Updated Health and Air Pollution in New Zealand Study

Volume 2: Technical Reports

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Introduction

This report contains all of the technical reports (in their entirety) that were prepared as part of the updated Health and Air Pollution in New Zealand (HAPINZ) study and used in the preparation of the main report and the health effects model.

It is intended as a companion to the *Updated Health and Air Pollution in New Zealand Study Volume 1 - Summary Report* for those readers who would like more detailed information on the methodology.

The technical reports included as appendices are as follows:

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Appendix 1: PM_{10} Exposure Assessment Methodology

Prepared by Emily Wilton (Environet Ltd), Gerda Kuschel (Emission Impossible Ltd) and Jayne Metcalfe (Emission Impossible Ltd)

Executive Summary

This appendix describes the methodology for assessing PM_{10} exposure, in terms of estimating annual average concentrations by CAU, used in the updated HAPINZ study.

Key Features of the Updated Methodology

1. **Using actual monitoring data for PM_{10} exposure, typically averaged for the period 2006-2008, where available**. The available data cover 83 per cent of the population living in urban areas and 73 per cent of the population overall.

2. **For unmonitored areas, estimating annual concentrations based on comparisons with monitored areas with the same urban/rural classification**.

3. **Correcting PM_{10} data for gravimetric (HiVol) equivalency** based on a combination of known relationships (applies to areas with 84 per cent of the monitored population) and estimated relationships (remaining 16 per cent monitored population). These corrections were applied to the three-year annual averaged data.

4. **Undertaking sensitivity analyses for equivalency** for those locations without known relationships. The base case being all data adjusted for HiVol equivalency (using the default correction factor of 1.18) with a low case of 0.85 (=1.0/1.18 assuming no adjustment needed) and a high case of 1.15 (assuming more adjustment needed).

5. **Allocating sources by estimating natural source concentrations (from source apportionment studies where available) and then attributing the remaining (anthropogenic) concentrations by the emissions inventory proportions for domestic fires, motor vehicles, industry, open burning and other major sources (if applicable)**.

A1.1 Introduction

A1.1.1 Exposure Assessment Approach

Mortality impacts of particulate exposure are calculated based on annual average PM_{10} or $PM_{2.5}$ concentrations. The reason for this is because the majority of the health effects owing to PM_{10} or $PM_{2.5}$ occur as a result of prolonged exposure, as opposed to peak events. This is demonstrated in the difference between dose-response relationships for mortality based on time series studies (typically around 1 per cent per 10 $\mu g/m^3$ increase in daily PM_{10}) compared with more than 4 per cent increase in baseline mortality from the longitudinal cohort studies comparing impacts of annual average concentrations.

Historically the approach used for risk assessments has been to estimate effects based on dose/exposure-response relationships and concentrations measured at ambient air quality monitoring sites within urban areas. This approach is appropriate because it is consistent with the way the dose-response relationships have been derived. Potential inaccuracies associated with some higher and lower exposures are likely to balance out as they would have in the derivation of the dose/exposure-response relationships.

It is noted, however, that the term 'dose-response' is probably less accurate in this respect than the term 'exposure-response' or even 'concentration-response'. We use the term 'exposure-response' in this study. In this sense 'exposure' is meant to refer to the exposure to ambient particulate concentrations and does not take into account localised influences on personal exposure. As indicated above this is appropriate given the way the exposure-response relationships are derived. In addition to the dearth in exposure data from localised sources, no exposure-response relationships are available to assist with characterising impacts at this finer scale.

The exposure-response relationships derived from the longitudinal cohort studies are based on concentrations of particulate measured using gravimetric sampling (ENHIS 2007). In New Zealand particulate monitoring is carried out using a variety of methods including gravimetric (reference method), BAM (reference method equivalency status), TEOM (reference method equivalency status) and TEOM/FDMS (reference method equivalency status). Bluett *et al*. (2007) demonstrates that both the BAM and the TEOM tend to under-measure PM_{10} relative to gravimetric methods. The extent of under-representation appears to vary with location with some areas showing BAM concentrations under-measuring by nearly 30 per cent (e.g., Wilton & Baynes 2010). Because the exposure-response relationships are based on gravimetric methods, using unadjusted PM_{10} data results in an underestimate of the effects of PM_{10}. Adjusting all PM_{10} data for gravimetric equivalency is therefore a recommendation of ENHIS (2007) when preparing health risk assessments.

A1.1.2 Air Quality Data for New Zealand

In New Zealand air quality monitoring data are now available for more than 40 urban areas. These range in size and location from Bluff to Auckland. In some locations PM_{10} data are available for a number of monitoring sites. Areas without monitoring data are typically small in size and have generally been deemed by regional councils as being lesser risk of experiencing elevated PM_{10}.

The key factors which influence PM_{10} concentrations are considered by regional councils in making these evaluations. These include:

- Emissions from anthropogenic sources (typically domestic home heating, motor vehicles, industry and open burning)
- Natural sources of PM_{10} (sea spray, windblown dust, volcanoes etc.)
- Meteorology and topography

A1.2 Review of Original HAPINZ Methodology

A1.2.1 Description of Original Exposure Assessment Method

The method used to estimate exposure for the original HAPINZ assessment (Fisher *et al.* 2007) was a land based regression (LBR) model. The approach used an emission estimate for each census area unit (CAU) which was based on:

- the number of houses using wood and coal from census data
- the vehicle kilometres travelled (VKT) estimates
- a combination of real and proxy industrial PM_{10} emissions data

The emission estimate was converted into a concentration using the relationship between emissions and concentrations for Christchurch (as modelled using TAPM[1]) and some other indicators of the potential relationship between emissions and concentrations (e.g., slope of topography and number of adjoining CAUs). An estimate of the natural sources contribution was also made. The original HAPINZ study did not adjust any PM_{10} monitoring data collected by either a BAM or a TEOM for gravimetric equivalency.

A good correlation was found between the LBR model PM_{10} concentrations and estimated concentrations in the 31 urban towns where PM_{10} monitoring data were sourced (reported as $r^2 = 0.86$). The method was published in at least two papers (Kingham & Fisher 2007, Kingham *et al.* 2007).

A1.2.2 2010 Evaluation of the Original LBR Model

A review of the original LBR model was carried out for this proposal to establish whether it could be applied directly for our 2010 update. Not surprisingly, since the release of the original HAPINZ study, more data are now available to better characterise concentrations in the urban areas included. Historical data were evaluated[2] and updated where appropriate and additional monitoring data from urban areas not previously included in original LBR model were obtained.

Firstly, the data were reviewed for those urban areas which had monitoring information available for the original HAPINZ study. Table A1-1 compares the currently reported PM_{10} information for 2001 with the data reported in original HAPINZ for urban areas which had monitoring data at that time. The results are not consistent with some urban areas recording significantly lower concentrations whilst others were significantly higher than

[1] The Air Pollution Model (TAPM) is a software package developed by the Commonwealth Scientific and Industrial Research Organisation (CSIRO) to estimate the spread and impact of air pollution. It is used by more than 190 national and international users in 25 countries.

[2] A number of the annual averages included in the model were biased seasonally towards winter months, e.g., calculating an annual average based on data from January to August or based on a higher frequency sampling rate undertaken in winter as opposed to non-winter months.

the original HAPINZ concentrations. Potential reasons for these differences include improved data availability, bias in averaging periods (e.g., reporting an annual average based on winter concentrations alone) and subsequent adjustments by Councils for gravimetric equivalency or quality assurance.

Table A1-1: Urban areas with revised annual PM_{10} averages for 2001

Urban Area	2001 Pop'n	HAPINZ 2001 PM_{10} Reported[3] ($\mu g/m^3$)	HAPINZ 2001 PM_{10} Monitored[4] ($\mu g/m^3$)	Revised PM_{10} Monitored[5] ($\mu g/m^3$)	% Change
Alexandra	4,407	25.9	25.0	24	-9.1%
Blenheim	21,195	18.1	17.0	16	-13.0%
Christchurch Inner	132,706	24.7	25.0	24	-4.6%
Christchurch Outer	183,512	17.4	18.0	23	31.2%
Dunedin	82,284	15.5	16.0	20	30.5%
Geraldine	2,205	17.2	17.0	19	10.5%
Gisborne	28,992	10.7	9.0	10	-3.7%
Gore	7,665	19.1	19.0	16	-14.9%
Hamilton	114,171	14.7	15.0	16	5.4%
Invercargill	41,964	19.7	18.8	20	-0.1%
Masterton	17,514	22.1	19.0	15	-33.4%
Napier	49,851	10.3	11.9	16	55.3%
Taupo	16,935	15.1	15.0	17	13.4%
Te Kuiti	4,374	15.3	15.0	18	15.6%
Timaru	24,732	21.2	20.0	28	31.1%
Tokoroa	14,019	21.5	23.0	18	-18.6%
Upper Hutt	32,904	18.6	14.5	12	-36.1%
Waimate	2,757	17.0	15.0	17	-2.0%
Total	**782,187**				

[3] From Table 1 in HAPINZ Report Appendices.
[4] From HAPINZ Exposure Calculations spreadsheet
[5] Based on an evaluation of best available data.

The following outline two examples explaining how and why data have been revised:

Example 1 Te Kuiti

HAPINZ reported an annual average concentration for Te Kuiti of 15.0 µg/m^3 based on 1998 data, as this was the only information available prior to 2003. The 1998 data were for 22 April – 4 November. Adjustment for seasonality was made but was limited particularly for the first part of the year (pre-May data were limited to 22-30 April). The PM_{10} data were collected using a TEOM and were not adjusted for gravimetric equivalency. Since 2003, PM_{10} has been measured in Te Kuiti continuously using an FH62 BAM. An evaluation of the relationship between the BAM and gravimetric methods suggests no adjustments of PM_{10} are required. The annual average concentration each year from 2003 to 2009 was either 17 or 18 µg/m^3 and the average over the whole period (and for just the period 2003-2005) is 17.7 µg/m^3. This represents a 16 per cent increase over the concentration used in the original HAPINZ evaluation.

Example 2 Timaru

Environment Canterbury have re-evaluated historical PM_{10} data for Timaru by comparing concentrations measured by the TEOM alone with those measured for the TEOM with the FDMS system (which captures the volatile PM_{10} not measured using the TEOM alone). The data were subsequently adjusted to account for this proportion of PM_{10} concentrations (mostly low molecular weight organic compounds) that were not previously allowed for in PM_{10} reporting. The adjustment equations were:

- FDMS = (TEOM@40+3.15)/0.75 for TEOM concentrations > 44 µg/m^3
- FDMS = (TEOM@40-2.23)/0.74 for TEOM concentrations < 44 µg/m^3

This accounts for the main difference in the HAPINZ 2001 monitoring concentrations for Timaru (reported as 20 µg/m^3) and the Environment Canterbury reported annual average concentration for Timaru of 28 µg/m^3 for 2001. This represents a 31 per cent increase over the concentration used in the original HAPINZ evaluation.

Secondly, the data were reviewed for those urban areas which relied on modelling in the original HAPINZ but for which monitoring estimates are now available. Table A1-1 compares the currently reported PM_{10} information for 2001 with the monitoring data reported in original HAPINZ for a number of urban areas. Table A1-2 compares the currently reported PM_{10} information for 2001 with the data reported in original HAPINZ for urban areas which did not have monitoring data at that time. Again, the results are not consistent with some urban areas recording significantly lower concentrations whilst others were significantly higher than the original HAPINZ concentrations used.

As a final check, data from new sites (now with monitoring estimates) were combined with that for the existing sites (using revised data where appropriate) and checked against the original LBR predictions, yielding a revised r^2 of 0.4 (compared with 0.86 reported in

Kingham *et al*. 2007). A comparison was also made of the HAPINZ predicted PM_{10} concentrations (annual average) to the measured PM_{10} concentrations for areas where monitoring hadn't been carried out or included. Results indicated the performance of the model in predicting PM_{10} concentrations in areas not previously monitored was also extremely poor (r^2= 0.09).

Table A1-2: Estimated 2001 annual average PM_{10} concentrations in areas where monitoring data were not previously available

Urban Area	2001 Pop'n	HAPINZ 2001 PM_{10} Reported[6] ($\mu g/m^3$)	Estimated 2001 PM_{10} Monitored[7] ($\mu g/m^3$)	% Change
Arrowtown	1,689	19.5	23.4	20%
Ashburton	14,202	19.5	21.6	11%
Balclutha	4,104	16.0	16.0	0%
Cromwell	2,667	23.8	17.7	-26%
Hastings	42,297	10.3	18.8	83%
Matamata	6,078	16.2	13.3	-18%
Milton	1,920	19.6	25.6	31%
Mosgiel	6,342	21.4	16.6	-22%
Oamaru	11,085	21.1	18.6	-12%
Palmerston North	70,836	12.7	12.7	0%
Putaruru	3,783	14.7	16.3	11%
Rangiora	8,607	19.7	19.0	-4%
Reefton	987	25.4	19.7	-22%
Richmond	10,578	18.1	22.0	22%
Wainuiomata	16,602	12.7	12.1	-5%
Westport	3,783	20.1	17.5	-13%
Whangarei	40,284	18.1	15.3	-15%
Winton	2,100	27.0	15.2	-44%
Total	**62,769**			

[6] From Table 1 in HAPINZ Report Appendices.

[7] Data are for a variety of years depending on what is available in each location. Where more than one year of data is available, averages (typically over three years) have been used to reduce meteorological impact. In most locations significant reductions in emissions will not have occurred from 1996 – 2009. Consequently monitoring data are likely to provide indicative annual average concentrations.

As a consequence, the overall conclusion was that the original LBR model should not be directly applied to the update.

Further evaluation was given to the allocation of PM_{10} concentrations by source. In the original LBR model, emissions from domestic heating and motor vehicles were estimated and these were converted to PM_{10} concentrations based on a TAPM estimated relationship for Christchurch. These were combined with concentration estimates from natural sources and industry to give the modelled annual average PM_{10} concentrations by source. It is uncertain how emissions were allocated by season. The resulting source allocations from the original HAPINZ study differ from proportions that have now been obtained using source apportionment techniques. A comparison of the source allocations using the LBR model and the proposed alternative method is covered in more detail in section A1-3.

A final concern is the application of the model to small towns and rural areas. There is inevitably a certain amount of guess work involved in establishing probable concentrations in small urban areas and rural areas because of the lack of monitoring data. The model aimed to fill this gap but the derivation method relied on relationships in higher density emission areas and the applicability to small urban towns and rural areas was essentially untested. Given the poor correlation observed in the medium to small size towns for which monitoring data are available, it was our conclusion that the model was not the best method to predict PM_{10} concentrations.

It should be noted that, at the time of preparation, the original HAPINZ study utilised the most current information available. It is highly likely that had the additional monitoring data and source apportionment results we now have at our disposal been available the original HAPINZ team would have been able to develop a more robust LBR model that could potentially have been used in the update.

A1.3 Methodology for the HAPINZ Update

A1.3.1 General Approach

Where available, PM_{10} monitoring data were used. These data were averaged monthly and where a full year of data was unavailable an annual average was derived based on an appropriate weighting[8] of winter and non-winter data.

Sources were assessed on a monthly basis as the relative contributions vary with season (e.g., domestic heating is greater during the winter). Meteorological conditions also vary with season (conditions inhibiting dispersion are more prevalent during the winter months). The relative contributions of sources to monthly average PM_{10} concentrations were determined and these contributions were averaged for a year to provide a more robust assessment of the resulting contributions to annual average concentrations.

[8] The weighting method was done at a monthly level – the exact details of what assumptions were made for each urban area are detailed in the Monitoring Data worksheet.

To assess the monthly contributions of sources, emissions from domestic heating, open burning and motor vehicles were estimated for each CAU based on a) inventory data if available and b) proxy data using Ministry of Transport VKT data (motor vehicles)[9], household wood and coal use data (domestic heating), household numbers and inventory derived relationships (open burning).

An evaluation of New Zealand source apportionment (receptor modelling) studies was carried out to identify the contribution of natural sources to PM_{10} concentrations in urban areas of New Zealand. This showed an average natural source contribution of 6.8 $\mu g/m^3$ which was used to determine approximate natural source contributions to PM_{10} in coastal and inland areas and smaller and larger towns. Average seasonal profiles were also established to account for seasonal variability. An estimate of the natural sources contribution by month of the year was then be made for each CAU.

The methodology for areas where there were no monitoring data varied depending whether the remaining CAUs were classified as urban or rural according to the Statistics New Zealand (2006) classifications detailed in Table A1-3.

A1.3.2 Adjustments of PM_{10} Data for Gravimetric Equivalency

It is international best practice that PM_{10} data be adjusted for gravimetric equivalency when preparing health risk assessments (ENHIS 2007).

The relationship between the gravimetric reference method and BAM or TEOM data was unavailable for all areas of New Zealand where monitoring has been conducted. While ENHIS (2007) provides a generic equation for adjustment of data in cases where relationships are unavailable, the equation is based on European data and sources and is unlikely to be applicable to New Zealand. Moreover, in the case of BAM adjustments, evaluation of existing relationships shows significant variations within New Zealand in the adjustments that would be required (e.g., Bluett *et al*. 2007). A case in point is the analysis undertaken by Environment Waikato which suggests BAMs under-represent PM_{10} concentrations by a factor of around 25 per cent in Taupo but that no adjustments are required in Te Kuiti.

In the exposure assessment PM_{10} data were adjusted for gravimetric equivalency for the purposes of calculating health impacts. Where site specific data were available on the relationship between reference methods and equivalent methods data were adjusted using this relationship. An average relationship was derived for each monitoring method (e.g., BAM, TEOM) and was applied for locations where no relationship existed, typically 1.18. In three areas where very low PM_{10} concentrations were found it was considered inappropriate to adjust PM_{10} data based on the average relationship. No adjustments were made in these areas. Around 16 per cent of the monitored population resided in areas

[9] Traffic count data by CAU which is then corrected to align with NZTA Motor Vehicle Registration statistics data

where these estimates were required. A sensitivity analysis was integrated into the model to include an estimate of health effects for such locations based on a lower limit (0.85=1.0/1.18 for no adjustment) and an upper limit (1.15=1.36/1.18 for increased adjustment) with all sites adjusted being the base case.

Annex A summarises the monitoring methods for different areas and identifies which areas had co-location data available, whether adjustments were required, the corrections applied and the population bases.

A1.3.3 Urban Areas with Monitoring Data

As discussed above mortality estimates from PM_{10} exposure form the basis of the assessment. The exposure estimates were based on annual average PM_{10} concentrations which were averaged over a period of up to three years, where possible. Although the HAPINZ updated assessment uses a base year of 2006, averaging over three years was appropriate to account for year to year variations in concentrations occurring as a result of meteorological variations. Where available, air quality monitoring data for the years 2006 to 2008[10] were used. In some locations only intermittent monitoring data were available (e.g., one or two years or winter only data or limited sampling days). These data can provide a better indication of likely annual averages than emissions based estimates and were used in the absence of specific data or more complete datasets for 2006 to 2008.

Where data were available for only one year in the 2006-2008 period but were available for an alternative year(s), e.g., 2009, annual averages were estimated based on two years of data including one year outside of the 2006-2008 period. If data were only available for years outside of 2006-2008 these data were used.

Existing Statistics New Zealand urban-rural classifications were used to classify CAUs as urban or rural[11].

An estimate of emissions for each CAU within the urban areas was made based on the emissions assessment approach described previously. Similarly, an estimate of the natural sources contribution in each CAU was made based on the method determined as a result of the source apportionment evaluation.

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[10] It is noted that the severe and on-going Christchurch earthquakes have likely influenced PM_{10} concentrations in the affected CAUs since late 2010, due to dust caused by building collapses and liquefaction as well as changes in home heating emissions caused by restrictions in electricity supply or chimney collapses. However, because the updated HAPINZ is based on the years 2006 and utilises data for 2006-2008, this major event has no impact on the results. Future evaluations of air pollution health effects in Christchurch will need to carefully re-assess the PM_{10} concentrations to see whether the overall outcome of the earthquakes is a benefit or lag to the plans already put in place by Environment Canterbury to improve PM_{10} concentrations.

[11] See http://www.stats.govt.nz/census/about-2006-census/2006-census-definitions-questionnaires/definitions/geographic.aspx for details

In urban areas, the relationship between emissions and concentrations is mainly influenced by the meteorology and topography. The most appropriate method for assessing this is atmospheric dispersion modelling of all sources. However, this approach was beyond the resources available for this project.

Table A1-3 shows that, as a consequence of the increase in air quality monitoring sites since 2001, the majority of the population likely to be exposed to elevated PM_{10} concentrations (those in urban areas) reside in locations where PM_{10} monitoring has been conducted. The original HAPINZ study had monitoring data available for 31 of the 67 urban areas reported in that study. Data are now available for 58 of the 67 urban areas reported in that study. In 2001, around 56 per cent of the population resided in urban areas for which monitoring data for PM_{10} were available. For our update based on 2006, the monitoring information now covers 83 per cent of the resident urban population.

