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**Study on mercury sources and emissions,
and analysis of cost and effectiveness of
control measures
“UNEP Paragraph 29 study”**



Division of Technology, Industry and Economics (DTIE)
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1. Executive Summary

1.1. Introduction

1. The purpose of this report is to provide an overview of mercury emissions to air, control options in selected sectors and their efficiencies and costs. It is prepared with the intention of supporting the on-going negotiation process to prepare a global legally binding instrument on mercury.

1.2. Emissions

2. This report on emissions is based on the comprehensive global inventory for mercury emissions to air which was prepared for the year 2005 (UNEP/AMAP, 2008). The current study includes an update of this inventory with an analysis of information on mercury emissions which has appeared in national submissions and published reports since 2008. In addition to this, an evaluation of global emission trends including a harmonisation of data and improved completeness of previously published inventories was performed for the period 1990 to 2005.

3. The global emissions of anthropogenic mercury to air for 2005 were estimated to be 1921 metric tonnes. The main source category is combustion of fossil fuels in power plants and industrial boilers which contribute about 500 tonnes (26 per cent of the global emissions to air). An additional 380 tonnes of mercury (20 per cent) are associated with fossil fuel combustion for residential heating. Artisanal small scale gold mining contributes about 323 tonnes (17 per cent). Other important sectors are cement production (190 tonnes, 10 per cent), non-ferrous metal industries (excl. gold)(130 tonnes, 7 per cent) and large-scale gold production (110 tonnes, 6 per cent). The remaining 15 per cent originate from iron and steel production, waste management, cremation, chlor alkali industry and mercury production.

4. The global emissions data described above are also available as geospatially distributed (gridded) emission inventories to be used as input for atmospheric modelling purposes. Information on mercury speciation between three main types of mercury/mercury compounds: gaseous elemental mercury, divalent mercury compounds, and particulate associated mercury are also available.

5. A re-analysis of global mercury inventories from 1990, 1995, 2000 and 2005 has been performed to prepare a series of more comparable historical global emission inventories and to allow an analysis of global trends in mercury emissions to air. This re-analysis also included new estimates of emissions from 'intentional use' of mercury for the years 1990, 1995 and 2000. This re-analysis indicate that global emissions have varied from 1967 tonnes in 1990, a slight decrease to 1814 and 1819 tonnes in 1995 and 2000, respectively, to 1921 tonnes in 2005. No major change in global mercury emissions can be seen in this re-analysis. The reason is a significant shift in emissions amounts between regions. While Asian emissions increased in the period 1990 to 2005, emissions in North America and Europe decreased during the same period.

6. A review of recent national emission inventory reports and research papers on mercury emissions concluded that in most cases the new emissions estimates were in the range of emission values reported in the UNEP/AMAP (2008) study.

7. For investigation of emission control options in this study, four sectors were selected based on the size of the emissions and that mercury emitted from these sources mainly originates from fuel, raw materials or waste which is treated or combusted at high temperature. These characteristics to some extent make control options similar and these sectors are thus suitable for a joint study. The selected sectors are coal combustion in power plants and industrial boilers, cement production, non-ferrous metal production (incl. gold) and waste incineration which together contribute about 51 per cent to the total global emissions to air of mercury in 2005.

1.3. Emission control options

1.3.1. General considerations on emission control in all the four selected sectors

8. The magnitude of the mercury emissions from the four selected sectors are dependent on a number of factors including mercury content and other compositional characteristics of fuels and raw materials, the technical configuration of the source facility and the presence of emission control equipment for air pollutants and mercury specific controls. These factors are highly variable in different regions of the world, as well as within regions, which makes identification of generally applicable control options challenging. For identification of suitable control options for a specific plant/facility, a detailed knowledge of the current technical status of this plant/facility is needed. For development of regional or national strategies for reducing mercury emissions, the economic status of the region also needs to be taken into account.

9. One important factor is existing equipment to reduce emissions of other air pollutants (e.g., particles, sulphur dioxide, nitrogen oxides, or other toxic pollutants) which can also reduce mercury emissions. The level of achieved co-control of mercury emission reduction can vary significantly, depending on the characteristics of the input materials and details of the controls. This aspect needs to be taken into account both when assessing current mercury emissions, when considering the need for additional measures and in developing future scenarios and strategies for mercury emission control.

10. Another general aspect applicable to the four sectors is that measures aimed at improved performance and increased energy efficiency will lead to reduced consumption of fuel and raw material and thus indirectly to reduced mercury emissions.

11. Estimating costs for control of mercury emissions requires knowledge of the current status of the emission sources including technological descriptions of current and planned air pollution control. Some degree of mercury emissions control is often achieved when applying controls for the purpose of controlling non-mercury pollutants to meet relevant air pollution standards. In this case, a certain level of mercury emission reduction is achieved at no extra cost.

1.3.2. Coal fired power plants and industrial boilers

12. Pre-combustion measures such as coal washing are often performed to reduce sulphur and ash contents of the coal. A variable fraction of the mercury in the coal is also removed in this operation. Pre-combustion measures have been shown to reduce mercury emissions by an average of 30 per cent. Blending of coal using low mercury coal will reduce emissions correspondingly.

13. Pre-combustion addition of halogens (esp. bromine) has the potential to improve mercury removal by enhancing oxidation of mercury in the flue gas and thus increasing the removal efficiency in downstream particulate matter control and flue gas desulphurisation equipment.

14. Activated Carbon Injection, when used at commercial scale in conjunction with a particle control device, e.g., ESP or fabric filter can produce significant reduction of mercury emissions. Reductions of more than 90 per cent have been seen. Chemically treated carbons (e.g., brominated carbons) are more effective than conventional, untreated activated carbon when treating flue gases containing higher amounts of elemental mercury vapour.

15. Detailed assessment of costs and efficiencies of mercury emissions control requires detailed information on the technical characteristics of the sources as well as on the availability of coal for blending and potential additives. Substantial data is available from field applications in the US.

16. Air pollution control devices may shift mercury from the flue gas to residues such as fly ash or FGD scrubber sludge. Assessments of total control costs should include potential increases in costs from management of these residues due to any potential impacts from changes in the composition of the waste, including increased mercury contents.

1.3.3. Cement production

17. Mercury emitted from cement production originates from coal and other fuels used and raw material such as limestone and other additives. Mercury concentrations are highly variable in fuels and raw materials and significant emission reductions can be achieved by switching to lower mercury fuels and raw materials.

18. Adsorption of mercury to cement kiln dust in cement production is highly dependent on the operating conditions in the plant, where lower temperatures promote adsorption. Removal of mercury from the process can be achieved by removal of dust from the particle control device, if the temperature of the exhaust gases is low. Since cement kiln dust removed from stack gases in cement production to a large extent is recirculated in the process, the use of activated carbon injection methods may require an additional particle removal step where the collected particles are not recirculated.

19. Other end of pipe controls for air pollutants in cement production can be achieved with similar control technologies as for coal fired power plants and significant mercury removal may be expected if flue gas desulphurization is introduced in order primarily to reduce acid gases.

1.3.4. Non-ferrous metal production

20. Non-ferrous metal production is a highly complex industrial process with different configurations depending on which metals are extracted, the characteristics of the ore and which basic process is used. The mercury content of the ore can vary widely. In pyrometallurgical processing of ores, the first step is roasting where sulphur in the ore is converted to sulphur dioxide gas. Any mercury in the ore will be released together with the gas. Large non-ferrous smelters are often equipped with high efficiency air pollution control devices to control particle and SO₂ emissions from roasters, smelting furnaces, and converters. Control of flue gas emissions is achieved by absorption of sulphur dioxide in the sulphuric acid plants, which are commonly a part of the smelting plants.

21. Specific mercury removal (about 90-95 per cent) can be achieved using e.g. the Boliden Norzink technique where mercury is adsorbed in a solution of calomel (mercurous chloride) prior to the sulphuric acid plant.

22. For other thermal units in the gold production industry such as kilns, retorts and furnaces, the best mercury-specific controls are sulphur-impregnated carbon filters, which achieve efficient mercury emission reduction efficiencies in the range of 93 - >99 per cent.

1.3.5. Waste incineration

23. Since all mercury emitted to air from waste incineration originates from mercury in the waste, the most important primary measure is to reduce the input of mercury to the incinerator. This can be accomplished by sorting out mercury containing products from the waste stream prior to incineration.

24. Most modern municipal waste incinerators are equipped with advanced emission control for different pollutants. Some of these devices can be optimised for mercury e.g. by controlling temperature.

25. Mercury specific control can be achieved by carbon injection or scrubbers or both. Examples of costs for Activated Carbon Injection were reported from US EPA from several waste combustion rule processes.

1.3.6. Conclusions

26. A range of technologies to reduce mercury emissions exist and are in use in various countries for each of the sectors examined. Many of these are in place to reduce other air pollutants (e.g., particles, sulphur dioxide, nitrogen oxides, or other toxic pollutants).

27. Existing equipment to reduce such emissions can also reduce mercury emissions. However, the level of mercury emission reduction can vary significantly, depending on the characteristics of the input materials and details of the controls. This aspect needs to be taken into account both when assessing current mercury emissions, in considering the need for additional measures and in developing future scenarios and strategies for mercury emission control.

28. General measures aimed at improved performance and increased energy efficiency will lead to reduced consumption of fuel and raw material and thus indirectly to reduced mercury emissions.

29. Estimating costs for control of mercury emissions requires knowledge of the current status of the emission sources including technological descriptions of current and planned air pollution control.

