Mercury Sources, Transportation and Fate in Australia

Final Report to the Department of Environment, Water, Heritage & the Arts
RFT 100/0607

Peter F Nelson, Hao Nguyen, Anthony L Morrison and Hugh Malfroy
Graduate School of the Environment
Macquarie University

Martin E Cope, Mark F Hibberd, Sunhee Lee, John L McGregor, Mick (CP) Meyer
The Centre for Australian Weather and Climate Research
A partnership between CSIRO and the Bureau of Meteorology

December 2009
(Revised April 2011)

About the Authors

Peter Nelson is Professor of Environmental Studies, Associate Dean (Research) Faculty of Science, and Head of the Graduate School of the Environment at Macquarie University.

Hao Nguyen is a Research Fellow in the Graduate School of the Environment, Macquarie University.

Anthony Morrison is a Senior Research Fellow in the Graduate School of the Environment, Macquarie University.

Hugh Malfroy is Director, Malfroy Environmental Strategies Pty Ltd and a Visiting Fellow in the Graduate School of the Environment, Macquarie University.

Martin Cope, Mark Hibberd, Sunhee Lee, John McGregor, and Mick Meyer are research scientists with the Centre for Australian Weather and Climate Research, a partnership between CSIRO and the Bureau of Meteorology.

Summary

BACKGROUND

Mercury is among the most bio-concentrated trace metals in the food chain. It is a naturally occurring metal found in small quantities throughout the environment in both the atmosphere and in aquatic and terrestrial ecosystems. While it is continuously released, transported, transformed and stored in and between these compartments, the atmosphere is considered to be the dominant transport medium of mercury in the environment.

At the *24th Session of the United Nations Environment Program Governing Council/ Global Ministerial Environment Forum* in 2007 it was concluded that:

- "current efforts to reduce risks from mercury are not sufficient to address the global challenges posed by mercury", and
- "further long-term international action is required to reduce risks to human health and the environment and that, for this reason, the options of enhanced voluntary measures and new or existing international legal instruments will be reviewed and assessed in order to make progress in addressing this issue."

On 20th February 2009 the UN Environment Programme's (UNEP) Governing Council agreed on a plan for a global approach to reduce population and ecosystem exposure to mercury. The landmark decision, taken by over 140 countries, sets the stage for the development of an international mercury treaty to deal with world-wide emissions and discharges of this pollutant. The Council also agreed that the risk to human health and the environment was so significant that accelerated action under a voluntary Global Mercury Partnership is needed whilst the treaty is being finalised.

In conjunction with the 2009 UN process, revised estimates (in 2008) of global emissions of mercury have been made. These estimates reveal that:

- The largest sectoral source is the combustion of fossil fuels, largely coal. This sector accounts for a total of ~46% of emissions of mercury to atmosphere, about 25% from electrical power plants and 20% from industrial and residential heating.

- Emissions from gold production arises from both large scale industrial production (~6% of total global emissions) and from small scale and artisanal gold mining and production (~18%, and largely in developing countries).
- The mining, smelting and production of metals other than gold, and cement production each account for ~10% of global emissions.
- The emission **estimates are subject to large uncertainties,** largely due to lack of data, uncertainty in the data that are available, and a reliance on data from other locations.

To date, there has been little systematic, coordinated effort to understand the nature of mercury emissions in Australia and as such there is significant uncertainty in our current understanding of the sources, fate and impacts of mercury in Australia. These uncertainties include:

- emission source strengths from stationary sources in Australia;
- emissions from natural sources (eg, bushfires, water bodies and vegetation), and re-emission of previously deposited mercury; and
- the relative contributions of the different chemical forms of mercury (ie, elemental, oxidised and particulate) in many sources.

In an effort to address these uncertainties and to improve the understanding of mercury in Australia, the Department of the Environment, Water, Heritage & the Arts (DEWHA) commissioned Macquarie University and CSIRO to carry out a study to determine the sources, transportation and fate of mercury in Australia. The study has six parts:

- Collection of Data on Mercury Emissions, Sources and Trends from Anthropogenic and Non-Anthropogenic Sources (part A)
- Study of the transport and fate of mercury in Australia (part B)
- The identification of gaps in the scientific data related to mercury in Australia. (part C)
- The identification of areas or populations especially at risk from mercury in Australia (part D)
- The collation of information into an inventory of mercury sources and emissions in Australia (part E)
- Study of the availability, efficiency and costs of control technologies (part F)

This Final Report addresses all parts of the study’s brief.

RESULTS

Anthropogenic emissions

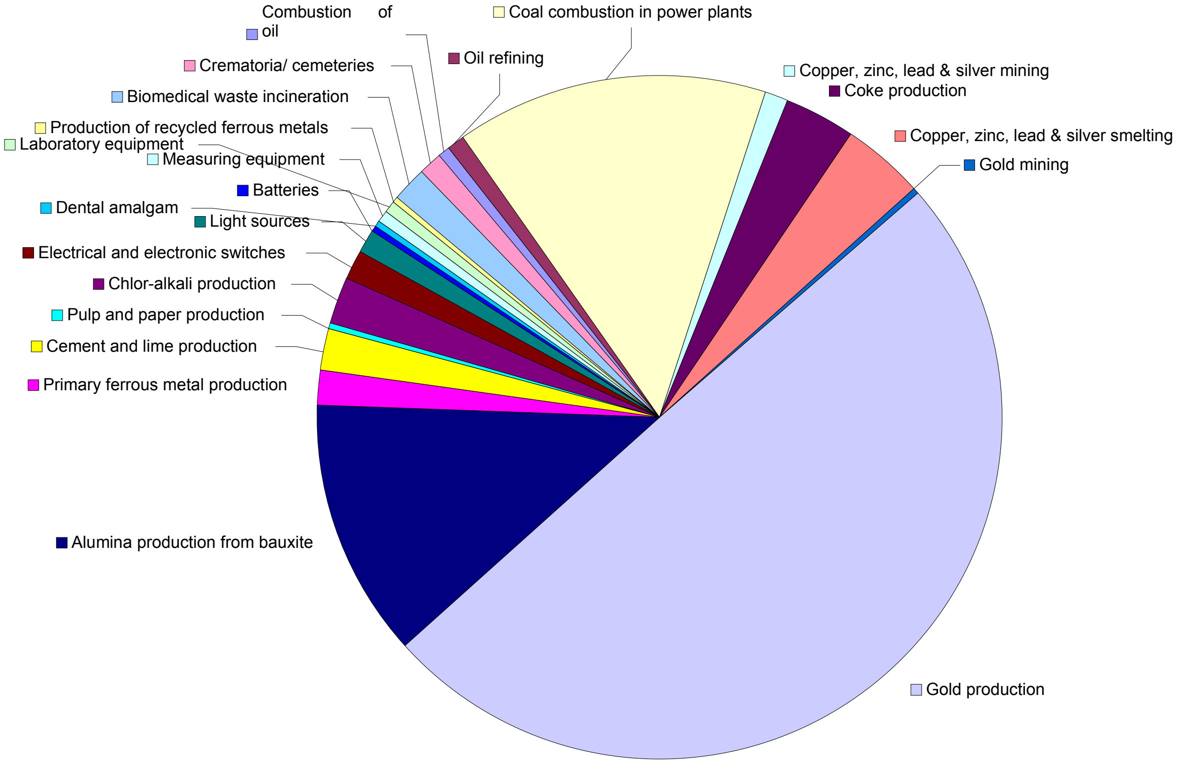
Derivation of an inventory of Australian emissions of mercury from anthropogenic sources in 2006 was undertaken using a range of data sources. These included the National Pollutant Inventory (NPI), and overseas protocols and emission factors (eg, those included in the *UNEP Toolkit for identification and quantification of mercury releases*). There is considerable uncertainty in the emission estimates so obtained, not least because of the very high reliance on overseas sources of information, assumptions and emission factors. Hence the mercury emission inventory should be used with caution, and the impacts predicted using it should recognise the limitations which these uncertainties impose on any conclusions or decisions which may be based on data from the emission inventory. Notwithstanding the preceding note of caution, it is considered that the new inventory represents a significant advance upon previous data, will enable qualified assessments to be undertaken (as is done in parts of this study) and provides a platform for further improvement with advances in knowledge and as resources permit.

The following conclusions may be made about the estimated emissions of mercury to the atmosphere from Australian anthropogenic sources:

- The best estimate of total emissions of mercury to the atmosphere in 2006 was around 15 tonnes. Using a very different methodology the most recent global emission estimate (in 2008) reports total anthropogenic emissions from Australia at ~34 tonnes/year. The difference between the two methods is largely due to a much higher estimate for emissions from stationary combustion in the global estimate. It can be convincingly argued that the estimate presented in the current report for stationary combustion (largely coal-fired power stations) is more accurate as it uses NPI reported emissions which incorporate estimates of mercury capture in air pollution control devices (the global estimate does not include any mercury capture), and is supported by a comparison of top-down and bottom-up estimates of mercury from Australian stationary sources.
- Three sectors contribute substantially to Australian anthropogenic emissions; these are gold production (49.7%), coal combustion in power plants (14.8%), and alumina production from bauxite (12.2%).
- A range of other diverse sectors contribute smaller proportions of the emitted mercury. These include industrial sources (mining, smelting, and cement production), and intentional use of mercury in products.
- It is difficult to determine historical trends in mercury emissions given the large uncertainties in the data. Past historical data is likely to be even more uncertain. However it is clear that the intentional

use of mercury in products is in decline. In addition all mercury cell-based chlor-alkali plants in Australia have now ceased operation, and emissions from this source have decreased significantly since this time.

The figure and table below summarise the relative contribution of sources and sectors to Australian anthropogenic emissions to the atmosphere in 2006.



Natural emissions

Mercury is a naturally occurring substance in a variety of environmental media and hence it is also emitted from vegetation, soil, water bodies and during fires. It is believed that a large part (up to 50 percent) of the mercury that is emitted from natural sources is actually of anthropogenic origin (Mason *et al.* 1994a) that is “re-emitted” from natural sources after having previously been emitted from an anthropogenic source to the atmosphere or to a water body. Evaporation of mercury from the oceans’ surface, emission of mercury from soil, vegetation and the release of mercury in forest fires, are consequently a mix of natural and re-emitted mercury. It is clear that care needs to be taken when referring to natural emissions since the term "natural" in this context may be somewhat misleading. In the context of this report "natural emissions" will, by definition, also include re-emissions.

Estimates of mercury emissions from the natural sources in Australia are highly uncertain, due to both the large uncertainties inherent in estimating these emissions, and also to the lack of relevant Australian data. The magnitude of the mercury emissions released depends on a number of biological, chemical, physical and meteorological factors, of which few are fully understood, and many are subject to very large uncertainties.

Relative Contributions of anthropogenic sources of mercury emissions to the atmosphere in Australia in 2006

Sector	Emissions, kg/year	Proportion of Total Emissions (%)
Gold smelting	7642	49.7
Coal combustion in power plants	2271	14.8
Alumina production from bauxite	1872	12.2
Copper, zinc, lead & silver smelting	629	4.1
Coke production	500	3.2
Chlor-alkali production	340	2.2
Cement and lime production	313	2.0
Primary ferrous metal production	247	1.6
Biomedical waste incineration	236	1.5
Electrical and electronic switches	207	1.3
Light sources	177	1.2
Crematoria/ cemeteries	172	1.1
Copper, zinc, lead & silver mining	169	1.1
Oil refining	101	0.7
Combustion of oil	101	0.7
Measuring equipment	92	0.6
Laboratory equipment	80	0.5
Production of recycled ferrous metals	63	0.4
Dental amalgam	59	0.4
Batteries	36	0.2
Gold mining	29	0.2
Pulp and paper production	14	0.1
Total	15346	

Previous works estimated emissions of 117 -567 tonnes of mercury per year from land and water surfaces in Australia (Nelson *et al.* 2004). In this study a more detailed approach based on land and vegetation classifications resulted in an estimate of about 148 tonnes emitted annually from vegetation and soil but not including emissions from the ocean. The mercury released from these various natural sources is mainly in the form of elemental mercury, although small quantities of dimethyl mercury are also released (Lindquist *et al.* 1991; Schroeder and Munthe 1998b).

Fires are also an important but highly uncertain source of mercury, and emit elemental, divalent and particulate forms of mercury (Porcella *et al.* 1996). Two recent estimates of Australian emissions from this source of 129 and 19 ± 9 tonnes/year have been made, and in this study a detailed modelling approach results in an estimate of 41.8 tonnes annually.

The natural sources are estimated to contribute 93% of the mercury emitted annually in continental Australia, demonstrating that natural emissions in Australia are significant in comparison to anthropogenic emissions but also highly uncertain. Future research should address this uncertainty.

Transport and fate of mercury

This component of the study entailed the use of numerical meteorological and transport models and the air emissions inventory for mercury (as discussed above) to generate best estimates of annual average ambient mercury concentrations and wet and dry deposition mass. *Wet deposition* is the transfer of a substance, in this case mercury, from the atmosphere to the surface via precipitation. In this regard it should be noted that although elemental gaseous mercury is relatively insoluble, reactive gaseous mercury is very soluble and particulate mercury is readily scavenged by cloud water droplets (Seigneur et al. 2001). Thus it may be expected that the majority of the mercury mass deposited by precipitation will be in the form of reactive gaseous mercury and particulate mercury. *Dry deposition* refers to the transfer of gas and aerosol phase mercury to “sinks” on vegetation (such as leaf stomata), soil and water surfaces by atmospheric turbulence and molecular diffusion. For particulate mercury, deposition rates may also be enhanced by gravitational settling of the particles.

The modelling has been undertaken over three spatial scales - for the Australian continent; for the urban regions of Melbourne and Sydney; and for five significant point source emitter groups.

The modelling has included best available estimates of natural and anthropogenic sources to estimate total mercury concentrations and deposition loadings. The natural source group considers the emissions from soils, vegetation, water and fires. The anthropogenic source group includes industrial emissions as well as emissions from the commercial, domestic and transport sectors. A mercury concentration of about 1.3

nanograms[1] per cubic metre of air (ng m^{-3}) was included in the model calculations to represent a global background contribution.

Natural emissions were estimated to contribute 93% of total mercury emissions in Australia with soil emissions being the largest single source (66% of total), followed by fires (20%), and vegetation (4%). Industrial sources (6.8%) dominated the anthropogenic emissions with only 0.4% coming from commercial and domestic sources.

Annual average mercury concentrations at the continental scale were dominated by the global background (1.1–1.3 ng m^{-3}), with increases evident at the regional scale in the vicinity of fires and major industrial sources (up to 3 ng m^{-3}). A similar range of concentrations was estimated at the urban scale for Melbourne and Sydney. However, fine scale modelling predicted concentrations up to 10 times larger within the first few kilometres of several significant industrial sources.

The modelled concentration results are consistent with observations taken at Macquarie University (Nelson *et al.* 2009) and also with measured global background concentrations. The concentration results are also generally consistent with results reported in the USA, considering that emissions there are significantly higher than in Australia.

It is also noted that the highest predicted annual average atmospheric concentrations are well below the World Health Organisation guideline for atmospheric mercury of 1 microgram[2] per cubic metre of air (1 μg m^{-3} or 1000 ng m^{-3}).

Wet and dry deposition was also modelled at the three spatial scales. The highest wet deposition rates occur in regions of higher rainfall or regions of local elevated mercury concentrations due to anthropogenic sources or combinations of these two factors. In contrast, dry deposition is generally dominated by natural emissions and the continental background, although enhanced dry deposition masses occur within the vicinity of fires and significant industrial sources.

At the continental scale, wet deposition peaks of up to 5 μg m^{-2} yr^{-1} were predicted. The total mercury mass deposited by precipitation onto the Australian land mass is estimated to be about 1.8 t yr^{-1} which is equivalent to about 0.8 % of the total emissions from the region in a year.

---

[1] A nanogram is one billionth of a gram - 10^{-9}

[2] A microgram is one millionth of a gram – 10^{-6}

At the continental scale, dry deposition rates were generally less than 20 µg m^{-2} yr^{-1}, although values up to 70 µg m^{-2} yr^{-1} were predicted near the largest industrial source in Kalgoorlie. Dry deposition is calculated to contribute about 21 t yr^{-1} which is equivalent to about 10 % of the emitted mercury from the region in a year and about ten times higher than the wet deposition amount.

Modelling of deposition is subject to considerable uncertainty and the results reported should be treated with caution. The uncertainties which will affect deposition rates include the extent and location of rainfall events, cloud processes resulting in incorporation of mercury in rainwater, net deposition velocities for various forms of mercury, and the assumed proportion of emitted mercury in an oxidised (and hence soluble form). In addition, the modelling at urban and point source scale showed significantly higher wet deposition than the continental scale modelling. This may be as a result of the higher resolution of the urban modelling (3 km grid spacing vs. 25 km grid spacing for the continental modelling) as well as better resolution of the rain processes. In spite of these uncertainties it is likely that the majority of mercury emitted from Australian anthropogenic and natural sources is transported out of the Australian continental airshed, and is hence incorporated in the global mercury pool.

Control Options for mercury

Summary of major issues

- The use of mercury in products and processes can occur either intentionally or incidentally;
- The intentional use of mercury is declining in many countries through substitution of new mercury-free products and processes and UNEP suggests that substitution is now possible for virtually all products that use mercury. Australia benefits from the development of alternatives elsewhere;
- A recent report prepared for the European Commission identified several intentional uses of mercury at levels higher than previously expected or known; (porosimetry and as a catalyst in polyurethane production);
- The intentional use of mercury in some products is increasing, most notably in compact fluorescent lamps (and some other electronics) despite the fact that the amount of mercury per lamp has declined substantially;
- Work is progressing on mercury free alternatives to compact fluorescent lamps, but commercially available alternatives are not yet on the market;
- At this stage, one can only prescribe production/use of energy-efficient lamps with a minimum mercury-content, and collection and treatment of spent lamps (UNEP 2002);

- Mercury in lamps and other electronic products can be recovered and recycled. Data worldwide on recycling rates is patchy but a recent report prepared for the European Commission would suggest that rates are modest in the European Union (but may be better in individual countries);
- Programs aimed at reducing mercury use and at recovering mercury containing products need to be supported by strong education and outreach programs and even incentives if they are to be successful, particularly in the difficult to manage area of domestic waste;
- Dental amalgam is a major contributor to mercury in waste water treatment systems.
  - Even in the EU with a strong substitution program in some countries, dental use of mercury is expected to rise in coming years. Use of mercury in dentistry has declined very little in the USA in recent years;
  - Particulate mercury emissions from dental surgeries can be readily controlled. Amalgam separators are part of best practice guidelines in the industry and are mandated in a number of countries;
  - Viable alternatives to amalgam fillings are available for most applications but are not yet widely known or accepted in many countries, as practitioners generally find it easier to continue using the techniques with which they are most familiar (UNEP 2002);
- The incidental use of mercury occurs mainly via its occurrence in fuels (coal) and metal ores;
- It is technically possible to remove a high proportion of mercury from the flue gases of most, if not all, industrial processes;
- In western economies, some industries are required to achieve a high degree of mercury control, most notably those involved in the combustion / incineration of waste (domestic, medical, hazardous);
- A significant amount of work on the control of mercury from coal-fired power stations has occurred in the USA in response to proposed legislation which would require a 70% reduction over existing levels. Following court action, this legislation is currently being reviewed by the USEPA;
- A degree of mercury capture is already occurring from many facilities as a co-benefit of existing air pollution control devices. Co-benefit forms a significant part of the USEPA control strategy;
- While feasible, the reduction in mercury from coal combustion due to fuel switching and or fuel substitution is unlikely in the absence of strong regulatory measures and / or financial incentives;

- Reductions in mercury emissions may result as a co-benefit from national and international measures aimed at reducing greenhouse gas emissions, and particularly carbon dioxide, such as energy conservation and the generation of electricity from technologies with negligible or zero mercury emissions (gas, renewables);
- A voluntary program between environmental regulators and gold producers in Nevada, USA resulted in a reduction in mercury emissions to the atmosphere of about 80%;
- Mercury emissions from crematoria are increasing in some countries (more cremations, bodies with more mercury), leading to requirements to control emissions from this source. No jurisdiction has mandated the removal of dental amalgam from bodies prior to cremation;
- A range of technologies exist for the treatment of soil, waste and water contaminated with mercury. New technologies are also being developed; and
- "*As is the case with the other management and policy options, it is important to consider both the potential reductions (and hence benefits) and the costs of the options. Any consideration of potential reductions should examine whether (and the extent to which) emissions reductions from the particular sources in question will yield reductions in risk to public health and the environment.*" (USEPA 1997a).

Future Work

The results of this study suggest a range of future activities to improve our knowledge of the sources, transport and fate of mercury in Australia:

- Efforts that could be made to reduce the uncertainties in emission estimates include:
    - The collection of more and higher quality local industrial emissions data to reduce the heavy reliance on international data sources (particularly from the USA). This could be pursued under the framework of the National Pollutant Inventory (NPI).
    - Area and diffuse commercial and domestic sources of mercury are similarly poorly understood, and the available data is inconsistent and / or inaccurate. Consistent, higher quality data should be collected.
    - A targeted research program to address the main uncertainties in estimating natural sources of mercury - vegetation, soils, water bodies and fires. These appear to exceed anthropogenic emission sources but are subject to large uncertainties.

- There is very little data on atmospheric mercury concentrations or deposition rates of mercury in Australia (and in the southern hemisphere in general). Initial results from the Macquarie team (Dutt *et al.* 2009; Nelson *et al.* 2009) are addressing this data gap but more comprehensive measurements of concentrations and deposition at representative sites are required. Modelling results provide a useful guide to the selection of suitable sites. These measurements are significant, since as a global pollutant improved knowledge of mercury concentrations and deposition in the southern hemisphere would enable more rigorous tests of global atmospheric models for mercury transport and fate.

Table of Contents

Appendix 1: Cope, M. E., Hibberd, M. F., Lee, S., Malfroy, H. R., McGregor, J. R., Meyer, C. P., Morrison, A. L., Nelson, P. F. (2009). The Transportation and Fate of Mercury in Australia: Atmospheric Transport Modelling and Dispersion. Appendix 1 to Report RFT 100/0607 to Department of Environment, Water, Heritage & the Arts, The Centre for Australian Weather and Climate Research, 60 pp.

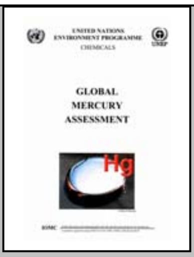
List of Figures

List of Tables

1 INTRODUCTION AND BACKGROUND TO PROJECT

1.1 International developments

Mercury is among the most bio-concentrated trace metals in the food chain. It is a naturally occurring metal found in small quantities throughout the environment in both the atmosphere and in aquatic and terrestrial ecosystems. While it is continuously released, transported, transformed and stored in and between these compartments, the atmosphere is considered to be the dominant transport medium of mercury in the environment (Fitzgerald *et al.* 1991; Lindquist *et al.* 1991).



UNITED NATIONS ENVIRONMENT PROGRAMME
CHEMICALS

GLOBAL
MERCURY
ASSESSMENT

The UN Global Mercury Assessment

"The UNEP Governing Council concluded after considering the key findings of the Global Mercury Assessment report, that there is sufficient evidence of significant global adverse impacts from mercury to warrant **further international action** to reduce the risks to humans and wildlife from the release of mercury to the environment. The Governing Council decided that **national, regional and global actions** should be initiated as soon as possible and urged all countries to adopt goals and take actions, as appropriate, to identify populations at risk and to reduce human-generated releases." (UNEP 2002)

A number of reviews have summarised published data concerning the long-range atmospheric transportation of mercury from industrial areas, and concluded that there is scientific evidence of a linkage between anthropogenic mercury emissions and elevated mercury concentrations in remote areas (Petersen *et al.* 1995; Jackson 1997a; Pai *et al.* 1997; Fitzgerald *et al.* 1998; Xu *et al.* 2000a; Xu *et al.* 2000b; Petersen *et al.* 2001; Wangberg *et al.* 2001). Measurements of mercury concentrations in ambient air support the conclusion that mercury deposited in remote areas may originate from anthropogenic sources far away. Thus, mercury becomes a global problem not only affecting local areas that are heavily industrialised, but also areas that are remote from emitting sources. In the view of some researchers, release of any mercury from anthropogenic sources which will lead to increases in the global pool, should be avoided since there is already evidence for significant impacts (Meili *et al.* 2003).

Recent international developments have led to a very careful examination of mercury emissions to the environment. For example, in December 2002, the United Nations Environment Programme (UNEP) published its Global Mercury Assessment (UNEP 2002). The Global Mercury Assessment provides an extensive overview of the current knowledge of mercury sources, and environmental impacts. The UNEP Governing Council concluded soon afterwards that:

- there is sufficient evidence of significant global adverse impacts from mercury to warrant further international action
- national, regional and global actions should be initiated as soon as possible to identify populations at risk and to reduce human-generated releases.

At the 24th *Session of the United Nations Environment Program Governing Council/ Global Ministerial Environment Forum* (UNEP 2007) it was further concluded that:

- "current efforts to reduce risks from mercury are not sufficient to address the global challenges posed by mercury", and
- "further long-term international action is required to reduce risks to human health and the environment and that, for this reason, the options of enhanced voluntary measures and new or

existing international legal instruments will be reviewed and assessed in order to make progress in addressing this issue” (UNEP 2007).

A range of potential actions were also identified at this meeting, including (UNEP 2007):

- substitution of products and technologies;
- technical assistance and capacity-building;
- development of national policy and regulation; and
- data collection, research and information provision, bearing in mind the need to provide assistance to developing countries and countries with economies in transition.

Specific measures were also identified, and include (UNEP 2007):

- reduction of atmospheric mercury emissions from human sources;
- development of environmentally sound solutions for the management of waste containing mercury and mercury compounds;
- reduction of global mercury demand related to use in products and production processes;
- reduction of global mercury supply, including considering curbing primary mining;
- development of environmentally sound storage solutions for mercury;
- remediation of existing contaminated sites affecting public and environmental health; and
- an increase in knowledge in areas such as inventories, human and environmental exposure, environmental monitoring and socio-economic impacts.

Inevitably developments such as these will focus attention on mercury emissions from all sources. In fact a number of countries (including the US, Canada and the EU) have already developed detailed strategies for reducing mercury use, and for controlling emissions.

On 20th February 2009 the UN Environment Programme's (UNEP) Governing Council agreed on a plan for a global approach to reduce population and ecosystem exposure to mercury. The landmark decision, taken by over 140 countries, sets the stage for the development of an international mercury treaty to deal with world - wide emissions and discharges of this pollutant. The Council also agreed that the risk to human health and the environment was so significant that accelerated action under a voluntary Global Mercury Partnership is needed whilst the treaty is being finalised.

1.2 Global mercury supply, trade and emissions

It is useful to consider a brief overview of mercury production, use, trade and emission sources. Historically, mercury was used extensively in a range of products, including:

- Electrolytic and chemical processes;
- Pesticides;
- Paints; and
- Batteries.

It is now recognised that the benefits of mercury use are exceeded by the risks, and mercury-free alternatives have been developed for most of the previous uses (the inclusion of mercury in energy-efficient lighting is an important exception). This reduction in mercury use is reflected in the data: the maximum annual global mercury use was ~10,000 tonnes in the 1960s (Swain *et al.* 2007), but had decreased to an estimated 3,500 tonnes in 2005 (Swain *et al.* 2007).

Mercury is also traded extensively, but the size of the market, and details of the subsequent uses of traded mercury, particularly in developing countries, are very difficult to quantify (Maxson 2005; Greer *et al.* 2006; Swain *et al.* 2007).

The atmosphere is the dominant transport medium for mercury in the environment, and hence emissions to the atmosphere are the major way in which the global pool of mercury can increase with consequent impacts on human and ecosystem health (Pirrone and Mahaffey 2005a; Pirrone and Mahaffey 2005b; Mergler *et al.* 2007; Munthe *et al.* 2007; Scheulhammer *et al.* 2007; Swain *et al.* 2007). It follows that regulations and voluntary agreements related to mercury will need to focus on the quantities of mercury emitted from a range of sources in order to meet the UNEP goals described above.

It is generally agreed (UNEP 2002) that there are four major categories of mercury emission:

- Natural sources, including rocks and soils and volcanic activity;
- Anthropogenic emissions from industrial activities where the feed materials for these processes contain mercury; these processes include combustion of fossil fuels, particularly coals, and metal smelting;
- Anthropogenic releases of mercury from the manufacture, use and/or disposal of mercury containing products (examples include batteries, thermometers, lighting, dental amalgam);
- Re-mobilisation of mercury originally released from anthropogenic sources and deposited in the environmental repositories such as soils, water bodies, sediments, and landfills.

A useful illustration of the current understanding of the global pathways of mercury is given in Figure 1, and of the sources of mercury emission are summarised in the following section. Nelson and co-workers (Nelson *et al.* 2004; Nelson 2007) have made estimates of Australian mercury emissions from both natural and anthropogenic sources.

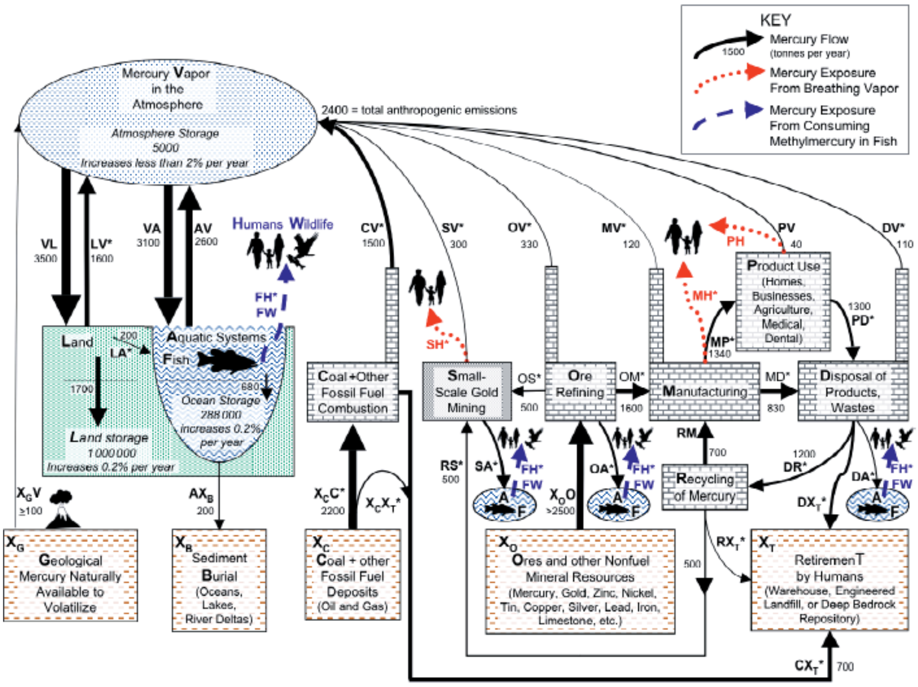


Figure 1: Important global pathways of mercury in commerce and the environment; from Swain et al (2007); codes used are defined in Table 1.

1.3 Sources of Mercury Emission

1.3.1 Natural Emissions

Mercury occurs primarily in the earth's crust and mantle. It occurs naturally in hydrothermal deposits in rocks as various minerals (eg. cinnabar, HgS), in coal, and in some sedimentary rocks, especially shales of high organic and sulphide content (Schroeder and Munthe 1998b). Mercury also exists as a trace element in numerous secondary sources in terrestrial environments (eg. soil and vegetation) and in the ocean (Jackson 1997b).

Divalent mercury, originating from both natural and anthropogenic sources, is the predominant form of mercury deposited to the planet (Lindberg and Stratton 1998; Bergan *et al.* 1999; Lindberg *et al.* 2007b). After deposition some of the mercury is reduced chemically and bio-chemically to elemental mercury which, due to its volatile nature, can be re-emitted back to the atmosphere. This bi-directional exchange (deposition-to-emission) of mercury across the air-surface interface makes it difficult to distinguish between emissions from a "pure" natural source and re-emission of previously deposited mercury.

Table 1: Important global compartments of mercury in commerce and the environment, as used in Figure 1; from Swain *et al* (2007)

Code	Mnemonic	Definitions
A	Aquatic system	Hg in wetlands, lakes, rivers, and oceans. Hg introduced to aquatic systems may become MeHg, which may be bioaccumulated by fish
C	Coal and other fossil fuel combustion	Hg mobilized by the processing and combustion of the fossil fuels coal, oil, and natural gas (XC)
D	Disposal	Hg in discarded products or process wastes from chlor-alkali or VCM plants
F	Fish	Hg in fish, virtually all of which is in the form of MeHg, which is produced by naturally occurring bacteria in aquatic systems
H	Humans	Hg absorbed by humans following exposure, generally through fish consumption or inhalation of vapor
L	Land	Hg in soil, mostly derived from atmospheric deposition of vapor, but can be elevated from mine waste, Hg waste disposal, or geologically rare mineral deposits containing Hg
M	Manufacturing	Hg used in the manufacture of Hg-containing products, or in processes that use Hg to make Hg-free products (e.g., chlor-alkali and vinyl chloride monomer processes)
O	Ore refining	Hg mobilized by the processing and refining of nonfuel mineral resources XO
P	Products	Hg contained in products, including thermometers, switches, fluorescent lamps, batteries, fungicides, preservatives, seed-coatings, pharmaceuticals, etc
R	Recycling	Hg that is extracted from discarded products or wastes, purified, and put into commerce or retired
S	Small-scale gold mining	Hg utilized by independent, artisanal, miners to concentrate geological gold through amalgamation
V	Vapor	Hg vapor in indoor and outdoor air
W	Wildlife	Hg absorbed by fish-eating wildlife, such as seal, whale, otter, mink, osprey, eagle, kingfisher, and loon
X	Out of the biosphere	Hg in the "X" compartments are not part of the Hg cycling in the biosphere and therefore do not harm humans or wildlife. "X" Hg may be mobilized at some point in the future, but for practical purposes is permanently stored unless humans intervene

XB	Buried	Hg, formerly in the biosphere, that has been buried in the sediments of oceans, lakes, and river deltas
XC	Coal and other fossil fuel deposits	Hg in buried fossil fuel deposits such as coal, oil, and gas, that may be extracted and burned
XG	Geological	Hg in geological materials that release Hg vapor to the atmosphere through natural processes
XO	Ores	Hg in non-fuel geological resources that are subject to mining and refining, including minerals containing Hg, gold, zinc, nickel, tin, copper, silver, lead, and iron. All geological materials contain some Hg, even limestone that is heated to make lime
XT	Retirement	Hg permanently stored, or ‘‘retired’’ by humans in warehouses, engineered landfills, or deep bedrock repositories

It is believed that a large part (up to 50 percent) of the mercury that is emitted from natural sources is actually of anthropogenic origin (Mason *et al.* 1994a). Evaporation of mercury from the oceans’ surface, emission of mercury from soil, vegetation and the release of mercury in forest fires, are consequently a mix of natural and re-emitted mercury. It is clear that care needs to be taken when referring to natural emissions since the term "natural" in this context may be somewhat misleading. In the context of the following discussion "natural emissions" will by definition also include re-emissions.

The mercury released from these many natural sources is mainly in the form of elemental mercury, although small quantities of dimethyl mercury are also released (Lindquist *et al.* 1991; Schroeder and Munthe 1998b). Forest fires emit elemental, divalent and particulate forms of mercury (Porcella *et al.* 1996). The magnitude of the mercury emissions released depends on a number of biological, chemical, physical, and meteorological factors, of which few are fully understood, and are subject to very large uncertainties.

1.3.2 Anthropogenic Mercury Emissions

A large proportion of the mercury present in the global atmosphere today is due to anthropogenic activities. These global activities have increased the overall mercury levels in the atmosphere by roughly a factor of three (UNEP 2002). Direct global anthropogenic emissions are believed to account for around 2000 tonnes/yr, which is approximately 35-60 percent of the annual total mercury emissions (UNEP 2002).

The Global Mercury Assessment (GMA) (UNEP 2002) provides an extensive overview of the current knowledge of mercury sources, and environmental impacts, and includes an estimate of total global emissions of mercury from anthropogenic sources for 1995. The estimate is based on data collected by Pacyna and Pacyna (2002), and is summarised in Table 2. Table 2 shows that the major anthropogenic sources of emissions of mercury to the atmosphere are:

- stationary combustion;
- non-ferrous metal production;
- pig iron and steel production;
- cement production; and
- waste disposal.

Approximately 1900 tonnes of anthropogenic mercury were estimated to be emitted, an apparent decrease of 10 percent since 1990 (Pacyna and Pacyna 2002). A major anthropogenic source, the use of mercury in artisanal gold mining, largely in developing countries, is not included and is highly uncertain in magnitude. However it could amount to more than 300 tonnes per year (Pacyna and Pacyna 2002), with some estimates as high as 800-1000 tonnes (Veiga *et al.* 2006). There have been both previous and more recent estimates (Pirrone *et al.* 1996; Pacyna *et al.* 2006) of global mercury emissions than those reported in the GMA, and maps have been developed to depict the spatial distribution of emissions (Pacyna *et al.* 2003; Wilson *et al.* 2006).

Table 2: Global emissions of mercury from major anthropogenic sources in 1995 (Mg yr^{-1}) [a]

Continent	Stationary Combustion	Non-ferrous metal production	Pig iron & steel Production	Cement production	Waste disposal	Total
Europe	185.5	15.4	10.2	26.2	12.4	249.7
Africa	197.0	7.9	0.5	5.2		210.6
Asia	860.4	87.4	12.1	81.8	32.6	1 074.3
North America	104.8	25.1	4.6	12.9	66.1	213.5
South America	26.9	25.4	1.4	5.5		59.2
Australia[a]	97.0	4.4	0.3	0.7	0.1	102.5
Oceania[a]	2.9	-	-	0.1	-	3.0
Total (1995)	1 474.5	165.6	29.1	132.4	111.2	1 912.8[b]
Total (1990[c])	1 295.1	394.4	28.4	114.5	139.0	2 143.1[d]

[a] Table from (Pacyna and Pacyna 2002; UNEP 2002) and personal communications with J. Pacyna

[b] 325 tonnes of mercury emissions from gold production is not included (>50% assumed to occur in Africa)

[c] Estimates of maximum values, which were regarded as close to the best value[a]

Table 3: Global anthropogenic emissions to air in 2005 from different sectors (UNEP Chemicals Branch 2008)

Sector	2005 emission, tonnes	Proportion (%) Of 2005 emission	Range of estimate, tonnes
Fossil Fuel combustion for power and heating	878	45.6	**595 – 1160**
Metal production (ferrous and non-ferrous, excluding gold)	200	10.4	**125 – 275**
Large scale gold production	111	5.8	**65 – 155**
Artisanal and small-scale gold production	350	18.2	**225 – 475**
Cement production	189	9.8	**115 – 265**
Chlor-alkali industry	47	2.4	**25 – 65**
Waste incineration, waste and other	125	6.5	**50 – 475**
Dental amalgam (cremation)	26	1.3	**20 – 30**
Total	1930		**1220 – 2900**

Although the detailed numbers vary, the overall conclusions on the major anthropogenic sources appear to be consistent. However, Nelson and co-workers (Nelson *et al.* 2004; Nelson 2007) have argued that the GMA estimates for Australian stationary combustion sources are a significant over-estimate.

Very recently revised estimates of global emissions of mercury have been made as a part of the UN process. The estimates are summarised in a United Nations Environment Programme (UNEP) report (UNEP Chemicals Branch 2008) prepared for the Governing Council of UNEP. The UNEP report is supported by two detailed technical reports (AMAP/UNEP 2008; Pirrone and Mason 2008). Data on global mercury emissions in 2005 to atmosphere are summarised in Figure 2 and Table 3.

It is worth making a number of points related to the data in Figure 2 and Table 3:

- The largest sectoral source is the combustion of fossil fuels, largely coal. This sector accounts for a total of ~46% of emissions to atmosphere, about 25% from electrical power plants and 20% from industrial and residential heating (UNEP Chemicals Branch 2008). The latter heating source is unlikely to be a significant source in Australia where coal is not often used for heating.
- Emissions of mercury from gold production arise from both large scale industrial production (~6% of total global emissions) and from small scale and artisanal gold mining and production (~18%, and largely in developing countries). The latter source is also unlikely to be significant in Australia.
- The mining, smelting and production of metals other than gold, and cement production each account for ~10% of global emissions.
- The emission estimates are subject to large uncertainties (see last column of Table 3).

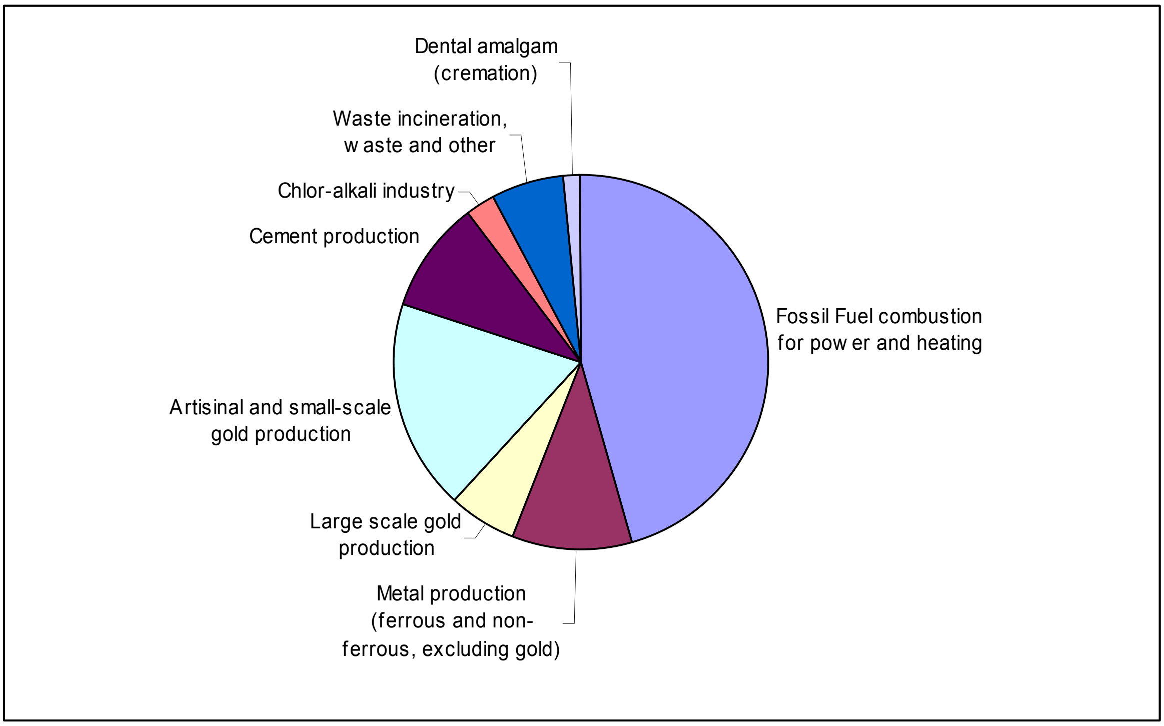


Figure 2: Proportion of global anthropogenic emissions to air in 2005 from different sectors (UNEP Chemicals Branch 2008) ; see Table 3 for details

The UNEP report (UNEP Chemicals Branch 2008) discusses uncertainties at length, and the issues it raises are worth noting in the context of the present study. According to this document, estimates of mercury emissions are affected by the following considerations:

- The lack of a sufficiently comprehensive database of emission measurements from large industrial sources;
- Regional differences and other variables that have an influence on emissions from waste disposal and incineration;
- Emissions from products containing mercury (eg dental amalgam, electrical switches) are highly uncertain due to uncertainties in both the product life cycles and emission factors;
- The accuracy of relevant statistics (eg, the amount of cement production or battery consumption);
- Accuracy of emission factors; and
- Assumptions about technology for mercury production and mercury control.

Uncertainties of 25-30% are estimated (UNEP Chemicals Branch 2008) for large industrial sources, and significantly larger uncertainties are likely for diffuse sources of mercury such as dental amalgam, and waste disposal and incineration.

1.4 Mercury Species in the Atmosphere and Mercury Deposition

Speciation of mercury determines atmospheric and environmental behaviour, so it is important to understand some of the fundamental aspects of mercury release from industrial sources. Mercury is released to the atmosphere in three main forms (EU 2004):

- elemental Hg (Hg^0);
- divalent Hg (Hg(II)); and
- particulate phase mercury (Hg_p).

The three different Hg species have, due to differences in physical and chemical properties, different atmospheric behaviour and residence times.

Mercury also exists in a monovalent form Hg(I) (e.g. Hg_2Cl_2). However, it is extremely unstable and will rapidly disproportionate to form Hg(II) and Hg^0 (McElroy and Munthe 1991). It is therefore assumed to have a minor importance in atmospheric mercury chemistry (Schroeder and Munthe 1998b). In addition to these species, methyl mercury is also believed to be emitted (mainly from industrial processes), however, in much smaller quantities (USEPA 1997b). Natural sources are assumed to emit mainly elemental Hg (Lindquist *et al.* 1991).

The prevailing Hg species in the atmosphere is elemental Hg (ca 98 %) (Lindquist *et al.* 1991). Due to its substantial vapour pressure it exists predominantly in the gaseous phase (Schroeder *et al.* 1991). The background concentration of Hg^0 in ambient air is approximately 1.3-1.5 ng m^{-3} in the Northern Hemisphere and 0.9-1.2 ng m^{-3} in the Southern Hemisphere (EU 2004).

Elemental Hg is:

- relatively unreactive (reacting slowly with atmospheric oxidants); and
- highly insoluble which prevents it from being removed efficiently through wet deposition and it is mainly transported back to the surface through dry deposition at a very low rate (Schroeder *et al.* 1991; Lin and Pehkonen 1999b; Lin and Pehkonen 1999a).

These properties combined lead to a global distribution and an atmospheric residence time of approximately one-year (Bergan *et al.* 1999). In addition, small amounts of elemental Hg may be removed from the atmosphere by being oxidised to divalent Hg or adsorbed onto particulate matter (Lindquist *et al.* 1991; EU 2004) (Hg(ads) and Hg(II)(ads) in Figure 1).

Divalent and particulate Hg, which are present in ambient air at concentrations of less than 2 percent of Hg^0, are:

- at least 10^5 times more soluble than Hg^0 (Lindberg and Stratton 1998) and
- readily removed after emission on local to regional scales via wet and dry processes (Slemr *et al.* 1985; Lindquist *et al.* 1991; Schroeder and Munthe 1998b).

These two inorganic Hg forms have residence times of a few hours to several months (Lindquist *et al.* 1991). However, some fine particles can approach the residence time of elemental Hg even after precipitation has occurred indicating that these may also be distributed on a global scale (Porcella *et al.* 1996). Furthermore, particulate Hg is exceptionally abundant in the atmosphere over polluted industrial areas where it may reach levels of 50 percent of the total Hg concentration (Schroeder *et al.* 1991; Keeler *et al.* 1995; Pirrone *et al.* 1996).

Divalent Hg, frequently referred to as reactive gaseous mercury (RGM), can react with a number of different ligands (OH^-, Cl^-, Br^-, I^-, SO_3^{2-} and CN^-) to form relatively stable inorganic complexes (e.g. $HgCl_2$ and $Hg(OH)_2$) (Seigneur *et al.* 1994; Travnikov and Ryaboshapko 2002). In addition, divalent mercury may react directly with organic molecules or through bacteria in aquatic systems, forming organic Hg compounds such as monomethyl mercury (MMM) (e.g. CH_3HgCl, CH_3HgOH, CH_3HgBr) and dimethyl mercury (DMM) (e.g. $Hg(CH_3)_2$) (Seigneur *et al.* 1994).

MMM is extremely toxic and of great environmental importance because of its ability to bio-concentrate in, for instance, fish tissues, which in turn affect human health (especially the central nervous system) following consumption (WHO 1990; WHO 1991). DMM is highly volatile and is rapidly released through the water phase to the atmosphere where it interacts with other chemical species (USEPA 1997b).

Particulate Hg is formed when divalent Hg complexes such as $Hg(OH)_2$, $HgCl_2$, $HgSO_3$ and $Hg(NO_3)_2$ are adsorbed onto particles particularly within atmospheric water droplets (Seigneur *et al.* 1994; Pleijel and Munthe 1995a; Pleijel and Munthe 1995b). Seigneur et al (1998) suggested that up to 35% of the dissolved divalent Hg species can be adsorbed onto particulate matter. In the gaseous phase, particulate divalent Hg consists mainly of sparingly soluble compounds such as HgO and HgS (Seigneur *et al.* 1998; Travnikov and Ryaboshapko 2002). These compounds are primarily removed via dry deposition, although approximately 50% of the Hg in rainwater occurs as particulates (Brosset and Lord 1991), indicating the importance of scavenging by the atmospheric aqueous phase.

Although elemental Hg is present as a vapour in the atmosphere, it may also adsorb onto particles and be subject to wet and dry deposition (EU 2004). The amount that is adsorbed depends upon the composition of the particle and the gas phase concentration of Hg. Adsorption is more likely to occur when the particulate matter is rich in elemental carbon (soot), which has a high adsorption coefficient for Hg (Petersen *et al.* 1998; Pirrone *et al.* 2000). Another source that incorporates Hg with particulate matter is combustion of fossil fuels where some of the Hg present in the fuel is emitted bound to particulate matter. This bound Hg is not released or engaged in any further reactions and is therefore deposited together with the particle (EU 2004). Figure 3 shows a summary of atmospheric processes important in the mercury cycle.

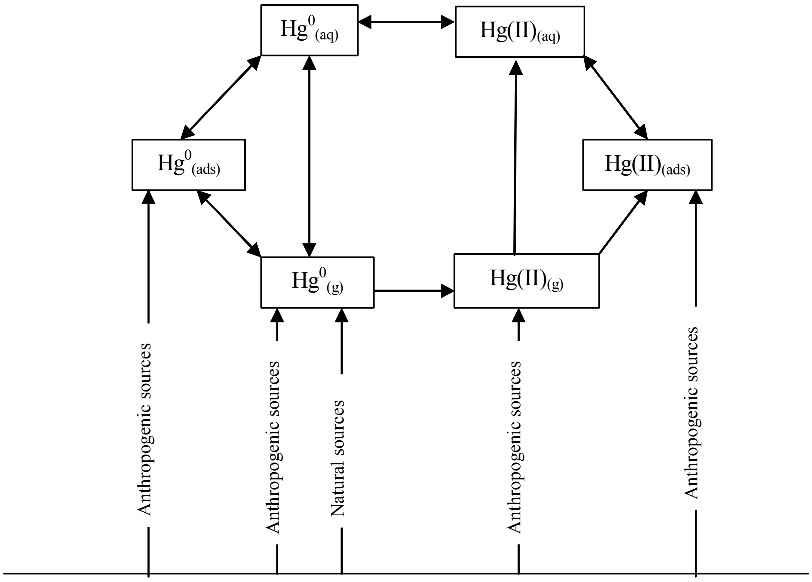


Figure 3: Mercury oxidation, reduction and mass transfer processes in the atmosphere. Natural sources (including re-emission of previous deposited Hg) also emit Hg_p (represented in the figure as Hg(ads) and Hg(II)(ads)) but in small quantities. The idea for the figure came from the front page of 2001 Special Issue of Atmospheric Environment (vol.35, no.17).

SUMMARY - SPECIATION OF ATMOSPHERIC MERCURY

Speciation of atmospheric Hg is critical to:

- removal rates
- transportation distance from emission sources.
- Environmental impact

Near-source contamination is most likely related to the **emission of divalent and particulate forms of Hg.**

Effects at greater distances from the source are associated with elemental Hg.

To evaluate the global cycling of Hg and its effects in the environment it is important to understand:

- the speciation of mercury in emissions
- the different transformation processes, including transitions between the gaseous, aqueous and soil phases, and
- chemical reactions in the gaseous and aqueous environments.

THESE CONSIDERATIONS INFLUENCE THE DATA REQUIREMENTS FOR THIS PROJECT AND THE APPROACH TO MODELLING

1.5 Mercury in Australia and Objectives of this Study

There is significant uncertainty in our current understanding of the sources, fates and impacts of mercury in Australia. These uncertainties include:

- emission source strengths from stationary sources in Australia;
- emissions from natural sources (eg, bushfires, water bodies and vegetation), and re-emission of previously deposited mercury;
- the relative contributions of the different chemical forms of mercury (ie, elemental, oxidised and particulate) in many sources; and
- limited data on the use of mercury in products and its fate.

The Department of Environment, Water, Heritage & the Arts (DEWHA) has commissioned the Graduate School of the Environment at Macquarie University (through AccessMQ) to carry out a study to determine the sources, transportation and fate of mercury in Australia. The study commenced in July 2008 and has six parts:

A Collection of Data on Mercury Emissions, Sources and Trends from Anthropogenic and Non-Anthropogenic Sources
B Study of the transport and fate of mercury in Australia
C The identification of gaps in the scientific data related to mercury in Australia.
D The identification of areas or populations especially at risk from mercury in Australia
E The collation of information into an inventory of mercury sources and emissions in Australia
F Study of the availability, efficiency and costs of control technologies

The work carried out on all these areas is collated in this final report.

2 Mercury Emissions from Anthropogenic Sources in Australia

2.1 Methodology

The project team resolved to use National Pollution Inventory (NPI) data where available and consistent with data reported elsewhere. Reporting of point source emissions in Australia has been mandated since 1998, under the National Pollutant Inventory (NPI) National Environment Protection Measure (NEPM). The NPI includes data on emissions of 93 substances to air, land and water. The NPI also includes estimation of some area sources, known as aggregated emission data. In 2007 the NPI NEPM was varied and the threshold for reporting mercury was reduced from 10 tonnes to 5 kg. This reduced threshold is mandatory for the 2007-08 NPI reporting year. Further details of the NPI program are given on the NPI website, www.npi.gov.au (DEH 2006). In this study emission estimates and modelling of mercury transport and fate were undertaken for 2006. Examination of more recent NPI data (ie, after the threshold variation discussed above) did not show any major changes in industrial source contributions.