A number of different methods were used to determine the 'exposure area' to which the monitoring data were applied. These included:

- Regression of PM_{10} concentrations versus emission density for urban areas represented by a number of monitoring sites and where a good correlation was observed between the emission density in the CAU where the monitoring was carried out and the measured PM_{10}. This method was applied in Auckland (r^2=0.63), Invercargill (r^2=0.78) and Rotorua (r^2=0.94). Extrapolation above the highest measured concentration was not allowed.

- Applying the annual average concentration to the area included in the airshed or emission inventory. Areas outside of these classified as urban 1 were evaluated for each urban area using the 'emissions' spatial emission density model (http://wrenz.niwa.co.nz/webmodel/emissions). Of these areas, those with higher emission densities or for other reasons likely to experience high concentrations (e.g., downwind of the main urban area) were included in the 'exposure area'.

A small number of exceptions to the above approaches were used in areas where topography was complex and monitoring data were limited. These were:

- **Tauranga:** Two monitoring sites were used to represent concentrations in the immediate and adjoining CAUs and an average of the two sites were used to represent PM_{10} concentrations in the remaining urban CAUs based on statistics New Zealand urban 1 classifications with the exception of Ohauiti-Ngapeke which was allocated rural PM_{10} concentrations because of the nature of the area and the low emission density.

- **Wellington**: The one monitoring site in the Wellington airshed was a roadside site and unlikely to be representative of PM_{10} exposure beyond the immediate roadside area. Alternative approaches were evaluated including using the lowest PM_{10} concentrations for other Category 1 urban areas in the region and historical PM_{10} monitoring in Wellington. Both approaches resulted in a proxy annual average concentration of 11 $\mu g/m^3$. This value was applied as an average to all Category 1 urban area CAUs for Wellington.

Table A1-3: Summary of populations where PM_{10} data are available by urban area classification

Category	Classification	Description	Total Pop'n	Pop'n in Monitored Areas	% in Monitored Areas
1	Urban	Main urban area	2,855,406	2,557,119	90%
2	Urban	Secondary urban area	275,373	194,031	70%
3	Urban	Minor urban area	331,713	129,066	39%
4	Rural	Rural area with high urban influence	81,510	8,967	11%
5	Rural	Rural area with moderate urban influence	483,075	50,502	10%
6	Rural	Rural area with a low urban influence	825	15	2%
Summary	Urban areas		3,462,492	2,880,216	83%
	Rural areas		565,410	59,484	11%
	Total		**4,027,902**	**2,959,700**	**73%**

A1.3.4 Urban Areas without Monitoring Data

Category 1 Urban Areas

An evaluation was made of CAUs classified as Category 1 (main urban areas) not represented by monitoring data. Unmonitored areas typically fell into one of three categories:

- Part of a main urban area but excluded from the analysis of exposure for this work because of the topography. For example, those residing in the Cashmere CAUs in Christchurch are unlikely to be exposed to typical Christchurch PM_{10} concentrations.
- An urban area where comprehensive monitoring for PM_{10} has not been undertaken because the Council has assessed it to be unlikely to experience high PM_{10} concentrations. For example, Wanganui is a reasonably large urban area with a population of around 35,000. Investigative monitoring carried out by Horizons Regional Council during the winter of 2001 suggests PM_{10} concentrations are unlikely to be elevated because meteorological conditions are very conducive to dispersion in Wanganui.
- Based on these observations and an evaluation of monitoring data and domestic heating emission densities in these areas, it seemed likely that in areas where

dispersion is typically good, annual average PM_{10} concentrations would be around 10-12 µg/m^3. A value of 10 µg/m^3 was used for the annual average PM_{10} concentrations in these areas with the following exceptions:

- Christchurch - because of the higher emission density and the potential for downwind exposure from the main urban areas
- Tauranga - because of the higher emission density within the areas outside of the 'exposure area'

Table A1-4 shows the different values used for Category 1 urban areas.

Table A1-4: Summary of annual average PM_{10} concentrations assumed for Category 1 urban areas outside of the designated exposure area

Category 1 Urban Area	PM_{10} Concentration Used	Basis
Christchurch zone 1	17	Lowest of area 1,2 or 3 concentrations in Canterbury
Tauranga	12	Average of concentrations from two monitoring sites
Other areas	10	Evaluation of all monitoring data

Category 2 Urban Areas

Category 2 urban areas are typically:

- medium sized urban areas, such as Greymouth and Fielding
- smaller urban areas located near to other urban areas, such as Weston which is located four kilometres inland from Oamaru

Table A1-3 shows that representative monitoring data were available for 70 per cent of the Category 2 urban areas. For a number of the medium sized urban areas indicative or investigative data were available but were not included in this study because of issues of completeness, time period or monitoring method. On-going monitoring in these areas has not been carried out by Councils because elevated concentrations are not indicated as a result of investigative studies.

The following methods were used for estimating annual average PM_{10} concentrations in Category 2 urban areas.

- For medium sized Category 2 areas for which there was some investigative monitoring, annual average PM_{10} concentrations were estimated based on the investigative monitoring.
- For Category 2 areas that were monitored but included low emission density CAUs, typically on the outskirts of town, concentrations were estimated for the

outside areas based on the lowest concentration from an urban area 4 or 5 CAU nationally. This was 9.2 µg/m^3 measured in Pongakawa.

- For remaining Category 2 areas, use the lowest annual average for a monitored Category 1, 2 or 3 urban area within the Region[12].

Table A1-5 shows the different values used for Category 2 urban areas.

Table A1-5: Summary of annual average PM_{10} concentrations assumed for unmonitored Category 2 urban areas

Category 2 Urban Area	PM_{10} Conc'n Used	Basis
Tuakau	15	Assume same as Pukekohe
Mangakaretu (near Tokoroa)	9	Used Category 3 concentration and distribution
Taupo outside area most influenced	9	Low housing density - treat as rural
Whakatane outside area most influenced	9	Low housing density - treat as rural
Hawera	14	Lowest of Category 1,2 or 3 concentrations
Feilding	13	MiniVol June to August 2003 (assumed 10 for non-winter months)
Levin	11	MiniVol 42 days around August 2001
Masterton outside exposure area	9	Low housing density - treat as rural
Greymouth	16	Lowest of Category 1,2 or 3 concentrations
Ashburton outside exposure area	9	Low housing density - treat as rural
Timaru outside exposure area	9	Low housing density - treat as rural
Oamaru outside exposure area	9	Low housing density - treat as rural
Gore outside exposure area	9	Only one CAU and very rural so treat as rural background

Category 3 Urban Areas

Category 3 urban areas are small urban towns typically comprising just one CAU but occasionally more. Examples include Alexandra, Cromwell, and Winton in the South Island and Carterton, Te Kuiti, and Putaruru in the North Island.

[12] The rationale for using the lowest concentration is that on-going monitoring in these areas is not included because they have been deemed by Councils to be less likely to have elevated concentrations than areas regularly monitored.

In Otago, most of the Category 3 urban areas have been monitored because of the higher propensity for elevated PM_{10} concentrations in small urban areas in that region. The Waikato Region has around 20 Category 3 urban areas and has monitored PM_{10} in seven of them. Annual average concentrations in the areas monitored range from 9 $\mu g/m^3$ (Turangi, population 3,240) to 17 $\mu g/m^3$ (Putaruru, population 3,768).

The annual average PM_{10} concentrations for Category 3 towns without monitoring data were estimated based on the lowest annual average for the Category 3 towns for the region. For example, all unmonitored Category 3 towns in the Waikato would be based on Turangi's annual average value of 9 $\mu g/m^3$. The values used for the Category 3 towns in each region are shown in Table A1-6.

Table A1-6: Summary of annual average PM_{10} concentrations assumed for unmonitored Category 3 urban areas

Category 3 Urban Area	PM_{10} Conc'n Used	Basis
Northland	11	Lowest of Category 1,2 or 3 concentrations
Auckland	11	Lowest of Category 1,2 or 3 concentrations
Waikato	9	Lowest of Category 1,2 or 3 concentrations
Bay of Plenty	12	Lowest of Category 1,2 or 3 concentrations
Taranaki	14	Lowest of Category 1,2 or 3 concentrations
Hawkes Bay	13	Used Taihape as no monitoring in area 2 or 3s
Gisborne	9	Lowest of Category 1,2 or 3 concentrations
Horizons	13	Lowest of Category 1,2 or 3 concentrations
Wellington	11	Lowest of Category 1,2 or 3 concentrations
Tasman	9	Used lowest of monitoring across Tasman, Nelson and Marlborough
Nelson	9	Used lowest of monitoring across Tasman, Nelson and Marlborough
Marlborough	9	Used lowest of monitoring across Tasman, Nelson and Marlborough
West Coast	16	Lowest of Category 1,2 or 3 concentrations
Canterbury	17	Lowest of Category 1,2 or 3 concentrations
Otago	11	Lowest of Category 1,2 or 3 concentrations
Southland	7	Lowest of Category 1,2 or 3 concentrations (excl Te Anau)

A1.3.5 Rural Areas

In rural areas, ambient air concentrations of PM_{10} are likely to be low because of the lower emission density. Exposure to PM_{10} in rural areas is likely to be dominated by natural source contributions, open burning and in some cases industry and transport from nearby

urban areas. These will vary from area to area depending on factors such as proximity to the coast, prevalence of dust sources (e.g., gravel river beds, exposed soils) and agricultural practices (e.g., stubble burning).

Concentration estimates for rural areas were based primarily on the natural sources contribution estimation methodology and industrial emissions for major dischargers. Table A1-7 shows monitored PM_{10} concentrations in areas designated as Category 4 and 5 rural areas based on Statistics New Zealand urban area classifications.

Table A1-7: Summary of monitored annual average PM_{10} concentrations in Category 4 and 5 rural areas

Rural Area	PM_{10} Annual Average	Rural Category
Pongakawa	9	5
Ranfurly	14	4
Nasby	11	5
Clyde	20	4
Edendale	10	4
Wallacetown	12	4

Unpaved roads may be a significant source of PM_{10} emissions in some locations. However, relationships have not been adequately characterised to allow this source to be integrated into the assessment.

Rural CAUs were allocated an annual average PM_{10} concentrations based on the results lowest measured annual average PM_{10} concentrations in a rural area with an additional industry contribution for those CAUs with a significant industry contributor. This gives a concentration of 9.2 $\mu g/m^3$ based on monitoring at Pongakawa. The background contribution based on the average of source apportionment studies is 6.8 $\mu g/m^3$ giving around 2.4 $\mu g/m^3$ to be allocated to anthropogenic sources.

In all areas where monitoring data were not used to estimate PM_{10} concentrations, industrial concentrations have been added to the values indicated in this section where these have been able to be estimated (see section A1.3.6). The industrial contributions have been assumed to apply only to the CAU in which the discharge occurs.

A1.3.6 Source Allocation

Once PM_{10} concentrations have been collated or estimated they need to be disaggregated by source so that the health effects likely to be attributable to specific sources can be assessed.

The method used to allocate PM_{10} concentrations by source by month in the update was as follows:

1. Estimate the **total monthly** PM_{10} contribution in $\mu g/m^3$

   These are generated from the Exposure Model as discussed previously.

2. Subtract the estimated monthly **natural sources** contribution in $\mu g/m^3$

   This is discussed in more detail in Appendix 2 which follows are based on source apportionment studies.

3. Subtract the estimated monthly contribution from **industry with 'tall stacks'** in $\mu g/m^3$ to determine the remaining monthly PM_{10} concentration that needs to be allocated between the other anthropogenic sources

   Emissions from industry with 'tall stacks' have unique dispersion characteristics relative to ground level sources and were handled separately. This is discussed in the section which follows on industry.

4. Estimate the **monthly emissions in $kg/km^2/day$ for the other anthropogenic sources** - domestic fires, motor vehicles, open burning and industry (excluding those with 'tall stacks')

   The methodology adopted for each source is discussed in the next sections which follow.

5. Allocate the remaining **monthly PM_{10} concentrations by each of these sources** based on their contribution to monthly emissions, e.g.

$$\text{Domestic fire concentration } (\mu g/m^3) = \frac{\text{Domestic fire emissions } (kg/km^2/day)}{\text{Total anthro less industry with tall stacks emissions } (kg/km^2/day)}$$

Table A1-8 provides an example of how this was done in practice for the Penrose, Auckland CAU.

If the averaged monitored PM_{10} concentrations for a given month was lower than the default natural source contribution for that month, the lower value was be used and assumed to be the natural source contribution.

The resulting output is a concentration (in $\mu g/m^3$) of PM_{10} by source for each month of the year. The annual average contribution for each source was then estimated by averaging the source concentrations for each month of the year.

Natural sources

Natural emissions were based on source apportionment data as discussed in detail in Appendix 2.

Industry

Industry emissions were separated into two categories – those denoted as having 'tall stacks' and those without.

The **'tall stack' industry** category was created to separate out the most major industries across New Zealand Industries to better account for their dispersion characteristics. Most of these sites emit from stacks that are much higher than ground level (typically above 20 metres). The industries included:

- combustion processes (coal-fired) included in the Ministry for the Environment's 2008 SO_2 emission inventory (MfE, *pers. comm.* 2011)
- other processes included in the 2008 SO_2 emission inventory for which PM_{10} data were supplied
- other industry with the potential for high PM_{10} emissions involving stacks higher than 25 metres for which information could be readily obtained.

The mass emissions (kg/day) for the each of the above sources was multiplied by a dispersion factor derived from modelling of SO_2 from industrial discharges for 24-hour averages (MfE, *pers. comm.* 2011). For sources included in the SO_2 inventory but without the site specific modelling result, a proxy dispersion factor based on Figure A1-1 was used.

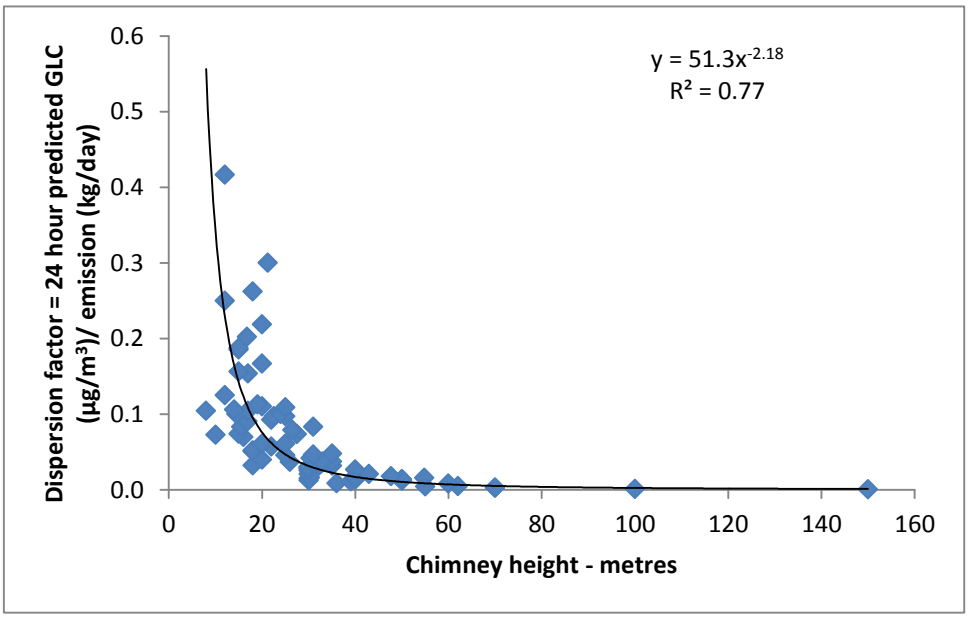

Figure A1-1: Relationship between chimney height and dispersion* of SO_2 emissions

* dispersion calculated as 24-hour average predicted GLC (µg/m^3) divided by daily emissions (kg/day) from MfE 2008 SO_2 emissions concentrations assessment (MfE pers comm, 2011)

The resulting 24-hour maximum ground level concentrations were converted to annual averages using modelling data from a range of resource consent applications. For the purposes of this study, this approach is referred to as the 'proxy dispersion modelling' approach.

The mass emissions (kg/day) for the "tall stack' industry were then subtracted from the emissions inventory estimates for all industry in the relevant CAU to get mass emissions for **industry (excluding 'tall stacks)**. These mass emissions were then used to arrive at concentrations as per Step 4 of the allocation method.

The **all industry** concentrations were the sum of the "tall stack' industry and industry (excluding 'tall stacks') concentrations.

Domestic fires

Emission estimates for domestic fires were based on inventory data distributed down to CAU level based on numbers of households using wood and coal from census data. Where inventory data were unavailable, the emissions were estimated based on the number of households burning wood and coal and default emission factors of:

Domestic fire emissions defaults = 160g/day for woodburners and 392g/day for coalburners

In this case (and in cases where inventory data did not include seasonal breakdowns), monthly variations in domestic fire emissions were allocated based on a default seasonal profile based on the national average domestic heating seasonal distribution.

Open burning

Open burning[13] was included as a separate source in the HAPINZ update because emission inventories for urban areas identified this as a key PM_{10} source. The method used to estimate open burning emissions in the inventories relies on households estimating both the frequency of burns by season and the quantity of material combusted per burn (typically in m^2) and the application of emission factors. While there may be some uncertainties in the quantities burnt, the source does appear to be significant in many areas and inclusion of this source in the HAPINZ update was considered prudent.

Open burning emission estimates were based on inventory data distributed to CAUs based on household data. For areas where open burning inventory data were unavailable, estimates were made based on the relationship shown in Figure A1-2.

[13] 'Open burning' is also referred to as 'outdoor burning' in many regions across New Zealand and covers the burning of any combustible material in the outdoors, such as household rubbish, garden clippings and agricultural waste. We use 'open burning' in this HAPINZ update.

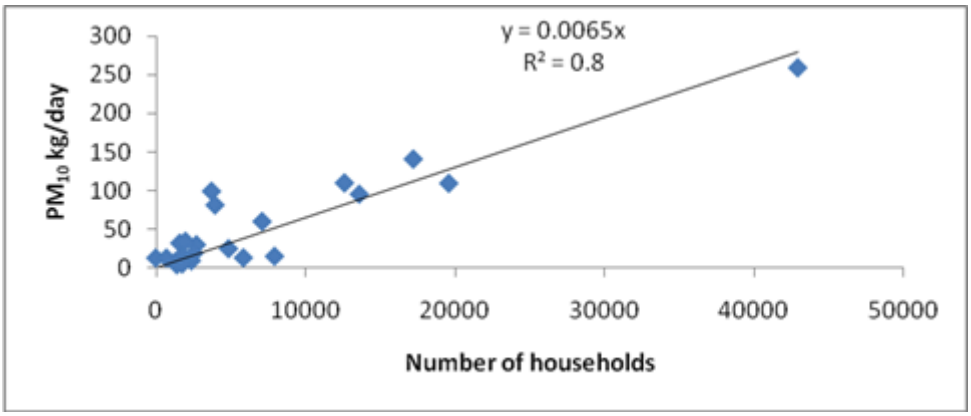

Figure A1-2: Relationship between number of households and PM_{10} emissions from open burning (based on inventory data).

Motor vehicles

Motor vehicle emissions estimates were based on Ministry of Transport vehicle kilometres travelled (VKT) data and emission factors from the Auckland Council Vehicle Emission Prediction Model (VEPM 3.0) (Metcalfe *et al*. 2009). The VKT data provided for each CAU included the number of kilometres travelled at vehicle average speeds less than or more than 80 km/hr[14] and were corrected by 0.9 based on advice from the Ministry of Transport to align the overall total with the actual total from the vehicle registration database. The national fleet profile for 2006 was applied to all CAUs. Regional fleet profiles do differ across New Zealand but the impact on fleet averaged PM_{10} emission factors is minimal. Cold start emissions were factored in as part of the default conditions selected in VEPM.

The following default emission factors were assumed:

Motor vehicle emissions defaults = 0.09g/km for speeds < 80km/h and 0.08g/km for speeds > 80 km/h

These emissions only apply to vehicles travelling on-road.

[14] The cut-off of 80 km/hr was used because it easily separates out the data collected for travel on state highways or motorways from data collected for travel on local roads. Most of the local travel is in urban CAUs that have average speeds that are much lower. Consequently the default emission factor of 0.09g/km shown above for local travel is from VEPM but reflects a more realistic average speed of 40km/hr.