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List of abbreviations

AMAP	Arctic Monitoring and Assessment Program
ASGM	Artisanal Small scale Gold Mining
BAT	Best Available Techniques
BREF	Best Available Techniques Reference Documents
By product emissions	Emissions occurring from sources where mercury is a constituent in fuel or raw material used in the process.
CCC	Clean Coal Centre of the IEA
CO	Carbon monoxide
ESP	Electrostatic Precipitator
ESPREME	Integrated Assessment of heavy metal releases in Europe. Research project funded by European Union, see reference list
EU	European Union
EU-27	The 27 member states of the European Union
FBC	Fluidized Bed Combustion
FF	Fabric Filter
FGD	Flue Gas Desulphurization
HEIMTSA	Research project funded by European Union, see reference list.
Hg ⁰	Elemental mercury
HgCl ₂	Mercuric Chloride
IEA	International Energy Agency
INC	Intergovernmental Negotiating Committee
Intentional use emissions	Emissions occurring from sources where mercury is intentionally used e.g. in products or as a part of an industrial process.
IPPC	Integrated Pollution Prevention and Control, a EU directive
LHV	Lower heating value (also known as net calorific value, or net CV)
LRTAP Convention, CLRTAP	Convention on Long-Range Transboundary Air Pollution. A convention on air pollution under the UN Economic Commission for Europe (UN ECE)
NH ₃	Ammonia
NO _x	Nitrogen Oxides
PCC	Pulverized Coal Combustion
PM	Particulate Matter
SCR	Selective Catalytic Reduction
SNCR	Selective Non-Catalytic Reduction

SO ₂	Sulphur Dioxide
VOC	Volatile Organic Compounds
NETL	National Energy Technology Laboratory
EPRI	Electric Power Research Institute
POG	Process Optimization Guidelines, see reference list.
O&M	Operating and Maintenance
GAO	Government Accountability Office (USA)
PAC	Powdered Activated Carbon
MWh	Megawatt hour (energy generation)
MWhe	Megawatt hour electricity

2. Introduction

2.1. Background, Scope and Mandate

1. As called for in paragraph 29 of decision 25/5 III of the Governing Council/Global Ministerial Environment Forum of the United Nations Environment Programme (UNEP), this study has been undertaken on various types of mercury-emitting sources and current and future trends in mercury emissions, including an analysis and assessment of the cost and the effectiveness of alternative control technologies and measures.
2. The purpose of the study is to inform the work of the intergovernmental negotiating committee (INC) and, as such, to give background information relevant to the development of measures for controlling mercury emissions that could be featured in the legally binding instrument that is being elaborated.
3. The main objectives of the study are:
 - (a) To present updated and new information on mercury emissions for selected countries and sectors and current trends in mercury emissions;
 - (b) To provide an overview of the technical characteristics of the main sources of mercury emissions in important sectors in the selected countries;
 - (c) To provide a summary of available information on costs and efficiencies of different emission control technologies.
4. The study focuses on the largest emission source categories. These were identified on the basis of previous UNEP emissions reports and information available from published literature. Also considered was the availability of information on costs and effectiveness of alternative control technologies and measures and the extent to which these may be informative for the INC. The identified priority mercury emission sources are coal-fired power plants and industrial boilers; non-ferrous metal production (in particular lead, zinc, copper and gold); waste incineration; and cement production.
5. This report provides a summary of available knowledge on mercury emissions to air; a short description of the sectors selected for this study; information on where mercury enters the processes and where/how it is released to air; and control options and the associated costs.
6. The report is thus intended to serve to inform ongoing discussions of a global agreement on reductions of mercury emissions from the selected sectors. Annexed to this report is guidance for countries who wish to perform more detailed assessment of control options in their countries.

2.2. Sources of information

7. This report is based on reports on global mercury emissions and qualitative assessment of costs and efficiencies of control options prepared for UNEP in 2008, submitted information from individual countries and organizations as well as recent information available from the published literature on emissions, control options and costs.

8. The reports prepared for UNEP in 2008 which were used in this study are; the Global Mercury Emission Inventory for 2005¹ (UNEP/AMAP, 2008) (prepared in cooperation between UNEP and AMAP), and “A general qualitative assessment of the potential costs and benefits associated with each of the strategic objectives set out in Annex 1 of the report of the first meeting of the Open Ended Working Group”². Additional information on control technologies was taken from published literature and results from EU-funded research projects such as ESPREME and HEIMTSA.

9. Countries were requested to provide information on emissions, control options and costs. To facilitate the collection of information, questionnaires and excel templates were circulated. The selected countries for the study were Brazil, China, India, the Russian Federation, South Africa, the United States of America (US) and the 27 countries of the European Union. Follow-up meetings were also held with some governments to assist them in collecting information.

10. Two reviews have been undertaken in the preparation of this report. The Zero Draft report of 4 March 2010 was directly sent to the above mentioned countries and made available publically for review on UNEP’s website. The final draft of 29 September 2010 was sent for review to all stakeholders that had submitted information.

11. A summary of the submitted information is given in Annex 1. National submissions are also available on the UNEP web site.

¹ http://www.chem.unep.ch/mercury/Atmospheric_Emissions/Technical_background_report.pdf

² [http://www.chem.unep.ch/mercury/OEWG2/documents/e52\)/English/OEWG_2_5_add_1.pdf](http://www.chem.unep.ch/mercury/OEWG2/documents/e52)/English/OEWG_2_5_add_1.pdf)

3. Global emissions of mercury

12. The purpose of this chapter is to provide an overview of the global anthropogenic emissions of mercury to the atmosphere, both as total global emissions and from the selected sectors and geographical regions.

3.1. Global anthropogenic mercury emissions to air

13. An inventory of the global anthropogenic emissions of mercury for 2005 was prepared in a joint UNEP/AMAP project in 2008. This inventory was based on nationally reported emissions, and emissions estimated for other countries based on activity data and relevant emission factors. Due to the general lack of direct measurement of mercury emissions, ‘estimating’ global emissions employs essentially the same methods used to derive most nationally reported data. Details on the methods, data sources and other information can be found in UNEP/AMAP (2008)

14. The 2005 inventory addressed ‘unintentional’ and ‘intentional-use’ emission sectors. ‘Unintentional’ mercury emissions to air are mainly from energy production and industrial sectors, where emissions of mercury are an ‘unintentional by-product’ associated with presence of mercury in fossil fuels, such as coal and in raw materials. ‘Unintentional’ emission sectors include: stationary combustion of fossil fuels in power plants and for residential heating³; pig iron and steel production; non-ferrous metal production; cement production; mercury production; large scale gold production; and certain ‘other’ activities. Mercury emissions from the chlor-alkali industry are also accounted in the ‘unintentional’ sector inventory since it is usually reported as a part of national emission inventories and for historical reasons (i.e. consistency with previously published inventories). For some countries, national emission reporting also includes incineration of waste and other sectors which can be defined as intentional use. In most parts of the world, however, emissions from intentional-use sources are mainly estimated using approaches involving substance flow analysis or mass balance estimations (UNEP/AMAP, 2008).

15. ‘Intentional-use’ sectors include artisanal and small-scale gold mining; emissions from cremation of individuals with dental work utilising mercury amalgam; secondary steel production; diffuse releases from product use (e.g. medical instruments, batteries, lamps etc.), and waste disposal (including incineration of mercury-containing waste).

16. As is evident from Figure 1, stationary combustion of coal in power plants and in residential heating and other combustion sources is the largest single source category of anthropogenic mercury emissions to air. Mercury is present in coal as a minor constituent which is released to air during combustion. Combustion of other fossil fuels for energy or heat production also contribute to mercury emissions, but to a significantly smaller extent than coal.

17. Mining and industrial processing of ores, in particular in primary production of iron and steel and in non-ferrous metal production (especially copper, lead and zinc smelting), release mercury as a result of both fuel combustion and mercury being present as impurities in ores. Metal production sources of mercury also include mining and production of mercury itself (a relatively minor source) and production of gold, where mercury is both present in ores and

³ In many countries, large amounts of (often poor quality) coal are used for heating and cooking in individual households. In other countries, boilers may serve a larger number of residential units.

used in some industrial processes to extract gold from lode deposits. Use of mercury to extract gold in artisanal and small-scale gold mining (ASGM) operations is considered an intentional use.

18. The third major source of ‘unintentional’ releases of mercury is associated with cement production. In cement production, emitted mercury originates from its presence in the fuel used to heat the cement kilns (mainly coal) together with any co-combusted fuels or waste, and the limestone and other minor raw materials and additives. Variations in origin as well as magnitude of mercury emissions from cement production can thus be considerable.

19. Intentional-use sectors in the UNEP/AMAP (2008) inventory include artisanal and small-scale gold production, which gives rise to large emissions both to air and discharges to water. Unlike most other sectors, discharges to water associated with ASGM activities are as large as or larger than the air emissions and may result in significant environmental impacts on the local scale. Other intentional-use emissions estimated in the UNEP/AMAP (2008) inventory include emissions from waste incineration and other waste disposal, emissions from dental mercury following human cremation (but not currently including releases during the preparation of dental amalgam), and diffuse releases from use of mercury in products such as batteries, lamps and measuring devices.

20. The estimated global anthropogenic emissions of mercury to air in 2005 from the various sectors are presented in Table 1 and Figure 1.

21. In Table 1 the emissions and the relative contribution to global mercury emissions to air in 2005 from the sectors selected for consideration in this current study are presented in the top five rows. In total, these five sectors account for some 50 per cent of the estimated global emissions, with coal combustion in power plants contributing 26 per cent; non-ferrous metal production (excluding gold production) contributing 7 per cent; large-scale gold production 6 per cent; cement production contributing 10 per cent; and large scale waste incineration contributing 2 per cent. These sectors were prioritised in the current study due to their respective relative contribution to global emissions of mercury to air and because they are sectors largely consisting of point sources with high temperature combustion or processes where installation and use of similar emission control technologies is feasible.

22. Both nationally reported and ‘estimated’ emissions are associated with large uncertainties. For ‘unintentional use’ sectors, uncertainties in global estimates of the order of ± 25 per cent for coal combustion and ± 30 per cent for ferrous/non-ferrous metal and cement production are quoted, although these may not reflect the uncertainties in estimates for individual countries. The uncertainties vary between countries and continents depending on availability and reliability of statistical data on activities and consumption of mercury, as well as information that allows the application of appropriate emission factors (e.g. mercury content in fuel, control technologies). For sectors such as gold production and emissions from waste incineration and waste disposal, the uncertainties are significantly greater and the high end estimates presented in UNEP/AMAP (2008) are to be regarded as maximum potential emission. The “conservative” estimates represent the most reasonable estimate of the emissions from the ‘intentional-use’ sectors presented in the UNEP/AMAP (2008).

Table 1. Estimated global anthropogenic emissions of mercury to air in 2005 from various sectors (revised from UNEP/AMAP, 2008). Selected sectors for this study in first five rows, below various other sectors.

