REPORT

METHODOLOGY FOR VALUING THE HEALTH IMPACTS OF CHANGES IN PARTICLE EMISSIONS – FINAL REPORT

NSW Environment Protection Authority (EPA)

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A PEL Company

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EXECUTIVE SUMMARY

Background

Air pollution is associated with detrimental effects on human health, natural ecosystems and climate. When evaluating the potential benefits of various pollution-reduction policies and measures it is desirable to quantify impacts in a simple and consistent manner, and economic appraisal is a common approach. An important factor in any economic appraisal of air pollution is the cost of health impacts. The health costs of air pollution are dominated by its effects on mortality. These in turn are dominated by the effects of airborne particulate matter (PM), and especially particles with a diameter of less than 2.5 μm ($PM_{2.5}$).

Cost-benefit analyses and other studies in Australia during the last decade have produced a range of values for the health impacts of PM, and have been limited in a number of ways. This Report reviews the valuation approaches taken overseas and in Australia, as well as the PM monitoring data, emissions inventory data and dispersion model results that are available in Australia to support valuation studies. Based on what is reasonably practical in Australia, a new and flexible methodology has been developed to enable the costs to society associated with changes in PM emissions to be quantified. The Report also includes a review of secondary particles, with a view to establishing how these can be better incorporated into any future valuation method.

Valuation methods

The most thorough and detailed method for valuing changes in air pollution is commonly referred to as the 'impact pathway' approach. This involves a calculation following the pathway from emissions to cost via ambient concentrations, exposure and health impacts. It is mostly used for setting standards, where data are available on current and projected (or desired) pollutant concentrations.

Applying the impact pathway approach to every policy impact assessment is very resource-intensive. As a result, many countries have adopted simple tables or models to allow direct valuation based on emissions alone. These are frequently referred to as 'damage costs', stated as a cost per tonne of emissions. Damage costs can therefore be used to evaluate policies and measures that are designed to reduce emissions. This has been the usual means of estimating the benefits of actions to improve air quality in Australia. Damage costs for a specific country or jurisdiction are usually generated via a full impact pathway approach, utilising location-specific inputs and data, but this has not been the case in Australia (for which damage costs from overseas studies have been used).

Review of overseas studies

The PM valuation approaches taken by overseas jurisdictions were reviewed. The review mainly covered the European Union (EU), the United Kingdom (UK) and the United States (US), and the methods used were described and compared. The review revealed a large number of similarities, with some overall consensus on the key issues, and a harmonisation on the main mortality risk function in the US and Europe. Indeed, most of the current methods are now dominated by one single health endpoint, mortality from chronic exposure to $PM_{2.5}$. However, there were large variations in the damage costs per tonne of pollutant.

Review of Australian studies

During the last decade the approaches used for monetising the health impacts of PM in Australia have generally involved the transfer of damage cost values from overseas studies, in some cases with an adjustment for Australian conditions. The range of unit cost values in the literature is quite wide, reflecting not only advances in the understanding of health impacts during the period but also differences in the underlying methods and assumptions. However, the later studies are broadly consistent in some respects. For example, typical average values for State capital cities are around A$250,000-A$300,000 per tonne of PM_{10} at 2010 prices. A particular challenge has been the valuation of impacts in rural areas with low population density. The unit cost values used in previous studies are also rather coarse in terms of spatial resolution, and there is little temporal resolution. Such deficiencies further highlighted the need for a new PM valuation method for Australia.

Review of Australian needs and conditions

The Report considers the availability of data and information to support the use of valuation methodologies in Australia. The work focussed on the emission inventories, modelling capabilities and monitoring activities in each Australian jurisdiction. It was concluded that Australia and NSW currently lack sufficient and readily available PM emission modelling information to undertake a full impact pathway process (and to generate a set of location-specific damage costs). To evaluate the full impact pathway approach in Australia in the future, the following information will be required:

- A detailed emissions inventory for primary particles and precursors of secondary pollutants (NO_x, VOCs, NH_3, SO_2, SO_3, elemental/organic carbon).
- A regional modelling platform capable of predicting the dispersion of primary pollutants and capable of predicting secondary particulate formation (chemical transport model).
- Detailed and reliable information on current air quality in relation to PM concentrations and composition.
- Detailed population statistics in order to assess exposure of 'stock at risk'.

These data collection and modelling exercises are likely to be expensive and time consuming.

Review of secondary particles

Another objective of the work was to review the international literature on secondary PM, to summarise the current knowledge, and to understand whether it would be possible to make inferences about Australia from the data in other countries.

The literature shows that secondary PM can be responsible for a large fraction of $PM_{2.5}$ and, to a lesser extent, PM_{10}. The secondary component is likely to represent some 25-75% of the total $PM_{2.5}$ exposure burden. On the whole, secondary PM is distributed more evenly than primary PM on a regional scale, with fewer (but still substantial) differences between urban and rural areas.

The modelling of secondary particles is an area of international development, with no clear consensus on methods. The understanding of secondary inorganic PM (nitrate and sulfate) formation is reasonably good, but the estimation of secondary organic aerosol (SOA) is highly uncertain, with the science being in an early stage of development. Data on secondary PM at Australian monitoring sites are also rather limited.

Given the uncertainties, the transfer of overseas secondary PM damage costs to Australia is not recommended. To conduct regional air quality modelling of the emission inventories including secondary PM will cost up to A$250,000 per jurisdiction. However, the validity of the modelling will be highly uncertain until initial studies have been completed and assessed against monitoring results.

Development of new valuation approach and unit damage costs

Taking into account the findings of the aforementioned reviews, this study has resulted in a new method for valuing the health impacts of PM in Australia, and has resolved some of the uncertainty arising from the use of different damage costs in recent projects.

It was concluded that the best approach would be to transfer damage cost values from the UK Department of Environment, Food and Rural Affairs (Defra). The UK values were selected primarily because of their greater sensitivity (with damage costs being available for areas with different population density). Rather than just taking geographically aggregated UK values in pounds sterling and converting to them to Australian dollars (2011 prices), a more sophisticated approach was used. Firstly, the UK damage costs were adjusted to take into account the difference between the Value of a Life Year (VOLY) in the UK and Australia, as well as differences in currency and inflation. A linear regression function was then fitted to the adjusted damage cost and population density data. This permitted a greater spatial discrimination of damage costs.

Unit damage costs were then developed for specific geographical areas of Australia using a simplified and standardised method which will allow users to relate the location of emissions to an approximate population-weighted exposure. The approach used is based on the ABS Significant Urban Area (SUA) structure for urban centres with more than 10,000 people. For each SUA in Australia the population density was used in conjunction with the regression function to determine a unit damage cost.

The following tables list the SUAs in each of the Australian jurisdictions and the associated unit damage costs (A$ per tonne of $PM_{2.5}$ emitted at 2011 prices). It is recommended that these unit damage costs are used for economic appraisals in NSW and Australia where there is no possibility of following the full impact pathway approach.

Guidance

Guidance on the calculation of damage costs in economic appraisals is provided in the Report. This includes advice on the adjustment of unit damage costs for future years (including 'uplift' to reflect future growth in GDP and a 'discount' to give net present values. The calculation of net economic impacts over several years is also explained.

Unit damage costs by SAU (rounded to two significant figures) - NSW

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$)
1030	Sydney	4,064	4,028,525	991	**$280,000**
1009	Central Coast	566	304,755	538	**$150,000**
1035	Wollongong	572	268,944	470	**$130,000**
1027	Port Macquarie	96	41,722	433	**$120,000**
1013	Forster - Tuncurry	50	19,501	394	**$110,000**
1023	Newcastle - Maitland	1,019	398,770	391	**$110,000**
1014	Goulburn	65	21,485	332	**$93,000**
1003	Ballina	73	23,511	320	**$90,000**
1018	Lismore	89	28,285	319	**$89,000**
1016	Griffith	56	17,900	317	**$89,000**
1033	Ulladulla	47	14,148	303	**$85,000**
1010	Cessnock	69	20,262	294	**$82,000**
1034	Wagga Wagga	192	52,043	272	**$76,000**
1025	Orange	145	36,467	252	**$71,000**
1022	Nelson Bay - Corlette	116	25,072	217	**$61,000**
1012	Dubbo	183	33,997	186	**$52,000**
1017	Kurri Kurri - Weston	91	16,198	179	**$50,000**
1015	Grafton	106	18,360	173	**$48,000**
1004	Batemans Bay	94	15,732	167	**$47,000**
1024	Nowra - Bomaderry	202	33,340	165	**$46,000**
1029	St Georges Basin - Sanctuary Point	77	12,610	164	**$46,000**
1031	Tamworth	241	38,736	161	**$45,000**
1005	Bathurst	213	32,480	152	**$43,000**
1032	Taree	187	25,421	136	**$38,000**
1001	Albury - Wodonga	628	82,083	131	**$37,000**
1011	Coffs Harbour	506	64,242	127	**$36,000**
1028	Singleton	127	16,133	127	**$36,000**
1007	Broken Hill	170	18,519	109	**$30,000**
1019	Lithgow	120	12,251	102	**$29,000**
1006	Bowral - Mittagong	422	34,861	83	**$23,000**
1002	Armidale	275	22,469	82	**$23,000**
1020	Morisset - Cooranbong	341	21,775	64	**$18,000**
1026	Parkes	235	10,939	47	**$13,000**
1021	Muswellbrook	262	11,791	45	**$13,000**
1008	Camden Haven	525	15,739	30	**$8,400**
1000	Not in any Significant Urban Area (NSW)	788,116	999,873	1.3	**$360**

Unit damage costs by SAU (rounded to two significant figures) - Victoria

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$)
2011	Melbourne	5,679	3,847,567	677	**$190,000**
2016	Sale	46	14,259	313	**$88,000**
2020	Wangaratta	58	17,687	307	**$86,000**
2004	Bendigo	287	86,078	299	**$84,000**
2003	Ballarat	344	91,800	267	**$75,000**
2005	Colac	55	11,776	215	**$60,000**
2010	Horsham	83	15,894	191	**$54,000**
2008	Geelong	919	173,450	189	**$53,000**
2017	Shepparton - Mooroopna	249	46,503	187	**$52,000**
2006	Drysdale - Clifton Springs	65	11,699	180	**$50,000**
2012	Melton	266	47,670	179	**$50,000**
20+22	Warrnambool	183	32,381	177	**$50,000**
2019	Traralgon - Morwell	235	39,706	169	**$47,000**
2014	Moe - Newborough	105	16,675	158	**$44,000**
2018	Torquay	126	15,043	119	**$33,000**
2015	Ocean Grove - Point Lonsdale	219	22,424	103	**$29,000**
2001	Bacchus Marsh	196	17,156	87	**$24,000**
2002	Bairnsdale	155	13,239	85	**$24,000**
2013	Mildura - Wentworth	589	47,538	81	**$23,000**
2007	Echuca - Moama	351	19,308	55	**$15,000**
2009	Gisborne - Macedon	367	18,014	49	**$14,000**
2021	Warragul - Drouin	680	29,946	44	**$12,000**
2000	Not in any Significant Urban Area (Vic.)	216,296	693,578	3	**$900**

Unit damage costs by SAU (rounded to two significant figures) - Queensland

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$)
3003	Cairns	254	133,912	527	**$150,000**
3008	Hervey Bay	93	48,678	523	**$150,000**
3006	Gold Coast - Tweed Heads	1,403	557,823	398	**$110,000**
3001	Brisbane	5,065	1,977,316	390	**$110,000**
3010	Mackay	208	77,293	371	**$100,000**
3004	Emerald	39	13,219	337	**$94,000**
3012	Mount Isa	63	20,569	328	**$92,000**
3007	Gympie	69	19,511	282	**$79,000**
3016	Townsville	696	162,291	233	**$65,000**
3002	Bundaberg	306	67,341	220	**$62,000**
3015	Toowoomba	498	105,984	213	**$60,000**
3018	Yeppoon	79	16,372	208	**$58,000**
3005	Gladstone - Tannum Sands	240	41,966	175	**$49,000**
3014	Sunshine Coast	1,633	270,771	166	**$46,000**
3011	Maryborough	171	26,215	154	**$43,000**
3013	Rockhampton	580	73,680	127	**$36,000**
3017	Warwick	159	14,609	92	**$26,000**
3009	Highfields	230	16,820	73	**$20,000**
3000	Not in any Significant Urban Area (Qld)	1,718,546	755,687	0.4	**$120**

Unit damage costs by SAU (rounded to two significant figures) – South Australia

SUA code	SUA name	Area (km^2)	Population	Population density (people/km^2)	Damage cost/tonne of $PM_{2.5}$ (A$)
4001	Adelaide	2,024	1,198,467	592	**$170,000**
4006	Port Pirie	75	14,044	187	**$52,000**
4008	Whyalla	121	21,991	181	**$51,000**
4003	Murray Bridge	98	16,706	171	**$48,000**
4002	Mount Gambier	193	27,754	144	**$40,000**
4005	Port Lincoln	136	15,222	112	**$31,000**
4007	Victor Harbor - Goolwa	309	23,851	77	**$22,000**
4004	Port Augusta	249	13,657	55	**$15,000**
4000	Not in any Significant Urban Area (SA)	980,973	264,882	0.3	**$76**

Unit damage costs by SAU (rounded to two significant figures) – Western Australia

SUA code	SUA name	Area (km^2)	Population	Population density (people/km^2)	Damage cost/tonne of $PM_{2.5}$ (A$)
5009	Perth	3,367	1,670,952	496	**$140,000**
5007	Kalgoorlie - Boulder	75	30,839	411	**$110,000**
5003	Bunbury	223	65,608	295	**$83,000**
5005	Ellenbrook	105	28,802	276	**$77,000**
5002	Broome	50	12,765	255	**$71,000**
5006	Geraldton	271	35,749	132	**$37,000**
5008	Karratha	134	16,474	123	**$34,000**
5010	Port Hedland	116	13,770	118	**$33,000**
5001	Albany	297	30,656	103	**$29,000**
5004	Busselton	1,423	30,286	21	**$6,000**
5000	Not in any Significant Urban Area (WA)	2,520,513	30,654	0.01	**$3**

Unit damage costs by SAU (rounded to two significant figures) - Other

State	SUA code	SUA name	Area (km^2)	Population	Population density (people/km^2)	Damage cost/tonne of $PM_{2.5}$ (A$)
Tasmania	6001	Burnie – Wynyard	131	29,050	223	**$62,000**
	6004	Launceston	435	82,222	189	**$53,000**
	6003	Hobart	1,213	200,498	165	**$46,000**
	6005	Ulverstone	130	14,110	108	**$30,000**
	6002	Devonport	290	26,871	93	**$26,000**
	6000	Not in any Significant Urban Area (Tas.)	65,819	142,598	2	**$610**
Northern territory	7002	Darwin	295	106,257	361	**$100,000**
	7001	Alice Springs	328	25,187	77	**$22,000**
	7000	Not in any Significant Urban Area (NT)	1,347,577	80,504	0.06	**$17**
ACT	8001	Canberra – Queanbeyan	482	391,643	812	**$230,000**
	8000	Not in any Significant Urban Area (ACT)	1,914	1,622	0.85	**$240**
Other	9000	Not in any Significant Urban Area (OT)	218	3,029	14	**$3,900**

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1 INTRODUCTION

1.1 Background and objectives

Air pollution is associated with detrimental effects on human health, natural ecosystems and climate. When evaluating the potential benefits of various pollution-reduction policies and measures it is desirable to quantify impacts in a consistent manner. Whilst this is difficult given the diversity of the impacts, approaches based on monetary valuation are the most common, and these have several advantages. They make explicit the real cost of pollution impacts on society, and enable alternative proposals to be compared directly using a single index (money).

A framework for the valuation of costs and benefits of policies, including the economic assessment of environmental impacts, has been established in Guidelines published by the **NSW Treasury (2007)**. The Guidelines aim to ensure that all public sector agencies undertake economic appraisals on a consistent basis. Economic appraisal is also an important prerequisite of any new statutory instrument.

An important factor in any economic appraisal of air pollution is the cost of health impacts. The health costs of air pollution are dominated by its effects on mortality, which in turn are dominated by the effects of airborne particulate matter (PM).

Ambient concentrations of PM are most commonly defined in terms of two metrics: PM_{10} and $PM_{2.5}$, the mass concentrations of particles with an aerodynamic diameter of less than 10 µm and 2.5 µm respectively. Airborne PM is derived from a wide range of natural and anthropogenic sources. When discussing PM sources and composition it is essential to distinguish between 'primary' and 'secondary' particles. Primary particles are emitted directly into the atmosphere as a result of natural processes (*e.g.* wind erosion, marine aerosols) and anthropogenic processes involving either combustion (*e.g*. industrial activity, domestic wood heaters, vehicle exhaust) or abrasion (*e.g.* road vehicle tyre wear). Secondary particles are not emitted directly, but are formed by reactions involving gas-phase components of the atmosphere. Various studies have shown that secondary particles contribute significantly to PM concentrations, especially $PM_{2.5}$ at background sites, although their characteristics vary significantly with both location and time.

The current approach to air quality management in Australia focuses on reducing exceedances of ambient air quality standards at specific locations[1]. The standards are designed to protect health. However, for PM_{10} and $PM_{2.5}$ there is no evidence of threshold concentrations below which adverse health effects are not observed (**WHO, 2006; COMEAP, 2009; USEPA, 2011a**). Therefore, whilst PM_{10} concentrations in Australian cities are significantly below the standards for most of the time[2] **(Commonwealth of Australia, 2010)**, the health costs are

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[1] The National Environment Protection (Ambient Air Quality) Measure (AAQ NEPM) sets national air quality standards for six air pollutants (CO, NO_2, SO_2, lead, O_3, PM_{10}). The NEPM was extended in 2003 to include advisory reporting standards for $PM_{2.5}$. Monitoring is required to determine whether the standards have been met within populated areas. The NEPM monitoring protocol states that some monitoring stations should be located in populated areas which are expected to experience relatively high concentrations, providing a basis for reliable statements about compliance within the region as a whole. These stations are called generally representative upper bound (GRUB) for community exposure sites. However, it is also necessary to ensure that a NEPM monitoring network provides a widespread coverage of the populated area in a region and provides data indicative of the air quality experienced by most of the population. Monitoring plans must demonstrate an adequate balance of GRUB and population-average measurements.

[2] High particle concentrations are usually a result of bushfires and dust storms.

actually driven by large-scale exposure to relatively low pollution levels[3]. The NSW Environment Protection Agency (EPA) has therefore promoted a 'net economic benefit' approach to air pollution control which supports continuous reductions in PM emissions and improvements in air quality as long as a net benefit can be demonstrated, taking into account costs and benefits to government, industry and the community.

Previous cost-benefit analysis (CBA) projects and other studies in Australia have used a range of cost values for the health impacts of PM. NSW EPA therefore commissioned PAEHolmes to develop a more robust general valuation methodology to replace the previous *ad hoc* assessments.

The principal objective of the project was to develop a methodology for valuing the health impacts of PM in Australia. The methodology needed to be applicable to pollution-reduction policies and measures such as possible national emission standards for non-road diesel engines. It also had to be simple, robust and capable of being updated to reflect changes in the health evidence.

This Final Report of the project describes the development of the methodology. The scope of the work is summarised below, and a glossary of terms and abbreviations used in the Report is provided in **Appendix A**.

1.2 Scope of work

To address the objectives of the study, the following work was undertaken:

- A **review stage**, which involved the following:
  - A summary of the approaches taken by overseas jurisdictions - including the European Union (EU), the United States (US), Canada and New Zealand - and Australian jurisdictions to valuing the health impacts of PM emissions and concentrations. This work is described in **Chapter 2**.
  - An analysis of Australian needs and conditions, and the availability of data and information to support the use of potential methodologies. This part of the study is described in **Chapter 3**.
  - A review of the literature on secondary particles, including Australian studies, in order to establish feasibility for inclusion in the proposed methodology. This review is provided in **Chapter 4**.
- The **Development of the methodology** for estimating the health costs associated with changes in PM emissions in NSW and Australia, supported by reference to the findings of the review stage. The proposed methodology is provided in **Chapter 5**.

[3] The development of an exposure-reduction framework for PM was an important recommendation of a review of the National Environment Protection Measure for Ambient Air Quality ('Air NEPM') (**NEPC, 2011**), and the NSW government is currently in the process of developing such a framework.

2 REVIEW OF METHODOLOGIES FOR VALUING THE HEALTH EFFECTS OF PM EMISSIONS

2.1 Overview

In recent years various methods have emerged for quantifying and valuing the health effects (and other environmental effects) of air pollutants, including PM. Many of these methods have adopted a very similar approach, and have even relied upon the same underlying health studies. This Chapter firstly provides brief descriptions of the two main approaches to valuing changes in air pollution: the 'impact pathway' approach and the 'damage cost' approach. Several important studies from the literature are then summarised.

2.2 Impact pathway approach

2.2.1 Summary

In broad terms, the approach taken for the detailed valuation of the health impacts of air pollution is often referred to as the impact pathway approach. This involves a 'bottom-up' calculation in which environmental benefits and costs are estimated by following the steps shown in **Figure 2-1**. This approach was developed through a series of joint EU-US research projects in the 1990s.

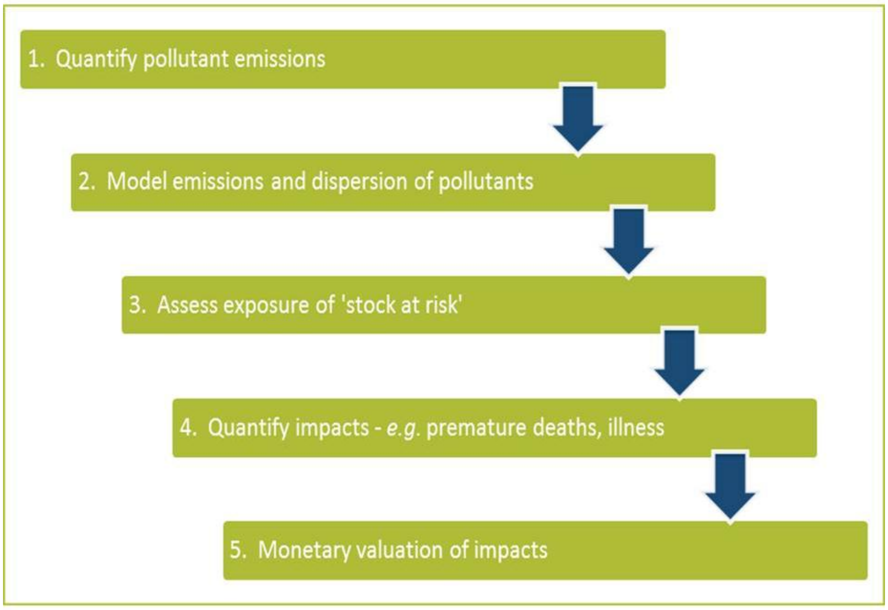

Figure 2-1 Impact Pathway Approach

Is some circumstances a variant of the impact pathway approach may be applied. For example, when setting air quality standards, Steps 1 and 2 may be disregarded and changes in exposure to pollutant concentrations between current and future scenarios (the latter being based on the proposed standard) are used to quantify the impacts on health.

Regardless of the complexity of the approach taken, the overall impacts are calculated using the following general relationships:

$$Impact = Concentration \times Stock\ at\ risk \times Response\ function \qquad \text{(Equation 1)}$$

$$Cost = Impact \times Unit\ cost\ of\ impact \qquad \text{(Equation 2)}$$

The main steps in the calculation are discussed in more detail below.

2.2.2 Quantifying emissions

The first step in the calculation involves the quantification of emissions, with disaggregated road-based or grid-based source apportionment. This requires a detailed emissions inventory.

2.2.3 Modelling air pollution

The second step involves an analysis of pollutant dispersion and chemistry across different spatial scales. Importantly, this includes the consideration of both primary pollutants (*e.g.* SO_2, primary PM) and secondary pollutants (secondary PM such as sulfates, or secondary gaseous pollutants such as ozone), and the assessment of changes in pollutant concentrations. A large amount of information is required on baseline emissions and pollution concentrations, as these determine the formation of secondary pollutants.

2.2.4 Determining exposure

The third step is the quantification of the exposure of people, the environment (*e.g.* crops) and buildings that are affected by air pollution (*i.e.* linking pollution with the 'stock at risk' using, for example, population data).

2.2.5 Estimating impacts

The fourth step involves the quantification of the impacts (health and non-health) of air pollution. The adverse health effects of ambient air pollution are divided into two categories: morbidity and mortality. Morbidity effects may range from the relatively mild sub-clinical effects such as increased coughing, reduction in lung function or increased medication usage, through to seeking medical attention by a general practitioner, emergency department attendances and hospital admission. Mortality is the most widely recommended health impact for use in studies quantifying the effects of air pollution **(COMEAP, 2009)**.

The assessment of mortality from chronic exposure is a critical issue because valuation approaches that look at long-term changes in air pollution associated with air quality standards may need a different approach to those approaches that estimate at short-term changes associated with specific policies. It is acceptable to assume that an air pollution standard will lead to a long-lasting change in air pollution, and thus lead to lifetime reduced exposure for the population. However, this requires analysis of costs and benefits over the longer term. In contrast, some air quality measures or policies are more transient. As an example, a low-emission zone which bans older vehicles from urban centres merely accelerates the introduction of cleaner vehicles (meeting more recent vehicle standards). It produces an air quality benefit for a few years but only minimally reduces lifetime exposure. This can be addressed with the use of an 'annual pulse' analysis of chronic effects.

PM is known to be the most damaging pollutant to human health in terms of overall health costs, particularly in the longer term. Many studies have used PM_{10} as an indicator of PM. However, there is increasing evidence that the adverse health effects - particularly mortality - are more closely associated with $PM_{2.5}$ **(Pope and Dockery, 2006)**. A recent UK report states that $PM_{2.5}$ is considered to be the best index of PM for quantitative assessments of the effects of policy interventions **(COMEAP, 2009)**.

There are two methods of calculating the proportion of deaths attributable to a change in PM exposure. The first method uses a 'static' concentration-response (C-R) function derived from epidemiological studies, in which:

$$\textit{Attributable proportion} = \textit{Annual death rate} \times \textit{Study population size} \times \textit{\% increase in health effect per increase in exposure} \times \textit{Change in exposure} \qquad \text{(Equation 3)}$$

The second method is based on 'life tables'. This approach follows a stratified (by age) study population over time. It takes into account the probability of each age band dying, and compares a baseline scenario with a scenario in which the exposure changes **(Hurley *et al*., 2005)**. The life-table method is based on a matrix defined simultaneously by the calendar years into the future and the age distribution of the study population. The effect of a specific exposure on health is given by the differences between the two matrices (between the exposure-changed scenario and the baseline). This estimation method expresses health impacts in terms of 'years of life lost' (YOLL) from air pollution.

2.2.6 Monetary valuation of impacts

In the final step a monetary value is assigned to the impacts. Health impacts from changes in PM emissions are often monetised using unit costs for the value of a statistical life (VSL), value of a statistical life year (VOLY), hospitalisation for respiratory disease and hospitalisation for cardiovascular disease. The single most important health endpoint in the valuation of PM health impacts is mortality, and specifically mortality from chronic exposure. This has dominated valuations in all studies to date. However, mortality from chronic exposure is also the most complex health endpoint to assess.

The monetary valuation of the VSL is often derived using the 'willingness to pay' (WTP) approach. In short, the WTP approach surveys individuals about their willingness to pay to avoid a specific health effect. The VSL is defined as an aggregate measure of a community's WTP to reduce the risk of premature mortality. Once the number of deaths saved or lost due to changes in PM emissions is established (static method of attributable deaths), the VSL is applied to the number, producing the cost or benefit of the change.

The other approach in the monetary valuation of premature mortality is the VOLY. The VOLY is usually calculated via an annualised equivalent of VSL estimates. The VOLY can then be applied to the YOLL to derive a cost due to changes in PM emissions. In their report for the Australian Commonwealth Department of Environment, Water, Heritage and the Arts (DEWHA), **Jalaludin *et al.* (2009)** recommend that the use of the VOLY is preferable to the use of the VSL in monetising the air pollution effects on premature mortality, and should be used whenever feasible and practicable.

The cost of hospital admissions and other morbidity outcomes are usually based on the average use of hospital or medicinal resources for a patient group.

The terminology in relation to the valuation of health impacts is summarised in the text box below.

Summary of health impact valuation terminology

The effect of chronic exposure to PM on mortality is expressed in two ways in health valuations:

- The loss of life expectancy is expressed as the total number of life years lost annually across the affected population.
- The number of deaths brought forward, is expressed as the number of cases (deaths) per year.

The loss of life expectancy is the preferred measure of impact on theoretical and practical grounds, although deaths brought forward are included for valuation purposes. The two estimates are not additive. However, they allow alternative valuation approaches to be adopted.

Some of the terms used in health valuations are described below.

Life table	A table which shows, for each age, what the probability is that a person of that age will die before his or her next birthday.
VOLY	Value of Life Year: an estimate of the value society places on reducing the risk of premature death, expressed in terms of saving a statistical life year.
VSL	Value of a Statistical Life: an estimate of the economic value society places on reducing the average number of deaths by one.
YOLL	Years of Life Lost: an estimate of the average years a person would have lived if he or she had not died prematurely (in this case due to exposure to air pollution).

2.3 Damage cost approach

Applying the impact pathway approach to every policy impact assessment is very resource intensive, and most likely prohibitively so. As a result, many countries have adopted simple 'look-up' tables to allow direct valuation based on emissions alone. These are frequently referred to as 'damage costs', and allocate dollar-per-tonne values to emissions.