Table A1-8: Example of the source allocation method used to estimate annual average PM_{10} concentrations for Penrose, Auckland CAU

	Ave Mthly PM_{10}	Nat Sources	Ind Tall only	PM_{10} left to Allocate	Dom Fires	Motor Veh	Open Burn	Ind (ex Tall)	Total Emis'ns (inv)	Dom Fires	Motor Veh	Open Burn	All Ind
	($\mu g/m^3$)		($\mu g/m^3$)		($kg/km^2/day$)					($\mu g/m^3$)			
January	12.74	8.67	0.54	3.54	0.00	4.34	0.05	0.65	5.04	0.00	3.04	0.04	0.99
February	12.65	8.91	0.54	3.20	0.00	4.34	0.05	0.65	5.04	0.00	2.76	0.03	0.95
March	13.13	9.70	0.54	2.89	0.01	4.34	0.05	0.65	5.05	0.01	2.49	0.03	0.91
April	12.26	7.51	0.54	4.21	0.03	4.34	0.05	0.65	5.07	0.03	3.60	0.04	1.08
May	14.24	6.68	0.54	7.02	0.31	4.34	0.05	0.65	5.35	0.41	5.69	0.07	1.39
June	14.96	8.19	0.54	6.24	0.67	4.34	0.05	0.65	5.71	0.74	4.74	0.06	1.25
July	14.59	6.82	0.54	7.23	0.91	4.34	0.05	0.65	5.95	1.11	5.27	0.06	1.33
August	14.09	6.48	0.54	7.07	0.86	4.34	0.05	0.65	5.90	1.03	5.20	0.06	1.32
September	12.78	7.15	0.54	5.09	0.21	4.34	0.05	0.65	5.24	0.20	4.21	0.05	1.17
October	13.40	6.24	0.54	6.62	0.03	4.34	0.05	0.65	5.07	0.04	5.66	0.07	1.39
November	13.46	9.80	0.54	3.12	0.01	4.34	0.05	0.65	5.04	0.00	2.68	0.03	0.94
December	11.98	8.38	0.54	3.07	0.00	4.34	0.05	0.65	5.04	0.00	2.64	0.03	0.93
Annual Average	13.36	7.88								0.30	4.00	0.05	1.14
Annual Proportion	**100.0%**	**59.0%**								**2.2%**	**29.9%**	**0.4%**	**8.5%**

Notes on the updated source allocation method

The source allocation approach used in this update will exclude a number of PM_{10} sources in areas outside of main urban areas or urban areas for which no inventory data exist, such as off-road vehicles, rail, aviation and marine. Improvements to the method used in this HAPINZ update, such as the collation of a national database of industrial discharges, were possible but were beyond the resources available for this project. Similarly the annual average PM_{10} estimates made for this study were unable to be commented on via direct contact with industry. As a result there may be some inaccuracies at the area unit level but the overall conclusions for the main urban areas should be reasonable.

In the original HAPINZ study, source allocation was done directly from the LBR model. This also used the same basic statistics we adopted in the update but combined these in regression relationships to predict PM_{10} concentrations over an annual period, based on the number of 'calm, cold' days and wood use relative to Christchurch and also considered the influence of other parameters such as topography.

We consider that the methodology used here is preferable because it reflects the dispersive characteristics of the individual areas and captures the seasonal influence of the different emissions sources.

A1.4 Comparison of the Update versus Original Source Allocation Methods

A comparison of source allocation methods (the HAPINZ update versus the original HAPINZ study) was made for Hastings because a source apportionment study was available for comparing estimates from both methods.

Source apportionment studies involve analysing PM_{10} collected on filters for concentrations of elements and then using the clustering of those elements to identify sources and quantify their contribution. In our view this is the most accurate method of determining the relative contribution of sources to annual average PM_{10}. Only a few places in New Zealand have conducted this type of analysis because it is resource intensive.

The source apportionment study for Hastings found around 38 per cent of the annual PM_{10} was from natural sources, around 43 per cent from domestic heating, 7 per cent from motor vehicles and 4 per cent from secondary particles[15]. As with most other source apportionment studies carried out in New Zealand no specific industrial profile was identified. This is likely a reflection of the small contribution made by industry to PM_{10} concentrations in most urban areas of New Zealand.

Table A1-9 shows how the combination of emissions data and PM_{10} concentrations was used to allocate source contributions in the HAPINZ update. The example shows an emissions-based industry only assessment (i.e., no 'tall stacks') with the natural sources contribution taken from source apportionment work for Hastings.

[15] This profile was dominated by sulphur and was referred to as sulphate or secondary particles.

Table A1-9: Emission inventory estimates for Hastings, combined with monthly average concentrations used to estimate contributions to annual average PM_{10} concentrations

	Domestic Heating (kg/day)	Industry (kg/day)	Motor Vehicles (kg/day)	Total Emissions (inventory) (kg/day)	Average Monthly PM_{10} ($\mu g/m^3$)	Natural Sources ($\mu g/m^3$)	PM_{10} left to Allocate ($\mu g/m^3$)	Motor Vehicles ($\mu g/m^3$)	Domestic Heating ($\mu g/m^3$)	Industry ($\mu g/m^3$)
January	14	33	33	80	13.5	7.7	5.8	2.4	1.0	2.4
February	14	33	33	81	14.6	6.9	7.7	3.2	1.4	3.2
March	24	31	33	88	13.5	4.4	9.1	3.5	2.5	3.2
April	138	31	33	203	14.4	4.2	10.1	1.7	6.9	1.6
May	620	31	33	684	28.7	5.3	23.4	1.1	21.2	1.1
June	1182	31	33	1246	32.5	6.6	25.9	0.7	24.6	0.6
July	1242	31	33	1306	29.9	5.1	24.8	0.6	23.6	0.6
August	1078	31	33	1142	24.1	4.3	19.8	0.6	18.7	0.5
September	394	31	33	458	15.5	5.2	10.4	0.8	8.9	0.7
October	101	31	33	165	12.6	6.0	6.6	1.3	4.0	1.2
November	25	33	33	91	13.2	10.3	2.9	1.1	0.8	1.1
December	14	26	33	73	10.8	7.7	3.1	1.4	0.6	1.1
Annual average					**18.61**	6.14	12.47	1.52	9.51	1.43
						33%		**8%**	**51%**	**8%**

A1.4.1 Results Using the Updated Methodology

From the results shown in Table A1-9, the updated methodology estimated contributions from natural sources at 33 per cent, domestic heating at 51 per cent and motor vehicles at 8 per cent.

This is a slight overestimate of the domestic heating contribution and a slight underestimate of the natural sources contribution relative to the actual source apportionment results but seems reasonable given the potential impact of year to year variations in meteorological conditions.

A1.4.2 Results Using the Original Method

In comparison, the source allocation method used in the original HAPINZ study attributed 19 per cent to domestic heating, 49 per cent to motor vehicles, 13 per cent to industry and 19 per cent to natural sources.

If the original HAPINZ study had had the benefit of the source apportionment natural sources concentration (around 6 $\mu g/m^3$ rather than 2 $\mu g/m^3$ assumed), the estimated relative contributions would have been 42 per cent natural sources, 13 per cent domestic heating and 35 per cent motor vehicles.

A1.5 Conclusions

Since the preparation of the original HAPINZ report, there have been significant improvements in the availability of data from which health based assessments are made. This includes data on air quality monitoring for urban areas (with more than 20 new sites now available), emission inventories for allocating concentrations to sources and source apportionment studies which provide information on natural sources contributions.

For the HAPINZ update, actual PM_{10} monitoring data (rather than modelled data) were used. Analysis showed that 73 per cent of the population live in areas where monitoring data are now available. A good proportion of the population in unmonitored areas reside in rural locations where PM_{10} concentrations are likely to be low. Others reside on the outskirts of main urban areas in locations where Councils consider high PM_{10} concentrations unlikely or in smaller urban centres that have not been prioritised for monitoring owing to lower probability of elevated concentrations. Because of the nature of these unmonitored areas and because of the limited success of the previous model as a predictive tool, a method used for estimating concentrations which relied on monitoring data from other 'low risk' locations.

A number of other options were considered for the exposure assessment. These included using the existing land based regression model used for the original HAPINZ study, updating the land based regression model, and applying monitoring data plus proxy methods to non-monitored urban areas.

Use of the original land based regression model was discounted because of the poor performance as a predictive tool. Updating the LBR model with current monitoring data was also considered but discounted because it was not entirely clear exactly what had been done and the because resources involved in establishing it were likely to be beyond what was available for this project. A third approach of establishing a proxy dispersion factor based on the relationship between emissions and concentrations from existing monitored areas was considered. However, after examining the nature of the unmonitored areas and the extent to which monitoring had been carried out in the county, the proposed approach was preferred.

An alternative approach for allocating source contributions was also used. This method appears to perform significantly better when allocating PM_{10} concentrations by source compared with the original HAPINZ method, although the validation we have undertaken has only been done for a single location where source apportionment data are available. Other sources identified in inventories as contributors to PM_{10} emissions, e.g., open burning, have also been added to source assessment in the HAPINZ update.

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Annex A1A: Summary of Co-location Data for Different PM_{10} Monitoring Methods

Region	Co-location Summary	Site	Method	Status	Pop'n
Northland					
	No collation data available	Kaitaia	High Vol	No adjustment required	5,205
		Whangarei	BAM	Unknown relationship	49,080
Auckland					
	Co-location data are inconclusive. Relationships are poor relative to other areas but suggest minimal differences between gravimetric methods and BAMs.	Queen St	Partisol	No adjustments required	1,303,671
		Khyber Pass	BAM		
		Penrose II	BAM		
		Takapuna	BAM		
		Henderson	BAM		
		Kingsland	BAM		
		Glen Eden	BAM		
		Botany Downs	BAM		
		Pakuranga	BAM		
		Orewa	BAM		
		Kumeu	BAM		
		Pukekohe	BAM		
		Warkworth	BAM		
		Waiuku	BAM		
		Waiheke	BAM		
		Patumahoe	BAM		
		Whangaparaoa	BAM		
Waikato					
	Co-location data are available for most sites. Data are provided as adjusted for gravimetric equivalency. Some sites data do not require adjustment.	Huntly	BAM	Unknown relationship	6,834
		Ngaruawahia	BAM	Unknown relationship	5,106
		Matamata	BAM	No adjustments required	6,306
		Hamilton	TEOM	PM_{10} = 1.19975 x RawTEOM - 3.9182	121,239
		Tokoroa	BAM	PM_{10} = 10 (1.09945logFH62 - 0.08595)	12,933
		Putaruru	BAM	PM_{10} = 1.106BAM - 2.38	3,768
		Te Kuiti	BAM	No adjustments required	4,419
		Turangi	BAM	Unknown relationship	3,240
		Taupo	BAM	PM_{10} = 1.255BAM – 1.538	17,061

Region	Co-location Summary	Site	Method	Status	Pop'n
		Waihi	Partisol	No adjustments required	4,500
Bay of Plenty					
	Data do not require adjustment	Rotorua 1 - Edmond Road	TEOM FDMS	No adjustments required	49,281
		Rotorua 1 – Pererika Site	TEOM	PM_{10}=1.5161x-2.2079	
		Rotorua PakNSave	TEOM	PM_{10}=1.2072x+0.176	
		Ngapuna	Partisol	No adjustments required	513
		Ngongotaha	Partisol	No adjustments required	1,101
		Kawerau	Partisol	No adjustments required	6,921
		Tauranga 1 Moreland Fox Park	TEOM FDMS	No adjustments required	46,191
		Tauranga Otumoetai	TEOM	PM_{10}=1.0342x+1.724	15,309
		Whakatane	FDMS TEOM	No adjustments required	9,360
		Pongakawa	Partisol	No adjustments required	2,928
Gisborne					
	Data do not require adjustment	Gisborne	Hi Vol	No adjustments required	32,535
Taranaki					
	No co-location data available	New Plymouth	BAM	Unknown relationship	45,870
Hawkes Bay					
	No co-location data available	Napier	BAM	Unknown relationship	53,235
		Hastings	BAM	Unknown relationship	49,011
Horizons					
	No co-location data available	Palmerston North	BAM	Unknown relationship	70,809
		Taihape	BAM	Unknown relationship	1,788
		Taumarunui	BAM	Unknown relationship	5,055
Wellington					
	No co-location data available	Upper Hutt	BAM	Unknown relationship	32,904
		Masterton	BAM	Unknown relationship	17,664
		Wainuiomata	BAM	Unknown relationship	16,677
		Wellington Central	BAM	Unknown relationship	156,798
		Lower Hutt	TEOM	Unknown relationship no adjustment applied*	62,070
		Karori	BAM	Unknown relationship	14,007
		Porirua	BAM	Unknown relationship	54,537

Region	Co-location Summary	Site	Method	Status	Pop'n
		Carterton	BAM	Unknown relationship	4,122
Tasman					
	Data are reported as adjusted	Richmond	BAM	PM_{10} = 1.16BAM-0.3	11,715
Nelson					
	Co-location studies show minimal differences (4%) and consequently data are not adjusted in data reported by Council	Nelson Airshed A	BAM	PM_{10}=1.06BAM-0.2	7,698
		Nelson Airshed B1	BAM	PM_{10}=0.98BAM-0.03	4,971
		Nelson Airshed B2	Partisol	No adjustments required	11,865
		Nelson Airshed C	Partisol	No adjustments required	10,920
Marlborough					
	Co-location studies are available - data are not adjusted in data reported by Council	Blenheim MMR	HiVol	No adjustments required	16,590
		Blenheim Redwoodtown	BAM	PM_{10} = 1.18BAM - 2.2	11,937
		Picton	HiVol	No adjustments required	2,928
West Coast					
	No co-location data available	Reefton	BAM	Unknown relationship	948
		Westport	HiVol	No adjustments required	3,900
Canterbury					
	Data do not require adjustment. There are a few sites where non adjusted data are available e.g., BAM at Burnside through MfE programme. The relationship is available for this site.	Rangiora	TEOM FDMS	No adjustments required	11,865
		Kaiapoi	TEOM FDMS	No adjustments required	7,410
		Christchurch St Albans	TEOM FDMS	No adjustments required	309,483
		Christchurch Burnside	BAM	PM_{10} = 1.23x - 2.7	
		Christchurch Woolston	TEOM FDMS	No adjustments required	
		Ashburton	TEOM FDMS	No adjustments required	14,754
		Timaru	TEOM FDMS	No adjustments required	24,801
		Washdyke	TEOM FDMS	No adjustments required	939
		Geraldine	TEOM FDMS	No adjustments required	2,244
		Kaikoura	TEOM	Unknown relationship no adjustment applied*	2,175
		Waimate	TEOM FDMS	No adjustments required	2,835
Otago					
	Co-location studies have been carried out in three locations. Data are currently reported unadjusted.	Oamaru	BAM	Unknown relationship	10,935
		Clyde	BAM	Unknown relationship	921
		Balclutha	BAM	Unknown relationship	4,062
		Alexandra	BAM	PM_{10} = 1.14BAM-0.08	4,824
		Cromwell	BAM	Unknown relationship	3,588

Region	Co-location Summary	Site	Method	Status	Pop'n
		Arrowtown	BAM	Unknown relationship	2,148
		Dunedin	BAM	PM_{10} = 1.07BAM -0.6	82,527
		Mosgiel	BAM	PM_{10} = 1.25BAM -0.1	10,494
		Milton	BAM	Unknown relationship	1,884
		Nasby	BAM	Unknown relationship	114
		Ranfurly	BAM	Unknown relationship	708
Southland					
	Co-location data are available for Invercargill and investigations underway for Gore. Data for Invercargill are presented as both adjusted and unadjusted in reports.	Gore	BAM	Unknown relationship	7,497
		Invercargill Miller St	BAM	PM_{10} = 1.3xBAM-1.1	42,039
		Invercargill Pomona	BAM	PM_{10} = 1.5xBAM-2.9	
		Invercargill Glengarry	HiVol	No adjustments required	
		Invercargill North Road	HiVol	No adjustments required	
		Winton	HiVol	No adjustments required	2,088
		Edendale	HiVol	No adjustments required	504
		Wallacetown	HiVol	No adjustments required	591
		Te Anau	BAM	Unknown relationship	1,899
		Bluff	HiVol	No adjustments required	1,791

Summary	Number of Sites	Monitored Population	Percent
Known relationship	21	576,666	20%
No adjustments required	48	1,888,641	64%
Unknown relationship	26	465,393	16%

* = no adjustments were applied because the characteristics of these locations were considered significantly different to areas where relationships had been derived. It was our view and that of the relevant regional council staff, that the difference between the TEOM and the gravimetric method in these locations would be smaller than measured elsewhere.

Appendix 2: Natural Source Contributions to PM_{10} Concentrations

Prepared by Emily Wilton (Environet Ltd)

Executive Summary

This appendix evaluates available source apportionment studies and outlines the methodology for estimating natural source contributions to PM_{10} concentrations by CAU used in the updated HAPINZ study.

Key Features of the Updated Methodology

1. Using **actual natural source contributions from source apportionment studies** for areas where these have been undertaken.
2. Applying a **default annual average natural source contribution of 6.8 µg/m^3** for CAUs in New Zealand where contributions are not known.
3. Establishing **average seasonal profile for natural sources** based on existing source apportionment studies to estimate monthly contributions to PM_{10} concentrations as per the HAPINZ update PM_{10} exposure assessment methodology.

A2.1 Introduction

A2.1.1 Natural Source Contributions to PM_{10}

The contribution of natural sources to PM_{10} concentrations in New Zealand requires identification for the purpose of allocating health effect estimates to sources and for estimating PM_{10} exposure outside of the urban areas of New Zealand.

The main natural sources of PM_{10} in New Zealand are sea spray (referred to as 'marine aerosol') and windblown dusts (referred to as 'soil'). Other sources such as volcanic eruptions and trans-Tasman emissions from bush fires and dust storms in Australia can be significant but are infrequent occurrences and are difficult to quantify. Only marine aerosol and soil were considered in this update.

Windblown dust and marine aerosol are formed through abrasive and mechanistic processes and are therefore more prevalent in the coarser ($PM_{10-2.5}$) size fraction, although they can also occur within the fine $PM_{2.5}$ size fraction.

The main method for identifying the contribution of natural sources to PM_{10} concentrations in New Zealand is source apportionment using receptor modelling. This involves the measurement of concentrations of elements collected on a filter using standard gravimetric methods and subsequent statistical analysis using Positive Matrix Factorisation (PMF).

In New Zealand source apportionment studies have been carried out in:

- Auckland
- Blenheim
- Christchurch (PM2.5 only)
- Hastings
- Lower Hutt
- Masterton
- Napier
- Nelson (Airshed B2 only)
- Wainuiomata

A2.1.2 The Approach used in the Original HAPINZ Study

Results from the source apportionment studies were unavailable at the time the original HAPINZ study was conducted by Fisher *et al.* (2007). The approach used in the original HAPINZ study to estimate PM_{10} concentrations from natural sources relied on an evaluation of concentrations of PM_{10} measured at a range of air quality monitoring sites. The exact nature of the evaluation is not detailed.

The sites used in the original study included Musick Point (Auckland), Whangaparaoa (Auckland), Whakatane (Bay of Plenty), Pongaweka (Bay of Plenty), Napier (Hawke's Bay), Huntly (Waikato), Gisborne (Gisborne), Kaikoura (Canterbury), Green Island (Dunedin), Alexandra (Otago), and Baring Head (Wellington). The results were used to develop 'background categories' as shown in Table A2-1 which were then applied to urban areas as shown in Table A2-2.

