30 °C, and at a constant relative humidity, within ±5% between 20% and 45%.

<table>
<thead>
<tr>
<th>A status of equivalent to the reference method can be achieved by a non-reference method sampler if it meets the following criteria:</th>
</tr>
</thead>
<tbody>
<tr>
<td>Three equivalence samplers collocated with 3 reference samplers for 10-15 days in 2 different locations give:</td>
</tr>
<tr>
<td>• differences in concentration within the larger of ±5 µg m(^{-3}) or ±7% of measure values; and</td>
</tr>
<tr>
<td>• correlation coefficients among sampled pairs of greater than 0.97.</td>
</tr>
</tbody>
</table>
Appendix 5: Emission factors for domestic solid fuel burning appliances

Emission factors are an estimate of the amount of contaminant (grams) resulting from the use of an appliance burning 1 kg of fuel. Emission factors for home heating will therefore be dependent on the type of fuel and the appliance it is being burnt, as well as other factors such as the operation of the appliance. Emission factors used in this research for different appliance types and fuel types were developed following a literature survey (USEPA, 1994; Economopoulos, 1993; Brady & Pullen, 1985; Todd, 1994) and with regard to installation standards for solid fuel burners specific to the Christchurch Clean Air Zones.

Clean Air Zone provisions have applied in central Christchurch since 1977 and most of the remaining urban areas of Christchurch since 1984. Restrictions on emissions from wood burners installed in these areas result in a more unique emission situation than could be expected in other cities in New Zealand. The emissions will vary primarily with respect to the installation standard to which an appliance was tested. Appliance testing criteria are outlined as follows:

- Prior to October 1988 there were no specific emission criteria to which solid fuel burners were required to meet. Emissions from solid fuel burners installed in Christchurch prior to 1989 are unlikely to differ from emissions from solid fuel burners installed in other areas during the same period.
- The Clean Air Council Standards (1987) applied from October 1988 and all appliances installed from this date were required to meet the test criteria specified in the standard. The criterion was specified in terms of gram per hour with an emission standard of 35 g/hr, average of 3 operating conditions (high, medium & low).
- In 1992 the joint NZ and Australian standard (Aus/NZ 7402-7404) for solid fuel burners was introduced. This was deemed an equivalent standard to the Clean Air Council (1987) requirements. Following its introduction appliances were tested to the joint NZ/Aus standard rather than the Clean Air Council (1987) criteria. NZ/Aus 7402-7404 specifies an emission criterion of 5.5 grams of particulate per kilogram of fuel burnt.

The emission test criteria and testing methods are designed to provide an indication of the comparative performance of appliances under repeatable test conditions. While the results of testing provide an indication of the performance capability of appliances under
ideal operating conditions they do not indicate performance under actual operating conditions. The latter is better reflected by the application of an average emission factor.

Emission factors for appliances installed prior to 1989 and for appliances installed post 1993 are based on emission factors derived by Todd (1994) for Tasmania for wood burners of the pre 1989 period and for wood burners meeting NZ/Aus 7403 respectively. Todd’s (1989) estimates of the percentages of incorrect operation were reduced from 80% to 70% for pre 1989 appliances and 60% for post 1993 appliances, in the derivation of the Christchurch emission factors, because of historical awareness and the reduced potential for incorrect operation of modern appliances.

The relationship between emission factors of the post 1993 appliances (Todd, 1994) and their total suspended particulate emissions from appliance test result was examined. This relationship, and the appliance test results for the 1989-1992 wood burners, was used to estimate emission factors for the 1989-1992 wood burners.

A summary of the home heating emission factors used in the emissions inventory and in subsequent calculations and the average fuel use for a 24-hour period are detailed in table A3.
Table A3: Emission factors for domestic heating appliances