Time series NPI data for the top 90% of reported mercury emissions were examined to determine data consistency in the period 2001-2007. Where appropriate this NPI data for NSW sources has been compared to the comprehensive 2003 emissions inventory (which included mercury) carried out independently by the NSW Department of Environment and Climate Change (DECC). It should be noted that substantial components of the DECC inventory are based on NPI data.

For some potential sources, and most notably area and diffuse sources, NPI data is not available, or is of inconsistent quality (as detailed in previous correspondence with DEWHA); these sources typically include distributed sources such as crematoria and cemeteries, motor vehicle emissions, and landfills, sewage treatment plants (STPs) and domestic combustion sources. In these cases alternate approaches were developed based on a range of data sources as detailed below.

In addition, the NPI reports substantial emissions from paved and unpaved roads. This data is very inconsistent across reporting jurisdictions. It is also likely that the major part of this emission will be related to very coarse dust particles which rapidly settle out in the atmosphere, and are unlikely to remain in the atmosphere for a significant period of time. For these reasons this source was not included in the inventory developed for this study.

2.2 Inventory of atmospheric emissions of mercury from Australian anthropogenic sources

Results of the data collection for Australian anthropogenic sources are summarised in Figure 4 and Table 4. Full details of the source of the data, including explanatory notes are provided in Table 5, which is organised in classes/source categories and arranged according to the format of the *UNEP Toolkit for identification and quantification of mercury releases* (UNEP 2005).

The detailed information in Table 5 includes discussion of the many uncertainties and assumptions in the derivation of the data summarised in these figures and tables, particularly the very high reliance on overseas sources of information, assumptions, and emission factors. Needless to say these uncertainties are substantial so that the mercury emission inventory should be used with caution, and any impacts predicted using inventory data should recognise the limitations which these uncertainties impose on any conclusions or decisions made which are based on these data.

Nonetheless some general comments can be made about the estimated emissions of mercury to the atmosphere from Australian anthropogenic sources:

- The best estimate of total emissions of mercury to the atmosphere in 2006 was around 15 tonnes. Using a very different methodology the most recent global emission estimate (AMAP/UNEP 2008) reports total anthropogenic emissions from Australia at ~34 tonnes/year; the difference is largely due to a much higher estimate for emissions from stationary combustion in the global estimate (AMAP/UNEP 2008). It can be convincingly argued that the estimate presented in this report for stationary combustion (largely coal-fired power stations) is more accurate as it uses NPI reported emissions which incorporate estimates of mercury capture in air pollution control devices (the global estimate does not include any mercury capture), and is supported by the comparison of top down and bottom up estimates of mercury from Australian stationary sources reported by Nelson (2007).
- Three sectors contribute substantially to Australian anthropogenic emissions; these are gold smelting (49.7%), coal combustion in power plants (14.8%), and alumina production from bauxite (12.2%). It is worth noting that the gold smelting emissions are from a single location at Kalgoorlie in Western Australia.
- A range of other diverse sectors contribute smaller proportions of the emitted mercury. These include industrial sources (mining, smelting, and cement production), and intentional use of mercury in products. It is difficult to determine historical trends in mercury emissions given the large uncertainties in the data. However it is clear that the intentional use of mercury in products is in decline. In addition the only remaining mercury-based chlor-alkali plant in Australia, at Orica in Sydney, has now ceased operation. Emissions from this source are predicted to decrease very significantly as the remaining mercury stocks are removed from the site. In fact Orica reports emissions of only 7.9 kg in 2007 (in contrast to reported emissions of 340 kg in 2006).

Table 4: Atmospheric emissions of mercury for 2006 from Australian anthropogenic sources, classified by sector

Sector	Emissions, kg/year	Proportion of Total Emissions (%)
Gold smelting	7642	49.7
Coal combustion in power plants	2271	14.8
Alumina production fron bauxite	1872	12.2
Copper, zinc, lead & silver smelting	629	4.1
Coke production	500	3.2
Chlor-alkali production	340	2.2
Cement and lime production	313	2.0
Primary ferrous metal production	247	1.6
Biomedical waste incineration	236	1.5
Electrical and electronic switches	207	1.3
Light sources	177	1.2
Crematoria/ cemeteries	172	1.1
Copper, zinc, lead & silver mining	169	1.1
Oil refining	101	0.7
Combustion of oil	101	0.7
Measuring equipment	92	0.6
Laboratory equipment	80	0.5
Production of recycled ferrous metals	63	0.4
Dental amalgam	59	0.4
Batteries	36	0.2
Gold mining	29	0.2
Pulp and paper production	14	0.1
Total	15346	

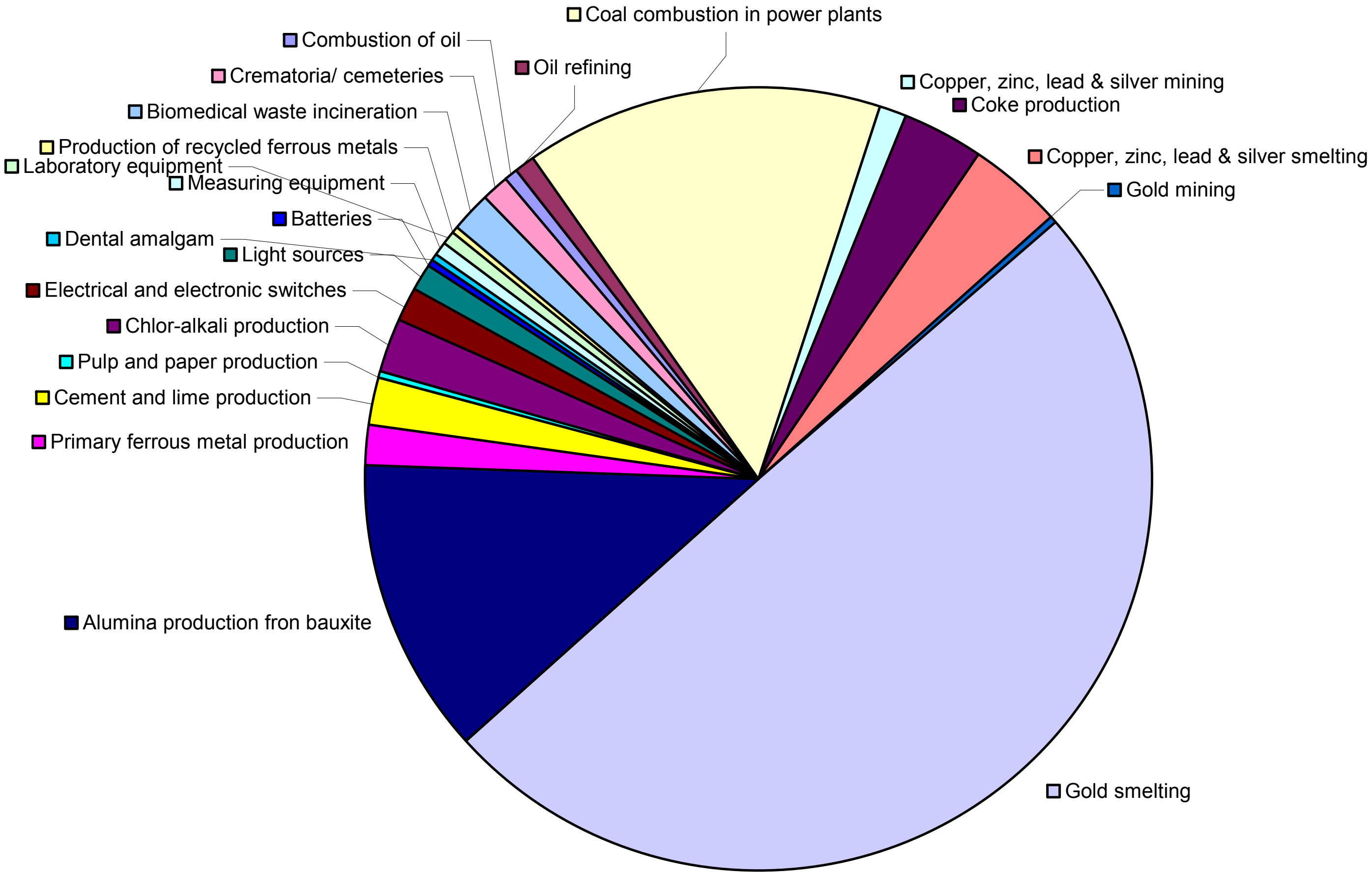


Figure 4: Anthropogenic sources of mercury emissions to the atmosphere in Australia in 2006

Table 5: Estimates of mercury emissions from anthropogenic sources in Australia organized according to the UNEP Toolkit (UNEP 2005) for construction of mercury inventories; the category and sub-category identifiers and categorizations refer to the Toolkit

Main source category	**Sub-category**	**Source category**	**Sources of information**		**Emission estimate kg/year**					**Comments**
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
5.1		**Extraction and use of fuels /energy sources**								
	5.1.1	Coal combustion in large power plants	As reported in the NPI [3]	As reported in the NPI	2270.5	60.7	0	0	0	See Note 1
	5.1.2	Other coal use								
		Coke production	As reported in the NPI	As reported in the NPI	500	0	0	0	0	Emissions consistent with DECC inventory
	5.1.3	Mineral oils – extraction , refining and use								
		Combustion of residual oil in residential heating	NPI mercury content of 15 mg/tonne [4]	6000 tonnes/yr[5]	0.1	0	0	0	0	Estimates using UNEP default emission factors 0.06 to 1.8 kg/yr
		Combustion of residual oil (other than in power station and residential heating)	NPI mercury content of 15 mg/tonne [4]	1.435 million tonnes/yr[6]	21.5	0	0	0	0	Estimates using UNEP default emission factors 14.3 to 430.5 kg/yr
		Combustion of distillate (diesel oil)	Mercury content of 5 mg/tonne	12.91 million tonne/yr[7]	64.5	0	0	0	0	Note 2 Estimates using UNEP default

---

[3] Emissions from brown coal plants increased by a factor of 10, as per explanation in Note 1

[4] http://www.npi.gov.au/handbooks/approved_handbooks/pubs/boilers.pdf

[5] http://data.un.org/Data.aspx?d=EDATA&f=cmID%3ARF%3BtrID%3A1234

[6] http://www.abareconomics.com/interactive/energy/excel/Tablek.xls

Main source category	**Sub-category**	**Source category**	**Sources of information**		**Emission estimate kg/year**					**Comments**
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
										emission factor range from 13.6 to 1367 kg/yr
		Combustion of gasoline	Mercury content of 1 mg/tonne[4]	17.89 million tonne/yr[8]	14.7	0	0	0	0	Note 3 Estimates using UNEP default emission factor range from 17.9 to 1790 kg/yr
		Oil Refining	As reported in the NPI	As reported in the NPI	101.1	2.6	6.2	0	0	Likely underestimate See Note 4
	5.1.4	Natural gas-extraction, refining and use								
		Extraction/refining	As reported in the NPI	As reported in the NPI	8.7	39.8	11.9			See Note 5
		Use of pipeline gas	UNEP mercury content of 0.4 µg/Nm3	26748 million Nm3/yr[9]	10.7	0	0	0	0	See Note 6
	5.1.5	Other fossil fuels – extraction and use	Not applicable		0	0	0	0	0	Combustion of peat
	5.1.6	Biomass fired power station	As reported in the NPI	As reported in the NPI	0.24	0	0	0	0	Emission from Rocky Point Green Power

---

[7] http://www.abareconomics.com/interactive/energy/excel/Tablek.xls
[8] http://www.abareconomics.com/interactive/energy/excel/Tablek.xls
[9] http://www.abareconomics.com/interactive/energy/excel/Tablee.xls

Main source category	**Sub-category**	**Source category**	**Sources of information**		**Emission estimate kg/year**					**Comments**
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
5.2		**Source category: Primary metal production**								
	5.2.1	Mercury extraction and initial processing			0	0	0	0	0	None in Australia
	5.2.2	Gold and silver extraction with amalgam processes			0	0	0	0	0	None in Australia
	5.2.3, 5.2.4 5.2.5	Copper, zinc, lead silver extraction and initial processing								
		Copper, zinc, lead and silver mining	As reported in the NPI	As reported in the NPI	168.7	31.1	4.7	0	0	Note 7
		Copper, Zinc, lead and silver smelting	As reported in the NPI	As reported in the NPI	628.6	1.2	70.0	0	0	Note 8
		Gold extraction and initial processing								
	5.2.6	Gold mining	As reported in the NPI	As reported in the NPI	29.0	7.7	4.1	0	0	
		Gold smelting	As reported in the NPI	As reported in the NPI	7641.5	0	0	0	0	Note 9
	5.2.7	Aluminum extraction & initial processing								
		Bauxite mining	As reported in the NPI	As reported in the NPI	5.8	0	0.4	0	0	

Main source category	**Sub-category**	**Source category**	**Sources of information**		**Emission estimate kg/year**					**Comments**
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
		Alumina production from bauxite	As reported in the NPI	As reported in the NPI	1871.6	0	0.2	0	0	Note 10
	5.2.8	Other non-ferrous extraction and processing								Included in other metals
	5.2.9	Primary ferrous metal production	As reported in the NPI	As reported in the NPI	247		0	0	0	Note 11
5.3		**Production of other minerals and materials with mercury impurities**								
	5.3.1	Cement and lime production	As reported in the NPI	As reported in the NPI	313.2	0	0	0	0	Note 12 Estimates using NPI emission factors vary from 187 (FF) to 858 (ESP) kg/yr
	5.3.2	Pulp and paper production	As reported in the NPI	As reported in the NPI	13.7	1.2	0	0	0	Note 13
	5.3.3	Production of lime and light weight aggregates from kilns	Included in 5.3.1							
	5.3.4	Other minerals (glass)	As reported in the NPI	As reported in the NPI	9.5	0	0	0	0	Note 14
5.4		**Intentional use of mercury in industrial processes**								
	5.4.1	Chlor-alkali production with	As reported in the NPI	As reported in the NPI	340	0	0	0	0	Orica chlor-alkali in Botany closed

Main source category	Sub-category	Source category	Sources of information		Emission estimate kg/year					Comments
			Emission factor	Activity rate	Air	Water	Land	Impurity in product	General waste	
		mercury technology								in 2002. Note 15
	5.4.2	VCM production with mercury dichloride as catalyst			0	0	0	0	0	No production in Australia
	5.4.3	Acetaldehyde production with mercury sulfate as catalyst			0	0	0	0	0	No production in Australia
	5.4.4	Other production of chemicals and polymers with mercury as catalyst			0	0	0	0	0	No production in Australia
5.5		**Consumer products with intentional use of mercury**								
	5.5.1	Thermometers and measuring devices	See Note 16	See Note 16	92	138	92		138	Note 16
	5.5.2	Electrical switches and relays with mercury	0.02 g per year per inhabitant[10]	Population	207	0	207	0	1656	
	5.5.3	Light sources with mercury	See Note 17	See Note 17	177	0	0	0	3360	Note 17
	5.5.4	Batteries with mercury			35.7				321.3	Note 18
		Gas and electric			0	0	0	0	0	Included in

---

[10] UNEP Toolkit default values 0.02-0.25 Hg/yr/inhabitant. Assume 0.1 g/yr/inhabitant, 20.7 million population, and UNEP values for pathway distribution to air, water,.land and waste

Main source category	**Sub-category**	**Source category**	**Sources of information**		**Emission estimate kg/year**					**Comments**
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
		cooking ranges with mercury								switches
	5.5.5	Biocides and pesticides			0	0	0	0	0	No longer used in any significant quantity
	5.5.6	Paints			0	0	0	0	0	No longer used
	5.5.7	Pharmaceuticals for human and veterinary use			0	0	0	0	0	No longer used
	5.5.8	Cosmetics and related products			0	0	0	0	0	No longer used
5.6		**Other intentional products/ process uses**								
	5.6.1	Dental amalgam	Average mercury in fillings[11]	3465300 restorative services/yr [12]	58.6	1348.8	0	351.9	1055.6	See Note 19
	5.6.2	Manometers and gauges								Incorporated in 5.5.1 Note 16
	5.6.3	Laboratory chemicals and equipment	Denmark emission per inhabitant[13]	Australia population	80	80				Note 20
	5.6.4	Mercury metal uses in religious rituals and folklore medicine			0	0	0	0	0	Not used in Australia in significant amounts

---

[11] 0.4 g mercury per one-surface filling, 0.8 g mercury per two-surface filling and 1.2 g mercury per three-surface fillings (as per Skarup et al 2003)

[12] Annual amalgam restorative services as per NHMRC 1999 report.

[13] UNEP reported emission in Denmark of 20-40 kg/yr. Denmark population of 5 million

Main source category	Sub-category	Source category	Sources of information		Emission estimate kg/year					Comments
			Emission factor	**Activity rate**	**Air**	**Water**	**Land**	**Impurity in product**	**General waste**	
	5.6.5	Miscellaneous product uses, mercury metal uses and other sources							5.3	Notes 21 and 22 televison, laptop computers and modems
5.7		**Production of recycled metals (secondary)**								
	5.7.1	Production of recycled mercury			0	0	0	0	0	No recycling of mercury in 2006. Note 23
	5.7.2	Production of recycled ferrous metals	As reported in the NPI	As reported in the NPI	62.6	0	0	0	0	Note 24
	5.7.3	Production of other recycled metals	As reported in the NPI	As reported in the NPI	3.4	0	0	0	0	
5.8		**Waste Incineration**								
	5.8.3	Incineration of medical waste	As reported in the NPI	As reported in the NPI	235.6	0	0	0	0	Note 25
5.9		**Waste deposition/ landfilling and waste water treatment**								
	5.9.1	Controlled landfills/ deposits	As reported in the NPI	As reported in the NPI	4.7	0	500.5	0	0	Note 26
5.10		**Crematoria and cemeteries**								
	5.10.1	Crematoria	1.92 g/cremation	89580 cremation/yr	172	0	0	0	0	Note 27
Total					**15389**	**1711**	**897**	**352**	**6536**	

Notes to Table 5

Note 1- Emissions from coal combustion in large power plants

The NPI appears to underestimate mercury emissions from brown coal power stations. Based on coal mercury content and tonnages of brown coal used, Nelson (2007) estimates that 2.1 tonnes of mercury are emitted if it is assumed that there is no capture of mercury from control equipment. US evidence suggests that Hg capture from plants fired with similar coals (lignites) could be as low as ~5%. Even if brown coals behave differently to lignite, a conservative assumption might be that 50% is captured (this is similar to the numbers observed for US bituminous coals in fabric filters (FFs). This would yield an estimated emission of 1 tonne per year from brown coal combustion.

The NPI presents estimates of mercury emission from brown coal fired plants of 0.1 tonnes per year, an order of magnitude lower than the estimate of Nelson (2007). On this basis, all estimates from brown coal plants have been increased by a factor of ten.

The following additional anomalies are noted:

- The reported 2006 mercury emission from Eraring power station is 28 kg per year, the lowest emission when compared to emissions from other years (see Table 6 below).
- Reported mercury emission from Tarong power station in 2006 was 305 kg, the highest reported;

The reasons for these differences are not known.

Table 6: NPI estimated mercury emissions for some power stations

Mercury emission kg/year							
Year	**2007**	**2006**	**2005**	**2004**	**2003**	**2002**	**2001**
Eraring	81	28	61	65	56	74	164
Tarong	150	305	146	118	88	142	169
Redbank	4.3	5.4	3.5	0.53	38	0.067	0.67

For the purpose of this project, estimates as reported by the NPI in 2006 are used. However it is desirable that these anomalies be investigated.

Note 2- Emission from combustion of distillate oil

The mercury content of distillate is expected to be variable; depending on the sources of crude oil. In general mercury content in distillate oil is much lower (by a factor of 3 or 4) than residual fuel oil due to the nature of the products. This is in agreement with the ranges of NPI default mercury content for residual fuel oil and distillate. For this inventory an emission factor of 5 mg/tonne i.e. one third of mercury content of residual oil was used for distillate. This is a compromise between the NPI default value, and the much lower concentrations reported in the literature.

Note 3- Emission from gasoline consumption

Data on mercury concentration in gasoline is very limited. To our knowledge there have been no reported measurements of mercury content in gasoline sold in Australia. The *NPI Emissions*

Estimation Technique Manual For Aggregated Emissions from Motor Vehicles (22 November 2000 – Version 1.0)[14] does not include an emission factor for mercury in gasoline.

There is very little extant data on mercury emissions from motor vehicles, particularly in the form of emission rates that could be used together with existing inventories of motor vehicle emissions to calculate emissions from this source.

Some data does exist for the mercury contents in petroleum products which may be used together with ABS data on the sales and production of motor vehicle fuels to calculate total mercury emissions from this source. The most recent data for mercury contents in various petroleum products is summarised in Table 7.

Table 7: Mercury contents in petroleum products

Petroleum product	Source	Mercury concentration (ppb)	Reference
Crude oil	Worldwide	1505 (mean) 3278 (std dev)	(Wilhelm and Bloom 2000)
Condensate	Worldwide	3694 (mean) 11665 (std dev)	(Wilhelm and Bloom 2000)
Light distillates	Worldwide	1.32 (mean) 2.81 (std dev)	(Wilhelm and Bloom 2000)
Utility fuel oil	Worldwide	0.67 (mean) 0.96 (std dev)	(Wilhelm and Bloom 2000)
Asphalt	Worldwide	0.27 (mean) 0.32 (std dev)	(Wilhelm and Bloom 2000)
Refined products			
Gasoline	US	0.22 -1.43 (mean 0.7)	(Wilhelm 2001)
	Foreign to US	0.72 – 3.2 (mean 1.5)	(Wilhelm 2001)
	US	0.38 ± 0.15[a]	(Landis *et al.* 2007)
	San Francisco Bay area	0.08 - 1.4	(Conaway *et al.* 2005)
	Korean	0.77 ± 0.01[a]	(Won *et al.* 2007)
Diesel	US	0.4 (one sample)	(Wilhelm 2001)
	Foreign to US	2.97 (one sample)	(Wilhelm 2001)
	US	0.073 ± 0.04	(Landis *et al.* 2007)
	San Francisco Bay area	0.05 - 0.34	(Conaway *et al.* 2005)
	Korean	0.22 ± 0.003	(Won *et al.* 2007)
LPG	Korean	2.32 ± 0.04	(Won *et al.* 2007)

[a] assuming a density of 0.74 g/cm^3
[b] assuming a density of 0.85 g/cm^3
[c] assuming a density of 0.53 g/cm^3

The data in Table 7 may be compared with that used by Pacyna and co-workers in their most recent estimates of global mercury emissions. They assumed a mercury content of 0.0006 g ton^{-1} of oil for the generic source of oil combustion (Pacyna *et al.* 2006).

---

[14] See http://www.npi.gov.au/handbooks/aedmanuals/pubs/motorvehicles.pdf, accessed 20th October 2008

These data have been used by a number of workers, together with total consumption data for the specific locations, to estimate total mercury emissions from motor vehicles. Based on the data of Landis *et al* (2007), US mercury emissions from motor vehicles were estimated to be 148 kg year^{-1}, 136 kg from gasoline, and 12 kg from diesel powered vehicles. Hence this is a relatively small source compared to industrial emissions such as those from coal-fired power stations (estimated to be greater than 30 tonnes per year in the US).

Similarly using data from the San Francisco Bay area, Conaway *et al* (2005) estimated emissions of 0.7-13 kg Hg yr^{-1} to the environment in that area, with an average of 5 kg Hg yr^{-1}. This represents less than 3% of the total atmospheric emissions in the San Francisco Bay area. (Conaway *et al.* 2005).

Based on these estimates, mercury emissions from motor vehicles in Australia are also likely to be small. For this estimate, based on the data summarised in Table 7, an emission factor of 1 mg/tonne is assumed.

Note 4 - Emissions from oil refineries

The NPI air emissions for refineries are summarised in Table 8 below.

Table 8: NPI reported emissions to air from Australian refineries

Mercury emission kg/yr								
Year	**Capacity mtpa** [15]	**2007**	**2006**	**2005**	**2004**	**2003**	**2002**	**2001**
Caltex (NSW)	4.9	2.9	2.5	2.5	2.1	2.4	2.6	**2.9**
Shell Refining (NSW)	3.2	0.8	1.2	164.4	15.9	33.6	3.4	**3.1**
Southern Oil Refining Pty Ltd		0.01						
BP (QLD)	2.6	18.6	12.9	35.2	26.8	10.5	11.4	**18.0**
Caltex (QLD)	3.4	3.98	3.7	2.79	5.47	3.19	2.39	**2.7**
Mobil (VIC)	5.0	8.3	11	8.08	6.03	2.59	3.1	**7**
Shell (VIC)	5.3	65	69	37	23.0	45	45	**32**
BP (WA)	5.7	7.39	0.8	0.7	0.82	0.77	0.81	**1.17**
Oil Energy Corporation Pty Ltd (WA)		0.02			0.11	0.12		
Total	**30**	**107**	**101**	**250.67**	**80.23**	**98.17**	**68.7**	**66.87**

From the above Table, significant variations in emissions between facilities are noted. For example, emissions from Shell Refinery in Victoria are consistently higher than those from BP (WA) for plants with similar capacity. Significant changes from year to year are also observed: 164.4 kg in 2005 compared to 0.8 kg in 2007 for the Shell Refinery in NSW.

The Mobil (Altona) Environmental Improvement Plan[16] indicates that the daily loads of mercury in wastewater have been reduced from 30 g/day (10.9 kg/yr) in 2003 to 5 g/day (1.8 kg/yr) in 2006.

---

[15] Estimates for 2006 based on publicly available information

[16] http://www.exxonmobil.com/Australia-English/PA/Files/publication_AltonaRef_EIP_2007_to_2009.pdf

However, the NPI does not record any mercury discharge from the Altona plant. Mercury discharge from Altona refinery has been included in the table.

According to the NSW DECC emission inventory, mercury emissions from refineries in NSW totaled 15.99 kg in 2003 calendar year. The NPI estimates of 18 kg of mercury in 2003/04 financial year are broadly consistent with those from the NSW DECC.

The NPI estimates are well below those estimated using the UNEP (lowest) emission factor of 10 mg/tonne. The reasons for these differences are not known.

If the UNEP range of emission default factors is used, mercury emissions from refineries in Australia range from 300 kg to 9000 kg per year.

Note 5- Emission from oil and gas extraction

The NPI includes 100 facilities classified as oil and gas extraction. The majority of these facilities have mercury emissions of less than 0.1 kg per year. Emissions to water are dominated by three sources:

Santos (QLD): 2.63 kg/yr
Esso (Vic): 27.2 kg/yr
Vermillion Oil & Gas (WA): 6.21 kg/yr

We have used the NPI estimates in the inventory.

Note 6 – Emission from combustion of natural gas

The NPI mercury content (default) of natural gas is 5.62×10^{-06} kg/tonne or 4 $\mu g/Nm^3$; which is 10 times higher than the maximum concentration (0.4 $\mu g/Nm^3$) as quoted in the UNEP Toolkit. The highest UNEP mercury content is used in this inventory.

Note 7 – Emissions from copper, silver, lead, zinc mining

The most significant emission for this category is from WMC Olympic Dam (90.5 kg/year) in SA. Emissions as shown in the NPI are used in the inventory. Based on historical data, the 2006 emission appears to be "representative" as shown below (Table 9).

Table 9: Mercury emissions reported in the NPI from WMC Olympic Dam

Mercury emission (kg/year)					
Year	2007	2006	2005	2004	2003
Olympic Dam	16.5	90.5	100	69.76	120.7

Note 8 – Emissions from copper, silver, lead & zinc smelting

Mercury emissions from silver, lead, and zinc smelting as reported in the NPI are shown in Table 10.

Table 10: Mercury emissions reported in the NPI from copper, silver, lead and zinc smelting

Mercury emission kg/year							
Year	**2007**	**2006**	**2005**	**2004**	**2003**	**2002**	**2001**
Mount Isa Mine	944	541.6	761.6	555.8	1356.1	758.4	640
Zinifex Hobart smelter	64.8	15	75.8	30.2		35.1	92.3
Zinifex Port Pirie		72	10.3	28	360	970.2	120
Total	1008.8	628.6	847.7	614	1716.1	1763.7	852.3

For a particular facility annual emissions vary significantly. The reasons for these variations are not known. In this study, emissions as reported in the NPI are used which likely result in an under-estimation of emissions. As Mt Isa mine is the largest source in this category, further investigation on the reasons of these variations is recommended.

Note 9- Emissions from gold smelting

Mercury emissions from Kalgoorlie Consolidated Gold Mining (KCGM) in Western Australia, are from two separate processing sources: the Finniston Carbon Regeneration Kiln, and the Gidji Roaster.

Note 10 - Emission from production of alumina from bauxites

Mercury emissions from production of alumina as reported in the NPI are summarised in Table 11.

Table 11: Mercury emissions (kg/year) from production of alumina as reported in the NPI

Plant	Capacity[17]	Year						
		2007	2006	2005	2004	2003	2002	2001
Alcan Gove Pty Ltd - Refinery	2	76	166	188				
Kwinana Alumina Refinery	2	280	320	300	300	300	320	320
Pinjarra Alumina refinery	4	450	485	430	430	430	400	420
Wagerup Alumina refinery	2.6	300	290	280	290	290	277	320
Worseley Alumina	3.1	580	417	360	375	432	687.6	547
Queensland Alumina Ltd	3.95	197	193.6	216	214	223	187.6	182
Total	17.65	1883	1871.6	1774				

---

[17] Millions of Tonnes of alumina per year. Figures are for 2006. Obtained from publicly available information

There are no significant variations in annual emissions for individual facilities. Mercury emissions vary from facility to facility probably due to the mercury content in bauxite.

Table 12: Summary of emission factors for alumina production

Source	NPI EET Manual 2007[18]	Alcoa Australia[19]	Alcoa Jamalco[20]	Bergsdal et al[21]	USEPA[22]
Emission factor mg/kg	0.9	0.07	0.18	0.042	0.1

It should be noted that the NPI Manual for alumina refinery Version 2 (2007) has revised the mercury content in bauxite from 30 mg/kg to a maximum of 0.9 mg/kg and that the UNEP Toolkit adopted the now superseded emission factor as a default mercury concentration.

In general it requires 2.46 tonnes of bauxite to produce 1 tonne of alumina[23]. Using the production of alumina, the NPI mercury content in bauxite and the amount of bauxite required to produce a tonne of alumina, the estimated emissions would have been ten times higher than those reported by facilities. The lower emission factor appears to be consistent with those provided elsewhere (Table 12).

Note 11- Emission from primary ferrous metal production

The NPI reported mercury emissions from primary ferrous metal production (iron smelting and steel production) are shown below (Table 13).

Table 13: NPI reported emissions from primary ferrous metal production

Facility	Mercury emission kg/yr						
	2007	2006	2005	2004	2003	2002	2001
Bluescope Steel	246	247	245	241	148	284	102

Note 12 -Emission from cement and lime manufacturing

Mercury emissions from cement and lime manufacturing as reported in the NPI are summarised below (Table 14).

These data show that emissions vary from year to year and from facility to facility. In 2006, emissions from a number of plants appear anomalous when compared to previous years but these differences may be related to changed operating procedures or throughput. NSW Department of Environment and Climate Change (DECC) estimated an emission of 152 kg from cement manufacturing in NSW for 2003; compared to the NPI of 90 to 100 kg/year.

---

[18] http://www.npi.gov.au/handbooks/approved_handbooks/pubs/falref.pdf

[19] http://www.aluminalimited.com/index.php?s=awac_biz&ss=global&p=global_op

[20] http://www.alcoa.com/jamaica/en/pdf/0506sustainabilityreport2.pdf

[21] http://www.alcoa.com/jamaica/en/pdf/0506sustainabilityreport2.pdf

[22] http://www.epa.gov/bns/reports/stakesdec2005/mercury/Cain2.pdf

[23] http://www.alcoa.com/jamaica/en/pdf/0506sustainabilityreport2.pdf

Table 14: Mercury emissions (kg/year) from cement and lime manufacturing reported in the NPI

Facility	Capacity Kt	2007	2006	2005	2004	2003	2002	2001
Blue Circle (Maldon)	250	16.2	18.5	16	20.5	18.4	23.4	23
Blue Circle (Berrima)	1400	20.95	8.71	31.41	32	24.7	9.1	3.6
Aus Cement Holdings (Kandos)	405	11.7	44.62	45.04	43.32	42	38	36.8
Hyrocks Charbon Works		1.55	1.1	1.82	5.6	5	4	
Unimin Lime(Tamworth)		5	0.6	0.6	5.6	5.6	5.8	5
Cement Aust (Fishermans Landing)	1600	26.2	54.3	74.6	64.2	55	56	71
Unimum Lime (Queensland)		1.43	2.53	2.93	0.02	2.49		
Cement Aus (Parkhurst)	140	0.96	0.45	0.56	4.06	0	0	
Adelaide Brighton Cement Ltd (Birkenhead)	1300	37	35	36	27	2.2	2	3
Adelaide Brighton Cement (Angaston)	220	0.26	1.2	5.86	5.44	14	13	1
Unimin Lime (Tasmania)		34.3	33.4	35.2	-	41	110	3.1
Cement Aust (Railton)	1120	14.82	123.8	11.86	120	119	120	118.7
Blue Circle (Waurn Ponds)	800	5.2	2.8	0	0.05	0.5	2	2.5
Unimin Lime (Victoria)		0.3	0.3	0.3	0.28	0.2	0.28	
Cockburn Cement (Munster WA)	570	7.38	3.6	4.2	190	180	0	0
Cockburn Cement –Lime (Dongara WA)		0.98	0.81	0.82	0.8	0.96	0	4.3
Total		184.23	313.22	267.2	518.87	511.0	247.3	272.

A general default input factor was not provided in the UNEP Toolkit due to lack of data. The USEPA[24] (1997a) developed an average atmospheric emission factor of 0.65 g mercury per tonne of clinker produced. If this factor were adopted, and based on the capacity data in Table 14, a mercury emission of about 5070 kg per year is obtained. The EMEP/CORINAIR emission guidebook[25] suggests an emission factor of 0.1 g/tonne where limited information is available; this translates into an emission of 780 kg per year.

The NPI emission factors of 0.11 and 0.024 g/tonne are applicable to cement kilns having different types of air pollution control devices (Electrostatic precipitators (ESPs) and fabric filters (FF) respectively).

Note 13 –Emissions from pulp and paper manufacturing

Mercury emissions from pulp and paper manufacturing as reported in the NPI are summarised in Table 15.

With the exception of Norske Skog in 2002 and 2003, reported emissions for a particular facility appear to be reasonably consistent. However emissions vary significantly from facility to facility especially when production rates are taken into account. For example, emissions from Norske Skog at

---

[24] http://www.epa.gov/ttn/chief/le/mercury.pdf

[25] EMEP/CORINAIR emission inventory http://reports.eea.europa.eu/EMEPCORINAIR5/en/B3314vs2.2.pdf

Boyer are consistently higher than those at Mary Vale, although throughput is higher at Mary Vale (almost twice).

Table 15: Mercury emissions from pulp and paper manufacturing as reported in the NPI

	Capacity[26] kt	Mercury emission kg/yr						
Facility		**2007**	**2006**	**2005**	**2004**	**2003**	**2002**	**2001**
Norske Skog Albury	265[27]	0.3	0.1	2.93	0	1.0	0	7
Visy Pulp & Paper Tumut	320	0.07	0	0	0	1	1	0.04
Norske Skog Boyer	290	10	2	19	5	261	114	14
Kimberly Clark Tantanoola		3	3	2	0	1.6	7	7
Australian Paper Burnie	130	2.91	2.63	2.89	2.75	2.85	2.9	2.72
Australian Paper Wesley Vale	44	0.19	0.2	0.2	0.75	0.2	0.2	0.2
Paper Aust Mary Vale	500	2.8	1	0.89	0.75	0.15	1.3	1.4

Note 14- Emissions from glass manufacturing

Mercury emissions from glass manufacturing as reported in the NPI are summarised below in Table 16. With the exception of emissions from Amcor Packaging and Insulation Solution in 2003, the NPI results are consistent between facilities and years.

The NPI mercury emission factor for glass manufacturing has been revised downward from 0.0019 to 0.00005 kg/tonne. The revised emission factor is consistent with those applied in the EU[25]. It should be noted that the NSW emission inventory used the 1998 NPI emission factor and reported a mercury emission of 722 kg from the ACI Penrith plant.

The UNEP toolkit does not consider Hg emission from glass container production as a major source nor does the UK NAEI inventory[28] where emissions of 2, 2 and 0 kg (not occurring) were reported for 2000, 2001 and 2002 respectively.

---

[26] Obtained from publicly available information

[27] 55% recycled fibre

[28] http://www.airquality.co.uk/archive/reports/cat08/0407081213_SimpleStudies_Yr2_report_issue1.pdf

Table 16: Mercury emissions from glass manufacturing as reported in the NPI

Facility	NPI estimated mercury emissions kg/yr						
	2007	2006	2005	2004	2003	2002	2001
ACI Operations Pty Ltd (NSW)	2	2	2	1	2	0	2
Pilkington (Aust)	1	1			2.8	2.5	2.3
Electric Lamp Manufacturers (Australia)							0.02
Pilkington (Australia)	0.439	0.406	0.62	0.0388	0.047	0.17	
Pilkington (Australia) Ltd	0.124	0.122	0.132	0.1098	0.112	0.082	0.134
ACI Operations P/L	1.8	1.8	2	2	1.9	1.1	1.6
ACI Operations	0.4	0.4	0.4	0.4	0.3	0.4	0.4
AMCOR Packaging Aust	0.62	0.657	0.526	0.16	269.1		
ACI Operations	1.91	1.83	1.05	1.9	1.7	1.8	2.23
Insulation Solutions	0.965	1.04	1.05	0.995	36.1	34	
P Q AUST	0.03	0.03	0.03	0.03	0.03	0.024	0.0121
Pilkington Aust	0.17	0.171	0.162	0.162	0.17	220	
Potters Industries	0.033	0.032	0.034	0.032	0.033	0.032	0.0319
ACI Glass Packaging[29]				0.208	0.55	0.6	0.4
Total	9.491	9.488	8.004	6.8276	314.292	263.108	8.728

Note 15- Emissions from Chlor-alkali production with mercury technology

The chlor-alkali plant at Botany closed in 2002. The building was not demolished until 2006/07. The 2007 emission was due to emission from the remaining building footings. The emission of 340 kg mercury in 2006 resulted from contaminated materials on site. Historical emissions from the site are summarised in Table 17.

Table 17: Mercury emissions from chlor-alkali production reported in the NPI

Facility	Mercury emission kg/year						
	2007	2006	2005	2004	2003	2002	2001
Orica Botany	7.9	340	340	340	240	140	120

Note 16- Emissions from use and disposal of thermometers and other measuring devices

Mercury has traditionally been used in devices measuring temperature and pressure given its properties of high and consistent thermal expansion and high density. Various types of measuring devices containing mercury exist on the market and can be found in households, laboratories, medical facilities, industries, and schools. These include: barometers, hydrometers, manometers, thermometers, sphygmomanometers and pyrometers.

---

[29] Manufacturing ceased in 2003

Non-mercury containing replacements for mercury containing instruments, such as digital thermometers are readily available[30] and it is likely that mercury released from inappropriate disposal of these devices will rapidly decrease in the future as the instruments containing mercury are replaced with suitable alternatives.

Mercury contents

Mercury concentrations in medical devices vary widely; from 0.5 g in medical thermometers to 200 g in industrial and special applications (UNEP 2005). Table 5-103 of the UNEP Toolkit (see Table 18) provides default mercury contents in thermometers.

Table 18: Default mercury contents in thermometers as provided in the UNEP Toolkit for Mercury Inventory Development

Thermometer type	Mercury content (g Hg/item)
Medical	0.5-1.5
Ambient air temperature	2-5
Industrial/ special applications	5-200
Miscellaneous glass (including laboratories)	1-40

Table 19 presents the average amount of mercury in each type of measuring devices sold in the US[31].

Table 19: Mercury content in measuring devices[31]

Product	Amount of mercury (g)
Barometers	400-620
Manometers	30-75
Sphygmomanometers	50-140
Thermometers	0.5-54
Psychometers	5-6

Disposal

In a report in 2002, the USEPA assumed that only 5% of thermometers are broken and required disposal. In contrast Barr Engineering (2001) assumes that 50% of thermometers in the USA are broken by consumers. Of the 50% of thermometer broken, Barr assumes that 20% of mercury ends up in wastewater through spill cleaning and 10% is lost through volatilisation. The balance is spread between municipal waste, infectious waste and recycling (Barr Engineering 2001).

Emissions from disposal of thermometers depend on the actual management practices employed. In the UNEP Toolkit, where no separate thermometer waste collection is available, as is the case in Australia, it is assumed that the distribution to air, water, land and general waste are 20, 30, 20 and 30 percent of the mercury disposed of in thermometers.

Thermometer quantity

There is no comprehensive data regarding imports of measuring devices. Data on import and export of thermometer-liquid filled (which may include non-mercury containing thermometers) and other

---

[30] http://www.ec.gc.ca/MERCURY/SM/EN/sm-mcp.cfm

[31] http://www.newmoa.org/prevention/mercury/imerc/factsheets/measuring_devices.pdf

measurement devices potentially containing mercury (hydrometers, pyrometer etc) in Australia[32] is provided below (Table 20). No further breakdown to types and sectors is available.

Table 20: Australian import of measuring devices

Year	Thermometers –liquid filled		Others[33]	
	Weight (kg)	**Quantity (item)**	**Weight (kg)**	**Quantity (items)**
2006	36427	767279	76392	620535
2005	81325	964089	179013	569793
2004	57893	1034023	51802	695686
2003		830372		716597
2002		647194		430113
2001		724893		501010

Estimated emissions from use and disposal

To estimate annual mercury emissions from disposal of products containing mercury, it is necessary to estimate the mercury in the disposed products. This is calculated by multiplying the number of disposed units by the mass of mercury within each unit. Distribution factors are then used to determine the fate of the disposed mercury i.e. whether it goes to landfill, is recycled or is released by breakage.

Because of the great variety of measuring devices included in this category, it is difficult to select an appropriate "average" mercury content for mercury containing measurement devices and it is also difficult to determine how many devices are broken or disposed of due to age or obsolescence. For the purpose of this calculation, we assume that all liquid filled thermometers and other devices imported (see Table 20) contain mercury and are replacements for those which have been broken (a likely overestimation). If it is also assumed that the average mercury content is 1g/item i.e. that mostly it was medical thermometers which were broken or replacements for those made obsolescent. The quantity of mercury available from these measuring devices is estimated as 1388 kg/yr in 2006. If 20% of this mercury is emitted into the atmosphere this would amount to an emission of 278 kg/year.

Similarly, the US inventory of mercury in measuring devices indicates that 4.8 tonnes of mercury was sold in 2004 (equivalent to 0.016 gram per person in the USA) [31]. Applying this factor would result in an estimation of 334 kg of mercury contained in measuring devices for the Australian population entering the community. Most of these devices are presumably to replace existing equipment either broken or which has otherwise been disposed of because of age or obsolescence. Assuming 20% of this mercury is emitted to atmosphere this results in a total atmospheric emission from this source of 67 kg.

The above figures have a high level of uncertainty and are likely to be an overestimation as the mercury emission depends on the types of products and yet the break down of product types is not available, the proportion of non-mercury containing fluid-filled thermometers and other devices imported is unknown as is the actual ratio of replacement devices for those previously broken or disposed of due to age or obsolescence.

---

[32] http://data.un.org/Data.aspx?q=+thermometer+and+australia&d=ComTrade&f=_l1Code%3a90%3bcmdCode%3a902511%3brtCode%3a36

[33] Includes hydrometer, pyrometer, hygrometer

The EU emission inventory guidebook[34] has a population-based emission factor for measuring equipment in Western Europe of 0.0044 gram per person. Applying this factor approximately 92 kg per year of mercury would be emitted from measuring devices in Australia.

For the purposes of this study the atmospheric emission rate of 92 kg/year from measuring devices is adopted, although it is acknowledged that it is possible that it maybe as high as 200-300 kg/year. Emissions from this source are likely to decrease in future years as more measurement devices are replaced by non-mercury containing equivalents. The amounts of mercury released to other pathways (ie water, land, and waste) are calculated using the default UNEP Toolkit (UNEP 2005) values of 20, 30, 20 and 30 percent to air, water, land and general waste respectively.

Note 17- Emissions from light sources with mercury

Limited information on the mercury content in lamps installed in Australia is available. Recent testing of mercury levels in new CFLs available in Australia found mercury levels varying from 0.1 mg to 13 mg[35]. Lamp manufacturers quote values of 3 mg and 5 mg per lamp (Parsons 2006). This is consistent with the European Union standard of 5 mg mercury per lamp[35].

Recent technological advances have allowed manufacturers to reduce the amount of mercury contained in linear fluorescent lamps. The mercury content has fallen from an average of 40 mg for a 1200 mm fluorescent lamp in 1985 to the present average of 12 mg[36]. This is consistent with values reported overseas. For example, in the US it was also reported that mercury from lamps has been reduced from an average of 48.2 mg per 1200mm lamp length in 1985 to 12 mg in 1999 (New Jersey Department of Environment Protection (NJDEP) 2002).

In this study, it is assumed that linear fluorescent tubes and CFL currently in the disposal stream contain 12 and 5 mg of mercury respectively.

While linear and compact fluorescent lamps are both used in domestic and commercial/industrial lighting another lighting type, high intensity discharge (HID) lamps are almost exclusively used in the non-domestic sphere. These lamps provide lighting often in outdoor situations such as stadia and parks, have a long life and have relatively high mercury contents[37] of 50-1000 mg/lamp. In this study a value of 500 mg/lamp is assumed for HID lamps.

The United Nation's data[38], provides information on export and import of commodities. In 2006, Australian net import of fluorescent lamps was 37 million (an insignificant number of lamps are manufactured in Australia). The split between linear fluorescent lamps and CFL is 60/40 which is generally consistent with those reported elsewhere (UNEP 2005). HID lamps imported into Australia in 2006 totalled 6.4 million[38], but no information on the split into various types of HID lamps was available.

Using the average mercury content and the quantities of lamps, and assuming that all lamps imported were to replace lamps destined for landfill as a result of obsolescence, it is estimated that 340 kg of mercury was contained in fluorescent lamps and a further 3200 kg in HID lamps. If disposal of these lamps results in 5% of the contained mercury being emitted into the atmosphere (UNEP 2005) this results in atmospheric emissions of 177 kg per year. The remainder is disposed of to land in the general waste stream.

---

[34] http://reports.eea.europa.eu/EMEPCORINAIR4/en/BMER.pdf
[35] http://www.energyrating.gov.au/pubs/2008-phase-out-session6-boughey.pdf
[36] http://www.lightingcouncil.com.au/pdf/Light_9.pdf
[37] http://www.newmoa.org/prevention/mercury/imerc/factsheets/lighting.pdf
[38] http://data.un.org/

Note 18-Emissions from batteries containing mercury

In the absence of available data on mercury content in batteries sold in Australia, the UNEP default mercury contents for mercuric oxide and silver oxide batteries were used to calculate emissions from this source (UNEP 2005). These mercury contents are 320 kg/tonne for mercuric oxide batteries and 4 kg/tonne for silver oxide batteries.

The United Nation's Data[39] provides information on export and import of commodities. In 2006, Australian net imports of mercuric oxide batteries and silver oxide batteries were 1 and 13.7 tonnes respectively[39]. Using the import information and the default mercury contents, it is calculated that 357 kg of mercury was contained in batteries imported in 2006. If we assume that all these batteries were to replace similar existing batteries and that 10% of the mercury content in batteries was emitted into the atmosphere through breaking/leakage (Kindbom and Munthe 2007), then the emission to air from batteries is estimated at 35.7 kg/yr. The balance of the mercury (321.3 kg) is retained on land in the general waste stream.

Note 19- Emissions of mercury from dental amalgam

Depending on the size and type of filling, approximate 0.4 to 1.2 g of mercury is used in each dental filling: 0.4 g per surface (Skarup *et al.* 2003). A National Health and Medical Research Council report (NHMRC 1999) showed the number of restorative services in Australia (Table 21).

Dental use of mercury has been steadily in decline; however more recent data than the 1997/98 is not available, therefore this data has been used to calculate mercury emissions from this source. Because of the decline in use the use of mercury amalgam and general improvements in dental health the use of this data is likely to overestimate the emissions from this source.

Table 21: Dental restorative services in Australia (NHMRC 1999)

Service type	Number of restorative services (x1000)	
	1993-1994	**1997-1998**
Amalgam (1 surface)	1264.6	720.5
Amalgam (2 surfaces)	2452.5	1624.1
Amalgam (3 surfaces)	1594.8	1120.7
Total	5311.9	3465.3

Using the above information it is estimated that 2.93 tonnes of mercury was used in dental services. This estimate represents consumption of 0.14 g mercury per inhabitant which is consistent with information from other countries of similar economic status which have consumption rates ranging from 0.01 to 0.22 g per inhabitant for this activity (UNEP 2005).

In detailed Danish studies (Magg *et al.* 1996; Skarup *et al.* 2003), it was estimated that 60% of amalgam is built into fillings, 25% is excess amalgam and disposed through general /special wastes and the balance (15%) is disposed to waste water system via dental chair filtering systems. It's likely that more responsible dentists separate their mercury wastes and have it collected for recycling.

---

[39] http://data.un.org/Data.aspx?q=battery+and+australia&d=ComTrade&f=_l1Code%3a85%3bcmdCode%3a850619%3brtCode%3a36

High efficiency filters are available to dental clinics and these filters can retain 95% of the amalgam in the waste water although many clinics may only have dental chair coarse filter/strainers which retain a much lower percentage of the waste amalgam. The percentage of dental clinics having high efficiency filters in Australia is not known.

The emission estimates given in Table 22 are calculated using UNEP default mercury output distribution factors for dental amalgam.

Table 22: Emission estimates for mercury from dental use in Australia

Phase in life-cycle	Emission kg/yr			
	Air	**Water**	**Products**	**Wastes**
Preparation of fillings in the teeth	58.6	410.5	351.9	351.9
Use from filling in the mouth		58.6		
Disposal –via clinic				
Dental clinics with high efficiency amalgam filters		58.6		1642
Dental clinics with only chair filters/strainers m filters		879.7		703.7
Total with high efficiency filter	58.6	527.7	351.9	1993.9
Total without high efficiency filter	58.6	1348.8	351.9	1055.6

Note 20- Emissions from laboratory chemicals and equipment

There are no data on the amount of chemicals containing mercury used in Australian laboratories. A number of chemicals such as mercuric chloride, mercuric oxide are controlled chemicals under the Rotterdam Convention.

The UNEP Toolkit indicated that in Denmark the use of mercury as laboratory chemicals was about 20-40 kg in 2001 (UNEP 2005). Denmark has a population of around 5 million. For the purpose of this estimation, it is assumed that use of mercury in Australian laboratories per inhabitant is the same as that in Denmark. The maximum emission of mercury is estimated to be 160 kg per year with the emission distributed equally between air and water.

Given that this category includes laboratory instruments, care needs to be taken to ensure that double counting does not occur with the earlier category including thermometers, barometers etc which are also used in the laboratory.

Note 21- Emissions of mercury in television sets

It had previously been reported that mercury had been used in television sets (cathode ray tube type). A report by the Danish Ministry of the Environment (Christensen *et al.* 2004) stated that this was not the case.

Mercury has been used in television relays having a concentration of less than 4 mg/set (Christensen *et al.* 2004). Based on this mercury content, the mercury consumption for television sets is estimated at 2 kg per year in Denmark (5.4 millions TV sets).

A survey of Australian electrical waste in 2005[40] found that 9.7 million TV sets are owned by households in Australia. It is not known whether all TV sets have relays containing mercury. If it is assumed that all television sets have mercury relays containing 4 mg of mercury and that 9% of TVs deemed obsolescent each year then the total mercury disposed in television sets amounts to 3.6 kg per year. It is assumed that all mercury would be disposed to land in general waste which may cause an over estimate based on e-waste collection programs operated in some locations in Australia.

Note 22 - Emissions of mercury in laptop computers/modems

Mercury switches and relays have never been used in personal computers. Although mercury was traditionally used in large mainframe computers, it is unlikely that the older mainframe computers are either still being used in Australia or represent a significant portion of computer waste currently being disposed of in landfills.

Flat screen laptops may use back-lit lamps for illumination. The average amount of mercury in a lap top computer is 0.12 to 5 mg[41]. The household waste survey in 2005 found that 1.24 million laptops are owned by householders in Australia and that 5% of these are disposed per year. Using the information, the maximum amount of mercury in the disposed laptops is estimated at 0.3 kg per year.

Relays used in modems may contain mercury in the order of 0.1 to 10 mg per relay (Christensen *et al.* 2004). According to the Australian household survey of electrical waste[40], approximately 3 million modems are used in Australia; of which 5% are disposed per year. The quantity of mercury disposed of in obsolescent modems is estimated to range from 0.015 to 1.5 kg per year.

Note 23 - Production of recycled mercury

There is one mercury recycler in Australia: CMA EcoCycle. This plant has been recently commissioned and no data on emissions is currently available.

Note 24- Production of recycled ferrous metals

Data for this source is derived from the NPI and summarised in Table 23.