Table A2-1: Natural background categories and PM_{10} values assigned in the original HAPINZ study

Category Background	PM_{10} Value
Inland (low population density)	2 $\mu g/m^3$
Urban flat	4 $\mu g/m^3$
Urban valley	6 $\mu g/m^3$
Coast - not exposed	2 $\mu g/m^3$
Coast - exposed	8 $\mu g/m^3$
Coast - highly exposed	16 $\mu g/m^3$

Table A2-2: PM_{10} exposure by source including natural sources contributions from the original HAPINZ study

Town	Population	PM_{10} from Domestic Heating ($\mu g/m^3$)	PM_{10} from Motor Vehicles ($\mu g/m^3$)	PM_{10} from Industry ($\mu g/m^3$)	PM_{10} from Natural Sources ($\mu g/m^3$)	Total Modelled PM_{10} ($\mu g/m^3$)
Alexandra	4,407	20.6	2.8	0.5	2.0	25.9
Arrowtown	1,689	14.8	2.2	0.5	2.0	19.5
Ashburton	14,202	13.1	3.5	0.9	2.0	19.5
Auckland	359,454	4.4	9.9	2.7	4.0	21.0
Balclutha	4,104	10.5	3.0	0.5	2.0	16.0
Blenheim	21,195	8.9	4.7	1.1	2.0	16.6
Cambridge	4,995	3.5	6.8	1.8	2.3	14.5
Christchurch Inner	132,706	13.6	5.5	1.6	4.0	24.7
Christchurch Outer	183,512	8.1	4.2	1.1	4.0	17.5
Clevedon	3,297	1.9	2.6	0.5	2.0	7.1
Cromwell	2,667	18.6	2.7	0.5	2.0	23.8
Dunedin	82,284	4.5	2.5	1.0	7.5	15.5
Fielding	12,705	5.6	3.5	1.1	2.0	12.3
Geraldine	2,205	9.7	2.9	0.5	4.0	17.2
Gisborne	28,992	4.6	3.2	0.9	2.0	10.7
Gore	7,665	13.4	3.1	0.6	2.0	19.1
Hamilton	114,171	3.4	5.7	2.0	2.0	13.1
Hastings	42,297	1.9	5.1	1.3	2.0	10.3
Hawera	7,830	2.5	4.5	1.0	2.0	10.0
Invercargill	41,964	13.3	3.5	0.9	2.0	19.7
Kaiapoi	9,258	16.7	3.6	1.2	2.0	23.4
Kaikoura	2,106	1.0	3.3	0.5	16.0	20.9
Leamington	6,249	4.1	6.0	1.9	2.0	13.9
Levin	12,315	5.7	6.7	0.8	2.0	15.2
Lower Hutt	78,426	4.9	7.2	0.9	2.0	15.1
Manukau	279,906	2.4	6.3	2.6	3.3	14.6
Masterton	17,514	15.9	3.5	0.7	2.0	22.1
Matamata	6,078	5.5	7.0	1.7	2.0	16.2
Milton	1,920	14.2	3.0	0.5	2.0	19.6

Town	Population	PM_{10} from Domestic Heating ($\mu g/m^3$)	PM_{10} from Motor Vehicles ($\mu g/m^3$)	PM_{10} from Industry ($\mu g/m^3$)	PM_{10} from Natural Sources ($\mu g/m^3$)	Total Modelled PM_{10} ($\mu g/m^3$)
Morrinsville	3,678	4.1	6.8	1.1	2.0	14.0
Mosgiel	6,342	16.2	1.9	1.3	2.0	21.4
Napier	49,851	2.0	5.1	1.3	2.0	10.3
Nelson	15,012	14.4	6.0	0.8	8.0	29.2
New Plymouth	49,047	0.5	2.2	1.0	8.0	11.7
North Shore	184,812	3.2	7.0	2.1	4.0	16.3
Oamaru	11,085	13.2	3.3	0.6	4.0	21.1
Opotiki	3,999	4.5	3.3	1.0	2.0	10.8
Orewa	26,559	3.6	5.4	1.2	2.0	12.1
Palmerston North	70,836	3.9	3.9	1.0	3.8	12.7
Papakura	29,328	4.6	3.8	2.0	2.2	12.6
Paraparaumu	21,372	2.9	3.4	0.8	8.0	15.1
Porirua	47,364	2.4	3.8	0.8	6.1	13.1
Pukekohe	11,076	4.4	4.4	1.7	2.0	12.5
Putaruru	3,783	7.0	4.6	1.1	2.0	14.7
Rangiora	8,607	13.5	3.3	0.9	2.0	19.7
Reefton	987	16.9	6.1	0.5	2.0	25.4
Richmond	10,578	10.3	5.2	0.6	2.0	18.1
Rotorua	45,597	5.9	4.7	1.3	4.0	15.9
Takanini	9,390	2.0	4.6	1.8	4.0	12.4
Taupo	16,935	6.0	6.0	1.0	2.0	15.1
Tauranga	70,854	2.2	4.2	1.1	6.7	14.1
Te Awamutu	9,165	4.8	4.5	1.5	4.0	14.8
Te Kuiti	4,374	6.7	5.8	0.8	2.0	15.3
Timaru	24,732	12.6	3.7	0.9	4.0	21.2
Tokoroa	14,019	15.7	2.7	1.3	2.0	21.7
Upper Hutt	32,904	6.1	7.2	1.0	4.4	18.6
Waiheke Island	7,137	2.6	5.6	0.4	8.0	16.6
Waimate	2,757	11.6	2.9	0.5	2.0	17.0
Wainuiomata	16,602	4.4	5.8	0.5	2.0	12.7
Waitakere	168,741	4.0	5.4	1.9	4.0	15.3
Waiuku	5,478	4.7	4.7	1.5	2.0	12.9

Town	Population	PM_{10} from Domestic Heating ($\mu g/m^3$)	PM_{10} from Motor Vehicles ($\mu g/m^3$)	PM_{10} from Industry ($\mu g/m^3$)	PM_{10} from Natural Sources ($\mu g/m^3$)	Total Modelled PM_{10} ($\mu g/m^3$)
Wanganui	34,767	3.9	6.8	0.9	2.0	13.7
Wellington	162,978	1.4	2.2	1.2	6.7	11.5
Westport	3,783	8.5	7.2	0.5	4.0	20.1
Whakatane	13,113	4.3	3.4	0.8	2.0	10.5
Whangarei	40,824	5.5	6.2	1.0	4.0	16.6
Winton	2,100	21.5	3.0	0.5	2.0	27.0

A2.2 Source Apportionment Studies in New Zealand

A2.2.1 Auckland 2005-2007

Source apportionment studies were carried out in Auckland between 2005 and 2007 and have been analysed for this evaluation. These include the following sites:

- Henderson - August 2006 – December 2007
- Gavin Street, Penrose - May 2006 – December 2007
- Takapuna - December 2005 – December 2007
- Queen Street – January 2006 – December 2007
- Kowhai School – December 2005 – December 2006
- Khyber Pass - December 2005 – December 2007

Data from these studies were made available for this assessment by the former Auckland Regional Council (now Auckland Council) and GNS (Petersen, *pers. comm*. 2006). These data were analysed for monthly average contributions of marine aerosol and soil. Results are shown in Tables A2-3 and A2-4.

Table A2-3: Monthly average marine aerosol contributions in Auckland (2005-2007)

Month	Takapuna (µg/m^3)	Queen Street (µg/m^3)	Kowhai (µg/m^3)	Khyber Pass (µg/m^3)	Penrose (µg/m^3)
January	7.3	6.3	8.2	7.7	7.2
February	6.7	6.6	4.9	7.5	9.1
March	8.2	7.2	7.4	8.4	7.2
April	5.1	3.7	7.4	5.7	7.2
May	4.7	4.8	6.9	5.0	3.8
June	7.6	5.4		7.2	6.7
July	5.5	4.3	6.0	5.6	4.7
August	6.6	3.9	3.7	5.8	5.1
September	7.4	3.6	5.1	7.7	5.2
October	5.7	4.6	3.7	5.0	4.6
November	10.7	4.9	5.9	10.2	9.9
December	7.3	4.4	7.7	7.4	8.5
Annual Average	**6.9**	**5.0**	**6.1**	**6.9**	**6.6**

Table A2-4: Monthly average soil contributions in Auckland (2005-2007)

Month	Takapuna (µg/m^3)	Queen Street (µg/m^3)	Kowhai (µg/m^3)	Khyber Pass (µg/m^3)	Penrose (µg/m^3)
January	1.8	1.2	1.3	1.5	0.9
February	1.8	1.7	2.1	1.7	2.5
March	2.3	1.7	2.2	1.9	2.0
April	1.2	3.0	1.1	1.3	1.9
May	1.5	2.9	0.9	1.9	1.0
June	0.8	1.9		1.6	1.5
July	1.5	1.1	1.2	2.2	1.9
August	1.0	1.7	0.9	1.7	2.0
September	1.2	1.6	1.1	1.5	1.4
October	1.4	2.1	0.8	1.5	1.8
November	2.0	1.8	0.8	1.5	1.2
December	1.6	1.2	0.6	1.1	1.8
Annual Average	**1.5**	**1.8**	**1.2**	**1.6**	**1.7**

A2.2.2 Blenheim 2007

In Blenheim, a source apportionment study was undertaken by Marlborough District Council in 2007 (Wilton & Trompetter 2007).

Like most other locations in New Zealand, domestic fires used for home heating were found to be the main source of PM_{10} on high pollution days contributing around 78 per cent of total emissions. The main natural sources of PM_{10} in Blenheim are soil and marine aerosol. For winter months, soil is the main natural source contributor with an estimated contribution of around 10 per cent. Marin aerosol is the dominant natural source contributor during other months, contributing to around 40 per cent of the daily PM_{10} during spring and summer. The results showed that in spring and summer, that natural sources were responsible for around two thirds of the daily PM_{10} on average. Annually, natural sources were found to comprise around 3.6 $\mu g/m^3$ of PM_{10}.

A2.2.3 Christchurch 2001-2002

A source apportionment study for $PM_{2.5}$ in Christchurch was carried out by Environment Canterbury from November 2001 to August 2002.

Around 3 per cent of the $PM_{2.5}$ concentrations measured during the winter were identified as marine aerosol (Scott 2005). During summer, the contribution was higher at 21 per cent of the daily $PM_{2.5}$ on average. Soil was not identified as a main $PM_{2.5}$ source as soil particles typically reside in the coarse size fraction. Table A2-5 shows the monthly average $PM_{2.5}$ concentrations by source for $PM_{2.5}$ concentrations in Christchurch.

Table A2.5: Monthly average $PM_{2.5}$ concentrations measured in Christchurch (Scott, *pers. comm.* 2010)

Month	Wood Combustion ($\mu g/m^3$)	Marine Aerosol ($\mu g/m^3$)	Aged Aerosol ($\mu g/m^3$)	Secondary Particulate ($\mu g/m^3$)	Motor Vehicles ($\mu g/m^3$)	PMF Mass ($\mu g/m^3$)
Nov-01	1.8	1	1.3	0.5	0.8	5.4
Dec-01	1.3	1.3	1.8	0.2	0.7	5.3
Jan-02	0.4	0.4	2.7	2.2	0.3	6
Feb-02	1.5	1.6	1.8	0.4	0.6	6
Mar-02	2.3	1.7	1.7	0.4	0.5	6.6
April						
May-02	25.7	0.9	1.8	1.6	0.7	30.8
Jun-02	25.3	0.5	1.9	1.9	1	30.7
Jul-02	25.2	0.7	1.1	3.6	1.6	32.2
Aug-02	13.7	1.1	1.1	2.8	1.5	20.1

Results suggest an annual average marine aerosol $PM_{2.5}$ concentration of around 1 µg/m^3. It is also noted that the aged aerosol includes elements that may be of marine origin. This source contributed 2 µg/m^3 of $PM_{2.5}$ during the study period. Results from the Christchurch source apportionment study are excluded from subsequent analysis of natural source contributions because of the absence of the coarser PM_{10} - $PM_{2.5}$ size fraction.

A2.2.4 Hastings 2006-2007

A source apportionment receptor modelling assessment was carried out of PM_{10} and $PM_{2.5}$ concentrations measured in Hastings (Wilton *et al*. 2007.). The project was a collaboration between Hawkes Bay Regional Council, GNS and the Foundation for Research, Science and Technology (FRST) *Protecting New Zealand's Clean Air* programme. Filters were collected during 2006 and 2007.

Analysis showed five sources were found to contribute to the PM_{10} concentrations. These were identified as domestic heating, marine aerosol, motor vehicles, sulphate and soil. Marine aerosol was found to contribute to 53 per cent of the PM_{10} measured during the summer and soil contributed 17 per cent of the summer concentrations. The average PM_{10} concentration during this period was 12 µg/m^3 (24-hour average). In winter, domestic heating was found to be the main source of PM_{10}. Marine aerosol was found to contribute to 9 per cent of PM_{10} concentrations during winter and soil contributed 4 per cent. The average winter PM_{10} concentration was 44 µg/m^3 (24-hour average). Overall natural sources were found to contribute around 5.6 µg/ m^3 (25 per cent) of the annual average of 22 µg/m^3.

A2.2.5 Lower Hutt 2005-2007

A source apportionment study commissioned by the Greater Wellington Regional Council was carried out in Lower Hutt with filters collected from July 2005 to July 2007 (Davy *et al*. 2008). Around 150 filters of $PM_{2.5}$ and $PM_{10\text{-}2.5}$ were collected at the site using a gravimetric GENT sampler.

The study found that marine aerosol and crustal matter/ soil were the main contributors to PM_{10} concentrations. The marine aerosol was more prevalent in the coarse $PM_{10\text{-}2.5}$ size fraction (5.3 µg/m^3 per year) with an additional 1.1 µg/m^3 being present in the finer $PM_{2.5}$ size fraction. Similarly the majority of the soil was in the $PM_{10\text{-}2.5}$ size fraction (2.7 µg/m^3) compared with 0.4 µg/m^3 in the finer $PM_{2.5}$ size fraction. A separate road dust profile was also identified in the coarser size fraction and this contributed around 1.6 µg/m^3 of PM_{10}. The contribution of this source is not included in subsequent evaluations of the natural sources contributions to PM_{10}.

A2.2.6 Masterton 2002-2004

A source apportionment study was carried out from April 2002 to November 2004 in Masterton (Davy 2007). Sampling was carried out for both fine and coarse mode particulate using a GENT sampler.

Four main sources were identified in the coarse mode. These were soil, marine aerosol wood burning and road dust. In the fine mode, the main sources were wood burning, marine aerosol, motor vehicles and sulphate. Marine aerosol was found to contribute around 0.5 $\mu g/m^3$ in the fine ($PM_{2.5}$) mode in the winter and 1 $\mu g/m^3$ in the summer. In the coarse mode, marine aerosol contributed 2.7 $\mu g/m^3$ in the winter and 3.8 $\mu g/m^3$ in the summer. Soil was found to contribute around 2 $\mu g/m^3$ in the coarse mode and just under 1 $\mu g/m^3$ in the fine mode. Overall natural sources were estimated to contribute around 6.9 $\mu g/m^3$ of PM_{10} per year.

A2.2.7 Napier 2008-2009

The Napier source apportionment study was commissioned by the Hawkes Bay Regional Council and was based on monitoring of $PM_{2.5}$ and $PM_{10-2.5}$ between February 2008 and November 2009 (Wilton *et al*. 2010).

In the coarse mode, three main sources were identified as soil, marine aerosol and domestic heating. In the fine mode, the main sources were domestic heating, marine aerosol, motor vehicles and secondary particles. Marine aerosol was found to contribute around 1 $\mu g/m^3$ in the fine ($PM_{2.5}$) mode compared with 3 $\mu g/m^3$ in the coarse mode. Soil was found to contribute 2 $\mu g/m^3$ all in the coarse mode. Overall natural sources were estimated to contribute around 5.9 $\mu g/m^3$ of PM_{10} per year.

A2.2.8 Nelson – Tahunanui 2008-2009

A source apportionment study commissioned by Nelson City Council was conducted in the Tahunanui airshed (Airshed B2) of Nelson from September 2008 to September 2009 (Davy *et al*. 2010). Around 185 filters were collected and analysed for source contributions to PM_{10}.

Seven sources were found to contribute to PM_{10} concentrations with the main contributor to annual average concentrations being domestic home heating (35 per cent). Marine aerosol and soil were the next most significant sources contributing 18 per cent and 16 per cent while industry was found to contribute 13 per cent. Motor vehicles were found to contribute 11 per cent of the PM_{10} and secondary sulphate around 7 per cent. The relative contribution of domestic home heating increased during the winter months and was the predominant PM_{10} source on days when the National Environmental Standard (NES) for PM_{10} of 50 $\mu g/m^3$ was breached (24-hour average).

The annual average contribution to PM_{10} concentrations were 3.4 $\mu g/m^3$ for marine aerosol and 3.1 $\mu g/m^3$ for soil.

A2.2.9 Wainuiomata 2006-2008

In Wainuiomata a source apportionment study commissioned by the Greater Wellington Regional Council was carried out based on 220 filters collected from July 2006 to September 2008 (Davy *et al*. 2009).

The study found that marine aerosol and crustal matter/ soil were the main contributors to PM_{10} concentrations. Six sources were identified with marine aerosol and soil being the only two sources that were present in both size fractions. As with Lower Hutt both natural sources were more prevalent in the coarse $PM_{10-2.5}$ size fraction (4.7 $\mu g/m^3$ per year for marine aerosol and 1 $\mu g/m^3$ for soil). The marine aerosol contribution in the $PM_{2.5}$ size fraction was 1 $\mu g/m^3$ and a further 0.2 $\mu g/m^3$ of soil was found in the $PM_{2.5}$ size fraction. Other sources contributing to PM_{10} concentrations in Wainuiomata were biomass burning (domestic heating), motor vehicles, sulphate, and road dust. The latter contributed 1.6 $\mu g/m^3$ of PM_{10} per year and is not included in subsequent evaluations of the natural sources contributions.

A2.3 Estimation of Marine Aerosol and Soil

Daily source contribution data from seven of these studies were obtained and analysed for monthly average contributions of marine aerosol and soil. Christchurch data were not included because it was obtained for $PM_{2.5}$ only and Masterton data were unavailable as monthly averages. Data from Auckland sites were averaged to give a single value for Auckland.

Tables A2-6 and A2-7 compare the contributions from marine aerosol and soil respectively over the range of years for which source apportionment data are available. Table A2-8 shows that the annual average natural source contributions from these natural sources alone range from 3.6 $\mu g/m^3$ (Blenheim) to 9.5 $\mu g/m^3$ (Lower Hutt).

A comparison of the source apportionment natural sources contribution to PM_{10} to the estimated values from the original HAPINZ study is shown in Table A2-8 (bottom rows). Results suggest that while the original study underestimated natural source contributions in most locations, estimates of higher than average natural source contributions in some locations were correct.

The complexities depicted in Tables A2-6 to A2-8 around marine aerosol and soil contributions in urban locations in New Zealand make it difficult to derive a land/ topography based approach to allocating natural source contributions to CAUs. Results observed are not entirely consistent with topographical features used previously such as valleys, distance to coast and the nature of the coast line.

For example, Nelson was predicted to experience high (relative to other locations) natural source contributions based on its exposure to the sea. However, marine aerosol contributions in the area were relatively low and the main natural source contributor at the monitoring site (industrial area) was soil/dust.

Table A2-6: Monthly average marine aerosol contributions to PM_{10} concentrations

Month	Auckland average ($\mu g/m^3$)	Blenheim ($\mu g/m^3$)	Hastings ($\mu g/m^3$)	Lower Hutt ($\mu g/m^3$)	Napier ($\mu g/m^3$)	Nelson ($\mu g/m^3$)	Wainiuomata ($\mu g/m^3$)
January	7.3	2.2	5.1	4.5		6.3	6.1
February	7.0	3.3	5.4	6.6	6.0	4.0	7.2
March	7.7	2.0	6.6	8.0	3.5	3.1	6.7
April	5.8	2.7	3.8	4.5	3.1	4.5	3.6
May	5.1	1.5	1.6	6.5	2.4	2.5	4.5
June	6.7	0.7	3.0	7.7	4.4	0.6	6.0
July	5.2	2.0	5.7	5.9	4.0	0.6	3.8
August	5.0	0.7	2.0	5.2	3.4	4.4	5.0
September	5.8	3.1	3.6	6.5		4.6	6.3
October	4.7	3.9	3.8	8.1	2.5	2.8	11.0
November	8.3	3.1	1.9	7.0	7.2	6.2	7.1
December	7.1	2.5	4.1	6.6		5.9	7.4
Annual average	**6.3**	**2.3**	**3.9**	**6.4**	**5.9**	**3.8**	**6.2**

Table A2-7: Monthly average soil contributions to PM_{10} concentrations

Month	Auckland average ($\mu g/m^3$)	Blenheim ($\mu g/m^3$)	Hastings ($\mu g/m^3$)	Lower Hutt ($\mu g/m^3$)	Napier ($\mu g/m^3$)	Nelson ($\mu g/m^3$)	Wainiuomata ($\mu g/m^3$)
January	1.3	1.5	1.5	2.9		5.7	1.4
February	2.0	2.0	1.5	4.3	0.9	2.7	1.6
March	2.0	1.4	2.4	3.7	0.9	2.8	1.9
April	1.7	0.2	1.5	2.9	1.1	2.5	0.7
May	1.6	0.6	0.8	4.2	2.9	1.8	0.9
June	1.5	1.0	0.8	2.6	2.2	2.5	1.2
July	1.6	1.1	1.2	3.1	1.1	2.1	0.6
August	1.5	1.8	2.4	2.7	0.9	1.4	1.3
September	1.4	1.8	4.2	2.5		3.2	1.4
October	1.5	1.7	1.7	3.0	3.6	5.7	1.2
November	1.5	0.8	0.2	3.3	3.1	3.3	1.7
December	1.3	1.4	1.8	2.0		4.0	1.4
Annual average	**1.6**	**1.3**	**1.7**	**3.1**	**1.9**	**3.1**	**1.3**

Table A2-8: Monthly average natural sources contributions to PM_{10} concentrations

Month	Auckland average ($\mu g/m^3$)	Blenheim ($\mu g/m^3$)	Hastings ($\mu g/m^3$)	Lower Hutt ($\mu g/m^3$)	Napier ($\mu g/m^3$)	Nelson ($\mu g/m^3$)	Wainiuomata ($\mu g/m^3$)
January	8.7	3.7	6.6	7.4		12.1	7.5
February	8.9	5.4	6.9	10.8	6.9	6.6	8.9
March	9.7	3.4	9.0	11.8	4.4	5.9	8.7
April	7.5	2.9	5.3	7.4	4.2	7.0	4.2
May	6.7	2.1	2.4	10.7	5.3	4.3	5.4
June	8.2	1.7	3.8	10.2	6.6	3.1	7.3
July	6.8	3.1	7.0	9.0	5.1	2.6	4.4
August	6.5	2.5	4.5	7.9	4.3	5.8	6.4
September	7.2	4.8	7.8	9.0		7.8	7.6
October	6.2	5.6	5.5	11.1	6.0	8.6	12.2
November	9.8	3.9	2.1	10.3	10.3	9.5	8.8
December	8.4	3.9	5.9	8.6		9.9	8.8
Annual average	**7.9**	**3.6**	**5.6**	**9.5**	**5.9**	**6.9**	**7.5**
Original HAPINZ	4	2	2	6.7	2	8	2

Similarly natural source contributions in Wainuiomata were assumed to be low owing to the topography/hills between the sea and the town, yet the marine aerosol contribution to PM_{10} at this site was almost as high as for Lower Hutt and the Auckland average. A further study (not included in these tables because summary data only were available) showed that natural sources contributions inland in Masterton were as significant as those nearer the coast (Davy 2007).