<table>
<thead>
<tr>
<th>Appliance</th>
<th>Average PM₁₀ (g/kg)</th>
<th>CO (g/kg)</th>
<th>NOₓ (g/kg)</th>
<th>SOₓ (g/kg)</th>
<th>VOC (g/kg)</th>
<th>CO₂ (g/kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gas burner</td>
<td>1</td>
<td>0.1</td>
<td>0.4</td>
<td>2</td>
<td>0.01</td>
<td>0.2</td>
</tr>
<tr>
<td>Oil burner</td>
<td>4</td>
<td>1.3</td>
<td>0.6</td>
<td>2.2</td>
<td>3.8</td>
<td>0.25</td>
</tr>
<tr>
<td>Open fire - wood</td>
<td>14.6</td>
<td>15</td>
<td>120</td>
<td>1.6</td>
<td>0.2</td>
<td>30</td>
</tr>
<tr>
<td>Open fire - coal</td>
<td>10.3</td>
<td>33</td>
<td>60</td>
<td>1.5</td>
<td>18</td>
<td>15</td>
</tr>
<tr>
<td>Pre 1989 wood burner</td>
<td>14.9</td>
<td>12.8</td>
<td>104</td>
<td>1.4</td>
<td>0.2</td>
<td>26</td>
</tr>
<tr>
<td>1989-1992 wood burner</td>
<td>14.8</td>
<td>6.9</td>
<td>55</td>
<td>0.8</td>
<td>0.2</td>
<td>14</td>
</tr>
<tr>
<td>Post 1992 wood burner</td>
<td>14.7</td>
<td>5.9</td>
<td>47</td>
<td>0.7</td>
<td>0.2</td>
<td>12</td>
</tr>
<tr>
<td>New standard wood burner</td>
<td>14.7</td>
<td>2.9</td>
<td>23</td>
<td>0.3</td>
<td>0.2</td>
<td>6</td>
</tr>
<tr>
<td>Enclosed coal burner - wood</td>
<td>9.3</td>
<td>14.3</td>
<td>114</td>
<td>1.6</td>
<td>0.2</td>
<td>29</td>
</tr>
<tr>
<td>Enclosed coal burner - coal</td>
<td>13.5</td>
<td>31</td>
<td>57</td>
<td>1.4</td>
<td>18</td>
<td>14</td>
</tr>
<tr>
<td>Incinerator - wood</td>
<td>15</td>
<td>15.6</td>
<td>125</td>
<td>1.7</td>
<td>0.3</td>
<td>31</td>
</tr>
<tr>
<td>Incinerator - coal</td>
<td>12.5</td>
<td>34</td>
<td>62</td>
<td>1.6</td>
<td>28</td>
<td>16</td>
</tr>
<tr>
<td>Potbelly - wood</td>
<td>10.2</td>
<td>14.3</td>
<td>114</td>
<td>1.6</td>
<td>0.2</td>
<td>31</td>
</tr>
<tr>
<td>Potbelly - coal</td>
<td>10</td>
<td>34.3</td>
<td>62</td>
<td>1.6</td>
<td>18</td>
<td>16</td>
</tr>
</tbody>
</table>

Emission factors for wood burners meeting the test criterion of 1.5 g/kg were based on average test results (TSP) with consideration of the relationship between test results and emission factors for post 1993 appliances. This indicates emission factors of 1 g/kg for correctly operated burners and 4 g/kg for incorrectly operated burners. These compare reasonably well with Todd’s (1996) estimates of 1-1.5 g/kg and 2-3 g/kg for correctly and incorrectly operated “best available” wood burners.

One management option examines the effectiveness of education in reducing the percentage of incorrectly operated wood burners. Emission factors for wood burners with the percentage of incorrect operation reduced by 25% 50% and 100% are described in table A4.
Table A-4: Emission factors (PM$_{10}$) for wood burners based on reduced incorrect operation

<table>
<thead>
<tr>
<th></th>
<th>Incorrect operation reduced by 25%</th>
<th>Incorrect operation reduced by 50%</th>
<th>Incorrect operation reduced by 100%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pre 1989 wood burner</td>
<td>12.8</td>
<td>11.4</td>
<td>9.3</td>
</tr>
<tr>
<td>1989-1992 wood burner</td>
<td>6.9</td>
<td>6.3</td>
<td>5.7</td>
</tr>
<tr>
<td>Post 1992 wood burner</td>
<td>5.9</td>
<td>5.3</td>
<td>4.7</td>
</tr>
<tr>
<td>3 g/kg wood burner</td>
<td>4.5</td>
<td>4.0</td>
<td>2.5</td>
</tr>
<tr>
<td>New standard wood burner (1.5 g/kg burners)</td>
<td>2.9</td>
<td>2.5</td>
<td>2.0</td>
</tr>
</tbody>
</table>

Table A-5: Emission factors used to assess coincidental reductions in CO, NO$_x$ & SO$_x$ emissions.