Damage costs for a specific country or jurisdiction are usually generated via a full impact pathway approach utilising location-specific inputs and data. The level of detail used to generate damage costs varies. Some approaches involve the quantification of health impacts as well as monetary values, whereas others use disaggregated values that differentiate emissions according to the sector or location of emissions.

Damage costs provide a simple way to value changes in PM. They are estimates of the costs to society due to the impacts of changes in emissions. Damage costs assume an average impact on an average population affected by changes in air quality.

2.4 Review of studies in the literature

2.4.1 Overseas studies

This Section compares the main features of various overseas air pollution valuation studies. More detailed descriptions of the most relevant studies can be found in **Appendix B**.

Different countries have adopted different approaches for valuing the health impacts of PM. The most advanced and detailed studies have been those undertaken in Europe and the US, where independent scientific committees have provided advice on health quantification and valuation. These studies have examined major changes in air pollution standards, capturing the complexity associated with chronic health effects using the impact pathway approach. For other policy applications (including revisions to air pollution standards), simpler damage costs have been used. As an example, in the Clean Air for Europe (CAFE) programme damage costs have been applied to a range of sectoral and policy-specific contexts, whilst the United States Environmental Protection Agency (USEPA) has used damage costs (for secondary PM) when *updating* air quality standards for NO_2 and SO_2.

The main studies identified in the literature and considered in detail were:

- *European Union - CAFE programme.* The objectives of the CAFE programme were to establish the capacity to assess the costs and benefits of air pollution policies, and to conduct a CBA of the effects of these policies. The impact pathway approach was used to value the health impacts of air pollution (environmental endpoints such as crop damage were also assessed), although damage costs were also generated **(AEA Technology Environment, 2005)**.

- *United Kingdom – Review of Air Quality Strategy.* The UK has a long tradition of CBA for air pollution. The analysis of impacts and external costs has been taken forward by the Department of Health's Committee on the Medical Effects of Air Pollutants (COMEAP) and the Interdepartmental Group on Costs and Benefits (IGCB). IGCB undertook an economic analysis of the UK Air Quality Strategy using an impact pathway approach. IGCB also generated damage costs by sector, with further disaggregation for transport-related emissions according to population density **(Defra, 2007)**.

- *United States – National Air Quality Standards.* The US has long adopted CBA for air quality regulations and impact assessment. The USEPA has significantly developed the cost-benefit method for air pollution as part of the Benefits and Costs of the Clean Air Act **(Fann *et al.*, 2009)**. The general benefits analysis framework used an impact pathway approach, using detailed air quality models. The USEPA did not publish PM damage costs.

Studies undertaken in Canada and New Zealand were also examined. The analysis in Canada **(RWDI, 2005)** follows the USEPA literature. The analysis in New Zealand **(New Zealand Ministry for Environment, 2004)** predates most of the recent literature and the complexity of long-term PM exposure. Consequently, these studies were not considered further.

The principal characteristics of the approaches used in the EU, UK and US studies are summarised in **Table 2-1**. All three studies used complex modelling of emissions and ground-level concentrations, as well as various mortality and morbidity end points. Different methods for valuing end points were used. The single most important health endpoint in these studies is mortality from chronic exposure.

Table 2-1: Summary of International approaches

Aspect	CAFE	UK Air Quality Strategy Review	USEPA
General approach	Impact pathway and damage cost	Impact pathway and damage cost	Impact pathway (damage costs for SO_2 and NO_x)
Pollutants considered	Primary and secondary	Primary and secondary	Primary and secondary
Emission inventory	Various	NAEI – 11 sectors including point source, agriculture and transport	USEP NEI - point, non-point, on-road, non-road, and event
Approach for air quality	Detailed models (RAINS)	Detailed national models (plus EMEP)	Detailed air quality models (CMAQ)
Population assumptions and inputs	Detailed population and life tables	Detailed population and life tables	Detailed population and life tables
Mortality - chronic analysis of PM	$PM_{2.5}$, 6% hazard rate, all equally casual, no lag between exposure and effect, annual pulse, using life tables	PM_{10}, 6% hazard rate, all equally casual, various lag effects, life tables (UK specific), annual pulse and sustained pollution changes	$PM_{2.5}$, 6% hazard rate, all equally casual, lag distribution
Morbidity	Infant mortality Chronic bronchitis Respiratory hospital admissions Cardiac hospital admissions Restricted activity days Respiratory medication use Lower respiratory symptom days	Respiratory and cardio-vascular hospital admissions only	Infant mortality Bronchitis: chronic and acute Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Non-fatal heart attacks (myocardial infarction) Lower and upper respiratory illness Minor restricted-activity days Work loss days Asthma exacerbations (asthmatic population) Respiratory symptoms (asthmatic population)
Application of health functions (% of baseline rates, values per population).	Various	Baseline rates	Baseline rates
Functions used for estimating health endpoints	Pope *et al.* (1995, 2002) for chronic effects	Pope *et al.* (1995, 2002) for chronic effects	Pope *et al.* (1995, 2002) for chronic effects
Valuation of health endpoints	VSL and VOLY	VOLY	VSL
Overall economic framework	Current prices, no uplift or discounting	Current prices, then uplift at 2% per year, followed by declining discount rate starting at 3.5%	Projected real income growth (split by endpoint)

Watkiss (2008) compared the EC CAFE and UK Defra approaches in more detail. He reported that the two approaches had many similarities - they used the same methodological framework (impact pathway) and focussed on the same two key pollutants: PM and O_3. Moreover, the results were dominated by mortality from chronic exposure (PM). For this endpoint the studies used the same function and hazard rate from **Pope *et al.* (2002)**, and were based on the same set of life tables. However, there were also several differences:

- Defra applied the Pope function for mortality from chronic exposure (derived for $PM_{2.5}$) to marginal PM_{10} pollution. In the CAFE approach the function was applied only to $PM_{2.5}$ pollution. The Defra method therefore led to a larger estimate of the population-weighted increment in mortality from chronic exposure (up to 1.3 times greater, depending on the policy being examined).
- In the Defra method functions were applied to a fixed population in the year 2000, whilst in CAFE a 2020 UK population was used. This led to a higher stock at risk (*i.e.* 7% higher population) in CAFE.
- Defra applied the function for mortality from chronic exposure (6% hazard rate) with (a) no lag, (b) a 40-year lag, and (c) a weighted distribution of lag and hazard rate. In the CAFE work a single central value (6%) was used, with a phased introduction (lag) over 11 years. More importantly, the Defra approach worked with a sustained pollution change over 100 years, which was then annualised for valuation. The CAFE method used a one year marginal pulse only.
- The Defra method estimated YOLL only, whilst CAFE estimated YOLL but also expressed this same health endpoint as premature deaths (to allow valuation with a VSL).
- The Defra method used a VOLY of £29,000 (€43,500 at 2010 prices). CAFE used a higher VOLY (2000 prices) of €120,000 (mean). CAFE valued the predicted number of deaths using VSL as well as valuing the overall loss of life years with VOLY. Defra applied a 2% annual uplift to health values, so VOLY estimates in later years were much higher. It also then discounted using declining discount rates to generate a net present value and an equivalent annualised value. CAFE did not apply an uplift or discount (*i.e.* it worked with a static one-year value).
- CAFE included a significantly larger number of morbidity impacts than Defra, which increased the value of the impacts (£) by 10% to 30% (on the high and low estimates respectively).
- Defra assessed UK impacts only (from UK policy). CAFE included trans-boundary effects as well. This led to higher values in CAFE when assessing a UK only policy, and also led to higher CAFE UK damage costs. For the latter, this increased damages by 20% to 60%, depending on the pollutant.

Overall, there was no systematic bias towards higher or lower values for either method across all impact pathway stages (some of the differences in approach are likely to cancel each other out).

A further comparison of the European methods with the USEPA method shows a number of similarities and differences:

- USEPA included a very similar list of health endpoints to CAFE.

- USEPA used the same **Pope et *al*. (2002)** function, but also used a much wider range of functions including expert consensus functions. This led to a much more complex set of outputs.
- USEPA used $PM_{2.5}$ (as the CAFE method) and has a distributed lag phase, similar but different in the exact distribution to the UK approach.
- USEPA used VSL estimates only, using much higher values for a VSL than CAFE (approximately US$5-6 million versus around €1 million).
- USEPA applies specific inflators and discounts at 3% and 7%.

For comparison, selected Defra and CAFE unit damage costs are shown in **Table 2-2**, where they have been converted into Australian dollars at 2010 prices. These values are for primary PM emissions.

Table 2-2 UK and EU unit damage cost values (2010 Prices, AUD)

Defra	Low Central	High Central
PM Transport Average	$56,981	$82,700
PM Transport Central London	$260,402	$377,942
CAFE	**Low VOLY**	**High VSL**
CAFE UK Average	$35,054	$130,514
CAFE EU-25 Average	$40,040	$115,500

2.4.2 Australian studies

The following paragraphs summarise the methods used in Australia to value the health impacts of changes in PM emissions. More detailed descriptions of several of the studies mentioned can be found in **Appendix C**. The approaches used in Australia have varied, but have generally involved the use of unit damage cost values from overseas studies, in some cases with small adjustments for Australian conditions. The studies have not included complex valuation of long-term exposure to PM.

Early valuations of the health impacts of air pollution were presented by **NSW EPA (1997, 1998)** and **Environment Australia (2000)**. The damage costs from these studies were summarised by **Coffey (2003)** – though it was noted that many of these will not have taken mortality from chronic exposure into account, and so cannot be directly compared with more recent estimates. This would explain in part the much lower values obtained in these earlier studies.

Beer (2002) used published Australian transport-related health costs to estimate the costs associated with the road transport contribution to ambient PM_{10}. The work by Beer is cited as being the only valuation study based on Australian data, although it uses an equation developed to represent US airshed conditions in the early 1990s.

Unit damage costs for PM emissions were derived for Australia as part of the Fuel Taxation Inquiry by **Watkiss (2002)**. The original unit damage costs were taken from the EC ExternE study. Values were determined for Australian locations by transferring values for European locations based on similarity of population density. Unit costs were determined for areas in four population density bands, ranging from inner areas of large cities to non-urban areas, hence improving the spatial resolution of previous methods.

The Commonwealth Fuel CBA was an assessment of a potential policy measure, and provided a basic estimate of the resulting ambient air quality for PM **(Coffey, 2003)**. The study also involved an assessment of the average health saving per tonne of national transport emissions. Coffey used the on-road and total emissions in each airshed from jurisdiction inventories, and assumed the resultant ambient air concentrations were linearly related to the reduction in overall particle emissions. Health cost estimates were limited to mortality and hospital admissions as there was insufficient information for prediction of less severe impacts. Coffey did not take account for the role of NO_x and SO_2 in secondary PM formation.

In a study of health costs of existing air quality in the NSW GMR, **DEC (2005)** derived PM_{10} damage costs for 'Hunter', 'Sydney' and 'Illawarra'. The damage costs were calculated using the PM emissions inventory for the GMR. Modelling of secondary particle formation was not available for the analysis.

In 2005 the Centre for International Economics **(CIE, 2005)** undertook an evaluation of Sydney's then present and future transport infrastructure. As part of the study, CIE assessed the costs of air pollution by using PM_{10} as an index pollutant. The authors used estimated PM_{10} emissions from motor vehicles in Sydney and associated costs from a study by the Bureau of Transport and Regional Economics **(BTRE, 2005)**. The year 2000 vehicle emission estimates amounted to 4,750 tonnes, which were associated with a total cost of between $613 million and $1.5 billion. Working off the central cost estimate, the authors calculated that Sydney incurred a cost of $293,185 per tonne of PM_{10} emitted (at 2010 prices).

The Commonwealth Department of Infrastructure, Transport, Regional Development and Local Government **(DIT, 2010)** reviewed health benefits as part of a Regulatory Impact Statement for consideration of the Euro 5 and Euro 6 emissions standards for light-duty vehicles. The study used damage costs from a range of sources to predict the avoided health costs, with monetary values (in $/tonne) assigned to HC, NO_x and PM. The values in **Coffey (2003)**, **Watkiss (2002)** and **Beer (2002)** were averaged to calculate the total health benefit. Unit damage cost values for capital cities were calculated by taking the average of the estimates from the three studies. Unit values for the rest of Australia were based on the average of the estimates for Band 3 and Band 4 contained in **Watkiss (2002)**.

A consultation regulation impact statement (RIS) conducted by the **Non-road Engines Working Group (2010)** examined whether there was a case for government action to reduce adverse impacts of non-road spark ignition engines and equipment on human health and the environment. Costs were calculated by averaging the four European estimates from the CAFE programme **(AEA Technology Environment, 2005)**. For $PM_{2.5}$ the authors used a value of A$82,490/tonne at 2008 prices (from the European Commission). For PM_{10} the unit damage costs from **BTRE (2005)** were used. The study assumed that a linear relationship existed between the tonnage of emissions and health impacts. The study also noted that the impacts of emissions are directly related to the population size exposed to the emissions.

An economic appraisal of measures to control wood smoke was undertaken for the NSW OEH in 2011 **(AECOM, 2011)**. AECOM used the PM_{10} damage costs for capital cities from the Euro 5/6 study, but adjusted the value for regional areas using the population density ratio between Sydney and the particular area. They arrived at an overall value for NSW of $72,114/tonne (at 2010 prices). **BDA (2006)** assessed the benefits and costs across six urban Australian airsheds of changing national standards for particle emissions and energy efficiency for wood heaters.

The unit health cost for PM_{10} used in the study for Sydney ($133,543 at 2005 prices) was based on data supplied by the Department of the Environment and Heritage.

It is worth noting that **Beer (2002)**, **Coffey (2003)** and **DEC (2005)** calculated health costs based on a simplified impact pathway approach. In short, unit damage costs were estimated by comparing the health costs for ambient PM concentrations with total emissions for the area of interest. Whilst this method has the advantage of utilising local conditions and health incidence, they assumed a linear relationship between emissions and air quality. In addition, the unit damage costs are limited in terms of application to areas with different population density. **Watkiss (2002)** transferred damage cost values from overseas based on similarities of population density, which provided some flexibility when applying the values to different areas. The study did not, however, include adjustments for Australia-specific health values.

The unit damage cost values for PM resulting from the Australian studies are presented in **Table 2-3**. The original damage costs have been converted to 2010 prices to enable comparison. Whilst there is a wide range of cost values, the later studies are broadly consistent in some respects. For example, typical average values for State capital cities are around A$250,000-A$300,000 per tonne of PM_{10} at 2010 prices. A particular challenge has been the valuation of impacts in rural areas with low population density. The unit costs are also rather coarse in terms of spatial resolution and there is little temporal resolution.

Table 2-3: Summary of PM damage cost values from Australian studies (2010 prices)

Study	Metric	Details	A$/tonne
NSW EPA (1997[(a)]	PM_{10}	N/A	3,747
NSW EPA (1998)[(a)]	PM_{10}	N/A	642
Environment Australia (2000)[(a)]	PM_{10}	N/A	23,659
Beer (2002)[(b)]	PM_{10}	From transport	184,326
Watkiss (2002)	PM_{10}	1: Inner areas of larger State capital cities	427,155
	PM_{10}	2: Outer areas of larger State capital cities	116,500
	PM_{10}	3: Other State capital cities and urban areas	116,500
	PM_{10}	4: Non-urban areas	1,550
Coffey (2003)	PM_{10}	State capital cities	282,243
CIE (2005)	PM_{10}	Sydney	293,185
DEC (2005)[(b)]	PM_{10}	Sydney	273,000
	PM_{10}	Hunter	73,000
	PM_{10}	Illawarra	54,000
BDA (2006)	PM_{10}	Sydney	154,617
Non-Road Engines Working Group (2010)	$PM_{2.5}$	N/A	86,381
DIT (2010)[(b,c)]	PM_{10}	State capital cities	241,955
	PM_{10}	Rest of Australia	57,415
AECOM (2011)[(c)]	PM_{10}	NSW	72,114

(a) Cited in Coffey (2003)
(b) Central estimate
(c) Based on a review

3 AUSTRALIAN NEEDS AND CONDITIONS

3.1 Overview

In order to assess the potential application of methodologies discussed in Chapter 2 in Australia, an understanding of relevant conditions in Australia was required. This included:

- Air quality management resources compared with overseas jurisdictions.
- Atmospheric modelling capacity.
- The availability of data on air quality and health (including health costs).
- Variability in data across jurisdictions.
- Variability across urban and regional communities (e.g. in terms of air quality impacts and the benefits of emission reductions).

3.2 Air emission inventories

Within Australia, two types of regional pollutant inventories exist: the National Pollutant Inventory (NPI) and regional air emission inventories.

The NPI is a broad-based emissions inventory which contains data on pollutant emissions to air, land and water, and pollutant transfers to designated destinations. Data are collected and published annually for industrial facilities that trigger certain reporting thresholds (such as fuel used or total pollutant handled). Emissions from diffuse sources (e.g. domestic wood heaters) are required to be reported by jurisdictions on a period agreed by each jurisdiction. Emissions data from the NPI are aggregated into total stack and total fugitive emissions from each facility point or diffuse source. Information on the temporal and spatial variation in emissions – as required for air quality modelling purposes - are only collected on an annual basis (i.e. no temporal variation information is collected) and emissions are allocated spatially to the centre of the facility (not the specific emission source location). Furthermore, there is no requirement to provide source parameters required for air quality modelling such as stack height, exit temperature, exit velocity or stack diameter.

Regional inventories are developed and maintained by some jurisdictions in order to inform air quality management decisions and policy analyses. Regional air emission inventories contain more detailed information than that stored and collected under the NPI NEPM.

The following key differences between the two inventory types are:

- Emissions are stored on a source level - i.e. emissions across a facility can be separated according to the source (e.g. coal-fired boiler, coal stockpile, front-end loader).
- Temporal variation for each source is recorded to enable air quality modelling and seasonal analysis (e.g. monthly, weekday/weekend day, hourly variation).
- Source parameters are generally recorded within the emissions inventory to enable emissions data to be used for air quality modelling purposes.
- No threshold for inclusion of sources exists in the regional emissions inventories (all practical sources of emissions are included).
- There is no defined list of pollutants for a regional air emissions inventory (jurisdictions decide which pollutants to include in order to suit the planned inventory objectives).

Based on consultation with jurisdictions and a literature search as part of this project, five jurisdictions in Australia were found to use air emission inventories to manage air quality in some way. A summary of each air emissions inventory is provided in **Table 3-1**. Diffuse emission estimates exist for the major population centres in the other three Australian jurisdictions. However, the emission estimates are out of date having been completed close to the inception of the NPI, with all urban centres having a base year of 1999. The emission estimates are published on the NPI database.

No official methodology or guidebook exists for compiling regional air emissions inventories in Australia, such as the EMEP/EEA Air Pollutant Emission Inventory Guidebook in Europe. Handbooks and emission estimation manuals are published by the Commonwealth Government for estimating emissions for the NPI. These manuals have facilitated a certain level of consistency in constructing regional emission inventories. However, the techniques presented in aggregated manuals are largely outdated and have not received much attention in updates. Consequently, some jurisdictions now prefer to use more up-to-date methodologies, such as those outlined in the following:

- *CARB's Emissions Inventory, Area-Wide Source Methodologies, Index of Methodologies by Major Category* **(CARB, 2008)**
- *EMEP/EEA air pollutant emission inventory guidebook 2009* **(European Environment Agency, 2009)**
- *USEPA AP-42, Fifth Edition, Compilation of Air Pollutant Emission Factors, Volume 1: Stationary Point and Area Sources* **(USEPA, 1995)**
- *USEPA Emission Inventory Improvement Program, EIIP Technical Report Series, Volumes 1-10* **(USEPA, 2007)**
- *USEPA 2008 National Emissions Inventory Data* **(USEPA, 2011b)**
- *USEPA Non-road Engines, Equipment, and Vehicles* **(USEPA, 2011c)**

Furthermore, each jurisdiction constructs a regional air emissions inventory to perform a range of functions (the inventory scope). The scope and content of the inventory is tailored to each jurisdiction's requirements at the time of construction, resulting in differences in the sources that are included, how each source is estimated, and how each source is represented in the inventory.

A comparison of the sources included in each operational regional air emissions inventory in Australia is shown in **Table 3-2**. As noted above, the methodology to estimate emissions from each source is likely to differ significantly between jurisdictions. A comparison of source coverage for each urban area in the remaining three jurisdictions is provided in **Table 3-3**.

The substances included in each air emissions inventory are also variable between jurisdictions. Substances that could be relevant to particulate matter include primary pollutants (TSP, PM_{10} and $PM_{2.5}$) and precursor pollutants (NO_x, NH_3, SO_2, SO_3, VOC and elemental/organic carbon). The coverage of each regional inventory for these substances changes depending on the inventory. Furthermore, as the methodologies used to estimate emissions for each inventory are significantly different, even if an inventory contains a particular substance the source coverage of each inventory is likely to vary considerably between each inventory. This is particularly true for precursor substances such as ammonia and sulfur trioxide. A summary of pollutant coverage for each regional air emissions inventory is provided in **Table 3-4**.

Table 3-1: Summary of active regional air emission inventories in Australia

Regional Air Emissions Inventory	Latest Base Year	Summary
NSW GMR air emissions inventory **(DECC, 2007; OEH, 2012a)**	2008	The study area covers 57,330 km^2 (including ocean), which includes the greater Sydney, Newcastle and Wollongong regions, known collectively as the GMR. Approximately 75% of the NSW population resides in the GMR (approximately 5.3 million people in 2008). OEH (now EPA) aims to update the inventory every 5 years **(OEH, 2012b)**.
Victoria air emissions inventory **(Delaney & Marshall, 2011)**	2006	The study area covers the whole state and includes the airsheds of Port Phillip, Latrobe Valley, Bendigo and Mildura. The population of the region was estimated to be 5.1 million people in 2006. EPA Victoria is currently updating the air emissions inventory to a base year of 2011.
South east Queensland **(QEPA & BCC, 2004)**	2000	The study area covers 23,316 km^2 (land-based area), which includes the Sunshine Coast, Brisbane, Toowoomba and the Gold Coast regions, known collectively as the South-East Queensland Region (SEQR). Approximately 70% of the Queensland population resides in the South-East Queensland Region (approximately 2.5 million people in 2000). Queensland Department of Science, Information Technology, Innovation and the Arts is currently updating the SEQ air emissions inventory for all emission sources with completion expected at end of 2012 **(DSITIR, 2012)**.
Perth air emissions inventory **(DEP, 2003; Rostampour V., 2010)**	1998/1999	The Perth air emissions inventory was constructed in order to report emissions to the National Pollutant Inventory (NPI). The original Perth airshed emissions inventory was compiled for the year 1992, with a later update based on the 1998/1999 period (DEP, 2003). In addition to these inventories, a diffuse emissions study was undertaken by a consultant on behalf of DEC based on the 2004/2005 period. Due to the rapidly increasing number of motor vehicles in the Perth metropolitan area, an update of the vehicle emissions inventory has recently been completed based on the years 2006/2007. The vehicle emissions inventory is generally updated every five years. The vehicle kilometres travelled (VKT) map will be updated for the vehicle emissions inventory for 2011-2012. The inventory is provided to universities on request and the National Pollutant Inventory and may be used for background information in the development of airshed studies **(DEC, 2012)**. The study area covers 8,613 km^2, which includes the major population centres and emission sources in Western Australia Approximately 70% of the Western Australia population resides in the Perth airshed (approximately 1.3 million people in 1998/1999). It is noted that the Perth diffuse air emissions inventory is not in a model-ready format (gridded emissions are not readily available)
Adelaide air emissions inventory **(Ciuk, 2002)**	1998/1999	The South Australian air emissions inventory was constructed in order to report emissions to the National Pollutant Inventory (NPI). The emissions inventory is based on activity that occurred during the 1998/1999 period. The study area covers the five major regional areas of South Australia. Approximately 76% of the South Australia population resides in the study regions (approximately 1.1 million people in 1998/1999). The South Australia EPA also recently completed a gridded air emissions inventory for the entire state covering motor vehicle emissions. The base year for the study was 2006. It is noted that the Adelaide air emissions inventory is not in a model-ready format (gridded emissions are not readily available).

Table 3-2: Summary of source coverage for each regional air emissions inventory (over major urban areas)

Source Type	Source	Inventory/Airshed				
		NSW GMR	Victoria	SEQ	Perth	Adelaide
Biogenic /Geogenic	Agricultural burning	✓	✓	✓	✓	×
	Bushfires and prescribed burning	✓	✓	✓	✓	×
	Fugitive/windborne - agricultural lands and unpaved roads	✓	✓	×	×	×
	Soil nitrification and de-nitrification	✓	✓	✓	✓	×
	Tree canopy	✓	✓	✓	✓	×
	Un-cut grass and cut grass	✓	×	×	×	×
	Marine aerosol	✓	×	×	×	×
Industrial	All industrial sources	✓	✓	✓	✓	✓
Commercial	All major commercial sources	✓	✓	✓	✓	✓
Off-Road	Aircraft (flight and ground support operations)	✓	✓	✓	✓	✓
	Commercial boats	✓	✓	✓	✓	✓
	Commercial off-road vehicles and equipment	✓	✓	✓	✓	×
	Industrial off-road vehicles and equipment	✓	✓	✓	✓	×
	Locomotives	✓	✓	✓	✓	✓
	Recreational boats	✓	✓	✓	✓	✓
	Ships	✓	✓	✓	✓	✓
Domestic-Commercial	Aerosols and solvents	✓	✓	✓	✓	✓
	Barbecues	✓	✓	✓	×	×
	Cutback bitumen	✓	✓	×	✓	✓
	Gaseous fuel combustion	✓	✓	✓	✓	✓
	Graphic arts	✓	✓	✓	✓	✓
	Lawn mowing and garden	✓	✓	✓	✓	✓
	Liquid fuel combustion	✓	✓	✓	✓	✓
	Natural gas leakage	✓	✓	✓	✓	×
	Portable fuel containers	✓	×	×	×	×
	Solid fuel combustion	✓	✓	✓	✓	✓
	Surface coatings	✓	✓	✓	✓	✓
On-Road	All - evaporative	✓	✓	✓	✓	✓
	All - non-exhaust PM	✓	✓	×	✓	✓
	Heavy duty commercial diesel - exhaust	✓	✓	✓	✓	✓
	Light duty commercial petrol - exhaust	✓	✓	✓	✓	✓
	Light duty diesel - exhaust	✓	✓	✓	✓	✓
	Others - exhaust	✓	✓	✓	✓	✓
	Passenger vehicle petrol - exhaust	✓	✓	✓	✓	✓
Other	Architectural and industrial surface coatings	✓	✓	✓	✓	✓
	Pets and humans	×	✓	×	×	×
	Tobacco smoking	×	×	✓	✓	×
	Swimming pools	×	×	×	✓	×

Table 3-3: Summary of source coverage for diffuse emission estimates performed by other jurisdictions

Emission Source	ACT Canberra	Tas Tasmania	NT Darwin
Aeroplanes	✓	✓	✓
Architectural surface coating	✓	✓	✓
Backyard incinerators	×	✓	✓
Bakeries	✓	✓	✓
Barbeques	✓	×	✓
Burning (fuel reduction, regeneration, agricultural)/Wildfires	✓	✓	✓
Cigarettes	✓	×	×
Commercial shipping/boating	NA[(a)]	✓	✓
Cutback bitumen	✓	✓	✓
Domestic/commercial solvents/aerosols	✓	✓	✓
Fuel combustion - sub threshold	✓	✓	✓
Lawn mowing	✓	✓	✓
Liquid fuel combustion	✓	✓	✓
Gaseous fuel burning	✓	✓	✓
Motor vehicles	✓	✓	✓
Motor vehicle refinishing	✓	✓	✓
Paved/unpaved roads	×	✓	×
Print shops/Graphic arts	×	✓	×
Railways	×	✓	✓
Recreational boating	NA	✓	✓
Service stations	✓	✓	✓
Solid fuel burning	✓	✓	×
Structural metal product manufacturing n.e.c.	×	×	✓
Traffic (road line) marking	✓	×	×

(a) NA = not available

Table 3-4: Substance coverage for each regional air emissions inventory

Pollutant type	Pollutant	Inventory/Airshed NSW GMR	Victoria	SEQ	Perth	Adelaide
Primary pollutants	TSP	✓	✓	✓	×	×
	PM_{10}	✓	✓	✓	✓	✓[(a)]
	$PM_{2.5}$	✓	✓	✓	✓	✓[(a)]
Secondary - nitrates	NO_x	✓	✓	✓	✓	✓[(a)]
	NH_3	✓	✓	✓	✓	✓[(a)]
Secondary - sulfates	SO_2	✓	✓	✓	✓	✓[(a)]
	SO_3	✓	✓	×	×	×
Secondary - organic	VOCs	✓	✓	✓	✓	✓[(a)]
	Elemental carbon	×	✓	×	×	×
	Organic carbon	×	✓	×	×	×

(a) Not all sources are included

3.3 Regional air quality modelling

Air quality modelling for policy development is typically undertaken by the jurisdictions, often with support from the Commonwealth Scientific and Industrial Research Organisation (CSIRO). The modelling is generally based on 'hindcasting' in which a series of representative historical air quality episodes or seasons or years are modelled in detail for a business–as–usual emissions base case, and one or more scenarios which represent a potential change in a significant source group **(Cope *et al.*, 2006)**.

The extent to which each jurisdiction uses regional air quality modelling to inform the air quality management decisions varies considerably. Each jurisdiction was sent a questionnaire requesting information on:

- Regional air dispersion modelling currently undertaken.
- Whether regional PM modelling is conducted, and whether secondary particulate formation is assessed.
- Resources available internally to perform regional air dispersion modelling.