Table 23: Mercury emissions from recycled ferrous materials based on NPI reporting

Facility	Mercury emission kg/yr						
	2007	**2006**	**2005**	**2004**	**2003**	**2002**	**2001**
Tyco Water Pty Ltd	1.42	0.54			11.7		0.54
Commonwealth Steel	56.1	56.1	59	56	56.1	78.5	282
Smorgon Steel	0.1	4.3	0.1	0.1	0.1	0.1	0.1
Bluescope Steel (NSW)	1.4	1.0					
OneSteel (SA)	0.15	0.62					
Total	**59.1**	**62.6**	**59.1**	**56.1**	**67.9**	**78.6**	**282.6**

---

[40] http://www.environment.nsw.gov.au/resources/warr/spd060220_ewaste_newsletter.pdf

[41] http://www.oecd.org/dataoecd/44/46/2741576.pdf

Note 25- Biomedical waste incineration

To the best of our knowledge there are currently no operating municipal waste incinerators in Australia. There are, however, two biomedical waste incinerators, Ace Waste (in Queensland) and Stericorp (in New South Wales). Emissions of mercury as reported in the NPI from these two facilities are summarised in Table 24.

Table 24: Mercury emissions from Australian medical waste incinerators reported in the NPI

Facility	Mercury emission kg/yr						
	2007	2006	2005	2004	2003	2002	2001
Ace Waste	230	232	191	189	180	170	175
Stericorp	18	3.6	0.59	0.32	0.11	0.26	
Total	248	235.6	191.59	189.32	180.11	170.26	175

Note 26- Emission from landfills

The NPI for 2006 includes a significant number of landfills. With one exception, all reported very low emissions of mercury (<0.01 kg/yr). By contrast, Leanyer Waste Disposal in Darwin reported emissions of 500 kg to land.

In Australia there is very limited information on the extent and nature of mercury emissions during or after landfill operation. No study on the potential fate of mercury in landfills has been undertaken nor has there been any detailed study on the composition of mercury-containing wastes in landfills.

To estimate mercury contained in landfills, the following assumptions are made:

- 8.7 million tonnes of municipal waste is generated per year in Australia (Australian Bureau of Statistics (ABS) 2006);
- mercury content of this waste is 2 ppm (New Jersey Department of Environment Protection (NJDEP) 2002). This is at the low end of the UNEP Toolkit (UNEP 2005) default range (1 to 10 ppm).

Using the above information, it is estimated that 17400 kg of mercury is deposited in landfills per year.

Rates of emissions from landfills are a function of waste type, landfill age, landfill status (active or closed) and operation characteristics (compacting, venting etc). To calculate landfill gas emissions, the following methodology is adopted.

The DECC inventory provided an estimate of VOC emissions from the Greater Metropolitan Region of 461,700 kg/year (2003 year). By applying the USEPA AP-42 VOC emission concentration of 235 ppmv (expressed as hexane) and the mercury content in the landfill gas of 2.92×10^{-4} ppmv, it can be estimated that 1.32 kg of mercury was emitted from all solid landfills in NSW (Department of Environment and Climate Change NSW (DECC) 2007). This is equivalent to 1.76 kg/yr for NSW or 0.26 kg per million population (Greater Metropolitan Region accounted for 75% of total NSW population).

Applying this emission factor of 0.26 kg per million people it can be calculated that 5.46 kg/yr of mercury is emitted from landfills into the atmosphere in Australia. This value is consistent with the air emissions as reported by the NPI.

Note 27–Emissions from crematoria

Significant amounts of mercury may be released from the volatilisation of mercury amalgam fillings in crematoria. Estimates of mercury release per cremation of a human body vary considerably and depend on the number of dental fillings per individual in a particular age category and on the size of the fillings.

Due to the lack of source test data for crematoria in Australia, reliance on international data is required to estimate emissions. Emission factors for cremation as reported in the literature are summarised below:

Country	Average emission per cremation (g)
UNEP Toolkit	1-4
Sweden[42]	2.0
Germany[42]	0.1-0.5
Belgium [42]	0.1
Switzerland [42]	1.1
France[42]	2.5
UK (2004) [43]	1.92
USEPA (1997a)	1.5
US 1999 National emission inventory	0.2027
Reindl (2005)	3.4

For the purpose of this exercise, the UK emission factor of 1.92 g per cremation was assumed to be the most appropriate for the Australian situation. The Australian Bureau of Statistics (ABS[44]) reported that there were 133700 deaths in 2006. The Australasian Cemeteries & Crematoria Association [45] (ACCA,) indicates that the cremation rate in Australia is 67%.

Using the emission factor of 1.92 g per cremation, emissions of mercury from cremation in 2006 are estimated to be 172 kg per year.

---

[42] http://www.ospar.org/documents/dbase/publications/p00179_Mercury%20emissions%20from%20crematoria.pdf

[43] Defra Mercury emission from crematoria Second Consultation 2004 http://www.defra.gov.uk/environment/ppc/old-consultations/crematoria-two/consultation.pdf

[44] http://www.abs.gov.au/ausstats/abs@.nsf/mf/3302.0

[45] Personal communication with the Australasian Cemeteries & Crematoria Association

3 Mercury Emissions from Natural Sources in Australia

3.1 Background and previous work

Natural sources of mercury emissions are broadly described in Section 1.3.1 above. Mercury is emitted from vegetation, soil, and water bodies. A large part (up to 50 percent) of the mercury that is emitted from natural sources is believed to be of anthropogenic origin (Mason *et al.* 1994a). As discussed above (Section 1.3.1), evaporation of mercury from the oceans, soil, vegetation and the release of mercury in forest fires, are a mix of natural and re-emitted mercury. Hence "natural" in this context may be somewhat misleading; in the context of this report "natural emissions" will by definition also include re-emissions.

In this section, the derivation of mercury fluxes from continental Australia is described. Included are emissions from vegetation, soil, water bodies and fires.

In order to derive estimates of the annual natural mercury flux from Australia, individual flux measurements of mercury must be integrated over space and time, taking into account three major complexities:

- Firstly, there appear to have been no flux measurements from natural surfaces in climates similar to those of Australia (eg. temperate forests, tropical savannas, rainforests and deserts); all available measurements of natural mercury emissions originate from the temperate Northern Hemisphere, which has different types of climate, vegetation and very different land use patterns than Australia.
- Secondly, many variables (eg. temperature, solar radiation, mercury content in the emitting media, precipitation, diurnal and seasonal trends) affect the magnitude of mercury flux across the air-surface interface. The impact of these variables on mercury the flux is likely to be significant due to the peculiarities of the Australian climate, with high average temperatures and levels of solar radiation.
- Thirdly, there are considerable uncertainties involved in the integration of studies carried out over limited space (or area) and time to represent the complexities of a continent, particularly to account for natural cycles exceeding years.

In carrying out calculations of Australian natural source mercury emissions the following information sources are used and/or simplifying assumptions are made:

- Studies of mercury flux during the Northern Hemisphere summer are assumed to represent average Australian climatic conditions;
- Fluxes for Northern Hemisphere forests are assumed representative of Australia even if the particular species do not exist in Australia. Emission rates for low altitude forest (Lindberg *et al.* 1998b) are used, and forested area is estimated for all forest types;
- Emission values from Northern Hemisphere lakes are used to represent all permanent inland water bodies in Australia for which the area can be determined; and
- For desert, bush and grassland areas, Northern Hemisphere flux measurements from rural grassy sites are adopted, though it is recognised that desert areas are likely to have lower emission factors than Northern Hemisphere grasslands due to the absence of significant biomass cover – and very different regimes of water and soil moisture.

Mercury emission from anthropogenic sources can also be deposited to aquatic and terrestrial surfaces through wet and dry processes. Following the deposition on the surface, some of mercury is then reduced chemically or biologically to highly volatile species which are re-emitted back to the atmosphere (Stein et al 1996). This bi-directional exchange between air-surface interfaces makes it difficult to distinguish between the natural source and re-emission of previously deposited Hg. For the purpose of this report, natural emissions include re-emissions.

Natural emissions exhibit diurnal and seasonal variations and strongly depend on temperature, radiation, vegetation coverage and mercury concentrations in soil/ ambient air (Shetty et al 2007). The global emission inventory suggested that Hg emission from natural sources including re-emission is of the same order of magnitude as anthropogenic sources (200-4,000 tonnes) (Mason *et al.* 1994b; Bergan and Rodhe 2001).

The relative impact or contribution from natural sources in Australia has previously been reported by Nelson *et al* (2004). In that study it was estimated that 117 -567 tonnes per year of mercury is emitted from land and water surfaces in Australia. Approximately 45-65 tonnes of Hg re-emitted from Australian natural sources originate from previously deposited Hg. This inventory assumed a minimum and maximum range of fluxes from forest and lake. For areas that are not forest or lakes, a range of Hg flux of 1.5 to 3 ng $m^{-2}hr^{-1}$ was used. No emissions from bush fires were included in the that earlier inventory (Nelson *et al.* 2004).

In the present report, mercury emissions from natural sources have been re-examined by a thorough review of published data and by a modelling approach summarised in Section 4.

3.2 Bi-directional exchange of mercury

There are a number of studies that have estimated the fluxes of mercury on a global scale to and from the atmosphere. Recent research indicates that forests act as a dynamic exchange surface which can behave either as a source or sink depending on ambient Hg concentration and Hg concentration in the transpiration stream (Hanson *et al.* 1995). In general, there is a net emission of Hg out of vegetation (Lindberg *et al.* 1998a). Canopy area (total leaf surface area), temperature, solar radiation and wind turbulence influence the Hg flux. The flux is also indirectly influenced by soil Hg concentration through the plant uptake of mercury. For grassland and forest bark and stems the Hg flux is 5.5 to 9.5 times less than from the canopy due to a much lower exposed plant surface area (Hanson *et al.* 1997).

Leonard et al (1998a; 1998b) developed a linear regression equation relating Hg flux from forested areas to the soil Hg concentration. That equation (as below) has been used in estimating natural emission from vegetation in Canada (Richardson *et al.* 2003), and Table 25 summarises the data used in the calculations for Canada.

Mercury emission from vegetation shows a strong diurnal variation with higher values during daytime and lower at night. The diurnal variation of Hg emission flux can be estimated using the correction factors of solar irradiation and temperature (Lin *et al.* 2005).

$$F_i = F_{s,i} * C_T * C_L$$

Where:

F_i is the estimated mercury emission flux (ng m^{-2} h^{-1}) for a given land use type or vegetation species

F_{si} is the average daytime emission factor for species i

C_T are temperature corrections

C_L is a solar radiation correction

The typical summer ranges for C_L and C_T are 0-1.6 and 0.5-1.8, while the typical winter ranges of C_L and C_T are 0-1.5 and 0.05- 0.16 respectively. It should be noted that the values for winter are for surfaces covering in snow which seldom occur in Australia.

Table 25: Method for determining mercury vapour flux due to evasion from terrestrial plants (from Richardson *et al* (2003))

			Values for variable distributions from probabilistic computations			
Variable	**Description**	**Units**	**Minimum**	**Most Likely**	**maximum**	**Distribution**
$F_{Hg0\text{ - }Plants}$	Flux of Hg vapour to atmosphere due to evasion of vapour from terrestrial plants	kg a^{-1}	calculated			
$C_{Hg\text{-}soil}$	Concentration of Hg in soil	µg kg^{-1}	22	62	102	Triangular
SA_{forest}	Surface area of forested land in Canada	km^2		5 350 000		Constant
$SA_{grassland\ /bushland}$	Surface area of land covered by grassland and brushland in Canada	km^2		606 011		Constant
$R_{SA\text{-}forestcapony.SAland}$	Ratio of the surface area of the forest canopy to the surface area of land covered by forest	unitless	5.5	7.5	9.5	Triangular

Table 26 summarises average emission flux for various vegetation types. It should be noted that the range between and within species and types is very large.

Table 26: Average mercury emission fluxes for various types of vegetation as reported in the literature

Type /Location	Hg flux ng m^{-2} h^{-1}	Reference
Mature hardwood, TN USA	8-66	Lindberg et al.(1998a)
Young pines , TN, USA	1-35	Lindberg et al.(1998a)
Pines USA	4-18	Lindberg et al.(1998a)
Maples	4-37 [46]	Hanson et al.(1995); Lindberg et al.(1998a)
Oaks	5.3-37[46]	Hanson et al.(1995); Lindberg et al.(1998a)
Cattail canopy , FL , USA	17±43	Lindberg et al. (2002)
Agricultural North East , USA	52	Xu et al. (1999)
Agricultural and other forest	4	Lindberg et al.(2002)
Irrigated agricultural USA	4.0	Lin et al. (2005)
Irrigated agricultural China	21.5	Hou et al. (2005)
Rural grassy site, Quebec	0.62-8.3	Poissant & Casimir (1998)
Mixed forest[47]	28	Xu et al. (1999)
Deciduous forest[47]	30	Xu et al. (1999)
Deciduous forest	7-290	Lindberg et al.(1998a)
Forest (foliage)	5.3	Hanson et al.(1995)
Bark/stems	1.2-2.5	Hanson et al.(1997)
Forest	5.5	Fang & Wang (2004)
Vegetation China	0-66	Shetty et al. (2007)
Forest sites Chongqing, China	3.5±1.2-8.4±2.5	Wang et al.(2006a; 2006b)

---

[46] Estimated flux due to mercury contaminated soils

[47] Model

Farming field site Chongqing, China	85.8±32.4	Wang et al.(2006a; 2006b)
Grass land and manmade forest sites Chongqing, China	12.3±9.8 -733.8±255	Wang et al.(2006a; 2006b)
Shaded forest MI USA	-0.2-2.4 Mean 1.4	Zhang et al. (2001)
Various deciduous forest soils with liter TN USA	2.0-6.98 Apr to Aug 1995 0.98 -5.95 May to June 1995 Mean 2.7	Capri and Lindberg (1998)
Pasture QC Canada	3.0	Schroeder et al. (2005)
Pasture ON Canada	1.1	Schroeder et al. (2005)
Fallow field ON Canada	2.9	Schroeder et al. (2005)
Field MI USA	7.6	Zhang et al. (2001)
Open agricultural sites TN, USA	-0.66-1.21	Capri and Lindberg (1998)
Open agricultural sites TN, USA	16.8 [48] 44.83[49]	Capri and Lindberg (1998)
Rural un-vegetated Italy	1.5-30	Ferrara et al. (1997)

3.2.1 Air-soil exchange

Mercury emission from soil can be classified into emissions from bare soil and emission from soil under a vegetation canopy. Research has shown that mercury emissions from soil are dependent on meteorological and parameters such as soil temperature, soil moisture and solar radiation (Kim *et al.* 1995). The above factors are important in controlling temporal variations; however, the dominant factor controlling the magnitude of the flux is substrate Hg concentration and that is dictated by the geological properties of the area.

The influence of meteorological parameters on mercury emissions is not fully understood. Consequently considerable uncertainty exists in scale-up of mercury flux measurements to regional scales. Mercury emission from bare soils depends on soil temperature, solar radiation, moisture and mercury concentration. A strong relationship exists between the log of the soil Hg concentration and the log of flux from soil (Rasmussen *et al.* 1998). Richardson (2003) developed a regression relationship which was used in estimating emissions from Canadian soils.

Similarly Gbor et al. (2006) developed the flux-temperature relationship for bare soil:

$$\ln F_s = -\beta/T_s + n*\ln [Hg]_s + \gamma$$

Where:

F_s is Hg flux from soil (ng m^{-2} h^{-1})
T_s is soil temperature (K)
β is related to the activation energy and equals to $1.26*10^4$ K
γ =38.67
n equals 1.

Emission of Hg from soil under the canopy correlates better with solar radiation than soil temperature (Gbor *et al.* 2006). Daytime flux measurements indicated that mercury emissions from the forest floor are usually much smaller than above the canopy, an average of 7.5 vs 100 ng m^{-2} h^{-1} was estimated by Lindberg et al. (1992). Table 27 summarises average emission flux for various soils.

---

[48] Mean fluxes for shaded soil

[49] Mean Flux for soil exposed to sunlight

Table 27: Average emission fluxes of mercury for various soils

Type/location	Period	Hg flux rate (ng m^{-2} h^{-1})	Reference
Urban area –Semi arid Reno Nevada USA	Spring/summer	2.2±0.4	Stamekovic et al.(2007)
Urban area Reno Nevada USA	Winter	2.5 ±0.8	Stamekovic et al.(2007)
Urban area USA		2.0-4.1	Gabriel et al. (2005a; 2005b)
Urban area Reno Nevada USA		2.3	Stamekovic et al.(2007)
Urban area (Northern Hemisphere)		1.5-2	Stamekovic et al.(2007)
Rural area[50]- (Canada)		1.55±0.39	Kellerhals et al. (2003)
National park – Softwood forest Canada		1.33±0.25	Kellerhals et al. (2003)
Deciduous forest[51]-Canada		1.9±0.43	Kellerhals et al. (2003)
Mining area Ivanhoe [52] Nevada		9.2-17.1	Engle et al. (2001)
Mining area Ivanhoe [53] Nevada		17.5	Gustin et al. (2002)
New Idria Mining District CA USA		3.2	Gustin et al. (2002)
Steamboat Springs area Nevada		300	Gustin et al. (2002)
Flowery Peak area Nevada		28	Gustin et al. (2002)
Medicine Lake CA		2.6	Gustin et al. (2002)
Sulphur Bank[53] USA		436	Gustin et al. (2002)

3.2.2 Air-water exchange

Atmospheric deposition of mercury to water is driven by particle dry deposition and wet scavenging through precipitation mechanisms (Schroeder and Munthe 1998a). Table 28 summarises reported measurements of fluxes of mercury from water surfaces.

Table 28: Average emission fluxes of mercury from water surfaces

Country	Surface type	Period	Hg Flux rate ng m^{-2} h^{-1}	Reference
USA	Wetland	Summer	40	Lindberg et al. (2002)
	Wetland	Winter	20	Lindberg et al. (2002)
North Sea			1.6-2.5	Cossa et al.(1996); Ferrara et al. (2000)
Scheldt estuary (Belgium)			5.8	Ferrara et al. (2000)
Sweden			20.5	Ferrara et al. (2000)

---

[50] Northeast coast of Canada, area largely devoid of vegetation

[51] Lake &deciduous forest-surrounded by urban & industrial centres

[52] Mining area –undisturbed substrate

[53] Mining area, substrate high in Hg and high geothermal activity

Coastal Atlantic			2.7	Gardfeldt et al. (2003)
River Sweden		Summer	-2.5-88.9 Mean 11.0	Gardfeldt et al. (2001)
Sea surface Sweden		Summer	-0.35-0.42	Gardfeldt et al. (2001)
Marsh CT USA		Summer	-3.3 to 13.2 Mean 4.5	Lee et al. (2000)
Mediterranean sea		Summer	4.0-5 (day) 1 (night)	Pirrone et al. (2001)
Mediterranean sea		Winter	1.5	Pirrone et al. (2001)
Watershed/rivers			1 -3 Mean 1.7	Grigal (2002)
Water Chongqing, China			5.9±12.6 - 618±339	Wang et al. (2006a; 2006b)
Northern-Tyrrenian [54]Italy		Summer	11.25 (day) 2.4 (night)	Ferrara et al. (2000)
Northern-Tyrrenian [55]Italy		Summer	0.7-10.1 (day) 1 (night)	Ferrara et al. (2000)
St Anicet Quay QC Canada		Summer	0-0.96	Schroeder et al. (2005)
St Lawrence River ON		Summer	0.02-9.28 Mean 3.07	Schroeder et al. (2005)
Lake Ontario North Shore ON Canada		Summer	0-2.8 Mean 0.82	Schroeder et al. (2005)
Lake Ontario South Shore ON Canada			0.9-9.07 Mean 3.39	Schroeder et al. (2005)
Fresh water lake , norther QC		Summer	0.59-4.5 Mean 0.31	Schroeder et al. (2005)
Various fresh water lakes NS		Summer	0.6-2.2	Schroeder et al. (2005)
Halifax Harbour Saltwater NS		Summer	0.7	Schroeder et al. (2005)
Wetland QC Canada		Summer (dry)	4.5	Schroeder et al. (2005)
Wetland QC Canada		Fall	-1.5-2.4	Schroeder et al. (2005)
Wetland QC Canada		Summer (wet)	-0.46-7.13 Mean 0.95	Schroeder et al. (2005)
Lake Gardsjon Sweden		Summer	2.5-10.4 Mean 6.9	Xiao et al. (1991)
Lake Skarsjon Sweden		Summer	1.3-6.4 Mean 4.2	Xiao et al. (1991)
Lake Hallungen Sweden		Summer	1.6-8.0 Mean 4.5	Xiao et al. (1991)
Atlantic Ocean			3.17	Xu et al. (1999)
Everglades FL,USA		Summer	17	Lindberg and Meyers (2001)
Wetlands QC Canada		Summer	-110-278 Mean 32.1	Poissant et al. (2004)

---

[54] Polluted coastal zone

[55] Un-polluted coastal zone

In addition to emission fluxes from soil, vegetation and water, mercury is emitted due to wind erosion. Metal concentrations in soil particles entrained into the atmosphere due to wind erosion are more or less equivalent to those found in surface soil (Eltayeb *et al.* 1993). Richardson et al. (2003) has developed a method to estimate Hg flux due to entrainment of soil particles shown in Table 29 below.

Table 29: Method for determining Hg flux due to entrainment of soil particles (from Richardson et al. (2003))

			Values for variable Distributions from probabilistic computations			
Variable	**Description**	**Units**	**Minimum**	**Most Likely**	**maximum**	**Distribution**
F_{Hg-SP}	Flux of Hg vapour to atmosphere due to suspension of soil particulate matter	Kg a^{-1}	Calculated			
$F_{SP-shrubland}$	Flux of soil particles to atmosphere for shribland	kg km^{-2} a^{-1}	526	61 015	431 225	Triangular
RF_{biomei}	Frequency of dusty periods for other biomes, relative to shrubland	unitless	Calculated			
C_{Hg-SP}	Concentration of Hg in soil particles	kg kg^{-1}	0.022	0.062	0.102	Triangular
SA_{biomei}	Surface area of biome i	km^2	Based on input data			

In this approach the total flux for Canada is given by:

$$F_{Hg-SP}\ (\mathrm{kg\ a^{-1}}) = F_{SP\text{-}shrubland}\ (\mathrm{kg\ km^{-2}\ a^{-1}}) \times RF_{biomei} \times C_{Hg-SP}\ (\mathrm{mg\ kg^{-1}}) \times SA_{biomei}\ (\mathrm{km^2})$$

summed over all biomes.

3.2.3 Speciation of mercury in natural emissions

Hg is emitted to the atmosphere in both elemental Hg(0) (primarily gaseous), divalent Hg(II) (gaseous or particulate), and organic (as methylmercury) form. Anthropogenic emission sources primarily emit Hg in any of three forms: elemental Hg(0), gas-phase inorganic Hg(II), and particulate Hg(p). The natural sources emit mercury mostly as gaseous Hg(0), although there is some mercury associated with "natural" particulate phase emissions such as those from volcanoes and dust (Mason 2008).

3.3 Calculation of mercury emissions from natural sources in Australia

The review of previous work on mercury emissions from natural sources summarised above, reveals the very large variability in the fluxes measured. In addition the review revealed no measurements made under Australian conditions. For this reason any estimate of mercury emissions from natural sources in Australia will be highly uncertain.

A number of approaches were taken to derive an estimate for various natural sources in Australia. In Section 4 an estimate for all sources is made using a modeling approach. The inputs for this estimate are described in detail in that section. A key input is the soil mercury content and a description of the existing Australian data for mercury content in soils is given below, together with an estimate of mercury emissions from vegetation using an approach similar to that of Richardson *et al* (2003).

3.3.1 Australian soil mercury contents

There is very little extant Australian soil mercury data. The best source of data is from a CSIRO study (Carr *et al.* 1986) of the use of mercury as a pathfinder for base metal and Uranium deposits. The data from this study is summarised in Table 30.

Table 30: Australian soil mercury contents for a range of soil types (data from Carr *et al* (1986)

Soil type (locality where measured)	Background mercury (ppb)
Transported sandy soil (Woodcutters, NT)	5-20
Acid peats, peaty podsols (Western Tasmania)	70-290
Duplex soils (Currawang, Eastern NSW)	10-70
Red earth (Elura, Western NSW)	5-25
Lateritic soil (Ranger, NT)	25
Thin lithosols (Dugald River, Western Qld)	20

In most localities where mineralization occurs there are significant increases in the soil mercury content in the immediate vicinity of the deposit. These mercury 'anomalies' can be 5-10 times greater than the background mercury content of the soils in the surrounding area.

Geoscience Australia is currently undertaking a national geochemical survey[56] of the Australian continent on a 100 km grid. This survey might have provided further data on soil mercury but the work will not be completed until at least 2010. Because of the low levels of mercury in the regolith and the need to adopt special techniques to achieve detection at the ppb level it is possible that this survey may not result in a meaningful addition to our knowledge on soil mercury concentrations. The

---

[56] http://www.dpi.nsw.gov.au/minerals/geological/about/mapping/national-geochemical-survey-of-australia

earlier geochemical pilot surveys of the Riverina Region (NSW and Victoria), Gawler Region (South Australia) and the Thomson Region (Northern NSW) had a mercury detection limit of 5 ppm (5000 ppb), and in all cases mercury contents in these samples were below this limit[57].

These soil mercury values were used to derive emissions from this source as described in Section 4.

3.3.2 An estimate of mercury emissions from vegetation

An estimate of the emissions from vegetation was made using the Australian continental vegetation files[58]. Table 31 shows the emission factors used for the various vegetation types. Because of the lack of specific Australian vegetation data, five limited grouping were used to derive an estimate of total emissions from this source.

The emission factors in Table 31 were sourced from the review of mercury emissions from vegetation as described in Section 3.2. The computation used these emission factors and the area of the areas of the groupings of vegetation types to derive the following estimations (Table 32) based on the high and low emission factors.

The low estimate figure is of similar magnitude to that estimated previously by Nelson *et al* (2004). It is useful to compare the present estimate with that previously made by Nelson *et al* which is based on a report prepared for the CRC in Sustainable Development (Peterson *et al.* 2004). This report summarises natural emission estimates for mercury as follows:

> Using the Hg flux data presented in table 9, a total annual emission of Hg from natural sources (including urban areas) to the atmosphere is estimated to be in the range of 130 - 270 tonnes/yr (with a mean of 200 tonnes/yr), representing an emission rate of 17 - 35 $\mu gm^{-2}yr^{-1}$ (or 2 - 4 ng $m^{-2}h^{-1}$). Since there are no similar investigations performed in Australia it is difficult to verify the results. If the result in this study is compared to the average annual natural Hg emissions from Europe of approximately 250-300 tonnes/yr (25-30 $\mu gm^{-2}yr^{-1}$or 2.9-3.4 $ngm^{-2}h^{-1}$)[59] (Axenfeld et al., 1991; Pacyna et al., 2001) and to the estimated emission from the earth's total landmasses of 1000 - 3200 tonnes/yr (7.7 - 24 $\mu gm^{-2}yr^{-1}$ or 0.8 - 2.7 $ngm^{-2}h^{-1}$)[60,61](Mason et al., 1994; Lindberg et al., 1998), the upper end of the calculated range of Hg emissions for Australia is somewhat high. If the average emission rate from the earth's landmasses is used instead of the fluxes listed in table 9 (reproduced as here)the average emission from land sources in Australia is within the range of 54 - 182 t/yr, with a mean of 118 t/yr. Thus, an emission rate of 130 tonnes/yr is therefore probably the most reasonable assumption based on these estimates.

---

[57] http://www.crcleme.org.au/Pubs/OPEN%20FILE%20REPORTS/OFR%20234/OFR%20234.pdf

[58] See: http://www.ga.gov.au/nmd/mapping/atlas_soil.jsp ; http://www.ga.gov.au/nmd/mapping/atlas_veg.jsp

[59] The area of Europe is 9 892 923 km^2 (Geoscience Australia, 2002, http://www.auslig.gov.au/facts).

[60] The total global landmass (excluding the Antarctic continent) is 135 774 000 km^2 (ABS, 2002a).

[61] There is a large variation concerning emission from land base sources, for instance, Bergan et al., 1999 estimated the emission from land sources in a model simulation to be 500 tonnes/yr. However, when the study was revised two years later the corresponding value had risen to 1320 tonnes/yr, i.e. roughly by a factor of 3 (Bergan and Rohde, 2001).

Table 31: Mercury emission factors for vegetation types used to calculate total Australian mercury emissions from vegetation

CODE	VEGETATION DESCRIPTION	MERCURY EMISSION (ng/m2/h)		REFERENCE	
		LOW	HIGH	LOW	HIGH
0	ocean	1.6	3.17	20	18
1	tall dense forest	4	66	8	8, 4
2	tall mid-dense forest	4	66	8	8, 4
3	dense forest	4	66	8	8, 4
4	mid-dense forest	4	66	8	8, 4
5	sparse forest (woodland)	1	35	8	8
6	very sparse forest (woodland)	1	35	8	8
7	low dense forest	1	35	8	8
8	low mid-dense forest	1	35	8	8
9	low sparse forest (woodland)	1	35	8	8
10	tall mid-dense shrubland (scrub)	1	35	8	8
11	tall sparse shrubland	1	35	8	8
12	tall very sparse shrubland	1	35	8	8
13	low mid-dense shrubland	1	35	8	8
14	low sparse shrubland	1	35	8	8
15	low very sparse shrubland	1	35	8	8
16	sparse hummock grassland	0.62	8.3	12	12
17	very sparse hummock grassland	0.62	8.3	12	12
18	dense tussock grassland	0.62	8.3	12	12
19	mid-dense tussock grassland	0.62	8.3	12	12
20	sparse tussock grassland	0.62	8.3	12	12
21	very sparse tussock grassland	0.62	8.3	12	12
22	dense pasture/herbfield (perennial)	1.1	4	11	7
23	dense pasture/herbfield (seasonal)	1.1	4	11	7
24	mid-dense pasture/herb (perennial)	1.1	4	11	7
25	mid-dense pasture/herb (seasonal)	1.1	4	11	7
26	sparse herbfield*	1.1	4	11	7
27	very sparse herbfield	1.1	4	11	7
28	littoral	0	2.8	13	13
29	permanent lake	0.6	10.4	13	21
30	ephemeral lake (salt)	0.8	0.8	29	29
31	urban	1.55	2.5		22

REFERENCES

4	Hanson et al 1995
7	Lin et al
8	Lindberg 1998
11	Lindberg et al 2004
12	Poissant & Casimir 1998
13	Schroeder et al 2005
18	Cossa et al 1996
20	Gardfeldt et al 2003
22	Stamekovic et al 2007
29	Lee, Benoit and Hu 1999

Table 32: Annual estimation of total Australian mercury emission from vegetation

	Total Hg (low)	Total Hg (high)
Annual Hg (tonnes)	63	1315
Average hourly emission rate ($ng/m^2/h$)	0.93	19.5

Table 33: Estimated emission of Hg from natural land surfaces in Australia (reproduced from Table 9 in Petersen *et al* (2004))

Land cover	Q range (ng m^{-2} h^{-1})	Q_{min} (ng m^{-2} h^{-1})	Q_{max} (ng$m^{-2}$$h^{-1}$)	Area km^2	M_{min} ton/yr	M_{max} ton/yr
Forest[a]	7-290, 22, table 8	3.5	7.7	1644120	50.4	111
Lake[b]	1, 2.05-20.5	0.5	10	15267	0.1	1.5
Rural grassy sites/Soil/ Other[c]	1,4-5,0.62-8.3(2.95), 9.67, 32, 12-45	1.5	3	6032573	80	156
Total[d]				7691960	130.5	268.5

[a] BRS, National Forest Inventory, 2001. http://www.brs.gov.au/npi/. 3.5 ng$m^{-2}$$h^{-1}$is the mean of 7, however, it is also close to the average estimates of 3.3 ng$m^{-2}$$h^{-1}$ from forests presented in table 8. The flux of 7.7 is from table 8.

[b] Geoscience Australia, 2002. (Area of major lakes). http://www.auslig.gov.au/facts/landforms/. The two estimated fluxes are the average of the presented range.

[c] Area by difference. There is no flux measurement from eg, urban and alpine areas. The flux is regarded as a mean for all thinkable surfaces other than the two previously listed. Qmin is the mean of 2.95 and Qmax is the mean of 1 + 5 ng$m^{-2}$$h^{-1}$which is measured in South Europe.

[d] The total area is from Geoscience Australia, 2002. http://www.auslig.gov.au/facts/dimensions/.

The estimated average annual emission rates are also in keeping with the overall emission rates from vegetation calculated for China by (Shetty *et al.* 2008), who calculates an average continental emission rate of 5.5 ng/m^2/h based on annual emissions of 462 tonnes and a land area of 9,596,960 km^2.

3.4 Mercury Emissions from Fires

As vegetation contains mercury, mercury emissions from all types of vegetative burning will release mercury into the atmosphere. There have been two recently published estimates of this potentially large source which are of relevance to Australia.

Packham *et al* (2009) made measurements or estimates of mercury in biomatter and soils for 29 locations around Australia. Using these results and by identifying bushfire fuel zones with an estimated burning frequency they estimate total emissions of approximately 129 tonnes/year.

In a very recent study Friedli et al (2009) estimated global emissions of mercury from biomass burning. They claim that “frequently burning grasslands in Africa and Australia…contribute relatively little to the mercury budget”, and estimate total Australian emissions of 19 ± 9 tonnes/ year.

The companion modelling study to this report (appendix 1, (Cope *et al.* 2009)) estimated bushfire emissions for 2006 to be 41.8 tonnes, which falls between these two estimates. The methodology used there was to estimate the burned areas using satellite fire-scar and hotspot data, and then combine these with estimated fuel loads and emission factors to determine mercury emission rates at a temporal resolution of 1 hr and a spatial resolution of 1 km. For 2006, approximately 62×106 ha of vegetation was estimated to be burnt, not including emissions due to agricultural burning.

It is clear that this source is very significant but also subject to considerable uncertainty.

4 Transport and Fate of Mercury in Australia

Detailed results for the transport and fate of mercury in Australia are given in the reports attached as an appendix to this report (Appendix 1; (Cope *et al.* 2009)). Here a summary of the results of this modeling only is given.

This component of the study entailed the use of numerical meteorological and transport models and the air emissions inventory for mercury (as discussed above) to generate best estimates of annual average ambient mercury concentrations and wet and dry deposition rates. *Wet deposition* is the transfer of a substance, in this case mercury, from the atmosphere to the surface via precipitation. In this regard it should be noted that elemental gaseous mercury is relatively insoluble, reactive gaseous mercury is very soluble and particulate mercury is readily scavenged by cloud water droplets (Seigneur et al. 2001). Thus it may be expected that the majority of the mercury mass deposited by precipitation will be in the form of reactive gaseous mercury and particulate mercury. *Dry deposition* refers to the transfer of gas and aerosol phase mercury to "sinks" on vegetation (such as leaf stomata), soil and water surfaces by atmospheric turbulence and molecular diffusion. For particulate mercury, deposition rates may also be enhanced by gravitational settling of the particles.

The modelling was undertaken over three spatial scales - for the Australian continent; for the urban regions of Melbourne and Sydney; and for five significant point source locations.

The modelling included best available estimates of natural and anthropogenic sources to estimate total mercury concentrations and deposition rates. The natural source group considers the emissions from soils, vegetation, water and fires. The anthropogenic source group includes industrial emissions as well as emissions from the commercial, domestic and transport sectors. A mercury concentration of 1.3 nanograms[62] per cubic metre of air (ng m^{-3}) was included in the model calculations to represent the global background contribution advected through the model boundaries.

Natural emissions were estimated to contribute 93% of total mercury emissions in Australia with soil emissions being the largest single source (69% of total), followed by bushfires (20%), and vegetation (4%). Industrial sources (6.8% of the total) dominated the anthropogenic emissions with only 0.4% coming from commercial and domestic sources.

Annual average mercury concentrations at the continental scale were dominated by the global background (1.1–1.3 ng m^{-3}), with increases evident at the regional scale in the vicinity of fires and major industrial sources (up to 3 ng m^{-3}). A similar range of concentrations was modelled at the urban scale for Melbourne and Sydney. However, fine scale modelling predicted concentrations to be up to 10 times larger within the first few kilometres of several significant industrial sources. This is consistent with the finer scale modelling providing better resolution of the high concentration gradients near point sources and hence predicting higher maximum concentrations.

The modelled concentration results may be compared with concentrations determined at the Macquarie University campus in Sydney (Nelson *et al.* 2009). Although these measurements are not part of the formal brief, Macquarie University is undertaking atmospheric measurements of total and particulate mercury as part of a broader study. Macquarie University has made a major investment in continuous instrumentation for the measurement of trace atmospheric mercury species. The Tekran® Model 2537A performs continuous measurement of total gaseous mercury (TGM) in ambient air with a sampling time as low as 2.5 minutes and a detection limit of <0.1 ng/m^3. This facility has recently been enhanced by the addition of the Tekran® Model 1130 Mercury Speciation Unit which gives the Model 2537A Mercury Vapor Analyzer the ability to simultaneously monitor both elemental and reactive gaseous mercury (RGM) species in ambient air.

---

[62] A nanogram is one billionth of a gram - 10^{-9} g

Figure 5 show results for measurement of total gaseous mercury (TGM) on the Macquarie University campus in December 2007. The data clearly reveal a background TGM concentration of ~ 1 ng/m^3. This is consistent with the few previous measurements for the global Southern Hemisphere background (Lindberg *et al.* 2007a). There is also evidence for early morning concentrations of TGM which significantly exceed this background concentration, suggesting some significant urban sources, probably ground based.

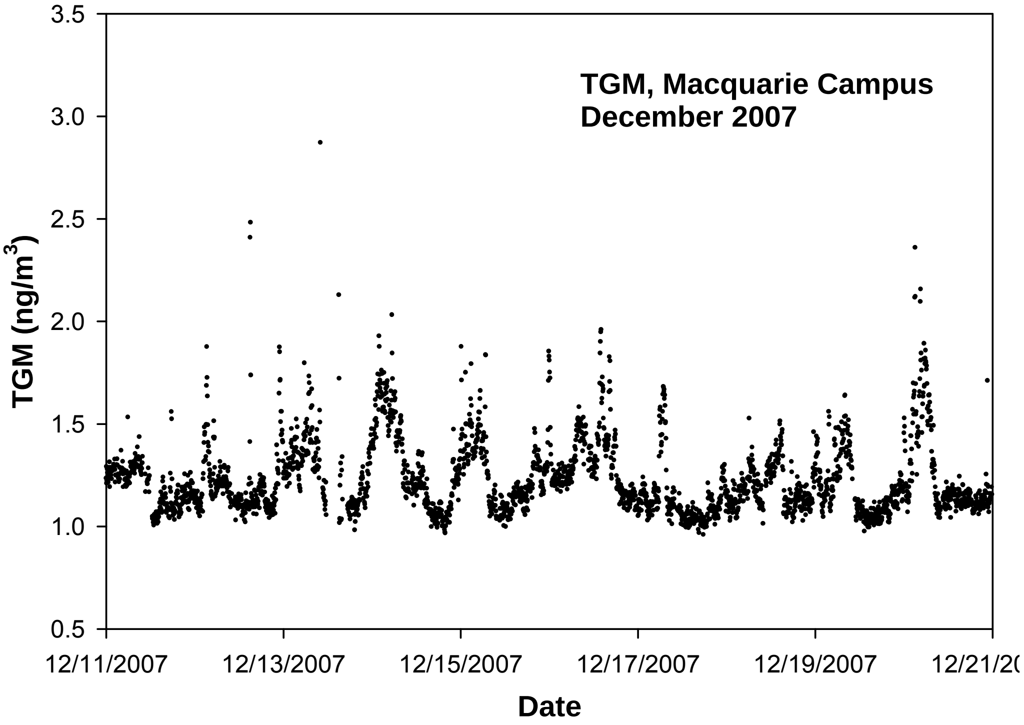


Figure 5: Total gaseous mercury (TGM) measured on Macquarie University campus, December 2007

The concentration results are also generally consistent with results reported in the USA, considering that emissions there are significantly higher than in Australia. Comparison with the results reported in Appendix 1 also shows that the measured and modelled results are in good agreement.

It is also noted that the highest predicted annual average atmospheric concentrations are well below the World Health Organisation guideline for atmospheric mercury of 1 microgram[63] per cubic metre of air (1 µg m^{-3} or 1000 ng m^{-3}).

For comparison, a time series of hourly averaged total gaseous mercury concentrations (Figure 6) was extracted from the urban-scale modelling (Appendix1; (Cope *et al.* 2009)) for the grid point nearest to Macquarie University and for a period in the same month but a different year than the observations. The modelled background is slightly higher than the observations because of the inclusion of a 1.3 ng m^{-3} background in the modelling. The model results show similar diurnal variations of 0.5–0.7 ng m^{-3} as the measurements, but the occasional short-term peaks almost 2 ng m^{-3} above background in the measurements do not show up in the modelling because of the much longer averaging time for the model results (60 versus 2.5 minutes). The good agreement indicates that the model captures the local influences of meteorology on the dispersion.

---

[63] A microgram is one millionth of a gram – 10^{-6}

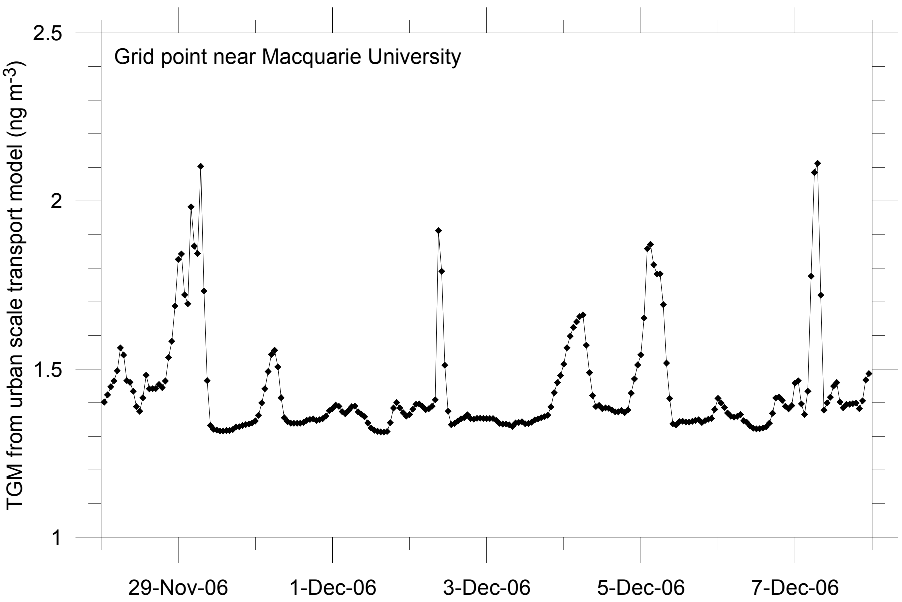


Figure 6: Time series of hourly average total gaseous mercury concentration from urban-scale modelling of Sydney (see Appendix: (Cope *et al.* 2009)) for a grid point closest to Macquarie University. The modelling includes a fixed background of 1.3 ng m^{-3}.

Wet and dry deposition was also modelled at the three spatial scales. The highest wet deposition rates occur in regions of higher rainfall or regions of local elevated mercury concentrations due to anthropogenic sources or combinations of these two factors. In contrast, dry deposition is generally dominated by natural emissions and the continental background, although enhanced dry deposition occurs within the vicinity of the bushfires and significant industrial sources.

At the continental scale, wet deposition rate peaks of up to 5 µg m^{-2} yr^{-1} were predicted. The total mercury mass deposited by precipitation onto the Australian land mass is estimated to be about 1.8 t yr^{-1} which is equivalent to about 0.8 % of the total emissions from the region.

At the continental scale, dry deposition rates were generally less than 20 µg m^{-2} yr^{-1} , although values up to 70 µg m^{-2} yr^{-1} were predicted near the largest industrial source in Kalgoorlie. Dry deposition is calculated to contribute about 21 t yr^{-1} which is equivalent to about 10 % of the emitted mercury from the region and about ten times higher than the wet deposition amount.

The urban scale modelling showed significantly higher wet deposition than the continental scale modelling (20 µg m^{-2} yr^{-1} near the Geelong refinery and 200 µg m^{-2} yr^{-1} within the Illawarra for the Sydney modelling. This is a result of the higher resolution of the urban modelling (3 km grid spacing vs. ~30 km grid spacing for the continental modelling) as well as better resolution of the rain processes.

In contrast to the wet deposition (which is dominated by the local distribution of rainfall), dry deposition in Melbourne is dominated by the natural emissions and the continental background concentrations. Dry deposition peaks of up to 20 µg m^{-2} yr^{-1} were predicted for forested regions to the north-east of Melbourne. These maxima are comparable to the highest wet deposition totals. The natural emission and continental background contribution in the Sydney region is similar to that predicted for Melbourne. However, the natural emission and background continental contributions has proportionally less impact on the predicted Illawarra region dry deposition fluxes of up to 120 µg m^{-2} yr^{-1} .

The near-source modelling predicted maximum wet deposition fluxes range from 5 µg m^{-2} yr^{-1} to over 200 µg m^{-2} yr^{-1}. The deposition patterns reflect the prevailing winds associated with rain and also the proximity of elevated terrain. In contrast, the maximum dry deposition fluxes range from 1–120 µg m^{-2} yr^{-1} with the spatial distributions being similar to the concentration distributions, but influenced locally by land use and vegetation differences. These high deposition fluxes are a result of the high resolution of the near-source modelling (1 km grid spacing vs. ~30 km grid spacing for the continental modelling) as well as the selection of the modelling domains to include the highest mercury emitters. For example, the Kalgoorlie emissions of 7918 kg yr^{-1} dwarf the largest power station emissions of about 100 kg yr^{-1}. Table 34 summarises the modelling results at the various scales (continental, urban and local) and compares them against some observations.

Table 34: Mercury concentrations and deposition fluxes for various modelling regimes, compared to some observations and a WHO ambient concentration guideline.

	Hg Concentration (ng m^{-3})	Hg Wet Deposition (µg m^{-2} yr^{-1})	Hg Dry Deposition (µg m^{-2} yr^{-1})
WHO Guideline	1000	-	-
Background (advected into model domain)	1.3	n/a	n/a
Observations reported in literature from Europe and USA	2 – 20[1]	2 – 25[2]	3 – 15[3]
Continental-scale modelling (Australia) – without fires	1.2 – 2.6	0.03 – 4	0.7 – 70
– with fires	1.2 – 3.0	0.04 – 6	0.7 – 70
Urban-scale modelling (3 km grid)	up to 1[4]	up to 200	up to 120
Near source modelling (1 km grid) near major sources	up to 30[4]	up to 800	up to 160

[1]WHO (2000); [2]NADP (2009); [3] Seigneur et al (2001); [4] Not including background or contribution from fires.

Table 35: Comparison of modelled wet deposition against measurements in Sydney and the Hunter Valley reported by Dutt et al (2009).

Wet deposition	North Ryde (µg m^{-2} yr^{-1})	Cessnock (µg m^{-2} yr^{-1})
Measurements (Dutt et al, 2009)	3.2	3.8
Continental scale modelling (approx 25 x 25 km grid)	1.5	2.1
Urban scale modelling (3 x 3 km grid)	8.3	-

The modelling of wet deposition may also be compared with measurements obtained by Dutt *et al* (2009) at the Macquarie University campus and in the Hunter Valley at Cessnock. This comparison is provided in Table 35, and shows reasonable agreement between measured and modelled values. For the North Ryde site, the two model results are approximately a factor of 2 low and high compared to the observation, which can be considered reasonable agreement given the (unquantified) uncertainties in both the observations and the modelling.

It is worth emphasising, that modelling of deposition is subject to considerable uncertainty and the results reported should be treated with caution. The uncertainties which will affect deposition rates include:

- the extent and location of rainfall events;
- cloud processes resulting in incorporation of mercury in rainwater;
- net deposition velocities for various forms of mercury;
- the assumed proportion of emitted mercury in an oxidised (and hence soluble form). This is particularly important for large industrial point sources where there is little data on the proportion of mercury emitted as elemental, oxidised or particulate mercury.

In addition, the modelling at urban and point source scale showed significantly higher wet deposition than the continental scale modelling. This may be as a result of the higher resolution of the urban modelling (3 km grid spacing vs. 25 km grid spacing for the continental modelling) as well as better resolution of the rain processes, but could also be affected by the emission rates of soluble mercury from individual industrial sources.

In spite of these uncertainties it is likely that the majority of mercury emitted from Australian anthropogenic and natural sources is transported out of the Australian continental airshed, and is hence incorporated in the global mercury pool.

5 Mercury Control – Inputs to Outputs

5.1 Summary of major issues

- The use of mercury in products and processes can occur either intentionally or incidentally;

- The intentional use of mercury is declining in many countries through substitution of new mercury free products and processes. UNEP suggests substitution is now possible for virtually all products that use mercury. Australia benefits from the development of alternatives elsewhere;

- A recent report prepared for the European Commission identified several intentional uses of mercury at levels higher than previously expected or known; (porosimetry and as a catalyst in polyurethane production);

- The intentional use of mercury in some products is increasing, most notably in compact fluorescent lamps (and some other electronics) despite the fact that the amount of mercury per lamp has declined substantially;

    - Work is progressing on mercury free alternatives to compact fluorescent lamps, but commercially available alternatives are not yet available;

    - At this stage, one can only prescribe production/use of energy-efficient lamps with a minimum mercury-content, and collection and treatment of spent lamps (UNEP 2002);

- Mercury in lamps and other electronic products can be recovered and recycled. Data worldwide on recycling rates is patchy but a recent report prepared for the European Commission would suggest that rates are modest in the European Union (but maybe be better in individual countries);

- Programs aimed at reducing mercury use and at recovering mercury containing products must be supported by strong education and outreach programs and even incentives if they are to be successful, particularly in the difficult to manage area of domestic waste;

- Dental amalgam is a major contributor to mercury in waste water treatment systems;

    - Even in the EU with a strong substitution program in some countries, dental use of mercury is expected to rise in coming years. Use of mercury in dentistry has declined very little in USA in recent years;

    - Particulate mercury emissions from dental surgeries can readily be controlled. Amalgam separators are part of best practice guidelines in the industry and are mandated in a number of countries;

    - Viable alternatives to amalgam fillings are available for most applications but are not yet widely known or accepted in many countries, as practitioners generally find it easier to continue using the techniques with which they are most familiar (UNEP 2002);

- The incidental use of mercury occurs mainly via its occurrence in fuels (coal) and metal ores;

- It is technically possible to remove a high proportion of mercury from the flue gases of most, if not all, industrial processes;

- In western economies, some industries are required to achieve a high degree of mercury control, most notably those involved in the combustion / incineration of waste (domestic, medical, hazardous);

- A significant amount of work on the control of mercury from coal-fired power stations has occurred in the USA in response to proposed legislation which would require a 70% reduction over existing levels. Following court action, this legislation is currently being reviewed by the USEPA;

- A degree of mercury capture is already occurring from many facilities as a co benefit of existing air pollution control devices. Co-benefit forms a significant part of the USEPA control strategy;

- While feasible, the reduction in mercury from coal combustion due to fuel switching and or fuel substitution is unlikely in the absence of strong regulatory measures and or financial incentives;

- Reductions in mercury emissions may result as a co benefit from national and international measures aimed at reducing greenhouse gas emissions, and particularly carbon dioxide, such as energy conservation and the generation of electricity from technologies with negligible or zero mercury emissions (gas, renewables);

- A voluntary program between environmental regulators and gold producers in Nevada, USA resulted in a reduction in mercury emissions to the atmosphere of about 80%;

- Mercury emissions from crematoria are increasing in some countries (more cremations, bodies with more mercury), leading to requirements to control emissions from this source. No jurisdiction has mandated the removal of dental amalgam from bodies prior to cremation;

- A range of technologies exist for the treatment of soil, waste and water contaminated with mercury. New technologies are also being developed; and

- As is the case with the other management and policy options, it is important to consider both the potential reductions (and hence benefits) and the costs of the options. *Any consideration of potential reductions should examine whether (and the extent to which) emissions reductions from the particular sources in question will yield reductions in risk to public health and the environment, (USEPA 1997a).*

5.2 Background

As discussed in Section 1.3, mercury emissions to the environment can be grouped into four major categories:

- New releases from naturally occurring sources such as volcanic activity and weathering of rocks;
- Re-emission of previously deposited mercury from either natural or anthropogenic sources;
- From the intentional use of mercury, either in industrial processes or its inclusion in products– even though the actual emission may be accidental; and
- From its incidental use in processes such as electricity generation, cement, pulp and paper manufacturing and metal smelting when the fuel or raw materials contain mercury, some of which is released to the environment.

This section of the report reviews the measures that are available to prevent and minimise the release of mercury to the environment from the intentional and incidental use of mercury. The review is based upon the significant amount of work undertaken elsewhere over the past several decades and particularly in the Great Lakes region of North America, in the European Union and by the United Nations Environment Programme (UNEP 2002).

A report prepared for the USEPA in 2000 (Ross and Associates 2000) for the Great Lakes Binational Toxics Strategy considered the options for mercury control under 6 categories:

- Substitution: The use of alternative products or process inputs that are either lower in mercury concentration or mercury free and either less toxic, or non-toxic can reduce or eliminate mercury emissions;
- Recycle/Disposal: Involving the proper end-of-life handling of products containing mercury to ensure the potential for air and water contamination is minimised;
- Energy Efficiency: The more efficient use of energy can result in less emission of mercury due to less energy production;
- Emission Control: Where mercury is a pollution by-product of manufacturing, energy generation or waste disposal in process or end-of-pipe control technologies can be very effective in reducing emissions of mercury and other hazardous substances;
- Clean-up: Proper spill response planning, clean-up and disposal can help reduce the severity of contamination if and when mercury spills occur; and
- Education: Increasing the community's understanding of the use of, and potential hazards associated with the, use of mercury use, can lead to a reduction in contamination from households and work places. Education can also lead to a greater willingness to engage in alternative practices such as buying alternative consumer products, increasing energy efficiency, for example.