<table>
<thead>
<tr>
<th></th>
<th>CO</th>
<th>SO$_x$</th>
<th>NO$_x$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gas</td>
<td>0.4</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Oil</td>
<td>0.6</td>
<td>3.8</td>
<td>2.2</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Wood</th>
<th>Coal</th>
<th>Wood</th>
<th>Coal</th>
<th>Wood</th>
<th>Coal</th>
</tr>
</thead>
<tbody>
<tr>
<td>Open fire</td>
<td>120</td>
<td>60</td>
<td>0.2</td>
<td>18</td>
<td>1.7</td>
<td>1.5</td>
</tr>
<tr>
<td>Incinerator</td>
<td>125</td>
<td>62</td>
<td>0.2</td>
<td>18</td>
<td>1.7</td>
<td>1.6</td>
</tr>
<tr>
<td>Potbelly</td>
<td>114</td>
<td>57</td>
<td>0.2</td>
<td>18</td>
<td>1.6</td>
<td>1.4</td>
</tr>
<tr>
<td>Coal range</td>
<td>114</td>
<td>57</td>
<td>0.2</td>
<td>18</td>
<td>1.6</td>
<td>1.4</td>
</tr>
<tr>
<td>Pre 89 wood burner</td>
<td>102</td>
<td>-</td>
<td>0.2</td>
<td>-</td>
<td>1.4</td>
<td>-</td>
</tr>
<tr>
<td>1989-1992 wood burner</td>
<td>55</td>
<td>-</td>
<td>0.2</td>
<td>-</td>
<td>0.8</td>
<td>-</td>
</tr>
<tr>
<td>Post 1992 wood burner</td>
<td>47</td>
<td>-</td>
<td>0.2</td>
<td>-</td>
<td>0.6</td>
<td>-</td>
</tr>
<tr>
<td>3 g/kg wood burner</td>
<td>59</td>
<td>-</td>
<td>0.2</td>
<td>-</td>
<td>0.4</td>
<td>-</td>
</tr>
<tr>
<td>New standard wood burner</td>
<td>22</td>
<td>-</td>
<td>0.2</td>
<td>-</td>
<td>0.3</td>
<td>-</td>
</tr>
</tbody>
</table>
Appendix 6: Weighting of emissions to allow for variations in meteorological conditions over a 24-hour period

There are distinct differences between the diurnal pattern of emission rates from the domestic, industrial and transport sectors. Because of changes in meteorological conditions over a 24-hour period, on a typical winter's day there is a large temporal difference in the ability atmosphere to disperse pollutants. Therefore the timing of emissions will have significant effects on 24-hour average concentrations.

The emission model was used to determine the relative effect of each sector's "time of day" emission variations. The effect of variances in the temporal distribution of emissions, over 24-hours, was compared for each sector. A single value of total emissions was entered into the model for one sector at a time, with the temporal distribution of this total value occurring as it would for each sector. The concentration outputs, for each sector, were compared and used to establish a weighting factor for each source. The weighting factor for each sector is illustrated in table A6.

Table A6: Weighting factors used to compare the effect on concentrations of emissions from the different sectors.

<table>
<thead>
<tr>
<th>Sector</th>
<th>Weighting factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Domestic home</td>
<td>0.46</td>
</tr>
<tr>
<td>heating</td>
<td></td>
</tr>
<tr>
<td>Transport</td>
<td>0.21</td>
</tr>
<tr>
<td>Industry</td>
<td>0.33</td>
</tr>
</tbody>
</table>

Weightings were based on the urban area of Christchurch and the estimated variations in emission with time of day as predicted using the emission inventory data. It was assumed that temporal variations in emissions from each sector would remain constant in the future.