Sector	Emissions in 2005* (tonnes=1000 kg)	Per cent contribution of total emissions to air
Coal combustion in power plants and industrial boilers	498 (339-657)	26
Non-ferrous metals (Cu, Zn, Pb)	132 (80-185)	7
Large scale gold production	111 (66-156)	6
Cement production	189 (114-263)	10
Waste incineration	42	2
Residential heating/other combustion	382 (257-506)	20
Artisanal and small-scale gold production	323	17
Other waste	74	4
Pig iron and steel, sec. steel	61 (35-74)	3
Chlor alkali industry	47 (29-64)	2
Dental amalgam (cremation)**	27	1
Other	26	1
Mercury production	9 (5-12)	0.5
Total	1921	

*Represents best estimates: estimate (uncertainty interval), or conservative estimate (no associated range). See UNEP/AMAP (2008) for discussion on uncertainties.

** Does not include other releases from production, handling, use and disposal of dental amalgam.

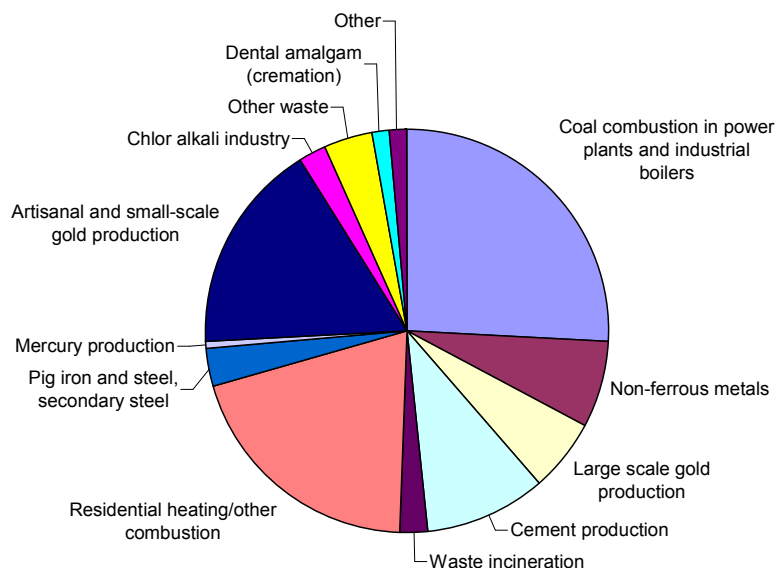


Figure 1. Proportion of global anthropogenic emissions of mercury to air in 2005 from various sectors (revised from UNEP/AMAP, 2008).

23. The global inventory of anthropogenic emissions to air for 2005, described in UNEP/AMAP (2008) and summarized in UNEP (2008) was the most comprehensive such (global) inventory developed thus far. A revision undertaken in 2010 re-examined this inventory together with previously published inventories for the nominal years 1990, 1995 and 2000; for 2005 this revision resulted in some small changes in the global total emissions of mercury to air, essentially concerning the intentional-use sectors as a result of improved information on mercury consumption in related sectors. This is further described in chapter 3.2 below.

24. Of the total estimated mercury emissions to air in 2005 of 1921 tonnes, combustion of fossil fuels in power plants and industrial boilers contributes about 500 tonnes (26 per cent). An additional 380 tonnes of mercury (20 per cent of global emissions) are associated with fossil fuel combustion for residential heating (UNEP/AMAP, 2008, AMAP, 2010). In many countries, large amounts of (often poor quality) coal are used for heating and cooking in individual households. In other countries, boilers may serve a larger number of residential units. Although a significant source of mercury emissions, residential heating is not further covered in the current study as suitable emission control options are very different from power plants and industrial boilers. Alternative controls such as fuel switch or conversion to other sources of energy for cooking and heating should be considered, depending on local conditions and availability of resources.

25. Almost 70 per cent of the total mercury emissions from non-ferrous industries originate in Asia (see section 2.2. for further details on emissions by geographical region). Non-ferrous metal industries (from large-scale operations) are estimated to contribute about 130 tonnes of the total mercury emissions to air worldwide in 2005, with about 50 per cent of the estimated emissions from this sector coming from China, followed by South Korea (about 5 per cent) Chile, and Australia each also emit about 5 per cent. Large-scale gold production represents 6 per cent of the global emissions (about 110 tonnes). In 2005, China was estimated to be the largest emitter with 40 per cent of the sector emissions, followed by Australia, the United States and Indonesia all contributing between 5 and 7 per cent of the sector emissions. UNEP has received information recently indicating that China has stopped the use of mercury in large-scale gold production.

26. About 190 tonnes (10 per cent) of the total global emissions of mercury to air from anthropogenic sources in 2005 originate from the manufacture of cement. About 74 per cent of the total mercury emitted from the cement sector are emitted in Asia, with China the main contributor, responsible for 45 per cent of total emissions to air from cement manufacturing followed by India (6 per cent) and Japan (5 per cent). The United States is the next largest contributor (3 per cent) (UNEP/AMAP, 2008).

27. Mercury emissions from household and other wastes are highly uncertain. Most countries dispose of large amounts of waste in landfill, or by open-burning. Many countries do not have facilities for large-scale high-temperature incineration of solid waste. Emissions from this sector are therefore often either not reported, or are included under a general category of 'other' emissions. Countries that do report emissions from large scale municipal, medical and hazardous waste incineration facilities include the United States, Japan and countries in Europe – and for these reasons they may be somewhat 'over-represented' in the emissions for this part of the waste sector inventory. In other countries, waste incineration may be accounted under the category 'waste and other' in the global inventory. Waste sector emissions calculated on the basis of substance flow analysis or mass balance approaches, together with information on waste treatment practises within countries can be compared with reported estimates of waste

incineration, where these are available. Under the most recently revised estimates (AMAP, 2010) for emissions from intentional-use sectors in 2005 (see para. 30), incineration of municipal solid waste in large-scale facilities is estimated to contribute approximately 2 per cent of the global mercury emissions to air (with a further 4 per cent accounted under the 'other waste' category). Noting the uncertainties in allocating emissions to different parts of the waste sector, North America and Europe each contribute about 20 per cent of the mercury that is directly associated with large-scale waste incineration worldwide and Asia about 45 per cent.

28. For atmospheric modelling purposes, geospatially distributed (gridded) emission inventories are used as input. In addition to this, models also require that these emissions are 'speciated', to divide primary emissions between three main types of mercury/mercury compounds: gaseous elemental mercury (GEM, also abbreviated as Hg⁰ or Hg⁰), divalent mercury compounds (Hg²⁺), and particulate associated mercury (Hg-P); together these comprise the total mercury (HgT) emissions. The different mercury species have distinctly different transport behaviour and information on speciation is thus a requirement for determining the fate of emitted mercury. Information on speciation and stack heights is available in the AMAP/UNEP (2008) report.

3.2. Trends in global mercury emissions to air

29. As part of its 2010 assessment of mercury in the Arctic, the Arctic Monitoring and Assessment Programme (AMAP) undertook a re-analysis of global mercury inventories from 1990, 1995, 2000 and 2005 (Pacyna and Pacyna, 2002; Pacyna and Pacyna, 2005; Pacyna et al., 2006) in an attempt to prepare a series of more comparable historical global emission inventories and to allow an analysis of global trends in mercury emissions to air (AMAP, 2010). This re-analysis employed a common methodology, a more consistent information base for estimating certain emissions, and updating of the earlier inventories for various countries and sectors to account for improved knowledge gained during the process of preparing the inventories over the last 15 years. It also involved further revising the 2005 inventory using newly available data on regional mercury consumption that form the basis for estimates of emissions associated with 'intentional-use' sectors. Details of the revisions can be found in the AMAP, 2010 report.

30. For 2005, the revision resulted in an estimate approximately 5 tonnes lower, or 1921 tonnes compared to the 1926 tonnes of global mercury emissions to air estimated in the UNEP/AMAP (2008) report. 'Unintentional' emissions were essentially unchanged from those presented in the UNEP/AMAP (2008) report; however, emissions estimates for particular countries from 'intentional-use' sectors were revised somewhat. In addition to revising the emissions from the 'unintentional' emissions sectors for all years (1990, 1995, 2000 and 2005), the re-analysis introduced estimates of emissions from 'intentional use' of mercury for the years 1990, 1995 and 2000, which (apart from emissions from the chlor-alkali industry) were previously not available for these 'historical' inventories.

31. Revised estimates of emissions of mercury to air in 1990, 1995, 2000 and 2005 from 'unintentional emissions' and 'intentional-use' sectors are presented in Figure 2. It is important to note that the data presented for emissions from 'intentional-use' sectors represent low end 'conservative estimates'.

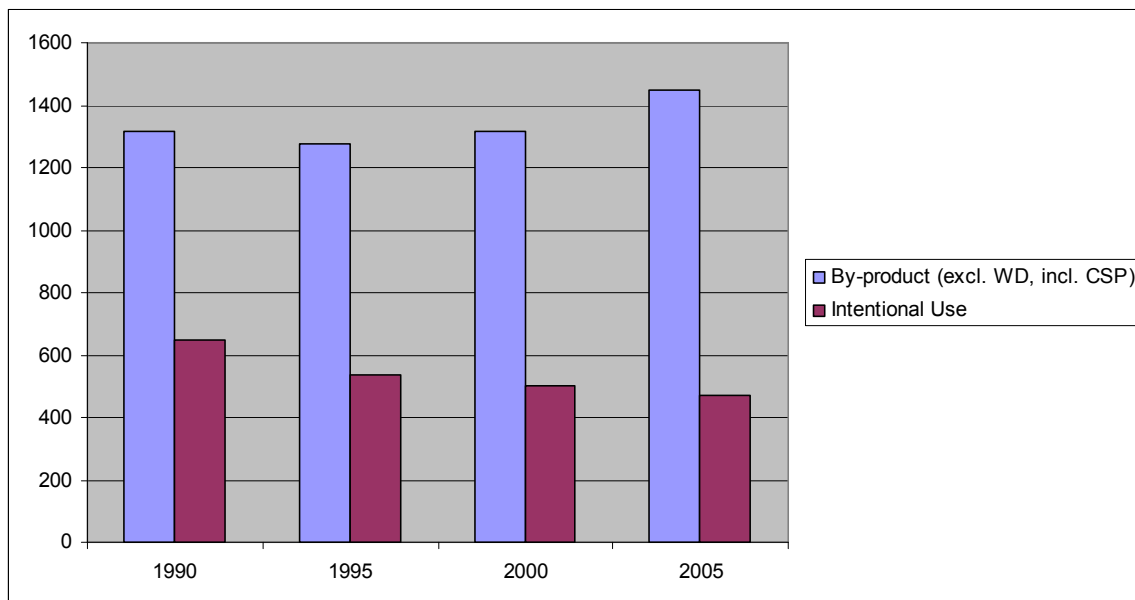


Figure 2. Revised estimates of total global anthropogenic mercury emissions to air (tonnes) from ‘by-product’ and ‘intentional-use’ emission sectors in 1990, 1995, 2000 and 2005. (WD = Waste Disposal, CSP = Caustic Soda Production) (Source: AMAP, 2010)

32. In Figure 2 the left bar for each year shows the revised estimates of ‘by-product’ emissions for the respective years. The right bar presents the estimated emissions of mercury to air from the ‘intentional-use’ sectors. According to the revised estimates, ‘by-product’ emissions have increased somewhat from 1990 to 2005, while the ‘intentional-use’ sector emissions have decreased over the same period of time. Overall, when adding ‘unintentional’ and ‘intentional use’ sector sources the total level of mercury emissions to air on the global scale has been relatively stable from 1990-2005.