One notable feature is the consistency of the marine aerosol contribution in the $PM_{2.5}$ size fraction which was around 1 $\mu g/m^3$ at all four sites where the $PM_{2.5}$ size fraction was investigated namely Christchurch, Wainuiomata, Lower Hutt and Napier. In addition, not reported separately here because data were available for PM_{10}, the $PM_{2.5}$ component was investigated at the Auckland sites and the marine aerosol component averaged around 1 $\mu g/m^3$ at Penrose, Khyber Pass and Kowhai and around 3 $\mu g/m^3$ at Queen Street and Takapuna.

Further source apportionment studies may improve the ability to extrapolate from existing studies. However, in the absence of more conclusive relationships between topography and natural source contributions, a natural sources contribution to CAUs was based on the average of 6.8 $\mu g/m^3$ from the seven source apportionment study areas. Exceptions were made in cases where the natural sources contribution was known. In particular the study

areas included in the source apportionment studies detailed herein were evaluated based on the results of that apportionment.

A further complexity to the relationship was the absence of a strong and consistent seasonal pattern to the natural sources contributions. It was predicted that higher concentrations would occur during the windier months of November and December. Figure A2-1 shows the seasonal profile for marine aerosol at each location over the range of years for which source apportionment data are available. The most notable variability in marine aerosol contribution is for Nelson which has average contributions of less than 1 $\mu g/m^3$ during June and July compared with concentrations above 4 $\mu g/m^3$ for most non winter months.

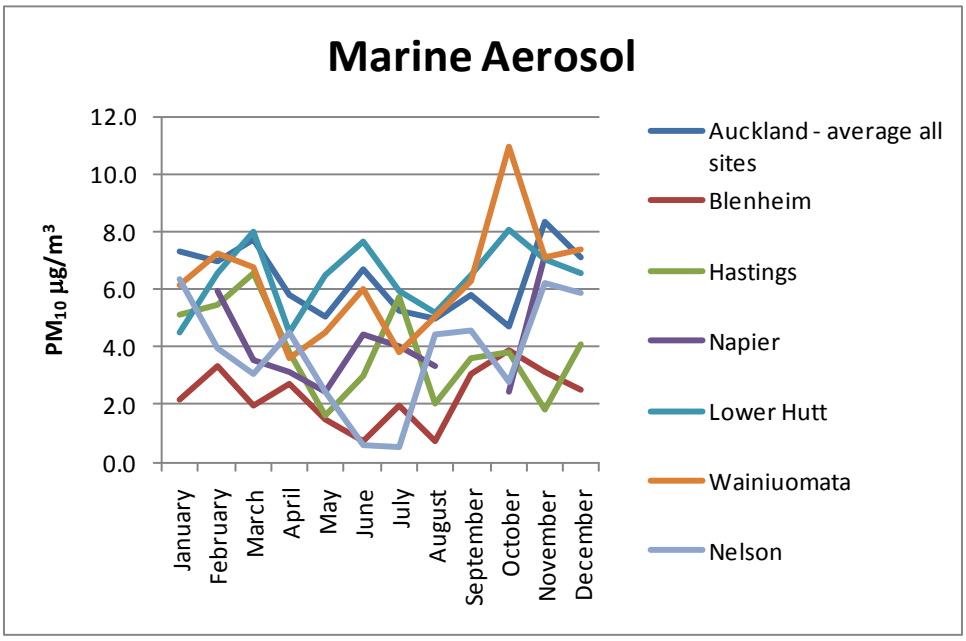

Figure A2-1: Seasonal variations in marine aerosol contributions to PM_{10}

Figure A2-2 shows the seasonal variations in soil contributions at each location over the range of years for which source apportionment data are available.

An average seasonal profile was derived from the seven locations and used to allocate natural source contributions by month, as required by the HAPINZ update methodology for source allocation. Figure A2-3 shows the average soil and marine aerosol contributions per month across seven sites and the combined seasonal profile of soil and marine aerosol. The average depicted is consistent with expectations of lower contributions during the calmer winter months and higher contributions during the summer months.

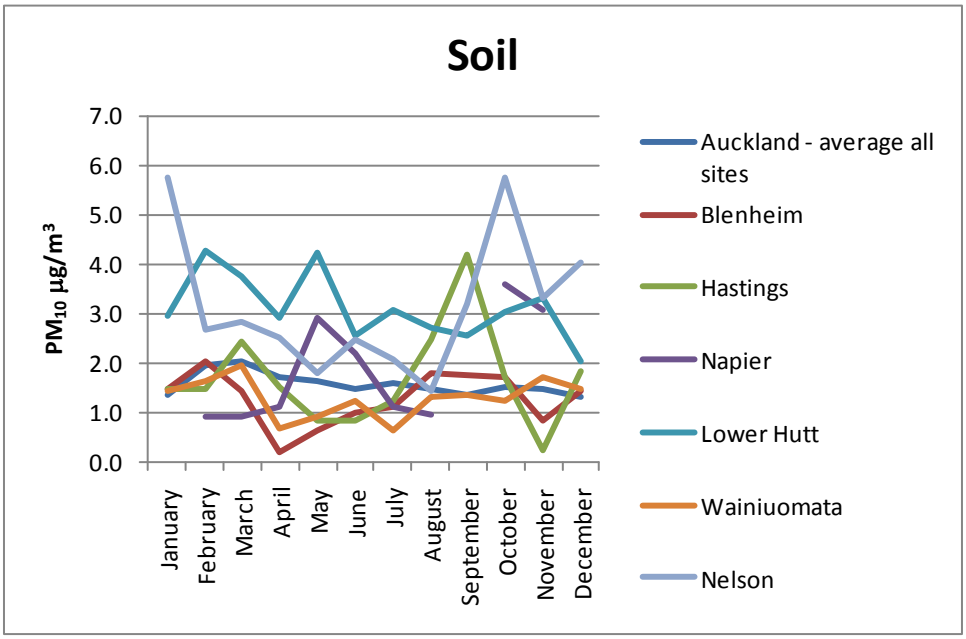

Figure A2-2: Seasonal variations in soil contributions to PM_{10}

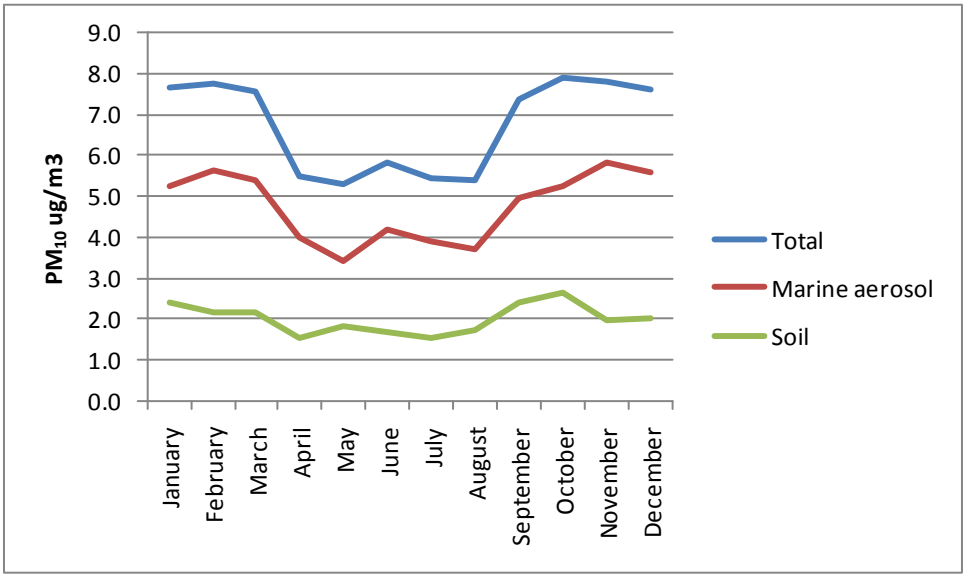

Figure A2-3: Average seasonal variations in soil and marine aerosol across source apportionment monitoring sites

A2.4 Summary of Method

The following method was derived based on an evaluation of the marine aerosol and soil contributions from source apportionment studies for New Zealand:

- Natural source contributions from source apportionment studies were used for areas where these were known.
- An annual average contribution of 6.8 $\mu g/m^3$ was used for CAUs in New Zealand for which contributions were unknown.
- An average seasonal profile was established based on existing source apportionment studies and was used to estimate monthly contributions to PM_{10} concentrations as per the HAPINZ update methodology.

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Appendix 3: Health Outcomes and Exposure-Response Relationships

Prepared by Simon Hales (University of Otago)

Executive Summary

This appendix describes and justifies the following components of the health effects assessment methodology used in the updated HAPINZ study involving:

- Exposure-response functions and methodology for estimation of premature mortality from PM_{10} exposure
- Whether to specifically assess $PM_{2.5}$ effects based on literature and available $PM_{2.5}$ data
- Exposure-response functions and methodology for morbidity effects of PM pollution based on review of recent meta-analyses
- Review of meta-analyses and available data for other pollutants and
- Confirmation of methodology to quantify health effects for Māori and non-Māori sub-groups.

Key Features of the Updated Methodology

Mortality, all non-external causes: exposure-response relationship, per 10 µg/m^3 PM_{10}

1. **Mortality, adults** (ages 30 years and over), annual mean, **all ethnicities**: 7% (3% to 10%) (Hales *et al*. 2010).

This is the best available local evidence of mortality effects and is consistent with the international evidence on long-term effects of particle exposure.

2. **Mortality, adults** (ages 30 years and over), annual mean, **Māori**: 20% (7% to 33%) (Hales *et al*. 2010).

This is the only available source of ethnically specific data but, since the 95% confidence intervals overlap, there is a small chance that the true relationship is no different for Māori compared to other ethnicities.

3. **Mortality, babies** (ages 1 month to 1 year), annual mean: 5% (2% to 8%) (Lacasaña *et al*. 2005).

This is an important emerging category of health effect that is included in current international assessments.

Morbidity, exposure-response relationship, per 10 µg/m^3 PM_{10}

4. **Cardiac hospital admissions, all ages**, daily mean: 0.6% (0.3%-0.9%) (APHEIS 2004).

Likely to be representative of effects found in New Zealand cities. Local studies are reasonably consistent, but are only available for Christchurch.

5. **Respiratory hospital admissions, all ages**, daily mean: 1 % (0.6%-1.7%) (APHEIS 2004).

Likely to be representative of effects found in New Zealand cities. Local studies are reasonably consistent, but are only available for Christchurch.

6. **Respiratory hospital admissions, ages 1-4 years**, daily mean: 2% (1-4%) and **ages 5-14 years**: 3% (0-5%) (Barnett *et al*. 2005).

Overall findings for Australasian cities, including PM_{10} in Christchurch.

Morbidity, exposure-response relationship, per 10 µg/m^3 $PM_{2.5}$

7. **Restricted activity days, all ages**, annual mean [16]: 0.9 (0.5-1.7) days per person (American Lung Association 1995 based on Ostro 1987).

Numerically large effect, though uncertain exposure-response relationship.

Potential PM_{10} Health Effects Which Were Considered but Not Included:

- Short-term effects on mortality. This impact is largely included as part of the assessment of long-term effects and separate consideration would lead to 'double counting'.
- Adverse reproductive outcomes (low birth weight, preterm births), stroke incidence, chronic obstructive airways disease and asthma *incidence* were not included due to limited scientific consensus on relationships with air pollution.
- Note the distinction between incidence of disease and worsening of pre-existing disease. Effects of air pollution on stroke and on exacerbations of respiratory diseases, including respiratory infections, asthma and chronic airways diseases are partly accounted for in the assessments of morbidity (hospital admissions and restricted activity days: categories 4 to 7, above).

[16] Assuming 60 per cent of annual PM_{10} in urban areas and 40 per cent of annual PM_{10} in rural areas is $PM_{2.5}$ for New Zealand.

Potential Other Health Effects Which Were Considered but Not Included:

These are any effects of air pollution for which PM_{10} is an inadequate indicator. In particular, there is increasing evidence that NO_2 exposure has important health effects, (especially respiratory symptoms in children). Several studies show that proximity to busy roads is a risk factor for respiratory effects. However, it is difficult to quantify these exposures in New Zealand. This is because very accurate information on the location of individuals is needed for this type of assessment. In New Zealand, routinely available health information is geocoded to census area units, which is not sufficiently accurate for an epidemiological study of the potential health effects based on the proximity to road traffic. In addition, differences between characteristics of traffic exposure overseas and in New Zealand make it difficult to extrapolate results from overseas studies to the New Zealand population.

Not being able to robustly assess NO_2 exposure means that results of this update most likely under-estimate the health impacts of motor vehicle-related air pollution.

Rationale for Selection of Exposure-Response Functions

Urban air pollution contains a complex mixture of gases and particles. The precise health effects of exposure depend, in part, upon a range of subtle factors related to the composition of the pollutant mixture, the level and duration of exposure (effective dose) and factors related to the exposed population, such as age, sex, ethnicity, pre-existing illnesses and access to health services (population sensitivity). Individual pollutants are often highly correlated and their independent effects are only partly resolved by epidemiological methods. In practice, in designing or extrapolating epidemiological studies to assess health impacts in a population and time period of interest, it is necessary to simplify the air pollution effects assessment by using summary indicators of exposure.

Particulate matter (PM_{10}) is the best available summary indicator of air pollution exposure for the purposes of the HAPINZ update study. More extensive local observations are now available for PM_{10} than for other pollutants. Major New Zealand based and international epidemiological studies have used PM_{10} as one of the exposure metrics. Although international assessments increasingly use $PM_{2.5}$ rather than PM_{10} as the exposure metric, we did not do this due to the lack of $PM_{2.5}$ monitoring data currently available[17]. Estimates of annual average and daily average PM_{10} are available for all 2006 census areas.

Basing the update on PM_{10} rather than $PM_{2.5}$ means that proportion of air pollution health impacts attributed to anthropogenic sources, in particular motor vehicles and domestic

[17] However, as a cross check, we developed a potential exposure-response function for premature mortality resulting from annual $PM_{2.5}$ in New Zealand by taking the Hales *et al.* (2010) relationship for PM_{10} and scaling by the typical ratios in the premature mortality exposure-response functions for $PM_{2.5}$ to PM_{10} seen in overseas studies. More information is provided in Section A3.2 later.

fires used for home heating, will be lower as these sources make a greater contribution to finer particulate fractions than natural sources.

We have used nationally representative New Zealand epidemiological studies where available. This avoids the additional uncertainties involved in extrapolating effects observed in international studies to the New Zealand population. Nevertheless, some extrapolation is still required even where New Zealand epidemiological studies are available, e.g., from one or more cities to the New Zealand urban population or from effects observed in the past to the present day.

A3.1 Summary of Health Outcomes Used in Previous Assessments

We reviewed one US assessment (USEPA 2010), one global assessment (Cohen *pers. comm.* 2011), one European impact assessment guide (ENHIS 2007) and one global impact assessment guide (Ostro 2004)[18]. The following summarises the health outcomes included in these overseas studies and that were considered for our updated New Zealand study.

A3.1.1 USEPA Assessment of $PM_{2.5}$ (USEPA 2010)

- total, cardiopulmonary, and lung cancer mortality associated with long-term $PM_{2.5}$ exposures
- mortality (total non-accidental, cardiovascular, and respiratory), morbidity (hospital admissions for cardiovascular and respiratory causes), associated with short-term exposures
- respiratory symptoms (not requiring hospitalisation) associated with short-term exposures

  "... consistent with those endpoints ... having a causal or likely causal relationship with $PM_{2.5}$ exposures." (USEPA 2010)

A3.1.2 Global Burden of Disease Assessment (Cohen *pers. comm.* 2011)

- mortality from all causes and from selected causes of death, adults, associated with long-term $PM_{2.5}$ exposures (ACS study: based on Krewski *et al*. 2009)
- cardiovascular morbidity (short-term $PM_{2.5}$ exposures)
- acute lower respiratory mortality, ages 0-4 years (thought to be the main contributor to mortality in this age group globally, but less appropriate for NZ)

[18] Other assessments were reviewed, such as COMEAP and EPHC Seven Cities, but we found that the key epidemiological relationships of interest were contained in the major studies outlined in this section so we did not mention the other ones separately.

The following outcomes were considered but not included:

- adverse reproductive outcomes
- respiratory disease mortality (related to ozone)
- stroke incidence
- COPD incidence
- asthma prevalence and incidence

A3.1.3 European Guidelines for Health Impact Assessment of Air Pollution (ENHIS 2007)

This included the following exposure-response data:

Health Outcome	Population	Pollutant	Period	Mean Type	RR (for 10 µg/m³ increase	References
Mortality						
Total mortality excluding external causes(ICD9<800 - ICD10 A00-R	All ages	PM_{10} annual mean	Year	Annual	1.043 (1.026-1.061)	Künzli et al. 2000[4]
All causes mortality(ICD9 0-999 - ICD10 A00-Y98) Cardiopulmonar mortality (ICD9 401-440 and 460-519 - ICD10 I 10- J00-J99) LCA mortality (ICD9 162 - ICD10 C33-C34)	All ages	$PM_{2.5}$ annual mear	Year	Annual	1.06 (1.02-1.11) 1.09 (1.03-1.16) 1.14 (1.04-1.23)	Average Pope, 2002[6]
Total postneonatal mortality Postneonatal respiratory mortality (ICD9 460-519 - ICD10 J00-J99) Postneonatal Sudden Infant Death Syndrom Mortality (ICD9 798.0 -I R95)	1 month- 1 year	Corrected PM_{10}	Year	Annual	1.048 (1.022-1.075) 1.216(1.102-1.342) 1.12 (1.07-1.17)	Lacasaña et al 2005 [23] Woodruff 1997[24]

Table 5: Concentration-Response Funcion (CRF) selected for long-term HIA

By default, in the framework of ENHIS, only the Concentration-Response Function for mortality excluding external causes and PM_{10} is available: CRF (IC95%) = 1.043 (1.026-1.061); reference: Künzli et al. 2000[4].