The information received indicated that no jurisdiction in Australia models regional PM through a regional modelling platform. Past and current efforts in regional air quality modelling have focussed on understanding ozone formation in NSW, Victoria and South-East Queensland. The jurisdictions that have performed regional air quality modelling include:

- NSW (Sydney)
- Victoria (Melbourne)
- Queensland (South-East Queensland)
- Western Australia (Perth)

Information provided by each of these jurisdictions on resources available to perform regional air quality modelling is summarised in **Table 3-5**.

Table 3-5: Resources available to conduct regional air quality modelling

Jurisdiction	Resources Available
NSW	EPA has a team of four modellers working on regional air quality modelling
Victoria	EPA Victoria do not conduct regional PM modelling specifically as there is low confidence in the current 2006 emission estimates for windblown PM **(EPAV, 2012)**. No information is available on resources available to perform regional air quality modelling internally by EPAV.
Queensland	Resources limited to one person that can undertake regional dispersion modelling. Current priorities would need to be considered if Queensland were to reallocate these to PM modelling.
Western Australia	DEC does not currently have the resources to undertake regional air dispersion modelling of PM.

NSW, Victoria and Queensland conduct regional air quality simulations using the TAPM-CTM model. The Air Pollution Model (TAPM) **(Hurley, 2008)** is an integrated prognostic meteorological/air quality model. TAPM is widely used in Australia and was developed by CSIRO Marine & Atmospheric Research. The Chemical Transport Model (CTM) add-on to TAPM is used for urban airsheds requiring more complex treatment of chemistry (such as through using the Lurmann Carter Coyner (LCC) or Carbon Bond (CB) 04 mechanisms) **(Cope *et al.*, 2009)**. The TAPM-CTM model includes modules for simulating inorganic aerosol formation and secondary organic aerosol formation.

3.4 Population statistics

Population statistics for use in a full impact pathway approach are available in Australia. In previous censuses, 'collection districts' were used for both the collection and dissemination of data. From 2011, the ABS introduced the Australian Statistical Geographic Standard (ASGS), in which the basic structural element is the Mesh Block. Mesh Blocks are so small that they can be aggregated reasonably accurately for different geographical regions, as well as administrative, management and political boundaries.

Population data for the latest census year (2011) are available from the ABS at the following levels of aggregation:

- Statistical Areas Level 1 (SA1)
- Statistical Areas Level 2 (SA2)
- Statistical Areas Level 2 (SA3)
- Statistical Areas Level 2 (SA4)
- State/Territory
- Australia

Population data are also described in a number of other ways within the ASGS. A useful concept in the context of this project is that of the Significant Urban Area (SUA) **(ABS, 2012)**. The SUA structure provides a geographical standard for the publication of statistics on concentrations of urban development with a population of 10,000 people or more. The regions are constructed from whole SA2s. They do not necessarily represent a single urban centre, as they can represent a cluster of related urban centres with a core urban population over 10,000. They can also include related peri-urban and satellite developments, as well as the area into which the urban development is likely to expand **(ABS, 2012)**. An SUA is identifiable by a unique 4 digit non-hierarchical code.

3.5 Monitoring

Air quality monitoring is performed by each jurisdiction in Australia in accordance with the AAQ NEPM requirements. Monitoring methods vary between each jurisdiction. The most common method for measuring and reporting PM_{10} and $PM_{2.5}$ concentrations is the Tapered-Element Oscillating Microbalance (TEOM).

The reference method for monitoring $PM_{2.5}$ is the manual gravimetric method. The method is a non-continuous (batch), 1-day-in-3 technique that requires pre- and post-laboratory filter weighing. This introduces a significant time delay in acquiring data. The main advantage of the

TEOM is that concentrations are reported on a continuous basis. However, the TEOM does not have reference or equivalence status through the USEPA designations for monitoring of $PM_{2.5}$ due to issues related with the loss of volatile components.

It was noted in the Air NEPM Review that as the high-volume sampler (a NEPM reference method) is labour-intensive and there are advantages of obtaining continuous measurements, TEOMs have almost universally been adopted by jurisdictions to measure PM_{10}. The PRC's '*Technical paper no. 10: Collection and reporting of TEOM PM_{10} data'* (2001) provides guidance on the handling of TEOM PM_{10} data by way of an adjustment factor to generate equivalent information to the NEPM reference method. These recommendations have not been implemented consistently by all jurisdictions, and equivalence remains an area of concern for PM_{10} data **(NEPC, 2011)**.

PM monitoring data are useful for validating regional air quality models that could be used to develop a full impact pathway approach in Australia. Care would need to be applied when using the monitoring data to account for the sensitivity of the measured concentration to the measurement method used. A summary of methods used to report concentrations of particulate matter by each jurisdiction is provided in **Table 3-6**.

Table 3-6: Particulate matter monitoring methods used by jurisdictions(a)

Jurisdiction	PM_{10}	$PM_{2.5}$
New South Wales	Gravimetric reference method TEOM	Gravimetric reference method TEOM
Victoria	TEOM	Gravimetric reference method TEOM
Queensland	FDMS TEOM TEOM	FDMS TEOM TEOM DOAS
Western Australia	TEOM	TEOM
South Australia	TEOM	TEOM
Tasmania	Gravimetric reference method TEOM Microcal air sampler DustTrak	Gravimetric TEOM DustTrak
Australian Capital Territory	Gravimetric reference method BAM	Gravimetric reference method BAM
Northern Territory	Partisol dichotomous sampler TEOM	Partisol dichotomous sampler

(a) FDMS: Filter Dynamic Measurement System; DOAS: Differential Optical Absorption Spectroscopy; BAM: Beta-Attenuation Monitor

3.6 Summary

The analysis of Australian air quality management conditions and capabilities has focused primarily on each jurisdiction's emission inventories, modelling capabilities and, to some extent, monitoring activities.

The starting point in developing damage costs for a specific location is to identify local emission profiles. This information is then used as an input into pollution modelling to estimate ground-level concentrations. Contributions to ground-level PM concentrations are influenced by emissions from point sources and area sources of secondary particles and primary particles.

At present, there is no consistency across the jurisdictional air emissions inventories, with some inventories not being suitable for regional air quality modelling. Furthermore, no jurisdiction in Australia is currently simulating regional PM. As understanding the "emission-to-impact" relationship is an essential part of the impact pathway process, this greatly reduces the capability to develop optimally localised damage cost values at this point in time.

A summary of the current status of elements required to develop a full impact pathway is outlined in **Table 3-7**.

It is noted that NSW almost has all the required information to develop localised damage costs based on a full impact pathway. In order to develop the full impact pathway, the NSW air emissions inventory would need to be supplemented with elemental and organic carbon emission estimates for input into a regional modelling platform. This could be achieved relatively simply by using PM speciation profiles consistent with those used to construct the 2008 air emissions inventory.

Similar to NSW, it is also noted that Victoria almost has all the required information to develop localised damage costs based on a full impact pathway. EPA Victoria has also included emission estimates for elemental carbon and organic carbon in order to be used with secondary particulate models. However, it is noted that EPA Victoria has a low confidence in the current emission estimates for windblown PM.

It is expected that the south east Queensland air emissions inventory once updated at the end of 2012 will contain all the required information to develop localised damage costs based on a full impact pathway. However, it is likely that this information will also need to be supplemented with estimates of organic and elemental carbon for each source.

CSIRO have been working with NSW EPA and other jurisdictions developing PM modelling capabilities utilising the TAPM-CTM and CSIRO's Cubic Conformal Atmospheric Model (CCAM). Currently CSIRO has been studying the sensitivity of secondary particulate formation to changes to secondary precursor emissions. This information will be useful when developing localised damage costs. Efficiencies in developing regional modelling are likely to be realised if jurisdictions partner with CSIRO to develop the modelling requirement. Furthermore, jurisdictions are encouraged to consult CSIRO prior to updating air emission inventories to ensure that emission inventories are suitable for modelling secondary air pollution formation.

Table 3-7: Summary of current status of essential elements to develop localised damage cost values based on a full impact pathway analysis

Jurisdiction				
NSW GMR[a]	**Victoria**	**SEQ[b]**	**Perth**	**Adelaide**
Emissions inventory				
All major sources included?				
True	False The most significant source not included is marine aerosol. TAPM could be used to supplement this source. Current update should include estimates for this source.	False Fugitive windborne, marine aerosols and emissions from paved roads (wheel generated dust) not included in 2000 inventory. Current update likely to include estimates for these sources.	False Fugitive windborne and marine aerosols were not included in the diffuse air emissions inventory.	False Biogenic/Geogenic emission sources have not been estimated for the Adelaide airshed.
Model ready?				
True The NSW GMR air emissions inventory is suitable for regional air quality modelling and readily exportable in model-ready file formats.	True EPA Victoria is currently updating the air emissions inventory to a base year of 2011.	False The air emissions inventory will be in a format suitable for regional air quality modelling when the current update (expected at end of 2012) is completed	False Inventory designed for diffuse sources only. Spatial and temporal variation of emissions not assigned.	False Inventory designed for diffuse sources only. Spatial/ temporal variation of emissions not assigned. Significant emission sources (e.g. biogenic) excluded.
Primary pollutants?				
True All primary pollutants are included (TSP, PM_{10}, $PM_{2.5}$)	True All primary pollutants are included (TSP, PM_{10}, $PM_{2.5}$)	True All primary pollutants are included (TSP, PM_{10}, $PM_{2.5}$)	False PM_{10} and $PM_{2.5}$ are included in the emission estimates but not TSP.	False PM_{10} and $PM_{2.5}$ are included in the emission estimates but not TSP.
Secondary precursor pollutants?				
False Does not include emissions of elemental carbon	True Includes emissions of all substances	False Does not include emissions of SO_3 or elemental carbon	False Does not include emissions of SO_3 or elemental carbon	False Does not include emissions of SO_3 or elemental carbon
Regional Modelling				
Modelling platform				
TAPM-CTM	TAPM-CTM	TAPM-CTM	Not applicable	Not applicable
Resources available				
EPA has a team of four modellers working on regional air quality modelling. EPA does not currently model regional PM concentrations.	EPAV does not model regional PM as there is low confidence in the 2006 estimates for windblown PM (**EPAV, 2012**). No information on resources for regional air quality modelling.	Resources limited to one person that can undertake regional dispersion modelling. Current priorities would need to be considered for PM modelling.	DEC does not currently have the resources to undertake regional air dispersion modelling of PM.	Not applicable
Population statistics				
Population statistics are available for Australia (2011 census year)				
Monitoring				
All jurisdictions conduct ambient air quality monitoring of PM. Care will need to be practised when using monitoring data for model validation in considering the differences in monitoring techniques between sites. Generally PM monitoring data are available for capital cities and industrial areas but not elsewhere (such as rural areas). A summary of methods used to report concentrations of particulate matter by each jurisdiction is provided in Table 3-6.				

[a] NSW GMR: NSW Greater Metropolitan Region

[b] SEQ: South East Queensland

3.7 Cost estimates

In order to perform regional air quality modelling (with chemistry), emission inventories would need to be updated and be model ready.

In order to develop a regional emissions inventory the costs detailed in **Table 3-8** are estimated. It is noted that the annualised cost of $156,800 per annum is the estimated required annual cost whether or not a jurisdiction has an existing regional air emissions inventory or not (as emission inventories need to be maintained).

In order to retrofit existing 'model-ready' emission inventories with additional substances such as elemental and organic carbon. The estimated cost is considered negligible at approximately $20,000. However, the task is likely to take some time (two to three months) for data handling (computer run times) depending on each jurisdiction's inventory configurations.

Table 3-8: Estimated costs to update and maintain an emissions inventory

Cost Item	Value	Unit
Salary (including 10% superannuation)	$132,000	$/year/person
Number of people	3	people
Years	1.5	years
Salary Cost	*$594,000*	*per inventory update*
Fees (buying data, domestic surveys)	**Value**	**Unit**
Domestic survey	$50,000	per update
Other fees (purchasing data, stationary)	$20,000	per update
Hardware (server, programs)	$20,000	per update
Software (custom built or off the shelf)	$100,000	per update
Total Fees to update air emissions inventory	*$190,000*	*per inventory update*
Total cost to update air emissions inventory	**$784,000**	
Annual cost to update and maintain air emissions inventory (based on a five year update period)	**$156,800**	**(per annum)**

To conduct regional air quality modelling of the emission inventories including secondary PM, ballpark cost estimates for initial studies are estimated to be up to $250,000 per jurisdiction. However, the validity of the modelling will be highly uncertain until initial studies have been completed and assessed against monitoring results. It is noted that there is a high level of uncertainty associated with significant primary PM sources such as windblown dust and marine aerosols. Furthermore, the estimation of secondary organic aerosols is highly uncertain, with the state of science relating to secondary organic aerosol formation in the early stages of development. However, it is noted that the understanding of nitrate and sulfate formation is better understood.

4 REVIEW OF SECONDARY PARTICLES

4.1 Background and objectives

Secondary PM is generally formed as a result of atmospheric oxidation reactions involving both inorganic and organic gaseous precursors **(USEPA, 2009)**. The oxidised substances may be either natural or anthropogenic in origin. The process by which secondary PM is formed is termed 'nucleation', whereby molecules of low volatility condense to form solid or liquid matter. There are two distinct types of nucleation process. Most secondary PM formation occurs by 'heterogeneous' nucleation in which newly formed substances condense onto existing particles, thereby causing the growth of those particles. The second process is called 'homogeneous' nucleation. Some newly formed molecules have extremely low vapour pressure and, in the absence of an abundance of pre-existing particles (which would favour heterogeneous nucleation), will condense with one another to form wholly new particles **(AQEG, 2005)**.

It was noted in **Chapter 2** that the US, EU and UK valuation studies now include secondary PM, and that damages due to secondary PM are generally stated in terms of the cost per tonne of emission of the precursor gases.

The objectives of this part of the work were to review the international literature on secondary particles, to summarise the current knowledge, to summarise modelling studies, and to understand whether it would be possible to make inferences about Australian damage costs from the data in other countries.

4.2 Formation and sources of secondary particles

4.2.1 Secondary inorganic particles

The formation of secondary inorganic particles is comparatively well understood, although a number of mechanistic details still remain to be determined **(USEPA, 2009)**. Secondary inorganic particles are composed mainly of ammonium sulphate ($(NH_4)_2SO_4$) and ammonium nitrate (NH_4NO_3). These originate from the conversion of sulphur and nitrogen oxides in the atmosphere to acids, which are then neutralised by atmospheric ammonium (NH_4^+).

The best known process of homogeneous nucleation occurs when sulphuric acid (H_2SO_4) is formed from the atmospheric oxidation of SO_2. In addition, gaseous nitric acid (HNO_3) is formed from the oxidation of nitrogen dioxide (NO_2), which itself is mainly derived from the oxidation of NO released during fossil fuel combustion. H_2SO_4 and HNO_3 are scavenged by existing particles and droplets to form sulphate and nitrate aerosols. The precursor to atmospheric ammonium is ammonia (NH_3). NH_3 emissions are dominated by agricultural sources, which are mainly due to the decomposition of urea and uric acid in livestock waste. Other significant anthropogenic sources of ammonia include waste disposal (landfills) and composting facilities. NH_3 is efficiently taken up into acidic sulphate (SO_4^{2-}) and nitrate (NO_3^-) aerosols, formed by the processes described above, leading to the formation of ammonium aerosol **(AQEG, 2005)**.

Water is a typical component of PM, but the amount measured is very variable and depends on the measurement method. Water binds to hydrophilic components in PM such as sulphate, ammonium, nitrate and sea salt. Reducing emissions of SO_2, NO_X and NH_3 lowers the concentration of their secondary PM components and therefore reduces the overall $PM_{2.5}$ concentration. Lower secondary PM levels may also reduce the uptake of water by fine particles.

This leads, in turn, to a further reduction in the $PM_{2.5}$ concentration. In this way water can magnify trends in secondary PM **(Matthijsen and Brink, 2007)**.

4.2.2 Secondary organic particles

Secondary organic particles – also commonly referred to as secondary organic aerosols (SOA) - are linked to the formation and continuing transformation of low-volatility organic compounds in the atmosphere. Hydrocarbon precursors of SOA can be either biogenic or anthropogenic in origin. The formation of low-volatility compounds is governed by a complex series of reactions involving a large number of organic species. **Kroll and Seinfeld (2008)** identified three general types of chemical process that can reduce the volatility of organic compounds:

1. *Oxidation reactions in the gas phase*. These can reduce volatility by the addition of polar functional groups or increase it by the cleavage of carbon-carbon bonds.

2. *Reactions in the particle phase*. These include oxidation reactions as well as accretion reactions (non-oxidative processes) leading to the formation of high-molecular-weight species.

3. *Continuing chemistry (in either phase) over several generations*. Organic carbon in the atmosphere is continually subject to reactions throughout its atmospheric lifetime (until lost by physical deposition or oxidised to CO or CO_2), implying continual changes in volatility over timescales of several days. The composition of SOA also evolves through repeated cycles of volatilisation and condensation of chemical reaction products in both the particle and gas phases.

The atmosphere contains many thousands of different organic oxygenates possessing a wide range of properties and, therefore, different propensities to undergo gas-to-particle transfer. However, the gas-phase oxidation of each organic precursor broadly follows the same pattern **(AQEG, 2005)**. Certain classes of compound are more likely to lead to aerosol formation by virtue of their high reactivity and the types of oxidation product formed. Of particular significance are large, cyclic, unsaturated compounds such as toluene and xylenes from vehicle exhaust. A review of SOA studies by **USEPA (2009)** noted that oligomers are likely to be a major component of organic carbon in aerosol samples, and that small but significant quantities of organic aerosol are formed from the oxidation of isoprene (released predominantly from vegetation). **USEPA (2009)** concluded that ambient samples can contain mixtures of SOA from different sources at different stages of processing, some with common reaction products.

As a result of this complexity a great deal of uncertainty exists around the process of SOA formation, and source identification presents a substantial challenge **(Kroll and Seinfeld, 2008; USEPA, 2009)**. **Warren *et al.* (2009)** found that temperature has a large effect on the total SOA formation. Regional SOA models use environmental chamber data for a single temperature, and this is likely to lead to errors when the data are applied to geographical regions with other temperatures.

4.2.3 Formation rates

The formation of secondary particles happens relatively slowly; the overall oxidation rates of SO_2 and NO_2 are around 1% per hour and 5% per hour respectively. The relatively slow rate of these reactions means that secondary particles are usually observed many kilometres downwind of the sources of the precursor emissions. This is further compounded since the size of the resulting particles means they generally have a relatively long atmospheric lifetime. SO_2 emission sources typically contribute to particulate sulfate hundreds to thousands of kilometres downwind, whereas NO_X emission sources typically contribute to particulate nitrate tens to hundreds of kilometres downwind. As a consequence, there is a reasonably even distribution of secondary PM on a regional scale, with fewer differences between urban and rural areas than for primary particles **(Laxen *et al*., 2010)**.

4.2.4 Australian studies

There have been relatively few studies of secondary PM - and in particular SOA - in urban areas of Australia. The main activities and existing literature are summarised below.

The main data available in Australia for aerosol sampling are from the Australian Nuclear Science and Technology Organisation (ANSTO). ANSTO has been sampling $PM_{2.5}$ – mainly along the east coast of Australia - since 1991. During this time fine particles have been routinely collected at selected urban, rural and industrial sites. Ion beam analysis and positive matrix factorisation have been used to characterise particles and to identify sources. This long term aerosol sampling study is the only one of its kind taking place in Australia **(ANSTO, 2010)**. One of the largest components of $PM_{2.5}$ at the ANSTO sites is ammonium sulfate. Between 1998 and 2008 the average ammonium sulfate concentration at 10 sites was 25% (range 18-31%) **(ANSTO, 2008)**. Data for specific sites are available from the ANSTO web site[4].

Receptor modelling of various PM size fractions has been undertaken extensively in Brisbane - and to a lesser extent in Melbourne, Sydney and Adelaide - by Griffith University **(Chan *et al.*, 1997, 1999, 2000, 2008, 2011)**. These studies have shown that secondary particles form a significant component of PM_{10} and $PM_{2.5}$. It was observed by **Chan *et al.* (1999)** that secondary organics and secondary sulfates accounted for 21% and 14% of $PM_{2.5}$ respectively at a suburban site in Brisbane surrounded by forest. Most of the secondary products were related to motor vehicle exhaust. In a study in the four cities mentioned above, **Chan *et al.* (2008)** found that, on average, secondary nitrates/sulfates contributed about 25% of the mass of the $PM_{2.5}$ samples. Secondary sulfates and nitrates were found to be spread out evenly within each city. The average contribution of secondary nitrates to fine particles was also rather uniform in different seasons, rather than being higher in winter as found in other studies. It was suggested that this could be due to the low humidity conditions in winter in the Australian cities which makes the partitioning of the particle phase less favourable in the NH_4NO_3 equilibrium.

The composition of $PM_{2.5}$ was determined by **Friend *et al.* (2011)** for two sites in the South-East Queensland region (Rocklea and South Brisbane), and sources were analysed using a receptor model. The five common sources of $PM_{2.5}$ at both sites were motor vehicle emissions, biomass burning, secondary sulfate, sea salt and soil. Secondary sulfate was the most significant contributor (up to 40%) to $PM_{2.5}$ aerosols at the South Brisbane site, and the second most important at the Rocklea site. Biomass burning was the most significant source at the Rocklea site. In addition, dust storms that caused the $PM_{2.5}$ concentration to exceed the NEPC standard were observed at both sites.

The earliest estimates of the contribution of SOA to particulate mass in Australian cities were obtained by **Gras *et al.* (1992)** and **Gras (1996)**, although SOA was grouped with secondary inorganic aerosol. The first study to determine the specific contribution of SOA to $PM_{2.5}$ in an

[4] http://www.ansto.gov.au/discovering_ansto/what_does_ansto_do/live_weather_and_pollution_data/aerosol_sampling_program

Australian urban context (Melbourne) was by **Keywood *et al.* (2011)**. SOA was estimated indirectly using the elemental carbon tracer method. The median annual SOA concentration was found to be 1.1 μg/m^3, representing 13% of $PM_{2.5}$. Significantly higher SOA concentrations were determined when bushfire smoke affected the airshed, and SOA displayed a seasonal cycle. The SOA fraction of $PM_{2.5}$ was greatest during the autumn and early winter months when the formation of inversions allowed build-up of particles produced by domestic wood-heaters. **Keywood *et al.* (2011)** also suggested that biogenic VOCs are a source of SOA at both urban and non-urban sites. During summer the oxidation of biogenic VOCs oxidation is the most likely source of SOA, whereas during winter the oxidation of volatile species associated with wood-smoke emissions are a probable source of non-fossil SOA.

An important issue in Australia is biomass burning. In rural towns, smoke from biomass burning such as prescribed burning of forests, bushfires and stubble burning is often claimed to be the major source of air pollution. **Reisen *et al.* (2011)** measured $PM_{2.5}$ at two rural locations in southern Australia. Monitoring clearly showed that, on occasions, air quality in rural areas was significantly affected by smoke from biomass combustion, with $PM_{2.5}$ showing the greatest impact. Biomass burning emits a complex mixture of air pollutants, both as gases and particulate matter. Gaseous species include carbon monoxide, hydrocarbons and a large range of trace gases. Significantly higher SOA concentrations have been observed when bushfire smoke affects an airshed.

4.2.5 Secondary PM damage costs

In contrast to primary PM, an analysis of the trends with secondary PM precursor NO_x from **Watkiss (2002)** presents a different picture. A comparison of the damage costs of NO_x is presented in **Figure 4-1** below and shows that emissions and impacts do not differentiate greatly between urban and rural locations. Local population density therefore has less impact in determining the damage costs of emissions from secondary PM precursors.

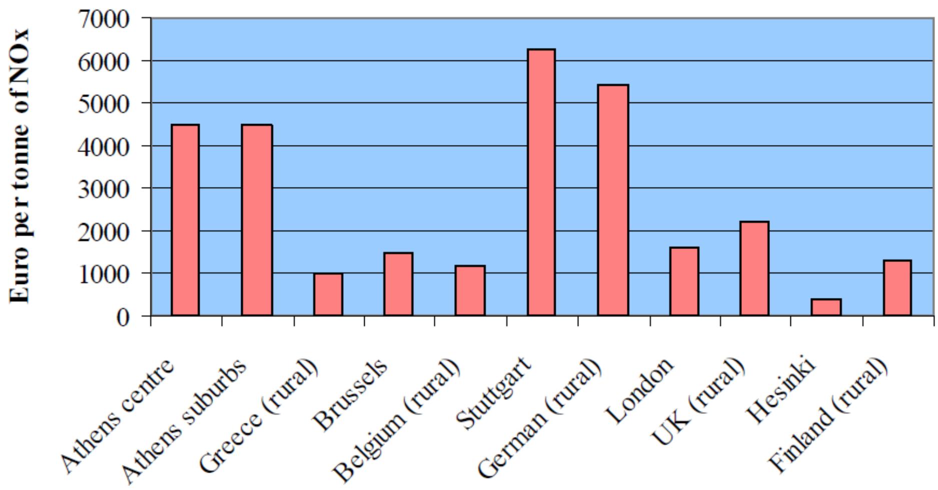

Figure 4-1 Damage Costs for NO_X from transport for rural and urban locations in Europe (Watkiss, 2002)

For this reason, the impacts at the local level are less important for secondary PM. The formation of secondary pollutants is also extremely regionally specific, depending on the local levels of other pollutants and the specific meteorological conditions. As a result, it is more difficult to accurately transfer unit pollution costs of secondary PM precursors between countries.

In some countries the damage costs are higher in rural areas, which is possibly due to secondary pollutants forming over time and distance in the atmosphere, therefore often manifesting at distances away from the souce.

The transfer of secondary PM damage costs to Australia is extremely uncertain, because of differences in factors affecting formation and partly because of the low population density and hence exposure outside of urban centres in Australia.

Therefore, in line with the indicated uncertainties involved, it is recommended that the transfer of secondary PM damage costs is not undertaken at this point and further modelling and analysis of the Australian conditions is undertaken to produce a set of appropriately developed values.

4.3 Summary and implications for Australian conditions

4.3.1 Summary

Very few studies in Australia have dealt with SOA. There is very little information on the proportion of $PM_{2.5}$ from secondary organic processes. This could be due to measurement techniques commonly used in Australia for $PM_{2.5}$. Commonly, the measuring instruments are heated (e.g. TEOM), and as such the (semi-volatile) secondary organic component of $PM_{2.5}$ is unlikely to be captured.

Data on secondary PM at Australian sites are rather limited. **Watkiss (2002)** noted that within Australia nitrate formation will be extremely site-specific, with significant variations between different states and cities. To evaluate the role of nitrates a detailed assessment is needed to understand the levels of particulate nitrate aerosol in urban PM_{10} levels, the types of aerosol species present, the background concentrations of other pollutants involved (*e.g*. ammonia) and the regional scale photochemical production of particulate nitrate.

Notwithstanding the above, there are some broad similarities between Europe, the US and Australia in terms of $PM_{2.5}$ composition and the contribution of secondary particles. For example, the sulfate contribution to $PM_{2.5}$ in Eastern Australia seems to be similar to that in the Western United States. However, the formation of secondary particles is complex, the understanding is incomplete, and the variability in the data is large. Moreover, some different metrics and reporting formats are in use. There may be some important differences in how secondary particles are formed in the three regions, but these cannot yet be quantified.

4.3.2 Implications

The above summary implies that, given the knowledge gaps in Australian secondary PM characteristics, if damage cost values from Europe and the US are applied to Australia, there will be a good deal of uncertainty in the outcome. Were damage costs are transferred to Australia from other countries, then it would probably be more beneficial to focus on more easily quantifiable differences, such as population density than on secondary particle formation processes. In addition, regional-level impacts dominate the damage costs associated with secondary pollutants.

5 DEVELOPMENT AND APPLICATION OF NEW PM VALUATION METHODOLOGY

In this Chapter we propose a new methodology for monetising the health impacts of changes in **primary $PM_{2.5}$ emissions**. This methodology is designed for use in policy assessment and other work relating to air quality management in NSW and in Australia.

5.1 Rationale

5.1.1 Primary $PM_{2.5}$

The application of the impact pathway approach to derive Australia-specific unit damage costs for changes in PM emissions is not currently feasible on account of the time and resource implications. It is estimated that the development of an adequate emission inventory for each Australian jurisdiction would take approximately 1.5 years and would cost around $800,000. Moreover, the cost of conducting the required regional air quality modelling is estimated to be up to $250,000 per jurisdiction. We have therefore adopted a pragmatic approach in which existing damage cost values from the UK **(Defra, 2012)** have been transferred to Australia and adjusted accordingly.

The health impacts and costs of PM emissions depend on population exposure. The local population density is therefore a critical parameter; an emission reduction in a densely populated area will have a greater relative health benefit than one in a less densely populated area. This is demonstrated in **Figure 5-1**, which shows the relationship between unit damage cost and population density for $PM_{2.5}$ emissions at various urban and rural sites across London and the UK, covering a wide range of population density. There are orders-of-magnitude differences between the damage costs associated with central urban and rural locations.