3.2.1. Emissions by geographical region

33. Even though the total level of global emissions of mercury to air has been relatively stable since 1990, there has been a considerable regional shift in where the emissions originate. Regional trends in (combined) emissions from ‘unintentional’ and ‘intentional-use’ sectors for 1990, 1995, 2000 and 2005 are summarized in Figure 3. The figure shows that anthropogenic mercury emissions to air have increased substantially in Asia, and to a much lesser extent in Africa and South America, while emissions in Europe, Europe-Asia (Russia) and North America have decreased from 1990 to 2005.

34. According to the UNEP/AMAP (2008) report, in 2005, Asian countries contributed about 67 per cent to the global mercury emissions to air from anthropogenic sources, followed by North America and Europe (UNEP/AMAP 2008). Russia, with its contribution of about 4 per cent to global emissions is considered separately due to its territories in both Europe and Asia.

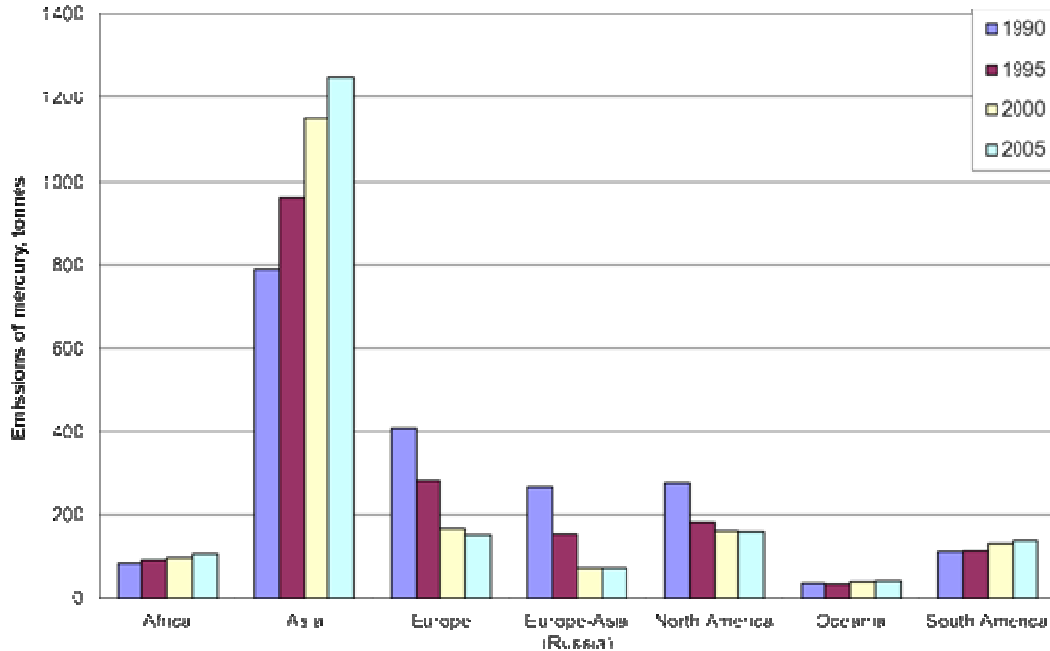


Figure 3. Revised estimates of anthropogenic mercury emissions to air (tonnes) in 1990, 1995, 2000 and 2005 from different continents/regions. (Source: AMAP, 2010)

35. China, with its large number of coal-fired power plants, was estimated to be the largest single emitter of mercury to air worldwide in 2005. Power plant emissions are only a part of the total combustion emissions of mercury in China. Equally significant are emissions from combustion of poor quality coal mixed with various kinds of wastes in small residential units to produce heat and cook food in rural areas. Emissions from the power sector in China have likely decreased since 2005 (see section 3.3.2)

36. Together, three countries, China, India, and the United States were responsible for 57 per cent of the total estimated global emissions of mercury to air in 2005 (1097 out of 1921 tonnes).

37. Figure 4 illustrates the global distribution of anthropogenic emissions of mercury in 2005 following their geospatial distribution according to the methodology described in UNEP/AMAP, 2008.

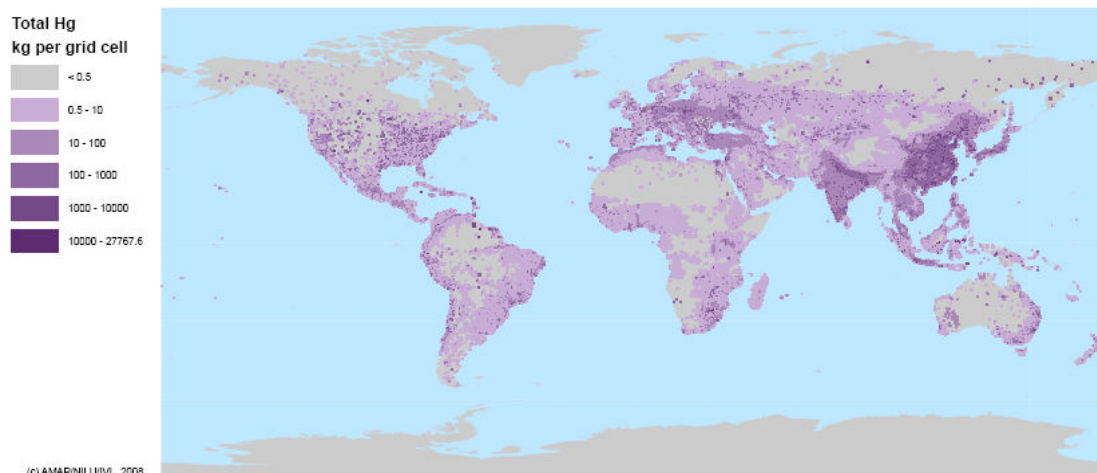


Figure 4. *Spatially distributed emissions of mercury (from UNEP/AMAP, 2008).*

3.3. Comparison of information on emissions

3.3.1. Comparison of available global emission studies

38. The UNEP/AMAP (2008) inventory of global anthropogenic emissions for the year 2005, can be compared with inventories presented in other such assessments, such as that of global emissions of mercury from anthropogenic and natural sources by Pirrone et al (2010). That inventory estimates global anthropogenic emissions to air at 2320 tonnes, based on data from different countries for the years 2003-2006, including several of the same sources used in the development of the UNEP/AMAP inventory for 2005 and, for Europe, Russia and South America, the UNEP/AMAP estimates. The total inventory (2320 tonnes) is well within the range of the global inventory estimated in the UNEP/AMAP (2008) report, 1221-2950 tonnes (best estimate 1921 tonnes), as are the estimates for some individual sectors (see Table 2). For example, Pirrone et al (2010) estimated emissions of 810 and 236 tonnes, respectively, for stationary combustion of coal (power stations and residential heating combined) and for cement production, compared with the UNEP/AMAP estimate of 878 (597-1163) and 189 (114-263) tonnes. For non-ferrous metal production, however, the agreement is not as good (310 tonnes compared with the UNEP/AMAP estimate of 132 (80-185) tonnes. Similarly, for the chlor-alkali industry, the estimate presented by Pirrone et al (2010) of 163 tonnes, is significantly higher than that presented in the UNEP/AMAP work of 47 (29-64) tonnes. There are also differences in the emissions estimates presented for individual countries. In some cases these can partly be explained by use of different emission factors. For example, the mercury content of coal is highly variable and in the absence of measured concentrations different assumptions of mercury content in coal are made for different inventories.

Table 2. *Estimated global anthropogenic emissions of mercury to air in 2005 from selected sectors – comparison of global inventory compilations (tonnes)*

Selected sectors	UNEP/AMAP 2008 (revised)*	Pirrone et al (2010)	Hylander and Herbert (2008)
	tonnes	tonnes	tonnes
<i>Stationary combustion</i>			
Coal combustion in power plants and industrial boilers	498 (339-657)	810 (of which 747 from coal combustion in power plants)	
Residential heating/other combustion	382 (257-506)		
<i>Non-ferrous metal production</i>			
Non-ferrous metals (Cu, Zn, Pb)	132 (80-185)	310	275
Large scale gold production	111 (66-156)	-	
<i>Cement production</i>			
Cement production	189 (114-263)	236	
<i>Waste incineration</i>			
Waste incineration	42	187 (waste disposal)	
Other waste	74		
<i>Other major sectors</i>			
Pig iron and steel, sec. steel	61 (35-74)	43	
Artisanal and small-scale gold production	323	400	
Chlor alkali industry	47 (29-64)	163	
Overall inventory**	1921 (1221-2950)	2320	

* Represents best estimates: estimate (uncertainty interval), or conservative estimate (no associated range). See UNEP/AMAP (2008) for discussion on uncertainties.

** Includes other sectors not listed in rows above.

39. Hylander and Herbert (2008) estimated that about 275 tonnes of mercury were emitted globally in 2005 from the non-ferrous pyrometallurgical treatment of copper (Cu), lead (Pb), and zinc (Zn) sulphide ores, with almost half emitted from Zn-smelters and the rest equally divided between Cu and Pb smelters. Again, this is higher than the emission estimate reported in the UNEP/AMAP (2008) inventory, and can be attributed to use of higher emission factors than those applied in the UNEP/AMAP work (but lower than the emission factors used in some of the work presented in the Pirrone et al (2010) inventory). The assessments by Pirrone et al (2010) and by Hylander and Herbert (2008) indicate that the contribution of the non-ferrous metal sector in particular to global emissions to air may be even higher than that estimated in the 2008 UNEP/AMAP assessment, and this in part may be related to emissions associated with small-scale smelting operations in countries such as China.