Health outcome	Population	Pollutant	Period	Mean Type	RR (for 10 µg/m³ increase	References
Mortality						
Total mortality excluding external causes(ICD9<800 - ICD10 A00-R99) Cardiovascular mortality (ICD9 390-459 - ICD10 I00-I99) Respiratory mortality (ICD9 460-519 - ICD10 J00-J99)	All ages	$O_3$8hmax	Summer	Daily	1.0031 (1.0017-1.0052) 1.0046 (1.0022-1.0073) 1.0113 (1.0074-1.0151)	Gryparis et al 2004[20]
Total mortality excluding external causes(ICD9<800 - ICD10 A00-R99) Cardiovascular mortality (ICD9 390-459 - ICD10 I00-I99) Respiratory mortality (ICD9 460-519 - ICD10 J00-J99)	All ages	PM_{10} daily mea	Year	Daily	1.006 (1.004-1.008) 1.009 (1.005-1.0021) 1.013 (1.005-1.021)	Anderson et al 2004[21]
Total mortality excluding external causes(ICD9<800 - ICD10 A00-R99) Cardiovascular mortality (ICD9 390-459 - ICD10 I00-I99) Respiratory mortality (ICD9 460-519 - ICD10 J00-J99)	All ages	BS	Year	Daily	1.006 (1.004-1.009) 1.004 (1.002-1.007) 1.006 (0.998-1.015)	Anderson et al 2004[21]
Morbidity						
Cardiac hospital admissions(ICD9 390-429 - ICD10 I00-I52) Respiratory hospital admissions (ICD9 460-519 - ICD10 J00-J99)	All ages	PM_{10} daily mea	Year	Daily	1.006 (1.003-1.009) 1.0114 (1.0062-1.0167)	Apheis 3[5]
Cardiac hospital admissions(ICD9 390-429 - ICD10 I00-I52) Respiratory hospital admissions (ICD9 460-519 - ICD10 J00-J99)	All ages	BS	Year	Daily	1.011 (1.004-1.019) 1.0030 (0.9985-1.0075)	Apheis 3[5]
Respiratory hospital admissions (ICD9 460-519 - ICD10 J00-J99)	< 15 years	PM_{10} daily mea	Year	Daily	1.010 (0.998-1.021)	Anderson et al 2004[21]
Respiratory hospital admissions (ICD9 460-519 - ICD10 J00-J99)	15 - 64 years	O_3 8h max	Summer	Daily	1.001 (0.991-1.012)	Anderson et al 2004[21]
Respiratory hospital admissions (ICD9 460-519 - ICD10 J00-J99)	>64 years	O_3 8h max	Summer	Daily	1.005 (0.998-1.012)	Anderson et al 2004[21]

Table 4: Concentration-Response Funcion (CRF) selected for short-term HIA

By default, in the framework of ENHIS, only the Concentration-Response Function which relates mortality excluding external causes and PM_{10} is available: CRF (IC95%) = 1.006 (1.004-1.008); reference: Anderson et al. 2004[21].

A3.1.4 WHO Guide to the Assessment of Outdoor Air Pollution Effects (Ostro 2004)

The current WHO guide to the assessment of outdoor air pollution effects (Ostro 2004) recommends basing the assessment on estimates of either $PM_{2.5}$ or PM_{10} exposures. The recommended endpoints in this guide include:

- mortality from all causes associated with short-term exposures (all ages)
- respiratory mortality associated with short-term exposures (age <5 years)
- mortality from all causes and from lung cancer, adults, associated with long-term exposure

A3.2 Discussion of Health Endpoints Used in this Update

Our health impact assessment has four major components:

1. an estimate of effects of long-term exposure on mortality in adults, (found to be the dominant health impact in previous studies)
2. separate estimates of effects of long-term exposure on mortality in sensitive subpopulations, including babies and Māori.
3. estimates of the effects of short-term exposure on hospital admissions for cardiovascular and respiratory diseases
4. an estimate of restricted activity days

These address the need to provide policy-relevant estimates which are based on well-established epidemiological results, while avoiding double-counting of effects, and including some less substantial evidence relevant to social justice and equity.

We assessed the effect of air pollution on mortality in adults based on the results of a recent New Zealand study (Hales *et al*. 2010). This cohort study of mortality in adults aged 30-75 over the years 1996-1999 was based on exposure estimates from the original HAPINZ study (Fisher *et al*. 2007). The result is consistent with the international evidence on long-term effects of particle exposure. Any inaccuracy in these exposure estimates is likely to have biased the dose-response relationship towards null.

Hales *et al*. (2010) reported substantially different responses to air pollution in different ethnic groups. These differences were numerically substantial (20 per cent in Māori as compared to 7 per cent in all ethnicities combined), but need to be treated with some caution by policymakers as the ethnic difference was not statistically significant in this study. There is limited evidence of varying sensitivity to air pollution effects according to socio economic position in international studies.

As discussed in the previous section, many international studies recommend use of $PM_{2.5}$ as the exposure metric. $PM_{2.5}$ refers to particulate matter less than 2.5 µm as opposed to PM_{10} which is less than 10 µm. Some epidemiologists consider that the main life-

shortening effects of air pollution may be due to $PM_{2.5}$ or even smaller size fractions – known as 'ultra-fine particles'. $PM_{2.5}$ and ultra-fine particles have larger contributions from motor vehicles and domestic home heating, e.g. opposed to natural sources and open burning.

However, the dearth of monitoring data and relevant exposure-response functions makes it impossible to robustly quantify the impacts of these smaller fractions currently in New Zealand so we were unable to base our assessment on $PM_{2.5}$. Nonetheless, as part of the sensitivity analyses, we undertook a rudimentary cross-check for the most significant health effect - premature mortality in all adults aged 30 years and over. The New Zealand-specific relationship determined by Hales *et al*. (2010) for PM_{10} was scaled using a ratio based on the premature mortality exposure-response functions for $PM_{2.5}$ to PM_{10} seen in overseas studies to develop an indicative relationship in New Zealand for $PM_{2.5}$ exposure. In this case, the exposure-response functions selected were taken from the ENHIS (2007) guidelines – specifically for total mortality due to:

- annual average $PM_{10:}$ = 1.043 (1.026-1.061) from Künzli *et al*. (2000)
- annual average $PM_{2.5}$ = 1.06 (1.02 – 1.11) from Pope (2002)

Based on these relationships, we estimated an indicative exposure-response function for premature mortality in all adults aged 30 years and over in New Zealand of 1.09 (=1.07*1.06/1.043) per 10 $\mu g/m^3$ $PM_{2.5}$. This has been applied to estimates of $PM_{2.5}$ concentrations across New Zealand (largely taken from source apportionment work) to check the figures derived based on PM_{10}.

For relatively wealthy countries such as New Zealand, the strongest evidence for effects on mortality in children relates to the post neonatal period (ages 1 month to 1 year). We assessed the effect of air pollution on post-neonatal mortality based on the meta-analysis by Lacasaña *et al*. (2005) as cited in a European guide to air pollution impact assessment (ENHIS 2007). There is insufficient evidence on which to base estimates of mortality impacts of long-term exposure in older children and young adults (aged <30 years).

We included estimates of effects on hospital admissions for respiratory diseases in children, based on the results of a multi-city Australasian study (Barnett *et al*. 2005) and the effect on hospital admissions in adults, based on the results of a European meta-analysis, APHEIS (2004), as cited in a European guide to air pollution impact assessment (ENHIS 2007).

Some New Zealand and international studies have also estimated restricted activity days (in which air pollution exposure causes symptoms sufficient to prevent usual activities such as attendance at work or study). We included an estimate of restricted activity days, for all ages, (American Lung Association 1995 based on Ostro 1987).

The proposed health endpoints and exposure-response estimates (above) are consistent with recommendations of other recent or current international studies of air pollution effects. We did not undertake a full literature review but examined the health outcomes included in recent assessments. We reviewed one US assessment (USEPA), one global

assessment (Cohen *pers. comm.* 2011) one European impact assessment guide (ENHIS 2007) and one global impact assessment guide (Ostro 2004).

The USEPA assessment of health effects of $PM_{2.5}$ exposures included the following endpoints considered to have *"... a causal or likely causal relationship with $PM_{2.5}$ exposures."* (USEPA 2010)

- associated with long-term exposures:
    - all causes mortality
    - ischemic heart disease
    - cardiopulmonary and
    - lung cancer mortality
- in relation to short-term exposures (24-hour averages):
    - mortality (non accidental, cardiovascular, respiratory)
    - hospital admissions, (cardiovascular, respiratory) and
    - respiratory symptoms (not requiring hospital admission)

An assessment of global air pollution effects on health in the year 2005, currently in progress, will assess mortality from all causes and from selected causes of death, in adults, (associated with long-term $PM_{2.5}$ exposures), cardiovascular morbidity (associated with short-term $PM_{2.5}$ exposures) and acute lower respiratory mortality, in children ages 0-4 years. Mortality from respiratory diseases is thought to be the main contributor to mortality in the 0-4 years age group in poor countries, and is appropriate for a global assessment (Cohen *pers. comm.* 2011). However, this endpoint may be less appropriate in a rich country such as New Zealand, where respiratory diseases are less important as a cause of mortality in this age group. For relatively wealthy countries such as New Zealand, the strongest evidence for effects on mortality in children relates to the post neonatal period (ages 1 month to 1 year). We therefore used an exposure-response coefficient for all cause post neonatal mortality from a meta-analysis (Lacasaña *et al.* 2005). This is a potentially important category of health effect which does not overlap with the assessment of mortality in adults.

A WHO guide notes that the short-term mortality effects are largely included in the long-term mortality estimates, and that these two estimates should not be added together, to avoid double counting (Ostro 2004, Table 1, p4-5). This author noted that there is evidence of effect modification by social status but did not recommend providing separate estimates by social status, pending further evidence of such effects (Ostro 2004, p19).

Sensitivity analyses were conducted for selected outcomes, using either the 95% confidence intervals or the upper and lower bounds for the exposure–response function.

A3.2.1 Linear, No Threshold Assumption

We assumed linear, no threshold exposure-response functions for all outcomes. This is in line with current thinking for exposures in the range typically experienced in New Zealand (Schwartz *et al*. 2002, Schwartz *et al*, 2008).

A3.3.3 Estimates We Consider Outside the Scope of the Present Study

The assessment of global air pollution effects on health (referred to previously) considered but decided not to include the effects of air pollution on adverse reproductive outcomes, mortality from respiratory diseases (related to ozone), incidence of stroke, asthma and chronic obstructive pulmonary disease. These decisions were based on assessment of the strength of epidemiological evidence, problems of extrapolation at global scale, and the availability (or lack thereof) of required baseline data (Cohen *pers. comm*. 2011).

There is evidence that both proximity to busy roads and NO_2 exposure have important health effects, especially on respiratory symptoms and lung development in children (Cohen *pers. comm*. 2011, Barnett *et al*. 2005, Gauderman *et al*. 2007). Quantifying these exposures and exposure-response relationships in New Zealand would require considerable additional research which is outside the scope of the present project due to limited data availability. However, some of these health effects are captured by studies of short-term exposure to PM_{10} for which there is also New Zealand based evidence. Effects on respiratory morbidity in children have been included as detailed above, based on findings of Barnett *et al*. (2005).

A3.3 Conclusions and Data Used

A3.3.1 Overall Conclusions

- Given the current status of monitoring and modelling resources available for this project, ambient (outdoor) particulate matter (PM_{10}) is the best available summary indicator of air pollution exposure. This does not imply that all effects are caused by PM_{10}. Basing the update on PM_{10} rather than finer particulate matter fractions, such as $PM_{2.5}$, means that proportion of air pollution health impacts attributed to anthropogenic sources, in particular motor vehicles and domestic fires used for home heating, will be lower relative to natural sources.
- Estimates of annual average and daily average PM_{10} were developed for all 2006 census areas.
- We used NZ based studies where possible.
- We avoided double counting where the scope of this effect could not be clearly identified. Note, however, that not all of the endpoints chosen are separate. The estimates of mortality by ethnic subgroups are included in the overall estimates of mortality. The effect of short-term exposure on respiratory hospital admissions in

children is included in the estimates of respiratory hospital admissions for all age groups.

- Although there is increasing evidence that both proximity to busy roads and NO_2 exposure have important health effects, (especially respiratory symptoms in children), it is difficult to quantify these exposures in New Zealand and we did not estimate health effects based on these exposures. Consequently, the results of this update most likely under-estimate the health impacts of motor vehicle-related air pollution.
- Although exposures to benzene and CO were included in the original HAPINZ study these were excluded from our update because of the risk of double counting (for CO) and low levels combined with low exposure risk functions (for benzene).
- We assumed linear, no threshold exposure-response functions for all endpoints.

A3.3.2 Health Effects and Exposure Measures

1. Mortality from all non-external causes, ages 30 and over, all ethnicities combined, 2005-2007 (*annual average PM_{10} by CAU*).
2. Mortality from all non-external causes, ages 30 and over, separately for Māori, 2005-2007 by CAU (*annual average PM_{10} by CAU*).
3. NZ total average annual mortality from all non-external causes, ages 1 month to 1 year, *2005-2007 (population weighted NZ annual average PM_{10})*
4. Hospital admissions, cardiovascular diseases, (ICD10 chapter I) all ages, all ethnicities (*daily average PM_{10} by CAU*)
5. Hospital admissions, respiratory diseases, (ICD10 chapter J) all ages, all ethnicities (*daily average PM_{10} by CAU*)
6. Hospital admissions, respiratory diseases, (ICD10 chapter J) ages 1-4 and 5-14, all ethnicities (*daily average PM_{10} by CAU*)
7. For the estimate of restricted activity days: total NZ population, all ages (*annual average $PM_{2.5}$ by CAU*)

A3.3.3 Potential Health Effects Which Were Considered but Not Included

- Short-term effects on mortality. This impact is largely included as part of the assessment of long-term effects. Separate consideration would lead to 'double counting' which is difficult to quantify precisely.
- Adverse reproductive outcomes (low birth weight, preterm births), stroke *incidence*, chronic obstructive airways disease and asthma *incidence* were not included due to limited scientific consensus on relationships with air pollution.

- Note the distinction between incidence of disease and worsening of pre-existing disease. Effects of air pollution on stroke and on exacerbations of respiratory diseases, including respiratory infections, asthma and chronic airways diseases are *partly* accounted for in the assessments of morbidity (hospital admissions and restricted activity days: categories 4 to 7, above).
- Any effects of air pollution for which PM_{10} is a not an adequate indicator. In particular, there is increasing evidence that NO_2 exposure has important health effects, (especially respiratory symptoms in children).
- Any effects of air pollution which are specific to pollutants other than PM_{10}.

A3.3.4 Exposure-Response Functions and Target Populations

Mortality, all non-external causes: exposure-response relationship, per 10 µg/m^3 PM_{10}

1. Mortality, adults (ages 30 years and over), annual mean, all ethnicities, 7% (3% to 10%) (Hales *et al*. 2010).

This is the best available local evidence of mortality effects, and the above dose-response relationship is reasonably consistent with the international evidence on long-term effects of particle exposure.

2. Mortality, adults (ages 30 years and over), annual mean, Māori, 20% (7% to 33%) (Hales *et al*. 2010).
   Note the risk of double counting: as this is also included in the previous category.

This is the only available source of ethnically specific data. However, since the 95% confidence intervals overlap with the previous relationship, there is a small chance (of the order of 10%) that the true relationship is no different for Maori compared to other ethnicities.

3. Mortality, babies (ages 1 month-1 year), annual mean, 5% (2% to 8%) (Lacasaña *et al*. 2005).

This is a potentially important category of health impact that is included in the current international assessment.

Morbidity, exposure-response relationship, per 10 µg/ m^3 PM_{10}

4. Cardiac hospital admissions, all ages, daily mean 0.6% (0.3%-0.9%) (APHEIS 2004).

Likely to be representative of effects in New Zealand cities. Local studies are reasonably consistent, but are only available for Christchurch.

5. Respiratory hospital admissions, all ages, daily mean 1.1% (0.6%-1.7%) (APHEIS 2004).

Likely to be representative of effects found in New Zealand cities. Local studies are reasonably consistent, but are only available for Christchurch.

6. Respiratory hospital admissions, ages 1-4 years, daily mean 2% (1-4%) and ages 5-14 years, daily mean 3% (0-5%)
   (Barnett *et al.* 2005)
   Note the risk of double counting: as this is also included in the previous category.

Overall findings for Australasian cities, including PM_{10} in Christchurch.

Morbidity, exposure-response relationship, per 10 µg/ m^3 $PM_{2.5}$

7. Restricted activity days, all ages, annual mean: 0.9 (0.5-1.7) days per person (American Lung Association 1995 based on Ostro 1987).

Numerically large effect, though uncertain exposure-response relationship.

A3.3.5 Health Outcome Data Used

- Mortality: all non-external causes, ages 30 years and over, all ethnicities, NZ totals, 2005-2007
- Mortality: all non-external causes, ages 30 years and over, separately by ethnicity: Māori and non-Māori, non-Pacific by 2006 CAU, 2005-2007
- Mortality data: all non-external causes, post-neonatal, all ethnicities, NZ total
- Hospital discharges, ICD10 chapters I and J, all ages, all ethnicities, by CAU
- Hospital discharges, ICD10 chapter J, ages 1-4 years and 5-14 years, all ethnicities, by CAU

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Appendix 4: VOSL and Other Social Costs due to Air Pollution

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Executive Summary

This appendix report outlines our recommendations and justification for the following health effect costs resulting from PM_{10} air pollution, used in the updated HAPINZ study:

- Value of Statistical Life (VOSL) for premature mortality
- Social costs for morbidity effects

Key Features of the Updated Methodology

1. Adopting a transport risk (road safety) based **VOSL of NZ$3.56M** (June 2010) for all cases of premature mortality from air pollution because this is the only official value used in New Zealand and is based on a more thorough estimation procedure than any other study carried out in our country.
    - Use of the same VOSL as in transport definitely does not overestimate the social costs. Many overseas countries use VOSLs to evaluate air pollution that are much higher than their transport risk (road safety) VOSLs to account for the suffering caused by chronic illness. Therefore we have undertaken a sensitivity analysis for a VOSL twice as high as that we are recommending to reflect higher international values for environmental risk.
2. Using average costs of **NZ$6,350 and NZ$4,535** (June 2010) **for cardiovascular and respiratory hospital admissions** respectively, caused by air pollution.
    - These are based on the average length of stay in hospital assuming that the average cost per day of hospitalisation is the same for all diseases, in the absence of more detailed information. The costs include medical costs plus loss of output during (but not after) the stay and so are likely to be at the lower end. Therefore we have undertaken a sensitivity analysis using a range of likely loss of life quality and a variation in medical costs.
3. Using a value of **NZ$62** (June 2010) for **restricted activity days** (RADs).
    - This value is based on the average loss of output per day (irrespective of a working or non-working day) and does not include additional costs associated with working care givers having to stay at home to look after sick relatives. Therefore we have undertaken a sensitivity analysis using a range of possible output losses.

A4.1 Background on VOSL, QALY and Health Effects of Pollution

Air pollution results in mortality and morbidity health consequences. This amounts to loss of life and life quality of people exposed to pollution. The health consequences of pollution depend on the level and type of pollution. Once the health consequences are estimated, our task here is to estimate the total cost to society resulting from these consequences.

This includes the values of loss of life and life quality and also the associated resource costs, mainly the costs of required medical treatment and rehabilitation. The costs to society of loss of life and life quality account for the major part of the total social cost.

Once the health effects of pollution changes are estimated, we need to translate them into equivalent monetary costs or benefits to society resulting from those changes. This discussion includes ways of measuring social costs of loss of life and life quality of health effects due to pollution.

A4.1.1 Measuring Loss of Life

Depending on severity of pollution effects, people exposed to pollution may suffer from some diseases and health conditions resulting in death. In some cases, though health is affected, the person may not die from pollution related health effects but due to some other causes. In that case the loss to society is the value of loss of life quality before death.

Since death is inevitable to a person, it is the premature death that is of concern to us in terms of estimating the cost to society of loss of life.

A premature death means there is loss of life years and hence it is often argued that only the life years lost should be valued and not the loss of life itself. Loss or gain of life years is commonly used in health sector evaluations. Since the value of a life year should vary by status of health of the person, it is common to use a standardised value known as quality adjusted life year or QALY. There are other measures such as health adjusted life year (HALY) or disability adjusted life year (DALY)[19]. QALY concept is more comprehensive and also more widely used in health project evaluations. So we discuss QALY in more details here.

QALY of a life year is a measure of life year adjusted for quality of health or health status, relative to perfect health. For a year in perfect health, the QALY is equal to 1. If the quality of health is assessed as equivalent to only four-fifths of perfect health, then a year of life can be measured by 0.8 QALY.

Following this analogy, a loss of life can be measured by QALYs lost due to death. It is often argued that cost to society of a loss of life should be the value of consequent loss of

[19] For a discussion, see Gold e*t al*. (2002).

QALYs. In this appendix we discuss usefulness and weaknesses of using QALYs in cost benefit analysis.

A4.1.2 Valuing Avoidance of Death

Like any other risk area, pollution also increases the risk of death to those exposed to pollution. The value of a change in risk is generally measured by society's willingness to pay (WTP) for that change. The amount of money a society is willing to pay to reduce the risk of death so that one premature death is prevented is known as WTP based value of statistical life (VOSL).

The official VOSL in New Zealand used by the transport sector and many others is regularly updated by the Ministry of Transport. The VOSL at June 2010 prices is $3.56 million (MoT 2010).