It can readily be appreciated that the approaches to mercury control outlined above are consistent with the well known "waste" hierarchy, an example of which is shown in Figure 7, where options which prevent and minimise the generation of "waste' or pollution are preferred over options which require treatment and/or disposal of a waste product or pollution, for example.

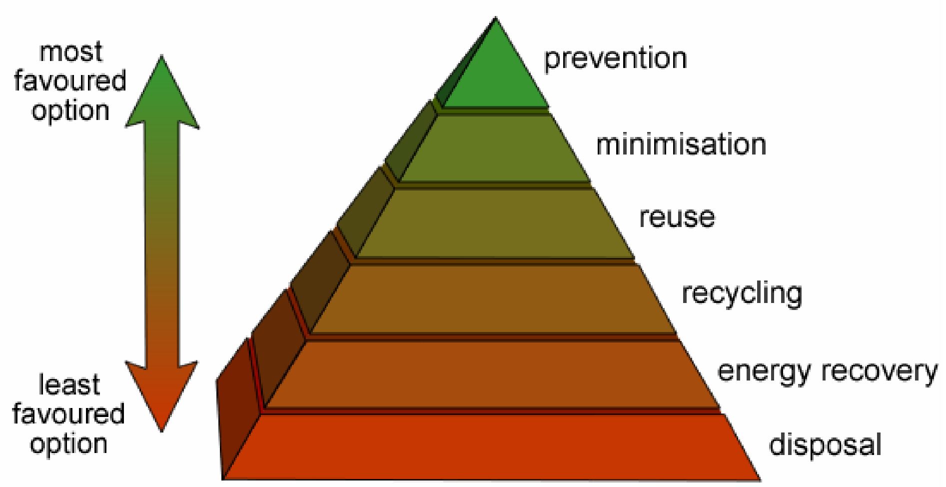


Figure 7: The waste management hierarchy

From the above, it can be appreciated that options higher up the hierarchy (ie more preferred) are more likely to be applicable to the intentional uses of mercury, particularly where alternative products or processes are able to eliminate its use, compared with processes in which the use of mercury is incidental. In the latter case, the control options are more likely to involve the use "end-of-pipe" technologies and improved disposal practices – options which sit lower on the hierarchy of options, although fuel substitution and cleaning may be an option in some circumstances.

Different approaches have been adopted in presenting mercury control options in several recent reports which have comprehensively addressed the topic. The report prepared for the USEPA in 2000 (Ross and Associates 2000) presented control options for different source categories:

- Energy Production;
- Manufacturing Industry;
- Waste Disposal;
- Medical Field;
- Dental Field;
- Schools and Laboratories; and
- Consumer Mercury Use.

For each of these source categories the report discusses relevant options from substitution, energy efficiency, control technologies etc.

An alternative approach in discussing mercury control was used in the UNEP Global Mercury Assessment report (UNEP 2002) and in the USEPA report to Congress (USEPA 1997a) which are organised first according to the type of control – Prevention, Substitution, Waste Management Practices, Reducing Emissions - being considered.

In this review, we draw significantly on the above mentioned, and other, reports in providing a summary of efforts to minimise the use of and emission of mercury to the environment in the recent past. World-wide-web links to most of the reports referenced provide ready access to these reports for the reader requiring more detail.

5.3 SUBSTITUTION

The United Nations Global Mercury Assessment Report of 2002 (UNEP 2002) makes the following observations (direct quotes in italics) about substitution as a way of addressing mercury in the environment.

"*Substitution of products and processes containing or using mercury with products and processes without mercury may be one of the most powerful preventive measures for influencing the entire flow of mercury through the economy and environment. It may substantially reduce mercury in households (and reduce accidental releases, as from a broken thermometers), the environment, the waste stream, incinerator emissions and landfills. Substitutions are mostly cost-effective, especially as they are demanded by a larger and larger market. This group of measures would also include the conversion of a fossil-fuelled generating plant to a non-fossil technology.*[64]

As the general awareness of mercury's adverse effects on human health and the environment has increased, a number of countries have made special efforts to address mercury in these applications, and have had particular success in reducing mercury use. Canada, Denmark, Norway, Sweden and the USA, among others, have seen the number of applications as well as the quantities of mercury used per application decrease significantly, particularly during the last 15-20 years. Nevertheless, since many mercury-containing products have long technical lives, it should be kept in mind that even if a country decides to ban the marketing and use of mercury in most products, it may take decades before most of the mercury in use is collected and removed from human circulation.[65]

A review of recent reports from the USA and Europe indicates that, notwithstanding some obstacles, in "*virtually all applications of mercury, substitution is possible*" and happening under existing or

---

[64] http://www.chem.unep.ch/mercury/Report/Final%20report/final-assessment-report-25nov02.pdf; paragraph 119.

[65] http://www.chem.unep.ch/MERCURY/Report/Chapter8.htm

proposed future measures. While the results from the European and American programs are not necessarily directly applicable to Australia, they do provide a clear indication of the potential to substitute mercury use and suggest areas where future action might be focussed. Obstacles to the introduction of substitutes, including possibly higher costs, competition and attitudes to, and knowledge of, alternative techniques are not insurmountable, but commitment and a clear strategy, including outreach and incentives are required to achieve program objectives, as evidenced by the following: "*During the implementation of the Swedish ban on mercury in products (except those few products with an exemption), an investigation of substitutes for mercury-containing measuring instruments and electrical components was carried out. It was discovered that while several applications of mercury were being phased out, some new applications, surprisingly, were appearing - as in electronic equipment - even though alternative technologies were available*.

5.3.1 Products and application

Table 36 provides a summary, not necessarily including all of the many application of mercury, of the degree of substitution considered feasible in many industrial processes and products, ranging from where substitution is not currently occurring (although development may be underway) to situations where mercury use is fully, or almost fully substituted. Information in the table is sourced from two recent reports (EU 2008; Northeast Waste Management Officials Association (NEWMOA))/ IMERC 2008) and Chapter 8 of the UN Global Mercury Assessment (UNEP 2002).

Table 36: Summary of mercury substitution (figures for the European Union)

Application area	% of total use in EU	Level of Substitution	Comments
Chlor alkali production	**41.2**	2-3	Many plants in Europe and elsewhere converted to mercury free (membrane technology) with most of the remaining plants to be phased out by 2020. No mercury-based chlor alkali plants in operation in Australia.
Light sources	**3.1**		Currently, there are no mercury-free energy-efficient alternatives to the energy-efficient lamps on the market, although there are reports of a high-efficiency non-mercury lamp based on the field-emission effect and lamps based on diode technology are developing. The decrease in mercury per tube is being offset by the increasing use of compact fluorescent tubes.(IMERC 2008) Low-mercury lamps are slightly more expensive than those with a bit more mercury. For a manufacturer to be allowed to use the European Ecolabel on a single-ended compact fluorescent lamp, the mercury content must not exceed 4 mg, and the life of the lamp must exceed 10,000 hours. Recently introduced and fashionable auto headlamps containing mercury are a concern, as they are inconvenient to recover and recycle, and acceptable non-mercury alternatives are available. ***One can only prescribe production/use of energy-efficient lamps with a minimum mercury-content, and collection and treatment of spent lamps.(UNEP 2005)***
Fluorescent tubes	0.9	0	
Compact fluorescent tubes	0.5	1	
HD lamps	0.3	0	
Other lamps (non-electronics)	0.4	1	
Lamps in electronics	0.9	1	
Batteries	**3.8**		**Note EU Directive 2006/66/EC regulating use and disposal of batteries and accumulators** The cost of alternatives may often be higher than the mercuric-oxide and mercury-zinc batteries, but municipalities can avoid expensive collection and disposal schemes. Standard mercury-free batteries generally cost about the same as the batteries they replace. Rechargeable batteries, on the other hand, especially the cadmium-free rechargeables, are significantly more expensive to purchase, although they become relatively less expensive if recharged more than 10 or 15 times. Batteries remain a concern because (COWI 2008): ▪ Large volumes marketed ▪ Older batteries in waste stream with higher Hg content ▪ Evidence of continued use / trade of banned mercuric oxide batteries (and/or parts) in EU (button cell batteries up to 2% Hg)
Mercury button cells	0.1	2	
General purpose batteries	1.4	4	
Mercury oxide batteries	2.2	4	
Dental amalgams	**23.5**		Some alternatives are less expensive and some are more expensive than mercury amalgams, some are as easy to apply and others are more difficult, but none of the alternatives require the specialized wastewater treatment equipment that dental professionals need to meet environmental regulations in many countries. ***The viable alternatives are not yet widely known or accepted in many countries, as***
Pre-measured capsules	16.5	2	
Liquid mercury	7.1	3	

			***practitioners generally find it easier to continue using the techniques with which they are most familiar*.**
Measuring equipment	**2.8**		Electronic thermometers remain somewhat more expensive than glass mercury thermometers but price has come down substantially in recent years and other recently introduced Ga/In/Sn thermometers should approach the cost of old mercury thermometers in time. It should also be noted that, while the initial cost of a mercury glass thermometer is lower than an electronic device, the frequency of broken mercury thermometers is higher, and one electronic thermometer may replace several mercury ones. If an annual cost is calculated, the price of an electronic measuring device is probably no higher than the mercury device it replaces. Electronic alternatives have several advantages over mercury. One electronic device can be adjusted to several different measuring ranges, thereby substituting for several mercury thermometers. It is possible to read and record digitally. This could reduce the chance of human error, as well as reduce operating costs. For a very small number of precision applications, mercury thermometers are still preferred for technical reasons, e.g. for calibration of other thermometer types, for international standards, etc. Pressure measuring alternatives based on gas, other liquids or a mechanical spring show no significant differences in price, compared to mercury devices. Alternatives in the form of electric and electronic instruments are only slightly more expensive, but have several advantages over mercury.
Medical thermometers	0.5	3	
Other mercury thermometers	0.2	3	
Manometers	0.04	4	
Barometers	0.82	3	
Sphygmomanometers	1.1	3	
Hygrometers	0.01	3	
Tensiometers	0.01	4	
Gyrocompasses	0.004	3	
Reference electrodes	0.002	3	
Hanging drop electrodes	0.1	3	
Other uses	0.01		
Chemicals	**10.2**		It is entirely possible to restrict mercury use in school or university laboratories to a few specific, controllable uses (mainly references and standard reagents). This initiative has already been implemented in Swedish and Danish legislation. The alternatives are generally no more expensive, and the need for control of mercury sources in the laboratory is greatly reduced. The recent EU report (EU 2008) highlighted the previously less noted use of mercury as a catalyst in polyurethane elastomer production The use of mercury in pesticides and biocides has been discontinued or banned in many countries. These alternatives are in place in many countries. The range of products and applications is too diverse to make definitive statements about cost comparisons, although it is likely that in the majority of cases costs are roughly comparable and environmental benefits are considerable.
Intermediates and catalyst (exc PU)	3.5	2	
Catalyst in polyurethane (PU) production.	6.5	3	
Laboratories and pharmaceutical	1.5 0.1	3 3	
Preservatives in vaccines and cosmetics	1.6	4	
Preservatives in paints	0.4	4	
Disinfectant	0.1	3	
Other applications			
Switches, relays etc	**0.1**		With very few exceptions, there are no technical obstacles to replacing electrical components, conventional relays and other contacts (even when these are contained in level switches, pressure
Tilt switches	0.09	4	
Thermoregulators	0.01	4	

Read relays and switches	0.01	3	switches, thermostats, etc.) with equivalent mercury-free components There are no significant price differences between conventional mercury and mercury-free relays and contacts, except for very specific applications.
Others	0.02	4	
Miscellaneous uses	**15.2**		The recent report for the EU (EU 2008) highlighted the previously less reported use of mercury in porosimetry – 12.9% of use in EU.
Porosimetry and pycnometry	12.9	2	
Conductors welding	0.1	3	
machines	0.1	N	
Mercury slip rings	0.4	0	
Lighthouse maintenance	0.1	0	
Bearing maintenance	1.1		
Gold production (artisanal)	0.4		
Other applications			

Key to assigned substitution level:
0 No substitution indicated; development often underway
1 Alternatives are ready to be marketed, or are present on the market but with marginal market share
2 Alternatives are being marketed and have significant market share , but do not dominate th market
3 Alternatives dominate the market, but new products which contain mercury also have significant market share
4 Mercury use is fully substituted, or almost fully substituted
N Not enough data was found to assign an indicator

Denmark provides a good example of the potential achievements of a coherent substitution strategy, which strongly encourages substitutes for mercury products, including placing a ban on the sale and use of most mercury products. As in a number of other countries, a substantial decrease in mercury consumption for intentional uses has been observed. Table 37 shows that the annual consumption of mercury in intentional uses fell from about 16 metric tons in 1982/83 to 6 metric tons in 1992/93, and decreased further to 1.5 metric tons in 2000/2001. In the same period, releases to the environment were reduced from an estimated 6.9-9.9 metric tons in 1983, to 2.3-3.0 tons in 1993. The deposits in (controlled) landfills have increased during the same period from 1.7-2.9 metric tons to 2.3-4.5 tons, most likely as a result of increased hazardous waste collection (reflecting the mercury content of used products, batteries, etc.) and improved filtering of waste incinerator emissions (UNEP 2002).

Between 2001 and 2004 the total amount of mercury sold in fabricated and formulated products in the USA declined from about 131 to 117 tonnes, about 11% (Northeast Waste Management Officials Association (NEWMOA))/ IMERC 2008). The overall decline does not reveal the slight decline in one of the largest sources – dental amalgam or a significant increase in use of some lamps using mercury, including compact fluorescent lamps, High intensity discharge lamps, screens and monitors.

Switches, relays and dental amalgam capsules accounted for approximately 70% of the total mercury use in the USA in 2004. Use in switches and relays fell from 60 to 51 tons 2001 – 2004 (15%) and there was no substantial change in use of dental amalgam, although significant control of discharge from dentists' premises has occurred (see the discussion of management practices in the following section).

Standard fluorescent mercury use declined 14% whereas use of mercury in high intensity discharge lamps increased by about 15%. Decrease in mercury in fluorescent lamps was likely due to manufacturers efforts to reduce mercury content per bulb, while higher sales likely accounts for increase in HID lamps. The greatest change between 2001 and 2004 is a nearly 70% increase in mercury use associated with CFLs which is due to increased sales (and despite an average decrease in mercury per lamp). Since 2004 there has also been a significant increase in the number of electronic devices using fluorescent lamps, including LCD flat screens, GPS units, and digital cameras.

Table 37: Estimated changes in annual consumption of mercury in Denmark (metric tons/year). From Chapter 8 of UN Global Mercury Assessment Report (UNEP 2002)

Year/use	1982/83	1992/93	2000/2001
Chlor-alkali production (discontinued in 1997)	3.00	2.50	0
Dental amalgam	3.1	1.80	0.9
Mercury-oxide batteries	2.40	0.36	0
Other batteries	2.30	0.28	~ 0
Measuring and control equipment	0.53	0.50	0.3
Electric and electronic switches	0.34	0.30	~ 0
Light sources (lamps)	0.14	0.17	0.17
Medical thermometers	0.75	0.05	0
Other thermometers	1.55	0.10	0
Laboratory chemicals	0.50	0.09	0.09
Other intentional uses	1.48	0.03	0.03
Sub-total, intentional uses	16.09	6.18	1.5
Impurities in consumed fuels, minerals and high-volume materials (non-intentional mobilisation)	1.96	1.80	1.8
Total	18.05	7.98	3.3

5.3.2 Fuel Substitution

In the electricity generation industry it is possible that fuels with lower mercury content could be substituted for higher mercury containing fuels (natural gas for coal, for example) or zero mercury emitting technologies (renewables) substituted for mercury emitting technologies. However, it is improbable that such substitution would occur without strong regulatory measures and/or financial incentives to drive such a change.

Modelling undertaken by the US Department of Energy (US DoE 2005) predicted little fuel switching under the Environment Protection Agency's (EPA) proposed 15 tonne cap[66] and trade scheme. Under a stringent 90% Maximum Available Control Technology (MACT) scenario (under which all power stations would be required to reduce mercury emissions by 90%) the degree to which renewables and natural gas would be substituted for coal would depend on the effectiveness and availability of mercury control technologies. Using the assumption that control technology would be available and able to achieve 90% reduction on all coals, little fuel switching was predicted to occur. Under scenarios in which control is less effective, significant switching from coal to renewables, natural gas and oil was predicted. Also of note was the prediction that there would be switching from (western) sub bituminous coal to (eastern) bituminous coal – the control of mercury being more feasible from the latter coal type.

From the above discussion it is worth noting that any measures, such as energy conservation, end use efficiency, lower emitting technologies etc directed at reducing greenhouse gas emissions by displacing coal-fired generation would yield a co-benefit of reduced mercury emissions.

International discussions on the agreement to be enacted following the term of the Kyoto Protocol in 2012 may see countries committing to significant greenhouse reductions over the coming decades, such as the Australian Government Carbon Pollution Reduction Scheme.

---

[66] From existing emissions from the sector of about 48 tonnes

5.3.3 Further action

5.3.3.1 Fluorescent lamps

For mercury use in fluorescent lamps, which are known for their low energy consumption, no commercially mature alternatives are yet available. Work has been done, however, to reduce the amount of mercury needed in each lamp. From typical amounts of 20-40 mg of mercury per lamp, lamps with only 3 mg of mercury are commercially available today. Unfortunately these modern low-mercury lamps have difficulty in competing on price with the higher-mercury lamps, and consumers are generally unaware of the difference between them. In the USA it is reported that more than two thirds of CFLs now contain less than 5 mg and only a small percentage contain more than 10 (and up to 50) mg (Northeast Waste Management Officials Association (NEWMOA))/ IMERC 2008).

The use of diodes as lights – recently installed in some traffic lights - has been proposed as an energy-efficient substitute. The strength of the light source for this application would appear to be comparable to what is needed for some housing purposes. However, until mercury-free alternatives are widely deployed, the mercury in fluorescent lamps may be managed by regulating mercury content, collection of used lamps and recycling or proper waste treatment. This has been attempted in a few countries and localities, but it has been difficult in most cases to achieve significant collection rates.

5.3.3.2 Motor vehicles

While motor vehicle makers have phased out the use of mercury-containing switches the use of other mercury containing products is increasing in motor vehicles, as noted above - HID lamps, navigation systems and entertainment systems which use fluorescent lamps.

The US auto switch recovery program (Section 5.4.7) is an effective program directed at recovering and recycling an estimated 67 million (mercury) switches but as yet there does not appear to be a program directed at the newer mercury containing products in motor vehicles.

5.3.3.3 Dental amalgam

While there has been substantial developmental work on a range of alternatives, there is not yet a consensus that substitutes can adequately replace mercury amalgams in all dental applications. In Sweden and Denmark voluntary substitution agreements have been in place for a number of years and the consumption of mercury for dental use has decreased significantly (Table 37), although use in the European Union as a whole is expected to increase slightly over coming years. In Denmark mercury amalgams are permitted (until further notice) only in molars where the fillings are worn. However, as noted above, mercury use in dental amalgam in the USA has remained reasonably steady over recent years.

Increased awareness of mercury use in fillings and availability of alternatives may drive further reductions – although cost of alternatives is a factor, in some uses.

5.3.3.4 Product labelling

This has advantages and disadvantages, but has proven rather effective in some cases, in combination with other measures. For example, in the case of consumer batteries, consumers paid significant attention to labels concerning the content of mercury and cadmium.

5.3.3.5 Education – Information – Incentives

It is apparent from the above discussion that the uptake of alternatives requires the alternatives to not only be readily available and cost competitive, but also that information is readily available to product suppliers and consumers to assist them to firstly to understand the issue and secondly to enable them make informed, responsible decisions. The use of incentives to stock or purchase low mercury CFLs, for example may also assist in promoting a change in purchasing decisions.

5.3.3.6 Chemical standard analyses

A number of chemical standard analyses involve the use of mercury compounds. While mercury-free substitutes are generally available, this issue is mentioned in the EU report (EU 2008) because it may take time to change standards previously agreed upon. For example, a common analysis using mercury is the COD (chemical oxygen demand - measuring contents of organic matter) analysis, which is widely used to control and monitor the quality of wastewater. Other oxygen demand (e.g., the so-called BOD – biological oxygen demand) analyses are available and are often used. However, the problem is that the prescriptions of many mandatory analyses in regulations and individual wastewater release permits specify the COD analysis, and need to be changed. This is possible, but requires attention and time. The Swedish government was considering a ban on mercury use in chemicals for analyses and reagents from 1 January 2004.

5.3.3.7 Final comment

Australia has benefited and will continue to benefit from the development in other countries of new, zero and low mercury products. The EU and USA also provide examples of polices and program to reduce the impact of continued use of mercury.

5.4 MANAGEMENT PRACTICES

As noted above mercury has widespread use in many products, and while its use is being phased out as substitutes are developed and awareness increases, its occurrence in products will continue for many years, if not decades. At present, many of these products may end up in conventional solid and liquid waste streams leading to emissions to the environment (to air, water and land) from waste management facilities (landfills, water treatment, incinerators or combustors). In a number of cases management practices are available which can minimise, if not eliminate the risks posed by the continued use of mercury. The following discussion does not address all possible practices but provides an overview and some examples of the practices that have, or could potentially be implemented.

5.4.1 Fuel cleaning

Fuel cleaning is the practice of removing a contaminant from the fuel prior to its use. A good example of fuel cleaning is the removal of inorganic material (the "ash" fraction) from coal prior to combustion. This form of coal cleaning is normally undertaken to improve combustion of the coal and/or to lower transportation costs and less commonly to remove specific contaminants, although contaminants such as sulfur and mercury which are included with coal's mineral matter may be partially removed in the coal cleaning process, providing a co-benefit. Coal cleaning to remove sulfur has contributed to the reduction in sulfur dioxide emissions in the USA under the Acid Rain program.

Fuel cleaning is referenced as an option for mercury control by the UN mercury program and elsewhere, but like fuel substitution (see Section 5.3.2) is unlikely to occur in the absence of regulatory measures and/or financial incentives.

Unlike fuel substitution, fuel cleaning does not reduce or eliminate the "use" of the substance; while minimising potential air emissions it adds to solid and liquid waste product management requirements. Volume II of the Mercury Study to Congress (An Inventory of Anthropogenic Mercury Emissions in the United States) (USEPA 1997a) reported that mercury reductions due to current coal cleaning methods ranged from 0 to 64% with an average reduction of 21%.

Advanced coal cleaning techniques such as selective agglomeration and column froth flotation have the potential to increase the amount mercury removed compared with existing methods although the potential impacts on post combustion controls and control of the remaining mercury has not been thoroughly investigated. Mercury mass transfer limitations are encountered in emission control systems and advanced cleaning techniques may exacerbate this problem and could also result in a change in the form of mercury being emitted with possible impacts on capture efficiency (USEPA 1997c). In the USA, 77% of the eastern and midwestern bituminous coal is cleaned to meet customer specifications for heating value, ash content and sulfur content. The Australian Coal Association reports[67] that 80 percent of all coal mined and most of the black coal destined for export in Australia is washed to provide a twofold advantage:

- it upgrades the quality of the coal; and
- it improves the economics of transportation by removing most of the non-combustible material.

Although not documented, some removal of mercury would occur during the washing or cleaning process.

---

[67] See: http://www.australiancoal.com.au/the-australian-coal-industry_coal-preparation.aspx

Regulatory measures may be used to limit the occurrence of certain (undesired) substances in fuel inputs. It has been used to set a limit on the sulfur content of fuel burnt, in the case of coal-fired power stations, for example.

In cases where an industry burns material previously used in other processes (“waste” oil, for example) it may be appropriate to set a limit on the concentration of certain substances, as occurs in the cement industry and power generation industries. Such measures are fuel management rather than fuel cleaning.

5.4.2 Energy Efficiency and Conservation

As noted elsewhere, measures that reduce the amount of energy used for a given output, can provide wide ranging benefits, including a reduction in the amount of mercury emitted to the atmosphere or being stored in ash repositories. These reductions are most likely to come about through national and international measures aimed at reducing greenhouse gas emissions.

5.4.3 Separation - Diversion – Recycling -Secure storage

Efforts to separate mercury containing products with subsequent diversion from waste streams have occurred in a number of jurisdictions, with varying levels of success. Programs to separate and divert mercury containing products from waste streams must be accompanied by measures to manage the collected mercury in a manner which does not result in its subsequent release to the environment through either secure storage or recycling and reuse. As discussed elsewhere, these programs must be accompanied by far reaching and extensive information and education campaigns and possibly incentive schemes to achieve high diversion rates, particularly in the difficult to manage area of domestic waste.

Removing mercury containing products from the waste stream can be done without lowering the energy content in the waste – which is a consideration in places where waste to energy is practised. The higher mercury containing products (lamps, batteries, thermostats etc) typically make up a small (~ 1%) fraction of the waste (USEPA 1997c).

Since 2000 the North Eastern states of the USA[68] have enacted a range of measures, including bans and phase outs on the sale of certain products, product labelling and mercury disclosure and collection of mercury containing products, to reduce the use of mercury and its disposal to the environment (NE States 2007) which has resulted in the elimination of an estimated 14 tons of mercury in product sales and collection and recycling of an estimated 7.5 tonnes of mercury (2000 – 2006).

Some specific examples of mercury collection and recycling are presented later in this section.

5.4.4 Dental amalgam

Dental amalgam makes up a high proportion of mercury used in products - in the order of 25% in the US and EU (Table 36) and in the US, dental clinics are estimated to contribute up to 50 % of mercury discharges to public water treatment facilities (USEPA 2008).

The vast majority of mercury (>99.6%) of dental mercury discharges are in the solid form (elemental mercury bound to particulate) but discharge of other forms, inorganic (ionic), elemental and organic

---

[68] Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont.

also occurs. Dissolved mercury composes a small fraction of total mercury in dental waste water but is of concern because it can be converted to the toxic monomethyl mercury in the waste water.

The US EPA study reports that most dentists use some form of basic filtration system (for example filters in the chair-side traps). Recommendations for dental amalgam waste management (EPA office of solid waste management) and Best Management Practices for Dental Amalgam (American Dental Association) include, amongst other things such as recycling and education, the addition of amalgam separators that meet International Organisation Standardisation (ISO) Standard 11143 which requires an amalgam separator to remove at least 95% of amalgam particles by weight when subjected to a specific test method.

Of the best practice measure the USEPA refers to the installation of an amalgam separator as the key to success. The use of amalgam separators results in reductions in water treatment facilities input mercury concentrations and biosolids mercury concentrations, but does not always result in mercury reductions in treatment facilities (liquid) effluent since most mercury discharged by dentists is attached to larger particles which are likely to be removed in the grit or with biosolids and not discharged with effluent. Thus current best practices may result in reductions in the mercury content of biosolids but may not result in reductions in effluent mercury concentrations.[69]

The 2008 USEPA study reports on a number of state programs, both voluntary and mandatory, designed to reduce dental mercury emissions. The voluntary programs produced participation rates from 20 to 90%.

The cost of a BMP program includes:

- implementation costs to state and local agencies;
- purchasing, installing and maintaining an amalgam separator;
- collecting and recycling the collected amalgam; and
- education and training etc.

Costs for amalgam separator(s) vary according to the size of the dental clinic (number of chairs) The USEPA report (USEPA 2008) includes detailed cost information but also includes the summary table below (Table 38).

Table 38: Estimated annual costs for amalgam separators by size of dental clinic ($US 2008)

	Small (1 -4 chairs)	Medium (5 -12 chairs)	Large (+12 chairs)
Purchase	228 - 1,370	760 – 2,510	**2,850 – 10,000**
Installation	114 - 228	143 – 297	**228 – 1,140**
Maintenance	0 - 228	0 -228	**0 -228**
Replacement	57 - 856	86 – 856	**571 – 2,400**
Estimated Annual cost	211 – 1,073	293 – 1,110	**1,990 – 4,630**

Eleven states in the USA have mandatory programs for dental facilities, typically adopting the ADA BMPs, which includes the use of amalgam separators. Other states have voluntary programs, also largely based on the ADA's BMPs. US EPA does not think national, categorical pre-treatment standards for dental mercury discharges are appropriate at this time, arguing that there are opportunities for pollution prevention and voluntary adoption of BMPs within the industry and

---

[69] Human wastes (faeces and urine) from individuals with dental amalgam fillings are estimated to be the major domestic source of mercury to waste water treatment facilities The National Association of Clean Water Agencies http://archive.nacwa.org/pubs/mercury/mercury.cfm

without federal regulation. EPA's further argument that the use of dental mercury is decreasing in the USA (due to mercury free fillings and improved overall dental health) must be considered against the recent IMERC report (Northeast Waste Management Officials Association (NEWMOA))/ IMERC 2008) which indicated only a slight decrease (~1%) in dental mercury use between 2004 – 2008.

At the end of 2005 more than 50% of dental offices in US New England States and Eastern Canadian provinces had installed dental amalgam separators to reduce mercury going to waste water facilities. Amalgam separators became mandatory whereas previously their use was discretionary. Implementation is still occurring but results from the Massachusetts Water Resources Authority treatment plant for the period 2004 to 2006 when the majority of separators were installed show mercury in sewage sludge pellets dropped from 3.8 mg/kg to 1.2 mg/kg (King *et al.* 2008).

Dental amalgam use is very limited in some European countries (Sweden, Denmark, Finland Norway); Amalgam separators are mandatory in a number of EU countries and despite development and use of alternatives amalgam use is increasing slightly in EU due to better access to dental care in some countries where amalgam still the main material used (EU 2007).

In Australia there appears to be a limited and varied state by state approach to best practice management in the dental industry. In the NSW Liquid Transfer Waste Management Guidelines (NSW Department of Energy Utilities and Sustainability 2005) "*an amalgam separator is required, built into the cuspidor provided by the manufacturer*" for standalone dental clinics. No detail on the type / efficiency is provided and further the guidelines allow a "small amount" of mercury to be discharged from dental practices.

In the ACT, the Trade Waste Acceptance Note TW9 for Dentistry requires the installation of amalgam separation equipment capable of a "good industry practice" level of amalgam removal (currently 95%) and conformation to ISO 11143.

In Victoria, the Dentists for Cleaner Water Project was launched in 2008. The voluntary program offers a rebate to the dental profession to install amalgam separators complying with International Standard ISO 11143:2008[70].

5.4.4.1 Removal of dental amalgam prior to cremation

Mercury emissions from crematoria occur primarily due to the volatilisation of mercury contained in dental fillings and to a lesser extent from mercury stored in blood and body tissues.

The removal of dental amalgam from bodies prior to cremation has been discussed as an effective management practice to prevent mercury emission to the atmosphere[71] from a source that is increasing in importance in many jurisdictions due to:

- a rise in the number of fillings per person; and
- increasing use of cremation compared with burial (Concorde 2007).

---

[70] See: http://www.adavb.net/DentistsforCleanerWater/tabid/555/language/en-AU/Default.aspx

[71] See for example (all viewed 2/01/2009)
http://www.zeromercury.org/EU_developments/Maxson_Dental-Releases_25May2007.pdf
http://www.planetark.com/dailynewsstory.cfm/newsid/25201/story.htm
http://www.metroactive.com/papers/metro/06.08.00/cremations-0023.html
http://www.ejnet.org/crematoria/reindl.pdf

While it is common for jurisdictions to require the removal of certain products (pacemakers, prosthetic devices) prior to cremation, there appears to be no jurisdiction where the removal of dental fillings prior to cremation has been mandated. The state of Minnesota in the USA introduced 2 Bills (HF 0661 and SF 641) in 2005 to require that dental mercury be removed before cremation but neither bill has been adopted. Likewise in Maine, a similar bill was rejected.

The control of mercury emissions to the atmosphere from crematoria is considered in Section 5.5.4.

Reporting of mercury from Sewage Treatment Plants to the NPI

Emissions from sewage treatment plants (STPs) are potentially a very important inclusion in the NPI for mercury emissions, as they represent the point of emission for all mercury that is deposited “down drains” from domestic, commercial and some industrial sources.

Emission from STPs could be to air (volatilisation), to water (as treated effluent) or to land (from sludge disposal, with subsequent emissions to air and water).

In reviewing NPI data on emissions from the Water supply, sewage and drainage category (that is including STPs) a number of inconsistencies and potential anomalies were indentified, including:

- 16 facilities reporting - 5 from NSW but not it seems including one of the largest;
- While NSW includes 32% of the facilities reporting - these 5 facilities reported only 0.070 kg from a total of 52 kg across Australia (< 1%);
- All 0.07 kg reported from the NSW facilities were to air - none to water, and none to land;
- Of the 52 kg of emissions reported from STPs Australia wide: 4kg to air, 19kg to land, 28kg to water
- ACT reported 3.7kg to air - 92% of the emission to air;
- Victoria reported 19 kg to land (and 5.7 kg to water) - 100% of emissions to land;
- South Australia reported 22kg to water - 78% of the water emissions;
- Smaller regional / rural STPS do not appear to be included in jurisdictions reporting? (ie sub threshold); and
- The NPI data suggest that improvements could be made in understanding the emission of mercury from STPs.

5.4.5 Electrical and Electronic Equipment Waste in the EU

EU legislation restricting the use of hazardous substances in electrical and electric equipment (Directive 2002/95/EC) and promoting the collection and recycling of such equipment (Directive 2002/96/EC)[72] has been in force since February 2003. Amongst other things Waste Electrical and Electronic Equipment (WEEE) covers mercury containing lighting but also other products containing mercury – computer monitors, switches, relays etc.

The legislation provides for the creation of collection schemes where consumers return their used e-waste free of charge. The objective of these schemes is to increase the recycling and/or re-use of such products. It also requires heavy metals such as lead, mercury, cadmium, and chromium and flame

---

[72] See: http://ec.europa.eu/environment/waste/weee/index_en.htm

retardants such as polybrominated biphenyls (PBB) or polybrominated diphenyl ethers (PBDE) to be substituted by safer alternatives.

The WEEE Directive obliged the twenty-five EU member states to transpose its provisions into national law by 13 August 2004. Only Cyprus met this deadline. On 13 August 2005, one year after the deadline, all member states except for Malta and the UK had transposed at least framework regulations. As the national transposition of the WEEE Directive varies between the member states, a patchwork of requirements and compliance solutions is emerging across Europe.

Despite such rules on collection and recycling only one third of electrical and electronic waste in the European Union is reported as appropriately treated and the other two thirds are going to landfills and potentially to sub-standard treatment sites in or outside the European Union.

The report prepared for the European Commission (EU 2008) indicated recycling efficiencies of:

- 11% for light sources;
- 13% for batteries; and
- 50% switches, relays and related devices.

In December 2008 the European Commission proposed to revise the directives on electrical and electronic equipment in order to tackle the fast increasing waste stream of electronic waste. The aim is to increase the amount of e-waste that is appropriately treated and reduce the number that go to final disposal. The proposals also aim to reduce any administrative burden.

The Commission proposes to set mandatory collection targets equal to 65% of the average weight of electrical and electronic equipment placed on the market over the two previous years in each Member State. The recycling and recovery targets of such equipment now cover the re-use of whole appliances and weight-base targets will increase by 5%. Targets will also be set for the recovery of medical devices. Member States where the consumption of electrical and electronic equipment is widespread would have more ambitious targets under the new directive while others with smaller markets will have less ambitious targets.

5.4.6 Disposal of spent batteries and accumulators in EU Directive 2006/66/EC[73]

This legislation prohibits the placing on the market of most batteries and accumulators with a certain mercury or cadmium content and establishes rules for the collection, recycling, treatment and disposal of batteries and accumulators.

The Directive prohibits the placing on the market of certain batteries and accumulators with a proportional mercury or cadmium content above a fixed threshold. In addition, it promotes a high rate of collection and recycling of waste batteries and accumulators and improvement in the environmental performance of all involved in the life-cycle of batteries and accumulators, including their recycling and disposal.

The Directive prohibits:

- batteries and accumulators, whether or not incorporated in appliances, containing more than 0.0005% by weight of mercury (except for button cells, which must have a mercury content of less than 2% by weight); and

---

[73] See: http://europa.eu/scadplus/leg/en/lvb/l21202.htm

- portable batteries and accumulators, including those incorporated in appliances, with a cadmium content by weight of more than 0.002% (except for portable batteries and accumulators for use in emergency and alarm systems, medical equipment or cordless power tools).

To ensure that a high proportion of spent batteries and accumulators are recycled, Member States must take whatever measures are needed (including economic instruments) to promote and maximise separate waste collections and prevent batteries and accumulators being thrown away as unsorted municipal refuse. They have to make arrangements enabling end-users to discard spent batteries and accumulators at collection points in their vicinity and have them taken back at no charge by the producers. Collection rates of at least 25% and 45% have to be reached by 2012 and 2016 respectively.

Member States also have to ensure that, from 26 September 2009 at the latest, batteries and accumulators that have been collected are treated and recycled using the best available techniques. Recycling must exclude energy recovery. As a minimum, treatment must include removal of all fluids and acids. Batteries and accumulators must be treated and stored (even if only temporarily) in sites with impermeable surfaces and weatherproof covering, or in suitable containers.

If there is no viable end market, or if a detailed assessment of environmental, economic and social impact concludes that recycling is not the best solution, Member States may dispose of batteries and accumulators containing cadmium, mercury or lead in landfills or underground storage. Otherwise, it is prohibited to put waste from industrial and automotive batteries and accumulators into landfill, or to incinerate it; only residues from treating and recycling them may be disposed of in these ways.

Treatment and recycling may take place outside the Member State concerned or even outside the Community, provided EU legislation on the shipment of waste is respected. The producers have to bear the cost of collecting, treating and recycling industrial, automotive and portable batteries and accumulators, as well as the costs of campaigns to inform the public of these arrangements. Small producers may be exempted from this obligation if this does not impede the proper functioning of the collection and recycling schemes. All producers of batteries and accumulators have to be registered.

End-users are to be informed in various ways:

- through campaigns covering, among other things, the potential effects on the environment and human health of the substances used in batteries and accumulators, and the collection and recycling arrangements at the end-users' disposal;
- being directly informed by distributors that they can discard waste batteries and accumulators at sales points; and
- visible, legible and indelible markings on batteries, accumulators and battery packs with the following information: the symbol of the crossed-out wheeled bin (in Annex II to the Directive); the capacity of the accumulator or the portable battery; the chemical symbols Hg, Cd and Pb if the batteries, accumulators or button cells contain over 0.0005% mercury, over 0.002% cadmium or over 0.004% lead. If the battery, accumulator or battery pack are too small, this information appears on the packaging.

The Member States must send the Commission reports on the implementation of the Directive and the measures they are taking to encourage developments affecting the impact of batteries and accumulators on the environment (including new recycling and treatment techniques) - the first report will cover the period until 26 September 2012; subsequent reports are to be produced every three years. On the basis of these reports, the Commission must publish its own report on the implementation of the Directive and its impact on the environment and the functioning of the internal market.

A review of the Directive will be carried out after the second round of reports from the Member States. The Commission will examine the appropriateness of further risk management measures, minimum collection targets and minimum recycling obligations, and if necessary propose amendments to the Directive.

5.4.7 US National Vehicle Mercury Switch Recovery Program[74]

Figure 8: A typical mercury containing auto -switch

On August 11, 2006, the US EPA announced a national program to recover 80-90 % of all available mercury switches from scrap automobiles. The National Vehicle Mercury Switch Recovery Program is designed to recover an estimated 40 million mercury-containing light switches from scrap vehicles that are melted to make new steel. This will significantly reduce mercury air emissions from the fourth leading source in the United States – the furnaces used in steel making. The Program is the result of a two-year collaboration involving EPA, states, environmental organizations, and several industry sectors.

- Ten automakers created the End of Life Vehicle Solutions Corporation (ELVS), which will provide dismantlers with information and supplies needed for switch removal, collect and transport switches to proper recycling and disposal facilities, and track program performance.
- Participating dismantlers will remove mercury-containing switches and ship them to ELVS, giving the dismantlers the ability to market reduced mercury scrap and earn recognition and certain financial incentives.
- Participating scrap recyclers will build awareness of the mercury switch removal program in their own industry and in the dismantling industry, which is their chief supplier of scrap vehicles.
- Participating steelmakers will educate and encourage their supply chain to participate, and will take steps to purchase scrap metal generated from participating dismantlers and recyclers that have removed the mercury-containing switches.

In February 2008, the Program collected its millionth mercury-containing automotive switch, (of an estimated 67 million switches available for recovery over time) which represents more than one ton of mercury that has been removed from the environment. The goal of the program is to collect 80 to 90 percent of available mercury switches by 2017.

5.4.8 Mercury containing lamps

Efforts have been taken in a number of states in the USA, and in European and other countries to divert mercury containing lamps from landfill and direct them to recycling facilities. While targets for

---

[74] See: http://www.epa.gov/mercury/switchfs.htm#Program

recycling of mercury containing lamps as high as 80% have been adopted, reliable, consistent data on recycling rates are not always available.

As noted under the EUs WEEE Directive (see Section 5.4.5 above) mercury containing lamps are banned from disposal to waste but the directive has resulted in a patchwork of requirements and compliance solutions across Europe with a reported lamp recycling rate of 11% (EU 2008) against targets of 25% and 45% by 2012 and 2016 respectively. However, recycling rates in individual EU countries as high as 80% have been reported (Table 39).

Table 39: Recycling rates and targets in various locations for mercury-containing lamps

Country	Recycling rate of mercury-containing lamps	Target
US	24% in 2004	80% by 2009
Taiwan	87% (fluorescents only) in 2003	Not available.
Germany	70-80% in 1994	80% of all lighting products by 2006 (WEEE)
Switzerland	60-70% in 2005	80% of all lighting products by 2006 (WEEE)
The Netherlands	Well over 50% in 2005	80% of all lighting products by 2006 (WEEE)
Austria	50% in 2005	80% of all lighting products by 2006 (WEEE)

Table 39 is sourced from the Cities for Climate protection website (http://www.iclei.org/index.php?id=6625#c25489) which in turn references a Canadian report (Hilkene and Friesen 2005) which does not fully document the high recycling rates achieved in a number of countries.

The recycling varies considerably between States in the USA . Some States (Florida & Connecticut) have banned incineration of lamps; and others (i.e., Maine, Massachusetts New Hampshire Vermont, California and parts of Washington state) prohibit the disposal of mercury-containing lamps in domestic waste regardless of the mercury content.

The New Jersey mercury inventory indicated that 14% of lamps are recycled or disposed as hazardous waste. In a report for Environment Canada it was found that the percentage of mercury-containing lamps recycled in 2004 was 7% of all mercury-containing lamps disposed of. The balance were disposed of by other means, most likely to landfill.

In the USA it is estimated that promotion of CFLs plus declining cost will see a further increase in use of mercury in this category (Northeast Waste Management Officials Association (NEWMOA))/ IMERC 2008).

Under federal regulations, commercial and industrial entities are required to manage mercury-containing light bulbs as a hazardous waste after they burn out. Some manufacturers of fluorescent tubes produce "low-mercury lamps" that they claim pass the Toxicity Characteristic Leaching Procedure (TCLP) test for mercury. Under federal regulations, the TCLP determines if a lamp is a hazardous waste. The amount of mercury in a low-mercury bulb can range from 3.5 to 4 mg compared to a standard fluorescent bulb which ranges from 8 to 14 mg of mercury. These low mercury lamps may be identified by green end caps (often referred to as green-tipped lamps), or green etchings on the lamps.

Households are exempt from the federal regulations. The US EPA encourages the recycling of all mercury-containing lamps and is working with manufacturers and major retailers to develop, implement or expand recycling options. Expanding the accessibility and convenience of collection and recycling outlets for fluorescent lamps is a high priority for many government agencies, consumer groups and companies. Educating consumers and businesses about the importance of recycling is also important.

The phase out of incandescent lighting has focused attention on the management of end of life mercury containing lamps. In May 2009 the Environment Protection and Heritage Council (EPHC) announced Fluoro-cycle, a voluntary partnership between the Australian Government and industry to increase the recycling of mercury containing lamps in the commercial and public lighting sectors, which currently accounts for approximately 90% of all lighting waste. The scope of the program is expected to eventually be broadened to include lamps from the domestic or household sector. This may be necessary as the CFLs currently being bought to replace incandescent globes will reach end of life in the next three to five years.

Several states and local councils offer residential collection and recycling schemes and while robust data are unavailable, recycling rates are considered to be very modest (<5%)[75].

The NSW DECC Household Chemical Clean Out program collects fluorescent tubes which are transported to Victoria where the mercury is recycled.

Sutherland Council in Sydney introduced a scheme to recycle fluorescent tubes, its main focus being on tubes used in council facilities but also including collection points at a number of council community centres. Tubes are dismantled in Sydney and the mercury containing component transported to Melbourne for recycling. This program is undertaken at cost to Council and has limited promotion at this stage, although it is included on Council's website and in its waste management booklet. Since its inception in early 2008 Council has had many enquiries from other Councils (NSW and interstate). (Janelle Booth, personal communication)

5.4.9 Education – Information Campaigns

A number of examples from the above discussion, including:

- Evidence of continued use of / trade in banned mercuric oxide batteries (and/or parts) (EU 2008);
- Modest recycling rates reported despite strong regulatory programs;
- Appearance of new mercury containing products despite the availability of alternative technologies;

demonstrate how important it is that regulatory measures, policies and programs directed at minimising and eliminating the use of mercury and its disposal to the environment are supported by strong education and information campaigns if they are to be effective.

Some activities worthy of consideration:

- Dissemination of material targeted at specific groups – producers, suppliers, commercial, consumers etc;
- Dissemination of information on alternative products;
- Labelling of products;
- Purchasing procedures;
- Promotion of collection schemes;

---

[75] See: http://www.environment.gov.au/settlements/waste/lamp-mercury.html

- Spill procedures – proper handling schools, hospitals, laboratories etc;
- Targeting key areas to promote best practice – dentists; and
- End-of-life (for example, autos).

5.4.10 Rehabilitation

In many places the past use of land for industrial and waste disposal activities has left a legacy of land (and water bodies) contaminated with one or more hazardous substances which can pose ongoing risks to human health and the environment until properly rehabilitated, which requires that discharges or emissions from the site are controlled.

Superfund[76] is the name given to the best known environmental program established to address abandoned hazardous waste sites in the USA. It is also the name of the fund established by the Comprehensive Environmental Response, Compensation and Liability Act of 1980, (CERCLA). This law was enacted in the wake of the discovery of toxic waste dumps such as Love Canal and Times Beach in the 1970s. It allows the EPA to clean up such sites and to compel responsible parties to perform cleanups or reimburse the government for EPA-lead cleanups.

- The Superfund cleanup process is complex. It involves the steps taken to assess sites, place them on the National Priorities List, and establish and implement appropriate cleanup plans. This is the long-term cleanup process. In addition, the Agency has the authority;
- to conduct removal actions where immediate action needs to be taken;
- to enforce against potentially responsible parties;
- to ensure community involvement;
- involve the relevant States; and
- and ensure long-term protectiveness.

Contamination by mercury is a common way for a site to be placed on the Superfund National Priority List. A search of the NLP list reveals several hundred sites which include mercury, often with other substances, as the reason for the listing. Mercury is ranked third on the 2005 CERCLA Priority List of Hazardous Substances (below arsenic and lead). Table 40 lists the mercury contaminated sites by medium (with soil and groundwater being most common) and Table 41 shows the site types contaminated by mercury, with landfills being the most common. The number of sites in Table 40 exceeds the number in Table 41 because more than one type of media contaminated with mercury may be present at some sites.

Table 40: Number of Superfund sites with mercury as a contaminant of concern by media type

Media type	Number of sites
Soil	173
Groundwater	144
Sediment	92
Surface water	43
Debris	35
Sludge	24
Solid waste	18
Leachate	16
Other	11
Liquid waste	4
Air	4
Residuals	1

---

[76] See: http://www.epa.gov/superfund

Table 41: Number of Superfund sites with mercury as a contaminant of concern by site type

Site type	Number of sites
Landfills	102
Chemicals and allied products	15
Groundwater plume	14
Metals fabrication and finishing	12
Military	11
Batteries and scrap metal	9
Transportation equipment	9
Primary metals processing	7
Ordinance production	6
Mining	5
Electrical equipment	5
Chemical and chemical waste	5
Research and development	5
Other	85
Total	290

In the next section on controlling emission of mercury, some of the technologies used to manage and control discharges to water, waste and land are considered.

An Australian example of rehabilitation of a contaminated site

The last remaining mercury-based chlor-alkali plant in NSW closed in 2002. The emission of mercury to the atmosphere from the contaminated building which was left standing was estimated to be over 300 kg per year and was reported to the NPI. In 2006 the building was demolished and the bulk of the contamination material removed to a secure landfill site – effectively eliminating the sixth largest source of mercury in Australia. Ongoing treatment of contaminate material at the site should see further reductions in potential environmental risk in the future.

It is possible that in other cases where former industrial sites have been rehabilitated reduced risks due to mercury contamination have occurred.

In Australia, the Assessment of Site Contamination National Environment Protection Measure provides a nationally consistent approach to the assessment of site contamination and includes annual reporting by the jurisdictions. However, it does not include a register of contaminated sites and the management status of those sites.

5.5 REDUCING EMISSIONS

In many industrial applications, particularly where mercury is an (incidental) input contained in fuel or an ore, it may not feasible to eliminate the mercury from use and hence some form of 'end-of-pipe' control, may be required.

As is the case with the other management and policy options, in considering the control technologies presented in this section it is important to consider both the potential reductions (and hence benefits) and the costs of the options. *Any consideration of potential reductions should examine whether (and the extent to which) emissions reductions from the particular sources in question will yield reductions in risk to public health and the environment,* (USEPA 1997a).While some information on the costs of the options considered is included here with potential emission reductions, detailed discussion of cost benefit analysis specific to mercury control is beyond the scope of this report. The reader is referred to Volume VIII of the US EPA Mercury Study Report to Congress (USEPA 1997c) for detailed discussion of this issue.

In this section some detail is presented on industries where specific mercury controls are being considered or have been implemented, for example waste and/or municipal combustion or incineration, electricity generation from coal, crematoria. In a number of other industries, less attention has been paid to specific mercury controls, although existing pollution control measures, using devices such as electrostatic precipitators, bag filters and wet scrubbing systems for the control of particulate and acid gas emissions is likely also to result in the capture of some fraction of the mercury emissions.

As there are no longer any chlor alkali facilities in Australia, the control of mercury emissions from these facilities is not considered in this report, other than noting that the control of emissions of mercury from the chlor alkali industry has received attention and mostly focuses on the replacement of mercury-containing products with mercury-free alternatives.

5.5.1 Flue gas Treatment

Most metals have sufficiently low vapour pressure at typical air pollution control device (APCD) operating temperatures that condensation onto particulate matter – and high collection rates – are possible. Mercury, however, has a high vapour pressure at typical APCD operating temperatures and collection by particulate matter control devices is highly variable. Factors that enhance mercury capture are the presence of a sorbent and an effective method to collect the sorbent. In general, high levels of (unburnt) carbon in the flue gas enhance mercury capture. Additionally, the presence of hydrogen chloride can result in the formation of mercuric chloride which is more readily collected in APCDs. Conversely, elevated sulfur dioxide in the flue gas can act to reduce oxidised mercury to the elemental form which is more difficult to collect.

The control of atmospheric emissions of mercury from the combustion of coal used in the generation of electricity is dealt with in detail in this section as it is arguably the sector in which mercury control has received the greatest attention over more recent years. Also, a number of the technologies that are, or can potentially be applied in this industry, are also applied, or could be potentially be applied, to other source types of atmospheric mercury.

5.5.1.1 Coal combustion – Energy Production

Over recent years and in a number of countries the control of mercury from boilers used for the generation of electricity has gained significant attention, perhaps most prominently in the USA following the promulgation of the USEPA's Clean Air Mercury Rule in March 2005. The Clean Air Mercury Rule, which builds on EPA's Clean Air Interstate Rule (CAIR) would significantly reduce mercury emissions from coal-fired power plants. When fully implemented, these rules would reduce utility emissions of mercury from 48 tons a year to 15 tons, a reduction of nearly 70 percent.

The Clean Air Mercury Rule establishes "standards of performance" limiting mercury emissions from new and existing utilities and creates a market-based cap-and-trade program that will reduce nationwide utility emissions of mercury in two distinct phases. In the first phase, due by 2010, emissions will be reduced by taking advantage of "co-benefit" reductions – that is, mercury reductions achieved while reducing sulfur dioxide (SO_2) and nitrogen oxides (NO_x) under CAIR. In the second phase, due in 2018, utilities would be subject to a second cap, which will reduce emissions to 15 tons upon full implementation.

In February 2008 The United States Court of Appeals vacated (i.e., voided) EPA's rule removing power plants from the Clean Air Act list of sources of hazardous air pollutants. At the same time, the court vacated the Clean Air Mercury Rule. EPA is reviewing the court's decisions, evaluating its impacts, and developing alternate approaches to controlling mercury and perhaps other toxic metals[77].

Co-benefits

Control of mercury emissions from coal-fired boilers is currently achieved via existing controls used to remove particulate matter (PM), sulfur dioxide (SO_2) and nitrogen oxides (NO_x). This includes the capture of particulate mercury in PM APCDs, and soluble mercury (Hg^{2+}) in wet flue gas desulfurisation (FGD) systems. There are also data that indicate that the use of selective catalytic reduction (SCR) NO_x control enhances the oxidation of elemental (Hg^0) to Hg^{2+} resulting in its removal in wet FGD.

In the USA approximately 75 tons of mercury is contained in the coal delivered to power plants each year and about two thirds of this mercury is emitted to the air, resulting in about 50 tons being emitted annually. This 25-ton reduction is achieved by the PM, SO_2 and NO_x co-benefits described above (USEPA 1997c). This multi-pollutant approach has been central to the EPA's plan to reduce mercury from power plants, particularly the first phase reductions, noted above.