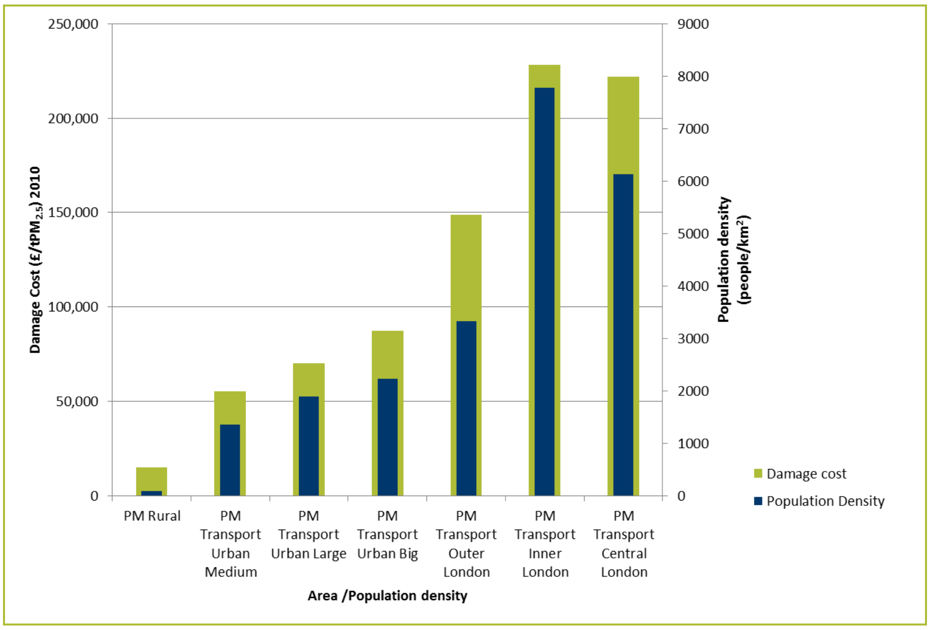

Figure 5-1: Unit damage costs for primary $PM_{2.5}$ emissions and population density in the UK (costs in pounds sterling at 2010 prices) (Defra, 2012)

The data in **Figure 5-1** were used as the basis for the development of Australian unit damage costs. At this stage it is important to note the following:

- Because the amount of ambient $PM_{2.5}$ monitoring data was rather limited, Defra expressed health impacts in terms of PM_{10}. However, the unit damage costs were dominated by mortality from chronic expposure, and the $PM_{2.5}$ function from **Pope *et al.* (2002)** was applied directly. Moreover, Defra dealt with the marginal costs associated with changes in emissions, and (for combustion sources), most of the PM_{10} emitted is $PM_{2.5}$. Therefore, the Defra values are effectively for $PM_{2.5}$.

- Apart from the 'rural' category, the data in the Figure relate to PM from transport sources. Defra also presented unit damage costs for several other categories which are not included in the Figure (including 'electricity supply', 'domestic', 'agriculture', 'waste' and 'industry'), as well as for gaseous pollutants. However, in line with the Defra methodology it has been assumed that $PM_{2.5}$ from any source has an identical health impact per population-weighted unit of exposure. This allows the method to be applied to all emission sources.

Whilst we have used data from the UK as the basis for the Australian approach, there would also have been strong grounds for using either an EU or US methodology. For example, as in the UK, unit damage costs have been developed in the EU and the US using multi-sector emission inventories with complex dispersion modelling, and studies have generated damage cost values for a range of population densities. However, the UK data were used here for the following reasons:

- The Defra analysis was more sophisticated than that used in the EU CAFE programme, due to a greater disaggregation of damage costs according to location and population-weighted exposure. The EU damage costs are only available as national aggregated values, and the uneven distribution of population density over large areas distorts the relationship between emissions and cost.

- The Defra analysis involved the use of the VOLY, but the available data from the US are based on the VSL. The Australian Commonwealth DEWHA study **(Jalaludin *et al.*, 2009)** recommended that the use of the VOLY is preferable to the use of the VSL when monetising the effects of air pollution on premature mortality. The response function for the chronic effects of exposure on mortality (from life tables) provides an output in terms of the change in life expectancy (of the entire community) that can be directly valued using VOLY estimates. From this perspective it is far more robust than outputs stated in terms of the number of deaths brought forward by air pollution, and accounts for the fact that air pollution is likely to be a contributory factor to death rather than the only, or indeed the primary, factor. However, weighing against this is strong empirical evidence from the valuation literature that VSL estimates are more robust as a metric and better reflect preferences.

- The Defra functions include a small number of morbidity functions. The functions used in the UK study are similar to those used in Australia, and neither country includes as many morbidity outcomes as the US and EU methods. The studies in the UK and Australia have also omitted non-health impacts, although this is because previous work **(Watkiss *et al.*, 2006)** has shown that these are not significant in overall monetary terms when compared with health effects.

- The Defra data displayed a clear relationship between the unit damage cost for primary PM and population density, as shown in **Figure 5-1**. The spatial analyses in the EU and US studies were less detailed.

We do accept, however, that the US valuation approach is statistically more rigorous than the Defra method. Defra relied on a single WTP study from the UK and asked questions to elicit preferences for avoiding small losses of life, though this was specifically undertaken for the air pollution context. The studies in the US (and Australia) used meta-analyses of a large range of studies, but these were not specific to air pollution.

5.1.2 Secondary $PM_{2.5}$

The transfer of secondary $PM_{2.5}$ damage costs to Australia is considerably more uncertain than the transfer of primary particle damage costs. Secondary particle formation is especially dependent upon local meteorological and geographical conditions. For these reasons, the costs associated with secondary particle precursors (SO_x, NO_x and NH_3) were excluded.

5.2 Development method

The unit damage costs for primary $PM_{2.5}$ published by **Defra (2012)** were stated in pounds sterling at 2010 prices, and were based on the UK VOLY[5] (relating to mortality from chronic exposure) at 2004 prices (the values were updated from the original WTP study **(Chilton *et al.*, 2004)**). An Australian VOLY (in A$, and based on the Australian valuation method) was available in 2008 prices from the Australian Safety and Compensation Council **(ASCC, 2008)**. The UK damage costs therefore had to be adjusted to account for the difference between the UK VOLY and the Australian VOLY, and had to take into account differences in currency and inflation.

The procedure used to make these adjustments involved the steps described in the following Sections. The method used minimised the effects of exchange rate fluctuation and differences in inflation in the UK and Australia. One disadvantage of the method was that all costs were related only to mortality, and ignored other effects (*e.g*. certain morbidity endpoints, atmospheric visibility). However, the costs associated with these effects would be small compared with those associated with mortality.

5.2.1 Step 1: YOLL per tonne of $PM_{2.5}$

In the first step, the number of life years lost per tonne of pollutant emitted was calculated for the various area types using the UK data.

[5] Is should be noted that in applying these VOLY values, the output from the life tables was used to estimate the profile of life years lost over time, taking into account that the life years lost do not all occur immediately, even from a one-year pollution pulse. The VOLYs in future years were adjusted in line with Government appraisal guidance (*i.e*. increased at 2% per year for each future year, and discounted using a discount rate of 3.5%).

Damage costs can be expressed in terms of the following equation[6]:

$$C = VOLY \times YOLL_{PM} \qquad \text{(Equation 4)}$$

Where:

C is the damage cost in the unit of currency per tonne of PM emitted

$VOLY$ is the value of one life year in the unit of currency

$YOLL_{PM}$ is the number of life years lost per tonne of PM emitted

This equation was rearranged to give the value of $YOLL_{PM}$ for each area type based on UK data for a common year. The actual year was not considered to be important[7]. For simplicity, 2010 was used as the UK damage costs are stated in 2010 prices.

The 2004 UK VOLY in pounds sterling was adjusted to a 2010 value (**Table 5-1**). The values of YLL_{PM} for the different UK area types were then calculated using the damage costs for the area types in 2010 and the UK VOLY for 2010 (**Table 5-2**).

Table 5-1: Adjustment of UK VOLY to 2010 prices

	Low[(a)]	Central	High[(a)]
2004 UK VOLY	£21,700	**£29,000**	£36,200
Inflator 2004 to 2010[(b)]	1.181	1.181	1.181
2010 UK VOLY	£25,628	**£34,249**	£42,752

(a) 95% confidence intervals
(b) http://www.thisismoney.co.uk/money/bills/article-1633409/Historic-inflation-calculator-value-money-changed-1900.html

Table 5-2: Calculation of $YOLL_{PM}$ per tonne of $PM_{2.5}$ by area type

UK area type	$PM_{2.5}$ damage cost (£/tonne, 2010 prices)			YOLL/tonne (based on central estimate)
	Low	Central	High	
Central London	£251,961	**£221,726**	£173,601	**6.47**
Inner London	£259,129	**£228,033**	£178,540	**6.66**
Outer London	£169,261	**£148,949**	£116,621	**4.35**
Urban Big	£99,241	**£87,332**	£68,377	**2.55**
Urban Large	£79,944	**£70,351**	£55,081	**2.05**
Urban Medium	£62,853	**£55,310**	£43,305	**1.61**
Rural	£17,091	**£15,041**	£11,776	**0.44**

[6] This approach introduces an error because the Defra damage costs were derived using life tables, and discount rates were applied to life years in future years. The equation will, in fact, lead to an underestimate in YOLL. However, the error is probably quite small, because most life years will be lost early on.

[7] Given that the UK and Australia have reasonably similar population profiles it is reasonable to assume that there would be similar epidemiological outcomes in both countries to exposure to similar levels of particulate matter. Moreover, the epidemiological response does not change much from year to year.

5.2.2 Step 2: Unit damage costs by area type

In the second step the YOLL per tonne of $PM_{2.5}$ for each UK area type was multiplied by the Australian VOLY for 2011 to give damage costs for 2011.

The Australian VOLY for 2008 was converted to 2011 prices (**Table 5-3**). For each type of area the YOLL values from **Table 5-2** were then multiplied by the 2011 Australian VOLY from **Table 5-3**. The results of this calculation are shown in **Table 5-4**.

Table 5-3: Adjustment of Australian VOLY to 2011 prices

	Low[(a)]	Central	High[(a)]
2008 Australian VOLY	$164,553	**$266,843**	$360,238
Inflator 2008 to 2011[(b)]	1.083	1.083	1.083
2011 Australian VOLY	$178,211	**$288,991**	$390,138

(a) 95% confidence intervals
(b) www.rba.gov.au/calculator/annualDecimal.html

Table 5-4: Damage costs in Australian dollars

UK area type	Damage cost (A$/tonne of $PM_{2.5}$, 2011 prices)		
	Low	Central	High
Central London	$1,153,727	**$1,870,910**	$2,525,729
Inner London	$1,186,545	**$1,924,129**	$2,597,573
Outer London	$775,040	**$1,256,823**	$1,696,710
Urban Big	$454,422	**$736,902**	$994,818
Urban Large	$366,064	**$593,617**	$801,383
Urban Medium	$287,799	**$466,702**	$630,048
Rural	$78,264	**$126,915**	$171,335

5.2.3 Step 3: Unit damage cost function

In the third step, linear regression functions were fitted to the damage cost data from **Table 5-4** and the associated population densities for the respective area types. This effectively permitted a more widely applicable spatial discrimination of damage costs. The regression fits for the 'central', 'low' and 'high' cases are shown in **Figure 5-2**. In each case the regression function has been forced through the origin, given that any offset on the *y*-axis would fall within the error associated with transferring damage costs.

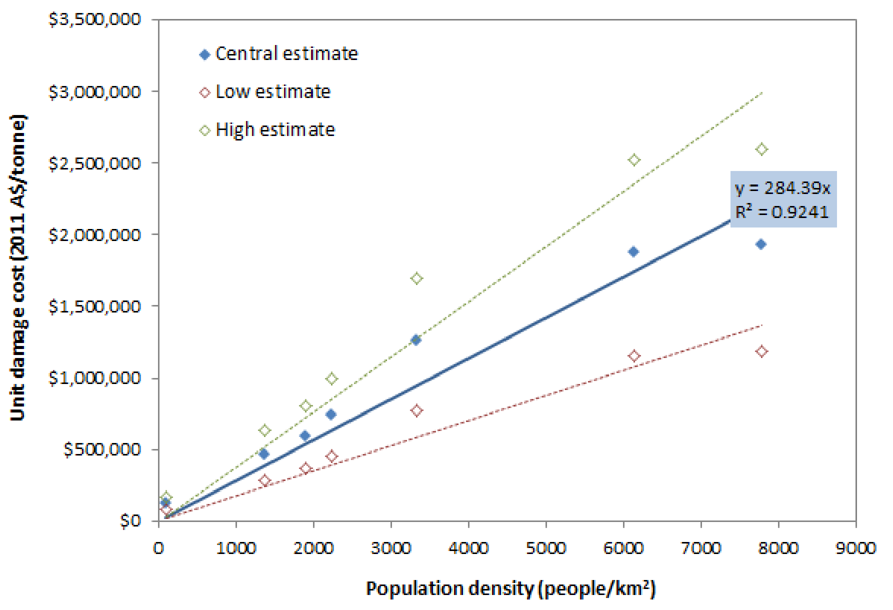

Figure 5-2: The relationship between population density (km^2) and primary $PM_{2.5}$ damage costs (A$/t$PM_{2.5}$) at 2011 prices

The regression fits resulted in the following equations:

Central value: $C_{central} = 280 \times D_{pop}$ (Equation 5)

Low value: $C_{low} = 170 \times D_{pop}$ (Equation 6)

High value: $C_{high} = 380 \times D_{pop}$ (Equation 7)

Where:

C = unit damage cost (A$ per tonne of $PM_{2.5}$ emitted at 2011 prices)

D_{pop} = population density (people/km^2)

The gradients of the equations have been rounded to two significant figures to reflect the uncertainty in the estimates.

5.2.4 Step 4: Unit damage costs by urban area

The application (by a user) of the equations derived in Step 3 in economic appraisal was not considered to be appropriate. Firstly, many users would not have detailed population density information to hand. Secondly, it involves an additional calculation step that could be a source of error. Moreover, the average population-weighted damage cost for an area increases as the level of spatial resolution in the population data increases. The effect will be greatest for a large area having a very spatially uneven distribution of population (such as NSW), whereby large empty spaces artificially dilute the impact/cost in the high-population areas.

It was therefore considered more appropriate to calculate unit damage costs for specific geographical areas using a simplified and standardised method which would allow users to easily relate the location of emissions to an approximate population-weighted exposure. The approach used was based on the ABS Significant Urban Area structure (see **Section 3.4**).

Table 5-5 to **Table 5-10** list the SUAs in each of the Australian jurisdictions. For each SUA the central estimate unit damage cost (A$ per tonne of $PM_{2.5}$ emitted) was calculated using Equation 5. These values are at 2011 prices and for population densities in 2011. It is recommended that these unit damage costs are used for economic appraisals in NSW and Australia where there is no possibility of following the full impact pathway approach.

Table 5-5: Unit damage costs by SAU (rounded to two significant figures) – NSW

SUA code	SUA name	Area (km^2)	Population	Population density (people/km^2)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
1030	Sydney	4,064	4,028,525	991	**$280,000**
1009	Central Coast	566	304,755	538	**$150,000**
1035	Wollongong	572	268,944	470	**$130,000**
1027	Port Macquarie	96	41,722	433	**$120,000**
1013	Forster - Tuncurry	50	19,501	394	**$110,000**
1023	Newcastle - Maitland	1,019	398,770	391	**$110,000**
1014	Goulburn	65	21,485	332	**$93,000**
1003	Ballina	73	23,511	320	**$90,000**
1018	Lismore	89	28,285	319	**$89,000**
1016	Griffith	56	17,900	317	**$89,000**
1033	Ulladulla	47	14,148	303	**$85,000**
1010	Cessnock	69	20,262	294	**$82,000**
1034	Wagga Wagga	192	52,043	272	**$76,000**
1025	Orange	145	36,467	252	**$71,000**
1022	Nelson Bay - Corlette	116	25,072	217	**$61,000**
1012	Dubbo	183	33,997	186	**$52,000**
1017	Kurri Kurri - Weston	91	16,198	179	**$50,000**
1015	Grafton	106	18,360	173	**$48,000**
1004	Batemans Bay	94	15,732	167	**$47,000**
1024	Nowra - Bomaderry	202	33,340	165	**$46,000**
1029	St Georges Basin - Sanctuary Point	77	12,610	164	**$46,000**
1031	Tamworth	241	38,736	161	**$45,000**
1005	Bathurst	213	32,480	152	**$43,000**
1032	Taree	187	25,421	136	**$38,000**
1001	Albury - Wodonga	628	82,083	131	**$37,000**
1011	Coffs Harbour	506	64,242	127	**$36,000**
1028	Singleton	127	16,133	127	**$36,000**
1007	Broken Hill	170	18,519	109	**$30,000**
1019	Lithgow	120	12,251	102	**$29,000**
1006	Bowral - Mittagong	422	34,861	83	**$23,000**
1002	Armidale	275	22,469	82	**$23,000**
1020	Morisset - Cooranbong	341	21,775	64	**$18,000**
1026	Parkes	235	10,939	47	**$13,000**
1021	Muswellbrook	262	11,791	45	**$13,000**
1008	Camden Haven	525	15,739	30	**$8,400**
1000	Not in any Significant Urban Area (NSW)	788,116	999,873	1.3	**$360**

Table 5-6: Unit damage costs by SAU (rounded to two significant figures) – Victoria

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
2011	Melbourne	5,679	3,847,567	677	**$190,000**
2016	Sale	46	14,259	313	**$88,000**
2020	Wangaratta	58	17,687	307	**$86,000**
2004	Bendigo	287	86,078	299	**$84,000**
2003	Ballarat	344	91,800	267	**$75,000**
2005	Colac	55	11,776	215	**$60,000**
2010	Horsham	83	15,894	191	**$54,000**
2008	Geelong	919	173,450	189	**$53,000**
2017	Shepparton - Mooroopna	249	46,503	187	**$52,000**
2006	Drysdale - Clifton Springs	65	11,699	180	**$50,000**
2012	Melton	266	47,670	179	**$50,000**
20+22	Warrnambool	183	32,381	177	**$50,000**
2019	Traralgon - Morwell	235	39,706	169	**$47,000**
2014	Moe - Newborough	105	16,675	158	**$44,000**
2018	Torquay	126	15,043	119	**$33,000**
2015	Ocean Grove - Point Lonsdale	219	22,424	103	**$29,000**
2001	Bacchus Marsh	196	17,156	87	**$24,000**
2002	Bairnsdale	155	13,239	85	**$24,000**
2013	Mildura - Wentworth	589	47,538	81	**$23,000**
2007	Echuca - Moama	351	19,308	55	**$15,000**
2009	Gisborne - Macedon	367	18,014	49	**$14,000**
2021	Warragul - Drouin	680	29,946	44	**$12,000**
2000	Not in any Significant Urban Area (Vic.)	216,296	693,578	3	**$900**

Table 5-7: Unit damage costs by SAU (rounded to two significant figures) - Queensland

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
3003	Cairns	254	133,912	527	**$150,000**
3008	Hervey Bay	93	48,678	523	**$150,000**
3006	Gold Coast - Tweed Heads	1,403	557,823	398	**$110,000**
3001	Brisbane	5,065	1,977,316	390	**$110,000**
3010	Mackay	208	77,293	371	**$100,000**
3004	Emerald	39	13,219	337	**$94,000**
3012	Mount Isa	63	20,569	328	**$92,000**
3007	Gympie	69	19,511	282	**$79,000**
3016	Townsville	696	162,291	233	**$65,000**
3002	Bundaberg	306	67,341	220	**$62,000**
3015	Toowoomba	498	105,984	213	**$60,000**
3018	Yeppoon	79	16,372	208	**$58,000**
3005	Gladstone - Tannum Sands	240	41,966	175	**$49,000**
3014	Sunshine Coast	1,633	270,771	166	**$46,000**
3011	Maryborough	171	26,215	154	**$43,000**
3013	Rockhampton	580	73,680	127	**$36,000**
3017	Warwick	159	14,609	92	**$26,000**
3009	Highfields	230	16,820	73	**$20,000**
3000	Not in any Significant Urban Area (Qld)	1,718,546	755,687	0.4	**$120**

Table 5-8: Unit damage costs by SAU (rounded to two significant figures) – South Australia

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
4001	Adelaide	2,024	1,198,467	592	**$170,000**
4006	Port Pirie	75	14,044	187	**$52,000**
4008	Whyalla	121	21,991	181	**$51,000**
4003	Murray Bridge	98	16,706	171	**$48,000**
4002	Mount Gambier	193	27,754	144	**$40,000**
4005	Port Lincoln	136	15,222	112	**$31,000**
4007	Victor Harbor - Goolwa	309	23,851	77	**$22,000**
4004	Port Augusta	249	13,657	55	**$15,000**
4000	Not in any Significant Urban Area (SA)	980,973	264,882	0.3	**$76**

Table 5-9: Unit damage costs by SAU (rounded to two significant figures) – Western Australia

SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
5009	Perth	3,367	1,670,952	496	**$140,000**
5007	Kalgoorlie - Boulder	75	30,839	411	**$110,000**
5003	Bunbury	223	65,608	295	**$83,000**
5005	Ellenbrook	105	28,802	276	**$77,000**
5002	Broome	50	12,765	255	**$71,000**
5006	Geraldton	271	35,749	132	**$37,000**
5008	Karratha	134	16,474	123	**$34,000**
5010	Port Hedland	116	13,770	118	**$33,000**
5001	Albany	297	30,656	103	**$29,000**
5004	Busselton	1,423	30,286	21	**$6,000**
5000	Not in any Significant Urban Area (WA)	2,520,513	30,654	0.01	**$3**

Table 5-10: Unit damage costs by SAU (rounded to two significant figures) - Other

State	SUA code	SUA name	Area (km^2)	Population	Population density ($people/km^2$)	Damage cost/tonne of $PM_{2.5}$ (A$, 2011)
Tasmania	6001	Burnie - Wynyard	131	29,050	223	**$62,000**
	6004	Launceston	435	82,222	189	**$53,000**
	6003	Hobart	1,213	200,498	165	**$46,000**
	6005	Ulverstone	130	14,110	108	**$30,000**
	6002	Devonport	290	26,871	93	**$26,000**
	6000	Not in any Significant Urban Area (Tas.)	65,819	142,598	2	**$610**
Northern territory	7002	Darwin	295	106,257	361	**$100,000**
	7001	Alice Springs	328	25,187	77	**$22,000**
	7000	Not in any Significant Urban Area (NT)	1,347,577	80,504	0.06	**$17**
ACT	8001	Canberra - Queanbeyan	482	391,643	812	**$230,000**
	8000	Not in any Significant Urban Area (ACT)	1,914	1,622	0.85	**$240**
Other	9000	Not in any Significant Urban Area (OT)	218	3,029	14	**$3,900**

5.3 Comparison with previous Australian studies

As noted earlier, unit damage cost values for Australian locations are available from several previous studies (see **Table 2-3**). Unit damage cost values obtained using the new method are compared with those from the more recent of these studies in **Table 5-11**. It should be noted that there are differences in the methodologies, definitions and metrics used, and the comparisons are unlikely to be 'like with like'. For example, the previous studies relate to PM_{10}, whereas the new method relates to $PM_{2.5}$, and the definitions of geographical areas are probably not consistent. For the purpose of comparison, the new average unit damage costs were determined using the information resented in the previous Section. The unit damage costs from the previous studies were inflated to 2011 prices.

Table 5-11: Comparison between unit damage costs produce using new method and those from previous studies (values rounded to 2 significant figures)

Study	Details	Unit damage cost (A$/tonne)	
		Original study[(a)]	New method
CIE (2005)	Sydney	$300,000	$280,000
DEC (2005)	Sydney	$280,000	$280,000
	Hunter	$75,000	$13,000-$82,000[(b)]
	Illawarra	$56,000	$130,000[(c)]
BDA (2006)	Sydney	$160,000	$280,000
DIT (2010)	State capital cities	$250,000	$200,000[(d)]
	Rest of Australia	$59,000	$41,000[(d)]
AECOM (2011)	NSW	$75,000	$190,000[(d)]

(a) Adjusted to 2011 prices, rounded to two significant figures
(b) Range for towns in the Hunter Valley.
(c) Value for Wollongong only.
(d) With weighting by SUA, rounded to two significant figures

It can be seen that the new method results in unit damage costs for Sydney and the State capitals that are very similar to those used in the previous studies. However, there may also be differences in geographical definitions (*e.g.* whether or not 'Sydney' includes the GMR). Definitive values for the Hunter Valley and the Illawarra could not be determined from the SUA data, as the proportions of the population in urban and non-urban areas were not readily available. The value determined for NSW was much higher than that used previously by **AECOM (2011)**, possibly due to the use of a finer spatial resolution in the population data in the current study.

5.4 Guidance on the calculation of damage costs in economic appraisals

5.4.1 Overview

This Report has provided a consistent set of damage costs for the analysis of air quality impacts due to changes in primary $PM_{2.5}$ emissions in SUAs in NSW. The methodology has been derived from UK values that relate to the marginal external costs due to each additional tonne of $PM_{2.5}$

emitted **in each year** – or conversely the benefits of reducing $PM_{2.5}$ emissions by one tonne. The Australian damage costs can be used to value the air quality impacts of policies or projects based on the annual change in $PM_{2.5}$ emissions (in tonnes), as described below. The use of the damage costs is not, however, a replacement for detailed modelling and analysis but represents the most reliable option when resources are limited.

5.4.2 Calculation of damage costs in a single SUA

Urban centres with more than 10,000 people (i.e. SUAs) should be included in economic appraisals. The unit damage costs should not be applied to smaller area categories. The unit damage costs define the cost associated with an emission of one tonne of primary $PM_{2.5}$ at 2011 prices, and for population densities in 2011.

In any economic appraisal a base year needs to be selected. Where 2011 is the base year of the appraisal, all emissions in future years need to be adjusted to express them in this price year. This firstly involves an adjustment for the forecast change in population density, followed by the application of an 'uplift' to adjust VOLY estimates to future years. The total benefits are then discounted back to the base year to reflect the fact that current benefits have greater value in the present than future benefits. In other words, all effects in future years have to be expressed in consistent terms (*e.g.* by calculating a net present value). These steps are described below.

5.4.2.1 Population adjustment

The unit damage costs for 2011 are based on population density statistics in 2011. In many locations it is likely that there will be a change (typically a growth) in population in future years, and this will affect population density. For the purpose of appraisal it can be assumed that the change in population density in an SUA is proportional to the change in population, and that the unit damage costs in future years can be adjusted linearly. Local data on forecast changes in population in an SUA should be used where these are available.

Table 5-12 shows a worked example of how the unit damage costs for future years are adjusted, in this case based on an annual increase in population density of 1.5%. The scenario involves a five-year appraisal with an initial damage cost for PM in a 2011 baseline year of A$20,000/tonne (equating to a population density of around 71.4 people per km^2).

Table 5-12: Unit damage costs in a single SUA adjusted for population change

	2011	2012	2013	2014	2015
Population density (people/km^2)	71.4	72.5	73.6	74.7	75.8
Unit damage cost (A$/tonne)	20,000	20,300	20,605	20,914	21,227

5.4.2.2 Uplift

The unit damage costs are then adjusted for future years to reflect the assumption that WTP for health will rise continue to rise in line with economic growth during the years of the economic analysis. It is recommended that a long-term growth in GDP per capita of 2% is used for consistency with the UK appraisal method, although it is recognised that there has been a tendency for uplifts not to be included in Australian studies. The uplift assumes that the real cost of each element of accident costs (such as the cost of medical treatment) will rise in line with increases in economic growth. A worked example using an uplift value of 2% is shown in **Table 5-13**. It should be noted that the uplift is compounded in each year.

Table 5-13: Uplift example for a single SUA

	2011	2012	2013	2014	2015
Unit damage cost from Table 5-12 (A$/tonne)	20,000	20,300	20,605	20,914	21,227
Uplift	1.000	1.020	1.040	1.061	1.082
Uplifted damage cost (A$/tonne)	20,000	20,706	21,437	22,194	22,977

5.4.2.3 Discount

The unit damage cost values for emissions in future years then need to be discounted back to the present year to reflect the net present value of air pollution impacts. The NSW Treasury has suggested an appropriate real discount rate for Australia of 7% **(NSW Treasury, 2007).**

Discounting is undertaken in a two-step process. Firstly, a discount factor for each year of the appraisal is generated. The discount factor can be calculated by the following equation.

$$\textit{Discount factor} = 1/(1.07)^t \qquad \text{(Equation 8)}$$

Where: 1.07 = 7% discount

t = the number of years into the future that value is from the base year

In the second step, the present value of air pollution impacts is calculated by multiplying the undiscounted value of impacts for each year by the discount factor.

$$\textit{Present value} = \textit{Undiscounted damage cost} \times \textit{Discount factor for year} \qquad \text{(Equation 9)}$$

Table 5-14 provides an example using the figures from **Table 5-13**.

Table 5-14: Example of applying discount to damage cost values

	2011	2012	2013	2014	2015
Year	0	1	2	3	4
Discount factor	1.000	0.935	0.873	0.816	0.763
Unit damage cost from Table 5-13 (A$/tonne)	20,000	20,706	21,437	22,194	22,977
Net present value unit damage cost (A$/tonne)	20,000	19,351	18,724	18,117	17,529
Net present value unit damage cost (A$/tonne) (rounded to 2 significant figures	**20,000**	**19,000**	**19,000**	**18,000**	**18,000**

5.4.2.4 Calculating the values of impacts

These uplift- and discount-adjusted damage costs can now be used to estimate the value of changes in $PM_{2.5}$ emissions for each year of the appraisal period. This calculation simply involves multiplying the expected change in emissions in tonnes by the adjusted damage cost figure for each year (**Table 5-15**). This assumes that all changes are reductions. Any increase in emissions would have to be taken into account by a change of sign (*i.e.* a negative reduction).