The VOSL is not a value of an identified individual life. It is the value society as a whole is willing to pay to reduce the risk of death so that one premature life is saved or, perhaps more accurately, a premature death is prevented. The VOSL is also often referred to as value of preventing a statistical fatality or VPF.

In the first HAPINZ study (Fisher *et al*. 2007), the value used for prevention of mortality was $750,000 at 2004 prices when the VOSL used by the Ministry of Transport was $2.725 million. This was based on the assumption that about five years of life would be lost on average when a person died due to air pollution.

The estimated value was derived based on the assumption that the average age at death in road traffic crashes was 35 years and the average loss of life years was 44 years. Using a 6 per cent discount rate, the study estimated the value per life year from the assumption that the discounted present value over 44 years would be $2.725 million. That would give the discounted present value over five years (only) at 6 per cent discount rate of about $750,000. Fisher *et al*. (2007) used this as the value to society of preventing one premature death.

We consider the approach used previously inappropriate because the value per life year is not necessarily constant. If it was constant, the VOSL would decline by age. We have no evidence of a decline trend of VOSL by age. Besides, once a person is diagnosed with a heart disease or cancer, the level of trauma suffered by the person and their close ones are very high. In addition, an OECD study recommends use of the same value for all ages (OECD 2010). We have used the official VOSL for the value to society of preventing one air pollution-related mortality. This is further discussed below.

A4.1.3 Loss of Life Quality

A reduction in pollution would reduce the risk of death and suffering and hence save lives and life qualities that would otherwise be lost. Some researchers strongly argue, e.g. Brunekreef *et al*. (2007), that death is unavoidable. Therefore, what happens is the change in life expectancy and thus the effect of pollution reduction is the gain in quality

adjusted life years. For avoidance of death, it is not just the number of life years lost but also the loss of life quality during the period of suffering from the disease resulting from pollution exposure.

It is often argued that even for suffering from chronic diseases, deaths occur only at an old age and hence number of life years lost is small. While this may be true, the loss of life quality since a person was affected by a chronic disease (such as cardiopulmonary, cardiovascular, respiratory disease or lung cancer) can be quite high.

Poorer health conditions may affect a person's life in many ways, such as reduced ability to work and hence loss of income, reduced ability to participate in sports, recreation and other social activities and these may also have psychological impact on the person's day to day quality of life. These are inter-related and hence complex to measure individually and adding them up to have a total picture. An alternative way is to measure it relative to normal healthy life or equivalent life years lost.

One way to measure the loss of life quality is to find the reduction in life quality relative to normal healthy life. In this case, the value is expressed as a percentage of VOSL. This is for permanent loss of life quality. If it is a temporary loss of life quality, the loss is much lower and can be ignored in most cases. An alternative approach is to measure the loss of QALYs. This is very commonly used in cost effectiveness analysis of health policies/treatments. There are problems in estimating equivalent monetary values to society and that is discussed next.

A4.1.4 Valuing QALY

In health sector evaluations, especially in cost-effectiveness analysis, it is common to compare treatments by money spent per QALY gained under each treatment. More information is needed in a cost benefit analysis. Here it is necessary to estimate the value of QALYs gained in monetary terms.

In a cost effectiveness analysis, it is an implicit assumption that all QALYs are equal and hence a treatment which costs less per QALY gained is preferable. A cost benefit analysis requires monetary value of QALYs gained to compare with the cost of providing the treatment.

Some studies consider WTP as an alternative to using QALY. As noted by Hammitt (2002, p 985),

> *"QALYs are used routinely in the medical and public-health fields, whereas WTP is widely used in evaluating environmental and transportation-related risks".*

Where the health effect is measured by QALY and equivalent monetary value is required for policy decisions (including cost benefit analysis), it becomes necessary to determine society's willingness to pay for QALY gains. In this exercise, it has been common to estimate values per QALY either directly from WTP surveys or most commonly from the VOSL being used. This idea of constant value per QALY at all ages and other circumstances can be challenged.

There are few studies estimating value of a QALY directly, i.e. the society's willingness to pay to gain a QALY. Gyrd-Hansen (2003) estimates WTP based value per QALY from a survey using elicited preferences for health states in the questionnaire. However, she raises a few theoretical and methodological issues with WTP for QALY in a later paper (Gyrd-Hansen 2005). Gyrd-Hansen argues that because cost effectiveness analysis where QALY plays an important role is based on maximising health and not welfare, a linear translation from QALY to WTP is theoretically unattainable. A second obstacle considered by Gyrd-Hansen is that since "*marginal utility of income is non-constant, and a function of income level and possibly health status*" (p 423), there cannot be a one unique WTP per QALY. Van Houtven *et al*. (2006) also come to the same conclusion based on a meta-analysis that the WTP per QALY should not be constant.

A4.1.5 QALY Derived from VOSL

The value per QALY is frequently estimated from a given VOSL. Since WTP based VOSL is often based on road travel risks, it is assumed to be the discounted present value of QALYs lost in road deaths. Using the average life years lost per road death, researchers in New Zealand and elsewhere have estimated the average value per QALY. This value then has been used for all ages.

There are mainly two problems in this approach. First it assumes that VOSL varies directly with QALY at a given age. Secondly, value per QALY is the same for all. The first point assumes that the value of marginal improvement in health status is the same irrespective of the base health status. The second point indicates that the value per QALY remains the same irrespective of age, sex, ethnicity and any other characteristic of the person.

A4.1.6 VOSL and QALY by age

If the value per QALY remains the same, the VOSL should have a downward sloping relationship with age. Two WTP studies on VOSL have been carried out in New Zealand. Neither of these two studies showed a relationship with age, though the first study (Miller and Guria, 1991) indicated a slightly lower value for those over 60 years of age. The second study (Guria *et al.*, 2003), which was more comprehensive, did not show any such relationship. However, some other studies show inverted U shaped relationships between VOSL and Age (Jones-Lee *et al*. 1985, Carthy *et al*. 1999, Aldy & Viscusi 2008).

If the value of a QALY is the same for all ages, then the VOSL for children should be considerably higher than the average. Since parents with small children are usually young with relatively low disposable income, the VOSL for such families was found to be smaller than those without small children in the New Zealand study carried out in 1997-98, the second study mentioned above (Leung & Guria 2006). Some studies find higher values for children but there is no consistent trend. Considering this, an international workshop involving expertise from Austria, Switzerland, France, Sweden, The Netherlands and Malta recommended in 2004 that the same value should be used for children as for adults until child specific values are available (Estreen & Friberg 2004).

None of the studies mentioned above supports the view of a constant value per QALY for all ages. Mason *et al*. (2009) estimates QALY for the downward sloping part of the VOSL curve, i.e., above 40 years of age. Following their approach, the value per QALY would be negative for the other part because VOSL increases when life expectancy decreases. Mason *et al*. justify their findings with the explanation that:

> *"the tendency for the valuation of safety to rise with age (and hence decline with remaining life expectancy) over early years of adult life, even when income effects are controlled for, is in fact a reflection of a fundamental change in attitude to risk and awareness of vulnerability to physical harm that would appear to be a common feature of the process of maturation for many people over the period from their late teens to their mid-20s. It would therefore seem reasonable to argue that since the VPF-age relationship over early years of adulthood is largely a result of a fundamental change in preferences and attitudes rather than a change in an individual's future hazard rates then estimation of the value of a gain in life expectancy should be based only on the time interval over which the VPF is a decreasing function of age" (p 938-9).*

It is difficult to accept this explanation. A more appropriate would be that such a value cannot be established for the upward sloping part of the VOSL-age relationship indicating the likelihood of variation of value per QALY by age, as shown by Aldy and Viscusi (2008). Even for the downward sloping part of the VOSL-age relationship, it is not obvious that the value per life year or per QALY should be the same.

Because of difficulties in estimating social value per QALY appropriately, VOSL (for the data available in New Zealand) is the preferred approach.

A4.1.7 Using Road Safety-based VOSL

It is often argued that VOSL may vary with risk environment, i.e., it can be different for different risk scenarios. In New Zealand, the WTP based VOSL established in 1991 and updated since then is being used for evaluation of transport safety programmes and policies and in many other areas. Our understanding is that this is the only official VOSL in New Zealand.

One study (BERL 2007) showed a lower value for fire safety relative to VOSL used in the transport sector. It is not clear to what extent this is a real reflection of risk environment. It could be an effect of the questionnaire design and its interpretation. The VOSL used in the transport sector is based on a WTP survey in which respondents had the option of valuing risk changes in realistic situations. It is a very important aspect of such (contingent valuation) studies. Comparing contingent valuation results with actual behaviours, Mitchell and Carson (1989) find that questions must be realistic to get realistic responses. Changed situations, for example, changes in base risk levels, real income distribution, etc. can provide different estimates as found in the two New Zealand studies (Miller & Guria 1991, Guria *et al*. 2003) with a gap of almost ten years.

The current WTP-based VOSL used for transport safety evaluations was based on people's willingness to pay for improvement in road safety risks. Internationally it is common to

use this value in some other areas. Mason *et al*. (2009) report that the UK Department for Transport (DfT), the Rail Industry, the Department for the Environment, Food and Rural Affairs (DEFRA) and other government agencies in the UK use the same VOSL originally estimated using transport risks.

As noted by NZIER (2009) in response to a question raised by a reviewer of the NZIER report on validity of using transport based VOSL for evaluating mortality risk changes from air quality improvements, there is no particular reason why the VOSL in the present context should differ drastically from VOSL estimated from traffic crash risk changes, except the possibility for higher values due to prolonged pain and suffering before death in some cases. Besides, it is not practical to estimate the WTP based VOSL for every risk environment due to the very high cost of conducting contingent valuation surveys. In some cases, relativity between WTP for different risk environments is estimated. Here also care should be taken on presenting realistic situations to the respondents as we have noted above and also emphasised in the NZIER report. Some such studies were conducted in the UK but as noted in Mason *et al*. (2009), different departments appear to be using the same values.

A4.1.8 Length of Suffering

In road traffic crashes most fatalities occur within a short-time period from the crash, whereas in many other areas victims suffer for a while before death occurs. Studies show that people are more willing to pay to save a cancer related death than a pedestrian death (Rowlett *et al*. 1998, Clinton *et al*. 2007). This is also reflected in the recommendation of the UK Health and Safety Executive that the same VOSL should be in all areas with the exception of prevention of cancer deaths, in which case the value should be twice as much (Mason *et al*. 2009).

As noted in NZIER (2009), pollution exposure results in some chronic diseases and many suffer for a long time before death. In such cases, the social costs of death can be much higher than social costs of death due to traffic crashes. It is unsurprising that in many jurisdictions the VOSL used for evaluating prevention of mortality due to environmental effects is much higher than the value used for prevention of mortality due to traffic crashes.

The VOSL used in evaluation of environmental effects in many countries are in fact higher than that used in transport risk change evaluations. The VOSL used by the US Environmental Protection Agency (EPA) is US$7.2 million at 2006 prices (National Center for Environmental Economics 2011) whereas the value used by the Department of Transport (DoT) is US$6 million at 2009 prices (US DoT 2009). The VOSL is updated from time to time by the DoT and it seems there is no system of updating the value on a regular basis as in New Zealand. The value was updated from US$3 million to US$5.8 million in

2008 and then to US$6 million in 2009 and as we understand, the same value is being used at present[20].

In Australia, the VOSL used by the Bureau of Infrastructure, Transport and Regional Economics is $2.67 million at 2006 prices (BITRE 2009). However according to this report, the Australian Office of Best Practice Regulation (2008) recommended a value of $3.5 million based on WTP based international estimates and Australian research. On the other hand, as noted by the Regulatory Impact Statement for amending the PM_{10} Air Quality Standards (MfE 2011), the Australian National Environment Protection Council recommends a value of $6 million at 2006 prices based on a meta-analysis and a sensitivity analysis using a range of $5 million - $7.1 million (NEPC 2009), based on the Access Economics (2008) report. While many countries use WTP based VOSL only limited information is available on separate values for environmental effects and transport safety. Many countries use the same value for convenience or in the absence of separate estimates. Values used in a few other countries are shown in Annex A.

Ministry of Transport (2009) carried out a meta-analysis to compare NZ VOSL with that of a few countries. This study compared values with respect to per capita GDP of these countries (varying VOSL proportionately with per capita GDP) and also based on a relationship (VOSL as a function of per capita GDP and traffic fatality risk per unit of population) developed in the study.

A4.2 Valuations for Mortality and Morbidity Effects

A4.2.1 Valuation of Premature Mortality

In a recent study on benefit cost analysis for amending the PM_{10} Air Quality Standards, NZIER (2009) used the VOSL being used by the Ministry of Transport for evaluation of safety programmes and policies. Some reviewers raised two issues with this approach. Their view was that many suffering from PM_{10} exposure related health effects die at a very advanced age and hence lose relatively low life expectancy. Secondly, the value for prevention of mortality may not be the same as for prevention of traffic crash related mortality.

As we have mentioned earlier, though some studies in the past have found lower values for older age groups, the two New Zealand studies did not establish any significant relationship between age and VOSL. Secondly, with increase in life expectancy and people's ability to be productively active for longer period and the level of endowments they acquire, the amount of money they would be willing to pay to reduce their risk of death is not necessarily lower than that of the younger population. An OECD study suggests that there is no evidence that VOSL vary with age. Besides, they report that a study found those above 60 are willing to pay more (OECD 2010).

[20] An e-mail from US DoT confirmed this on 9 February 2011.

Another point and perhaps a more important point to note here is that for many dying at an old age the loss is not just a few years of life expectancy but also the suffering before death. Considering this it is unlikely that the value to society of such deaths is any lower than the value considered for the average population. In some cases, as recommended by the UK Health and Safety Executive, the VOSL should be higher.

On the second issue, we would like to emphasise on two factors affecting the choice of VOSL. First, transport risk (road safety) based VOSL estimate is the only official value used in New Zealand and also it is based on more thorough estimation procedure than any other study carried out in New Zealand. Secondly, as we discussed above, the willingness to pay to avoid many diseases caused by pollution exposure is very likely to be higher and hence the VOSL used for this purpose should also be higher. Use of the same VOSL as in transport safety definitely does not overestimate the social cost. It most likely underestimates the social cost. The more appropriate VOSL for environmental effects should be higher and that requires a separate study. In its absence we use the same updated value as done by the Ministry of Transport. The latest updated value is $3.56 million at June 2010 prices.

Because some overseas countries adopt an environmental risk VOSL of twice the transport risk (road safety) VOSL, we have conducted a sensitivity analysis using the base case ($3.56 million) and double this as an upper bound ($7.12 million). As mentioned earlier the base case figure is likely to be an under-estimate so there is no need to assess a lower value of VOSL.

A4.2.2 Valuation of Morbidity Effects

All who suffer health effects do not necessarily die from those health effects. In that case, they suffer loss of life quality while alive since affected but from a different cause. In traffic crashes these are measured by non-fatal injuries. These injuries are mainly divided into serious and minor injuries. The loss of life quality due to serious injuries is measured as 10 per cent of VOSL on average and that for minor injuries by 0.4 per cent. The serious injuries are mostly injuries requiring hospitalisation. Minor injuries are cuts and bruises normally not expected to have long-term effects but some injuries, e.g., whiplash, though initially considered minor cause long-term effects later on. There may not be any equivalent health effects in the present contexts. However, there can be some health effects which may not be life threatening but cause loss of life quality. We are not aware of studies quantifying such effects. There are a few studies on loss of life quality due to asthma (e.g., Abelson 2003, Tolley *et al*. 1994, Mathers *et al*. 1999)

The NZIER (2009) considered only the likely medical costs and loss of output for hospitalised cases. Since average length of hospitalisation was less for PM_{10} related health effects, the medical cost was considered to be proportionately lower, though the actual cost may not necessarily be proportional. This aspect needs to be further looked at because it could be significantly underestimating costs. In absence of more precise estimates, however, we agree with the NZIER view and use the same method for estimating the social costs.

Though loss of life quality was not considered in the main estimates, the NZIER (2009) study carried out a sensitivity analysis with 10 per cent loss of life quality. At $3.56 million VOSL, the loss of life quality of hospitalised cases would be $356,000. These give two extreme values of social costs due to hospitalised health effects. In reality at least some of these hospitalised cases would have long-term health problems without threat to life.

The average loss of life quality may vary with length of hospitalisation as found in Guria (1990). However, no clear relationship between the two factors has been established.

Another consideration is that hospitalisation following a traffic crash treats the patient for the injury and may not require repeated hospitalisations. For a pollution-related disease, hospitalisation or continuous treatment over a long period of time or even the life time is more likely than due to traffic injuries. Therefore, length of initial hospitalisation may not provide a good indication of the seriousness of loss of life quality. It is likely to give a lower end estimate.

A4.2.3 Cost per Hospitalisation

The total costs per hospitalisation are a combination of the medical costs and the loss of output only incurred for each health effect – in this case respiratory admissions and cardiovascular admissions.

Medical costs per hospitalisation

The NZIER (2009) report estimated the average medical cost per hospitalisation as $7,700 at 2008 prices. This was based on average length of hospitalisation of 12.6 days for traffic accidents and 6.8 days for PM_{10} pollution (MfE 2004). New estimates suggest that the average length of hospitalisation for respiratory and cardiac admissions (potentially due to PM_{10} pollution) is only about 4.2 days. Assuming that the emergency costs would be the same as for hospitalised injuries and the costs of hospitalisation and follow on would be proportional to the number of days in hospital, as done by NZIER (2009), the average medical cost per hospitalised case would be about $5,233[21], using Ministry of Transport (2010) estimates.

Our estimates for average length of hospitalisation further indicate that it is 5 days for cardiovascular diseases and 3.3 days for respiratory diseases. Assuming that the average cost per day in hospital remains the same, the total medical cost for these two diseases would be about $6,040 and $4,330 respectively for cardiovascular and respiratory diseases. The same average cost has been assumed in the absence of better information. The possibility of substantially different estimates cannot be ruled out.

[21] Using the MoT 2010 value of $8,500 (medical cost)*4.2/12.6 plus $1,000 (emergency cost) plus $4,200 (follow on cost)*4.2/12.6.

Loss of output during hospitalisation

The days in hospital include both working and non-working days. So the average loss per day is estimated as the average weekly income divided by 7. Statistics New Zealand survey shows that the average income from these two sources for people aged 15 years and over is $557 at June 2010 prices. The total number of people in this range for whom this average is estimated is 3,421,300. This gives a total income of $1,906 million per week. For a total population of 4,367,700, this provides an average income per person per week as $436. This gives $62 per day (irrespective of a working or non-working day).

Therefore the average loss of output per cardiovascular hospital admissions is $310 (=5*$62) and per respiratory hospital admission is $205 (=3.3*$62).

Total costs per hospitalisation

The total costs for medical costs and loss of output (during hospitalisation) are $6,350 and $4,535 at June 2010 prices for cardiovascular and respiratory diseases respectively. These are approximate estimates in the absence of more detailed information.

The loss of output after hospitalisation has not been included in the above estimates. In some cases, this can be high. Thus the estimates are likely to be on the lower side.

As discussed previously, the length of initial hospitalisation may not provide a good indication of the seriousness of loss of life quality. Therefore a sensitivity analysis has been undertaken assuming up to 10 per cent loss of life quality which for hospitalised cases would be $356,000, based on the current VOSL of $3.56 million.

A4.2.4 Restricted Activity Days

The NZIER (2009) report assumes that only wages and salaries and income from self-employment would be lost during the restricted activity days (RADs).

RADs are estimated for the whole population, not just those employed. This includes adults (both employed and unemployed) as well as children. So RADs refer to loss of activity related to earning as well as non-earning time. Thus average loss per RAD is related to the average loss per affected person.

We follow the same methodology as for loss of output during hospitalisation to estimate the loss of income per RAD. This gives $62 per day (irrespective of a working or non-working day) and assume that the loss per RAD applies to the whole day on average.

In reality the loss could be less or more depending on the circumstances. A previous cost benefit analysis (MfE, 2004) assumed that each RAD would amount to a loss of 55 per cent work time on average (based on 90 per cent of RADs causing minor restrictions and the remaining 10 per cent causing major loss of work). On the other hand, a caregiver or parent may have to take time off from work to support the person affected by the RAD, particularly in cases involving children or the elderly, resulting in additional cost. Therefore, we have adopted limits for the sensitivity testing of $34 (=$62*0.55) and $87 (=$436÷5).

A4.3 Conclusions

To measure the social costs of pollution related health effects, we discuss three main components:

- loss of life and life quality
- costs of medical treatment and
- loss of output.