Figure 9 (from the USEPA (USEPA 2005)) shows potential mercury capture co- benefits for bituminous, sub-bituminous and lignite coal types for the following existing APCD types:

CS ESP	Cold side electrostatic precipitation;
HS ESP	Hot side electrostatic precipitation;
FF	Fabric filtration (bag filters);
FF/SDA	Fabric filtration in combination with a spray dry absorber;
CS ESP/FGD	Cold side electrostatic precipitation in combination with a wet FGD system;
HS ESP/FGD	Hot side electrostatic precipitation in combination with a wet FGD system.

---

[77] See: http://www.epa.gov/mercury/control_emissions/index.htm

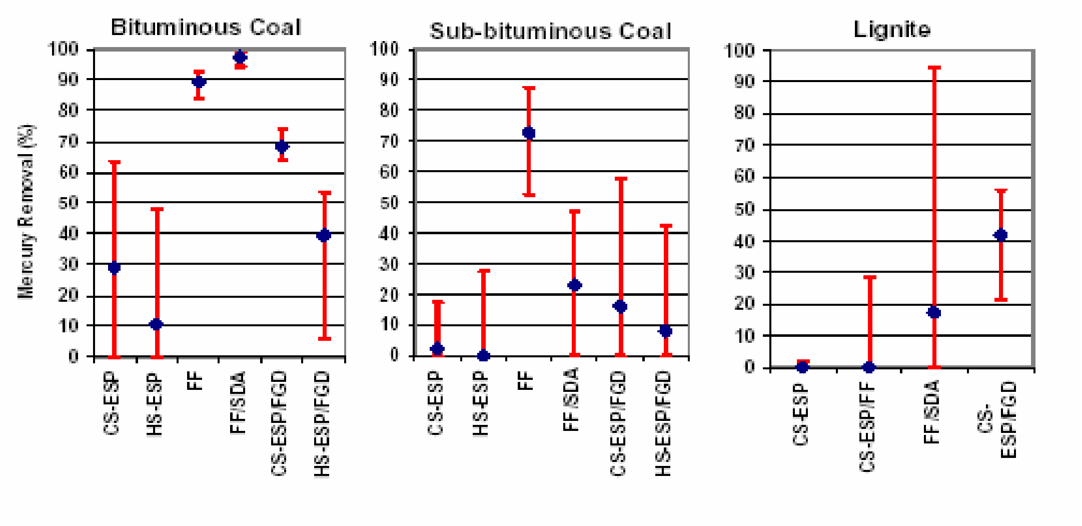


Figure 9: Mercury removal for different APCDs and coal types

The USEPA has also summarised average mercury captures by existing post-combustion control categories in coal-fired boilers, and this information is presented in **Error! Reference source not found.**. The data in **Error! Reference source not found.** and Figure 9 indicate:

- the highly variable nature of mercury capture for most APCDs and all 3 coal types;
- the potential to capture the majority of mercury from bituminous and sub-bituminous coal by using fabric filtration; and
- Very little mercury capture using CS ESP from lignite (a very low rank coal).

Of the APCDs listed in **Error! Reference source not found.** cold side electrostatic precipitation and fabric filtration are employed by the majority of coal fired power stations in Australia[78], as indicated in Table 43. Hot side ESP and Wet FGD systems are not used in Australian coal-fired power stations. Table 43 also provides information on the type of coal burnt in these power stations.

A consideration of the information in Table 43 in conjunction with the co-benefit data provide in Figure 9 suggests that a significant level of mercury control is occurring from coal-fired power stations in Australia with existing APCDs, particularly from power stations in Queensland and New South Wales equipped with fabric filters.

The existing level of mercury control is reflected in the emission estimation techniques manual for fossil Fuel Electric Power Generation used in the NPI (NPI 2005). This manual allows facilities to use either measurements or other estimation techniques in reporting emissions to the NPI.

In the case of mercury emissions from black (Bituminous and sub-bituminous) coal-fired generation, the Manual includes an estimation technique based on:

- The concentration of mercury in the coal which can be measured;
- The type of APCD used by the facility:
    - For ESPs the method assumes 54% of mercury in the coal is released to the air (46.5% retained in ash);

---

[78] Note that gas fired generation effectively emits no mercury and there is very little oil fired combustion in Australia

  - For fabric filters the method assumes 17% of mercury in the coal released to the air (83.1% retained in the ash); and
- The particulate collection efficiency of the APCD, which can be measured and/or calculated.

Table 42: Average mercury capture by existing post-combustion control configuration. Derived from Feeley et al (2005; 2008), based on EPA testing

Post-combustion Control Strategy	Post-combustion Emission Control Device Configuration	Average Mercury Capture by Control Configuration (%)		
		Coal Type		
		Bituminous	Sub-bituminous	Lignite
Particle Matter (PM) Control Only	CS-ESP	36	3	0
	HS-ESP	9	6	Not tested
	FF	90	72	Not tested
	Particulate Scrubber (PS)	Not tested	9	Not tested
PM and Spray Dryer Adsorber for S capture	SDA + CS-ESP	Not tested	35	Not tested
	SDA + FF	98	24	0
	SDA + FF + SCR	98	Not tested	Not tested
PM and Wet FGD system	PS + FGD	12	0	33
	CS-ESP + FGD	75	29	44
	HS-ESP + FGD	49	29	Not tested
	FF + FGD	98	Not tested	Not tested

The figures for the percentage of mercury emitted from black coal combustion were obtained from data prepared for the USEPA's Toxic Release Inventory[79]. The NPI emission estimation techniques for brown coal-fired generation are simple factors based on coal consumption and not requiring the input of site specific information as described for black coal power generation.

Stability of mercury collected with fly ash

Mercury captured in the ESPs and FFs is associated with the fly ash and is either disposed of to purpose built ash dams or sold for use in the building industry, most commonly as a replacement for cement in making concrete. While research is ongoing into the emission of mercury from flyash and FGD wastes, studies to date (Pavlish *et al.* 2003; Schroeder *et al.* 2007) have indicated that concentrations in the leachates from storage sites are very low and usually below detection limits and essentially no mercury emission from these material to air has been measured at ambient temperatures using very sensitive techniques. However, mercury is released from saturated sorbents above 135ºC.

---

[79] See USEPA Emergency Planning and Community Right-to-Know Act EPCRA Section 313: Guidance for Reporting Toxic Chemicals: Mercury and Mercury Compounds Category (EPA 260-B-01-004), August 2001 Mercury and Mercury Compounds Category Guidance, and http://www.epa.gov/tri/TWebHelp/WebHelp/guidance_materials/guidance_for_reporting_toxic_chemicals_mercury_and_mercury_compounds_category.htm

Table 43: Particulate matter APCDs used in Australian power stations and the type of coal burnt.

Power stations by state	APCD	Coal type
NSW		
Bayswater	FF	Bituminous
Eraring	FF	Bituminous
Liddell	FF	Bituminous
Munmorah	FF	Bituminous
Mt Piper	FF	Bituminous
Vales Point	FF	Bituminous
Wallerawang	ESP	Bituminous
Queensland		
Callide B	ESP	Bituminous
Callide C	FF	Bituminous
Collinsville		Bituminous
Gladstone	FF	Bituminous
Kogans Creek	FF	Bituminous
Millmerran	FF	Bituminous
Stanwell	ESP / FF	Bituminous
Swanbank B	FF	Bituminous
Tarong	ESP	Bituminous
Tarong North	FF	Bituminous
South Australia		
Northern	ESP	Low rank – sub bituminous
Thomas Playford B	ESP	Low rank – sub bituminous
Victoria		
Anglesea	ESP	Low rank Similar to lignite
Hazelwood	ESP	Low rank Similar to lignite
Loy Yang A	ESP	Low rank Similar to lignite
Loy Yang B	ESP	Low rank Similar to lignite
Morwell	ESP	Low rank Similar to lignite
Yallourn	ESP	Low rank Similar to lignite
Western Australia		
Collie	ESP	Low rank – sub bituminous
Kwinana WPC	ESP	Low rank – sub bituminous
Muja	ESP	Low rank – sub bituminous

Additional mercury control

Although mercury capture in APCDs can remove a significant fraction of the mercury released during coal combustion, as shown in the previous section, the implementation of the USEPA's Clean Air Mercury Rule would require consistently very high mercury capture efficiencies which in turn would require the development and implementation of new technologies. In the US, a range of approaches are being developed, many based on activated carbon injection (ACI). Under the Office of Fossil Energy's Innovations for Existing Plants (IEP) Program the US DoE's National Energy Technology Laboratory (NETL) in collaboration with many other organisations, including USEPA, a number of these technologies are being trialled at scales from bench to full scale, with the aim to improve performance and reduce cost in future years.

Activated Carbon Injection (ACI)

In ACI technology, powdered activated carbon (PAC) sorbent is injected into the flue gas at a location in the duct preceding the particulate matter (PM) control device, which usually is an electrostatic precipitator (ESP) or a fabric filter (FF). The sorbent binds with the mercury in the flue gas in the duct and in the PM control device. Subsequently, the mercury-containing PAC is captured in the PM control device. Greater mercury removal is obtained with a fabric filter compared to an ESP because of enhanced gas-particle contact in the filter cakes on the surface of the bags in a fabric filter. This approach is shown schematically in Figure 10.

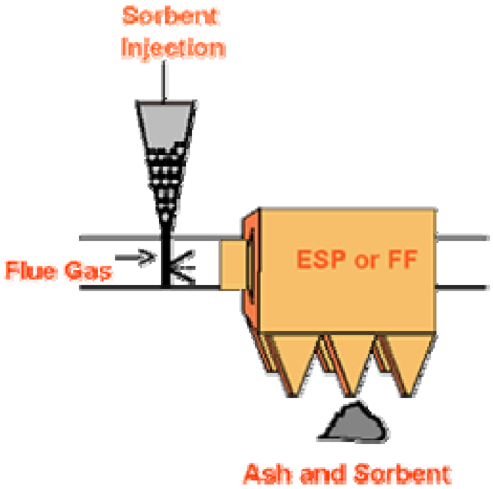


Figure 10: Schematic of Activated Carbon Injection for Mercury Control

The performance of activated carbon is related to its physical and chemical characteristics. Generally, the physical properties of interest are surface area, pore size distribution, and particle size distribution. The capacity for Hg capture generally increases with increasing surface area and pore volume. The ability of Hg and other sorbents to penetrate into the interior of a particle is related to pore size distribution. The pores of the carbon sorbent must be large enough to provide free access to internal surface area by Hg^0 and Hg^{2+} while avoiding excessive blockage by previously adsorbed reactants. As particle sizes decrease, access to the internal surface area of the particle increases along with potential adsorption rates.

Carbon sorbent capture is also dependent upon temperature, the concentration of Hg in the flue gas, the flue gas composition, and other factors. The selection of a carbon for a given application should take into consideration the total concentration of Hg, the flue gas composition, and the method of capture (i.e., ESP, FF or dry scrubber).

Activated carbon sorbents are made from coal or biomass; brominated activated carbon (BAC) sorbents are chemically treated with halogens such as bromine or chlorine and have been shown to be cost effective (Chang *et al.* 2008). None of the non-carbon sorbents tested to date have achieved high mercury removal rates. Boiler bromide additives can enhance ACI performance for low chlorine coals – as effectively as brominated ACI.

At present, ACI is the most widely studied of the mercury-specific control technologies for coal-fired power plants and shows the potential to achieve moderate-to-high levels of mercury control, as shown in Figure 11; the highest capture rates (>90%) are achieved with coal with moderate to high chlorine content and the use of fabric filters or TOXECON[80] technology.

---

[80] TOXECON involves the use of a relatively small fabric filter plant downstream of an ESP, with sorbent injection after the ESP

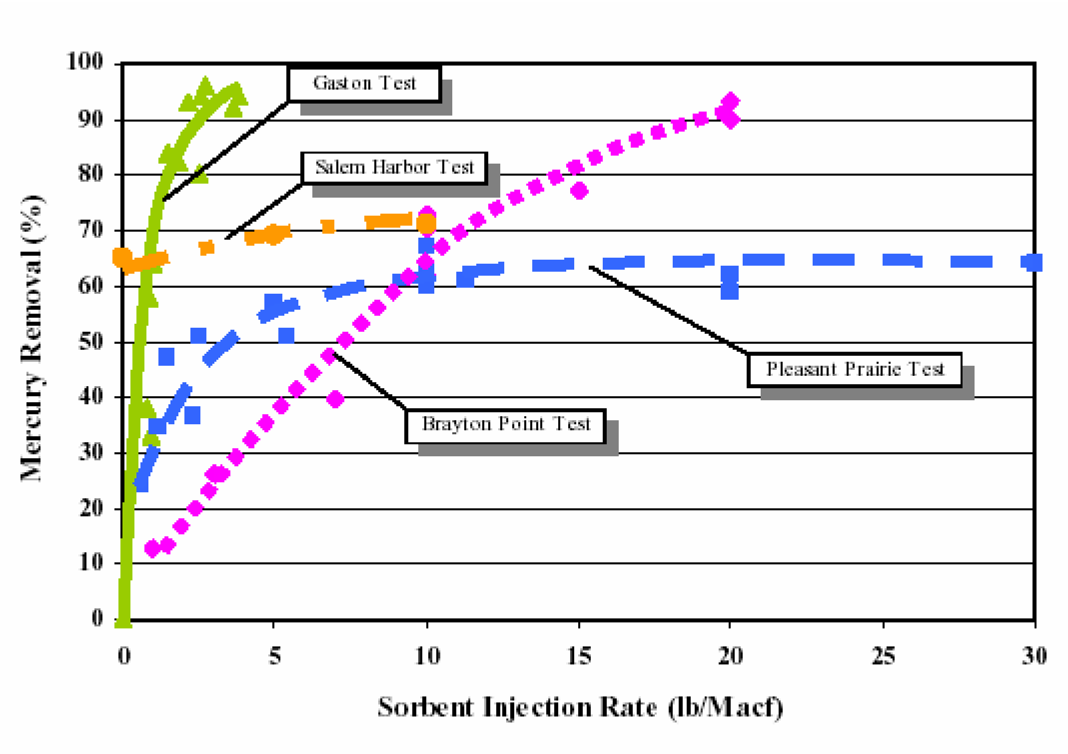


Figure 11: US DoE Activated Carbon Test data. Available at http://www.netl.doe.gov/coal/E&WR/pubs/mercuryR&D-v4-0505.pdf

The extensive experience gained with ACI applications on municipal waste combustion (MWC) and medical waste incineration (MWI) facilities over recent years is not directly transferable to applications on coal-fired boilers for several reasons:

- In general, the concentration of mercury in the flue gas of MWCs and MWIs is an order of magnitude higher than for coal-fired boiler systems, for example, 200-1000 micrograms/dscm compared to 5-30 micrograms/dscm. It is well known that removal of mercury by ACI is limited by the mass transfer (i.e., the transfer of mercury from the bulk gas to the surface of the carbon particle) in the duct and/or the ESP. This mass transfer is greater for higher concentrations of mercury in the flue gases of MWC and MWI systems. Based on this difference in mass transfer, the amount of mercury captured per unit mass of AC injected will, in general, be higher in MWCs and MWIs compared to coal-fired boilers;

- The flue gases of MWCs and MWIs have higher chlorine contents than those found in flue gases of coal-fired utility boilers, particularly boilers firing low-rank coals. Performance of ACI in situations with low levels of chlorine in the flue gas may be adversely affected as was evidenced at the Powder River Basin (PRB)-fired Pleasant Prairie plant. Consequently, ACI performance on coal-fired boilers will not, in general, be equivalent to that on MWCs and MWIs; and

- In general coal-fired power plants are much larger in size compared to MWCs or MWIs. For example a large MWC may be about the same size as a small, 40-50 MW, coal-fired plant. Accordingly, duct dimensions, generally, are much larger in coal-fired plants compared to those at MWCs and MWIs. Since mixing of injected AC and flue gas in the duct affects mercury capture performance, design of AC injection systems will, in general, be more involved for coal-fired boilers.

Testing of activated carbon and other sorbents has been undertaken at coal-fired utility boilers. Research on ACI and other mercury-specific technologies has been underway at the Department of Energy (DOE) for several years. The first full-scale commercial demonstration project for ACI was initiated by DOE in April 2004 and is projected to be completed in 2009. A number of similar projects will be necessary to establish this technology's effectiveness on other coal types.

Further the US Department of Energy [81] is of the view that "*there remain a number of critical technical and cost issues that need to be resolved through additional research before these technologies can be considered commercially available for all U.S. coals and the different coal-fired power plant configurations in operation in the United States*"**.**

The following issues are under further investigation:

- Difficulty in removing elemental Hg, particularly when combusting lower rank coals, where elemental mercury is the predominant species formed;
- Competition for activated carbon bonding sites between mercury and acid gases such as sulfur trioxide (SO_3) produced when burning high sulfur coal, which may limit the effectiveness of mercury control via ACI;
- The effect of continuous operation of ACI on performance of particulate control devices; and
- Potential impacts of mercury on coal by-product management and associated costs.

Costs of mercury Control with ACI

As discussed above, mercury can be removed from many power plants as a co-benefit of existing control devices, requiring little, if any, additional investment and/or operational costs and therefore the marginal costs of reduction using existing APCDs are close to zero.

The cost of control using ACI will vary significantly depending on a number of factors, including:
- Existing control technology (ESP, FF, FGD for example);
- Capture efficiency required;
- Coal type and chlorine, sulfur content; and
- Type and cost of sorbent.

The capital costs of ACI systems are usually very small compared to other equipment if additional PM control device retrofit is not required (USEPA 2005). Capital costs of ACI are about $US5 per kW (USEPA 2003); the fixed operating costs are also relatively low. This capital cost represents a situation where only the activated storage silo and injection system are required. However, if a completely new fabric filter is required, the ACI capital cost can increase to $126 per kilowatt[81]. Therefore the major costs associated with ACI system are the cost of sorbent and the disposal of additional fly ash materials.

For most units, mercury removal would add no more than about $US2 per MWh to the annualised cost of power production (Srivastava *et al.* 2005).

The operating cost of ACI depends on the rate of carbon injection needed to achieve the required mercury removal efficiency. This injection rate depends on the temperature of the flue gas and the type of coal used in the boiler. Increasing the carbon injection rate increases the carbon: mercury ratio and hence the removal rate. Performance can be maintained at lower carbon injection rates by

---

[81] See http://www.netl.doe.gov/technologies/coalpower/ewr/mercury/pubs/NETL%20Clarification%20on%20Mercury%20FINAL%200406.pdf. Accessed on 21 Octorber 2007

decreasing the temperature and the particle size of the activated carbon, and by increasing the contact time and the amount of chlorine present in the system (USEPA 2005a). Additional factors that can influence the economics of mercury control include, but are not limited to: economic/financial assumptions, process factors (for example, additional or extended outages) and required modifications to existing equipment.

In a recent summary Chang *et al* (2008) note that while brominated activated carbon costs about 30% more than untreated activated carbon, its performance is significantly better than ACI for US western (low rank coal) applications. They note that to date, no significant differences between ACI and brominated ACI have been observed for plants equipped with fabric filters.

Chang et al (2008) report that the costs for a 500MW plant achieving 90% mercury control, range from $US 2 million per year for a fabric filter plant using ACI, $US 10 million for ESP with ACI and $US 15 million for TOXECON with ACI. The authors note that the above costs assume the power plant was not selling flyash. At units where flyash sales are lost due to sorbent injection, annual control costs increase by about $US 2 million due to disposal costs and lost revenue.

Feeley *et al* (2008) report that a US DoE analysis in May 2007 indicates that the high capture efficiency of chemically treated sorbents has "drastically reduced the estimated cost of Hg control due to a reduction in the injection rate required to achieve a given level of control, which offsets the higher costs of these sorbents. The levelised incremental cost of 90% control using chemically treated ACI ranges from about $US 30,000 /lb (13,600 / kg) with ESP to less than US$10,000 /lb (4,500 / kg) for FF. The authors note that the results indicate that NETL has surpassed the Hg control cost goal of $US 40,000 /lb ($US 18,200 / kg) set by the IEP program.

Fly ash impacts associated with sorbent injection

As shown in Figure 10, typical ACI systems are located upstream of a PM_{10} control device to enable simultaneous capture of the spent sorbent and fly ash. This Hg control strategy leads to co-mingling of the sorbent and fly ash resulting in the potential limitations in the recycling of fly ash. One of the highest value reuse applications of fly ash is as a substitute for Portland cement in concrete production. The utilisation of fly ash in concrete production is particularly sensitive to carbon content as well as the surface area of the carbon present in fly ash. NETL is developing technologies which are designed to minimise fly ash contamination caused by ACI injection upstream of a PM_{10} control device:

- TOXECON does not impact on fly ash utlisation since the majority of the fly ash is collected in an ESP upstream of sorbent injection. Only a small percentage of the fly ash is collected with the sorbent in the downstream fabric filter In TOXECON II sorbent is injected directly into the downstream fields of an ESP, with most fly ash being collected in the upstream fields;
- "Ash-friendly" sorbents – various products are being developed / tested which might enable co-mingled sorbent and fly ash to be used in concrete production; and
- Sulfur trioxide interference – even at low concentrations sulfur trioxide can impede the performance of ACI by competing with Hg for adsorption sites on the sorbent. This could be an issue where higher sulfur coal is burnt or where SO_3 is used to condition the fly ash for collection by an ESP. Dual injection of Hg sorbents with alkaline materials, to mitigate the SO_3 effects, is being investigated (Feeley *et al.* 2008).

Other mercury Control

In addition to ACI and chemically treated ACI, there are other mercury control technologies under development. It should be noted that these estimates are considered preliminary and are expected to be refined as mercury control technologies mature to commercial status.

In the USA more than 20% of power plants use a wet FGD system to control SO_2 emissions. Oxidation of (insoluble) flue gas Hg^0 followed by absorption of Hg^{2+} across a wet FGD system therefore has the potential to be a cost effective mercury control strategy for some power plants. The development of a number of technologies that have the potential to promote the oxidation of elemental mercury are being funded by NETL.

Chemical additives – halogen compounds sprayed onto the coal as an aqueous salt solution or injected directly into the boiler have been investigated with low rank coals (Chang *et al.* 2008; Feeley *et al.* 2008).

Oxidation catalysts Feeley *et al* (2008) report on the progress using fixed bed catalysts to promote oxidation with subsequent capture of $Hg^{2+.}$ Total mercury capture at pilot scale in excess of 80% were reported compared with 36% at baseline conditions. Selective catalytic reduction (SCR) for nitrogen oxide control "*can act to oxidise a significant portion of the elemental mercury, which makes it easier to remove downstream in a wet FGD.*"

5.5.2 Waste Disposal

As a general comment and prior to considering technological mercury flue gas controls in medical waste incineration, domestic waste combustion and incineration and hazardous waste incineration, it is worth noting that the most cost effective form of control is to prevent mercury entering the waste stream in the first place by establishing effective domestic collection schemes, industrial / trade waste regulations and recycling centres. The Minnesota Pollution Control Agency reports that it costs $200 - $500 per pound of mercury emission reduction through source separation compared to $3,400 - $7,600 per pound for air pollution controls (Ross and Associates 2000).

The report prepared for the Great Lakes (Ross and Associates 2000) identified several air pollution control technologies that can be employed in Municipal Waste incinerators (MWIs) , Municipal Waste Combustors MWCs) Medical Waste Incinerators MedWIs and Hazardous Waste Incinerators (HWIs). These include:

- Carbon filter beds HWIs, MWIs;
- Wet scrubbers MWIs, MWCs, MedWIs;
- Selenium filters MWIs;
- Activated carbon injection MWIs, MedWIs; and
- Depleted brine scrubbers for chlor alkali plant (not included here).

It should be noted that facilities using these technologies might also employ particulate control devices (electrostatic precipitators, bag filters) which are capable of collecting a fraction of the mercury in the flue gas (see Section 5.5.1.1 on coal combustion).

The following is from the Great Lakes Report (Ross and Associates 2000) which references the Report to Congress 1997 (USEPA 1997a).

Carbon filter beds have been developed in Europe for use as a final cleaning stage in MWCs and utility boilers to remove heavy metals (including mercury), organic pollutants and acid gases. Reported mercury removal from a MWC > 99% Cost effectiveness studies indicate $513 - $1083 per pound mercury removed. Disposal of the spent carbon is a potential negative impact – may be combusted in a plant which has a wet scrubbing system or disposed to landfill. The possible negative impact of mercury released during the coal charring process of the carbon activation process has been dismissed as negligible.

Wet scrubbing systems are used principally to control acid gases, metals, PM, dioxins and furans. Their effectiveness in removing mercury depends on the amount of water soluble divalent mercury in the gas stream (elemental mercury is not water soluble). A 90% reduction of mercury is possible with a wet scrubber on a MWC.

Cost effectiveness is estimated to be $1,600 - $3,320 per pound of mercury removed from MWCs and $2,000 - $4,000 per pound for MedWIs. The resulting wastewater contains concentrated contaminants, including metal chloride complexes including mercury, which may be treated by use of precipitants.

Selenium filters have been developed for mercury removal from MWIs, smelters and also applied to a crematorium in Sweden. At higher mercury inlet concentrations the life of the selenium filter is short and alternative controls are recommended. Cost effectiveness has not been estimated and USEPA says application is "limited".

Activated carbon injection involves injection of powdered carbon into the flue gas upstream of an air pollution control device (ESP, bag filter, scrubber). Activated carbon is a specialised form of carbon produced by pyrolysing coal or various hard vegetative materials (wood) to remove the volatile matter. The resulting char then undergoes a steam or chemical activation process to produce an activated carbon that contains multiple internal pores and has a very high specific surface area. With this pore structure the activated carbon can adsorb a broad range of trace contaminants , including mercury. After injection into the flue gas and adsorption of contaminants, the activated carbon is captured in the PM_{10} control device.

Factors affecting performance include temperature, injection rate of AC, the concentration and species of mercury, the extent of contact between carbon and mercury and type of carbon used. High chlorine in the waste stream promotes formation of $HgCl_2$ which is effectively captured. With chemically impregnated AC the contaminant reacts with the chemical that is bound to the carbon. This technology has been used on MWCs and MWIs in Europe and the US with very high removal rates up to 96%. The cost of removal is estimated to be $211 - $870 per pound from MWCs and $2,000 to $4,000 from MedWIs. Testing showed mercury collected by the carbon was stable at a US MWC – low potential to be re-emitted to atmosphere.

Each of these technologies transfers wastes from air to either solid or liquid waste. Data indicates that mercury is not readily released (leached) from ash storage (see coal combustion section).

5.5.3 Mercury from gold production

Mercury commonly occurs with gold bearing rocks and the mining and processing of gold bearing ores is a potentially significant source of mercury emissions to the environment. The NPI reports that two gold mining and processing operations in Kalgoorlie, WA account for a large fraction of the mercury emissions from point sources in Australia.

Mercury emissions to the environment may occur from a number of points in the gold production process:

- For those ores that are unoxidised, the single largest source of mercury is from sulfide or carbonaceous ore roasters, autoclaves or other thermal processes that convert reduced gold ore to a form that can be more efficiently extracted with cyanide;
- The second major source of atmospheric release of mercury is from the carbon regeneration units which form part of the process that converts gold cyanide to dore bars; and
- A third type of atmospheric release of volatilised mercury is from waste rock dumps, tailings facilities and extracted heap leach piles (Jones and Miller 2005).

Jones and Miller (2005) report on measures that have been undertaken by gold mining facilities in Nevada, USA, to reduce mercury emissions. In 1998, Nevada was the second largest mercury emitting state in the USA (behind Texas) with gold mining being the largest source of emission in Nevada. Significant reductions in mercury emission from gold production facilities have occurred, largely as a result of Voluntary Mercury Air Emissions Reduction Program (VMRP) commenced in 2001 between the four largest emitting gold producers, the US EPA and the Nevada Division of Environmental Protection.

The aims of the VMRP[82] were to:

- "Achieve significant, permanent and rapid reductions in mercury air emissions from gold mining operations; and
- Achieve reductions through approaches that are most suitable for each individual mining facility".

The aim was to achieve a 33% reduction in mercury air emissions by 2003, and a 50% reduction by the end of 2005. The achieved reductions exceeded this aim, and reductions of 82% were achieved in mid 2005. Further reductions in mercury emissions are expected in the future, but not as dramatic as have occurred in the first years of the VMRP.

Jones and Miller (2005) summarise the data from the VMRP, including measures adopted at the four mines, including:

- Installation and implementation of control technology
    - The addition of carbon columns and carbon filtration units to adsorb mercury;
    - Chemical treatment of flue gases, including sodium hypochlorite and mecurous chloride injection and subsequent scrubbing;
    - Baghouses (fabric filters), ESPs and SO_2 scrubbing (some installed prior to VMRP);
- Pollution prevention; and
- Waste minimization.

Jones and Miller (2005) do not report on the cost of installing and operating these technologies.

The EPA adopted a voluntary approach for a number of reasons:

- There was no existing regulatory requirement for these mines to control emissions;
- The Clean Air Act regulations establishing Maximum Achievable Control Technology (MACT) emission standards did not require emission standards for the gold mining industry; and
- Establishment of a MACT would have been lengthy.

5.5.4 Crematoria

As discussed in Section 5.3.3.3 it can be expected that emissions from crematoria will increase in coming years in many jurisdictions and it is further noted that these facilities are often located in residential areas and have a relatively short stack (chimney). Therefore the control of mercury emissions from crematoria has gained attention in recent years.

The UK Department for Environment, Food and Rural Affairs (DEFRA 2003) estimates that, without intervention, mercury emissions from crematoria will increase by two thirds between 2000 and 2020, followed by a plateau or slight rise to 2035 followed by a decrease back to 2000 levels around 2055. By 2020 it is estimated crematoria will emit between 11 and 31% of UK mercury emissions to air.

---

[82] See http://ndep.nv.gov/mercury/docs/voluntar_mercury_q&a05.pdf; accessed 8th May 2006

According to the European Commission (Concorde 2007) most of the problem with mercury emission from crematoria is addressed by an OSPAR[83] recommendation. The recommendation which does not have the force of law calls for the use of Best Available Techniques (BAT) for controlling mercury emissions from crematoria and lists 4 types of control technology:

- Co-flow filters, using an absorbent for mercury with capture by a cloth filter;
- A solid bed filter, using absorbents such as cokes or zeolites;
- Traditional gas scrubbing techniques; and
- Honeycomb catalytic absorbers, using precious metal (gold/platinum) following particulate removal.

Concorde (2007) report that implementation of the recommendation has been "rather limited" with over 80% of the nearly 1000 crematoria in the EU-27 having little or limited emission controls. Fewer than 5% of crematoria have installed devices specifically for reducing mercury emissions, although more are planned in the next 5 years, possibly as a result of the implementation of emission standards for mercury, initially for new or large facilities. The UK, amongst a number of other countries[84] has established national standards regulating emissions from crematoria which require that new crematoria be fitted with mercury abatement technology (DEFRA 2004; DEFRA 2005). For new processes, the control of mercury (and dioxins and furans) is based on a system of:

"*cool capture and collect. The hot exhaust gases are cooled, using for example water tube coolers. Injecting dry lime, activated carbon and sodium sulphide into the gas stream captures pollutants. A dry filter captures the particulate matter and a reduction of between 90 and 98% in mercury concentrations is expected. Alternatives with equal or better performance may be accepted. However, conditions in a permit stating a percentage reduction are not recommended.*" (DEFRA 2004), p.17

The UK Department for Environment, Food and Rural Affairs (DEFRA 2003) concluded that the cost of abating mercury by gas cleaning crematoria exhausts in the UK is higher than other abatement costs that industry bears, but that gas cleaning (at around 55 pounds per cremation total cremation cost average 1215 pounds ~ 5%) is unlikely to affect the viability of the majority of crematoria. By 2020 crematoria will be, by far, the biggest single contributor to national mercury emissions unless steps are taken. DEFRA also report that 23% of responses to a survey of crematorium operators indicated that they would close rather than install gas cleaning equipment with lack of capital and space limitations being the main reasons quoted. It is more likely that the figure would be around 15%.

5.5.5 Other Industries

As discussed in the introduction to this section and the section on co-benefits in many industries where mercury is an incidental emission, existing APCDs are likely to remove some fraction of the mercury from the exhaust gases. The percentage of mercury removed will depend critically upon the form in which the mercury is emitted (particulate, metallic, ionic), the pollution control device and the characteristics of the gases, including temperature and chlorine content.

Some of the industries to which this co benefit might apply include:

- Primary aluminum production;

---

[83] OSPAR is the mechanism by which fifteen Governments of the western coasts and catchments of Europe, together with the European Community, cooperate to protect the marine environment of the North-East Atlantic. It started in 1972 with the Oslo Convention against dumping.

[84] National standards that require gas cleaning at new or large facilities in place in Austria, Belgium, The Netherlands, Norway Sweden and Switzerland. (Defra 2003)

- Portland cement manufacture;
- Brick and related clay product production;
- Primary metal smelting (copper, lead zinc);
- Iron and steel production; and
- Foundries.

These industries typically employ APCDs (electrostatic precipitators (hot and cold side); fabric filters; and wet scrubbers) which have been discussed previously, as well as cyclones, venture scrubbers, afterburners and catalytic incineration.

Selenium filters have been developed to reduce elemental mercury emissions in metallurgical processes (See Chapter 8 of the UN Global Mercury assessment (UNEP 2002); (USEPA 1997a)) and reported to have been applied to a crematoria in Sweden ((USEPA 1997a). Removal efficiencies of 90% are reported to have been achieved Chapter 8 of the UN Global Mercury assessment (UNEP 2002); (USEPA 1997a).

5.5.6 Treatment technologies for soil, waste and water

A recent report from the US Superfund program (USEPA 2007) provides a valuable summary of the availability, performance and cost of technologies used to treat mercury contamination in soil, waste and water from 57 projects in the Superfund program. The report notes that the information provided can "*serve as a starting point to identify options for mercury treatment*" and that the feasibility of particular technologies "*will depend heavily on site-specific factors, and final treatment and remedy decisions will require further analysis, expertise, and possibly treatability studies.*"

The Superfund report provides a significant amount of detail on the individual technologies and on their application at specific contaminated sites. Table 44 is a summary of the technologies and their applications.

Table 44: Overview of Selected Mercury Treatment Technologies (USEPA 2007)

Technology	Description
Soil and Waste Treatment	
Solidification/ Stabilisation	Physically binds or encloses contaminants within a stabilised mass and chemically reduces the hazard potential by converting the contaminants into les soluble, mobile or toxic forms. Most frequently used technology to treat soil and waste contaminated with mercury. Generates residual not requiring further treatment. Information on the long term stability of mercury containing soil and waste from this technology were not available, but presumably will be in the future as disposal sites age.
Soil Washing/ Acid Extraction	Uses the principle that some contaminates preferentially adsorb onto the fines fraction of soil. The soil is suspended in a wash solution and the fines are separated from the suspension, thereby reducing the contaminant concentrations in the remaining soil. Acid extraction uses an extracting chemical, such as hydrochloric acid, or sulfuric acid. Used primarily for soils with relatively low clay content as they tend to be separable into a highly contaminated fines fraction. Less effective with high organic content
Thermal Desorption/Retorting	Application of heat and reduced pressure to volatilise mercury from the contaminated medium, followed by conversion of the mercury vapours into liquid elemental mercury by condensation. Off-gases may require further

	treatment though additional air pollution control devices such as carbon units. Used to treat industrial and medical waste Generally not suitable for high clay or organic content soils
Vitrification	High temperature treatment that reduces the mobility of metals by incorporating them into a chemically durable, leach resistant vitreous mass. The process also may cause contaminants to volatilise, thereby reducing their concentrations in the soil and waste. Has been used with high organic content wastes.
Water Treatment	
Precipitation/ Co precipitation	Uses chemical additives to: (a) transform dissolved contaminants into an insoluble solid, or (b) form insoluble solids onto which dissolved contaminants are adsorbed. The insoluble solids are then removed from the liquid phase by clarification or filtration. The most commonly used process to treat mercury contaminated waste, with its effectiveness less likely to be affected by the characteristics of the waste compared with other water treatment technologies. Generally requires skilled operators so more cost effective at large scale.
Adsorption	Concentrates solutes at the surface of a sorbent, thereby reducing their concentration in the bulk liquid phase. The adsorption media usually packed into a column. Tends to be used when mercury is the only contaminant to be treated, for smaller systems and as a polishing technology for effluent from larger systems.
Membrane Filtration	Separates contaminants from water by passing water through a semi-permeable barrier or membrane. The membrane allows some constituents to pass while blocking others. High costs, larger volume of residuals
Biological Treatment	Involves the use of micro organisms that act directly on contaminant species or create ambient conditions that cause the contaminant to leach from soil or precipitate/co precipitate from water. Effective at pilot scale

The Superfund report also describes a number of innovative approaches have been applied at bench and pilot scale, including:

- Nanotechnology;
- Phyto-remediation;
- Air stripping; and
- In-situ thermal desorption.

Table 45 provides a summary of the application of the technologies outlined in Table 44. Table 46 provides further detail on treatment technologies, including factors that may affect performance and/or cost.

Table 45: Summary of applications used for treatment technologies

	Media Treated		Number of projects Identified		
Technology	***Soil and waste***	***Water***	***Pilot scale***	***Full Scale***	***Total***
Solidification/ Stabilisation	■		6	12	18
Soil washing	■		6	2	8
Thermal Treatment	■		5	3	8
Vitrification	■		2	1	3
Precipitation		■	0	11	11
Adsorption		■	2	4	6
Membrane Filtration		■	0	1	1
Bioremediation		■	2	0	2
TOTAL			**23**	**34**	**57**

Table 46: Mercury Treatment Technologies Screening Matrix

Technology	*Development Status*	*Treatment train (excludes off gas treatment)*	*Residuals*	*O&M or Capital intensive*	*Factors that may affect performance and or cost*	
					Matrix characteristics	*Operating parameters*
Soil and Waste						
Solidification/ Stabilisation	Full scale	No	Solid	Capital	▪ pH of media ▪ presence of organic compounds ▪ particle size ▪ moisture content ▪ oxidation state of mercury ▪	▪ type of binder and reagent ▪ mixing of waste and binder
Soil Washing/ Acid Extraction	Full scale	Yes	Solid, Liquid	Capital O&M	▪ Soil homogeneity ▪ presence of organic compounds ▪ particle size ▪ pH of media ▪ moisture content ▪	▪ temperature
Thermal Treatment	Full scale	No	Solid, liquid, vapour	Capital O&M	▪ presence of organic compounds ▪ particle size ▪ moisture content	▪ residence time ▪ system throughput ▪ temperature ▪ pressure ▪ e
Vitrification	Full scale	No	Solid, vapour	Capital	▪ lack of glass forming materials ▪ particle size ▪ moisture content ▪ subsurface air pockets ▪ presence of organic compounds	▪ temperature
Water						
Precipitation/ Co precipitation	Full scale	Yes	Solid	Capital O&M	▪ pH of media ▪ presence of other contaminants	▪ chemical dosage
Adsorption	Full scale	Yes	Solid	O&M	▪ pH of media ▪ presence of other contaminants	▪ fouling of adsorption media ▪ flow rate
Membrane Filtration	Full scale	Yes	Liquid	Capital O&M	▪ molecular weight of contaminants	▪ type of filter ▪ pressure

					▪ temperature ▪ presence of other contaminants	▪ temperature
Bioremediation	Pilot scale	Yes	Solid, Liquid	Capital	▪ pH of media ▪ presence of other contaminants	▪ available nutrients ▪ temperature

6 REFERENCES

AMAP/UNEP (2008). Technical Background Report to the Global Atmospheric Mercury Assessment, Arctic Monitoring and Assessment Programme/ UNEP Chemicals Branch, 159 pp.

Australian Bureau of Statistics (ABS) 2006. Environmental Isuues-People's views and Practices. Accessed at http://www.ausstats.abs.gov.au/ausstats/subscriber.nsf/0/C5C112B3DD1BCC72CA25722C00745DCD/$File/46020_mar%202006.pdf.

Barr Engineering (2001). Substance Flow Analysis of Mercury in Products. Prepared for Minnesota Pollution Control Agency

Minneapolis, Barr Engineering Company, 118 pp.

Bergan, T., Gallardo, L., Rodhe, H., 1999. Mercury in the global troposphere: a three-dimensional model study. Atmospheric Environment 33, 1575-1585.

Bergan, T., Rodhe, H., 2001. Oxidation of elemental mercury in the atmosphere: Constraints imposed by global scale modelling. Atmospheric Chemistry 40, 191-212.

Brosset, C., Lord, E., 1991. Mercury in precipitation and ambient air - a new scenario. Water Air and Soil Pollution 56, 493-506.

Capri, A., Lindberg, S. E., 1998. Application of a Teflon dynamic flux chamber for quantifying soil mercury fluxes: Tests and results over back-ground soils. Atmospheric Environment 32, 873-882.

Carr, G. R., Wilmshurst, J. R., Ryall, W. R., 1986. Evaluation of mercury pathfinder techniques: Base metal and Uranium deposits. J. of Geochemical Exploration 26, 1-117.

Chang, R., Dombrowski, K., Senior, C. 2008. Near and Long Term Options for Controlling Mercury Emissions from Power Plants. Power Plant Air Pollutant "Mega" Symposium, NETL, EPRI, EPA.

Christensen, C., Skarup, S., Maag, J., Jensen, S. H. (2004). Mass flow analyses of mercury 2001, Danish Minsitry of the Environment, 103 pp.

Conaway, C. H., Mason, R. P., Steding, D. J., Flegal, A. R., 2005. Estimate of mercury emission from gasoline and diesel fuel consumption, San Francisco Bay area, California. Atmospheric Environment 39, 101.

Concorde 2007. Mercury in dental use: Environmental Implications for the European Union. Accessed at http://www.zeromercury.org/EU_developments/Maxson%20Dental%2014May2007%20-%20A5colour.pdf.

Cope, M. E., Hibberd, M. F., Lee, S., Malfroy, H. R., McGregor, J. R., Meyer, C. P., Morrison, A. L., Nelson, P. F. (2009). The Transportation and Fate of Mercury in Australia: Atmospheric Transport Modelling and Dispersion. Appendix 1 to Report RFT 100/0607 to Department of Environment, Water, Heritage & the Arts, The Centre for Australian Weather and Climate Research, 60pp.

Cossa, D., Conquery, M., Gobeil, C., Martin, J. M. 1996. Mercury fluxes at the ocean margins. In Mercury as a global pollutant : Towards integration and synthesis. H. J. W. Boca Raton, Lewis Publisher: 229-247.

DEFRA (2003). Mercury emissions from crematoria. Consultation on an assessment by the Environment Agency's Local Authority Unit, UK Department for Environment, Food and Rural Affairs, pp.

DEFRA (2004). Process Guidance Note 5/2(04) Secretary of State's Guidance for Crematoria, UK Department for Environment, Food and Rural Affairs, pp.

DEFRA 2005. Additional guidance from the Department for Environment, Food and Rural Affairs, and from the Welsh Assembly Government. . Accessed at <http://www.defra.gov.uk/environment/ppc/localauth/pubs/guidance/notes/aqnotes/aq01(05).htm>; accessed 2/1/2009.

DEH 2006. National Pollutant Inventory. Australia's national database of pollutant emissions, Australian Department of Environment and Heritage. Accessed 18th April 2006, at www.npi.gov.au.

Department of Environment and Climate Change NSW (DECC) (2007). Air Emissions Inventory for the Greater Metropolitan Region in NSW -Industrial Emission Model Results, pp.

Dutt, U., Nelson, P. F., Morrison, A. L., Strezov, V., 2009. Mercury wet deposition and coal-fired power station contributions: An Australian study. Fuel Processing Technology 90, 1354-1359.

Eltayeb, M. A. H., Van Grieken, R. E., Maenhau, W., Annergarn, H. J., 1993. Aerosol-soil fractionation for Namib Desert samples. Atmospheric Environment 27A, 669-678.

Engle, M. A., Gustin, M. S., Zhang, H., 2001. Quantifying natural source mercury emissions from the Ivanhoe Mining District, north-central Nevada, USA. Atmospheric Environment 35, 3987-3997.

EU 2004. Ambient air pollution by mercury (Hg). Position paper, European Communities. Accessed at http://europa.eu.int/comm/environment/air/pdf/pp_mercury.pdf#page=5.

EU (2007). Preliminary report on the environmental risks and indirect health effects of mercury in dental amalgam Scientific Committee on Health and Environmental Risks European Commission, pp.

EU (2008). Options for reducing mercury use in products and applications, and the fate of mercury already circulating in society. Final report, European Commissions Directorate- General Environment, pp.

Fang, F. M., Wang, Q. C., 2004. Mercury concentration, emission flux in urban land surface and its factors. Environmental Chemistry 23, 109-110.

Feeley, T., Brickett, L., Palko, B., Jones, A. 2008. DOE/NETL's Mercury Control Technology R&D Program - Taking Technology from Concept to Commecial Reality. Power Plant Air Pollutant "Mega" Symposium, NETL, EPRI, EPA.

Ferrara, R., Maserti, M., Anderson, M., Edner, H., Ragnarson, F., Svanberg, S., 1997. Mercury degassing rate from mineralised areas in the Mediterranean basin. Water Air & Soil Pollution 93, 59-66.

Ferrara, R., Mazzolai .B, Lanzillotta .E, Nucaro.E, Pirrone.N, 2000. Temporal trends in gaseous mercury evasion from the Mediterranean Sea waters. Science of Total Environment, 115-121.

Fitzgerald, W. F., Engstrom, D. R., Mason, R. P., Nater, E. A., 1998. The case for atmospheric mercury contamination in remote areas. Environmental Science & Technology 32, 1-7.

Fitzgerald, W. F., Mason, R. P., Vandal, G. M., 1991. Atmospheric cycle and air-water exchange of mercury over midcontinental lacustrine regions. Water, Air and Soil Pollution 56, 745-767.

Friedli, H. R., Arellano, A. F., Cinnirella, S., Pirrone, N., 2009. Initial estimates of mercury emissions to the atmosphere from global biomass burning. Environ. Sci. Technol. Published on the web: 15 April 2009.

Gabriel, M. C., Williamson, D. G., Brooks, S., Lindberg, S., 2005a. Atmospheric speciation of Southeastern mercury in two contrasting US airsheds. Atmospheric Environment 39, 4947-4958.

Gabriel, M. C., Williamson, D. G., Brooks, S., Zhang, H., Lindberg, S., 2005b. Spatial variability of mercury emissions from soils in a southeastern US urban environment. Environmental Geology 48, 955-964.

Gardfeldt, K., Feng, X., Sommar, J., Lindqvist, O., 2001. Total gaseous mercury exchange between air and water at river and sea surfaces in Swedish coastal regions. Atmospheric Environment 35, 3027-3038.

Gardfeldt, K., Sommar, J., Ferrara, R., Ceccarini, C., Lanzillotta, E., Munthe, J., Wangberg, I., Lindqvist, O., Pirrone, N., Sprovieri, F., Pesenti, E., Stromberg, D., 2003. Evasion of mercury from coastal and open waters of the Atlantic Ocean and the Mediterranean Sea. Atmospheric Environment 37, S73-S84.

Gbor, P., Wen, D., Meng, F., Yang, F., Zhang, B., Sloan, J., 2006. Improved model for mercury emission, transport and deposition. Atmospheric Environment 40, 973-983.

Greer, L., Bender, M., Maxson, P., Lennett, D. 2006. Curtailing Mercury's Global Reach. In State of the World 2006. A Worldwatch Institute Report on Progress Toward a Sustainable Society. L. Starke. New York, W.W. Norton & Company**:** 96-114.

Grigal, D. F., 2002. Inputs and outputs of mercury from terrestrial watersheds. Environ. Review 10, 1-39.

Gustin, M. S., Coolbaugh, M., Engle, M. A. 2002. Emission of mercury to the atmosphere from natural sources in Nevada and California. 18 Nov 2008.

Hanson, P. J., Lindberg, S. E., Tabberer, T. A., Owens, J. G., Kim, K. H., 1995. Foliar exchange of mercury vapour: evidence for a compensation point. Water Air Soil Pollution 80, 373-382.

Hanson, P. J., Tabberer, T. A., Lindberg, S. E., 1997. Emissions of mercury vapour from tree bark. Atmospheric Environment 31, 777-780.

Hilkene, C., Friesen, K. (2005). Background Study on Increasing Recycling of End-of-Life Mercury-containing Lamps from Residential and Commercial Sources in Canada, pp.

Hou, Y. B., Feng, X. B., Wang, S. F., Qiu, G. L., 2005. Mercury flux at air-soil interface in Guiyang and its suburbs. Acta Pedologica 42, 52-58.

Jackson, T. A., 1997a. Long-range atmospheric transport of mercury to ecosystems, and the importance of anthropogenic emissions—a critical review and evaluation of the published evidence. Environmental Reviews 5, 99-120.

Jackson, T. A., 1997b. Long-range atmospheric transport of mercury to ecosystems, and the importance of anthropogenic emissions: a critical review and evaluation of the published evidence. Environmental Reviews 5, 99-120.

Jones, G., Miller, G. 2005. Mercury and modern gold mining in Nevada. Final report to US Environmental Protection Agency Region IX, University of Nevada. Accessed at http://www.chem.unep.ch/MERCURY/Sector-Specific-Information/Docs/Goldmining_Nevada.pdf.

Keeler, G., Glinsorn, G., Pirrone, N., 1995. Particulate Mercury In The Atmosphere - Its Significance, Transport, Transformation And Sources. Water Air And Soil Pollution 80, 159-168.

Kellerhals, M., Beauchamp, S., Belzer, W., Blanchard, P., Froude, F., Harvey, B., McDonald, K., Pilote, M., Poissant, L., Puckett, K., Schroeder, B., Steffen, A., Tordon, R., 2003. Temporal and spatial variability of total gaseous mercury in Canada: results from the Canadian Atmospheric Mercury Measurement Network (CAMNet). Atmospheric Environment 37, 1003-1011.

Kim, K. H., Lindberg, S. E., Meyers, T. P., 1995. Micrometeological measurement of mercury fluxes over background forest soils in eastern Tennessee. Atmospheric Environment 27, 267-282.

Kindbom, K., Munthe, J. (2007). Product-related emissions of Mercury to Air in the European Union, Swedish Environmental Research Institute (IVL), pp.

King, S., Hochbrum, S., Graham, J., Goldberg, T., Wienert, A., Wilcox, M., 2008. Reducing Mercury in the northeast United States. Air & Waste Management Association EM Magazine, May Issue.

Landis, M. S., Lewis, C. W., Stevens, R. K., Keeler, G. J., Dvonch, J. T., Tremblay, R. T., 2007. Ft. McHenry tunnel study: Source profiles and mercury emissions from diesel and gasoline powered vehicles. Atmospheric Environment 41, 8711-8724.

Lee, X., Benoit, G., Xu, X., 2000. Total gaseous mercury concentration and flux over a coastal saltmarsh vegetation in Connecticut USA. Atmospheric Environment 34, 4205-4213.

Leonard, T. L., Taylor, G. E., Gustin, M. S., Fernandez, G. C. J., 1998a. Mercury and plants in contaminated soils: 1. Uptake, partitioning, and emission to the atmosphere. Environmental Toxicology and Chemistry 17, 2063-2071.

Leonard, T. L., Taylor, G. E., Gustin, M. S., Fernandez, G. C. J., 1998b. Mercury and plants in contaminated soils: 2. Environmental and physiological factors governing mercury flux to the atmosphere. Environmental Toxicology and Chemistry 17, 2072-2079.

Lin, C. J., Lindberg, S. E., Ho, T. C., Jang, C., 2005. Development of a processor in BEIS3 for estimating vegetative mercury emission in the continental United States. Atmospheric Environment 39, 7529-7540.

Lin, C. J., Pehkonen, S. O., 1999a. Aqueous phase reactions of mercury with free radicals and chlorine: Implications for atmospheric mercury chemistry. Chemosphere 38, 1253-1263.

Lin, C. J., Pehkonen, S. O., 1999b. The chemistry of atmospheric mercury: a review. Atmospheric Environment 33, 2067-2079.

Lindberg, S., Bullock, R., Ebinghaus, R., Engstrom, D., Feng, X., Fitzgerald, W., Pirrone, N., Prestbo, E., Seigneur, C., 2007a. A synthesis of progress and uncertainties in attributing the sources of mercury in deposition. Ambio 36, 19.

Lindberg, S., Bullock, R., Ebinghaus, R., Engstrom, D., Feng, X. B., Fitzgerald, W., Pirrone, N., Prestbo, E., Seigneur, C., 2007b. A synthesis of progress and uncertainties in attributing the sources of mercury in deposition. Ambio 36, 19-32.

Lindberg, S. E., Hanson, P. J., Meyers, T., Kim, K. H., 1998a. Air/surface exchange of mercury vapour over forests-the need for a reassessment of continental biogenic emissions. Atmospheric Environment 32, 895-908.

Lindberg, S. E., Hanson, P. J., Meyers, T. P., Kim, K. H., 1998b. Air/surface exchange of mercury vapor over forests - The need for a reassessment of continental biogenic emissions. Atmospheric Environment 32, 895-908.

Lindberg, S. E., Meyers, T. P., 2001. Development of an automated micrometeorological method for measuring emission of mercury vapor from wetland vegetation. Wetlands Ecological Management 9, 333-347.

Lindberg, S. E., Meyers, T. P., Taylor, G. E., Schroeder, W. H., 1992. Atmosphere-surface exchange of mercury in a forest :results of modelling and gradient approaches. Geophysical Research 97, 2519-2528.

Lindberg, S. E., Mong, W., Meyers, T., 2002. Transpiration of gaseous elemental mercury through vegetation in a subtropical wetland in Florida. Atmospheric Environment 36, 5207-5219.

Lindberg, S. E., Stratton, W. J., 1998. Atmospheric mercury speciation: Concentrations and behavior of reactive gaseous mercury in ambient air. Environmental Science & Technology 32, 49-57.