Table 5-15: Example of total costs of changes in $PM_{2.5}$ emissions over 5 years

	2011	2012	2013	2014	2015	*TOTAL*
$PM_{2.5}$ emission change[(a)] (tonnes)	100	100	50	50	50	*350*
Net present value damage cost (A$/tonne)	20,000	19,000	19,000	18,000	18,000	-
Value (A$)	**2,000,000**	**1,900,000**	**950,000**	**900,000**	**900,000**	***6,650,000***

(a) It is assumed here that all changes are reductions.

5.4.2.5 Future base year

If an appraisal for an SUA is being undertaken for a future base year (*e.g.* 2014) then the unit damage costs need to be expressed in the prices of that year. Again, it is recommended that a value of 2% per annum is used to adjust the unit damage costs for 2011 for each future year of the programme being assessed. For example, if a policy appraisal scenario involves a damage cost for $PM_{2.5}$ of \$20,000 per tonne (in 2011 prices) and the required base year is 2014, the damage cost for the base year would be \$21,224 per tonne (A\$20,000 x $(1.02)^3$ = A\$21,224). A further adjustment should also be made to allow for any changes in the population density in the SUA.

5.4.3 Calculation of damage costs for multiple SUAs

Where larger geographical areas (containing a mixture of SUAs) are being appraised the change in $PM_{2.5}$ emissions – as described above - should be determined separately for each affected SUA. For each SUA in turn the change in emissions is then multiplied by the unit damage cost for the SUA to determine the actual change in damage cost in the SUA. The resulting changes in damage cost are then summated over all affected SUAs.

5.4.4 Other considerations

Emissions from non-transport sources will lead to a different population-weighted exposure compared with road transport. This is reflected in the Defra damage costs, which assign much lower levels to industry and electricity generation (as these are mostly emitted from tall stacks in rural areas). Population-weighted exposure from industrial stack emissions is not analysed separately in the UK for different areas. Further modelling work would be needed to address this issue accurately (both in the UK and Australia). It is therefore highlighted that the application of the new damage costs to industrial stack emissions will over-estimate population-weighted exposure, and it is recommended that industrial emissions are considered separately where industry dominates an area.

5.5 Assumptions and uncertainties

The following assumptions and uncertainties are inherent in the transferred damage costs:

- The academic community has not yet been able to determine whether the C-R functions derived in one area are transferable to another area. However, because so much data are needed to derive C-R functions, the US and EU have both used meta-analysis of C-R function from both continents to calculate the risks of air pollution exposure. Defra included studies from different parts of the UK and, for mortality from chronic exposure, the US - *i.e.* **Pope *et al.* (2002)**. There is therefore an (unknown) degree of uncertainty relating to the application of the C-R functions in Australia.

- The damage costs proposed here use the UK rate of all-cause mortality, as well as the UK rates of respiratory and cardio-vascular hospital admissions. There are likely to be different mortality and morbidity rates in Australian due to differences in health status, age, life expectancy, as well as other factors (incidence of smoking, *etc.*).

- External costs of air pollution vary according to a variety of environmental factors, including overall levels of pollution, geographic location of emission sources, and meteorology. Defra modelled the impact of air pollution in the UK, with UK source locations, population locations, geographic features and meteorology. Since conditions are different in Australia, additional uncertainty is incorporated into the damage costs. It is not possible to quantify the difference without modelling Australian air pollution.

- The Defra damage costs exclude several key effects, as quantification and valuation of these factors was not possible or was highly uncertain. These should be highlighted when presenting valuation results where appropriate. The key effects that have not been included are:

    - Effects on ecosystems (through acidification, eutrophication, etc.).
    - Impacts of trans-boundary pollution.
    - Effects on cultural or historic buildings from air pollution.
    - Potential additional morbidity from acute exposure to PM.
    - Potential mortality effects in children from acute exposure to PM.
    - Potential morbidity effects from chronic (long-term) exposure to PM or other pollutants.
    - Effects of exposure to ozone, including both health impacts and effects on materials.
    - Change in visibility (visual range).
    - Macroeconomic effects of reduced crop yield and damage to building materials.

5.6 Recommendations for a future valuation framework

The UK has adopted a two-level strategy in which either the impact pathway or damage cost approach is recommended based on the type of application and the anticipated effects of the changes proposed. A similar type of two-level approach would be useful for future consideration in Australia.

The proposed two-tier valuation framework is shown in **Figure 5-3**. The aim is to have a method that is fit for purpose and country-specific, whilst being flexible enough to allow the level of analysis to the match likely resources available.

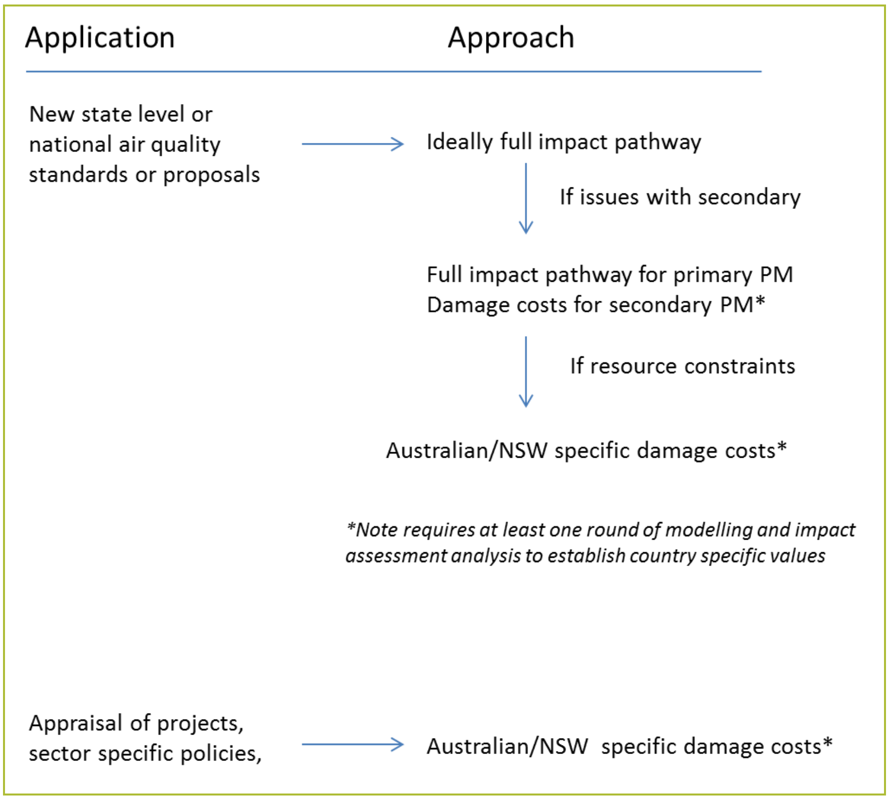

Figure 5-3: Two-level valuation framework

The framework addresses two types of application:

- Firstly, for new state or national air quality standards - which are associated with high costs - a full impact pathway approach is required. Whilst estimating the health benefits of proposed standards is possible (the current and proposed ambient air quality are sufficient), determining the costs of compliance requires emission inventories and air quality modelling.

- Secondly, for other applications (*e.g*. programmes to reduce emissions) the damage costs proposed here should be used in combination with emission modelling results. These damage costs can be improved in the future by applying the impact pathway approach to Australia (or NSW), and with a separation by region/area.

The recommended future approach will therefore involve some PM modelling, making sure that damage costs are area-specific and source-specific. For example, existing models could be run to establish marginal pollution increases in different areas (*i.e.* with different population density or with very different geographical characteristics such as 'coastal' or 'interior'), and ideally for different types of emission source (*e.g.* 'transport' or 'high stack').

This would ideally be complemented with some secondary PM modelling (for SO_2 and NO_x emissions), to provide Australian/NSW-specific damage costs for these secondary pollutants. To fully understand the relationship between source and receptors, this secondary PM analysis would ideally be undertaken for rural, urban and capital city locations.

To make the damage cost analysis as flexible as possible – and to allow amendments and changes in the future – the steps in the damage cost calculation need to be transparent. In other words, there should be sub-level information on:

- The population-weighted exposure per tonne of pollutant.
- The health impacts (physical cases) – split by health endpoint – per tonne of pollutant (*e.g.* respiratory hospital admissions per tonne).
- The economic value – split by health endpoint – per tonne of pollutant.

This allows the values to be updated in the future as new health endpoints emerge or as primary valuation studies provide new numbers for the specific Australian/NSW context.

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APPENDIX A

Glossary of terms and abbreviations

Table A1: Terms and abbreviations

AAQ NEPM	Ambient Air Quality National Environmental Protection Measure
ASCC	Australian Safety and Compensation Council
ALPHA	Atmospheric Long-range Pollution Health/environment Assessment
ANSTO	Australian Nuclear Science and Technology Organisation
AUD	Australian Dollar
CAFE	(EU) Clean Air for Europe (programme)
CAIR	(US) Clean Air Interstate Rule
CBA	Cost--benefit analysis
CO	carbon monoxide
COMEAP	Committee on the Medical Effects of Air Pollutants
CPI	Consumer Price Index
C-R	Concentration response
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CTM	Chemical Transport Model
Defra	(UK) Department of Environment, Food and Rural Affairs
DERM	(Queensland)(South Australia) Department of Environment and Resource
DEWHA	Department of Environment, Water, Heritage and the Arts
EAP	Environmental Action Programme
EC	European Commission
EDMS	Emissions Data Management System
EIA	Environment impact
EIS	Environmental Impact Statement
EMEP	European Monitoring and Evaluation Programme
EPA	(NSW) Environment Protection Agency
EPAV	Environmental Protection Agency Victoria
ESI	Electrical Supply Industry
EU	European Union
GAINS	Greenhouse Gas and Air Pollution Interactions and Synergies
GBD	Global Burden of Diseases
GIS	Geographical Information Systems
GMR	(Sydney) Greater Metropolitan Region
HNO_3	Gaseous nitric acid
H_2SO_4	Sulphuric acid
IA	Impact Assessment
IGCB	(UK) Interdepartmental Group on Costs and Benefits
LGA	Local Government Area
NAAQS	(US) National Ambient Air Quality Standard
NAEI	(UK) National Atmospheric Emissions Inventory
NEPC	National Environment Protection Council
NEPM	National Environment Protection Measure
NES	National Environmental Standards
NESHAP	National Emission Standards for Hazardous Air Pollutants
NH_3	Ammonia
NH_4^+	Ammonium
$(NH_4)_2SO_4$	Ammonium sulphate

NO_x	Oxides of nitrogen
NO_3^-	Nitrate
NPI	National Pollutant Index
NSW	New South Wales
OCM	Organic carbon mass
O_3	Ozone
PAHs	Polycyclic aromatic hydrocarbons
PM	Airborne particulate matter
PM_{10}	Airborne particulate matter with an aerodynamic diameter of less than 10
$PM_{2.5}$	Airborne particulate matter with an aerodynamic diameter of less than 2.5
RAINS	Regional Air pollution Information and Simulation
RIA	Regulatory Impact Analysis
REF	Review of Environmental Factors
SEPP	State Environmental Planning Policies
SEE	Statement of Environmental Effects
SEQR	South east Queensland region
SOA	Secondary organic aerosols
SO_2	Sulphur dioxide
SO_4^{2-}	Acidic sulphate
SSD	State-Significant Development
SSI	State-Significant Infrastructure
SUA	Significant Urban Area
TAPM	The Air Pollution Model
UK	United Kingdom
UN	United Nations
US	United States
VOCs	Volatile Organic Compounds
VOLY	Value of a Life Year
VSL	Value of a Statistical Life
WHO	World Health Organisation
WTP	Willingness to Pay
YOLL	Years of Life Lost

APPENDIX B

Overseas valuation studies

B1 European Union - CAFE programme

B1.1 Background

The European Commission's (EC) Sixth Environmental Action Programme (6^{th} EAP) aimed to develop long-term, strategic and integrated policy advice for '*achieving levels of air quality that do not give rise to significant negative impacts on and risks to human health and the environment'*. This also included '*no exceedance of critical loads and levels for acidification or eutrophication'*. The Programme identified the need for a Thematic Strategy on Air Pollution which considered the economic, social and environmental dimensions of policies designed to meet these objectives. In response, the EC launched the Clean Air for Europe (CAFE) Programme in 2001 – a knowledge-based approach for technical/scientific analyses and policy development, which led to the adoption of the Thematic Strategy on Air Pollution in 2005.

Consistent with all European regulatory policy proposals, the Thematic Strategy was subject to an Impact Assessment (IA). These IAs consider the likely economic, social and environmental impacts of different options. A cost-benefit analysis was undertaken as part of the CAFE programme to support the impact assessment of the Thematic Strategy. The approach was developed through the series of EC-funded research projects within the ExternE framework (**EC, 1995, 1999)**. The method was published as a series of reports **(Holland *et al.*, 2005a, b**; **Hurley *et al*., 2005b)**.

B1.2 Approach used

The objectives of the Clean Air for Europe (CAFE) programme were to establish the capacity to assess the costs and benefits of air pollution policies and to conduct a CBA on the effects of those policies. The CAFE programme used the impact pathway approach for valuing the health impacts of air pollution (environmental endpoints such as crop damage were also assessed) **(AEA Technology Environment, 2005)**. The CAFE programme used both the impact pathway and damage cost approaches for valuing health impacts from air pollution. Damage costs were generated using an impact pathway approach.

B1.3 Pollutants considered

The analysis assessed emissions of sulfur dioxide (SO_2), oxides of nitrogen (NO and NO_2, together referred to as NO_x), fine particles ($PM_{2.5}$), and volatile organic compounds (VOCs). Both primary and secondary PM were included, the latter via emissions of gaseous precursors (SO_2, NO_x and ammonia (NH_3)). Ozone (O_3) was also taken into account.

B1.4 Emissions and air quality modelling

The analyses, following the impact pathway approach, used two linked models. The first analysis was undertaken using the RAINS/GAINS[h] model **(Amann, 2008)**, which addresses the formation and dispersion of pollutants in the atmosphere. This model was used to assess the health impacts of $PM_{2.5}$ and ground-level O_3, as well as a number of ecosystem impacts from acidification and eutrophication. It was complemented with the use of the ALPHA[i] model, which covers a wider set of health and environmental impacts, and also includes monetary valuation.

[h] Regional Air pollution Information and Simulation/ Greenhouse Gas and Air Pollution Interactions and Synergies

[i] Atmospheric Long-range Pollution Health/environment Assessment

The detailed impact analysis used two models, as outlined below, working within a Geographical Information System (GIS) framework.

The GAINS and ALPHA models are complementary as they focus on different aspects of the impact pathway approach. Both the RAINS/GAINS model **(SERI, 2004, also 2007, 2009)**[j] and the CAFE cost-benefit analysis method **(Krupnick *et al.*, 2004)** and the ALPHA model **(Holland *et al.*, 2008)** have been extensively peer reviewed, and applied previously in EC policy impact assessment **(CEC, 2005b)**.

The RAINS/GAINS model explores cost-effective strategies to reduce emissions of greenhouse gases (GHGs) and other air pollutants. The model produces emission scenarios for any projection of future economic activity, abatement potential and cost, as well as interactions in abatement between various pollutants **(Amman *et al.*, 2011)**. It includes detailed atmospheric chemistry and transport models which allow the atmospheric modelling of emissions and the estimation of pollution concentrations, including both primary and secondary pollutants. These concentrations are combined with other necessary data such as critical loads and levels, relative risk factors, population, ecosystems areas, etc. This then allows the estimation of the effects on human health from exposure to fine particles and ground-level ozone, and damage to vegetation via excess deposition of acidifying and eutrophying compounds. The model also has a detailed abatement module which allows the analysis of abatement control to reduce these impacts, using a cost-effectiveness framework that can address multiple targets of health and ecosystem protection, as well as reducing GHG emissions. Thereby, GAINS allows for a comprehensive and combined analysis of air pollution and climate change mitigation strategies, which reveals important synergies and trade-offs between these policy areas.

The ALPHA model was developed to provide a detailed quantification of the benefits of pollution controls in Europe. It has been used extensively for European policy assessment, including work on the National Emission Ceilings Directive and the UN/ECE Gothenburg Protocol under the Convention on Long Range Transboundary Air Pollution (CLRTAP), directives on air quality including the CAFE Directive, directives on fuel quality and directives on emission limits for industry. The model takes dispersion data from the EMEP or GAINS models and provides a detailed quantification of effects on health, including various morbidity impacts (on chronic bronchitis, hospital admissions, etc.) and mortality, and effects on building materials and crops. Extension of the model for the quantification of effects on ecosystem services is currently under consideration. Analysis then continues to monetisation of quantified effects, permitting final results to be used in cost-benefit analysis using information on abatement costs from models such as GAINS. The model can be applied at any desired geographic scale and over any area of interest provided that appropriate pollution and population data are available.

The CAFE programme also provided damage costs per tonne of pollutant ($PM_{2.5}$, SO_2, NO_x, NH_3 and VOCs), accounting for variation in the location of emission by providing estimates for each country in the EU-25[k]. The impact pathway approach was used to estimate the damage costs.

Dispersion modelling was based on the EMEP model[l], with a 50 x 50 km resolution and updated chemistry and meteorology. The modelling was carried out for a series of scenarios in which emissions for the baseline 2010 scenario were changed individually by country and pollutant. The work was described by **Holland *et al.* (2005)**.

[j] See http://gains.iiasa.ac.at/index.php/documentation-of-model-methodology/model-reviews/gains-review-2009.

[k] Austria; Belgium; Bulgaria; Cyprus; Czech Republic; Denmark; Estonia; Finland; France; Germany; Greece; Hungary; Ireland; Italy; Lithuania; Luxembourg; Netherlands; Poland; Portugal; Romania; Slovakia; Slovenia; Spain; Sweden; United Kingdom

[l] See https://wiki.met.no/emep/page1/unimodopensource2011

The values used the CAFE approach and also took account of variation in the method used to value mortality, reflecting the use of the median and mean estimates of the VOLY and the VSL, This resulted in four alternative estimates, rather than a central value. The values are shown in **Table B1**.

Table B1: Average damages per tonne of emission of $PM_{2.5}$ for the EU25 (excluding Cyprus) and surrounding areas under different sets of assumptions (2010 prices)

VOLY median	VSL median	VOLY mean	VSL mean
A$53,823	A$82,805	A$105,576	A$155,259

Converted using approximate total change in cost from year 2000 of 34.4%, over 10 years, at an average annual inflation rate of 3.0%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

B1.5 Population and stock data

The study used United Nations (UN) population data, with additional factors for sensitivity of the population from Eurostat. The main sensitivities were a set of additional health functions, where there are primary studies that show relationships.

B1.6 Health impacts

B1.6.1 Mortality

An earlier expert group was convened by the World Health Organization (WHO) under the CAFE Programme[m], which recommended that Health Impact Assessment should be performed against exposure to O_3 and fine particles, considering acute effects on mortality – as reflected by premature mortality (O_3) and mortality from chronic exposure (PM).

The health impact assessment was based on methods and quantification steps developed over a number of years. The methods were subject to extensive review **(***e.g.* **Krupnick *et al.*, 2005)** and were found to be fit for purpose and reflective of the current state of science[n]. In line with WHO advice, all particles were treated, irrespective of source and chemical composition, as equally harmful. The key outputs were reported as the cumulative Years of Life Lost (YOLL) from PM pollution and the additional cases of premature mortality from O_3 pollution. For mortality due to acute exposure to O_3, the analysis quantified the number of 'premature deaths' (deaths brought forward)[o].

[m] The recommendations of the WHO-CLRTAP Task Force on Health (TFH) (http://www.unece.org/env/documents) and the WHO 'Systematic Review of Health Aspects of Air Quality in Europe' (http://www.euro.who.int/document/e79097.pdf) were key to the development of quantification methods for assessing health impacts of air pollution, the WHO-sponsored meta-analyses of the acute effects of PM and ozone based on studies in Europe (http://www.euro.who.int/document/e82792.pdf). The process also drew on the answers to follow-up questions (http://www.euro.who.int/document/e82790.pdf) of the CAFE Steering Group.

[n] The health impact assessment has been further substantiated by recent epidemiological research, *e.g.* by **Smith *et al.* (2009)**, a paper that also demonstrates links between climate and air quality policies), **Pope *et al.* (2009a, b)**, and others.

[o] This wording signifies that many people whose deaths are brought forward by acute exposure to ozone in particular have serious pre-existing cardio-respiratory disease, and so in at least some of these cases the actual loss of life is likely to be small – the death might have occurred within the same year and, for some, may only be brought forward by a few days.

B1.6.2 Morbidity

The ALPHA model was used to assess other non-fatal health effects (morbidity). This model also assesses mortality and morbidity impacts in monetary terms. A summary of the health impacts covered is provided in **Table B2**.

Table B2: List of health impacts from $PM_{2.5}$ (annual average) quantified

Impact / population group	Population
Mortality from chronic exposure as life years lost or premature deaths	Over 30 years
Infant mortality	1 month to 1 year
Chronic bronchitis	Over 27 years
Respiratory hospital admissions	All ages
Cardiac hospital admissions	All ages
Restricted activity days	15 to 64 years
Respiratory medication use	5 to 14 years
Respiratory medication use	Over 20 years
Lower respiratory symptom days	5 to 14 years
Lower respiratory symptom days	Over 15 years

B1.7 Non-health impacts

The study used the ALPHA model to quantify and monetise impacts on building materials and crops, focusing on the two major categories of impact in Europe: crop losses from ozone exposure and damage to building materials from acidic deposition. The GAINS model was used to assess the impacts of air pollutant deposition on ecosystems. The analysis considered the area of forests and ecosystems that exceeded 'critical loads' for acidification and nitrogen deposition.

B1.8 Estimation of health impacts

The analysis of mortality from chronic exposure to PM pollution, following WHO guidance[p], used the central estimate of a 6% increase in mortality hazard rates per 10 µg/m^3 $PM_{2.5}$ from the US studies by **Pope *et al.* (1997, 2002)**. This was implemented for anthropogenic PM, with no threshold.

Consistent with WHO guidance, and a wider emerging consensus in favour of using life table methods, the analysis expressed health impacts in terms of YOLL from air pollution.

[p] WHO is involved in reviewing health impact data for both CLRTAP and the CAFE programme. As part of the latter, the recommendations of the WHO-CLRTAP Task Force on Health (TFH) (http://www.unece.org/env/documents) and the WHO 'Systematic Review of Health Aspects of Air Quality in Europe' (http://www.euro.who.int/document/e79097.pdf) were key to the development of quantification methods for assessing health impacts of air pollution, the WHO-sponsored meta-analyses of the acute effects of PM and ozone based on studies in Europe (http://www.euro.who.int/document/e82792.pdf), and also the process drew on the answers to follow-up questions (http://www.euro.who.int/document/e82790.pdf) asked by the CAFE Steering Group.

In addition, the analysis also included estimates of the number of deaths per year attributable to long-term exposure to ambient $PM_{2.5}$[q]. The approach estimated attributable deaths using a 'static' approach (without life tables), whereby the annual death rate is multiplied by a PM risk factor. This method is approximate, and is considered to over-estimate the true attributable fraction to some extent. Consequently, mortality effects of long-term exposure to PM were expressed as both YOLL and attributable cases of premature mortality, both being relevant for monetary valuation.

CAFE assumed no lag between exposure and effect, using a function based on an annual pollution pulse for valuation. This is consistent with recent guidance given by the WHO to the CAFE process, though it seems implausible that all health effects are effectively immediate.

The method used to assess morbidity was based on the CAFE CBA methodology **(Holland *et al.*, 2005a& b; Hurley *et al.*, 2005)** and response functions developed as part of the EC CAFE programme[r].

For PM and O_3 morbidity, impact functions were used to assess the health effects of acute exposures (from observation of response to day-to-day variations in ambient PM) and long-term (chronic) exposures.

B1.9 Valuation

There is a debate in Europe concerning the correct approach for the valuation of mortality risks relating to air pollution. These can be valued using a long-established metric, the Value of a Statistical Life (VSL), but changes in life expectancy can also be valued using the Value of a Life Year (VOLY). Both approaches are used in the literature and both have strengths and weaknesses.

In some regards the estimates of the VOLY are uncertain, (especially in relation to age specific VOLYs). The quantification of premature mortality benefits in terms of 'attributable deaths' and values using a VSL deaths also has uncertainties. 'Attributable deaths' can only relate to a specified time period, the net difference in deaths (when comparing two populations of the same size with higher and lower pollution) is zero since everyone in both populations will die at some point.

B1.9.1 Mortality

The CAFE Programme used both VSL and VOLY for mortality valuation. The analysis was able to take advantage of research under the EC NewExt Project[s]. There was some debate as to whether it is appropriate to take the mean or median values from the NewExt analysis of VSL and VOLY. Consistent with the external peer review guidance, the analysis used both VSL and VOLY approaches, with mean and median values, which gives four alternatives on valuation (**Table B3**).

[q] Estimates of attributable deaths have their own methodological problems. However, numbers of premature deaths appear to be easy to understand, and so are often made in HIAs of air pollution and health.

[r] The methodology was the subject of intense consultation in 2003 and 2004 with stakeholders from the European Union Member States, academic institutes, environment agencies, industry and non-governmental organisations. It was also subject to formal peer review by senior experts in the USA and Europe.

[s] NewExt (2004) New Elements for the Assessment of External Costs from Energy Technologies. Funded under the EC 5th Framework Programme (1998 – 2002), Thematic programme: Energy, Environment and Sustainable Development, Part B: Energy; Generic Activities: 8.1.3. Externalities ENG1-CT2000-00129.

Table B3: Values for use in CAFE cost--benefit analysis: effects of chronic exposure on mortality (year 2010 prices)

	VSL	VOLY
Median (NewExt)	A$2,028,722	A$108,614
Mean (NewExt)	A$4,140,249	A$248,415

Converted using approximate total change in cost from year 2000 of 34.4%, over 10 years, at an average annual inflation rate of 3.0%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

The actual difference in mortality damage using the VOLY and VSL methods is not as great as the above table might suggest. Much of the difference is cancelled out by the difference between the number of premature deaths quantified and with the number of life years lost, and there is extensive overlap in the ranges.

For morbidity impacts, a literature review assessed the most appropriate values. Note that for mortality due to acute exposure to O_3, the analysis quantifies the number of 'premature deaths' (deaths brought forward) and these cases are valued using a VOLY approach, assuming that on average, each premature death leads to the loss of 12 months of life.

B1.9.2 Morbidity

For PM morbidity a set of functions was used based on studies of the effects of acute exposure as well as chronic exposure. A similar approach was also adopted for ozone and morbidity. These were based on full WTP values. The same monetary values for mortality risk and morbidity are used across all European countries.

B1.10 Analysis of the health effect values

Given the large number of health endpoints in the CAFE programme (environmental endpoints were also assessed), it is useful to query which are important in the overall health effect values. Firstly, the values are dominated by health effects. Although material and crop damage are assessed in CAFE, these account for only around 1% of the total values. Secondly, for the health values only a few health endpoints are important. This can be seen in **Figure B1** below, which shows the breakdown of damages for PM by health endpoint.

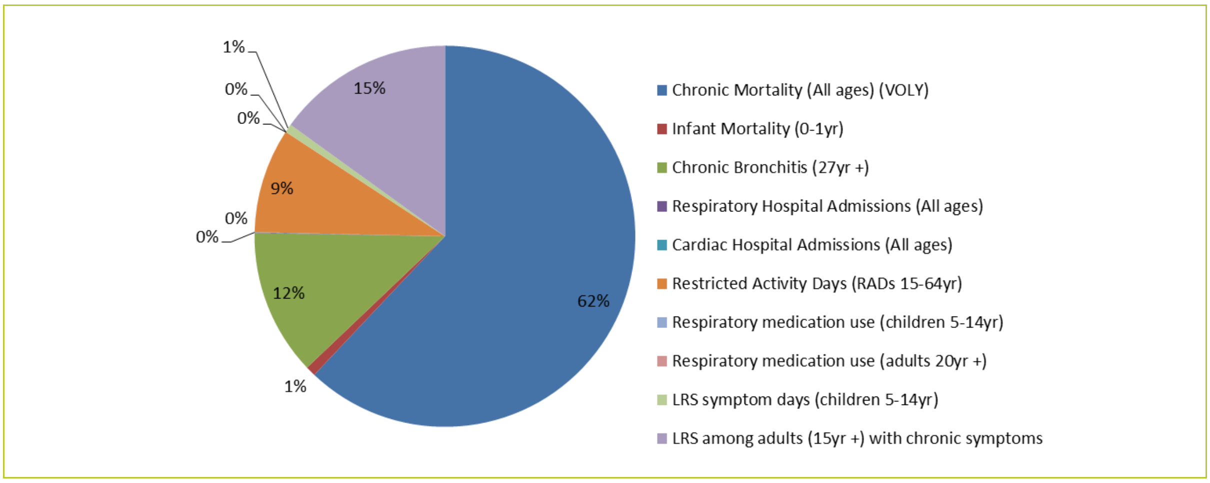

Figure B1: Percentage of PM-related health costs by health endpoint for the CAFE analysis.

In summary, around 60% of the costs are due to mortality from chronic exposure. For morbidity, three endpoints dominate the analysis: chronic bronchitis (12%), restricted activity days (9%) and low respiratory symptoms amongst adults (15%). All other endpoints are essentially insignificant (cumulatively they account for only 4% of the costs).

B1.11 Damage costs

As well as the impact pathway, the CAFE work provided damage costs per tonne of pollutant ($PM_{2.5}$, SO_2, NO_x, NH_3 and VOCs), accounting for variation in the location of emission by providing estimates for each country in the EU-25[t].

Dispersion modelling was based on the EMEP model[u], with a 50 x 50 km resolution and updated chemistry and meteorology. The modelling was carried out for a series of scenarios in which emissions for the baseline 2010 scenario were changed individually by country and pollutant. The work was described by **Holland *et al.* (2005)**.