The weighting factors are applied to the emissions estimates (grams) from each source as discussed in section 4.4.
### Appendix 7: 24-hour average PM$_{10}$ data by percentile

Table A-7 24-hour average PM$_{10}$ concentrations by percentile (1988-1996) and the reduction required to meet a 50 $\mu$g m$^{-3}$ air quality target

<table>
<thead>
<tr>
<th>Days exceeded</th>
<th>Percentile</th>
<th>Concentration ($\mu$g m$^{-3}$)</th>
<th>Reduction required %</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>99.7</td>
<td>189</td>
<td>74</td>
</tr>
<tr>
<td>2</td>
<td>99.5</td>
<td>158</td>
<td>68</td>
</tr>
<tr>
<td>3</td>
<td>99.2</td>
<td>143</td>
<td>65</td>
</tr>
<tr>
<td>4</td>
<td>98.9</td>
<td>126</td>
<td>60</td>
</tr>
<tr>
<td>5</td>
<td>98.6</td>
<td>120</td>
<td>58</td>
</tr>
<tr>
<td>6</td>
<td>98.4</td>
<td>113</td>
<td>56</td>
</tr>
<tr>
<td>7</td>
<td>98.1</td>
<td>108</td>
<td>54</td>
</tr>
<tr>
<td>8</td>
<td>97.8</td>
<td>104</td>
<td>52</td>
</tr>
<tr>
<td>9</td>
<td>97.5</td>
<td>98</td>
<td>49</td>
</tr>
<tr>
<td>10</td>
<td>97.3</td>
<td>92</td>
<td>46</td>
</tr>
<tr>
<td>11</td>
<td>97.0</td>
<td>88</td>
<td>43</td>
</tr>
<tr>
<td>12</td>
<td>96.7</td>
<td>85</td>
<td>41</td>
</tr>
<tr>
<td>13</td>
<td>96.4</td>
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Full data set is available from the Canterbury Regional Council.
## Table A-8: Estimated numbers of households with changes in home heating methods

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<th></th>
<th>Solid fuel burners removed</th>
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<th>New houses installing solid fuel burners</th>
<th>Solid fuel burners replacing open fires</th>
<th>Existing non solid fuel households converting to solid fuel burners</th>
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### Appendix 9: Estimated standard errors

#### Table A-9: Estimated standard error associated with input parameters

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<th>Variable</th>
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<th>Estimated standard error</th>
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<td></td>
<td>68% CI</td>
<td>95% CI</td>
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<td>No. of households with each solid fuel domestic heating method (from the emission inventory survey)</td>
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<td>Emission factors</td>
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<td>Average fuel use per night</td>
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<tr>
<td>Weighting factor</td>
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<td>9%</td>
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<tr>
<td>Transport emissions - inventory</td>
<td>33%</td>
<td>50%</td>
</tr>
<tr>
<td>Industrial emission – inventory</td>
<td>33%</td>
<td>50%</td>
</tr>
<tr>
<td>Projected households using solid fuel burning at 2020</td>
<td>30%</td>
<td>40%</td>
</tr>
<tr>
<td>Projected households using solid fuel burning for years n = 1997 - 2019</td>
<td>(n+1)-1%</td>
<td>(n+1)-1%</td>
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<tr>
<td>The number of households using older solid fuel burners that replace them with new solid fuel burners</td>
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<td>20%</td>
</tr>
<tr>
<td>The number of new houses installing solid fuel burners</td>
<td>5%</td>
<td>10%</td>
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<td>The number of existing houses with non-solid-fuel burners converting to solid fuel burners</td>
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<td>Transport projections</td>
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<tr>
<td>Industrial projections</td>
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Appendix 10: Air shed box model

Equation EA.1 describes a model derived by Gimson & Fisher (1997) of the relationship between PM$_{10}$ emissions and concentrations in the Christchurch air shed.

\[
C = \frac{1}{24} \sum_{j=1}^{24} C_j
\]

where

\[
C_{j+1} = \frac{100}{h_j} E_j + (1 - 0.41 u_j) C_j
\]

*Equation EA.1 Relationship between emissions and concentrations in Christchurch airshed*

- \(C\) is the 24-hour mean PM$_{10}$ concentration in $\mu$gm$^{-3}$
- \(C_j\) is the PM$_{10}$ concentration at hour \(j\)
- \(j = 0\) refers to 1400h, \(j=1\) refers to 1500h, and \(j= 24\) refers to 1400h the next day
- \(h_j\) is the inversion layer height (= mixing depth layer) in metres, between hours \(j\) and \(j+1\)
- \(E_j\) is the total emission, from all sources, in g/ha between hour's \(j\) and \(j+1\)
- \(u_j\) is the wind speed in m/s$^3$