Lindquist, O., Johansson, K., Aastrup, M., Andersson, A., Bringmark, L., Hovsenius, G., Hasansson, L., Iverfeldt, A., Meili, M., Timm, B., 1991. Water Air and Soil Pollution 55, 1-261.

Magg, J., Lassen, C., Hansen, E. 1996. Substance flow assessment for mercury, Danish Environment Protection Agency. Accessed at www.mst.dk.

Mason, R. 2008. Mercury emissions from natural processes and their importance in the global mercury cycle. In Mercury fate and transport in the global atmosphere: measurements, models and policy implications. Interim report of the UNEP Global Partnership on Atmospheric Mercury Transport and Fate Research. N. Pirrone and R. Mason, United Nations Environment Program**:** 130-144.

Mason, R. P., Fitzgerald, W. F., Morel, F. M. M., 1994a. The Biogeochemical Cycling of Elemental Mercury - Anthropogenic Influences. Geochimica Et Cosmochimica Acta 58, 3191-3198.

Mason, R. P., Fitzgerald, W. F., Morel, F. M. M., 1994b. The biogeochemical cycling of elemental mercury :Anthropogenic influences. Geochemica Cosmochimica Acta 58, 3191-3198.

Maxson, P. A. 2005. Global mercury production, use and trade. In Dynamics of mercury pollution on regional and global scales - atmospheric processes and human exposures around the world. N. Pirrone and K. R. Mahaffey. New York, Springer**:** 25-50.

McElroy, W. J., Munthe, J., 1991. The oxidation of mercury (I) by ozone in acidic aqueous solution. Acta Chemica Scandinavica 45, 254-257.

Meili, M., Bishop, K., Bringmark, L., Johansson, K., Munthe, J., Sverdrup, H., de Vries, W., 2003. Critical levels of atmospheric pollution: criteria and concepts for operational modelling of mercury in forest and lake ecosystems. The Science of The Total Environment 304, 83-106.

Mergler, D., Anderson, H. A., Chan, L. H. M., Mahaffey, K. R., Murray, M., Sakamoto, M., Stern, A. H., 2007. Methylmercury exposure and health effects in humans: A worldwide concern. Ambio 36, 3-11.

Munthe, J., Bodaly, R. A., Branfireun, B. A., Driscoll, C. T., Gilmour, C. C., Harris, R., Horvat, M., Lucotte, M., Malm, O., 2007. Recovery of mercury-contaminated fisheries. Ambio 36, 33-44.

NE States (2007). Northeast succeed in reducing mercury in the environment, NE Interstate Water Control Commission, NE States for Coordinated Air Use management and NE Waste Management Officials Association, pp.

Nelson, P. F., 2007. Atmospheric emissions of mercury from Australian point sources. Atmospheric Environment 41, 1717-1724.

Nelson, P. F., Dutt, U., Morrison, A. L., Strezov, V., 2009. Atmospheric gas phase mercury in Sydney Australia. unpublished data, in preparation for publication.

Nelson, P. F., Petersen, C., Morrison, A. L., 2004. Atmospheric emissions of mercury - sources and chemistry. Clean Air and Environmental Quality 38, 48-55.

New Jersey Department of Environment Protection (NJDEP) 2002. New Jersey mercury Task Force Volume III Chapter 3 Sources of Mercury in New Jersey. Accessed 7 Sep 2008, at http://www.state.nj.us/dep/dsr/Vol3-chapter3.pdf.

NHMRC (1999). Dental amalgam and mercury in dentistry, Commonwealth of Australia, 93pp.

Northeast Waste Management Officials Association (NEWMOA))/ IMERC (2008). Trends in mercury use in products: Summary of the Interstate Mercury Education and Reduction Clearing house (IMERC) Mercury-added Products database., pp.

NPI 2005. Emission Estimation Technique Manual for Fossil Fuel Electric Power Generation Version 2.4, Department of Environment and Heritage. Accessed 1st May 2006, at http://www.npi.gov.au/handbooks/approved_handbooks/pubs/elec-supply.pdf.

NSW Department of Energy Utilities and Sustainability (2005). NSW Liquid Transfer Waste Management Guidelines, pp.

Packham, D., Tapper, N., Griepsma, D., Friedli, H., Hellings, J., Harris, S., 2009. Release of mercury in the Australian environment by burning: A preliminary investigation of biomatter and soils. Air Quality and Climate Change 43, 24-27.

Pacyna, E. G., Pacyna, J. M., 2002. Global emission of mercury from anthropogenic sources in 1995. Water Air and Soil Pollution 137, 149-165.

Pacyna, E. G., Pacyna, J. M., Steenhuisen, F., Wilson, S., 2006. Global anthropogenic mercury emission inventory for 2000. Atmospheric Environment 40, 4048-4063.

Pacyna, J. M., Pacyna, E. G., Steenhuisen, F., Wilson, S., 2003. Mapping 1995 global anthropogenic emissions of mercury. Atmospheric Environment 37, S109-S117.

Pai, P., Karamchandani, P., Seigneur, C., 1997. Simulation of the regional atmospheric transport and fate of mercury using a comprehensive Eulerian model. Atmospheric Environment 31, 2717-2732.

Parsons, D., 2006. The environmental impact of compact fluorescent lamps and incandescent lamps for Australian conditions. The Environmental Engineer 7, 8-14.

Pavlish, J. H., Sondreal, E. A., Mann, M. D., Olson, E. S., Galbreath, K. C., Laudal, D. L., Benson, S. A., 2003. Status review of mercury control options for coal-fired power plants. Fuel Processing Technology 82, 89-165.

Petersen, G., Bloxam, R., Wong, S., Munthe, J., Kruger, O., Schmolke, S. R., Kumar, A. V., 2001. A comprehensive Eulerian modelling framework for airborne mercury species: model development and applications in Europe. 35, 3063-3074.

Petersen, G., Iverfeldt, A., Munthe, J., 1995. Atmospheric Mercury Species over Central and Northern Europe - Model-Calculations and Comparison with Observations from the Nordic Air and Precipitation Network for 1987 and 1988. Atmospheric Environment 29, 47-67.

Petersen, G., Munthe, J., Pleijel, K., Bloxam, R., Kumar, A. V., 1998. A comprehensive Eulerian modeling framework for airborne mercury species: Development and testing of the Tropospheric Chemistry Module (TCM). Atmospheric Environment 32, 829-843.

Peterson, C., Nelson, P., Morrison, A. (2004). Quantifying Natural and Anthropogenic Sourced Mercury Emissions from Australia in 2001 – A local scale modeling assessment of transport and deposition patterns for anthropogenic mercury emissions., CRC for Coal in Sustainable Development, 119pp.

Pirrone, N., Costa, P., Pacyna, J. M., Ferrara, R., 2001. Mercury emissions to the atmosphere from natural and anthropogenic sources in the Mediterranean region. Atmospheric Environment 35, 2997-3006.

Pirrone, N., Hedgecock, I. M., Forlano, L., 2000. Role of the ambient aerosol in the atmospheric processing of semivolatile contaminants: A parameterized numerical model (Gas-Particle Partitioning (GASPAR)). Journal Of Geophysical Research-Atmospheres 105, 9773-9790.

Pirrone, N., Keeler, G. J., Nriagu, J. O., 1996. Regional differences in worldwide emissions of mercury to the atmosphere. Atmospheric Environment 30, 2981-2987.

Pirrone, N., Mahaffey, K. R., Eds. (2005a). Dynamics of mercury pollution on regional and global scales: atmospheric processes and human exposures around the world, Springer.

Pirrone, N., Mahaffey, K. R. 2005b. Where we stand on mercury pollution and its health effects on regional and global scales. In Dynamics of mercury pollution on regional and global scales: atmospheric processes and human exposures around the world. N. Pirrone and K. R. Mahaffey, Springer**:** 1-24.

Pirrone, N., Mason, R., Eds. (2008). Mercury Fate and Transport in the Global Atmosphere: Measurements, Models and Policy Implications. Interim Report of the UNEP Global Partnership on Atmospheric Transport and Fate Research, UNEP.

Pleijel, K., Munthe, J., 1995a. Modeling The Atmospheric Chemistry Of Mercury - The Importance Of A Detailed Description Of The Chemistry Of Cloud-Water. Water Air And Soil Pollution 80, 317-324.

Pleijel, K., Munthe, J., 1995b. Modeling The Atmospheric Mercury Cycle - Chemistry In Fog Droplets. Atmospheric Environment 29, 1441-1457.

Poissant, L., Casimir, A., 1998. Water-air and soil-air exchange rate of total gaseous mercury measured at background sites. Atmospheric Environment 32, 883-893.

Poissant, L., Pilote, M., Constant, C., Beauvais, H., Zhang, H., Xu, X., 2004. Atmospheric mercury speciation and deposition in the Bay St Francois wetlands. Geophysical Research 109, D11301.

Porcella, D. B., Chu, P., Alla, M. A. 1996. Inventory of North America Hg Emissions to the Atmosphere; Relationship to the global Hg cycle. In Global and Regional Mercury Cycles: Sources, Fluxes, and mass Balances. W. Baeyens, R. Ebinghaus and O. Vasiliev, NATO ASI Series, 2. Environment – Vol. 21**:** 179-190.

Rasmussen, P. E., Kemp, R. J., Fitzgerald-Hubble, D. R., Schroeder, W. H. 1998. Towards an improved natural sources inventory for mercury. In Proceedings of Metals and the Environment;an international symposium,

Richardson, G. M., Mitchell, I. A., Mah-Paulson, M., Hackbarth, T., Garrett, R. G., 2003. Natural emissions of mercury to the atmosphere in Canada. Environmental Review 11, 17-36.

Ross and Associates (2000). Mercury Reduction Options: Great Lakes Binational Toxics Strategy, Draft report for U.S. EPA Great Lakes National Program Office, pp.

Scheulhammer, A. M., Meyer, M. W., Sandheinrich, M. B., Murray, M. W., 2007. Effects of environmental methylmercury on the health of wild birds, mammals, and fish. Ambio 36, 12-18.

Schroeder, K., Cardone, E., White, F., Rohar, P., Kim, A. (2007). Long-Term Column Leaching of Phase II Mercury Control Technology By-products, US Department of Energy National Energy Technology Laboratory pp.

Schroeder, W., Munthe, J., 1998a. Atmospheric mercury -an overview. Atmospheric Environment 32, 809-822.

Schroeder, W. H., Beauchamp, S., Edwards, G., Poissant, L., Rasmussen, P., Tordon, R., 2005. Gaseous mercury emissions from natural sources in Canadian landscapes. Geophysical Research 110, D18302.

Schroeder, W. H., Munthe, J., 1998b. Atmospheric mercury - An overview. Atmospheric Environment 32, 809-822.

Schroeder, W. H., Yarwood, G., Niki, H., 1991. Transformation processes involving Hg species in the atmosphere - results from a literature survey. Water Air and Soil Pollution 56, 653-666.

Seigneur, C., Abeck, H., Chia, G., Reinhard, M., Bloom, N. S., Prestbo, E., Saxena, P., 1998. Mercury adsorption to elemental carbon (soot) particles and atmospheric particulate matter. Atmospheric Environment 32, 2649-2657.

Seigneur, C., Wrobel, J., Constantinou, E., 1994. A chemical kinetic mechanism for atmospheric inorganic mercury. Environmental Science & Technology 28, 1589-1597.

Shetty, S., Lin, C. J., Streets, D., Jang, C., Ho, T., Chu, H., Vanjani, T. 2007. Estimate of mercury from natural sources in East Asia. CMAS conference. Chapel Hill.

Shetty, S. K., Lin, C.-J., Streets, D. G., Jang, C., 2008. Model estimate of mercury emission from natural sources in East Asia. Atmospheric Environment 42, 8674.

Skarup, S., Christensen, C., Maag, J., Jensen, S. 2003. Substance flow analysis for mercury . Environmental project no 808. The Danish EPA. Accessed at http://www2.mst.dk/Udgiv/publications/2006/87-7052-133-6/pdf/87-7052-134-4.pdf.

Slemr, F., Schuster, G., Seiler, W., 1985. Distribution, speciation and budget of atmospheric mercury. Journal of Atmospheric Chemistry 3, 407-434.

Srivastava, R. K., Staudt, J. E., Jozewicz, W., 2005. Preliminary estimates of performance and cost of mercury emission control technology applications on electric utility boilers: An update. Environmental Progress 24, 198-213.

Stamenkovic, J., Lyman, S., Gustin, M. S., 2007. Seasonal and diel variation of atmospheric mercury concentrations in the Reno (Nevada, USA) airshed. Atmospheric Environment 41, 6662-6672.

Swain, E. B., Jakus, P. M., Rice, G., Lupi, F., Maxson, P. A., Pacyna, J. M., Penn, A., Spiegel, S. J., Veiga, M. M., 2007. Socioeconomic consequences of mercury use and pollution. Ambio 36, 45-61.

Travnikov, O., Ryaboshapko, A. (2002). Modelling of mercury hemispheric transport and depositions. Moscow, Meteorological Synthesising Centre-East, pp.

UNEP 2002. Global Mercury Assessment Report, UNEP Chemicals. Accessed at http://www.chem.unep.ch/mercury/Report/Final%20report/final-assessment-report-25nov02.pdf.

UNEP 2005. Toolkit for identification and quantification of mercury releases. Accessed at http://www.chem.unep.ch/mercury/Toolkit/UNEP-final-pilot-draft-toolkit-Dec05.pdf.

UNEP 2007. Decisions adopted by the Governing Council/Global Ministerial Environment Forum at its twenty-fourth session, United Nations Environment Program. Accessed at http://www.unep.org/gc/gc24/docs/GC24_decisions.pdf; accessed 6th August 2007.

UNEP Chemicals Branch (2008). The Global Atmospheric Mercury Assessment: Sources, Emissions and Transport. Geneva, Switzerland, UNEP-Chemicals, 42pp.

US DoE (2005). Analysis of Alternative Mercury Control Strategies Energy Information Administration Office of Integrated Analysis and Forecasting, US Department of Energy, pp.

USEPA 1997a. Mercury Study Report to Congress. Accessed at http://www.epa.gov/mercury/report.htm.

USEPA (1997b). Mercury Study Report to Congress. Fate and transport of mercury in the Environment, vol. III. Washington, DC, US Environmental Protection Agency, 376pp.

USEPA (1997c). Mercury Study to Congress Volume VIII An Evaluation of Mercury Control Technologies and Costs, USEPA Office of Air Quality Planning and Standards and Office of Research and Development, pp.

USEPA (2003). Performance and Cost of Mercury and Multipollutant Emission Control Technology Applications on Electric Utility Boilers, pp.

USEPA (2005). Control of Mercury Emissions from Coal Fired Electric Utility Boilers: An Update, Air Pollution Prevention and Control Division, National Risk Management Laboratory, Office of Research and Development, pp.

USEPA (2007). Treatment Technologies for Mercury in Soil, Waste, and Water, USEPA Office of Superfund Remediation and Technology Innovation, pp.

USEPA (2008). Health Services Industry Detailed Study Dental Amalgam, pp.

Veiga, M. M., Maxson, P. A., Hylander, L. D., 2006. Origin and consumption of mercury in small-scale gold mining. Journal Of Cleaner Production 14, 436-447.

Wang, D. Y., He, L., Shi, X. J., Wei, S. Q., Feng, X. B., 2006a. Release flux of mercury from different environmental surfaces in Chongqing, China. Chemosphere 64, 1845-1854.

Wang, D. Y., He, L., Wei, S. Q., Feng, X. B., 2006b. Estimation of mercury emission from different sources to atmosphere in Chongqing, China. Science of the Total Environment 366, 722-728.

Wangberg, I., Munthe, J., Pirrone, N., Iverfeldt, A., Bahlman, E., Costa, P., Ebinghaus, R., Feng, X., Ferrara, R., Gardfeldt, K., Kock, H., Lanzillotta, E., Mamane, Y., Mas, F., Melamed, E., Osnat, Y., Prestbo, E., Sommar, J., Schmolke, S., Spain, G., Sprovieri, F., Tuncel, G., 2001. Atmospheric mercury distribution in Northern Europe and in the Mediterranean region. Atmospheric Environment 35, 3019-3025.

WHO (1990). Methyl Mercury. Geneva, World Health Organisation, pp.

WHO (1991). Inorganic Mercury. Geneva, World Health Organisation, pp.

Wilhelm, S. M., 2001. Estimate of mercury emissions to the atmosphere from petroleum. Environmental Science & Technology 35, 4704-4710.

Wilhelm, S. M., Bloom, N., 2000. Mercury in petroleum. Fuel processing technology 63, 1.

Wilson, S. J., Steenhuisen, F., Pacyna, J. M., Pacyna, E. G., 2006. Mapping the spatial distribution of global anthropogenic mercury atmospheric emission inventories. Atmospheric Environment 40, 4621-4632.

Won, J. H., Park, J. Y., Lee, T. G., 2007. Mercury emissions from automobiles using gasoline, diesel, and LPG. Atmospheric Environment 41, 7547.

Xiao, Z. F., Munthe, J., Schroeder, W. H., Lindqvist, O., 1991. Vertical fluxes of volatile mercury over forest soil and lake surfaces in Sweden. Tellus Series B-Chemical and Physical Meteorology 43, 267-279.

Xu, X., Yang, X., Miller, D., Helble, J., Carley, R., 1999. Formulation of bi-directional atmospheric surface exchanges of elemental mercury. Atmospheric Environment 33, 4345-4355.

Xu, X. H., Yang, X. S., Miller, D. R., Helble, J. J., Carley, R. J., 2000a. A regional scale modeling study of atmospheric transport and transformation of mercury. I. Model development and evaluation. Atmospheric Environment 34, 4933-4944.

Xu, X. H., Yang, X. S., Miller, D. R., Helble, J. J., Carley, R. J., 2000b. A regional scale modeling study of atmospheric transport and transformation of mercury. II. Simulation results for the northeast United States. Atmospheric Environment 34, 4945-4955.

Zhang, H., Lindberg, S. E., Marsik, F. J., Keeler, G. J., 2001. Mercury air/surface exchange kinetics of background soils of the Tahquamenon River watershed in the Michigan Upper peninsula. Water Air & Soil Pollution 126, 151-169.

Appendix 1

Mercury Sources, Transportation and Fate in Australia

Final Report to the Department of Environment, Water, Heritage & the Arts
RFT 100/0607

The Transportation and Fate of Mercury in Australia: Atmospheric Transport Modelling and Dispersion.

Cite as:

Cope, M. E., Hibberd, M. F., Lee, S., Malfroy, H. R., McGregor, J. R., Meyer, C. P., Morrison, A. L., Nelson, P. F. (2009). *The Transportation and Fate of Mercury in Australia: Atmospheric Transport Modelling and Dispersion*. Appendix 1 to Report RFT 100/0607 to Department of Environment, Water, Heritage & the Arts, The Centre for Australian Weather and Climate Research, 60 pp.

Australian Government
Bureau of Meteorology

The Centre for Australian Weather and Climate Research
A partnership between CSIRO and the Bureau of Meteorology

CSIRO

The Transportation and Fate of Mercury in Australia. Atmospheric transport and deposition modelling. Draft Final Report.

Martin Cope[1], Mark Hibberd[1], Sunhee Lee[1], Hugh Malfroy[3], John McGregor[1], Mick Meyer[1], Anthony Morrison[2], Peter Nelson[2]

4 June 2009

[1]Centre for Australian Weather and Climate Research
[2]Macquarie University
[3]Malfroy Environmental Strategies.

www.cawcr.gov.au

Enquiries should be addressed to:
Dr. Martin Cope
martin.cope@csiro.au

Copyright and Disclaimer

Important Disclaimer

Contents

List of Figures

List of Tables

1. INTRODUCTION

This report documents work undertaken toward the atmospheric transport modelling task of the project “Sources, Transport and Fate of Mercury in Australia”. This task entails the use of numerical meteorological and transport models and an air emissions inventory for mercury to generate best estimates of annual average ambient mercury concentrations and wet and dry deposition mass. The modelling has been undertaken over three spatial scales- for the Australian continent; for the urban regions of Melbourne and Sydney; and for five significant point source emitter groups. This information will form one input into a broader environmental assessment of mercury impacts in Australia which is being undertaken by Macquarie University.

Section 2 of the report provides a brief overview of the modelling system and includes an introduction to the weather model, the anthropogenic and natural air emission inventories, and to the transport models. Sections 3, 4 and 5 present the concentration and deposition results from the continental, urban and local scale, respectively.

2. METHODOLOGY

The continental-scale transport atmospheric modelling uses the coupled system shown in Figure 1. The system comprises a model for simulating weather (summarised in section 2.1), an inventory of mercury emissions from anthropogenic and natural sources (summarised in sections 2.2.1–2.2.3), and a model for simulating the atmospheric transport and its subsequent fate via wet and dry deposition (Section 2.3). The modelling includes global background concentrations of mercury which are advected into the Australian region by the prevailing winds (section 2.3.2).

The transport modelling has been undertaken for gaseous elemental mercury (Hg0), reactive gaseous mercury (RGM) and particulate mercury (Hgp). Although the oxidation–reduction chemistry of mercury has not been modelled (out of scope for this project), account has been taken of the different solubility properties of each mercury species and the subsequent differing rates of deposition. The continental transport modelling system has been used to generate ambient concentrations and deposition patterns of mercury for the Australian continent at a grid spacing of 0.25º (~25 km) in the horizontal for the year 2006. This year was selected for study because it includes a number of significant bushfires in the southern region of Australia, and thus may provide a worst case estimate of the contribution of mercury emissions from this source. The urban scale modelling for Melbourne and Sydney was undertaken with a grid spacing of 3 km, and the near-source modelling at five sites with a grid spacing of 1 km.

A brief description of the each component of the atmospheric modelling system is now given.

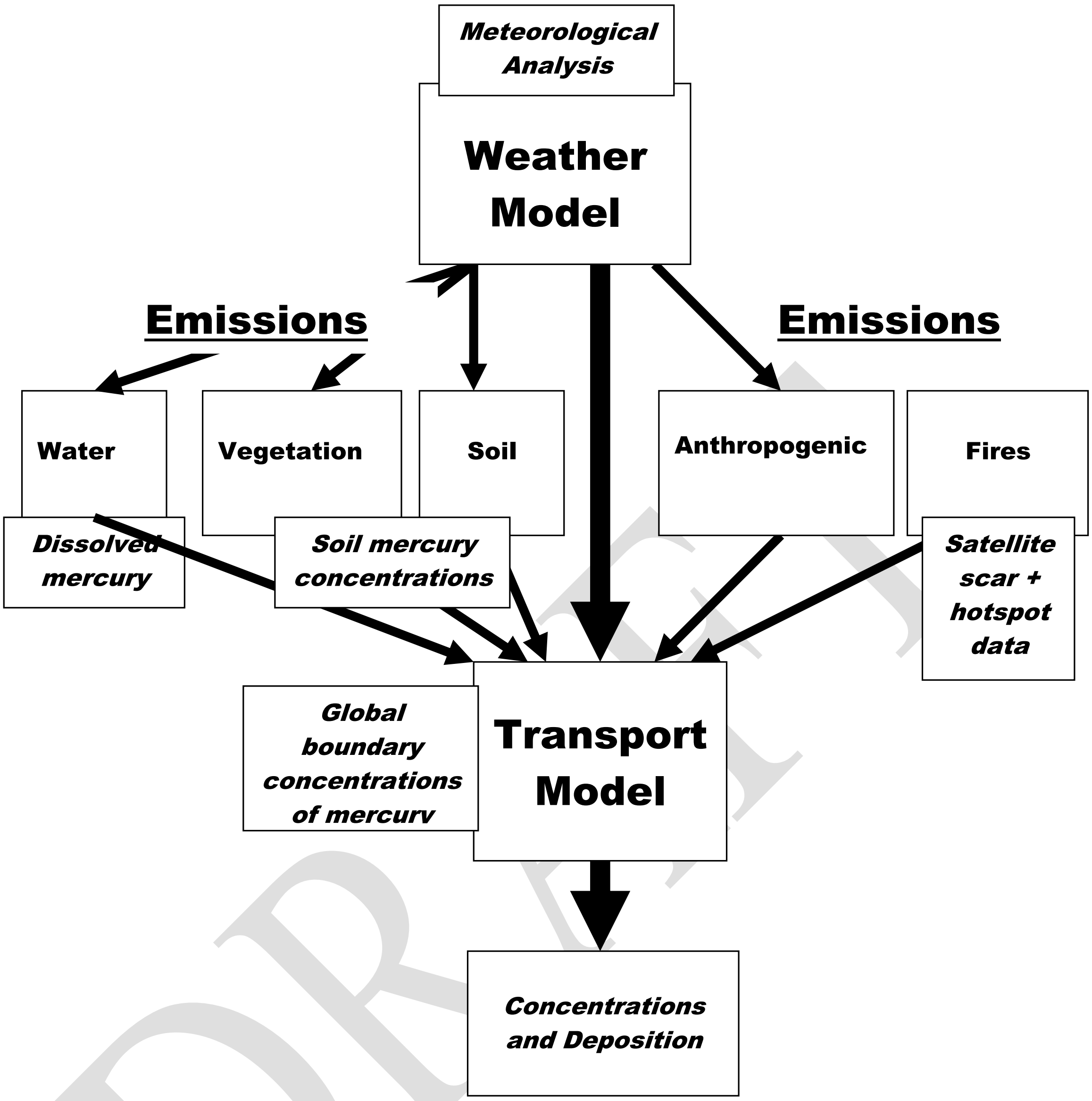


Figure 1. Schematic diagram showing the continental-scale atmospheric mercury transport modelling system.

2.1 Meteorological modelling

2.1.1 Continental Scale

Weather conditions for the continental-scale modelling have been generated by the CSIRO Cubic Conformal Atmospheric Model (CCAM, McGregor 2005; McGregor and Dix 2001). This model has been used extensively for weather-related studies in Australia including a recently completed study which investigates the impact of climate on air quality (Cope et al. 2009). In this study the modelled weather and air pollution fields for the decade 1996–2005 were compared with observations and CCAM was seen to perform well. In the current study the model has been used to generate pressure, wind, turbulence, temperature, humidity, cloud and rainfall, and evapotranspiration fields for the Australian region for 2006. Figure 1 shows that these meteorological data are used by the modelling system to drive the emission and transport simulations. For example the emissions of mercury from water are a function of the aqueous mercury concentrations and near-surface wind speed (with the latter provided by the meteorological model). Emissions of mercury from vegetation are proportional to the latent heat flux (and thus the water flux) via the transpiration of water from the stomata in leaves. Emissions from soil are a function of soil temperature; the height of emissions from buoyant stack plumes are a function of wind speed and vertical temperature gradient (in addition to the plume characteristics); and the diurnal variation of emissions from fires is a function of the wind speed, temperature and relative humidity. Additionally, the horizontal and vertical transport of mercury is linked to the three-dimensional wind and turbulence fields, both of which are generated by the weather model.

Figure 2 shows the stretched, global computational grid used by CCAM for the current study. Note that CCAM models the weather of the entire world, but uses a stretched grid to provide the greatest density of grid points over the Australian region where the weather conditions have to be resolved at a resolution suitable for the mercury emission and transport modelling.

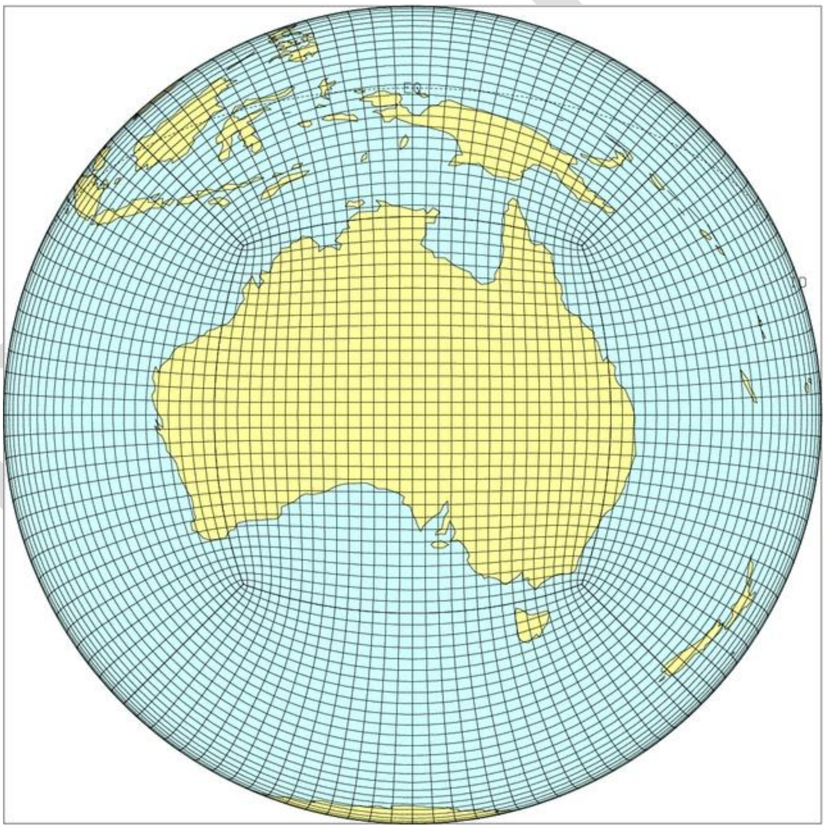


Figure 2. Computation grid domain used by CCAM for the 2006 continental-scale mercury transport modelling.

The CCAM grid is divided into 18 layers in the vertical, with the lowest layer centred on 38 m above ground level. Six layers are located below 2 km in order to resolve low level vertical wind shear and temperature inversions. The former is important because mercury may be emitted from tall chimneys in buoyant plumes that rise a considerable distance in the atmosphere. Vertical wind shear can cause these plumes to be transported in directions that are quite different from the surface wind direction. Good resolution of the height of temperature inversions is also important because it dictates the height over which surface emissions of mercury are diluted by atmospheric turbulence and hence the subsequent concentrations.

CCAM includes a vegetation canopy scheme (Kowalczyk et al. 1994) which is used to model the exchange of heat, moisture, momentum and gases between the atmosphere and plants and soil. Outputs from this scheme have been combined with soil mercury concentrations and used to generate hourly varying mercury emissions from vegetation and soils (section 2.2.2). CCAM also models the generation of convective and large scale cloud systems and the associated rain. Cloud characteristics and precipitation rates are used by the transport model (see section 2.3) to calculate the wet-deposition rates for mercury.

The accuracy of the simulated meteorological fields is optimised by nudging the CCAM weather towards the large-scale features of an observation-based weather analysis that is updated at six hourly intervals.

2.1.2 Urban and Near-Source Modelling

The urban and near-source modelling was undertaken using TAPM version 4 (Hurley 2008). The meteorological component of TAPM uses the large-scale synoptic analyses from the GASP (Global Analysis and Prediction) analyses at a horizontal grid spacing of 1° longitude × 1° latitude (about 100 km × 100 km) at 6-hourly intervals, as input boundary conditions for the model outer grid. TAPM uses nested grids to model local scales at a finer resolution. Other inputs to the meteorological component of the model include databases of terrain height, land use, soil type, vegetation, and leaf-area-index.

The urban environments were modelled using three nested grids, the outer two at grid resolutions of 20 km and 8 km, and the inner grid at 3 km grid resolution, for:

- Sydney, domain of inner grid 180 km (east-west) x 210 km (north-south)
- Melbourne, domain of inner grid 210 km (east-west) x 180 km (north-south).

Near-source modelling was undertaken for five sites with an inner grid resolution of 1 km on domains of 50 km x 50 km for:

- Kalgoorlie (WA) – centred on 30° 45′ S, 121° 28.5′ E
- Pinjarra (WA) – centred on 32° 38′ S, 115° 54′ E
- Mt Isa (QLD) – centred on 20° 44′ S, 139° 29′ E
- NSW Central Coast (NSW) – centred on 33° 8.5′ S, 151° 32′ E
- Latrobe Valley (VIC) – centred on 38° 14′ S, 146° 29′ E.

2.2 Emissions

Mercury emissions have been estimated for anthropogenic emissions (industrial, commercial-domestic and motor vehicle sources), soils, vegetation, water surfaces and fires.

2.2.1 Anthropogenic Emissions

A discussion of the methodology used to generate annual emission totals for mercury from anthropogenic sources is given by Nelson et al. (2008). These sources were provided to the transport modelling team in three categories (Table 1):

1. Large (> 10 kg yr^{-1}) industrial sources where the characteristics of the emission sources required for calculating plume rise (stack height, stack diameter, plume velocity and temperature) were provided from measurements or engineering estimates. This category includes large metal ore processing facilities, large coal-fired power stations as well as other major sources. These sources are treated as explicit point sources and plume rise is calculated for each hour of a simulation by the transport model using meteorological data provided by the meteorological models (Section 2.1).
2. Small (0.1–10 kg yr^{-1}) industrial sources. These sources are given nominal emission release characteristics. A few are ground-level sources but most are stacks and these have been given a nominal release height of 25 m, stack diameter of 1 m, efflux velocity of 10 m s^{-1}, and efflux temperature of 330 K.
3. Commercial-domestic sources (such as crematoria; which are not captured by the source groups described above) and other distributed sources (such as motor vehicles). The emissions from these source groups are spatially distributed using a gridded population database.

Figure 3 shows the spatial distribution of the commercial-domestic sources of anthropogenic mercury emissions, and Figure 4 shows the point source emissions with the major sources highlighted (>100 kg yr^{-1} emission rates). The inventory team also provided speciation factors to split the total mercury mass (kg yr^{-1}) into Hg0, RGM and Hgp. Anthropogenic emissions are assumed to be time invariant for all sources except for motor vehicles and domestic oil combustion for heating, cremations and dental amalgam. These sources were allocated temporal profiles taken from the Sydney air emissions inventory (DECC 2007a, b). Table 1 shows the annual emissions of the anthropogenic and natural sources (the latter will be discussed in the next section). It can be seen that the industrial source group comprises 93% of the anthropogenic emissions. Total mercury emissions from the industrial source group have an average (mass–weighted) percentage breakdown of 77%, 17% and 6% into the Hg0, RGM and Hgp species. The commercial–domestic and diffusive sources have a percentage breakdown of 80%; 10% and 10% into Hg0, RGM and Hgp species.

Table 1. Annual emissions of mercury from anthropogenic and natural sources in Australia. Ocean fluxes have not been added to the emission totals because they depends on the model domain size. Hg0 - gaseous elemental mercury; RGM - reactive gaseous mercury; Hgp - particulate mercury.

	Hg0	**RGM**	**Hgp**	**Total Hg**	
	(tonnes)	(tonnes)	(tonnes)	(tonnes)	(%)
	Anthropogenic				
Industrial	10.7	2.4	0.9	14.0	6.8
Commercial, Domestic + diffuse	0.8	0.1	0.1	1.0	0.4
	Natural				
Vegetation	7.9	0	0	7.9	3.9
Canopy-soil	54.2			54.2	26.5
Bare soil	86.0	0	0	86.0	42.0
Fires	33.4	2.9	5.4	41.8	20.4
Ocean	598.7	0	0	598.7	n/a

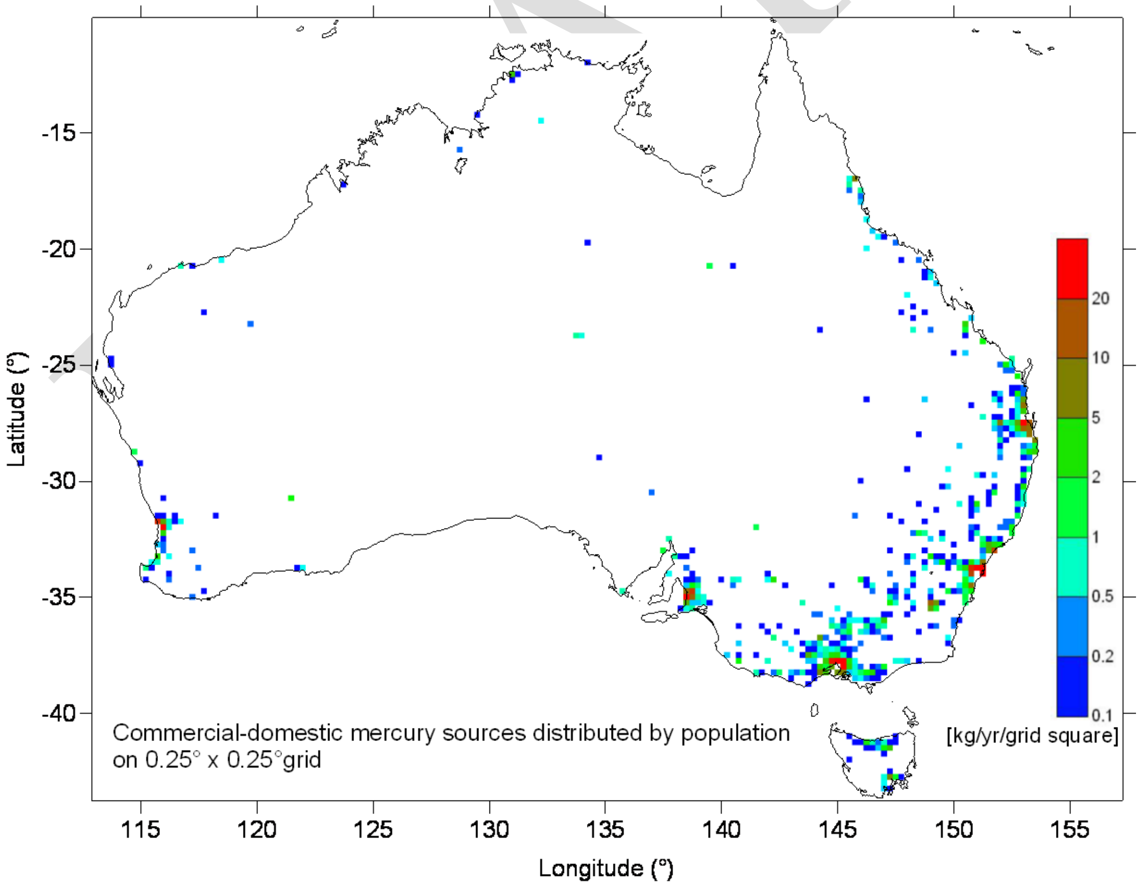


Figure 3. The spatial distribution of anthropogenic commercial-domestic mercury emissions. The locations of grid squares (0.25° x 0.25°) where total Hg > 0.1 kg yr^{-1} are shown.

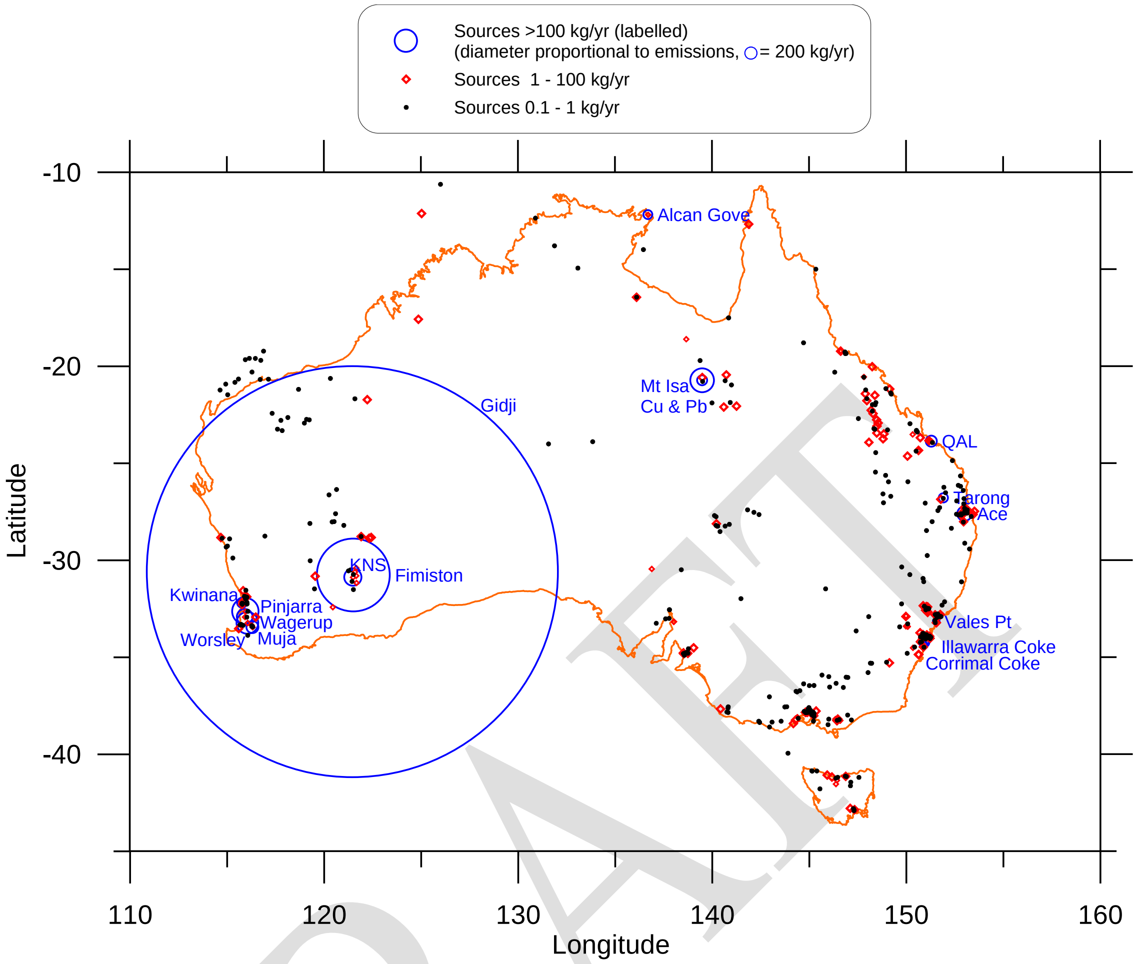


Figure 4. The spatial distribution of anthropogenic point source emissions as a bubble plot. The location of individual sources where total Hg > 0.1 kg yr^{-1} are shown. The largest sources (> 100 kg yr^{-1}) are labelled and shown as a bubble plot with the diameter proportional to the source strength.

2.2.2 Natural Emissions- vegetation, soil and water

Natural emissions for the Australia region have been modelled using the approach outlined in Shetty et al. (2008) and references therein. Emissions from vegetation are assumed to be caused by the uptake of mercury in the soil-water via the porous plant root system. The plant vascular system then transports the mercury into the canopy atmosphere within water vapour released via stomata in the leaves (evapotranspiration). The resistance to vapour transport through the stomata varies with radiation, temperature, ambient water mixing ratio and soil water availability. For example, plant stomata are only open when the leaves are exposed to solar radiation and moreover will close to regulate water vapour losses if the soil becomes dry or if leaf temperatures are too high. Because of the high temporal variability of stomata behaviour, mercury emissions from evapotranspiration are calculated on any hourly basis using stomatal resistances derived from the weather model outputs (Kowalczyk et al. 1994). Emissions are scaled up from leaf-level to grid scale using gridded fields of leaf area index (surface leaf area per m^2 of ground) and land cover and land cover dynamical products derived from MODIS satellite observations.

The fluxes of mercury from soils are divided into two categories – mercury emitted from shaded soil (located under a canopy) and mercury emitted from a bare soil surface (Gbor et al. 2006). For bare soil, emissions are parameterised using the soil temperature and the soil mercury concentration while the emissions from shaded soils are expressed as a function of the under-canopy solar radiation flux and the soil mercury concentration. Shaded and bare soil mercury emissions are calculated on an hourly basis using soil temperatures and radiation fluxes taken from the weather model.

The emissions of mercury from vegetation and soils are reliant on estimates of the soil mercury concentration. In the absence of a comprehensive set of soil-mercury observations, gridded soil mercury concentrations have been generated using relationships between soil characteristics and mercury uptake (Table 2). For example, peaty soils with a high humic acid content retain mercury more readily than sandy soils and thus the mercury concentrations in sandy soils are lower. Soil characteristics (on a 0.05° grid) for the Australian continent have been taken from the 1980 Atlas of Australian Resources. The soil water mercury concentrations required for the vegetation mercury emission modelling are derived from soil mercury concentrations using a partitioning coefficient of 0.2 g L^{-1} (Lyon et al. 1997).

Table 2. Mapping between soil characteristics and soil mercury concentrations

Soil Characteristics	**Soil mercury concentration (ng g^{-1})**
Sand/sandy soils	15
Peat soils	100
Saline lakes	500

Emissions from a water surface also use an approach described by Shetty et al. (2008). Here the mass transfer rate is driven by the difference between the equilibrium dissolved mercury concentration (derived from the modelled near-surface atmospheric mercury concentration using a Henry’s law approach) and the ambient dissolved mercury concentration. In the absence of dissolved mercury concentration observations for Australian coastal waters, we have used a mean aqueous concentration of 0.04 ng L^{-1} (Xu et al., 1999).

Figure 5 shows the modelled average hourly natural emission fluxes from vegetation and soil, averaged over the twelve month simulation period. Annual total emission fluxes for the Australian region are given in Table 1. From the table it can be seen that the combined mercury emissions from soil and vegetation comprise about 72% of the total mercury emissions from the Australian landmass and that the total emissions from anthropogenic sources are about a tenth of this at about 7%. Figure 5 shows that the average vegetation emission fluxes are generally less than 1 ng m^{-2} h^{-1}, and that the highest fluxes occur along the coastal regions where vegetation coverage and moisture availability peak. Soil mercury emission fluxes are generally in the range 1–4 ng m^{-2} h^{-1} although peaks of up to 10 ng m^{-2} h^{-1} are predicted for the dry salt lake regions where mercury-soil concentrations have been classified as high (Table 2).

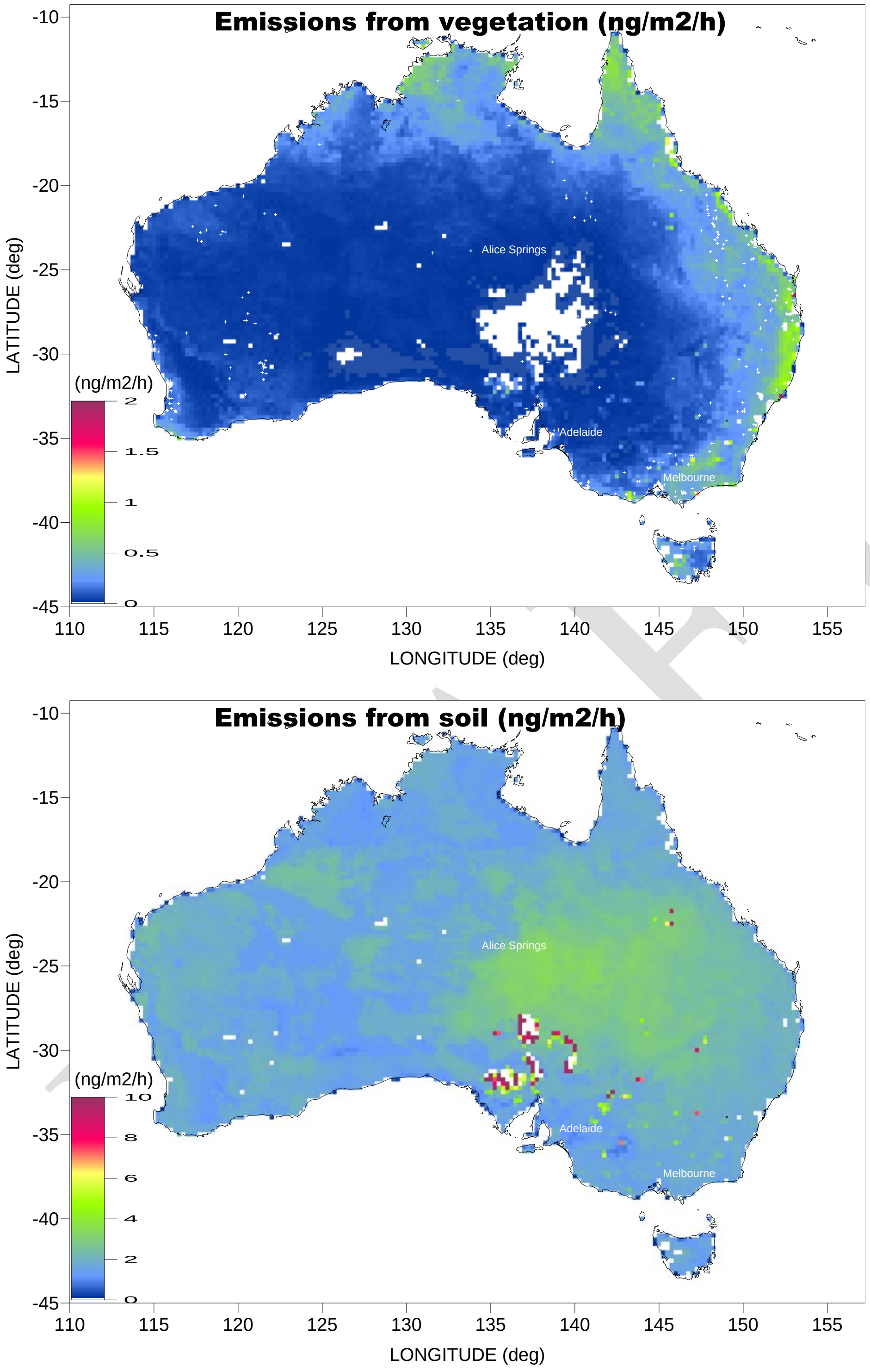


Figure 5. Average emission rates (ng m^{-2} h^{-1}) of mercury (gaseous elemental) from [top] vegetation and [bottom] soil for the Australian continent (note the change of scale).

Figure 6 shows the hourly variation of vegetation and soil mercury fluxes for the region of highest vegetative mercury emission fluxes (32.5°S 152.25°E) for January 2006. It can be seen that the fluxes from vegetation and soil have a strong diurnal variation, which is consistent with solar radiation being the main driver of these mercury sources. The hourly mercury fluxes from the vegetation can be seen to peak at 10–12 ng m^{-2} h^{-1}, which is consistent with the ranges reported in Gbor et al. (2006) and Bash et al. (2004) for the U.S and Canada, but is lower than the range reported by Shetty et al. (2008) for Asia. Wetter conditions and higher soil mercury concentrations may be reasons that the latter emissions are higher.

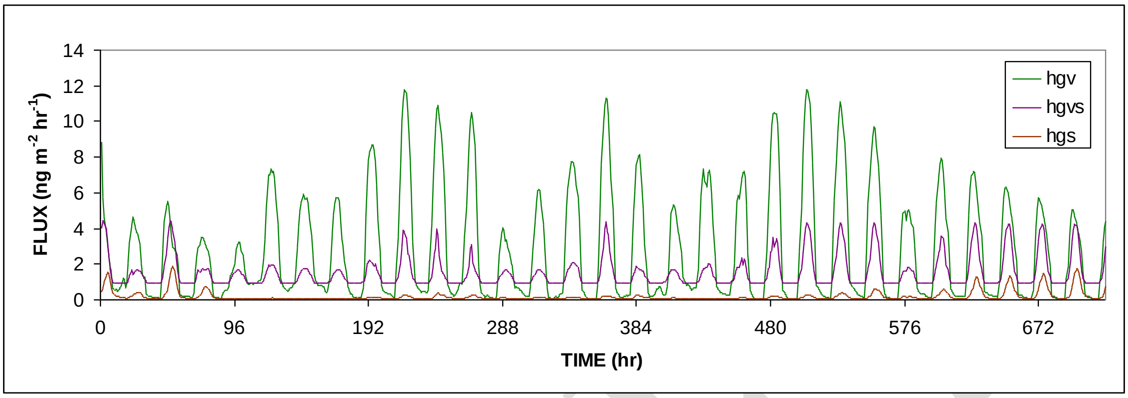


Figure 6. Modelled diurnal variation (for January 2006) of the hourly fluxes of Hg0 from vegetation (hgv); shaded soil (hgvs) and bare soil (hgs) for a location on the east coast of Australia where the maximum vegetation fluxes were generated by the modelling system.

The total budgets of natural mercury emissions reported in Table 1 can be compared with estimates generated by alternative approaches. For example Peterson et al. (2004) estimated mercury emissions from forests, lakes, grasses and soils in the range 131–269 tonne yr^{-1} for the Australian land mass, and one of us (Morrison) has estimated the annual emissions from vegetation to be in the range 63–1315 tonne yr^{-1}. The emissions from vegetation and soils shown in Table 1 total 148 tonne yr^{-1} of which transpiration from vegetation contributes 8 tonne yr^{-1} (5.3%) and emissions from soils contribute the remainder (94.7%). While this total falls within the range reported by Peterson et al. (2004), the vegetation flux is an order of magnitude lower than the lower bound estimate of Morrison. Two potential reasons why the vegetative fluxes of mercury calculated by the continental-scale modelling are lower are as follows.

1. Evapotranspiration (and hence the mercury fluxes) from vegetation are a function of the modelled soil moisture content (which varies on an hourly basis) and may limit the mercury fluxes in many parts of the continent.

2. The magnitude of the mercury fluxes from vegetation is proportional to the soil mercury concentration. In the modelling presented here, this is typically estimated to be 25 ng g^{-1} for many of the vegetated regions in Australia. From sampling undertaken within six Walker (1981) fuel zones across Australia, Packham et al. (2009) measured an average soil mercury concentration of 87 ng g^{-1} (range < 10–145 ng g^{-1}), which is 3–4 times higher than that used in the modelling here. However, it is not clear from Packham et al. (2009) whether the proximity of old gold mining site may have elevated the concentrations at some of the sampling sites.

2.2.3 Natural Emissions – bushfires

Here we summarise the methodology that has been used to estimate the distribution of mercury emissions from biomass burning for the Australian continent. In brief, the burned areas are first identified using satellite fire-scar and hotspot data. These data are then combined with estimated fuel loads and emission factors used to determine mercury emission rates (or source strengths). Information on the daily variation of emissions is obtained from hotspot data, while the diurnal variation is estimated from a fire danger meter using output from the weather model. The resultant emission fields are generated at a temporal resolution of 1 h and a spatial resolution of 1 km×1 km. This method was originally used by Meyer et al. (2008) to generate fuel loadings for the Top End of Australia.