The values used the CAFE approach and also took account of variation in the method used to value mortality, reflecting the use of the median and mean estimates of the VOLY from NewExt (2004) (€50,000 and €120,000 respectively ($84,000 and $201,600 AUD)). and the median and mean estimates of the VSL, also from NewExt (€980,000 and €2,000,000 respectively ($1,646,400 and $3,360,000 AUD)). This resulted in four alternative estimates, rather than a central value.

The EU25 average values are shown in **Table B4**. It should be noted that effects caused by secondary PM are not assigned to the PM_{10} damage costs per tonne, but to damage costs per tonne of the primary pollutant from which they are formed (*e.g.* SO_2 for sulfate aerosol, NO_x and NH_3). Values vary between Member States, reflecting trans-boundary pollution effects, as shown in **Tables B5 to B7** (Converted to AUD using approximate average market price over year 2000 of $1.54 to €).

SOMO 35 and SOMO 0 are the sum of Ozone Means Over 35 and 0 ppb, respectively. They are indicators for health impact assessment recommended by WHO and are defined as the yearly sum of the daily maximum of 8-hour running averages over 35 and 0 ppb, respectively. For each day the maximum of the running 8-hours average for O_3 is selected and the values over 35 and 0 ppb are summed over the whole year.

Comparing the effects of urban and rural release of NH_3, SO_2 and VOCs by would make little difference to the results, given that the effects of these pollutants are mediated here through formation of secondary aerosols and O_3 whose formation in the atmosphere requires time.

No urban uplifts (i.e. percentages applied each year to reflect the assumption that values will increase in line with long term economic growth)were included for primary PM.

A revision of these numbers (for 2005) was prepared for the **European Environment Agency (EEA, 2011)**. The EEA report also provides updates of the values for each Member State, finding similar differences across countries as outlined above.

[t] Austria; Belgium; Bulgaria; Cyprus; Czech Republic; Denmark; Estonia; Finland; France; Germany; Greece; Hungary; Ireland; Italy; Lithuania; Luxembourg; Netherlands; Poland; Portugal; Romania; Slovakia; Slovenia; Spain; Sweden; United Kingdom

[u] See https://wiki.met.no/emep/page1/unimodopensource2011

Table B4: Average damages per tonne of emission of NH_3, $PM_{2.5}$, SO_2 and VOCs for the EU25 (excluding Cyprus) and surrounding areas under different sets of assumptions (Year 2000 prices)

Damage cost metric	Method 1	Method 2	Method 3	Method 4
Damage Cost Assumptions				
PM mortality	VOLY median	VSL median	VOLY mean	VSL mean
O_3 mortality	VOLY median	VOLY median	VOLY mean	VOLY mean
Health core	✓	✓	✓	✓
Health sensitivity	×	×	✓	✓
Crops	✓	✓	✓	✓
O_3/health metric	SOMO 35[(a)]	SOMO 35[(a)]	SOMO 0[(b)]	SOMO 0[(b)]
EU25 Average Damage Cost (€ per tonne of emission)				
NH_3	€11,000 ($16,940 AUD)	€16,000 ($24,460 AUD)	€21,000 ($32,340 AUD)	€31,000 ($47,740 AUD)
NOx	€4,400 ($6,776 AUD)	€6,600 (12,628 AUD)	€8,200 ($12,628 AUD)	€12,000 ($8,480 AUD)
$PM_{2.5}$	€26,000 ($40,040 AUD)	€40,000 ($61,600 AUD)	€51,000 ($78,540 AUD)	€75,000 ($115,500 AUD)
SO_2	€5,600 ($8,624 AUD)	€8,700 ($13,398 AUD)	€11,000 ($16,940 AUD)	€16,000 ($24,640 AUD)
VOCs	€950 ($1463 AUD)	€1,400 ($2,156 AUD)	€2,100 ($3,234 AUD)	€2,800 ($4,312 AUD)

(a) Sum of means over 35 ppb (parts per billion)
(b) Sum of means over 0 ppb

Table B5: Marginal NO_x, damage (€) per tonne emission for 2010 (in year 2000 prices), with three sets of sensitivity analysis.

Damage cost metric	Method 1	Method 2	Method 3	Method 4
Damage Cost Assumptions				
PM mortality	VOLY - med	VSL - med	VOLY – mean	VSL – mean
O_3 mortality	VOLY - med	VOLY - med	VOLY – mean	VOLY – mean
Health core	✓	✓	✓	✓
Health sensitivity	×	×	✓	✓
Crops	✓	✓	✓	✓
O_3/health metric	SOMO 35[(a)]	SOMO 35[(a)]	SOMO 0[(b)]	SOMO 0[(b)]
Average Damage Cost (€ per tonne of emission)				
Germany	€ 9,600 ($14,784 AUD)	€ 15,000 ($23,100 AUD)	€ 18,000 ($27,720 AUD)	€ 26,000 ($40,040 AUD)
Mediterranean Sea	€ 530 ($816 AUD)	€ 760 ($1,170 AUD)	€ 990 ($1,525 AUD)	€ 1,400 ($2,156 AUD)

(a) Sum of means over 35 ppb (parts per billion)
(b) Sum of means over 0 ppb

Table B6: Marginal $PM_{2.5}$ damage (€) per tonne emission for 2010 (in year 2000 prices), with three sets of sensitivity analysis.

Damage cost metric	Method 1	Method 2	Method 3	Method 4
Damage Cost Assumptions				
PM mortality	VOLY - med	VSL - med	VOLY – mean	VSL - mean
O_3 mortality	VOLY - med	VOLY - med	VOLY – mean	VOLY - mean
Health core	✓	✓	✓	✓
Health sensitivity	×	×	✓	✓
Crops	✓	✓	✓	✓
O_3/health metric	SOMO 35[a]	SOMO 35[a]	SOMO 0[b]	SOMO 0[b]
Average Damage Cost (€ per tonne of emission)				
Estonia	€ 4,200 ($6,468 AUD)	€ 6,500 ($10,010 AUD)	€ 8,300 ($12,782 AUD)	€ 12,000 ($18,480 AUD)
Netherlands	€ 63,000 ($9,702 AUD)	€ 96,000 ($147,840 AUD)	€ 120,000 ($184,800 AUD)	€ 180,000 ($277,200 AUD)

(a) Sum of means over 35 ppb (parts per billion)
(b) Sum of means over 0 ppb

Table B7: Marginal SO_2 damage (€) per tonne emission for 2010 (in year 2000 prices), with three sets of sensitivity analysis.

Damage cost metric	Method 1	Method 2	Method 3	Method 4
Damage Cost Assumptions				
PM mortality	VOLY - med	VSL - med	VOLY – mean	VSL – mean
O_3 mortality	VOLY - med	VOLY - med	VOLY – mean	VOLY – mean
Health core	✓	✓	✓	✓
Health sensitivity	×	×	✓	✓
Crops	✓	✓	✓	✓
O_3/health metric	SOMO 35[a]	SOMO 35[a]	SOMO 0[b]	SOMO 0[b]
Average Damage Cost (€ per tonne of emission)				
Greece	€ 1,400 ($2,156 AUD)	€ 2,100 ($3,234 AUD)	€ 2,700 ($4,158 AUD)	€ 4,000 ($6,160 AUD)
Netherlands	€ 13,000 ($20,020 AUD)	€ 21,000 ($32,340 AUD)	€ 26,000 ($40,040 AUD)	€ 39,000 ($60,060 AUD)

(a) Sum of means over 35 ppb (parts per billion)
(b) Sum of means over 0 ppb

B1.11 Recent updates

B1.11.1 Valuation updates

More recent analysis has provided updated mortality valuation. **Alberini *et al.* (2006)** used a contingent valuation stated preference technique to derive VSL values. The results are particularly useful for application in the European context since they are derived from pooled observations in three different EU countries. These VSL estimates were converted to equivalent VOLY estimates. Thus, the central VSL from **Alberini *et al.* (2006a)** of €1.11 million

($1,837,000 AUD) equates to €59,000 ($98,530 AUD) for a VOLY (Converted to AUD using approximate average market price over year 2006 of $1.67 to €). These results are supported by the findings of a second study **(Alberini *et al.*, 2006b)** which applied a similar method to Italy and the Czech Republic. The following provides updated health effect values.

VOLY low (€37,500) ($62,625 AUD)

VOLY mid (€60,000) ($100,200 AUD)

VOLY high (€215,000) ($359,050 AUD)

VSL low (€1.1 million) ($1,837,000 AUD)

VSL mid (€3.8 million) ($6,346,000 AUD)

VSL high (€5.6 million) ($9,352,000 AUD)

B1.11.2 CAFE Updates using cause-specific health endpoints

A number of other recent developments have emerged **(**see **Hurley *et al.*, 2011)**. Firstly, recent research in the US **(***e.g.* **Pope *et al.*, 2009; Krewski *et al.*, 2009)** has provided the basis for quantifying mortality impacts of PM exposure against cause-specific mortality rates (respiratory, cardio-vascular and lung cancer) rather than the all-cause mortality rates used in the CAFE analysis. This is an area of potential future work, though **Amann and Schopp (2011)** suggest that using the cause-specific approach would lead to higher estimates. Secondly, the values use UK life tables. While life table analysis in Italy and Sweden suggests only small differences to these, greater differences do arise for Poland, Bulgaria, Hungary, Romania, the Czech Republic and Slovakia. The use of cause-specific mortality outcomes is considered a priority for future assessments.

New European evidence on the association between PM exposure and the development of new cases of chronic bronchitis has appeared in a Swiss study **(Schindler *et al.*, 2009)**. However, it must be noted that the analysis uses US study functions. This suggests similar levels of analysis are a priority for inclusion in future studies. Recent epidemiological research has also expanded the evidence base. There was previously concern that exposure to fine particles from air pollution was associated with larger risks of cardiovascular mortality than would be expected from linear extrapolation of the more extensively researched risks of smoking. This question was specifically investigated by **Pope *et al.* (2009a)**, who concluded that observed relationships for air pollution were robust, with the exposure response relationship flattening out at higher exposure levels. Another paper by **Pope *et al.* (2009b)** provides evidence that air quality improvements in the USA have contributed to measurable improvements in human health over the period 1980 to 2000, equivalent to an increase in life expectancy across the population of 0.61 years for a 10 $\mu g/m^3$ fall in fine particle exposure, accounting for as much as 15% of the observed increase in life expectancy for the period across the study areas. **Smith *et al.* (2009)** provide evidence for significant effects of sulphate particles, which in part addresses questions over the risks from exposure to secondary aerosols.

B2 United Kingdom – Review of Air Quality Strategy

The UK has a long tradition of CBA for air pollution. The analysis of impacts and external costs has been taken forward by the Department of Health's Committee on the Medical Effects of Air

Pollutants (COMEAP)[v] and the Interdepartmental Group on Costs and Benefits (IGCB). IGCB undertook an economic analysis of the UK Air Quality Strategy as part of an overall review of the Strategy **(Defra, 2007)**.

B2.1 Approach used

IGCB used an impact pathway approach and generated damage costs by sector, with further disaggregation for transport-related emissions according to population density. The modelling was undertaken within a GIS framework and combined with population data to estimate the population-weighted mean concentration change. These were combined with the impact functions to provide values.

B2.2 Pollutants and health impacts considered

The IGCB approach focussed on the following air pollutants and their associated health effects:

- PM – Mortality from chronic exposure, mortality from acute exposure, and all respiratory and all cardiovascular hospital admissions.
- SO_2 – Mortality from acute exposure and all respiratory hospital admissions.
- O_3 – Mortality from acute exposure and all respiratory hospital admissions.
- NO_2 – All respiratory hospital admissions (only for sensitivity analysis).

B2.3 Emissions and air quality modelling

IGCB used the UK National Atmospheric Emissions Inventory (NAEI) and projections of emissions from the motor vehicle sector, made using road traffic emission factors, fed into a national air quality model. Simple empirical and statistical models, in which air quality from low-level sources is assumed to be proportional to emissions rates, and more sophisticated deterministic models in the case of PM, O_3, NO_2 and SO_2, were used to estimate future air pollution concentrations and to derive air pollution levels for geographic points for which there were no ambient air monitoring data. It is stressed that the UK approach only takes account of pollution that arises in the UK; it does not include trans-boundary pollution (a major difference to the CAFE approach).

B2.4 Estimation of health impacts

The analysis relied upon risk estimates based on analyses of the American Cancer Society (ACS) cohort by **Pope *et al.* (1995)** (updated in 2002), with a central estimate of 6% per 10 $\mu g/m^3$ of $PM_{2.5}$. The outputs were the change in longevity aggregated across the population ('years of life lost'). **IGCB (2001)** applied the risk estimates for $PM_{2.5}$ from the Pope study directly to the change in marginal PM_{10} concentrations, rather than $PM_{2.5}$ as in the original study.

As noted by IGCB, there remains no consensus on the lag period for chronic effects (*i.e.* the period between exposure and impact). **COMEAP (2001)** gave a range of health impacts assuming a lag of between 0 and 40 years. However, in its 2006 Interim Statement COMEAP stated that, although the evidence was limited, its judgement tended towards a greater proportion of the effect occurring in the years soon after a pollution reduction rather than later.

[v] www.advisorybodies.doh.gov.uk/comeap/

The analysis of chronic effects was undertaken using a life-table approach. The analysis assessed both the changes from a sustained pollution pulse (over 5, 20 and 100 years) and an annual pollution pulse and relied upon risk estimates from **Pope *et al.* (1995)** (updated in 2002), with a central estimate of 6% per 10 μg/m^3 of $PM_{2.5}$. The risk estimates for $PM_{2.5}$ from the Pope study directly to the change in marginal PM_{10} concentrations.

The approach did not separately quantify mortality due to acute exposure to PM_{10} and add these to the mortality from chronic exposure, as this would have led to the double counting of some impacts.

IGCB provided recommendations on valuation, drawing upon research in the area, particularly the Department of Environment, Food and Rural Affairs (Defra)-led study by **Chilton *et al.* (2004)** which aimed to identify the willingness to pay to reduce the health impacts associated with air pollution, using a survey-style contingent valuation approach. **Table B8** provides health effect values.

Table B8: Valuation of health effects

Health effect	Form of measurement	Valuation (2010 prices)	
		Central	Sensitivity
Mortality from chronic exposure	Number of years of life lost due to air pollution (life years). Life-expectancy losses assumed to be in normal health.	£29,000 ($72,500 AUD)	£21,700 - £36,200 ($64,487 - $107,578 AUD) (sensitivity around the 95% confidence intervals)
Respiratory hospital admissions	A hospital admission of average duration 8 days.	£1,900 – £9,100 ($5,646 - $27,043 AUD)	£1,900– £9,600 ($5,646 - $28,529 AUD)
Cardiovascular hospital admissions	A hospital admission of average duration 9 days.	£2,000 – £9,200 ($5,944 - $27,340 AUD)	£2,000 – £9,800 ($5,944 - $29,123 AUD)

Converted using approximate total change in cost from year 2004 of 18.9%, over 6 years, at an average annual inflation rate of 2.9%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

Further complexity was added by the type of pollution reduction, noting that life tables assume sustained reductions or increases in pollution over the lifetime of the population. For national air quality, the IGCB used life tables to assess the change in mortality from long-term exposure to PM. This assumes that pollution reductions are sustained over time (*i.e.* that, once reduced, the mortality hazards remain at their lower levels). The main life table calculations are based on a % change (decrease) in mortality hazard rates, applied to the population of England and Wales. The analysis is based on a 100-year sustained analysis. The approach applies changes in mortality hazards only at ages 30 years or more, because the underlying ACS cohort study only examined adults at ages 30 years or more.

Some policies do not lead to sustained pollution pulses. The life tables were therefore used to investigate the impact of a 1-year change in hazard rates (the rate at which a health outcome appears in the population under study) (after which time, hazard rates return to the previous levels). For this annual pulse approach it is important to stress that the impacts of a 1-year pollution pulse are followed up over a 100-year period. To illustrate this, with no lag and a 1-year pulse, the *mortality risks* change in year 1 only and then they revert to previous levels. The reason for following up the population over a full lifetime is that the lower mortality risks in year 1 under a pulse reduction imply fewer deaths in year 1 (*i.e.* the number of deaths 'saved'), also and necessarily a slightly *increased* population in Year 2 and subsequently; and so slightly *more* deaths in years 2 and onwards, because of the slightly larger population at risk. The analysis over a full lifetime 'tracks' how all this plays out. Another way of expressing this is that analysis is conducted over 100 years to see when the deaths 'saved' in year 1 actually occur later, because necessarily they will occur. These 1-year pulse implementations were based *only* on the population alive at the time of the pulse – for analysis of that 1-year pulse, no account was taken of new birth cohorts born in later years. Note, that discounting is applied to YOLLs in future years during the valuation step.

The approach did not separately quantify mortality due to acute exposure to PM_{10} and add these to the mortality from chronic exposure, as this would have led to the double counting of some impacts. For morbidity, a much smaller list of health endpoints was considered (**Table B9**).

Table B9: List of endpoints considered for morbidity

Impact	Function
Respiratory hospital admissions - PM_{10}	Risk estimate of 0.8% increase in Respiratory hospital admissions per 10 µg/m^3 PM_{10}Baseline rate 980 per 100,000
Cardiovascular hospital admissions - PM_{10}	Risk estimate of 0.8% increase in Cardiovascular hospital admissions per 10 µg/m^3 PM_{10} Baseline rate 981 per 100,000

B2.5 Valuation

IGCB provided recommendations on valuation, drawing upon research in the area, particularly the Department of Environment, Food and Rural Affairs (Defra)-led study by **Chilton *et al.* (2004)** which aimed to identify the willingness to pay to reduce the health impacts associated with air pollution, using a survey-style contingent valuation approach. **Table B10** provides health effect values. For morbidity, the estimates include resource costs and dis-utility (opportunity costs, *i.e.* lost productivity, are considered in the sensitivity).

Table B10: Valuation of health effects

Health Effect	Form of measurement to which the valuations will apply	Valuation – (2004 prices)	
		Central	Sensitivity
Mortality from chronic exposure	Number of years of life lost due to air pollution (life years).Life-expectancy losses assumed to be in normal health.	£29,000 ($72,500 AUD)	£21,700 - £36,200 ($54,250 - $90,500 AUD) (sensitivity around the 95% confidence intervals)
Respiratory hospital admissions	A hospital admission of average duration 8 days.	£1,900 – £9,100 ($4,750 - $22,750 AUD)	£1,900– £9,600 ($4,750 - $24,000 AUD)
Cardiovascular hospital admissions	A hospital admission of average duration 9 days.	£2,000 – £9,200 ($5,000 - $23,000 AUD)	£2,000 – £9,800 ($5,000 - $24,500 AUD)

Converted using approximate average market price over year 2004 of $2.50 to £.

B2.6 Analysis of the health effects values

In contrast to the CAFE approach, there is very little coverage of morbidity in the IGCB approach. This has the effect that mortality from chronic exposure completely dominates the entire analysis (accounting for more than 99% of health costs), to such an extent that morbidity is irrelevant.

B2.7 Damage Costs

As part of the work of the IGCB for the Air Quality Review, Defra produced damage costs **(Watkiss *et al*., 2006)**. This used the impact pathway approach, running additional pulses of emissions over the baseline, looking at the modelled concentrations, stock at risk, and health impacts, and producing monetary values. The analysis was undertaken within a GIS framework. The analysis then looked at marginal emission reductions, by reducing emissions by 10% in each sector, or by a suitable marginal quantity (*e.g.* 50,000 tonnes). The impact pathway analysis is re-estimated (changes in emissions, changes in air pollution concentrations, changes in impacts, changes in values). The marginal change in values was then normalised against the change in emissions (in tonnes) to produce damage cost. It was assumed that the model response to different marginal changes will be linear (*i.e.* for smaller or larger changes than 10%). This approximation was appropriate for primary PM and the secondary PM analysis (where the model does effectively behave linearly[w]). This was combined with population data to estimate the population-weighted mean concentration change. These were combined with the impact functions to provide values. For chronic effects, the central estimate in the damage costs was based around a distribution that accounted for hazard rate and lag. The damage costs also used the annual pollution pulse.

The resulting values are presented in **Table B11**. The damage costs for SO_2 as sulfates (PM) and NO_x (as nitrate) have been presented as single values irrespective of sector or location. As these are secondary (regional) pollutants, this approximation is acceptable (as the formation in the atmosphere takes time). For primary PM there are much greater differences between sectorial emissions, because of the population-weighted increases from different emission sources. The analysis therefore used a sectoral split, with a more disaggregated division of road transport by area type.

Damage costs for PM (transport) are at a UK-wide level, with disaggregated damage costs presented below split by current National Transport Model area.

[w] Note that the approach assumes that secondary particles reduce across the UK in the same proportions. In practice, there will be different reduction patterns according to the exact spatial pattern of emission changes.

Table B11: IGCB- Air Quality Damage costs per tonne (2010 prices)

Metric	Sensitivities									
	Central estimate[(1)]		Low central range[(2)]		High central range[(2)]		Low sensitivity[(3)]		High sensitivity[(3)]	
	GBP	AUD	GBP	AUD	GBP	AUD	GBP	AUD	GBP	AUD
NO_X	**£955**	**$1,433**	£744	$1,116	£1,085	$1,628	£187	$281	£2,164	$3,246
SO_X	**£1,633**	**$2,450**	£1,320	$1,980	£1,856	$2,784	£520	$780	£3,452	$5,178
Ammonia	**£1,972**	**$2,958**	£1,538	$2,307	£2,241	$3,362	£733	$1,100	£1,069	$1,604
PM Domestic	**£28,140**	**$42,210**	£22,033	$33,050	£31,978	$47,967	£3,033	$4,550	£79,131	$118,697
PM Agriculture	**£9,703**	**$14,555**	£7,598	$11,397	£11,027	$16,541	£1,046	$1,569	£27,286	$40,929
PM Waste	**£20,862**	**$31,293**	£16,335	$24,503	£23,708	$35,562	£2,248	$3,372	£58,666	$87,999
PM Industry	**£20,862**	**$31,293**	£19,753	$29,630	£28,669	$43,004	£2,720	$4,080	£70,945	$106,418
PM ESI	**£25,229**	**$37,844**	£1,900	$2,850	£2,757	$4,136	£495	$743	£6,257	$9,386
PM Transport Average	**£2,426**	**$3,639**	£37,987	$56,981	£55,133	$82,700	£9,897	$14,846	£125,134	$187,701
PM Transport Central London	**£221,726**	**$332,589**	£173,601	$260,402	£251,961	$377,942	£45,229	$67,844	£571,859	$857,789
PM Transport Inner London	**£228,033**	**$342,050**	£178,540	$267,810	£259,129	$388,694	£45,229	$67,844	£588,126	$882,189
PM Transport Outer London	**£148,949**	**$223,424**	£116,621	$174,932	£169,261	$253,892	£30,383	$45,575	£384,160	$576,240
PM Transport Inner Conurbation	**£117,899**	**$176,849**	£92,309	$138,464	£133,975	$200,963	£24,050	$36,075	£304,074	$456,111
PM Transport Outer Conurbation	**£73,261**	**$109,892**	£57,362	$86,043	£83,252	$124,878	£14,944	$22,416	£188,951	$283,427
PM Transport Urban Big	**£87,332**	**$130,998**	£68,377	$102,566	£99,241	$148,862	£17,815	$26,723	£225,240	$337,860
PM Transport Urban Large	**£70,351**	**$105,527**	£55,081	$82,622	£79,944	$119,916	£14,351	$21,527	£181,443	$272,165
PM Transport Urban Medium	**£55,310**	**$82,965**	£43,305	$64,958	£62,853	$94,280	£11,283	$16,925	£142,652	$213,978
PM Transport Urban Small	**£34,932**	**$52,398**	£27,351	$41,027	£39,696	$59,544	£7,126	$10,689	£90,096	$135,144
PM Rural	**£15,041**	**$22,562**	£11,776	$17,664	£17,091	$25,637	£3,068	$4,602	£38,791	$58,187

(1) The central damage cost is derived from the lag probability distribution developed for Monte Carlo analysis to reflect the fact that, although evidence is limited, COMEAP tend towards a greater proportion of the health effect occurring in the years sooner after the pollution rather than later. This estimate is intended for use only where a single point estimate is necessary and should always be accompanied by the central range.

(2) Variation between the central values reflects uncertainty about the lag between exposure and the associated health impact. The presented figures show the range between a 0 and 40 year lag. This sensitivity should be reported as the central sensitivity.

(3) In addition to the lag range this sensitivity also applies the recommended COMEAP typical high (12%) and typical low (1%) hazard rate sensitivity. This sensitivity is intended for use only as a second round sensitivity.

B2.8 Guidance

In deciding when to use the impact pathway or damage cost approach, a useful approach has been adopted in the UK. The user is guided on the appropriate method to take based on a decision tree. This leads to a two-phase strategy based on the type of application and the materiality of the changes proposed. In very simple terms, for major air quality policy, especially new air quality strategies, an impact pathway approach is recommended, because of the high potential cost implications. This is also applied for national level projects that involve major pollution changes or high costs/benefits. For appraisal of individual projects, or smaller specific policies, or where air quality is not the primary driver of the legislation, then damage costs are recommended. This is shown in **Figure B2**.

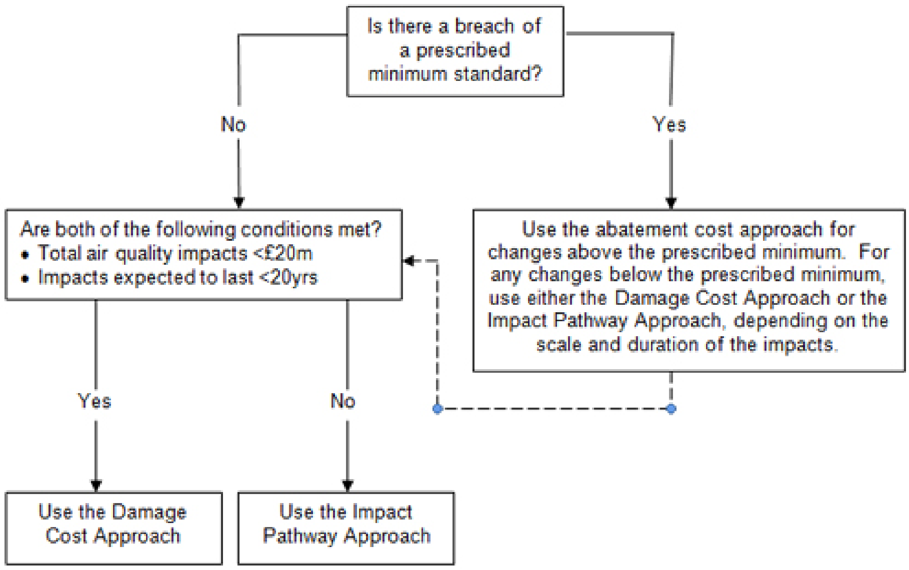

Figure B2: Guidance decision tree for valuation decisions (Defra, 2008)

B2.9 Recent Updates

A recent addition has been the use of activity costs. Activity costs are an extension of the damage cost approach. The evaluation process builds on the damage cost approach and links an activity or policy measure directly to the cost of the emissions it generates. In practice this means that to estimate damage costs the only information required is the change in activity. Previously the change in emissions from this change in activity was required.

B3 United States – National Air Quality Standards

B3.1 Background

The US has long adopted CBA for air quality regulations and impact assessment. More recently, the United States Environmental Protection Agency (USEPA) has significantly developed the cost-benefit method for air pollution as part of the Benefits and Costs of the Clean Air Act[x]. The USEPA conducted a benefits and costs analysis of the national air quality standards for particle pollution, which the Agency strengthened in 2006[y].

[x] To ensure careful consideration of all aspects of regulations under the Clean Air Act, Congress added to the 1990 Amendments a requirement under section 812 that EPA conduct periodic, scientifically reviewed, studies to assess the benefits and the costs of the entire Clean Air Act. Since this requirement was established, EPA has conducted three comprehensive studies on the benefits and costs of the Clean Air Act

[y] http://www.epa.gov/ttnecas1/ria.html

The USEPA did not publish PM damage costs. However, **Fann *et al.* (2009)** published PM damage costs, adjusting for location, using broadly the EPA approach. They undertook modelling (using CMAQ and BenMAP) for each of nine urban areas and one nationwide area to estimate monetised health benefits from changes in air quality. Upon modelling the air quality change, they divided the total monetised health benefits by the $PM_{2.5}$ precursor emission reductions to generate $/ton metrics. Reductions in directly emitted carbonaceous particles offered the largest $/ton. This relatively large estimate is likely due to the fact that these particles are emitted as total $PM_{2.5}$ and thus do not undergo any additional transformation in the atmosphere before affecting population centres. The results for directly emitted carbonaceous particles are shown below (**Table B12**).

Table B12: The monetised US$/ton in 2015 of reductions in $PM_{2.5}$ by area of the US (Fann *et al*, 2009).