40. Comparing inventories is also complicated by differences in the included sectors, for example, large scale gold mining is not separately categorised in the emissions sectors presented in the Pirrone et al (2010) inventory, but they do include emissions from coal bed fires (32 tonnes) and vinyl chloride monomer (VCM) production (24 tonnes) that were not included in the UNEP/AMAP inventory (the former was considered non-anthropogenic and the latter was excluded due to lack of adequate information on this source).

41. The UNEP/AMAP (2008) report includes a more detailed description of some of the comparisons presented above; but all of these comparisons point to the basic need for better information on key parameters such as mercury content of coal, appropriate emission factors (for different applied technologies, etc.), statistical data on mercury consumption and use, and activities in different countries as well as control measures applied. For countries where global inventories indicate large mercury emissions, the lack of assured national research or statistical data to support the verification of emission estimates remains an obstacle to reducing the uncertainties associated with these emissions estimates, although recently reported research on the coal fired power plant sector submitted to UNEP has shown significant progress (MEPC, 2010).

42. Much of the work under the current study, aimed to obtain improved information of this type, and other relevant information obtained from several countries, is discussed in the following section.

3.3.2. Comparison of inventories in individual countries

43. In recent years several countries have produced their own national inventories of mercury emissions to air, and made these available to UNEP for use in the current study. Some of these are discussed below, including comparisons with the emission estimates presented in the UNEP/AMAP (2008) report.

Australia

44. For Australia, Nelson et al, (2009) estimated the anthropogenic emissions of mercury to air to be around 15 tonnes in 2006, which are less than half the emissions estimated in the UNEP/AMAP inventory for 2005. Gold production, contributing around 7.6 tonnes is identified in both the national and global inventory as a major contributor to Australian anthropogenic emissions. Coal combustion in power plants is identified as the second largest source in the national inventory, contributing around 2.3 tonnes, whereas the UNEP/AMAP inventory puts the emission at nearer 8.8 tonnes as a result of use of differing mercury content in coal; coal combustion for residential heating is not specified in the Nelson et al (2009) inventory. Other major sources in Australia are alumina production from bauxite (1.9 tonnes) and mining and smelting in non-ferrous metal production (other than gold smelting; 0.89 tonnes), which is again lower than the UNEP/AMAP estimate of around 6 tonnes per year for these sectors. Emission estimates from activities such as coke production (0.5 tonnes), the chlor-alkali industry (0.34 tonnes) and emissions from cement and lime production (0.31 tonnes) are comparable with those in the UNEP/AMAP inventory.

Canada

45. The Canadian national report reports emissions from coal-fired power plants, non-ferrous metal production and cement production of 2.0, 1.4 and 0.3 tonnes, respectively, out of a total emission of approximately 7.1 tonnes in 2007. These data, based on the National Pollution Release Inventory (NPRI) for 2007 are similar to the values that were reported in the UNEP/AMAP inventory for 2005 which were based largely on the NPRI 2005 data and estimated total Canadian emissions in 2005 at 8.0 (4.0-12.0) tonnes. In the UNEP/AMAP work, emissions from waste incineration in 2005 were estimated at 0.8 tonnes based on the mass-flow approach, compared to the reported value of 1.1 tonnes in 2007. The NPRI data indicate slightly decreasing emissions from coal-fired power plants and non-ferrous metal production and slightly increasing emissions from cement production between 2005 and 2007.

China

46. While detailed national emission estimates for China are lacking, various relevant studies have recently been published. Wu et al, 2006, estimated the anthropogenic emissions of mercury in China to be 696 tonnes in 2003, with an uncertainty interval of ± 307 tonnes. This can be compared to the 794 tonnes (range 477-1113 tonnes) estimated in the UNEP/AMAP report. Li et al (2010) estimate mercury emissions from primary zinc production in China at 80.7 – 104.2 tonnes per year (for the period 2002-2006). In Streets et al (2009) data for coal combustion was updated and extrapolated to conditions in 2005, resulting in estimated mercury emissions from coal combustion in power plants and industry amounting to almost 295 tonnes in 2005. In a UNEP study prepared for the Ministry of Environmental Protection of China by Tsinghua University, Beijing (MEPC in prep, 2010), mercury emissions from coal combustion in coal-fired power plants in 2005 are preliminary estimated at 108.6 tonnes (confidence interval 65.2 – 195.4); the UNEP/AMAP (2008) estimate for this sector is 195 (146-243 tonnes) also for 2005. Wu et al's (2006) estimate for emissions from coal combustion in power plants and industry (combined) was 225 tonnes (similar to the UNEP/AMAP estimate for this combination). They also estimated emissions from large scale non-ferrous metals smelting at 290 tonnes (considerably higher than the UNEP/AMAP estimate of 77-143 tonnes for non-ferrous metal with large-scale gold production included), cement production at 35 tonnes (lower than the UNEP/AMAP estimate in the range 59-110 tonnes) and household waste burning approximately 10 tonnes.

India

47. In connection with the Mercury Air Transport and Fate Research Partnership area activities emissions of mercury to air from industrial sources in India were estimated by Mukherjee et al (2009) for the years 2000 and 2004. Due to scarcity of country specific information the estimates were based on emission factors for the European Union (EU), for the U.S., from the literature and from the limited information received from India. According to these estimates, emissions decreased from 321 tonnes in 2000 to 253 tonnes in 2004, due to strongly decreasing emissions from chlor-alkali plants; this can be compared with the UNEP/AMAP estimate of 180 (108-252) tonnes in 2005. India is a large coal producer and the largest source of emissions of mercury to air in India in 2004 was estimated by Mukherjee et al (2009) to be coal fired power plants, at around 121 tonnes. The difference between this estimate and the UNEP/AMAP (2008) estimate for 2005 (52-87 tonnes) is largely explained by the fact that Mukherjee et al used a higher value for mercury in coal (0.376 mg/kg) than that used in the UNEP/AMAP study. Kumari (2010) has estimated an increase of mercury emissions from Indian thermal power plants from 95 to 112 tonnes between 2006 and 2008, with an interval of uncertainty from 59-200 tonnes for 2008. According to Mukherjee et al (2009), non-ferrous metal production, where several different technologies are used for copper, lead and zinc-production, contributed 15.5 tonnes of mercury to air in 2004 in India, an increase from around 8 tonnes in 2000. Kumari (2010) estimated emissions from non-ferrous smelters to be in the interval 7.6 - 21.7 tonnes in 2007; the UNEP/AMAP estimate for 2005 is 3-5.6 tonnes. The cement industry in India, which according to Mukherjee et al (2009) is the second largest in the world, was estimated to emit around 4.7 tonnes of mercury in 2004, lower than the 8-14.8 tonnes estimated in the UNEP/AMAP report. In India, some Portland cement plants are quite modern and advanced. Mukherjee et al (2009) point out that the energy consumption in cement production is quite high, but it was not known if any plants use wastes as alternative fuel. Municipal solid waste in India is generally disposed of in landfills, by open dumping and open burning. Medical waste is incinerated at some hospitals, but the information was not complete.

Republic of Korea

48. An emission inventory for the Republic of Korea for 2007 (Kim et al 2010) resulted in an estimated total mercury emission to air of 12.8 tonnes (range 6.5-20.2 tonnes), where 54.8 per cent originated from industrial sources, 45.0 per cent from stationary combustion sources and 0.02 per cent from mobile sources. Major emitters were; thermal power plants (26 per cent), oil refineries (25 per cent), cement kilns (20 per cent) and waste incineration (municipal, industrial, medical and sewage sludge) (20 per cent). Emissions from non-ferrous metal production were minor, and much lower than those included in the UNEP/AMAP estimates.

49. Although lower than the 32.3 (19.4-45.2) tonnes estimated for Republic of Korea in the UNEP/AMAP work, Kim et al (2010) note that emissions decreased sharply from 2000 to 2007 associated with the introduction of control technology; also it does not quantify emissions from residential coal combustion. Power plant emissions estimated by Kim et al (2010) at 1.1-4.5 tonnes, are a factor of three lower than those estimated in the UNEP/AMAP report at 6.7-11.3 tonnes, however, estimates for waste incineration (1.8 tonnes) are higher than those in the UNEP/AMAP estimates (0.3 tonnes).

50. The study reported by Kim et al (2010) included an extensive measurement programme on emissions from various sources in the Republic of Korea. This allows the development of country specific emission factors, significantly increasing the accuracy of the inventory reported by Kim et al (2010). The emission factors obtained were generally lower than those published in the literature. Korea uses efficient air pollution control devices (APCD) and good environmental management practices. The practices in place in Korea include mercury regulations for products, mercury waste and source emission control, as well as increased co-benefit control of mercury in existing APCD, cleaner fuels, closing of high polluting and less efficient facilities, stringent air pollution regulation and increasing awareness among facility owners and public to limit the use of mercury in products (Kim et al 2010).

South Africa

51. Masekoameng et al (2010) have estimated mercury emissions to air from South African sources for the time period 2000-2006. By using a combination of annual information on activity in combination with South Africa specific emission factors and UNEP-toolkit based emission factors mercury emissions to air were estimated from each activity. Overall, there was an estimated increase in total atmospheric mercury emissions from around 34 tonnes in 2000 to 50 tonnes in 2006. Coal-fired power plants were the largest contributor of mercury emissions, 38.9 tonnes, followed by cement production with 3.9 tonnes in 2006. This emission inventory for South Africa is in close agreement with that presented in the UNEP/AMAP work for both the total emissions and almost all individual sectors considered.

52. For the coal fired power plants, calculations were based on plant specific coal consumption and mercury-content of coals, as well as emission control devices and their assumed removal efficiencies.

National inventories based on the UNEP Toolkit

53. Several countries have developed, or are in the process of developing a national mercury emission inventory using the UNEP toolkit (UNEP, 2005 and UNEP, 2010). The inventories referred to below have all been officially submitted to UNEP and are available on the UNEP website.