The spatial distribution of fuel loading (tonnes of Carbon per ha) is derived using a semi-empirical method known as VAST (Vegetation And Soil Carbon Transfer) given by Barrett (2002). The method is based on a biogeochemical production model which relates the drivers of production (intercepted radiation, temperature, soil moisture, rainfall, and vegetation class) to biomass and soil pools of carbon. Figure 7 shows the VAST prediction of total carbon loading (assuming that the carbon fraction is 45% of the total loading) for the year 2006. It can be seen that the fuel loading ranges between 0–32 t C ha^{-1} (0–71 t ha^{-1} total mass) which spans the range of 2–20 t ha^{-1} (total mass) reported by NPI (1999) and the range of 2–12 t ha^{-1} reported by Packham et al. (2009). It can be seen from Figure 7 that the highest fuel loading occurs along the Great Dividing Range to the north-east of Melbourne and in Tasmania.

Satellite observations of fire scars (burned areas) and surface hotspots are used to identify when and where a fire has occurred. Fire scar data are generated from NOAA-AVHRR (Advanced Very High Resolution Radiometer) satellite images and hotspot data are taken from MODIS (Moderate Resolution Imaging Spectroradiometer) observations. Fire scar data are used to identify the location of a fire (note however that not all fires have detectable fire scars – see below). Being calculated from the differences in surface reflectance from successive satellite passes at 10 day intervals, the fire scars do not have the temporal resolution (hourly) required by the transport modelling. However the hotspot data (which are observed daily) may be used to provide a commencement and cessation time for a fire and additionally to provide day-to-day variations of fire intensity.

The hotspot data are used as follows. Within each grid cell with a fire scar, the total number of hotpots occurring within a 45-day period (encompassing 40 days prior to the fire scar date and five days after the fire-scar date) is first determined. This sampling time was found to be the best compromise between removing hotspots that were not associated with the scar and errors in the timing of the actual scar.

The fraction of fire scar area occurring at a particular grid on a given day is determined by the number of hot spots detected on that day, divided by total number of hot spots detected over the time window mentioned above. The carbon loading at the grid point is then given by the product of total carbon loading at the grid and the fraction of scar area (burned area).

If no hotspots are detected in the scar, then it is assumed that the fire commenced 5 days prior to the fire scar date, i.e. the midpoint of the 10 day interval between image pairs. This is equivalent to assuming that the fire started mid-way through the sampling period before the reporting date, and ended mid-way through the sampling period following the reporting date.

An additional case can exist where hot spot data are present but a corresponding fire scar is not. In this case, a fire is assumed to be present for the 24 hours centred on the time that the hot spot was observed. In determining the size of the burnt area, a hotspot size of 392 ha is used (from a regression analysis between hot spot number and scar area undertaken by Meyer et al. (2008)).

Mercury emissions are calculated from the carbon loading using an emission factor of 112 µg of mercury per kg of dry fuel and assuming a 45% carbon content of the fuel (Friedli et al., 2003). This emission factor is representative of mercury emission from fires in a temperate forest. The mercury released during forest fires is predominantly Hg0 and the ratio between particulate and elemental mercury varies widely depending on the fuel type. Following Friedl et al. (2003), we have assumed that 13% of total emissions are particulate mercury (Hgp) and 80% are elemental mercury emissions (Hg0). We assigned the remaining 7% of the emission mass to reactive gaseous mercury (RGM). Following the calculation of a daily mercury emission rate, an hourly emission rate was calculated using a fire danger meter approach (Luhar et al. 2008).

A potential issue with the approach described above is that there can be a significant number of hot spots that do not have corresponding fire scars. For example, Luhar et al. (2008) found that 27% of hot spots have no corresponding fire scars for the 2004 burning season in the Top End. For these ‘anomalous’ hotspots, the carbon loading at each grid point is taken as the total carbon loading at that grid point under the assumption that a hot spot emits for 24 hours for the date of the hot spot occurrence. The impact of anomalous hot spots on the fire emission estimates was investigated by calculating mercury emissions for January 2006 for cases in which the anomalous hot spot data were included and omitted. These calculations showed that emissions from the anomalous hotspot regions contributed 88% of the total mass. A similar analysis undertaken for other months indicated that the anomalous hot spots can contribute between 10% and > 90% of the burnt area. This result is of particular concern in areas of agricultural burning where the hotspots may be assigned a forest fuel load rather than a crop fuel load and thus may over estimate the emissions by a factor of 2–10. Because of this concern, a second set of fire emission data sets were generated for input into the continental scale modelling. In these data sets the hotspots that are not coincident with areas of natural vegetation or plantations were removed. This was done for all regions except Queensland where most of the hotspots appear to be associated with land clearing and therefore were included.

Figure 8 shows the estimated spatial distribution of fire emissions for 2006. Using the approach described above, approximately 62×10^6 ha of vegetation was estimated to be burnt, which falls within the range $32–80\times10^6$ ha reported in the 2006 State of the Environment report (Beeton et al, 2006, section 5.2) for a typical year in Australia. The mercury emissions from fires for 2006 is estimated to be 41.8 tonnes (Table 1), which comprises 36% of total emissions (anthropogenic and natural) from the Australian land mass. The estimated annual emissions of 41.8 tonnes falls within the range of 2.6 tonnes reported by Nelson et al. (2004), who used the NPI (1999) emission factors but appears to have assumed a much smaller area burnt, and 129 tonnes reported by Packham et al. (2009), who estimated a larger average burn area of 146×10^6 ha per year.

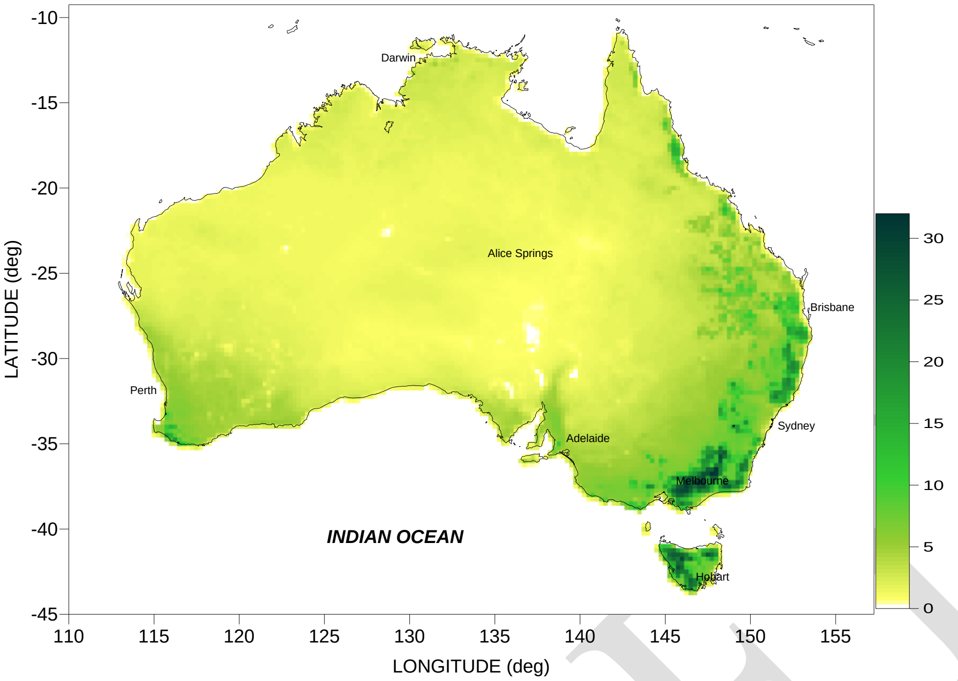


Figure 7. The spatial distribution of fuel carbon loading (t C ha $^{-1}$) for the year 2006.

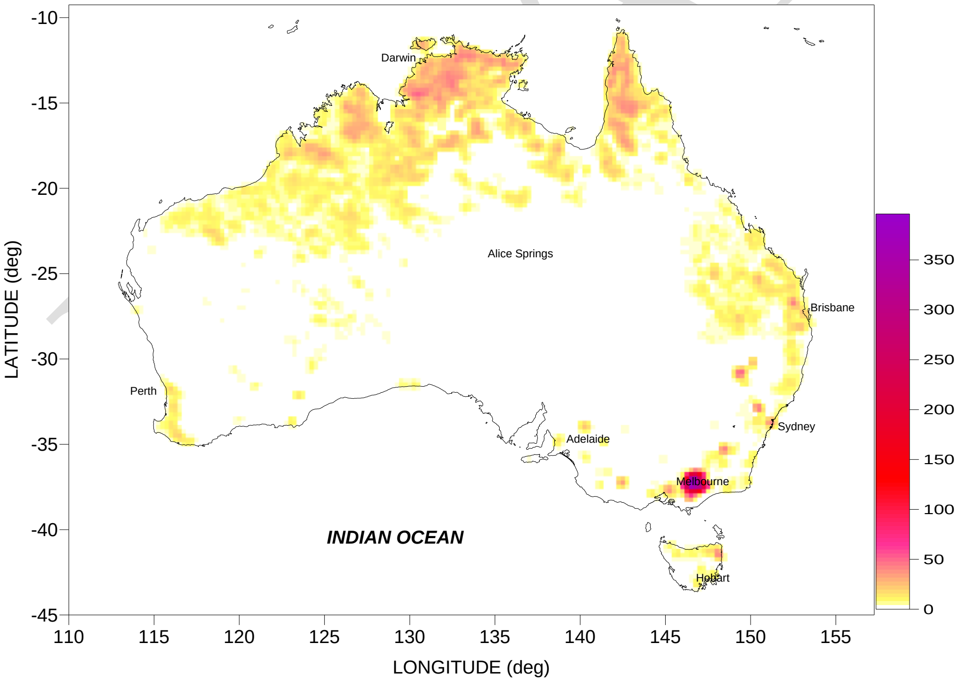


Figure 8. The spatial distribution of annual fire emissions for mercury (kg per grid square (0.25° x 0.25°) for 2006.

2.3 Transport Modelling

2.3.1 Continental scale transport modelling

The transport model uses the weather model output and the mercury emissions to calculate three-dimensional time varying mercury concentrations and two dimensional wet and dry deposition fields. The transport modelling has been undertaken with the CSIRO Chemical Transport Model (CTM) which was originally developed for the Australian Air Quality Forecasting System (Cope et al. 2004) and used to generate short-term 36 hour dust and smoke forecasts for Australian continent. For the current project, the CTM has been modified to calculate the natural emissions of mercury (from soil, vegetation and water) using hourly meteorological data from the weather model. Additionally the CTM wet deposition module has been coupled to the output of the CCAM cloud algorithms and the CTM default dry deposition algorithms have been enhanced to include additional sink pathways in a vegetation canopy which are important for mercury (Gbor et al. 2006). The coefficients used by the CTM for the calculation of wet and dry deposition are similar to those used by TAPM for the urban and local scale modelling (see Table 3).

These changes give the CTM the ability to model the processes of mercury emission (natural and anthropogenic) and plume rise (for the anthropogenic point sources), transport in the prevailing winds, dilution due to turbulence and mercury losses through wet and dry deposition at the surface.

2.3.2 Boundary concentrations

Unlike CCAM, which uses a stretched grid which spans the globe, the transport model uses a limited area grid which spans the region shown in Figure 4. Because the model has finite spatial boundaries in the horizontal, it is necessary to specify the concentrations of mercury which are present at the boundaries under in-flow wind directions.

To our knowledge, background mercury concentrations have not been routinely observed for the Australian region. As a result we have derived these background concentrations from a global model simulation reported in Seigneur et al. (2001). Figure 9 shows the annual-average near-surface concentrations of mercury predicted by their global model. It can be seen that the modelled Hg0 concentrations are in the range 1.2–2.2 ng m^{-3} and that a north–south gradient of decreasing concentration is predicted. Concentrations of RGM and Hgp are in the range 1–200 pg m^{-3}. However, in contrast to Hg0, the highest concentrations of these two species are located closer to local source regions and there is less evidence of a consistent north–south concentration gradient, which may be a result of the shorter atmospheric lifetimes of these mercury species.

Seigneur et al. (2001) compared the predicted mercury concentrations with observations taken at Mace Head, Ireland and reported that the modelled Hg0 mercury concentrations fell within the observed range and the modelled RGM and Hgp mercury concentrations fell at the lower end of the observed range. They also undertook a comparison of the vertical concentration gradients using aircraft measurements and reported that the modelled results were consistent with the observations.

Using the Seigneur et al. modelling results as guidance, we have prescribed a boundary concentration of 1.3 ng m^{-3} for elemental mercury, 10 pg m^{-3} for reactive gaseous mercury, and 2 pg m^{-3} for particulate mercury. These boundary conditions give a total gaseous mercury background concentration that is consistent with a background of ~1 ng m^{-3} observed by Nelson et al. (2008) for the Sydney region.

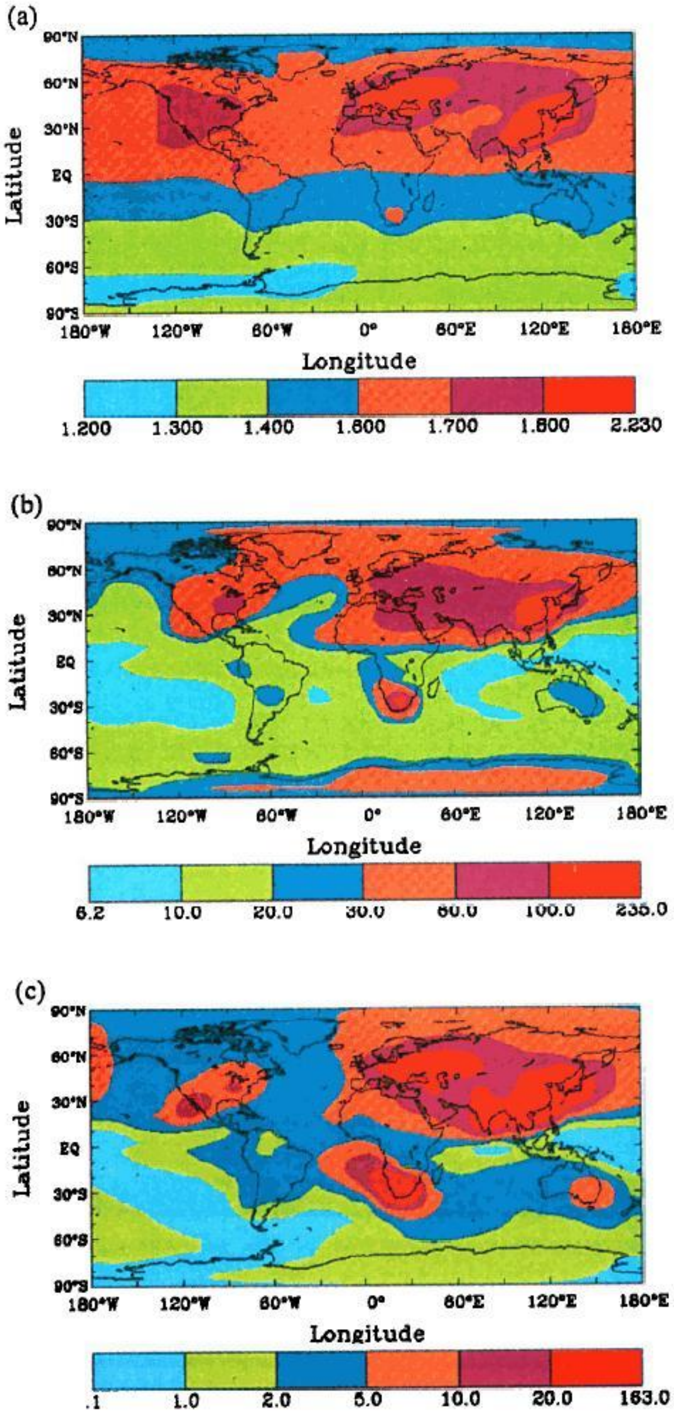


Figure 9. Annual average surface concentrations of (a) elemental gaseous mercury (ng m^{-3}), (b) reactive gaseous mercury (pg m^{-3}) and (c) particulate mercury (pg m^{-3}). Reproduced from Plate 1 in Seigneur et al. (2001).

2.3.3 Urban and near-source transport modelling

The transport and dispersion modelling for the urban and near-source regimes was carried out using the air pollution component of TAPM version 4 (Hurley, 2008), which was run at the same time a the meteorological component described in section 2.1.2. The three mercury species were treated as chemically inert and modelled in tracer mode. Hard-wired coefficients were adjusted in a special version of the TAPM code. Particulate mercury was modelled as PM2.5 with a scavenging coefficient of 1.0. Dry and wet deposition of elemental and reactive gas mercury was modelled using the standard TAPM formulation but with coefficients listed in Table 3.

Table 3. TAPM coefficient for wet and dry deposition of gas phase mercury species

Species	**MW**	**R_{soil} [s/m]**	**R_{water} [s/m]**	**Henry's Law constant [M/atm]**
Hg0	200.0	3.60×10^{7}	3.87×10^{3}	0.110
RGM	271.5	28.4	3.04×10^{-4}	1.40×10^{6}

The urban scale TAPM modelling for Melbourne and Sydney explicitly modelled the anthropogenic sources, both the industrial point sources and the commercial-domestic sources, which were distributed according to population. These results are presented in this report, together with the total including the contribution from natural sources, which were derived from the output of runs of the continental scale model that only included natural mercury sources

Near-source modelling was undertaken for five significant anthropogenic source groups of mercury (see Figure 4). This modelling only included the anthropogenic point sources (as per the project brief).

- Kalgoorlie –includes the two largest point sources at Gidji and Fimiston (based on data in the 2006 National Pollutant Inventory) as well as Kalgoorlie Nickel Smelter;
- Pinjarra (WA) – alumina refinery, third largest point source in Australia;
- Mt Isa – copper and lead stacks are the fourth and twelfth largest point sources;
- NSW Central Coast Power Stations (Vales Point, Eraring and Munmorah) – significant black coal-fired generators within the Sydney modelled domain; and
- Latrobe Valley Power Stations (Loy Yang, Hazelwood, Yallourn, and Energy Brix – the former Morwell power station) – significant brown coal-fired generators in Victoria.

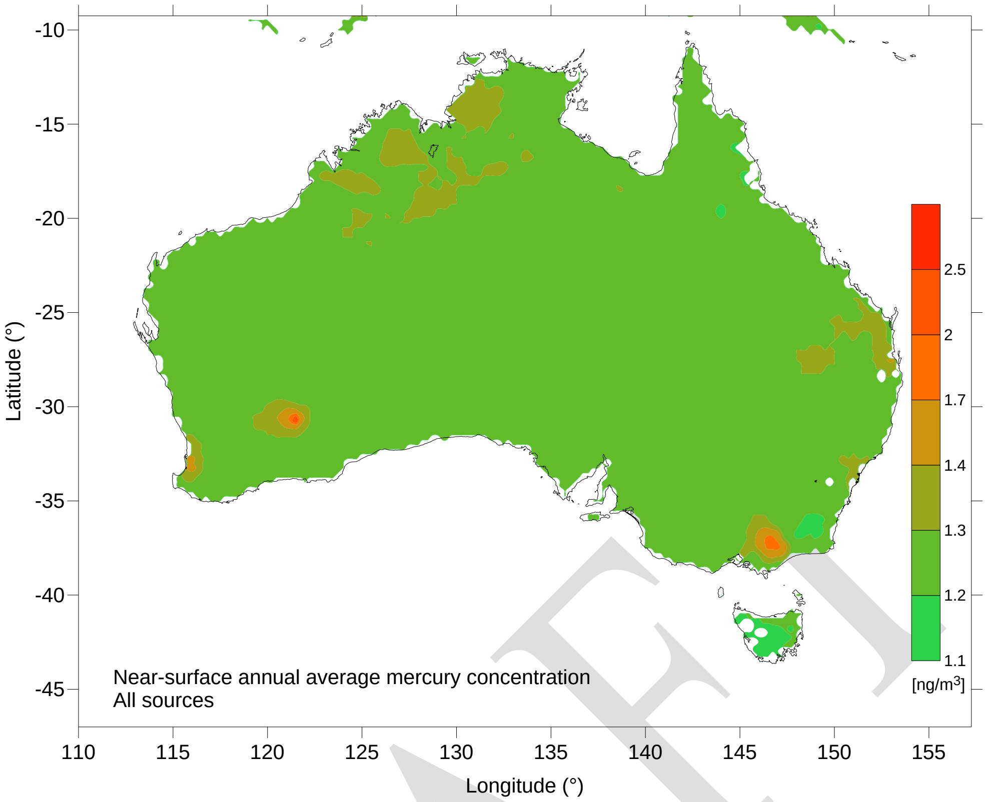
3. CONTINENTAL TRANSPORT MODELLING RESULTS

The mercury modelling system described in section 2 was run for the year 2006. Model results for the separate mercury species have been combined and used to generate monthly average concentrations and monthly average cumulative wet and dry deposition masses. Because of remaining uncertainties in the mercury emission from fires, separate scenarios with and without fire emissions were modelled.

3.1 Continental Concentrations

Figure 10 shows the annual average near-surface total mercury concentrations for the Australian landmass for the two emission scenarios. For the case in which bushfire emissions are excluded Figure 10 (bottom) it can be seen that the modelled concentrations are 1.2–1.3 ng m^{-3} over most of the continent. This is consistent with the observations taken at Macquarie University (Nelson et al. 2008). The highest modelled concentration of 2.6 ng m^{-3} is located in the vicinity of Kalgoorlie gold mine and reflects the significant mercury emissions associated with that source. Note that the peak modelled concentrations at this location is limited by the effective grid resolution of the model (~25 x 25 km^2) in the horizontal and thus underestimates local concentrations. Near-source modelling presented in Section 5.1 shows that at 1 x 1 km^2 resolution the peak modelled concentration is about ten times larger. Small changes in mercury concentrations associated with urban sources (and to a lesser extent natural sources) together with reductions associated with deposition are also evident from Figure 10 (bottom), although the magnitude of the changes are small.

Figure 10 (top) shows the effect of including the mercury emissions from bushfires and indicates that the emissions from this source group can lead to local and regional impacts which are comparable to, or larger than, the impacts resulting from the largest anthropogenic sources (wind blown dust may also be another significant source, but an investigation of the contribution from dust was outside the scope of this study). It should be noted that in this study the bushfire emissions have been modelled as surface sources. While this may be a good approximation for the emissions from small fires (i.e. savannah burning in the Top End), it could lead to over estimates of the ground level concentrations from large forest fires where the smoke plumes may rise several kilometres above the surface.



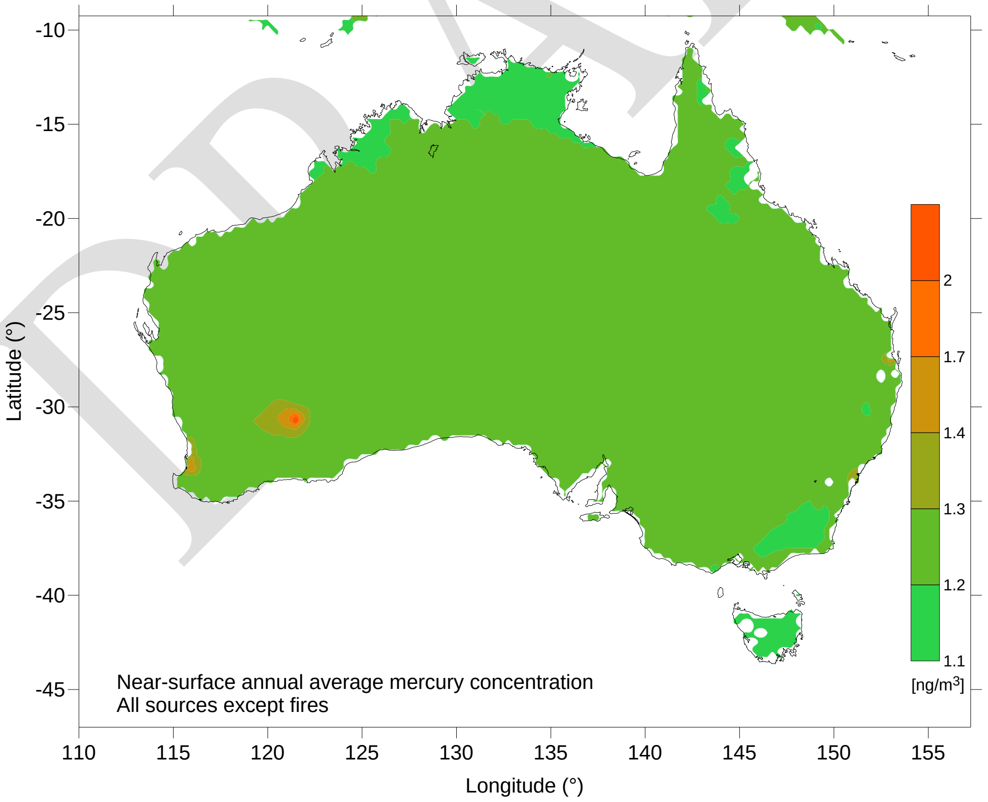


Figure 10. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the Australian region. Top –including bushfire emissions. Bottom – bushfire emissions have been omitted (not change in colour scale).

3.2 Continental Wet Deposition

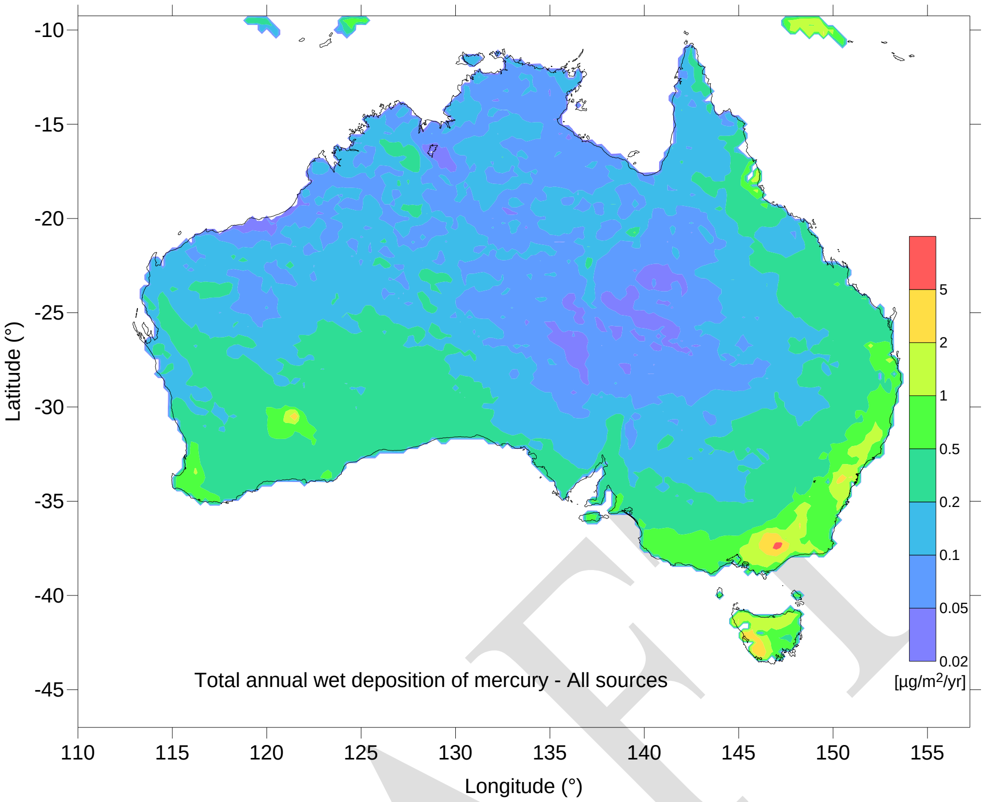
Wet deposition is calculated by the transport model whenever precipitation occurs within a model column. Wet deposition is a function of the precipitation rate, the concentration of mercury within the precipitating cloud, the cloud-water concentration and the Henry's law constant/scavenging coefficient for each of the mercury species. In this regard it should be noted that elemental gaseous mercury is relatively insoluble ($H = 0.11$ M atm^{-1}), reactive gaseous mercury is very soluble ($H = 1.4x10^6$ M atm^{-1}) and particulate mercury is readily scavenged by cloud water droplets (Seigneur et al. 2001). Thus it may be expected that the majority of the mercury mass deposited by precipitation will be in the form of RGM and Hgp.

Figure 11 shows the total annual (2006) wet deposition mass of all three mercury species for the two emission scenarios (with and without fires). For the non-fire scenario it can be seen that the wet deposition mass peaks on the western coast of Tasmania (2 μg m^{-2} yr^{-1}), within the Kalgoorlie region (2 μg m^{-2} yr^{-1}) and along the Great Dividing Range in eastern Australia. These locations correspond either to regions of higher rainfall or regions of elevated mercury concentrations or combinations of these two factors. The enhanced wet deposition mass predicted along the Tasmanian coastline is a result of the interaction of the persistent global background mercury concentrations and enhanced rainfall occurring along the west coast of Tasmania.

Comparing the top and bottom parts in Figure 11, it can be seen that the wet deposition patterns for the no-bushfire and bushfire scenarios are similar (with the exception of a region of enhanced deposition (up to 5 µg m^{-2} yr^{-1}) to the north-east of Melbourne for the bushfire scenario), which suggests that the wet deposition is limited by the availability of precipitation for the simulation period. The total mercury mass deposited by precipitation onto the Australian land mass is estimated to be 1.5 t yr^{-1} in the absence of bushfires and 1.8 t yr^{-1} when bushfires are included. This is equivalent to about 0.8 % of the total emissions from the region (Table 1).

3.3 Continental Dry Deposition

Figure 12 shows the spatial distribution of annual total dry deposition mass for the two emission scenarios. Dry deposition refers to the transfer of gas and aerosol phase mercury to sinks on vegetation (such as leaf stomata), soil and water surfaces by atmospheric turbulence and molecular diffusion. For particulate mercury, deposition rates may also be enhanced by gravitational settling of the particles. Comparing Figure 10 top and bottom it can be seen that enhanced rates of dry deposition are modelled to occur within the vicinity of the bushfires and also close to dry salt lakes which are modelled to have significant soil mercury concentrations and surface fluxes (Figure 5). Dry deposition is calculated to contribute 19.5 t yr^{-1} (no bushfires) to 21 t y^{-1} (including bushfires) to the Australian land mass, which is equivalent to about 10 % of the emitted mercury from the region (Table 1).



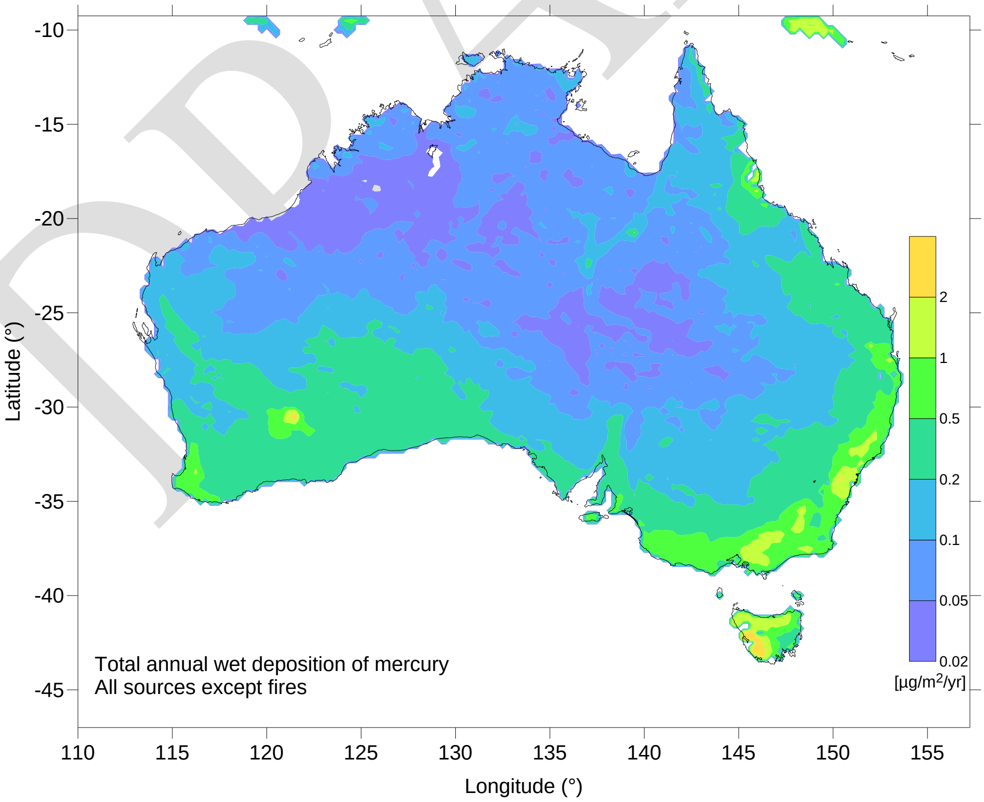
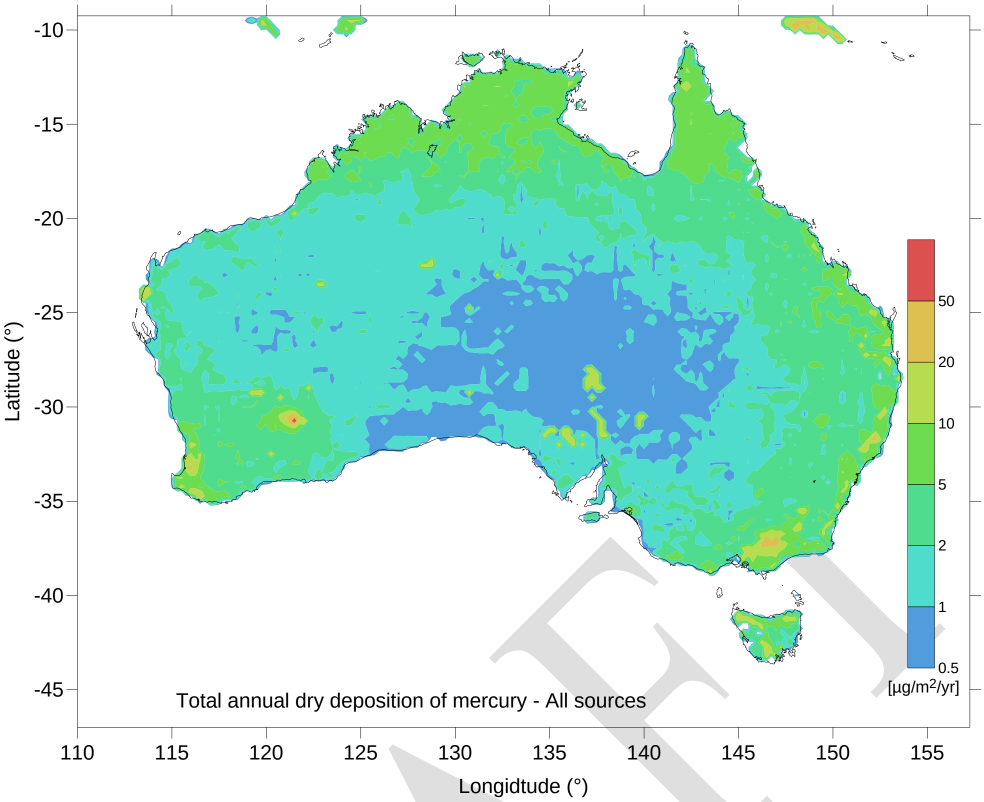


Figure 11. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the Australian region. Top – including bushfire emissions. Bottom – bushfire emissions have been omitted.



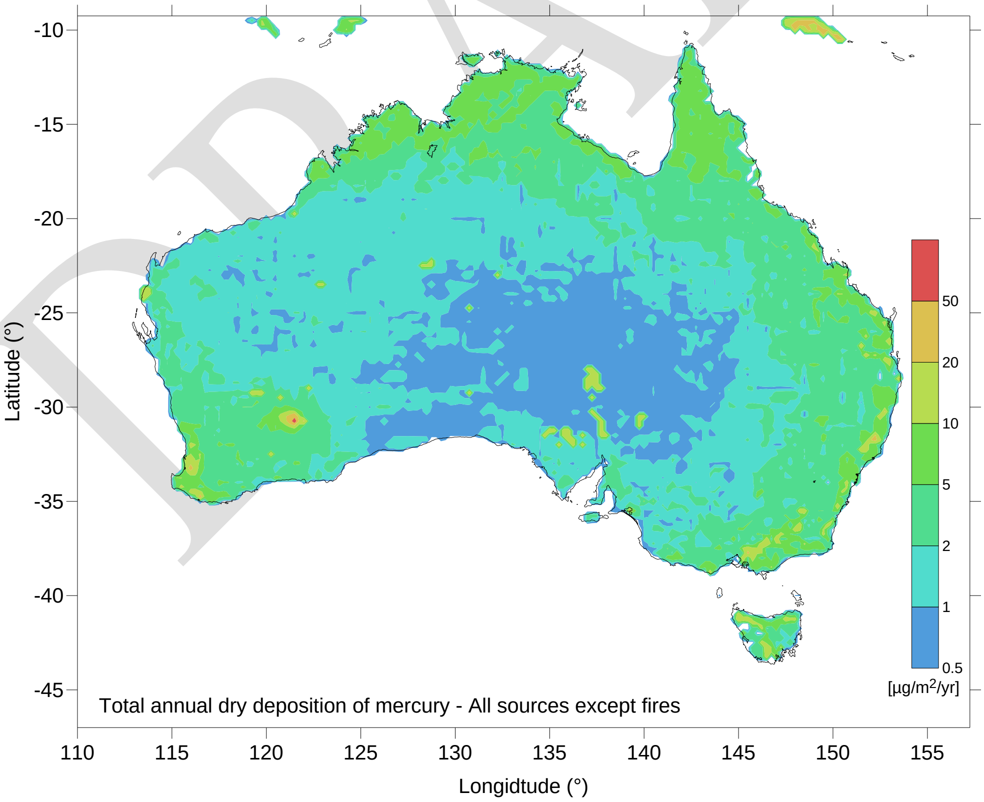


Figure 12. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the Australian region. Top – including bushfire emissions. Bottom – bushfire emissions have been omitted.

4. URBAN-SCALE TRANSPORT MODELLING RESULTS

4.1 Urban-scale Concentrations

Figure 13 shows the annual average near-surface mercury concentrations for the urban area centred on Melbourne. The contribution of the anthropogenic emissions is given in the top part of the figure. It shows the maximum concentration is 0.7 ng m^{-3} from an industrial source east of the city (Lilydale quarry) but within the most of the urban area, the contribution from anthropogenic sources is less than about 0.1 ng m^{-3}.

The lower part of Figure 13 includes the contribution from natural sources (vegetation, soil, water, and the continental background), which was derived from output of a special continental scale run that included only these sources. This output on a 0.25° x 0.25° grid was interpolated to the 3 km x 3 km grid of the urban run and the natural and anthropogenic contributions added. The contribution from natural sources is approximately 1.2 ng m^{-3} across the whole of the modelled domain, and is thus the dominant contributor to ambient concentrations in this urban area.

Figure 14 shows the same pair of graphs for the urban area centred on Sydney. The maximum concentrations of 1.2 ng m^{-3} are near the point source coke works in the vicinity of Wollongong. Away from these sources, the anthropogenic contribution is less than 0.2 ng m^{-3}. (Note that a major mercury source in the 2006 NPI database due to contaminated soil at Orica's site near Botany Bay was omitted from the modelling because of uncertainty in the emission rate, which was reduced by a factor of 40 in the 2007 and 2008 NPIs.)

The lower part of Figure 14 shows the concentrations due to all sources except the fires. The non-anthropogenic sources contribute most of the 1.2 ng m^{-3} away from the urban areas and total annual average concentrations (due to natural and anthropogenic emissions) away from the Illawarra are below 1.5 ng m^{-3} except near the Illawarra point sources where they reach 2.5 ng m^{-3}.

Overall for the two urban areas, it can be seen that the contribution from urban anthropogenic emissions to average mercury concentrations is generally smaller than that from natural and background sources, but can be up to the same magnitude close to major mercury sources.

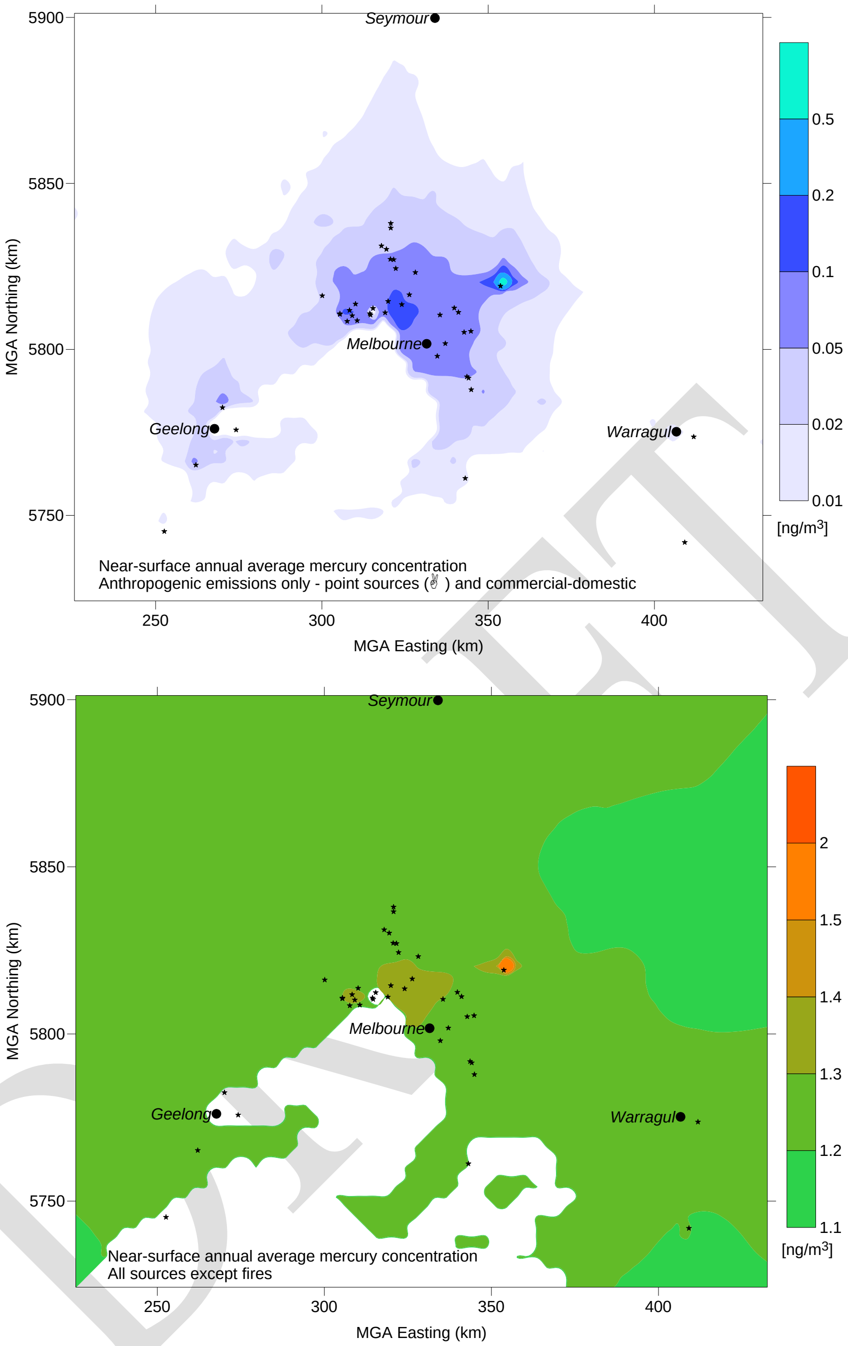


Figure 13. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Melbourne urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.

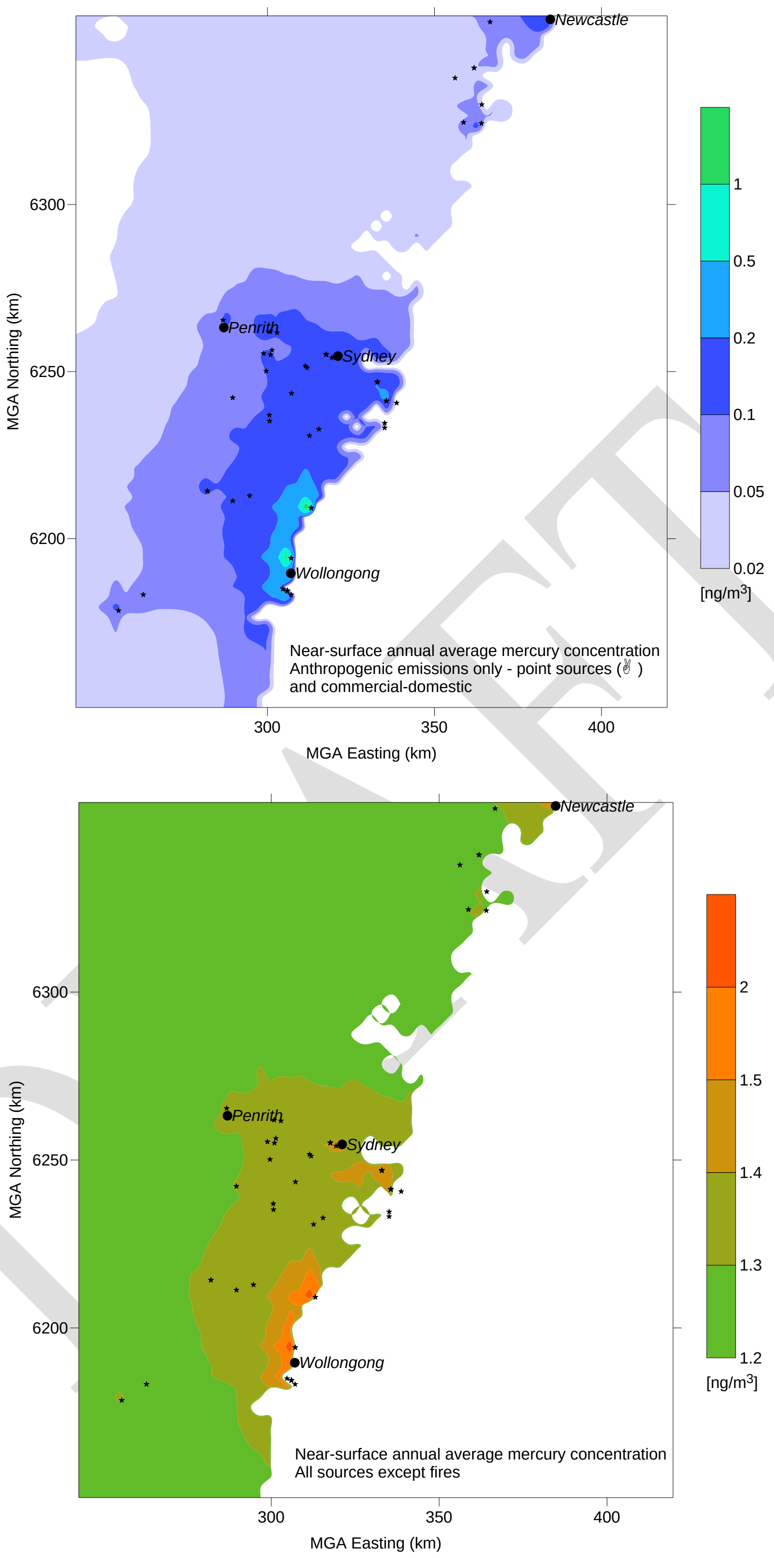


Figure 14. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Sydney urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.

4.2 Urban-scale Wet Deposition

The annual total wet deposition mass for the urban scale modelling of Melbourne is shown in Figure 15. The contribution of anthropogenic sources only is given in Figure 15 (top) and shows that the highest mass (over 20 µg m^{-2} yr^{-1}) generally deposited over regions of elevated topography surrounding Melbourne to the north, west and east. The other regions of higher deposited mass are located downwind of significant mercury sources.

Figure 15 (bottom) includes the contribution from natural sources and the global background which together typically add 0.5–1.5 µg m^{-2} yr^{-1} to the totals. In contrast to the mercury concentrations, it can be seen that the local sources are the dominant contributors to the wet deposition mass within the Melbourne region.

Figure 16 (top) shows the wet deposition mass from anthropogenic emissions for the urban scale Sydney modelling. It can be seen that the wet deposition totals are higher than predicted for Melbourne. The highest wet deposition masses (over 100 µg m^{-2} yr^{-1}) are predicted for the Illawarra and Central coast regions. These peaks are caused by the proximity of significant sources to elevated terrain plus enhanced rates of modelled rainfall in these regions. As for Melbourne the magnitude of the wet deposition does not change noticeably when the natural sources are included.

This urban-scale modelling shows significantly higher wet deposition than the continental scale modelling. This is a result of the higher resolution of the urban modelling (3 km grid spacing vs. 25 km grid spacing for the continental modelling), as well as better resolution of the rain processes. The average wet deposition across the whole of the Melbourne domain shown in Figure 15 (bottom) is about 3 µg m^{-2} yr^{-1}, which is close to the continental scale result (Figure 11) of about 2 µg m^{-2} yr^{-1} near Melbourne. On the other hand, the equivalent averages for Sydney are 20 µg m^{-2} yr^{-1} for the urban-scale modelling and 2 µg m^{-2} yr^{-1} for the continental scale modelling. This appears to be due to the better resolution of fine scale rain processes in the TAPM modelling with higher rainfall in the Sydney region, but is an issue that requires further investigation.

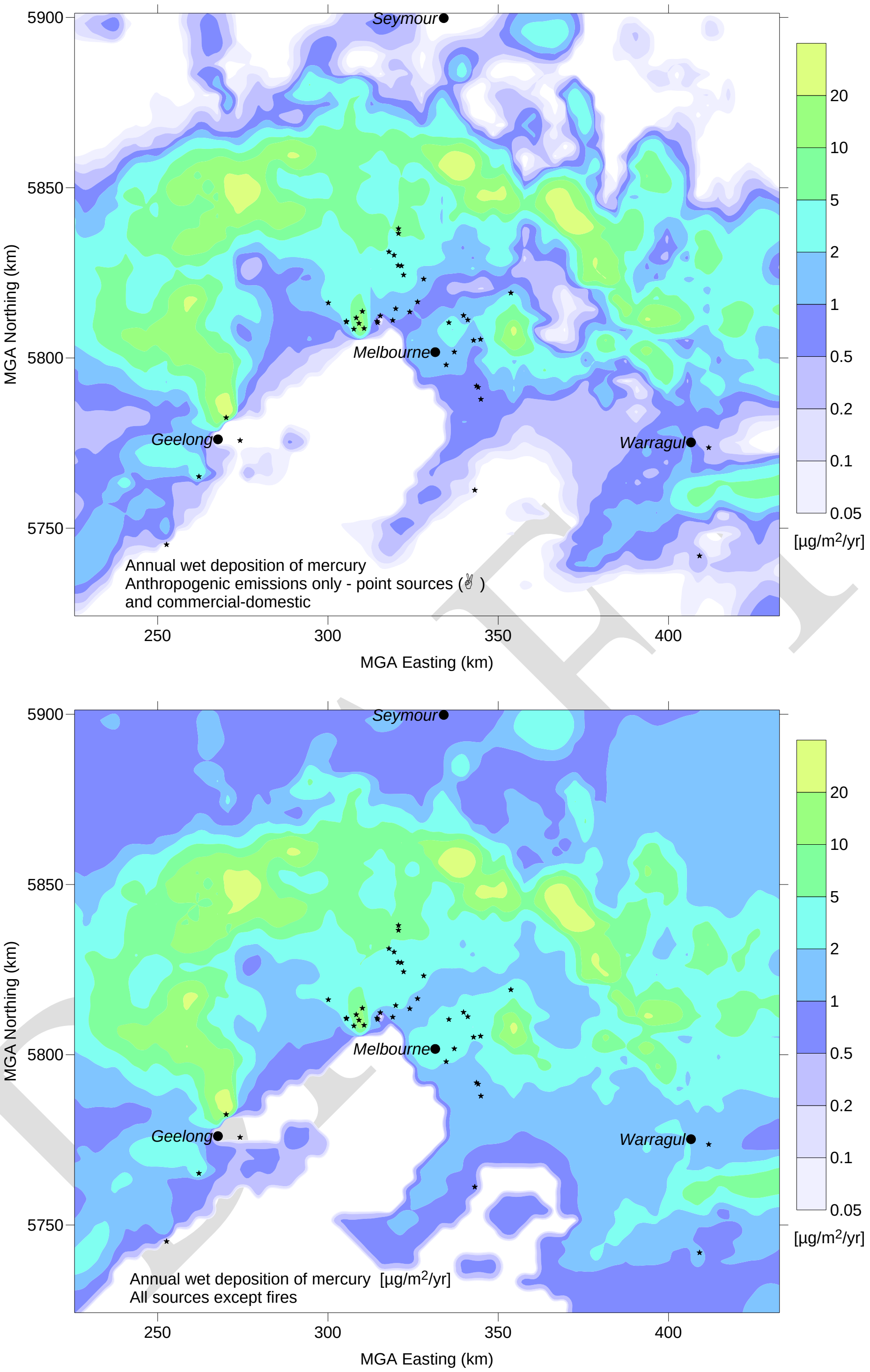
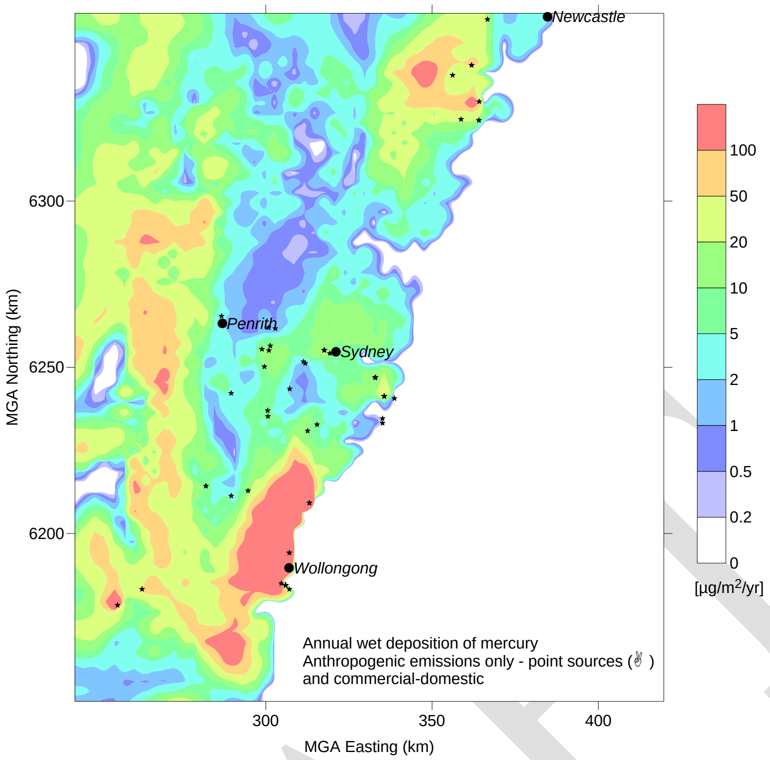


Figure 15. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Melbourne urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.