	Area source carbon	Mobile source carbon	EGU[z] and non-EGU carbon
National	$720,000	$550,000	$460,000
Atlanta	$670,000	$590,000	$620,000
Chicago	$510,000	$580,000	$600,000
Dallas	$1,100,000	$790,000	$1,100,000
Denver	$280,000	$450,000	$220,000
NY/Phi	$570,000	$710,000	$780,000
Salt Lake	$140,000	$150,000	$65,000
San Joaquin	$910,000	$560,000	$720,000
Seattle	$500,000	$570,000	$720,000

B3.2 Approach used

The general benefits analysis framework used an impact pathway approach, using detailed air quality models. The current methodology diverged from the previous work, moving the assessment of uncertainties from its ancillary analyses into its main benefits presentation and updating the projections of mortality incidence rates. The current methodology diverged in four areas from the previous CAIR:

- Rather than presenting both a 'primary' estimate of the benefits and a separate characterisation of the uncertainty associated with that estimate, the current analysis follows the recommendation of NRC's 2002 report 'Estimating the Public Health Benefits of Proposed Air Pollution Regulations' to begin moving the assessment of uncertainties from its ancillary analyses into its main benefits presentation through the conduct of probabilistic analyses.
- Since the publication of CAIR, the USEPA has completed a full-scale expert elicitation (synthesis of opinions of experts) designed to better characterise the state of their understanding of the concentration-response (C-R) function for PM-related premature mortality. The elicitation results from a major component of the current effort to use probabilistic assessment techniques to integrate uncertainty into the main benefits analysis.

[z] Electrical generating units

- The USEPA has updated the projections of mortality incidence rates to be consistent with the US Census population projections that form the basis of future population estimates. Compared with the methodology used in the CAIR analysis, this change will result in a reduction in mortality impacts in future years, as overall mortality rates are projected to decline for most age groups.

B3.3 Pollutants and health impacts considered

The impacts from PM considered in the analysis are outlined in **Table B13**. Note that the analysis performed by the USEPA included visibility which was an important outcome in the overall valuation results.

Table B13: Human health and welfare effects of pollutants controlled to simulate attainment with $PM_{2.5}$ standards

Pollutant/Effect	Quantified and Monetised Effects	Unquantified Effects
PM/Health[b]	Premature mortality based on cohort study estimates[c] Bronchitis: chronic and acute Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Non-fatal heart attacks (myocardial infarction) Lower and upper respiratory illness Minor restricted-activity days Work loss days Asthma exacerbations (asthmatic population) Respiratory symptoms (asthmatic population) Infant mortality	Low birth weight Pulmonary function Chronic respiratory diseases other than chronic bronchitis Non-asthma respiratory emergency room visits UVb exposure (+/-)[d]
PM/Welfare	Visibility in Southeastern, Southwestern, and California Class I areas	Visibility in residential and non-Class I areas UVb exposure (+/-)[d] Global climate impacts (+/-)[d]

Source: USEPA http://www.epa.gov/ttn/ecas/regdata/RIAs/Table%20of%20Contents.pdf

[a] Reductions in certain $PM_{2.5}$ precursors such as NO_x and VOC may also lead to changes in ambient concentrations of ozone. These changes in ozone will also have health and welfare effects. However, for this Regulatory Impact Analysis (RIA), because the majority of the illustrative strategies evaluated do not affect NO_x and VOC emissions, we focus on estimating the health and welfare effects associated with changes in ambient $PM_{2.5}$. For a full listing of health and welfare effects associated with ozone exposures, see the Ozone Criteria Document (U.S. EPA, 2006), and Chapter 4 of the RIA for the Clean Air Interstate Rule (U.S. EPA, 2005).

[b] In addition to primary economic endpoints, there are a number of biological responses that have been associated with PM health effects including morphological changes and altered host defence mechanisms. The public health impact of these biological responses may be partly represented by our quantified endpoints.

[c] Cohort estimates are designed to examine the effects of long term exposures to ambient pollution, but relative risk estimates may also incorporate some effects due to shorter term exposures (see Kunzli et al, 2001 for a discussion of this issue). While some of the effects of short term exposure are likely to be captured by the cohort estimates, there may be additional premature mortality from short term PM exposure not captured in the cohort estimates included in the primary analysis.

[d] May result in benefits or disbenefits.

The analysis included long-term (chronic) and shorter-term (acute) exposures for mortality and morbidity. A number of key methodological issues are highlighted below. All fine particles, regardless of their chemical composition, were assumed to be equally potent in causing premature mortality (as in the UK and EC).

Mortality due to chronic exposure was quantified using the Pope *et al.* **(2002)** functions (as in the UK and EC). Specifically, since the most recent Science Advisory Board (SAB) review, an extended follow up of the Harvard Six Cities study, undertaken by Dockery, has been published **(Laden *et al*., 2006)**, which was used in the study.

The benefits of attaining alternative standards were estimated using BenMAP, a computer program developed by USEPA that integrates a number of the modelling elements used in previous Regulatory Impact Analysis (RIA) and calculates the impact on specific areas (available at http://www.epa.gov/ttn/ecas/benmodels.html).

B3.3 Emissions and air quality modelling

The study used the data from the National Emissions Inventory (NEI). The NEI is a national compilation of emissions sources collected from state, local, and other air agencies as well as from emissions information from the EPA emissions programs. It is a comprehensive and detailed estimate of air emissions of both Criteria and Hazardous air pollutants from all air emissions sources and includes five data categories Point, Non-point, On-road, Non-road, and Event. The NEI is prepared every three years by the USEPA based primarily upon emission estimates and emission model inputs provided by State, Local, and Tribal air agencies for sources in their jurisdictions, and supplemented by data developed by the USEPA.

The emissions data in the NEI are compiled for detailed emissions processes within a facility for large "point" sources or as a county total for smaller "nonpoint" sources and spatially dispersed sources such as on-road and non-road mobile sources.

Community-Scale Air Quality (CMAQ) model is a three-dimensional grid-based Eulerian air quality model designed to estimate the formation and fate of oxidant precursors, primary and secondary particulate matter concentrations and deposition over regional and urban spatial scales (e.g., over the contiguous U.S.). Consideration of the different processes (e.g. transport and deposition) that affect primary (directly emitted) and secondary (formed by atmospheric processes) PM at the regional scale in different locations is fundamental to understanding and assessing the effects of pollution control measures that affect PM, ozone and deposition of pollutants to the surface.

The key inputs to the CMAQ model include emissions from anthropogenic and biogenic sources, meteorological data, and initial and boundary conditions.

B3.4 Estimation of health impacts

The analysis included long-term (chronic) and shorter-term (acute) exposures for mortality and morbidity. A number of key methodological issues are highlighted below.

- Mortality from chronic exposure is quantified using the Pope *et al.* **(2002)** functions (as in the UK and EC). Specifically, since the most recent Science Advisory Board (SAB) review, an extended follow up of the Harvard Six Cities study, undertaken by Dockery, has been published **(Laden *et al*., 2006)**, which was used in the study.

- All fine particles, regardless of their chemical composition, were assumed to be equally potent in causing premature mortality (as in the UK and EC). However, the study provided estimates of the proportions of benefits that are attributable to specific components of $PM_{2.5}$ such as ammonium sulphate, ammonium nitrate, elemental carbon, organic carbon, and crustal material (which included metals).
- The C-R function for fine particles was approximately linear within the range of ambient concentrations under consideration (above the assumed threshold of 10 µg/m^3). The use of a cut point is different to the UK and EC approaches. However, the document also stated that the available evidence neither supported nor refuted the existence of thresholds for the effects of PM on mortality across the range of concentrations in the studies.
- A detailed lag structure was assumed, characterised by 30% of mortality reductions occurring in the first year, 50% occurring evenly over years 2 to 5 after the reduction in $PM_{2.5}$, and 20% occurring evenly over the years 6 to 20 after the reduction in $PM_{2.5}$. The distribution of deaths over the latency period was intended to reflect the contribution of short-term exposure in the first year, cardiopulmonary deaths in the 2- to 5-year period, and long-term lung disease and lung cancer in the 6- to 20-year period.

B3.5 Valuation

As previously mentioned, the benefits of attaining alternative standards were estimated using BenMAP. The valuation of human health benefits were adjusted upwards to account for projected growth in real US income. The VSL used was USD 5,500,000 (1990 income levels). A single VSL was used for all reductions in mortality risk. These were applied to the incidences of premature mortality related to PM exposures. These exposures occurred in a distributed fashion over the 20 years following exposure and the valuation included an annual 3% discount rate to the value of premature mortality occurring in future years.

Given all the variables for mortality from chronic exposure, with sensitivity and expert judgement, a very large number of outputs are produced, especially as 3 and 7% discount rates were used, and alternative levels assessed. These increase when potential threshold effects are included.

An example of the estimated monetary values is shown in **Tables B14** and **B15**.

Table B14: Illustrative strategy to attain 15/35: Estimated monetary value of reduction in risk if premature mortality (3-percent discount rate, in millions of 1999 USD) 90th percentile confidence intervals provided in parentheses

	Eastern U.S.		Western U.S. Excluding CA		California		National Total		
	Modeled Partial Attainment	Residual Attainment	Modeled Partial Attainment	Residual Attainment	Modeled Partial Attainment	Residual Attainment	Modeled Partial Attainment	Residual Attainment	National Total Full Attainment
Mortality Impact Functions Derived from Epidemiology Literature									
ACS Study[b]	$2,100 ($470 – $4,400)	$97 ($22 – $200)	$440 ($99 – $920)	$87 ($19 – $180)	$3,000 ($670 – $6,200)	$9,000 ($2,000 – $19,000)	$5,500 ($1,200 – $12,000)	$9,200 ($2,000 – $19,000)	$15,000 ($3,300 – $31,000)
Harvard Six-City Study[c]	$4,800 ($1,200 – $9,200)	$220 ($57 – $430)	$1,000 ($260 – $1,900)	$200 ($51 – $380)	$6,800 ($1,800 – $13,000)	$20,000 ($5,300 – $39,000)	$13,000 ($3,300 – $24,000)	$21,000 ($5,400 – $40,000)	$33,000 ($8,600 – $64,000)
Woodruff et al 1997 (infant mortality)	$6 ($1 – $11)	$0 ($0 – $0)	$4 ($1 – $8)	$2 ($0 – $4)	$8 ($2 – $15)	$28 ($7 – $55)	$17 ($4 – $35)	$30 ($7 – $59)	$47 ($12 – $94)
Mortality Impact Functions Derived from Expert Elicitation									
Expert A	$9,800 ($1,300 – $22,000)	$240 ($32 – $540)	$8,000 ($1,100 – $18,000)	$2,100 ($280 – $4,800)	$9,000 ($1,200 – $20,000)	$29,000 ($4,000 – $67,000)	$27,000 ($3,600 – $61,000)	$32,000 ($4,300 – $72,000)	$59,000 ($7,900 – $130,000)
Expert B	$7,800 ($650 – $21,000)	$200 ($21 – $510)	$6,100 ($390 – $17,000)	$1,700 ($120 – $4,500)	$7,400 ($740 – $19,000)	$24,000 ($2,300 – $62,000)	$21,000 ($1,800 – $57,000)	$26,000 ($2,400 – $68,000)	$47,000 ($4,200 – $120,000)
Expert C	$8,100 ($980 – $20,000)	$200 ($24 – $480)	$6,600 ($800 – $16,000)	$1,800 ($210 – $4,200)	$7,500 ($900 – $18,000)	$24,000 ($3,000 – $59,000)	$22,000 ($2,700 – $54,000)	$26,000 ($3,200 – $63,000)	$48,000 ($5,900 – $120,000)
Expert D	$5,300 ($800 – $11,000)	$130 ($19 – $270)	$4,300 ($650 – $9,100)	$1,200 ($170 – $2,400)	$4,900 ($730 – $10,000)	$16,000 ($2,400 – $34,000)	$15,000 ($2,200 – $31,000)	$17,000 ($2,600 – $36,000)	$32,000 ($4,800 – $67,000)
Expert E	$12,000 ($3,100 – $24,000)	$300 ($76 – $600)	$10,000 ($2,500 – $20,000)	$2,700 ($670 – $5,300)	$11,000 ($2,800 – $22,000)	$37,000 ($9,300 – $73,000)	$34,000 ($8,500 – $67,000)	$40,000 ($10,000 – $79,000)	$74,000 ($19,000 – $150,000)
Expert F	$7,200 ($1,900 – $13,000)	$170 ($47 – $330)	$5,800 ($1,600 – $11,000)	$1,500 ($420 – $2,900)	$6,600 ($1,800 – $12,000)	$22,000 ($5,900 – $40,000)	$19,000 ($5,300 – $37,000)	$23,000 ($6,300 – $44,000)	$43,000 ($12,000 – $80,000)
Expert G	$4,300 ($0 – $11,000)	$110 ($0 – $260)	$3,500 ($0 – $8,700)	$940 ($0 – $2,300)	$4,000 ($0 – $9,800)	$13,000 ($0 – $32,000)	$12,000 ($0 – $29,000)	$14,000 ($0 – $35,000)	$26,000 ($0 – $64,000)
Expert H	$5,300 ($17 – $15,000)	$130 ($0 – $370)	$4,300 ($14 – $12,000)	$1,200 ($4 – $3,300)	$4,900 ($16 – $14,000)	$16,000 ($52 – $46,000)	$15,000 ($47 – $42,000)	$17,000 ($56 – $49,000)	$32,000 ($100 – $91,000)
Expert I	$7,600 ($900 – $17,000)	$190 ($22 – $410)	$6,200 ($730 – $14,000)	$1,600 ($190 – $3,600)	$7,000 ($830 – $15,000)	$23,000 ($2,700 – $50,000)	$21,000 ($2,500 – $46,000)	$25,000 ($2,900 – $54,000)	$45,000 ($5,400 – $100,000)
Expert J	$6,800 ($1,100 – $16,000)	$160 ($28 – $390)	$5,500 ($930 – $13,000)	$1,500 ($250 – $3,500)	$6,200 ($1,100 – $15,000)	$20,000 ($3,500 – $48,000)	$18,000 ($3,100 – $44,000)	$22,000 ($3,700 – $52,000)	$40,000 ($6,900 – $95,000)
Expert K	$1,100 ($0 – $6,000)	$27 ($0 – $150)	$900 ($0 – $4,800)	$240 ($0 – $1,300)	$1,100 ($0 – $6,000)	$3,400 ($0 – $18,000)	$3,100 ($0 – $17,000)	$3,600 ($0 – $20,000)	$6,800 ($0 – $36,000)
Expert L	$5,300 ($480 – $13,000)	$140 ($20 – $330)	$3,800 ($110 – $10,000)	$1,100 ($59 – $2,800)	$5,300 ($720 – $12,000)	$17,000 ($2,100 – $40,000)	$14,000 ($1,300 – $36,000)	$18,000 ($2,200 – $43,000)	$32,000 ($3,500 – $79,000)

[a] All estimates are rounded to 2 significant digits. All rounding occurs after final summing of unrounded estimates. As such, totals will not sum across columns.
[b] The estimate is based on the concentration-response (C-R) function developed from the study of the American Cancer Society cohort reported in Pope et al (2002), which has previously been reported as the primary estimate in recent RIAs
[c] Based on Laden et al (2006) reporting of the extended Six-cities study; to be reviewed by the EPA-SAB for advice on the appropriate method for incorporating what has previously been a sensitivity estimate.

Table B15: Mortality threshold sensitivity analysis for 15/35 scenario (Using Pope et al., 2002 Effect Estimate with Slope Adjustment for Thresholds above 7.5ug) 90th percentile confidence intervals provided in parentheses

		Estimated Reduction in Mortality Incidence							
		Eastern U.S.		*Western U.S. Excluding CA*		*California*		*National Total*	
	Level of Assumed Threshold	*Modeled Partial Attainment*	*Residual Attainment*	*Modeled Partial Attainment*	*Residual Attainment*	*Residual Attainment*	*Modeled Partial Attainment*	*Residual Attainment*	*Modeled Partial Attainment*
Less Certainty That Benefits Are at Least as Large	No Threshold	620 (240 – 1,000)	15 (6 – 24)	510 (200 – 810)	140 (53 – 220)	2,000 (800 – 3,300)	570 (220 – 920)	1,900 (740 – 3,000)	1,700 (670 – 2,700)
	Threshold at 7.5 µg	610 (240 – 980)	15 (6 – 24)	320 (130 – 520)	110 (44 – 180)	2,000 (790 – 3,200)	560 (220 – 900)	1,900 (740 – 3,000)	1,500 (590 – 2,400)
	Threshold at 10 µg	360 (140 – 580)	17 (7 – 27)	80 (30 – 120)	15 (6 – 24)	1,600 (620 – 2,500)	520 (200 – 0,800)	1,600 (610 – 2,500)	960 (370 – 1,500)
	Threshold at 12 µg	38 (15 – 62)	2 (1 – 3)	12 (5 – 19)	0 (0 – 0)	1,200 (490 – 2,000)	430 (170 – 0,700)	1,200 (490 – 2,000)	480 (190 – 800)
More Certainty That Benefits are at Least as Large	Threshold at 14 µg	10 (4 – 16)	2 (1 – 3)	9 (3 – 14)	0 (0 – 0)	440 (170 – 700)	390 (150 – 0,600)	440 (170 – 700)	410 (160 – 700)

a All estimates are rounded to 2 significant digits. All rounding occurs after final summing of unrounded estimates. As such, totals will not sum across columns.

B3.6 Analysis of the health effect values

The analysis showed that mortality from chronic exposure dominates the health costs, at around 93% of the health values (**Figure B3**). Of the morbidity values, chronic bronchitis was relatively significant.

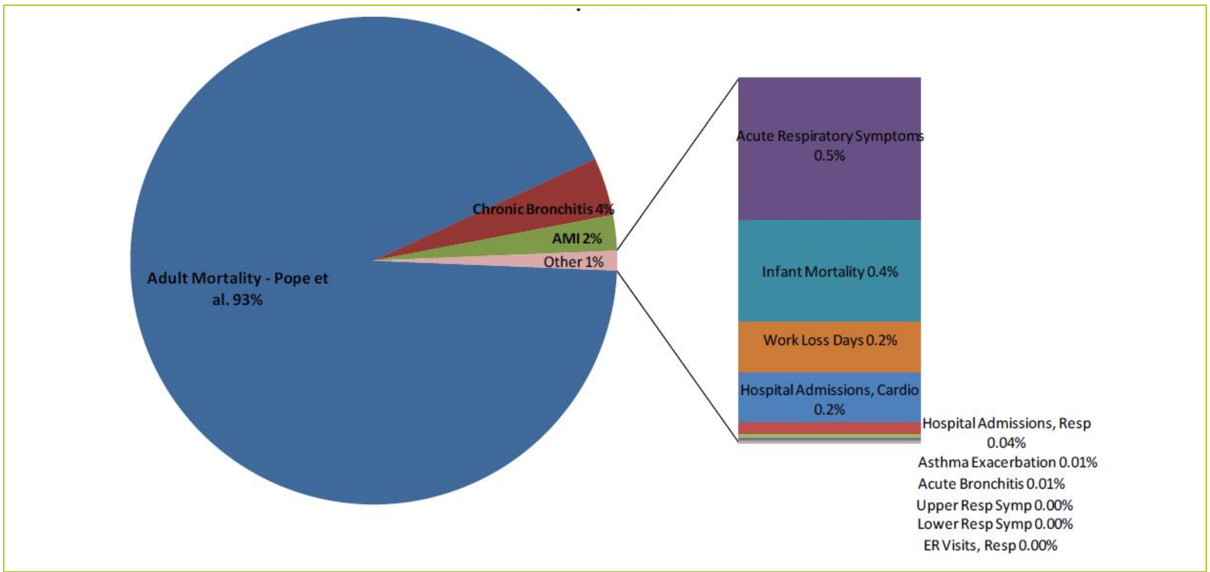

Figure B3: Breakdown of monetized $PM_{2.5}$ health benefits using mortality function from Pope *at al* (2002).

B3.7 Recent updates

In the more recent USEPA analysis of the national ambient air quality standard (NAAQS) for SO_2, which the Agency finalised on June 2, 2010, benefit per ton values are used for SO_2 secondary particulate formation **(Fann *et al.*, 2009)**.

The benefit-per-ton coefficients in this analysis were derived using modified versions of the health impact functions used in the PM NAAQS Regulatory Impact Analysis. Specifically, this analysis used the benefit-per-ton estimates first applied in the Portland Cement National Emission Standards for Hazardous Air Pollutants (NESHAP) Regulatory Impact Analysis (RIA) (USEPA, 2009a) It incorporated three updates: a new population dataset, an expanded geographic scope of the benefit-per-ton calculation, and the functions directly from the epidemiology studies without an adjustment for an assumed threshold. The benefit-per-ton estimates are provided in Table B16.

Table B16: $PM_{2.5}$ Co-benefits associated with reducing SO_2 emissions (2006 USD)

$PM_{2.5}$ Precursor	Benefit per ton estimate (Pope)	Benefit per ton estimate (Laden)
SO_2 EGU[b]	$42,000	$100,000
SO_2 non-EGU	$30,000	$74,000
SO_2 area	$19,000	$47,000

a Estimates have been rounded to two significant figures. Confidence intervals are not available for benefit per-ton estimates. Estimates shown use a 3% discount rate.

[b] EGU: Electricity Utility Generating Units

A similar approach was also used for CBA of the NAAQS for nitrogen dioxide, which the Agency finalised on January 22, 2010 (**Table B17**).

Table B17: PM$_{2.5}$ Benefit-per-ton estimates at discount rates of 3% and 7% (millions of 2006 USD) [1]

PM$_{2.5}$ Precursor	Benefit per Ton Estimate (Pope)	Benefit per Ton Estimate (Laden)
NO_x mobile 3% (no-threshold)[2]	$5,200	$13,000
NO_x mobile 7% (no-threshold)[2]	$4,700	$11,000

[1] Numbers have been rounded to two significant figures. This table includes extrapolated tons, spread across the sectors in proportion to the emissions in the county. PM$_{2.5}$ co-benefit estimates do not include confidence intervals because they are derived using benefit per-ton estimates. For the selected standard of 100 ppb, there would be zero costs and benefits as we project all areas to attain this standard without additional controls.

[2]The benefit-per-ton estimates using thresholds are $4,300 to $9,300 at 3% and $3,900 and $8,400 at 7%. These estimates assume a threshold at 10µg/m^{3}, and are provided as a historical reference only.

B3.8 Damage costs

The USEPA did not publish PM damage costs. However, **Fann *et al.* (2009)** published secondary PM precursor damage costs, adjusting for location, using broadly the EPA approach. Their results are shown in **Figure B4** below.

	Area source carbon	Mobile source carbon	EGU & Non-EGU carbon	Area source SOx	EGU SOx	Non-EGU SOx	VOC	Area source NH3	Mobile source NH3	EGU NOx	Non-EGU NOx	Mobile source NOx
National	$720,000	$550,000	$460,000	$40,000	$82,000	$59,000	$2,400	$38,000	$95,000	$15,000	$9,700	$10,000
Atlanta	$670,000	$590,000	$620,000	$48,000	$15,000	$42,000	$1,200	-$4,100	$56,000	$7,900	-$4,500	-$4,100
Chicago	$510,000	$580,000	$600,000	$29,000	$18,000	$15,000	$3,100	$36,000	$100,000	$1,100	$2,000	-$8,700
Dallas	$1,100,000	$790,000	$1,100,000	$140,000		$29,000	$600	$16,000	$36,000	$34,000	$920	$370
Denver	$280,000	$450,000	$220,000	$75,000	$6,400	$19,000	$1,400	$10,000	$58,000	$3,200	$3,800	$2,700
NY/Phi	$570,000	$710,000	$780,000	$14,000	$74,000	$50,000	$4,300	$53,000	$140,000	$1,500	-$2,600	-$8,200
Phoenix	$2,500,000	$1,700,000	$980,000	$73,000		$550,000	$2,000	$15,000	$43,000	$11,000	-$2,100	-$680
Salt Lake	$140,000	$150,000	$65,000	$15,000		$9,100	$2,600	$29,000	$43,000		$4,200	$1,500
San Joaquin	$910,000	$560,000	$720,000	$140,000	$350,000	$46,000	$5,700	$36,000	$140,000	$28,000	$28,000	$43,000
Seattle	$500,000	$570,000	$720,000	$54,000	$6,300	$52,000	$560	$18,000	$49,000	$120,000	-$2,300	-$8,100

Figure B4: The monetised USD/ton in 2015 of reductions in PM$_{2.5}$ precursor emissions by area of the country (using Laden et al (2006) mortality estimate, 2006USD, 3% discount rate)
Source: Fann *et al* (2009).

The ranges mirror the findings of the CAFE work, showing large variations across pollutants. The urban area-specific USD/ton estimates suggest a significant amount of inter-regional and intraregional variability.

B4 Canada – Ontario coal-fired electricity generation

Environment Canada has used CBA to guide air quality policy. The most recent study cited is 'Cost-Benefit Analysis: Replacing Ontario's Coal-Fired Electricity Generation' **(RWDI, 2005)**.This included analysis of secondary PM from SO_2 and NO_x.

Two premature mortality risk factors were used in this analysis. Previous air pollution health damage estimates for Ontario were based on time-series risk factors that only captured acute premature mortality risks. The report also considered mortality from cohort-based studies (mortality from chronic exposure) used for estimating health risks associated with exposure to air pollution by the USEPA, and argues (as do the EC and US studies) that cohort-based risk factors are more appropriate for this type of public policy analysis (since they capture more completely the negative effects of air pollution exposure).

The premature mortality risk associated with short-term exposure to air pollution, based on local studies, was included for comparison purposes only. For valuation, account was made of empirical studies that revealed WTP for low risk reductions. Future risk reductions were discounted relative to immediate risk reductions (at 5%). A VSL of CDN4.18 million (Canadian Dollars (CDN) 2004) was derived. For morbidity, hospital admissions, emergency room visits, and minor illnesses were quantified. Valuation was based on pain and suffering and cost of illness and loss productivity. Mortality from chronic exposure dominated the economic results, at over 99% of the total values.

A social discount rate of 5% was used after consultation with the Ontario Ministry of Energy.

B5 New Zealand – Air quality standards

Work undertaken in New Zealand included cost-benefit analysis of air quality proposals for air quality standards (National Environmental Standards (NES)) **(New Zealand Ministry for Environment, 2004)**. The study used risk factors for acute premature deaths (mortality) reported for the four major cities (Auckland, Christchurch, Wellington and Dunedin). It also estimated hospitalisations per year (e.g. asthma, bronchitis) and estimated restricted activity days (RADs) per year.

Premature deaths were assumed to be premature by 18 months. The benefits analysis included a monetary value for premature mortality, based on the value of a statistical life. It was adapted from Transfund (of NZ$2.5 million per fatality) and adjusted to reflect age, such that the 'value of life' figure was reduced to 75% of $2.5 million to reflect impacts on older members of the at-risk population. This gave a value per premature death of NZD1.88 million. The mortality value was discounted at a real rate of 10% per year and assumed growth in real gross domestic product (GDP) per capita of 1.5% per year.

The work was updated by **Clough *et al.* (2009)**[27]. The updated report reviewed the 2004 study and recommended a number of changes. It concluded that the benefits were likely to be underestimated because of the change in evidence (notably chronic effects). The updated study includes infant mortality in addition to the premature mortality cases. It did not include mortality from chronic exposure or other health endpoints, though these were discussed. However, the study adjusted the VSL to a full value, without any adjustment of the values to take account the predominance of effects in the elderly. It used VSL of NZ$3.35 million. It also used an 8% discount rate to reflect the change in NZ Treasury guidance.

[27] http://www.mfe.govt.nz/publications/air/national-air-quality-standards-nzier/index.html

APPENDIX C

Valuation studies in Australia

C1 Study by Beer (2002)

Beer (2002) carried out a valuation of pollutants, including PM, emitted by road transport based upon published assessments of Australian health impacts from air pollution and estimates of total air emissions from road transport. A valuation of the health benefits of reducing transport air pollutant emissions was developed and compared with assessments reported elsewhere. Transport emissions were estimated by combining Australian emission factors and estimates of vehicle-kilometres travelled (VKT) in 1999. Emissions to PM concentration estimates were not available and an interim approach was developed that used estimates of the percentage of PM that arise from road transport. Published Australian transport-related health costs were then divided by the estimated total transport emissions to provide a best estimate valuation for PM_{10}.

C2 Commonwealth Fuel Taxation Inquiry

Watkiss (2002) transferred damage costs from other countries as a straightforward way of providing values for health impacts in Australia. An earlier set of damage costs from the EC ExternE Transport study (2001) (a forerunner of the EC CAFE approach) was used. **Figure C1** provides an overview of the method for deriving the damage costs for Australia.

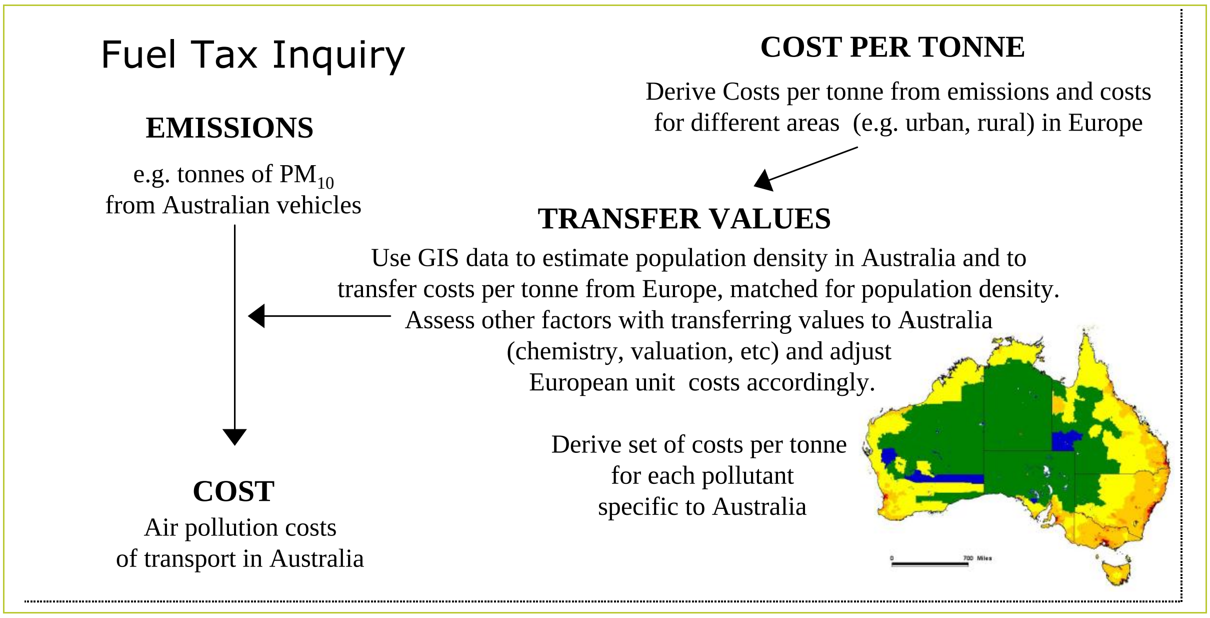

Figure C1: Deriving unit costs for use in Australia (Watkiss, 2002)

The ExternE approach quantified mortality due to acute exposure (deaths brought forward), mortality from chronic exposure (life expectancy), a number of acute morbidity (from respiratory hospital admissions through to minor restricted activity days) and chronic morbidity impacts. Valuation was based on WTP. Mortality was based on VSL (~A$6 million) but adjusted to reflect years of life lost.