54. In South and Central America estimates have been reported from Chile, Ecuador, the Dominican Republic, Mexico and Panama. The anthropogenic mercury emissions to air in Chile were estimated to be in the range 1.7 - 9 tonnes per year (Government of Chile, 2008);

this is lower than the estimate of 12.6 tonnes in the UNEP/AMAP (2008) report. An emission inventory for Panama, also based on the UNEP toolkit, resulted in estimates of emissions of mercury to air of 0.24 - 4.8 tonnes per year (National Environmental Authority, Panama, 2009), compared with estimated emissions of 0.8 tonnes in the UNEP/AMAP work. Ecuador estimated anthropogenic mercury emissions between 0.9 - 16 tonnes in 2005 (Ministerio del Ambiente, Ecuador, 2008), compared with UNEP/AMAP estimates of 6.5 tonnes. Mexico reported emissions of 50 tonnes in 2004; the UNEP/AMAP (2008) estimate for Mexico was 14.7 tonnes from all sources considered. The Dominican Republic reported emissions of mercury to air of 1.1 tonnes (Ministerio de Medio Ambiente y Recursos Naturales, 2010), compared to the UNEP/AMAP work estimate of .0.654 tonnes.

55. Several Asian countries have produced national estimates of mercury emissions to air using the UNEP Toolkit, as follows:

- Cambodia: 0.8-14.8 tonnes (although the breakdown between emissions to air and water is unclear in this case), compared with the UNEP/AMAP (2008) estimate of 2.3 (1.4-3.3 tonnes);
- Pakistan: 2.1-5.7 tonnes, compared with the UNEP/AMAP (2008) estimate of 5.9 (3.5-8.3 tonnes);
- Philippines: 80.7 tonnes (of which 32 tonnes is attributed to geothermal power production and 39.5 tonnes to gold production), compared with the UNEP/AMAP (2008) estimate of 14.4 (8.6-20.1 tonnes);
- Syrian Arab Republic: 1.3-7.1 tonnes, compared with the UNEP/AMAP (2008) estimate of 0.7 (0.4-1.0 tonnes);
- Yemen: 0.2-1.8 tonnes, compared with the UNEP/AMAP (2008) estimate of 0.4 (0.2-0.5 tonnes).

56. In Africa, UNEP toolkit based inventories have been produced for:

- Burkina Faso: 0.5 tonnes, compared to the UNEP/AMAP (2008) estimate of 2.4 (1.2-3.6 tonnes); and
- Madagascar: 15 tonnes (largely associated with waste disposal), compared to the UNEP/AMAP (2008) estimate of 0.9 (0.4-1.3 tonnes).

57. It should be noted that a number of these national inventories are still preliminary, and direct comparison of total emissions estimates with those from the UNEP/AMAP (2008) work may not always be appropriate due to the sources included (e.g. geothermal power production in the case of the Philippines, which is not addressed in the UNEP/AMAP work). Similarly, emissions from coal combustion sources other than power plants (e.g. use of coal for domestic heating and cooking) are not quantified in many national inventories, and are subject to large uncertainties in all inventories where they are included.

58. National inventories from most countries, including major emitting countries are still lacking or incomplete. In the absence of such inventories, calculations based on available (globally compiled) statistical data, emissions factors and assumptions regarding technology and practises are the only way to derive quantitative global emission estimates. These calculations are associated with some uncertainty, which was estimated in AMAP/UNEP (2008) to be 25 per cent for stationary fossil fuel combustion and 30 per cent for the sectors non-ferrous metal, iron and steel and cement production. For waste disposal, the uncertainty is significantly higher, up to a factor of 5. For mercury and gold production, no estimate of the uncertainty was given mainly due to the high uncertainty in estimating emissions from artisanal

gold production. Uncertainty of total emission estimates for different regions were also given and ranged from 27-30 per cent for North America, Europe and Australia, to 40-50 per cent for the rest of the world. No review of these uncertainties has been made in the present study but an increased availability of national emission inventories (as presented in this section) provides valuable information that will allow preparation of emission inventories with increased accuracy in the future. In addition to the uncertainties associated with using global statistical data and emission factors, it should also be noted that emissions may have changed since the base year 2005 used in the AMAP/UNEP (2008) report. Increased energy use and industrial production may have increased mercury emissions in some regions whereas measures to reduce air pollutant emissions or, specific measures to reduce mercury emissions may have led to reduced mercury emissions in other sectors and regions. An example of this is China where, according to information submitted by China to this study, emissions of mercury from coal fired power have decreased by 35 tonnes from 2005 to 2008, mainly as a result of improved energy efficiency in the sector and measures to reduce SO₂ emissions.

4. Emission control, efficiencies and costs

59. The purpose of this section is to provide an introduction to available technologies to reduce mercury emissions in the selected sectors and their associated efficiencies and costs, co-control of mercury and air pollutants as well as a short general discussion and examples of requirements for cost calculations.

4.1. Primary emission control

60. Mercury enters combustion or the high temperature industrial processes covered in this study as a minor constituent of the fuel, raw material used in the process or waste (in incineration) and a variable fraction of this mercury will be emitted to air after combustion. While the most practical method of reducing mercury emissions is likely through exhaust controls, in many instances other steps can play an important role. One primary control measure is thus to reduce the amount of mercury in the fuel, raw materials or waste before combustion takes place. Another general approach to reduce emissions is to increase the operating efficiency, thereby decreasing the amount of fuel or raw materials used and thus the resulting emissions of mercury and other pollutants.

61. There are a number of more general measures that can be regarded as general best practice and which can be effective in reducing mercury emissions of all kinds, generally at a low cost, particularly in comparison to overall costs of establishing and operating a plant (UNEP 2006, UNEP in prep, 2010):

- *Training, education and motivation of staff and operators.* Industrial processes are overseen by people. Therefore, appropriate and focused training of staff can be a very cost-effective way of reducing discharges of harmful substances.
- *Process control optimisation.* To be able to reduce different pollutants simultaneously, and to maintain low emissions, strict attention to process control is required.
- *Regular maintenance.* To maintain the efficiency of the technical units of industrial processes, and to keep the associated abatement systems operating at a high level, sufficient and routine maintenance has to be ensured.
- *An operational awareness of the importance of environmental management.* An environmental management system that clearly defines the responsibilities for environmentally relevant operations is a necessary tool of responsible management. It raises awareness while including goals and measures, process and job instructions, check lists and other relevant documentation, as necessary.

62. For control of mercury emissions in the flue gases after the combustion process, a number of different technical approaches can be taken. In many cases, co-control of mercury and other air pollutant emissions is achieved when employing air emission control technologies. For additional control of mercury emissions to air, specific control technologies have been developed often based on addition of sorbents, such as activated carbon (both chemically and non-chemically treated) to trap the mercury.

63. For the selected sectors in this study, many of the potentially applicable co-control and mercury specific control technologies are similar and are thus not presented separately for each sector here. However, due to variations among and within sectors, further sector-specific information is considered in chapters' 5.1 to 5.4.

4.2. Co-control of mercury and air pollutants

64. Modern combustion and industrial processes plants, and many older plants which have been updated, have a range of different emission abatement equipment (Table 3). For several of the abatement technologies the primary target is one type of air pollutant, but the technology may also contribute to removal of other pollutants from the flue gases, e.g. the fabric filter has a secondary function for acid gas control and similarly the wet scrubber for particle control. Often different control systems are needed in combination to achieve a desired control of several pollutants. Generally the abatement equipment installed are those targeted to reduce emissions of pollutants such as particles (PM, particulate matter), sulphur dioxide (SO₂) and nitrogen oxides (NO_x), which to a varying degree also may affect mercury emissions. To reach reductions of mercury exceeding the levels already achieved as co-benefit from the more common air pollutant abatement technologies, additional measures to specifically reduce mercury are needed.

Table 3. Some common air pollutant control technologies (Largely from EMEP/EEA air pollutant emission inventory guidebook, 2007).

Air pollutant control technologies	Description
Fabric filters (FF)	Consist of semi-permeable material in the form of bags or sleeves which trap particles and which are mounted in an airtight housing (baghouse), which is divided into a number of sections. Fabric filters are also used as a second stage in acid gas control systems (e.g. SO ₂ control).
Electrostatic precipitators (ESP)	Use the principle of electrostatic attraction to remove entrained particles from the flue gases. They consist of rows of discharge electrodes (wires or thin metal rods), through which a high voltage is applied, and which run between an array of parallel rows of metal plates which collect the charged particles.
Wet scrubbers	Remove acid gases (e.g. HCl, HF and SO ₂) by washing the flue gases in a reaction tower designed to provide a high gas-liquid contact. In the first stage the gases are cooled by water sprays, removing HCl, HF, some particulates and some heavy metals. In the second stage calcium hydroxide or another suitable alkali is used to remove SO ₂ and any remaining HCl. With lime/limestone wet scrubbing, SO ₂ reduction efficiency is > 90 per cent. Wet scrubbers are also sometimes employed with the prime objective of removing particles. The majority of FGD systems used worldwide are wet limestone scrubbers.
Semi-dry scrubbers/spray absorber systems (spray drying)	Make use of an alkaline reagent slurry (usually calcium hydroxide) which is introduced as a spray of fine droplets. The acid gases (e.g. SO ₂) are absorbed into the aqueous phase on the surface of these droplets and neutralised; the slurry is dried by the heat of the flue gas to form a dry product, which is collected in an electrostatic precipitator or fabric filter. The SO ₂ reduction efficiency is > 90 per cent.
Dry injection systems	Involve the injection of an alkaline reagent (e.g. calcium hydroxide or sodium bicarbonate) as a fine, dry powder to remove and neutralise acid gases. The neutralised product is normally collected in a fabric filter.
Selective non-catalytic reduction (SNCR) and Selective catalytic reduction (SCR)	Aims at reducing nitrogen oxides (NO _x) in the flue gases. The SNCR process involves injection of ammonia or urea near the furnace. An SCR system is based on selective reactions with injected additives in the presence of a catalyst. The additives used are mostly ammonia (gaseous and in solution) but also urea. Emission reduction of NO _x with SNCR can be limited (up to 50 per cent) but for SCR between 70 and 95 per cent.
Adsorption using activated carbon/activated lignite coke	Several different technologies have been developed for mercury, VOC and dioxin control. These systems can also be fairly effective at removing HCl and SO ₂ .

65. Fabric filters (FF) and electrostatic precipitators (ESP) are primarily aimed at reducing particulate matter from the flue gases, but they can also to some extent reduce atmospheric mercury emissions by removing mercury adsorbed to the particles. The mercury removal efficiency of the filters depends on the filter's capability for removal of small size particles on which mercury may be adsorbed and on the presence of constituents in the flue gas stream that may be captured on the filters and serve as mercury adsorption sites.