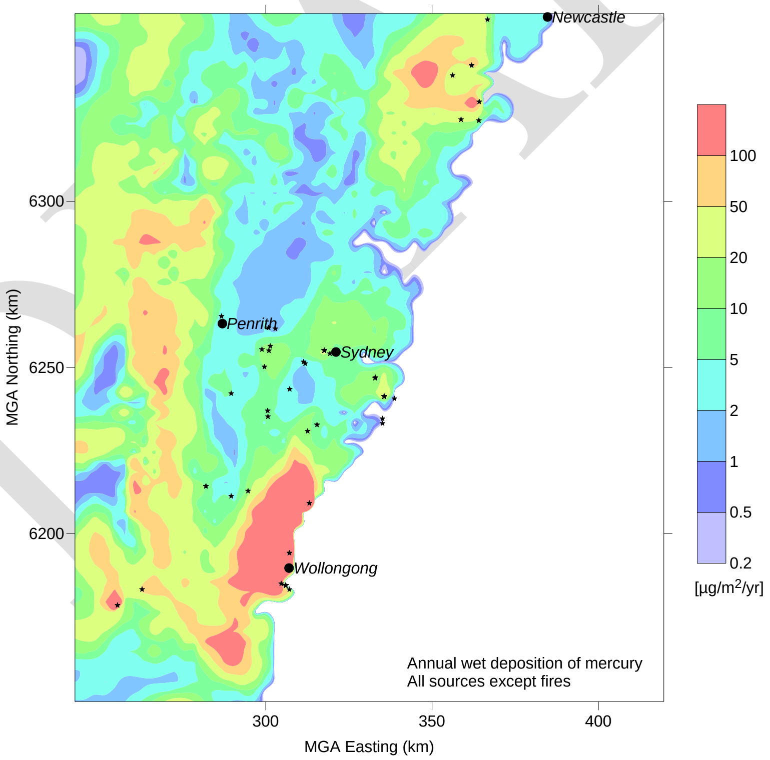


Figure 16. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Sydney urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.

4.3 Urban-scale Dry Deposition

Figure 17 (top) shows the contribution of the anthropogenic emissions to the dry deposition mass in the Melbourne region. In contrast the wet deposition (which is dominated by the spatial distribution of rainfall), the highest dry deposition masses (over 2 µg m^{-2} yr^{-1}) generally occur close to the major sources. These maxima are about one tenth of the highest wet deposition totals.

Figure 17 (bottom) shows the combined contribution of the anthropogenic and natural emissions, together with the continental background mercury concentrations to the dry deposition mass in the Melbourne region. It can be seen that the largest total dry deposition mass (which occur in the more heavily forested areas) are dominated by deposition from the natural emissions and continental background.

Figure 18 shows the dry deposition results for the Sydney urban scale modelling. In the case of the anthropogenic emission modelling it can be seen that the Illawarra region again has the highest deposition masses (marginally above 100 µg m^{-2} yr^{-1}). This is consistent with the presence of significant low-level emissions within that region. The natural emission and continental background contribution is similar to that predicted for the Melbourne region. However this is seen to have proportionally less impact in the Illawarra region.

There is good consistency between the continental and urban scale modelling results for dry deposition. The continental scale modelling (Figure 12) predicts dry deposition masses of about 5 µg m^{-2} yr^{-1} for the Melbourne region and 10 µg m^{-2} yr^{-1} for Sydney region. These are within 20% of the averages computed across the whole urban domains shown in the bottom parts of Figure 17 and Figure 18.

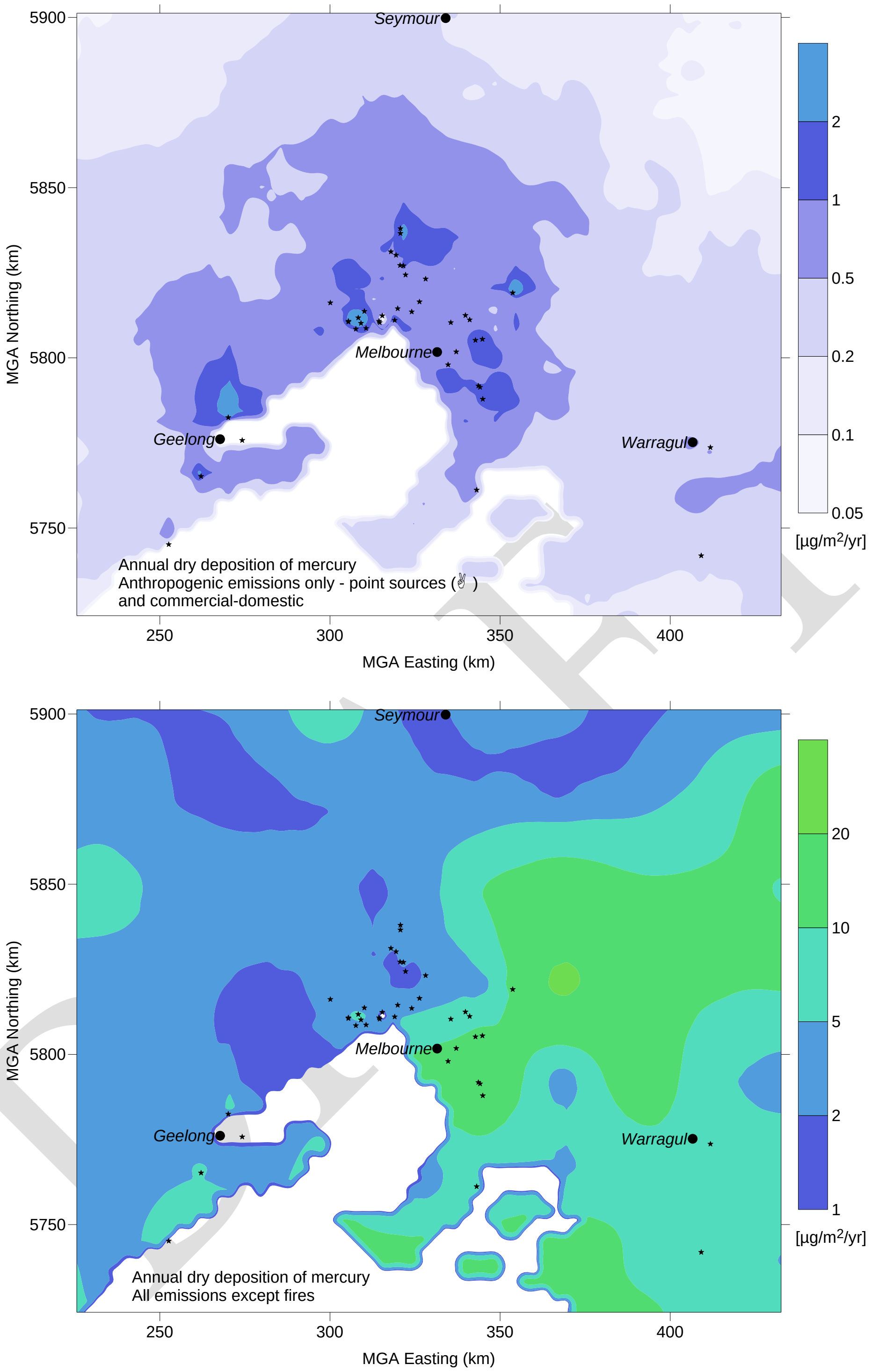


Figure 17. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled Melbourne urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.

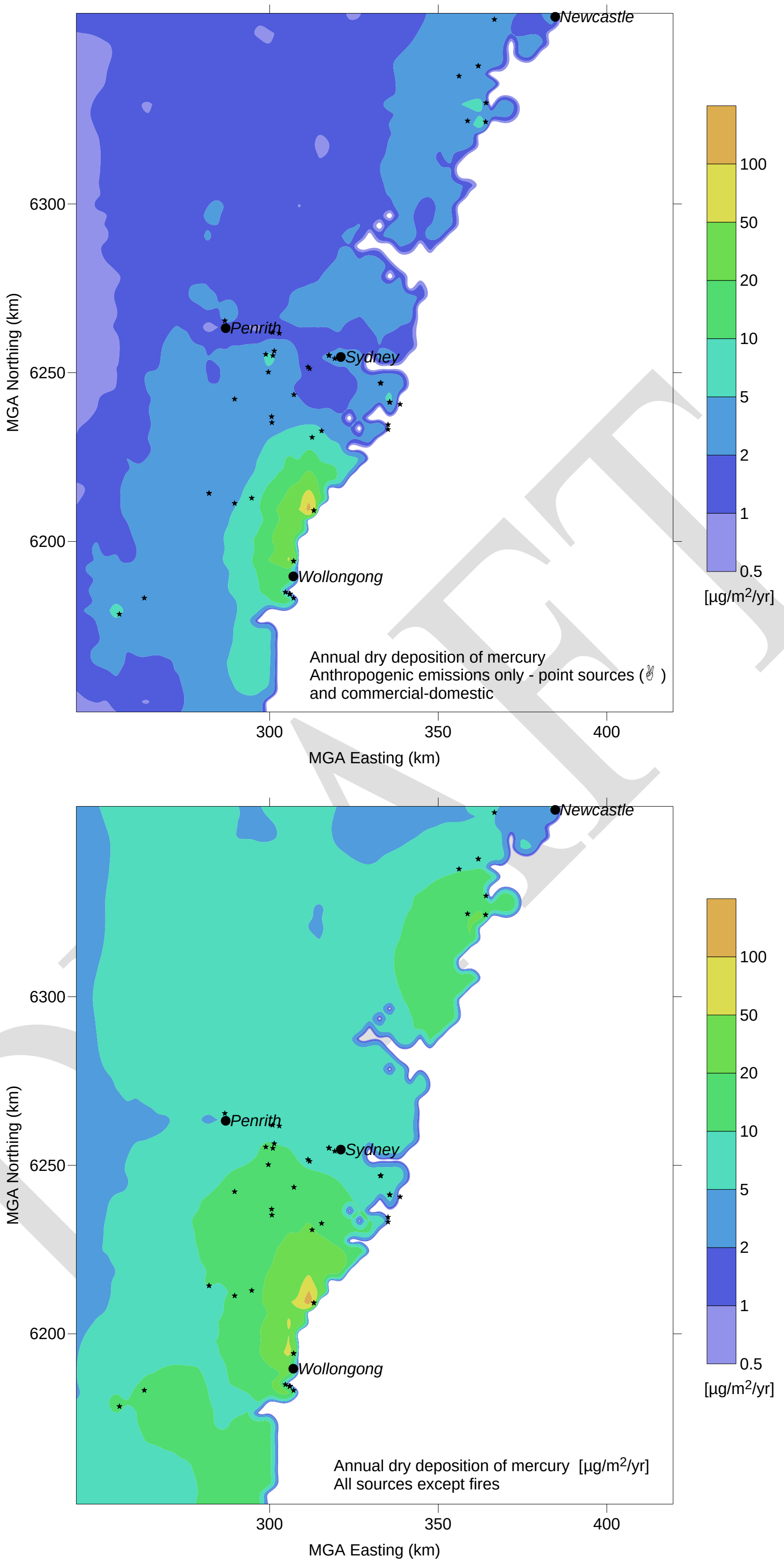


Figure 18. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled Sydney urban area. Top – only anthropogenic emissions (point sources and commercial-domestic). Bottom – all emissions except bushfires.

5. NEAR-SOURCE TRANSPORT MODELLING RESULTS

5.1 Near-source Concentrations

Figure 19 to Figure 23 show the modelled annual average mercury concentrations for the five significant anthropogenic source groups, which were modelled down to 1 km grid spacing. A description of the included anthropogenic point sources is given in section 2.3.3. It can be seen that a very wide range of maximum annual average concentrations is predicted (ranging from 0.005–20 ng/m^3) due to anthropogenic emissions. This results both from the observed range of emissions (100–7000 kg/yr) from the modelled sources and the range of final plume heights (which depends on the stack height, and the efflux temperature and speed). Additionally, the local meteorology influences the final plume rise and the downwind dispersion characteristics of the plumes.

The ground level concentrations for the near-source transport modelling are typically at least ten times higher than that predicted by the continental scale modelling. This is a result of the higher resolution of the near source modelling (1 km grid spacing vs. 25 km grid spacing for the continental modelling), but there is good consistency between the models. For example, the average concentration across the whole Kalgoorlie domain shown in Figure 19 of 0.9 ng m^{-3} agrees well with the continental scale result for anthropogenic contributions at Kalgoorlie, which can be estimated from Figure 10 to be approximately 1 ng m^{-3}.

The two major sources at Kalgoorlie (Figure 19) demonstrate the joint effects of emission rate and stack height on the resultant ground level concentrations. It can be seen that the highest ground level concentrations result from the Fimiston mercury emissions, even though the Gidgi source strength is six times larger. The much higher Gidgi stack (180 m vs. 30 m for Fimiston) and hence enhanced dispersion is the major reason for the lower ground level concentrations due to the emissions from this source.

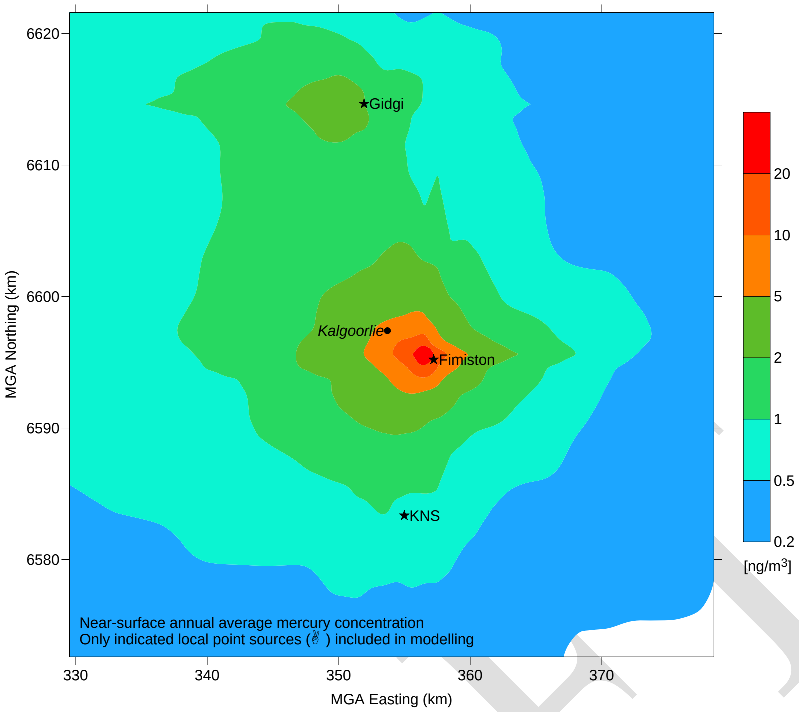


Figure 19. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Kalgoorlie near-source region. Only the indicated point sources of emissions are included in the modelling.

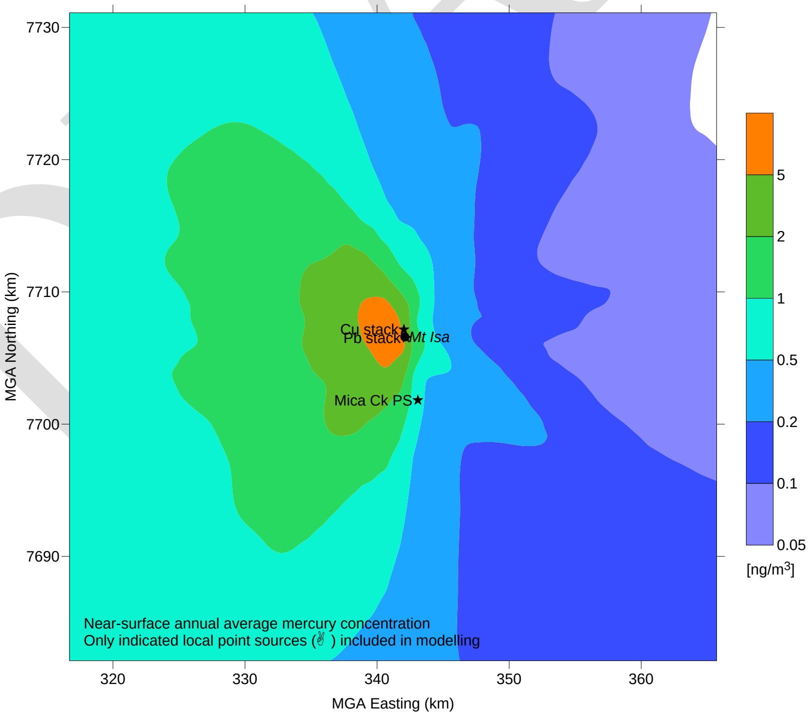


Figure 20. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Mt Isa near-source region. Only the indicated point sources of emissions are included in the modelling.

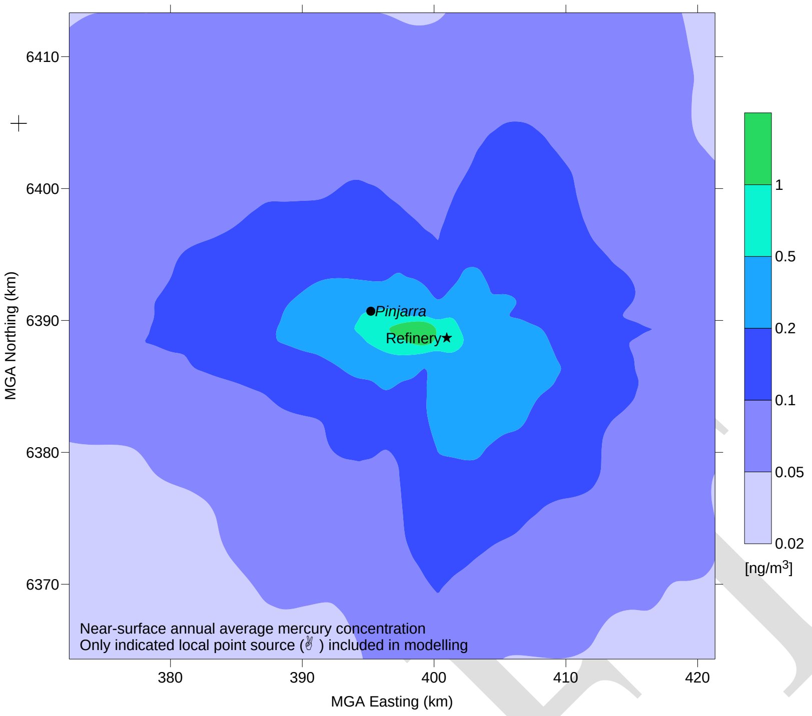


Figure 21. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Pinjarra WA near-source region. Only the indicated point sources of emissions are included in the modelling.

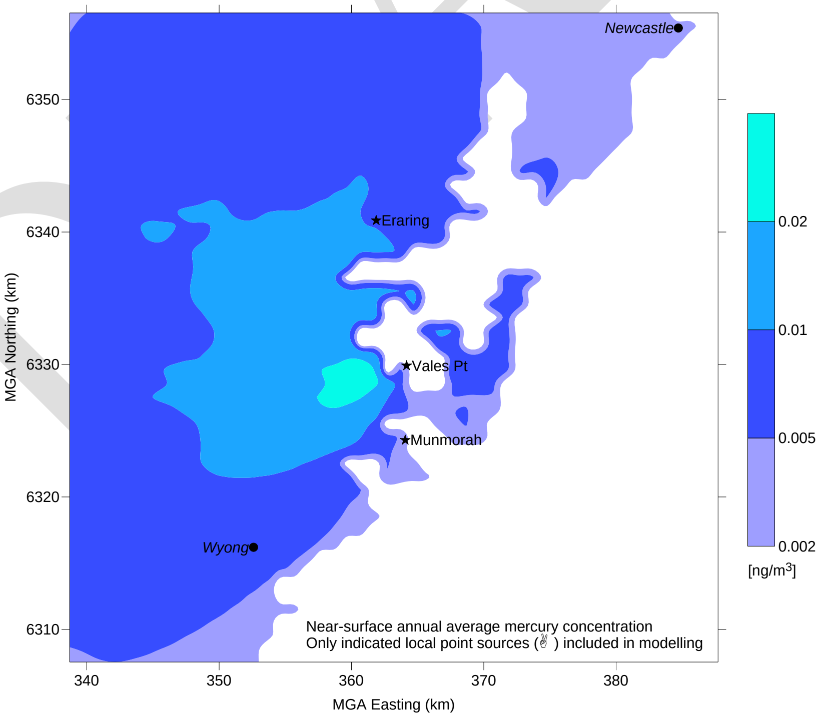


Figure 22. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled NSW Central Coast near-source region. Only the indicated point sources of emissions are included in the modelling.

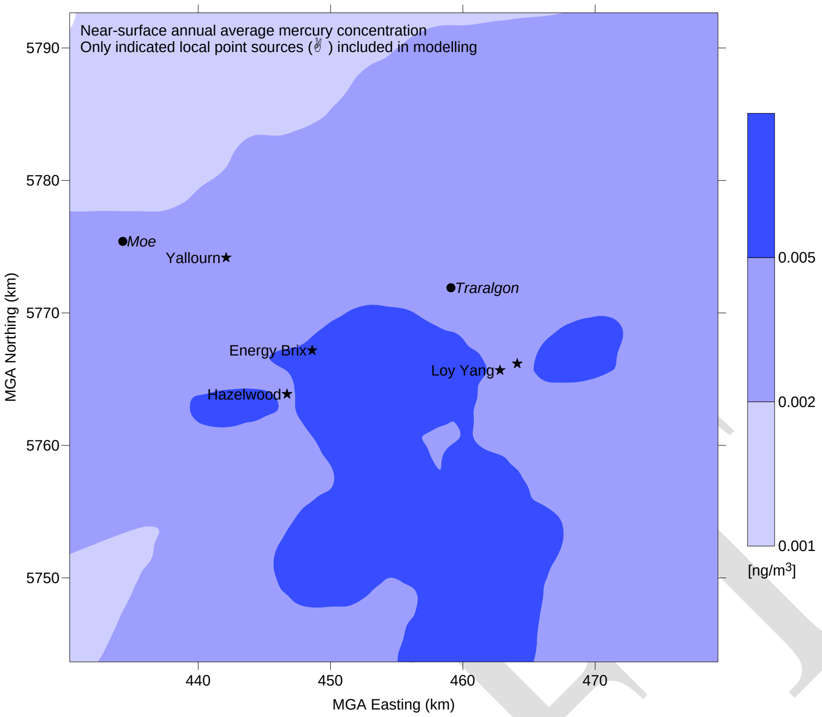


Figure 23. Annual average (2006) total (Hg0 + RGM + Hgp) near-surface ambient mercury concentrations (ng m^{-3}) for the modelled Latrobe Valley near-source region. Only the indicated point sources of emissions are included in the modelling.

5.2 Near-source Wet Deposition

Figure 24 to Figure 28 show the annual wet deposition masses for the five near source modelling domains. It can be seen that the maximum deposited masses range from 5 µg m^{-2} yr^{-1} to over 200 µg m^{-2} yr^{-1}. The spatial distributions of wet deposition mass are markedly different to the concentration distributions. This is because the wet deposition is strongly influenced by rain processes and less dependent on plume height (provided the plume resides within or below a precipitating cloud).

The deposition patterns reflect the prevailing winds associated with rain and also the proximity of elevated terrain. This is evident from Figure 25 for Mt. Isa where the maximum deposition occurs under north-easterly flows. Terrain effects are evident in the case of Pinjarra (Figure 26), NSW Central Coast (Figure 27) and for the Latrobe Valley (Figure 28).

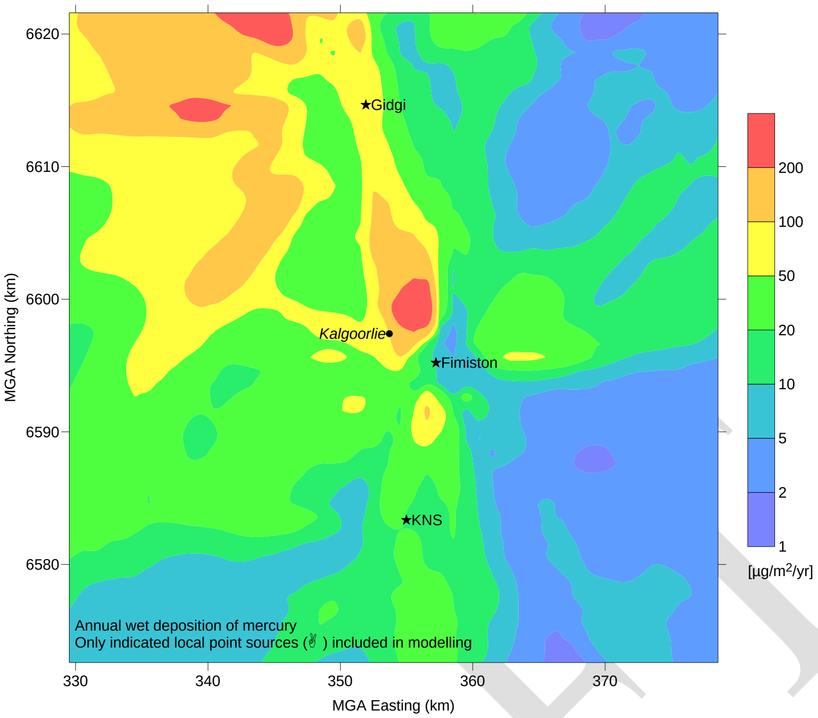


Figure 24. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Kalgoorlie near-source region. Only the indicated point sources of emissions are included in the modelling.

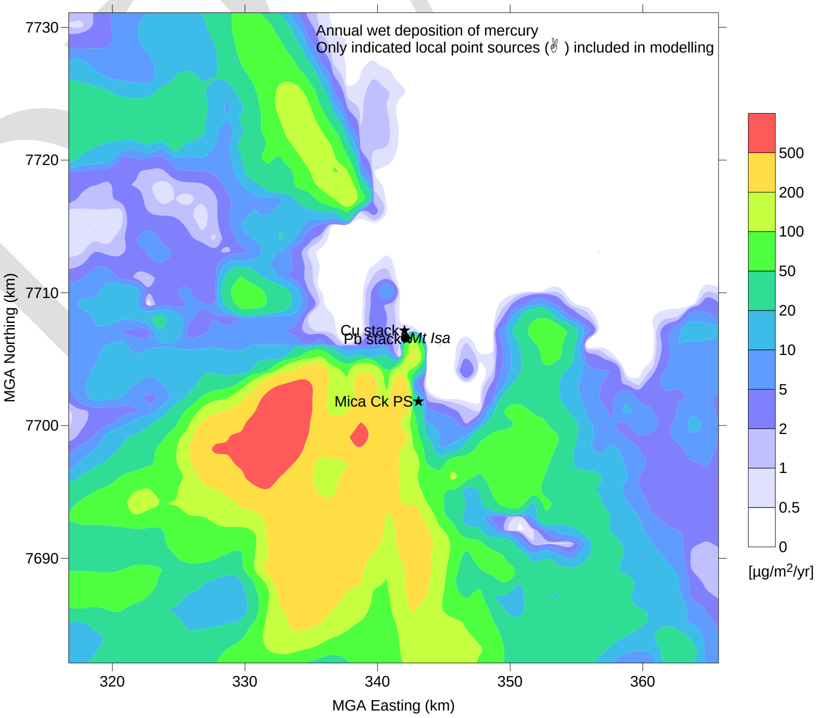


Figure 25. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Mt Isa near-source region. Only the indicated point sources of emissions are included in the modelling.

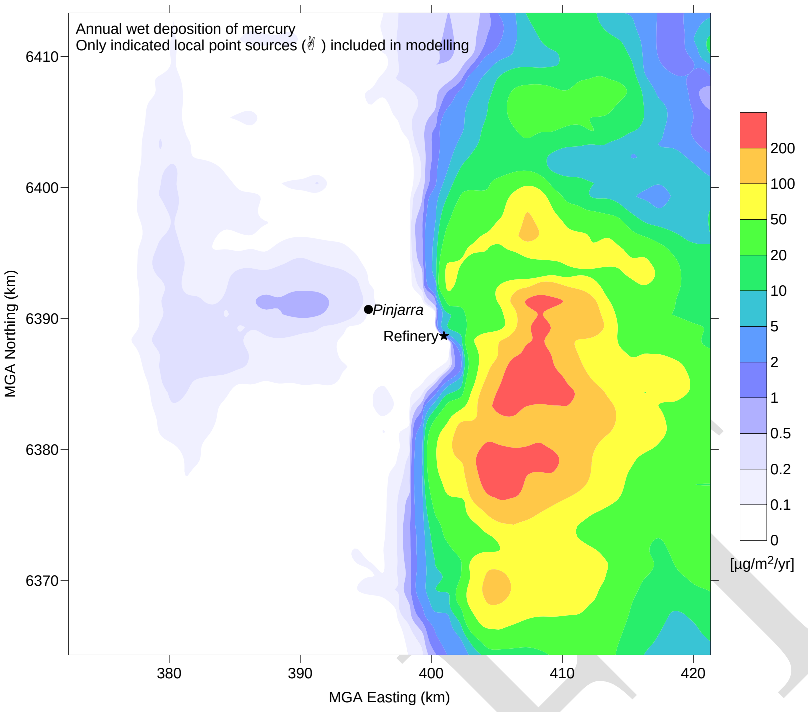


Figure 26. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Pinjarra WA near-source region. Only the indicated point sources of emissions are included in the modelling.

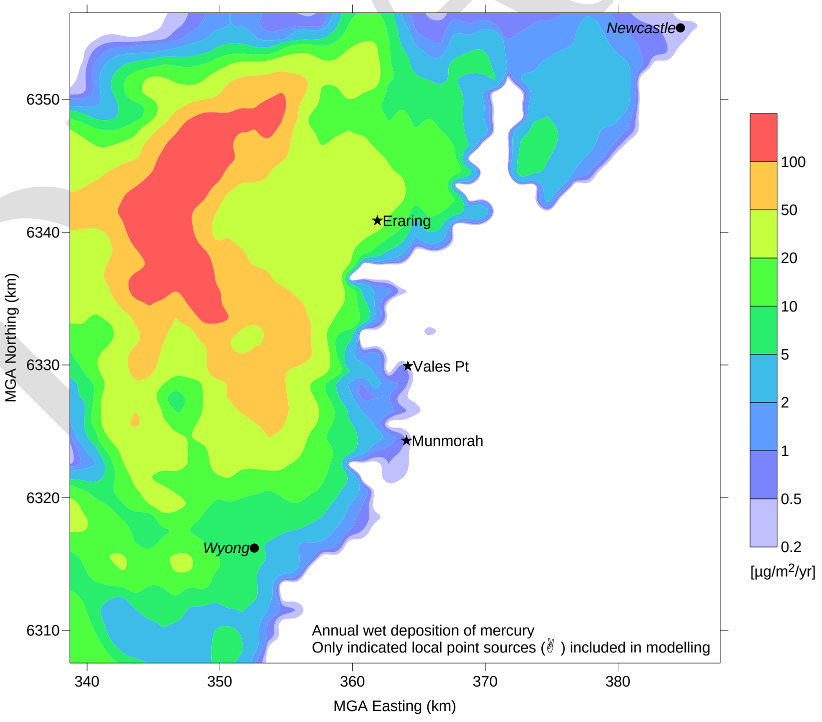


Figure 27. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled NSW Central Coast near-source region. Only the indicated point sources of emissions are included in the modelling.

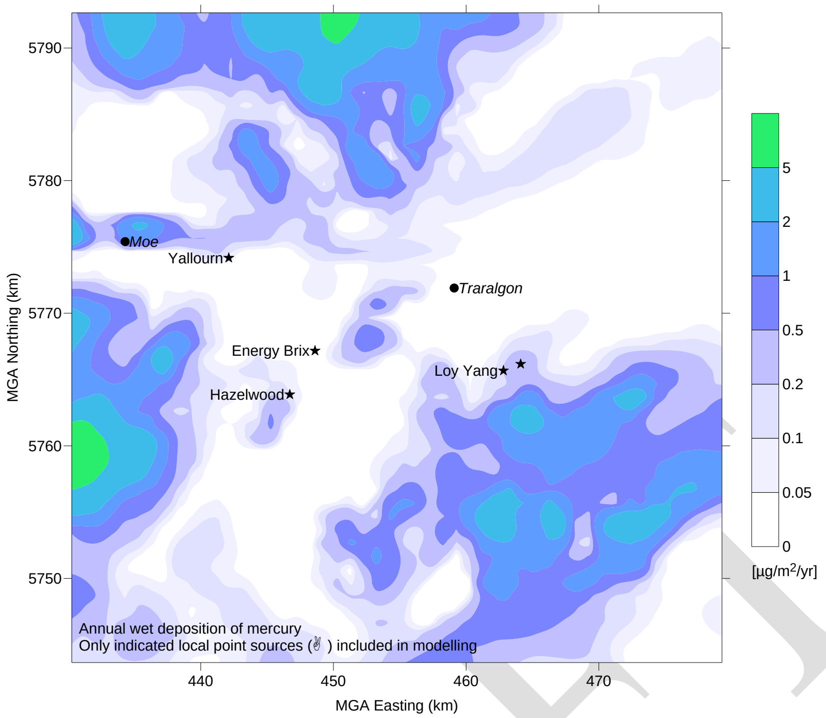


Figure 28. Total annual (2006) mercury (Hg0 + RGM + Hgp) wet deposition (µg m^{-2} yr^{-1}) for the modelled Latrobe Valley near-source region. Only the indicated point sources of emissions are included in the modelling.

5.3 Near-source Dry Deposition

Figure 29 to Figure 33 show the annual dry deposition masses for the five near source modelling domains. The maximum deposited masses range from 1–100 µg m^{-2} yr^{-1}. The spatial distributions of dry deposition mass are similar to the concentration distributions but influenced locally by land use and vegetation differences.

Comparing the near-source dry deposition with the continental scale results shows that the contribution of anthropogenic point source emissions from the electricity generators on the NSW Central coast (Figure 32) and the Latrobe Valley (Figure 33) is small compared to the contribution from natural and background sources. At Pinjarra (Figure 31) the anthropogenic contribution is of similar magnitude to that from the natural and background sources, whereas at Kalgoorlie (Figure 29) and Mt Isa (Figure 30), the near-source contributions are dominant.

Figure 29. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled Kalgoorlie near-source region. Only the indicated point sources of emissions are included in the modelling.

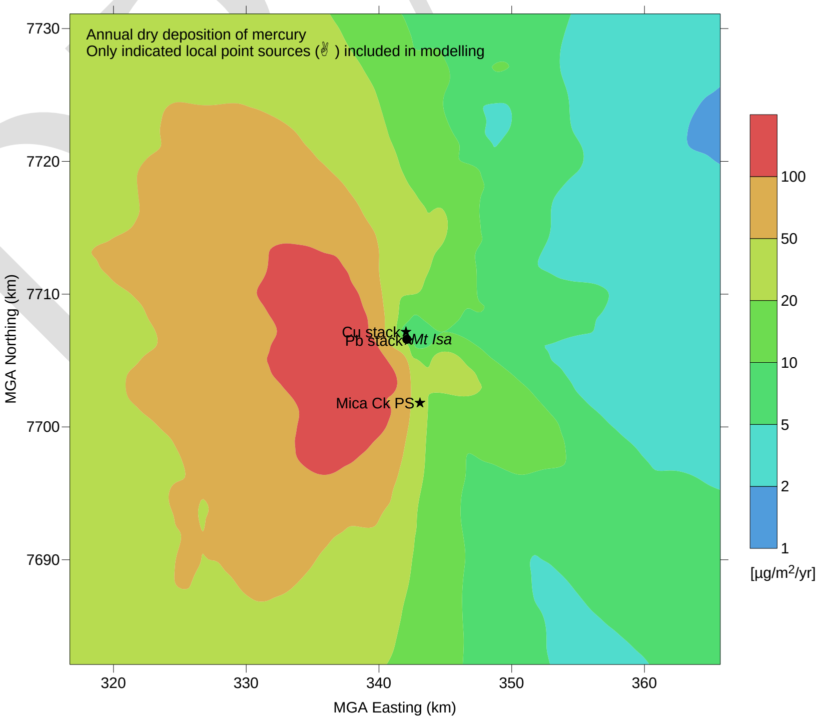


Figure 30. Total annual (2006) mercury (Hg0 + RGM + Hgp)dry deposition (µg m^{-2} yr^{-1}) for the modelled Mt Isa near-source region. Only the indicated point sources of emissions are included in the modelling.

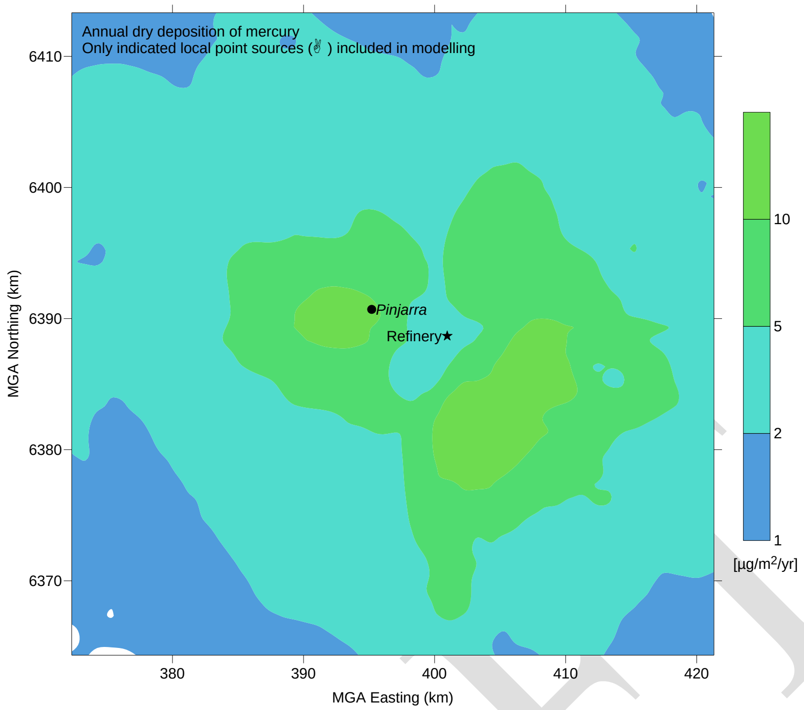


Figure 31. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled Pinjarra WA near-source region. Only the indicated point sources of emissions are included in the modelling.

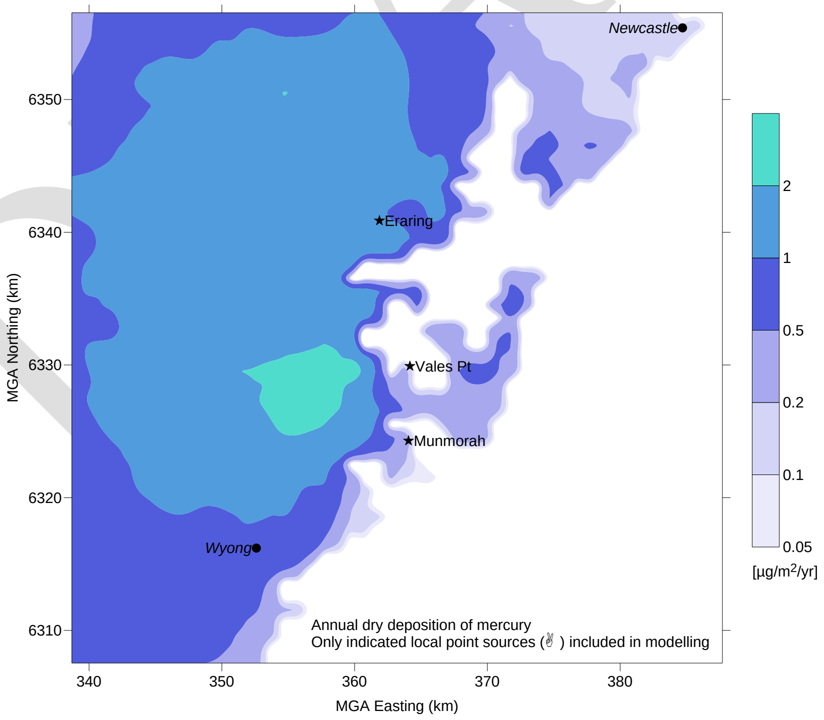


Figure 32. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled NSW Central Coast near-source region. Only the indicated point sources of emissions are included in the modelling.

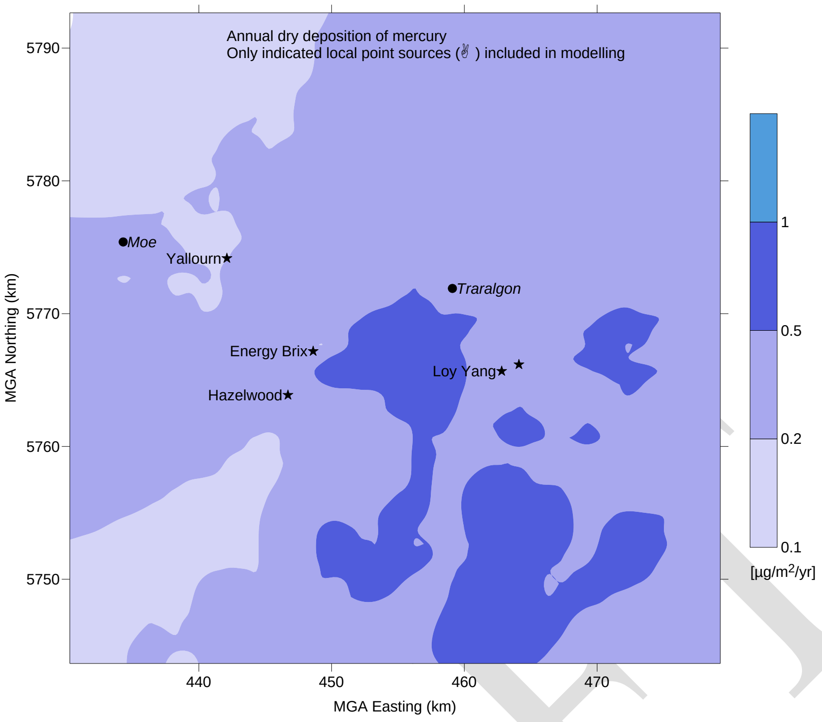


Figure 33. Total annual (2006) mercury (Hg0 + RGM + Hgp) dry deposition (µg m^{-2} yr^{-1}) for the modelled Latrobe Valley near-source region. Only the indicated point sources of emissions are included in the modelling.

6. CONCLUSIONS

In this project we have modelled the concentrations, and the wet and dry deposition of mercury across Australia for 2006. Meteorological and transport modelling of mercury has been undertaken at the continental scale and then focussing on two urban regions and five significant industrial source groups.

The modelling has included best available estimates of natural and anthropogenic sources to estimate total mercury concentrations and deposition loadings. The natural source group considers the emissions from soils, vegetation, water and fires. The anthropogenic source group includes industrial emissions as well as emissions from the commercial, domestic and transport sectors. A mercury concentration of about 1.3 ng m^{-3} was advected into the model domains through the boundaries to represent a global background contribution.

Natural emissions were estimated to contribute 93% of total mercury emissions in Australia with soil emissions being the largest single source (66% of total), followed by bushfires (20%), and vegetation (4%). Industrial sources (6.8%) dominated the anthropogenic emissions with only 0.4% coming from commercial and domestic sources.

Annual average mercury concentrations at the continental scale were dominated by the global background (1.1–1.3 ng m^{-3}), with increases evident at the regional scale in the vicinity of fires and major industrial sources (up to 3 ng m^{-3}). A similar range of concentrations was modelled at the urban scale for Melbourne and Sydney. However, fine scale modelling predicted concentrations to be up to 10 times larger within the first few kilometres of several significant industrial sources.

Wet and dry deposition was also modelled at the three spatial scales. The highest wet deposition masses occur in regions of higher rainfall or regions of local elevated mercury concentrations due to anthropogenic sources or combinations of these two factors. In contrast, dry deposition is generally dominated by natural emissions and the continental background, although enhanced dry deposition masses occur within the vicinity of the bushfires and significant industrial sources.

At the continental scale, wet deposition mass peaks of up to 5 µg m^{-2} yr^{-1} were predicted. The total mercury mass deposited by precipitation onto the Australian land mass is estimated to be about 1.8 t yr^{-1} which is equivalent to about 0.8 % of the total emissions from the region.

At the continental scale, dry deposition masses were generally less than 20 µg m^{-2} yr^{-1} , although values up to 70 µg m^{-2} yr^{-1}) were predicted near the largest industrial source in Kalgoorlie. Dry deposition is calculated to contribute about 21 t yr^{-1} which is equivalent to about 10 % of the emitted mercury from the region.

The urban scale modelling showed significantly higher wet deposition than the continental scale modelling (20 µg m^{-2} yr^{-1}) on the elevated topography surrounding Melbourne; 100 µg m^{-2} yr^{-1} within the Illawarra for the Sydney modelling. This is a result of the higher resolution of the urban modelling (3 km grid spacing vs. 25 km grid spacing for the continental modelling) as well as better resolution of the rain processes.

In contrast to the wet deposition (which is dominated by the local distribution of rainfall), dry deposition in Melbourne is dominated by the natural emissions and the continental background concentrations. Dry deposition peaks of up to 20 µg m^{-2} yr^{-1} were predicted for forested regions to the north-east of Melbourne. These maxima are comparable to the highest wet deposition totals. The natural emission and continental background contribution in the Sydney region is similar to that predicted for Melbourne. However this has proportionally less impact in the Illawarra region dry deposition masses of up to 100 µg m^{-2} yr^{-1} are predicted.

The near-source modelling of wet deposition predicted maximum deposited masses range from 5 µg m^{-2} yr^{-1} to over 200 µg m^{-2} yr^{-1}. The deposition patterns reflect the prevailing winds associated with rain and also the proximity of elevated terrain. In contrast, the maximum deposited masses range from 1–100 µg m^{-2} yr^{-1} with the spatial distributions being similar to the concentration distributions but influenced locally by land use and vegetation differences.

7. REFERENCES

Barrett, D.J., 2002: Steady state turnover time of carbon in the Australian terrestrial biosphere.*Global Biogeochem. Cycles*, 16, 1108.

Bash, J.O., Miller, D.R., Meyer, T.H., Bresnahan, P.A., 2004. Northeast United States and Southeast Canada natural mercury emissions estimated with a surface emission model. *Atmos. Environ*., 38, 5683–5692.

Beeton RJS (Bob), Buckley Kristal I, Jones Gary J, Morgan Denise, Reichelt Russell E, Trewin Dennis (2006 Australian State of the Environment Committee) 2006, *Australia State of the Environment 2006*, Independent report to the Australian Government Minister for the Environment and Heritage, Department of the Environment and Heritage, Canberra.

Cope, M.E., Lee, s.H., Physick, W.L., Abbs D., McGregor, J.L, Nguyen, K.C. (2009) A Methodology for Determining the Impact of Climate Change on Ozone Levels in an Urban Area. Clean Air Research Project 11. Final report to Department of Environment, Water, Heritage and the Arts (DEWHA).

Cope M. E., Hess G. D., Lee S., Tory K., Azzi M., Carras J., Lilley W., Manins P. C., Nelson P., Ng L., Puri K., Wong N., Walsh S., Young M., 2004. The Australian Air Quality Forecasting System. Part I: Project Description and Early Outcomes. *J. Appl. Meteorol.*, 43, 649-662.

DECC, 2007a. Air Emissions Inventory for the Greater Metropolitan Region in NSW, On-Road Mobile Emissions Module: Results, Department of Environment and Climate Change NSW, Sydney, NSW 2000, Australia.

DECC, 2007b. Air Emissions Inventory for the Greater Metropolitan Region in NSW, Domestic-Commercial Emissions Module: Results, Department of Environment and Climate Change NSW, Sydney, NSW 2000, Australia.

Friedli, H.R., Radke, L.F., Lu, J.Y., Banic, C.M., Leaitch, W.R., MacPherson, J.I., 2003. Mercury emissions from burning of biomass from temperate North American forests: laboratory and airborne measurements. Atmospheric Environment 3(2), 253–267.

Gbor, P.K, Wen, D., Meng, F., Yang, F., Zhang, B. and Sloan, J.J. 2006. Improved model for mercury emission, transport and deposition, *Atmos. Environ*, 40, 973-983.

Hurley P., 2008. TAPM v4. Part 1: Technical Description. CSIRO Marine and Atmospheric Research. ISBN: 978-1-921424-71-7. ISSN: 1835-1476.

Kowalczyk E. A., Garratt, J. R., and Krummel, P. B., 1994. Implementation of a soil-canopy scheme into the CSIRO GCM - regional aspects of the model response. CSIRO Div. Atmospheric Research Tech Paper No. 32: 59 pp.

Luhar, A. K., Mitchell, R. M., Meyer, C. P., Qin, Y., Campbell, S., Gras, J. L., Parry, D., 2008: Biomass burning emissions over Northern Australia constrained by aerosol measurements: II-Model validation, and impacts on air quality and radiative forcing. *Atmos. Environ.*, 42, 1647–1664.

Lyon, B.F., Ambrose, R., Rice, G., Maxwell, C.J., 1997. Calculation of soil–water and benthic sediment partition coefficients for mercury. *Chemosphere* 35, 791–808.

McGregor, J. L., 2005. CCAM: Geometric aspects and dynamical formulation [electronic publication]. CSIRO Atmospheric Research Tech. Paper No. 70, 43 pp.

McGregor, J. L., and Dix, M. R. 2001. The CSIRO conformal-cubic atmospheric GCM. In IUTAM Symposium on Advances in Mathematical Modelling of Atmosphere and Ocean Dynamics, P. F. Hodnett (Ed.), Kluwer, Dordrecht, 197-202.

Meyer, C.P., Luhar, A.K., Mitchell, R.M., 2008: Biomass burning emissions over northern Australia constrained by aerosol measurements: I-Modelling the distribution of hourly emissions. *Atmos. Environ*., 42, 1629–1646.

Nelson, P.F., Peterson, C. and A. Morrison, A., 2004. Atmospheric Emissions of Mercury – Sources and Chemistry. *Clean Air & Environmental Quality,* ***36*** *(4), 48–55*

Nelson, P., Morrison, A. Nguyen, H., and Malfroy, H., 2008. Mercury Sources, Transportation and Fate in Australia, Progress Report: RFT 100/0607 Graduate School of the Environment, Maquarie University.

NPI, 1999. NPI Emission Estimation Technique Manual for Aggregated Emissions from Bush fires and Prescribed Burning, Environment Australia.
www.npi.gov.au/handbooks/aedmanuals/pubs/burning_ff.pdf

Packham, D., Tapper, N., Griepsma, D., Friedli, H., Hellings, J., and Harris, S., 2009. Release of mercury in the Australian environment by burning: A preliminary investigation of biomatter and soils. *Air Quality & Climate Change,* ***43*** *(1), 24–27*

Peterson, C., Nelson, P., and Morrison, A., 2004. Quantifying natural and anthropogenic sourced mercury emissions from Australia in 2001. A local scale modelling assessment of transport and deposition patterns for anthropogenic mercury air emissions. Research report (Cooperative Research Centre for Coal in Sustainable Development Australia). Pullenvale, Qld. QCAT Technology Transfer Centre.

Seigneur, C., Karamchandani, P., Lohman. K., and Vijayaraghavan. K., 2001. Multiscale modelling of the atmospheric fate and transport of mercury, *J. Geophy Res., 106*, 27795-27809.

Shetty, S., Lin,C.,Streets, D. and Jang, C., 2008. Model estimate of mercury emission from natural sources in East Asia, *Atmos. Environ*, 42, 8674-8685.

Walker, J., 1981. Fuel dynamics in Australian vegetation, In: *Fire and the Australian Biota,* eds A.M. Mills, R.H. Groves and I.R. Noble. Australian Academy of Science, Canberra.

Xu, X., Yang, X., Miller, D., Helble, J., Carley, R., 1999. Formulation of bi-directional atmospheric–surface exchanges of elemental mercury. *Atmos*. *Environ*. 33 (27), 4345–4355.

The Centre for Australian Weather and
Climate Research is a partnership between
CSIRO and the Bureau of Meteorology.