For the Australian analysis the study transferred unit values (costs per tonne) based on local damages for similar population density sites in Europe. An assessment of the relative population densities in Australia was undertaken using the population distribution in major Australian cities. Interestingly, the average population densities in the central districts of Australian capital cities were found to be similar to those of cities in Europe. This match extended to the very high densities found in some central areas.

However, there were also some differences between Europe and Australia. The population densities in the outer areas of major cities are much lower in Australia. Moreover, once the capital cities are excluded, most urban areas in Australia have smaller populations and have lower population densities (*i.e.* urban areas are generally spread over larger areas than in Europe). Not surprisingly, the population density in remote rural locations was much lower in Australia.

The study used four different area categories, each of which has a different set of air pollution costs (**Table C1**).

- Band 1. Inner areas of larger capital cities (Melbourne, Sydney, Brisbane, Adelaide and Perth).
- Band 2. Outer areas of larger capital cities.
- Band 3. Other urban areas. Includes other capital cities (Canberra, Hobart and Darwin), and other urban areas.
- Band 4. Non-urban areas.

Table C1: Unit damage cost values for PM_{10} (Watkiss, 2002)

Unit Values A$/tonne (2010 prices)			
Band 1	**Band 2**	**Band 3**	**Band 4**
$427,155	$116,500	$116,500	$1,550

Converted using approximate total change in cost from year 2002 of 25.0%, over 8 years, at an average annual inflation rate of 2.8%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

C3 Commonwealth Fuel CBA

For the Commonwealth Fuel CBA, **Coffey (2003)** used scaled emission reductions to predict air quality for some pollutants but adopted marginal abatement benefit values ($/tonne avoided) for VOCs reductions. The Commonwealth Fuel CBA was an assessment of a potential policy measure, and provided a basic estimate of the resulting ambient air quality, for PM. For mortality, the joint C-R function estimate for mortality due to chronic exposure to PM_{10} from **Kunzli *et al.* (1999)** of 1.043 (1.026 to 1.061) for a 10 $\mu g/m^3$ increase in PM_{10} was adopted. The joint estimate was obtained by scaling the $PM_{2.5}$ estimate from the ACS study **(Pope *et al.*, 1995)** to PM_{10}.

For morbidity, the study included the relationships for respiratory hospital admissions, cardiovascular hospital admissions, acute bronchitis, chronic bronchitis, asthma, restricted activity days, and emergency department attendance (local).

The report summarises the functions adopted as below (**Table C2**).

For valuation, the study used a value of A$5,000,000 for an estimated cost of mortality as this was the approximate median of the reviewed values.

The study also made an assessment of the average health savings per tonne of national transport emissions. For one tonne of national emissions (carbon monoxide, oxides of nitrogen, hydrocarbons and PM were each considered separately) the health saving in each capital city was estimated. The change in concentration of the target air quality parameter was assessed for each city. Coffey found the on-road and total emissions in each air shed, and assumed the resultant ambient air concentrations were linearly related to the reduction in overall particle emissions.

Table C2: Summary of adopted risk estimates for air pollutants and air toxics (Coffey, 2003)

Health Outcome	Source	Applied Population	Exposure Response Relationship-Central Estimate
PM_{10} (for 10µg/m^3 change)			
Mortality (Long term exposure effects) All deaths excluding violent deaths/accidents	Kunzli *et al.* (1999)	Adults >30 years	1.043
Mortality (Short term exposure effects) All deaths excluding violent deaths/accidents	WHO (2000)	All ages	1.0074
Respiratory Hospital Admissions (ICD9 460-519)	WHO (2000)	All ages	1.008
Cardiovascular Hospital Admissions (ICD9 390-459)	Kunzli *et al.* (1999)	All ages	1.009

The health cost savings were assessed for each city based on the projected population for 2015. Health cost estimates were limited to mortality and hospital admissions as there was insufficient information for prediction of less severe impacts and such impacts were expected to be comparatively small. The average capital city health benefit for reductions of PM emissions in a capital city are presented below.

Table C3: Emission type air quality impact health savings (A$/tonne 2010) (Coffey, 2003)

Pollutant	Health Savings (A$/tonne), 2010 prices
Particulate matter (PM_{10})	$282,243

Converted using approximate total change in cost from year 2003 of 21.7%, over 7 years, at an average annual inflation rate of 2.8%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

The report noted that the savings per tonne of emissions varied from location to location according to the population and meteorological factors.

C4 Health Costs of Air Pollution in the Greater Sydney Metropolitan Region

The **DEC (2005)** derived local specific PM damage costs from the NSW GMR (defined as Hunter, Sydney and Illawarra). The study evaluated health impacts of a range of air pollutants. However, it followed **Kunzli *et al.* (1999)** in using PM_{10} as the index pollutant of the health impacts to avoid double counting. Health impacts were estimated using a PM_{10} threshold of 7.5 µg/m^3 and using no threshold.

Table C4 provides the health endpoints and associated concentration-response estimates used in the study.

Table C4: PM_{10} health endpoints and concentration-response estimates for the NSW GMR (DEC, 2005)

Health endpoint	Average exposure-response estimates for $10 \mu g/m^3$ change in PM_{10}
Mortality (long-term)	Low: 2.6% Central: 4.3% High: 6.1%
Respiratory hospital admissions	Low: 0.5% Central: 0.8% High: 1.1%
Cardiovascular hospital admissions	Low: 0.6% Central: 0.9% High: 1.3%
Asthma attacks (<15 years)	Low: 2.7% Central: 4.4% High: 6.2%
Asthma attacks (>15 years)	Low: 0.0% Central: 0.4% High: 0.8%
Restricted activity days	Low: 7.9% Central: 9.4% High: 10.9%
Acute bronchitis (<15 years)	Low: 13.5% Central: 30.6% High: 50.2%
Chronic bronchitis	Low 0.9% Central: 9.8% High: 19.4%

The population attributable risk method was used to estimate mortality impacts with low and high VSL estimates of A$1 million and A$2.5 million applied to estimate costs. Mortality was the main driver of total health costs.

In deriving damage costs, total regional health costs were divided by the total anthropogenic emissions of PM_{10} in each region. This method gave an indication of the health cost associated with a tonne of PM_{10}, shown below in **Table C5**. Thee value found was A$236 000 per tonne for Sydney, falling to A$47,000 for Illawarra.

Table C5: Costs per tonne of PM_{10} in the NSW GMR (estimated without a threshold) (A$2010)

Region	Low	High	Midpoint
Cost per tonne of PM_{10} ($ thousand)			
Sydney	55	521	288
Hunter	16	137	77
Illawarra	12	104	57

C5 Review of Euro 5/6 Light Vehicle Emissions Standards

The NSW Department of Infrastructure and Transport undertook a review of health benefits as part of a Regulatory Impact Statement for adopting the latest Euro 5 and Euro 6 emission standards for light vehicles, and their capacity to deliver significant emission reductions **(DIT, 2010)**.

The study reviewed eight existing studies. However, only the three most recent studies were selected as input for estimation **(Coffey, 2003; Watkiss, 2002; Beer, 2002)**. An 'avoided health cost' approach was used, whereby monetary values (in $/tonne) were assigned to individual pollutants (in this case HC, NO_x and PM). To calculate the total health benefit, the study split total emissions into those for capital cities and those for the rest of Australia, then derived and applied unit values. Unit values for capital cities were calculated by taking the simple average of the estimates from the three studies. Unit values for the rest of Australia were based on the simple average of the estimates for Band 3 and Band 4 contained in **Watkiss (2002)**.

Given the uncertainties surrounding the unit value estimates, an upper bound and a lower bound were established (an average ±50%) on the basis of observations made by **Coffey (2003)** and unit values were updated to 2009 prices using the Consumer Price Index (CPI). **Table C6** presents the unit cost values for calculating the health benefit and undertaking sensitivity analyses.

Table C6: Average health costs (A$/tonne) by geographical area (2010 prices)

Area & Sensitivity	PM_{10} Health Costs
Central	
Capital cities	A$241,955
Rest of Australia	A$57,415
Upper bound +50%	
Capital cities	A$362,932
Rest of Australia	A$86,123
Lower bound -50%	
Capital cities	A$120,977
Rest of Australia	A$28,707

Converted using approximate total change in cost from year 2009 of 2.8%, over 1 years, at an average annual inflation rate of 2.8%. Source: Reserve Bank of Australia Inflation Calculator http://www.rba.gov.au/calculator/annualDecimal.html

C6 Reducing emissions from non-road spark ignition engines and equipment

This consultation regulation impact statement (RIS) conducted by the Non-roads Engines Working Group examined whether there was a case for government action to reduce adverse impacts of non-road spark ignition engines and equipment on human health and the environment.

The study consisted of an assessment of health costs arising from lifetime emissions of air pollutants from non-road spark ignition engines in Australia. The study found that health costs arising from the use of non-compliant engines were far higher than those for engines compliant with overseas standards. The study concluded that a significant reduction in heath costs arising through air pollutant emissions from non-road spark ignition engines and equipment can be achieved through regulating the market.

A number of options to reduce emissions from non-road spark ignition engines and equipment were assessed in this RIS. Three options to deliver national emission standards were assessed for costs and benefits:

- A voluntary industry agreement, including sales target – outboard engines only
- Commonwealth regulation – all specified garden and marine engines/equipment
- National Environment Protection Measure (NEPM) – all specified garden and marine engines/equipment

The emission standards proposed for adoption in Australia are the most recent US standards, promulgated in 2008.

Two scenarios were assessed for impacts for each of the three feasible delivery options:

1. A 15% sales target to be met in 2020 and 2012 for the voluntary outboard industry option.
2. A phased (two-step implementation) and non-phased (one-step implementation) approach for each of the Commonwealth regulation and NEPM options.

A simplified explanation of the method to estimate the costs and benefits arising from the use of stock engines follows. For the more detailed analysis refer to **(Non-Road Engines Working Group, 2010)**. Firstly, the number of engines and age profiles was estimated. Then, the resultant yearly emissions were calculated using relevant USEPA emission factors. The total emissions arising from the entire stock of engines was the aggregate of emissions from each individual engine.

The costs were then calculated using data from The CAFE CBA **(AEA Technology 2005)**, which reported cost per tonne of emissions. For $PM_{2.5}$ a value of A$82,490/tonne of emission was used. For PM_{10} a range of A$324,000 to $A6,795/tonne of emissions was used. The study assumed that a linear relationship existed between the tonnage of emissions and health impacts. The study also noted that the impacts of emissions are directly related to the population size exposed to the emission.

The annual costs were then discounted back to 2008 and then the costs and benefits of the given legislation were calculated.

APPENDIX D

International studies on secondary PM

D1 Overview

Secondary particles have been quantified in various studies around the world, and some of these studies are summarised below. The composition of airborne PM is often stated in relation to the two most commonly used metrics – PM_{10} and $PM_{2.5}$, with the latter being more relevant to secondary particles on account of their small size. The emphasis here is on studies in Europe and the US, as this is where most of the health impact and valuation work has been conducted. Australian studies are also summarised.

D2 European studies

Putaud *et al.* (2010) carried out a detailed analysis of data on the chemical composition of PM_{10}, $PM_{2.5}$ and PM_{COARSE} (PM_{10} minus $PM_{2.5}$) from various locations across Europe. It was found that the main constituents of $PM_{2.5}$ (and PM_{10}) in Europe were generally organic matter, sulphate and nitrate. The $SO_4^{2-}/PM_{2.5}$ and $NO_3^{-}/PM_{2.5}$ ratios in southern Europe were significantly lower than in other regions of Europe. Differences in PM chemistry between types of site were also observed. The contributions of both sulfate and nitrate to PM_{10} generally decreased when moving from rural to kerbside sites. The same was observed for nitrate in PM_{COARSE} and for sulfate in both $PM_{2.5}$ and PM_{COARSE} in central Europe. The contribution of organic matter to $PM_{2.5}$ was similar at all types of site.

Putaud *et al.* (2010) also observed that in Europe there are still very few sites where all the major constituents of PM_{10} and $PM_{2.5}$ have been measured over time periods long enough to obtain representative averages. Another issue is the lack of control on data comparability. There are currently no reference methods for measuring aerosol characteristics, except for PM mass concentrations. There are therefore significant sampling and analytical artefacts.

Some examples of findings in specific European countries are given below.

In an investigation at background sites in the UK **Turnbull and Harrison (2000)** found that secondary particles contributed 28-35% of PM_{10}. **Charron *et al.* (2007)** determined that the regional background was the largest contributor to PM_{10} concentrations measured alongside the heavily trafficked Marylebone Road in London. Particulate nitrate constituted the largest part of the secondary aerosol, especially during air pollution episodes.

At urban background and rural background sites in the UK $PM_{2.5}$ is composed predominantly of ammonium sulfate, ammonium nitrate and organic carbon. Background $PM_{2.5}$ concentrations are therefore dominated by secondary PM rather than local sources **(Laxen *et al.*, 2010)**. The distribution of secondary $PM_{2.5}$ (organic and inorganic) across the UK is shown in Figure 7-1. Concentrations decline from 6 µg/m^3 in the south east of England to less than 2 µg/m^3 in the north-west of Scotland. **Laxen *et al.* (2010)** note that this demonstrates a strong trans-boundary contribution to $PM_{2.5}$ from mainland northern Europe. It also means that the secondary contribution to $PM_{2.5}$ varies from between 31% in urban Scotland to almost 60% in rural south-east England. Given the relative rates of formation, the continental contribution is likely to be higher for secondary sulfate than for secondary nitrate **(AQEG, 2005)**.

Specific studies have confirmed these values. **Yin *et al.* (2010)** found that around 60% of the $PM_{2.5}$ at an urban background site in Birmingham and a rural site 20 km to the west was of secondary origin. **Allan *et al.* (2010)** sampled urban background air in London and Manchester during the winter of 2006/07, and applied positive matrix factorisation to determine sources. SOA accounted for 28-53% of the organic aerosol. It was noted that SOA would be likely to dominate during the summer months. **Heal *et al.* (2011)** used radiocarbon analysis to identify sources of $PM_{2.5}$ at an urban background site in Birmingham. The proportion of total $PM_{2.5}$ at this location estimated to be biogenic SOA was 9–29%. The findings from this work were considered to be consistent with those from elsewhere in Europe, and support the conclusion of a significant and ubiquitous contribution from non-fossil biogenic sources to the carbon in terrestrial aerosol.

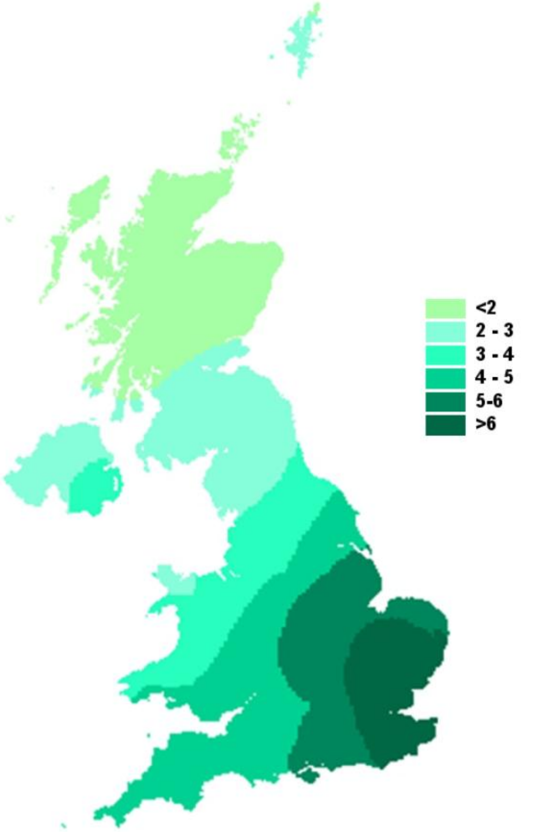

Figure 6-1: Annual mean secondary $PM_{2.5}$ concentrations (µg/m^3) in 2010 (Laxen *et al.*, 2010)

Yin *et al.* (2005) investigated the composition of PM_{10} and $PM_{2.5}$ at five different monitoring sites (roadside, urban, rural and coastal) in Ireland. In urban areas, the major components of $PM_{2.5}$ were organic compounds (~30%), elemental carbon (23–33%), and ammonium sulphate/nitrate (17–29%). In comparison, at the rural and coastal sites $PM_{2.5}$ mainly consisted of ammonium sulfate/nitrate (42% and 44%) and organic material (23% and 21%). Urban site PM_{10} also consisted of a larger fraction of carbonaceous aerosols, as well as a significant proportion of re-suspended materials due to the influence of road traffic. Sea salt and secondary components were predominant at non-urban sites. The average contributions of total organic material to the $PM_{2.5}$ fraction in urban areas were 27% in winter and 37% in summer, due to increased production of secondary organic compounds in summer. In comparison, a lesser contribution (18% in winter and 25% in summer) was observed at non-urban sites, indicative of road transportation in urban areas being a significant source of organic material.

Matthijsen and ten Brink (2007) estimated that more than 75% of the regional $PM_{2.5}$ in the Netherlands was composed of secondary inorganic constituents (sulfate, nitrate and ammonium), elemental carbon and organic carbon. SOA was not included in the calculations, but its contribution to $PM_{2.5}$ concentrations was expected to be less than 1 µg/m^3.

Viana *et al.* (2007) sampled $PM_{2.5}$ at five urban Spanish locations during the European Community Respiratory Health Survey II (ECRHS II). In an attempt to identify and quantify

$PM_{2.5}$ sources, principal component analysis (PCA) was performed on five datasets containing elemental composition of $PM_{2.5}$. Secondary aerosols accounted for 14–34% of $PM_{2.5}$ (2.6–4.5 µg/m^3).

The composition of PM_{10} and $PM_{2.5}$ has been investigated at urban background, roadside and tunnel sites in Milan, Italy **(Lonati *et al*., 2005; Giugliano *et al.*, 2005)**. The results pointed to a strong contribution from secondary sources. Organic and inorganic secondary material contributed 75% (50% inorganic, 25% SOA) of $PM_{2.5}$ mass in winter and 40% (30% inorganic, 10% SOA) in summer. For PM_{10} the secondary sources accounted for about 25% of the total mass in summer and up to 55% in winter.

In Zurich, Switzerland, **Szidat *et al.* (2006)** found that secondary aerosols with biogenic sources were responsible for 25% of organic carbon during winter and 48% during summer.

In the Helsinki metropolitan area **Koistinen *et al.* (2004)** reported a 46% contribution of secondary aerosols to $PM_{2.5}$ concentrations.

D3 US Studies

Many source apportionment studies have been conducted in the United States. These studies were reviewed by **USEPA (2009)**, and are therefore not repeated here. However, the analysis of data from various monitoring sites across the United States is instructive, and is summarised below.

Figure 6-2 shows the $PM_{2.5}$ compositional breakdown at sites across the country. The data are for at least one calendar year between 2005 and 2007. On an annual average basis, sulfate was a dominant PM component in the eastern US cities. For cities east of Houston the sulfate fraction of $PM_{2.5}$ ranged from 42% (Chicago) to 56% (Pittsburgh) on an annual average basis. Organic carbon mass (OCM) was the next largest component in the east, ranging from 27% in Pittsburgh to 42% in Birmingham. In the west, OCM was the largest constituent on an annual basis, ranging from 34% in Los Angeles to 58% in Seattle. Sulfate, nitrate and crustal material were also important in many of the western cities. In the west, the sulfate component ranged from 18% (Denver) to 32% (Los Angeles). The nitrate component was relatively large in Riverside (22%), Los Angeles (19%) and Denver (15%) but nitrate was less important on an annual basis in Phoenix (1%) and Seattle (2%). EC made up a small fraction of the $PM_{2.5}$ (4-11%), but it was consistently present in all included cities regardless of region **(USEPA, 2009)**. The seasonal variation in $PM_{2.5}$ composition was also examined in the USEPA review. Sulfate generally dominated $PM_{2.5}$ in most metropolitan areas in the summertime, while nitrate generally became important in the colder wintertime months.

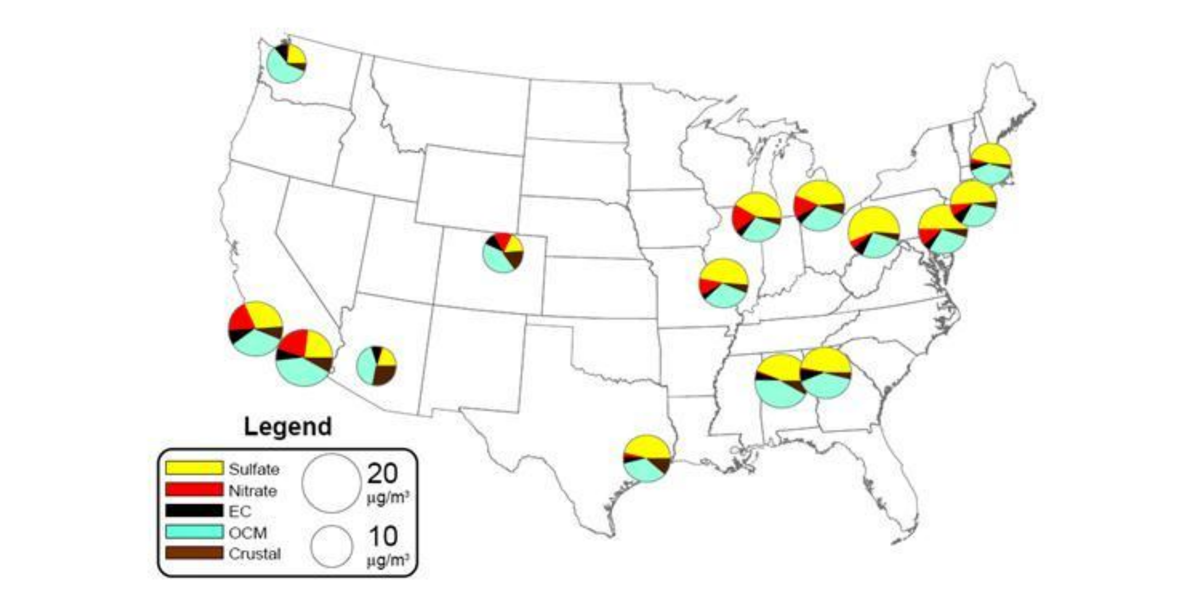

Figure 6-3: Average $PM_{2.5}$ speciation profiles for 2005-2007 in the United States. SO_4^{2-} and NO_3^- estimates include NH_4^+ and particle-bound water, and the circled areas are scaled in proportion to $PM_{2.5}$ mass (USEPA, 2009).

Ying *et al.* (2009) identified diesel engines as the largest source of secondary nitrate in central California, accounting for approximately 40% of the total $PM_{2.5}$ nitrate. Catalyst-equipped gasoline engines were also significant, contributing approximately 20% of the total secondary $PM_{2.5}$ nitrate. Agricultural sources were the dominant source of secondary ammonium. Sharp gradients of PM concentrations were predicted around major urban areas. The relative source contributions to $PM_{2.5}$ of different source categories varied with location due to the dominance of primary organic carbon in urban locations and secondary nitrate in rural areas.

Six-year trends in $PM_{2.5}$ constituents across the United States from the **USEPA (2009)** review are shown in Figure 7-3. Two seasons representing different temperature ranges – cool (October-April) and warm (May-September) – were considered since many $PM_{2.5}$ components exhibit a strong seasonal dependence. There were no trends in the sulfate, EC and crustal components in any of the regions or seasons. A slight decline in OC was observed for the Northeast during warm months and in Southern California year-round. The largest decreases were for nitrate in Southern California, and smaller decreases were observed for some other regions.

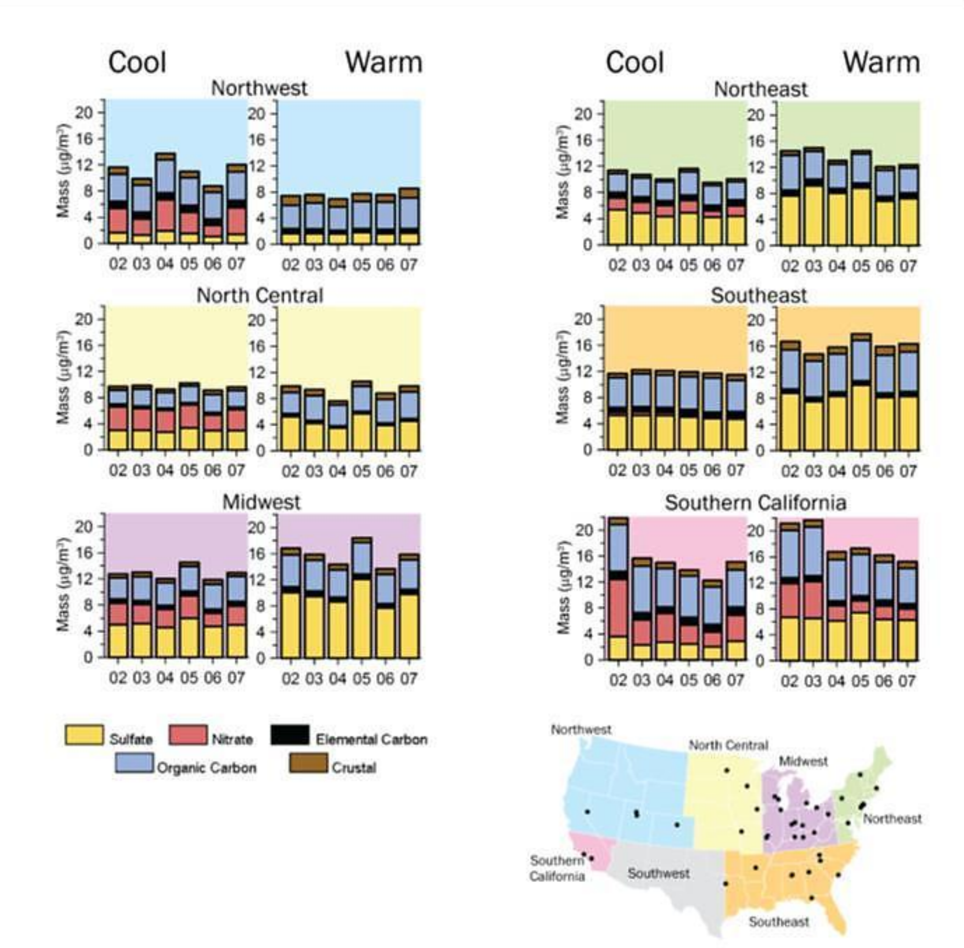

Figure 6-4: Regional and seasonal trends in annual $PM_{2.5}$ composition from 2002 to 2007. Data are shown separately for cool months (October-April) and warm months (May-September) (USEPA, 2009).

Some studies have specifically addressed SOA. In Los Angeles, **Docherty *et al*. (2008)** found that SOA comprised 70–90% of organic aerosol during midday periods and 45% of organic aerosol during peak traffic periods. The authors added that the SOA proportion had increased with time, possibly a result of primary organic aerosol having decreased due to targeted policies such as vehicle emission controls rather than a reduction of SOA precursors. In the Pittsburgh Air Quality Study, **Cabada *et al.* (2008)** estimated that around 35% of the organic carbon concentration in Western Pennsylvania during July of 2001 was estimated to be secondary in origin. **Chen *et al.* (2009)** predicted source contributions to secondary organic aerosol (SOA) formation in the San Joaquin Valley. Predicted SOA concentrations at Fresno, Angiola, and Bakersfield were 2.46 µg/m^3, 1.68 µg/m^3, and 2.28 µg/m^3, respectively, accounting for 6%, 37%, and 4% of the total predicted organic aerosol. The average SOA concentration across the entire valley was 1.4 µg/m^3, and SOA accounted for 20% of the total predicted organic aerosol. The major SOA sources were solvent use (28% of SOA), catalyst gasoline engines (25% of SOA), wood smoke (16% of SOA), non-catalyst gasoline engines (13% of SOA), and other anthropogenic sources (11% of SOA).

SOA has been observed in aged biomass burning plumes in the US. **Lee *et al.* (2008)** identified elevated $PM_{2.5}$ concentrations when a smoke plume from prescribed-burning affected Atlanta. Source apportionment suggested that the $PM_{2.5}$ had a significant fraction of secondary components, and included high molecular weight compounds that were likely to have low volatility.