The social cost of loss of life will be estimated by the official VOSL in New Zealand which is based on society's willingness to pay for safety improvement of road traffic. We have discussed the alternative approach of estimating QALY and then the equivalent monetary value. The main problem is that the value per QALY is unlikely to remain constant for all. Secondly, people's willingness to pay for two QALY is not necessarily twice the value for one QALY. In our view it is wrong to treat VOSL as a discounted present value of amounts people would be willing to pay for each of the future life years. Studies clearly show that the variation in VOSL by age is small and some studies also show that older people have higher willingness to pay. Thus the VOSL does not necessarily diminish by age which is an essential requirement for value per QALY to remain constant and also invariant by age.

The magnitude of average long-term health effects (i.e., permanent loss of life quality for those who suffer health problems but die due to other causes) is not yet known. For this, following the NZIER (2009) approach, only loss of output and costs of medical treatment will be considered for lower end estimate and additional 10 per cent loss of life quality at the higher end as used by the Ministry of Transport for hospitalised injuries.

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Annex A4A: VOSL in Different Countries

The information on VOSL used by different countries is available for specific years. Even here the values are often expressed in prices of previous years as shown in Table A4-1.

Table A4-1: VOSL in different countries

Country	Year	VOSL		Source
		Transport	Environment	
USA	2010	US$6.0M[a]	US$7.2M[b]	a: US Department of Transport (2009) b: National Center for Environmental Economics (2011)
Australia	2009	A$2.67M[c]	A$6M[d]	c: BITRE (2009) for transport d: Environment VOSL recommended by Australian National Environment Protection Council (NEPC 2009)
Canada	2011	C$6.7M[e]		Health Canada (e-mail correspondence)
UK	2008	£1.68M		NERA Economic Consulting (2011).
Austria	2006	€2.68M		Dahdah and McMahon (2008). No separate values for transport and environmental effects are available.
France	2005	€1.16M		
Germany	2004	€1.16M		

Note: a: at 2009 prices; b: at 2006 prices; c: at 2006 prices; d: at 2006 prices; e: at 2007 prices

Conversion of values in NZ dollars at 2009 prices

A problem with Table A4-1 is that the VOSLs are expressed in different currencies and also refer to prices of different years. This causes problems in comparing them. To avoid complication, we convert all values to prices of one particular year and express them in NZ$.

Different countries update their values in different ways. For example, NZ VOSL is updated by indexing it to ordinary time wage rate. In the UK it is updated by per capita GDP. Here we have updated values which would indicate what would be the values if they were all indexed to per capita GDP. Due to availability of per capita GDP of countries included in Table A to only up to 2009, we have updated all values to 2009 prices, using per capita GDP as the indexing factor. We assume here that VOSL would change directly with per capita GDP. This may not indicate the exact value used by different countries but provides a basis for reasonable comparison.

Conversion of all values to 2009 prices

If VOSL in a country (C) in one year (say 1) is V1, then in another (say 2), it is estimated as:

$V_2 = \frac{V_1 G_2}{G_1}$ where G1 and G2 are per capita GDP in year 1 and year 2 respectively.

The per capita values are obtained from The World Bank (2011).

Conversion of all values to 2009 NZ$

V2 is in the currency noted in Table A4-1. Now we want to express all values in comparable NZ dollar. The exchange rate does not always indicate the right purchasing power of the currency. For better comparison between countries, exchange rate is also expressed in purchasing power parity (PPP). The estimated VOSL in the country's currency can now be expressed in NZ PPP by multiplying V_2 by the ratio of the PPP of NZ and the country C.

Now the VOSL V_2 in country C can be expressed in NZ$ as:

$$V_2^{NZ} = V_2 \frac{PPP_2^{NZ}}{PPP_2^{C}} = \frac{V_1 G_2}{G_1} \cdot \frac{PPP_2^{NZ}}{PPP_2^{C}}$$

where PPP_2^{NZ} and PPP_2^{C} are values of 1 US$ in PPP NZ$ and in C's currency respectively. The PPP exchange rate data are from OECD (2011).

Comparison of VOSLs from different countries in NZ$ PPP

The estimated VOSL at 2009 in NZ$ PPP are as shown in Table A4-2.

Table A4-2: VOSL in different countries in 2009 NZ$ PPP

Country	VOSL in 2009 NZ$ PPP	
	Transport	Environment
New Zealand	$3.50M	
USA	$9.0M	$11.1M
Australia	$3.2M	$7.1M
Canada	$8.0M	
UK	$3.5M	
Austria	$5.4M	
France	$2.5M	
Germany	$2.6M	

Appendix 5: Emissions Inventory and Monitoring Data References

Prepared by Emily Wilton (Environet Ltd)

This appendix report summarises all of the emissions inventories and monitoring datasets used in the HAPINZ update.

A5.1 Emission Inventories

Area	Year	Reference	Notes
Whangarei	2006	Wilton E (2006). Air Emission Inventory - Whangarei 2006, Northland Regional Council Technical Report.	
Auckland	2006	Metcalfe J *et al (pending*). Auckland Regional Air Emissions Inventory 2006 - unpublished but data being supplied by ARC	
Huntly	2009	Wilton E & Baynes M (2009). Emission Inventory Taupo, Thames and Huntly 2009, Environment Waikato Technical Report 2010/13.	
Ngaruawahia	2006	Wilton E (2006). Emission Inventory for Te Awamutu, Turangi and Ngaruawahia 2006, Environment Waikato Technical report 2006/43.	
Matamata	2006	Smith & Wilton (2007). Air Emission Inventory Matamata, Putaruru and Waihi, Environment Waikato Technical Report 2007/13, ISSN 1172-4005.	
Hamilton	2005	Wilton E (2005). Hamilton Emission Inventory 2005, Environment Waikato Technical report 2005/R52, ISSN: 1172-4005.	
Te Awamutu	2006	Wilton E (2006). Emission Inventory for Te Awamutu, Turangi and Ngaruawahia 2006, Environment Waikato Technical report 2006/43.	
Tokoroa	2007	Wilton E & Baynes M (2008), Emission Inventory Tokoroa and Te Kuiti 2007, Environment Waikato Technical Report 2008/02.	
Putaruru	2006	Smith & Wilton (2007). Air Emission Inventory Matamata, Putaruru and Waihi, Environment Waikato Technical Report 2007/13, ISSN 1172-4005.	
Te Kuiti	2007	Wilton E & Baynes M (2008). Emission Inventory Tokoroa and Te Kuiti 2007, Environment Waikato Technical Report 2008/02.	
Thames	2009	Wilton E & Baynes M (2009), Emission Inventory Taupo, Thames and Huntly 2009, Environment Waikato Technical Report 2010/13.	
Turangi	2006	Wilton E (2006) Emission Inventory for Te Awamutu, Turangi and Ngaruawahia 2006, Environment Waikato Technical report 2006/43.	
Taupo	2009	Wilton E & Baynes M (2009). Emission Inventory Taupo, Thames and Huntly 2009, Environment Waikato Technical Report 2010/13.	
Waihi	2006	Smith & Wilton (2007). Air Emission Inventory Matamata, Putaruru and Waihi, Environment Waikato Technical Report 2007/13, ISSN 1172-4005.	
Rotorua	2005	Iremonger S. & Graham B (2006). Rotorua Emission Inventory 2005, Environment Bay of Plenty Technical Report 2007/02.	
Gisborne	2005	Wilton E (2005). Warm Homes Technical Report: Home heating methods and fuels in New Zealand, Ministry for the Environment Technical report 701.	Domestic Only

Area	Year	Reference	Notes
Napier	2005	Wilton E (2005). Air Emission Inventory - Hawkes Bay Region - 2005, Hawkes Bay Regional Council Technical Report.	
Hastings	2005	Wilton E (2005). Air Emission Inventory - Hawkes Bay Region - 2005, Hawkes Bay Regional Council Technical Report.	
Havelock North	2005	Wilton E (2005). Air Emission Inventory - Hawkes Bay Region - 2005, Hawkes Bay Regional Council Technical Report.	
Taihape	2010	Wilton E & Baynes M (2010). Air Emissions Inventory - Taumarunui and Taihape 2010, Horizons Regional Council Technical Report.	
Taumarunui	2010	Wilton E & Baynes M (2010). Air Emissions Inventory - Taumarunui and Taihape 2010, Horizons Regional Council Technical Report.	
Upper Hutt	2006	Wilton E & Baynes M (2006), Air Emissions Inventory - Wainuiomata and Upper Hutt 2006, Greater Wellington Regional Council Technical Report.	
Masterton	2008	Wilton E & Baynes M (2008). Air Emissions Inventory - Masterton 2008, Greater Wellington Regional Council Technical Report.	
Wainuiomata	2006	Wilton E & Baynes M (2006). Air Emissions Inventory - Wainuiomata and Upper Hutt 2006, Greater Wellington Regional Council Technical Report.	
Richmond	2005	Wilton E (2005). Richmond Emission Inventory 2004, Tasman District Council Technical Report.	
Nelson A	2006	Wilton E (2007). Air Emission Inventory Nelson 2006, Nelson City Council Technical Report.	
Nelson B1	2006	Wilton E (2007). Air Emission Inventory Nelson 2006, Nelson City Council Technical Report.	
Nelson B2	2006	Wilton E (2007). Air Emission Inventory Nelson 2006, Nelson City Council Technical Report.	
Nelson C	2006	Wilton E (2007). Air Emission Inventory Nelson 2006, Nelson City Council Technical Report.	
Blenheim	2005	Wilton E (2005). Blenheim Air Emission Inventory 2005, Marlborough District Council Technical Report.	
Reefton	2005	Wilton, E (2006). Reefton Air Emission Inventory 2005, West Coast Regional Council Technical Report.	
Westport	2005	Wilton E & Baynes M (2008). Westport Emission Inventory, Technical report prepared for Holcim NZ Limited.	
Rangiora	2007	Smithson J (2008). Inventory of emissions to air in regional Canterbury towns, 2007, Environment Canterbury Report R08/96.	
Kaiapoi	2007	Smithson J (2008). Inventory of emissions to air in regional Canterbury towns, 2007, Environment Canterbury Report R08/96.	
Christchurch	2006	Smithson J (2008). Inventory of emissions to air in Christchurch 2006, Environment Canterbury Report R08/70 ISBN 978-1-86937-885-1.	
Ashburton	2007	Smithson J (2008). Inventory of emissions to air in regional Canterbury towns, 2007, Environment Canterbury Report R08/96.	
Timaru	2008	Smithson J (2010). Inventory of emissions to air in Timaru and Washdyke 2008, Environment Canterbury Report.	
Geraldine	2007	Smithson J (2008). Inventory of emissions to air in regional Canterbury towns, 2007, Environment Canterbury Report R08/96.	

Area	Year	Reference	Notes
Waimate	2007	Smithson J (2008). Inventory of emissions to air in regional Canterbury towns, 2007, Environment Canterbury Report R08/96.	
Oamaru	2005	Wilton E & Baynes M (2008). Oamaru Emission Inventory, Technical report prepared for Holcim NZ Limited.	
Alexandra	2005	Wilton E (2006). Air Emission Inventory - Dunedin, Mosgiel and Alexandra 2006, Otago Regional Council Technical Report.	No MV or OB
Cromwell	2005	Wilton E (2005). Warm Homes Technical Report: Home heating methods and fuels in New Zealand, Ministry for the Environment Technical report 701.	Domestic Only
Arrowtown	2005	Wilton E (2005). Warm Homes Technical Report: Home heating methods and fuels in New Zealand, Ministry for the Environment Technical report 701.	Domestic Only
Dunedin	2005	Wilton E (2006). Air Emission Inventory - Dunedin, Mosgiel and Alexandra 2006, Otago Regional Council Technical Report.	No MV or OB
Mosgiel	2005	Wilton E (2006). Air Emission Inventory - Dunedin, Mosgiel and Alexandra 2006, Otago Regional Council Technical Report.	No MV or OB
Milton	2005	Wilton E (2005). Warm Homes Technical Report: Home heating methods and fuels in New Zealand, Ministry for the Environment Technical report 701.	Domestic Only
Balclutha	2005	Wilton E (2005). Warm Homes Technical Report: Home heating methods and fuels in New Zealand, Ministry for the Environment Technical report 701.	Domestic Only
Gore	2005	Wilton E (2005). Invercargill and Gore Air Emissions Inventory, Environment Southland Technical Report.	
Invercargill	2005	Wilton E (2005). Invercargill and Gore Air Emissions Inventory, Environment Southland Technical Report.	

A5.2 Source Apportionment

Area	Year	Reference
Blenheim	2006	Wilton E & Trompetter B (2007). Source Apportionment of PM_{10} in Blenheim, Marlborough District Council Report.
Hastings	2006/07	Wilton E, Davy P,& Smith J (2007). Source Identification and Apportionment of PM_{10} and $PM_{2.5}$ in Hastings and Auckland, NIWA Client Report prepared for FRST.
Napier	2009	Wilton E, Baynes M, & Zawar Resa P (2010). Source Apportionment of Particulate in Napier, Hawkes Bay Regional Council report - Envirolink 869-HBRC130.
Masterton	2002/04	Davy PK (2007). Composition and Sources of Aerosol in the Wellington Region of New Zealand, PhD thesis, Victoria University of Wellington.
Tahunanui	2008/09	Davy P, Trompetter B, & Markwitz A (2010). Source Apportionment of PM_{10} at Tahunuanui, Nelson, GNS Science Consultancy Report 2010/198.
Auckland	2006/07	Wilton E, Davy P,& Smith J (2007) Source Identification and Apportionment of PM_{10} and $PM_{2.5}$ in Hastings and Auckland, NIWA Client Report prepared for FRST. Petersen J, pers comm. (2010). Provision of Auckland source apportionment data, unpublished.

Area	Year	Reference
Christchurch	2001	Scott A. (2005). Source apportionment and chemical characterisation of airborne fine particulate matter in Christchurch, New Zealand, PhD thesis, University of Canterbury. Scott A, pers comm. (2010). Provision of Christchurch source apportionment data.
Wainuiomata	2006/08	Davy P, Trompetter B, & Markwitz A (2009). Source apportionment of airborne particles at Wainuiomata, Lower Hutt, GNS Science Consultancy Report 2009/188.

A5.3 Industry Allocations

Area	Year	Reference
Nationwide	2007	Wilton E, Baynes M & Iseli J (2008). New Zealand Sulphur Dioxide Industrial Emissions Inventory 2007, Environet report prepared for the Ministry for the Environment. Lawrence K. National assessment of industrial SO_2 emissions in New Zealand, Pacific Air and Environment Report 2726, provided by MfE, unpublished.
Waikato Region	2003	Wilton E (2004). Regional Energy Survey 2003, Environment Waikato Technical Report TR05/31.
Ashburton	2007	Wilton E, Baynes M, Anderson B, & Iseli J (2007). Cost effectiveness of policy options for boilers - Ashburton, Environment Canterbury Report.
Christchurch	2007	Wilton E, Baynes M, Anderson B, & Iseli J (2007). Cost effectiveness of policy options for boilers - Christchurch, Environment Canterbury Report.
Kaiapoi	2007	Wilton E, Baynes M, Anderson B,& Iseli J (2007). Cost effectiveness of policy options for boilers - Kaiapoi, Environment Canterbury Report.
Rangiora	2007	Wilton E, Baynes M, Anderson B, & Iseli J (2007). Cost effectiveness of policy options for boilers - Rangiora, Environment Canterbury Report.
Timaru	2007	Wilton E, Baynes M, Anderson B, & Iseli J (2007). Cost effectiveness of policy options for boilers - Timaru, Environment Canterbury Report.
Washdyke	2007	Wilton E, Baynes M, Anderson B, & Iseli J (2007). Cost effectiveness of policy options for boilers - Washdyke, Environment Canterbury Report.

A5.4 PM_{10} Monitoring Data

Region	Years (if not 2006-08)	Location	Method
Northland		Whangarei	BAM
Northland	2006	Kaitaia	HiVol
Auckland		various monitoring sites across Auckland	BAM plus other
Waikato		Huntly	BAM
Waikato	2008 only	Ngaruawahia	BAM
Waikato		Matamata	BAM

Region	Years (if not 2006-08)	Location	Method
Waikato		Hamilton	TEOM
Waikato		Tokoroa	BAM
Waikato		Putaruru	BAM
Waikato		Te Kuiti	BAM
Waikato	2009 only	Turangi	BAM
Waikato		Taupo	BAM
Waikato	2008 only	Waihi	Partisol
Bay of Plenty		Rotorua	TEOM FDMS
Bay of Plenty	2009	Ngongotaha	Partisol
Bay of Plenty		Pongakawa	Partisol
Bay of Plenty	2007-2008	Kawerau	Partisol
Bay of Plenty		Tauranga	TEOM FDMS
Bay of Plenty		Whakatane	TEOM FDMS
Gisborne	2007+2009-2010	Gisborne	HiVol
Taranaki	2010	New Plymouth	BAM
Hawkes Bay		Napier	BAM
Hawkes Bay		Hastings	BAM
Horizons		Palmerston North	BAM
Horizons		Taihape	BAM
Horizons	2009-2010	Taumarunui	BAM
Wellington		Upper Hutt	BAM
Wellington		Masterton	BAM
Wellington		Wainuiomata	BAM
Wellington	2010 (winter) only	Carterton	BAM
Wellington		Wellington Central	BAM
Wellington		Lower Hutt	TEOM
Wellington		Karori	BAM
Wellington		Porirua	BAM
Tasman		Richmond	BAM
Nelson		Nelson A	BAM
Nelson		Nelson B1 (Tahunanui)	BAM
Nelson	2010 only	Nelson B2	Partisol
Nelson	2008-2009	Nelson C	Partisol
Marlborough		Blenheim Redwoodtown	BAM
Marlborough		Blenheim MMR	HiVol
Marlborough		Picton	HiVol
Westcoast		Reefton	BAM

Region	Years (if not 2006-08)	Location	Method
Westcoast	2004 (winter) only	Westport	HiVol
Canterbury		Rangiora	TEOM FDMS
Canterbury		Kaiapoi	TEOM FDMS
Canterbury		Christchurch	TEOM FDMS
Canterbury		Ashburton	TEOM FDMS
Canterbury		Timaru	TEOM FDMS
Canterbury	2008-2009	Washdyke	TEOM FDMS
Canterbury	2007-2008	Geraldine	TEOM FDMS
Canterbury		Waimate	TEOM FDMS
Canterbury	2002	Kaikoura	TEOM
Otago	2008-2009	Oamaru	BAM
Otago	2008-2009	Clyde	BAM
Otago	2009 only	Balclutha	BAM
Otago		Alexandra	BAM
Otago	2008-2009	Cromwell	BAM
Otago		Arrowtown	BAM
Otago		Dunedin	BAM
Otago		Mosgiel	BAM
Otago	2008-2009	Milton	BAM
Otago	2007 only	Nasby	HiVol
Otago	2007-2008	Ranfurly	HiVol
Southland		Gore	BAM
Southland		Invercargill Miller Street	HiVol/BAM
Southland	2008-2009	Invercargill Pomona	HiVol/BAM
Southland	2006	Invercargill Glengarry	HiVol
Southland	2006	Invercargill North Road	HiVol
Southland	2007-2008	Winton	HiVol
Southland	2006 only	Edendale	HiVol
Southland	2010 only	Wallacetown	HiVol
Southland	2010 only	Te Anau	BAM
Southland	2007 only	Bluff	HiVol
Southland	2005-2006	Mataura	HiVol

A5.5 $PM_{2.5}$ Monitoring Data

Region	Years	Location / Comments	Method
Auckland	2007-2009	Patumahoe, Whangaparaoa, Takapuna, Penrose Gavin Street, Pukekohe, Kowhai	BAM & Partisol
Canterbury	2001-2005	Christchurch	TEOM
Hawkes Bay	2006/07	Hastings - data problematic	ANSTO sampler
Nelson	2008-2010	Airshed A - gravimetric survey sampling - 120 filters in total	Partisol

Note: There are a number of sites where $PM_{2.5}$ has been measured as part of **source apportionment studies** but the monitoring method is not reference or equivalent and the data are pretty sketchy

A5.6 NO_2 Hourly Average Monitoring Data

Region	Years	Location
Auckland	2006-2008	multiple sites
Canterbury	2006-2008	Christchurch - multiple sites
Canterbury	2005	Timaru
Nelson	2010	Airshed A, Airshed B1
Taranaki	2004-2008	Vector site
Bay of Plenty	2006-2007	Otemoetai
Wellington	2006-2010	Upper Hutt
Wellington	2006-2010	Masterton
Wellington	2006-2010	Wainuiomata
Wellington	2006-2010	Wellington Central - Corner V
Wellington	2006-2010	Lower Hutt
Wellington	2006-2010	Karori

Note: NZTA have an extensive **national network of NO_2 passive samplers** that have been in operation since 2007 however it is difficult to take the monthly results and convert them to hourly averages.

Appendix 6: Detailed Data by CAU

For detailed data by CAU, please see the *Exposure Model* spreadsheet.