66. Systems designed to remove SO₂ (and other acid gases) from the flue gases, also called flue gas desulphurisation (FGD) systems, are wet scrubbers, semi-dry scrubbers/spray absorber systems and dry injection. All are based on the reaction of SO₂ with an alkaline agent added as solid or as suspension or solution of the agent in water to form the respective salts. Use of FGD processes may also reduce emissions of particulate matter, mercury and other metal emissions. Other wet systems are mainly configured to remove particles, but will also reduce emissions of water soluble gases such as SO₂.

67. Selective non-catalytic reduction and selective catalytic reduction aim at reducing emissions of nitrogen oxides (NO_x), and may have a positive effect on mercury removal by enhancing oxidation of elemental mercury to divalent forms.

68. Co-control of mercury emissions from combustion sources is mainly determined by the capacity of the installed air pollution control system (filters or scrubbers) to adsorb mercury from the gas phase to solid or liquid phase, which can then be separated from the flowing gas. Coal combustion releases mercury in oxidised (Hg²⁺), elemental (Hg⁰) or particulate bound (Hg^p) form. Mercury is present in the coal in trace amounts, and the combustion process releases this into the exhaust gas as elemental mercury. This may then be oxidised as the flue gas cools down. Oxidised species are readily absorbed in liquid and solid sorbents and can thus be removed more efficiently from the flue gas. The mercury that is adsorbed onto solid surfaces is known as particulate-bound mercury. Formation of oxidised species is dependent on temperature but also on the composition of the flue gas, which in turn is dependent on the conditions of combustion, composition of the fuel and presence of additives etc. The presence of chlorine in coal is one example of an important parameter. Higher concentrations of chlorine in coal typically yield higher concentration of chlorine species in the flue gas and a more efficient oxidation of Hg⁰ to divalent form. In any practical case, the speciation of mercury is determined by a number of parameters and can also be highly variable. Oxidised mercury is more prevalent in the flue gas from bituminous coal combustion, and it is relatively easy to capture using SO₂ controls such as wet limestone scrubbers. Particulate bound mercury is also relatively easy to capture in existing particulate control devices. Elemental mercury, more prevalent in the flue gases of lignite and sub-bituminous coal combustion, is more difficult to capture with existing pollution controls.

69. Further information on relevant available techniques for the selected sectors can be found in, for example, the work of the European Integrated Pollution Prevention and Control (IPPC) Bureau (Best available technique REFERENCE (BREF) documents at <http://eippcb.jrc.es/reference/>, or in US Environmental Protection Agency (USEPA) emission factor handbook (USEPA, AP-42) at <http://www.epa.gov/ttnchie1/ap42/> and at USEPA's Clean Air Technology Center found at <http://www.epa.gov/ttn/catc/>.

4.3. Combining different control technologies for optimised mercury control

70. Control of mercury emissions from combustion sources can be achieved by a number of different measures or combination of control measures. Air pollution control technologies (aimed primarily at other pollutants but which will also reduce mercury emissions to some extent) are described in section 4.2. Examples of available mercury specific controls are described in chapter 5, under the relevant sub sections for each sector. For the development of relevant control strategies for mercury, combinations of air pollution control technologies and specific mercury control technologies should be considered. Which air pollution control devices are installed is of course also dependant on other priorities, such as reducing emissions of particulate matter or sulphur dioxide. It is thus difficult to present combinations of different control technologies which are generally applicable for optimal mercury emission control without knowledge of the current status of wider air pollution control. Other parameters which will affect the selection of potential control options are the technological specification of the emission source and the characteristics of the flue gas (e.g. other components, mercury speciation).

71. Of the four selected sectors in this study, by far the most information is available on mercury control in coal fired power plants, mainly based on research performed in the USA (see section 5.1). For this sector and under the conditions relevant for the studied power plants in the US, significant mercury emission reduction can be achieved by the application of activated carbon injection (ACI), possibly complemented by additional particle removal devices such as fabric filters. For example, ACI is already reducing mercury by 90 per cent at roughly 30 boilers in the USA (GAO, 2009; NESCAUM, 2010).⁴ Activated carbon can be untreated or, more recently, chemically treated, particularly with halogens such as bromine or chlorine. Chemical treatment increases the proportion of oxidised mercury (Hg^{2+}) in the flue gas leading to improvements in the performance of FGD and ESPs for removal of mercury (Weem, 2010). This is particularly relevant for low grade lignite or sub-bituminous coals. Chemically treated ACI continues to be developed and tested, but is also commercially available in the USA.

72. ACI has mainly been applied to power plants with existing air pollution control equipment (e.g. ESP, desulphurization, NO_x control). Further testing and study of ACI efficiency in coal fired power plants equipped with e.g. only ESP may be required. While ACI and other direct controls for mercury can operate well in conjunction with other air pollution controls, it is not necessarily the case that advanced emission controls are required for proper functioning of mercury controls. However, since mercury specific controls are aimed at increasing the amount of mercury adsorbed to particles, emission control of particles (e.g. ESP or FF) is a basic requirement. For development of national mercury emission projections and emission reduction strategies it is thus necessary to evaluate both the current and planned technical status of emission control of air pollutants as a first step. In regions with only control of particle emissions and where no additional air pollutant controls are planned, direct employment of sorbents or oxidizing agents may be considered. Pre-combustion measures such as coal washing or fuel blending (with fuel of lower mercury content) may also be considered.

⁴ The Institute of Clean Air Companies lists 155 ACI controls for mercury that have been applied to power plants in the US and Canada as of June 2010; many of these are already operational.
http://www.icac.com/files/members/Commercial_Hg_Bookings_060410.pdf

4.4. Cost and efficiencies for mercury control

73. Estimating costs for control of mercury emissions requires knowledge of the current status of the emission sources including technological descriptions of current and planned air pollution control. Thus if controls have been or are being applied for the purpose of controlling non-mercury pollutants (e.g., PM, SO_x, NO_x) to meet relevant air pollution standards, the cost of the resulting mercury emission reductions is considered to be zero. Real cost of mercury control should be considered as only the cost of those actions taken specifically to reduce mercury emissions. All cost information provided in this report should be evaluated in that context. Finally, the costs of mercury control are of course also dependant on the level of mercury control that is achieved.

74. In the following sector specific sections, general information on costs of mercury control is presented mainly based on studies in Europe and North America.

75. Costs of retrofitting pollutant control technologies to an existing plant are generally higher than when similar measures are included in the design of a new plant. This is due to factors such as; lower efficiencies for old plants, special requirements on space and design of the measure, and loss of revenues associated with installation and the following down time of the production. Annual investment expenditures may also be higher since the operational time of the existing plant might be shorter than a new one, leading to fewer years to spread the investment costs. Retrofitting costs for the more expensive technologies such as FGD and SCR are generally assumed to add 30 per cent to the annual capital costs, whereas costs for retrofitting technologies for particle control is significantly lower.

76. Mercury emission reduction resulting from different combinations of control installations is given in Table 4, and Table 5 and in the following chapters for each of the four sectors covered in this study. In some of the tables a single figure of removal efficiency is given for each control technology combination, as a best general estimate of reduction efficiency. In reality the removal efficiency should rather be seen as a range because the removal at any particular plant is dependent on a multitude of factors, such as coal quality, combustion conditions etc.

77. It is also important to note that effectiveness of mercury controls in this report is presented as per cent reduction in mercury emissions. Total emissions from a facility will depend on additional factors such as total activity and the level of mercury in the input or feed materials (e.g. coal, ores, waste).

78. Investment (capital) cost typically reflects the cost of purchasing the equipment and its necessary infrastructure, instruments and controls. In addition they include freight and installation, taxes, as well as engineering fees.

79. Operational and Maintenance (O&M) costs reflect labour cost, power cost, maintenance cost, periodic replacement of items, control cost, as well as variable cost from the adsorbing material input. They can be partly offset by the sale of particular by-products of pollution abatement, such as gypsum from wet limestone FGD. Some O&M costs, such as costs of labour and power, are typically country- specific and may, to some extent, differ from country to country. Costs are in this report primarily based on costs reported from the United States or Europe but more accurate country- specific costs may be calculated by adjusting for country-specific GDP/capita Purchase Power Parity. For dedicated mercury removal technologies, however, these costs are only a small fraction of the total O&M costs and such an adjustment may not be necessary. In some cases, costs of mandatory monitoring of emissions are also included in the O&M costs.

80. US EPA, (2005) reports an uncertainty range in the cost data in the order of -30 to +80 per cent. Typically, costs of technologies that are in the early phase of implementation will decrease over time. Costs are also highly site-specific, so they are normally presented as indicative costs collated from a range of literature being reported for different plant sizes and sites, which makes extrapolation to other plants less certain.

81. Based on an evaluation of control costs performed in the European research project ESPREME, a database of costs⁵ and efficiencies for some different control options was developed. Annualized costs of selected combinations of pollutant reduction technologies per activity and their efficiencies for mercury emission reduction in the coal combustion, non-ferrous metals, cement and waste incineration sectors as published in Pacyna et al. 2010, are illustrated in Table 4, Table 5 and Table 6. Note that discussion of additional control options are described in Chapter 5 of this report.

Table 4. Example of emission control costs and removal efficiencies for coal combustion from Pacyna et al 2010.

Emission control technology	Estimated Hg reduction (per cent)	Annual costs ^b (USD 2008/MWhe)		
		Investment cost	O&M costs	Total cost ^a
Dry ESP	> 63	0.5	0.9	1.4
Fabric filter (FF)	> 93	0.5	1.5	1.9
FF+wet or dry scrubber+sorbent injection	> 98	2.7	3.0	5.7
Dry ESP + wet or dry scrubber + sorbent injection	> 98	2.7	2.4	5.1

a. annual operating costs of about USD 20/MWh could be expected for emerging technologies such as Electro-catalytic Oxidation or Integrated Gasification Combined Cycle.

^b The accuracy of cost estimates in the table is within ± 50 per cent.

⁵ Investment/Capital costs are assumed from a technology lifetime of **15years**, including a 4 per cent discount rate.

Table 5. Examples of emission control costs and efficiencies for non-ferrous metal and cement production from Pacyna et al (2010).

Sector	Specific activity indicator (SAI)	Emission control technology	Hg red (per cent)	Annual costs ^a (USD 2008/SAI)		
				Investment costs	O&M costs	Total costs
Primary lead	metric ton primary lead	Dry ESP	5	0.1	0.04	0.1
	metric ton primary lead	FF	10	0.1	1.1	1.2
	metric ton primary lead	Activated carbon injection +FF+FGD	90	2.5	1.3	3.8
Primary zinc	metric ton primary zinc	Dry ESP	5	0.1	0.06	0.2
	metric ton primary zinc	FF	10	4.5	1.1	5.6
Primary copper	metric ton primary copper	FF	5	1.8	13.8	15.6
	metric ton primary copper	Fabric filters – state-of-the-art	10	3.9	25.7	29.5
Secondary lead	metric ton secondary lead	Dry ESP	5	0.1	0.06	0.2
	metric ton secondary lead	FF	10	6.8	1.1	7.9
Secondary zinc	metric ton secondary zinc	Dry ESP	5	0.1	0.06	0.2
	metric ton secondary zinc	FF	10	0.1	1.4	1.5
Secondary copper	metric ton secondary copper	Dry ESP	5	10.9	15.9	26.8
	metric ton secondary copper	FF	10	6.6	44.0	50.6
Cement production	metric ton cement	FF	5	0.2	0.2	0.4
	metric ton cement	FF – optimized	98	0.4	0.4	0.8
	metric ton cement	Wet FGD	90	1.4	0.5	1.8

^a The accuracy of cost estimates in the table is within ± 50 per cent.

Table 6 Example of emission control costs and removal efficiencies for waste incineration from Pacyna et al 2010.

Emission Control Technology	Hg reduction (per cent)	Annual costs ^a (USD 2008/metric ton of waste)		
		Investment costs	Operating costs	Total costs
Wet scrubber with alkaline additives – medium efficiency if emission control	20	0.1	0.1	0.2
Waste separation – medium	60	0.6	0.6	1.2
Dry ESP	70	1.8	7.0	8.8
ESP+wet scrubber+activated carbon with lime+FF	99	2.3	2.5	4.8
Two-stage scrubber+wet ESP –	90	2.3	1.8	4.1
Activated carbon injection +FF	80	2.2	4.0	6.2
Activated carbon injection +venturi scrubber+ESP –	95	5.3	6.2	11.4
Activated carbon injection +venturi scrubber with lime milk+caustic soda+FF	99	5.8	7.1	12.7

^a The accuracy of cost estimates in the table is within ± 50 per cent.

5. Characterisation of the selected sectors

82. The purpose of this chapter is to provide brief technical descriptions and information about mercury emissions to air, with focus on available options for controlling mercury emissions in the selected sectors and associated costs. Examples of cost estimates for individual facilities are also presented.

More detailed information on coal fired power plants can be found in the Draft UNEP Process Optimization Guidance document (UNEP 2010 in prep)

5.1. Coal combustion in power plants and industrial boilers

83. The purpose of coal combustion is to provide heat and electricity for industry and society in general. Combustion of coal can be achieved using several different technologies.

5.1.1. Origin of mercury and emissions from coal combustion

84. Mercury is naturally present in coal as a trace constituent and is released during combustion, entering the atmosphere via the flue gases. Concentrations of mercury in coal vary between geographical regions and are usually in the range 0.1 to 0.3 ppm but levels outside this range can be also found, and values as high as 1 ppm have been measured by accredited laboratories in lignite coal. The USGS has recently published a World Coal Quality Inventory of more than 1500 coal samples worldwide where one of the parameters analysed is the mercury content (available at <http://pubs.usgs.gov/of/2010/1196>).

5.1.2. Technologies for coal combustion

85. Pulverized coal combustion (PCC) is the dominant coal combustion technology worldwide and it is usually employed in larger units. It can be 1st generation traditional steam technology, with efficiencies up to 40 per cent, calculated based on Lower Heating Value (LHV) or 2nd generation super critical steam technology, with up to 47 per cent (LHV) efficiency. Similar efficiencies can be achieved by integrated gasification combined cycle technology (IGCC), although this is only now starting to be applied to coal combustion as a result of its particular advantage for carbon capture and storage (CCS). Fluidised bed combustion (FBC) is used to a lesser extent, although now in larger units and with efficiencies up to about 44 per cent (LHV). Both PCC and FBC require pre-treatment of the coal through grinding etc. to achieve suitable size fractions of the coal. Emission control systems on these, usually larger units, are common and economically feasible. Stoker systems have been used in small size boilers for over a century and use a lump coal feed and combustion. These systems are relatively inefficient and unstable due to restricted access of oxygen to the combustible material. IGCC technology converts the coal to a gas prior to combustion, thereby achieving low emissions without the use of expensive secondary abatement controls.

86. The PCC technology is well developed, and there are thousands of units around the world (IEA Clean Coal Centre). PCC can be used to fire a wide variety of coals, although it is not always appropriate for those with high ash content.

87. FBC have a bed of inert material which is agitated or "fluidised" by preheated air injected through a porous plate or grate from below. The result is a turbulent mixing of gas and solids. The tumbling action, much like a bubbling fluid, provides effective chemical reactions and heat transfer. The typical relatively low emissions of air pollutants from FBC are achieved by air

staging, limestone addition and low combustion temperatures of about 750– 950 °C. FBC is in particular adapted to coals rich in ash and fuels of varying quality (EMEP/EEA, 2009, EC 2006). However, it has the disadvantage of a large waste stream due to the inability to use the products of the desulphurisation process in the way that FGD gypsum from PCC is used.

88. Combustion in power plants converts the chemical energy stored in the fuels to either electrical power or heat, or both. Combined heat and power plants are more efficient in utilizing the released energy, whereas in older stoker type power plants, the total energy losses to the environment might be as high as 70 per cent of the chemical energy in the fuels, depending on the fuel and the specific technology. In a modern high efficiency power plant, losses are down to about half of the chemical energy contained in the fuels. In a combined heat and power plant a larger fraction of the energy in the fuel is delivered to final users, either as electricity or as heat (for industrial processes or residential heating or similar uses). The EU's Large Combustion Plant BAT Reference Document sets the standard for combined heat and power (CHP) generation for coal combustion at 70-90 per cent (LHV) efficiency.

5.1.3. Control measures in coal combustion

89. A primary control measure is to reduce the amount of mercury in the fuel for example by selecting coal with naturally low mercury content, by pre-treatment of the coal or by fuel substitution schemes (e.g., substitution of coal with natural gas or renewable energy sources). Another general approach to reduce emissions is to increase the operating efficiency thereby decreasing the amount of fuel required and thus the resulting emissions of mercury and other pollutants. To control mercury emissions after the combustion step various technical air pollution and mercury specific control measures are possible. Brief summaries of these options are given below.

Pre-treatment of coal

90. Apart from the quality and mercury content of the coal the chemistry of mercury within the coal also influences mercury emissions from coal combustion. Coal from different geographical regions may have very different characteristics. The major classes of coal are anthracite, bituminous coal, sub-bituminous coal and lignite where anthracite has the highest carbon content and energy value and lignite the lowest. In an alternative nomenclature system, the coal with the highest carbon content is sometimes named “hard coal” and the lowest “brown coal”⁶.

91. A decrease of mercury emissions from power plants may be achieved by the deployment of coal treatment technologies prior to combustion. Coal treatment technologies considered in the context of plant efficiency and mercury removal includes conventional coal washing, coal beneficiation for mercury content, coal blending, and coal additives.

⁶ Different systems for classification and nomenclature of coal types are in use internationally, mainly based on the degree of change undergone by a coal as it matures from peat to anthracite – known as coalification. Low rank coals, such as lignite and sub-bituminous coals are typically softer, friable materials with a dull, earthy appearance. They are characterised by high moisture levels and low carbon content, and therefore lower energy content. Higher rank coals are generally harder and stronger and often have a black vitreous lustre. They contain more carbon, have lower moisture content, and produce more energy. Anthracite is at the top of the rank scale and has a correspondingly higher carbon and energy content and a lower level of moisture. An alternative nomenclature is “Brown Coal” for the low rank coal and “Hard Coal” or “Black Coal” for the highest rank.

92. Although coal washing/treatments may produce relatively high reduction rates for mercury with some coals, they are not adequate for use as a reliable method for mercury reduction on all types of coals. Washing with the primary objective to reduce sulphur in bituminous coals also reduces the mercury concentration relative to in-ground concentrations. The removal efficiency varies according to the fraction of mercury in the sulphides in the coal and the efficiency of sulphide removal (Kolker et al, 2006, in Sloss, 2008). Other coal cleaning processes such as froth flotation, selective agglomeration, cyclones and chemical approaches are also aimed at sulphur removal and therefore any mercury reduction is a co-benefit. Mercury reduction in these processes ranges from 10 per cent to as much as 70 per cent, with an average of 30 per cent on an energy equivalent basis (Sloss, 2008).

Improved operating procedures

93. Improvement of plant efficiency (e.g. aiming at lowering production costs) may involve a number of measures designed to conserve fuel (coal) and, as a result, reduce the amount of mercury emissions. Some of the most commonly applicable measures at coal-burning facilities include: new burners, improved air preheater, improved economizer, improved combustion measures, minimization of short cycling, minimization of gas-side heat transfer surface deposits, minimization of air infiltration and upgrading of turbines. In addition, operation and maintenance (O&M) practices have a significant impact on plant performance, including its efficiency, reliability, and operating cost. A well operated and maintained plant will experience less rapid deterioration of heat rate; hence, O&M practices themselves influence coal use and mercury emissions. Good O&M practices should be an on-going concern in daily plant operation. However, existing PCC plants are limited to a maximum efficiency of about 40 per cent, with further improvements only being achieved by upgrading to new supercritical boilers.

Co-control of mercury in coal combustion facilities

94. The multi-pollutants control technologies described in chapter 3 are widely used on coal combustion facilities. The reported effectiveness of existing control technologies and co-benefit approaches on mercury capture in coal combustion facilities have been compiled by Sloss (2008) and are summarised in Table 7. The table shows that the ranges of mercury capture for a certain combination of controls may be quite large, and that the degree of capture also depends on the quality of the coal (bituminous coal, sub-bituminous coal or lignite). Thus, evaluation of capture of mercury as co-benefit in air pollution control equipment must be made on a plant specific basis and include information on fuel quality, flue gas composition and specific details about the installed air pollution control.

Table 7. Example of air pollution controls and their efficiency (per cent) to capture mercury in coal fired power plants, adapted from Sloss (2008). PM=particulate matter, SO₂= sulphur dioxide, NO_x= nitrogen oxides.

PM Controls	Bituminous coal*	Sub-bituminous coal*	Lignite*
CS-ESP	0-63	0-18	0-2
HS-ESP	0-48	0-27	- [□]
FF	84-93	53-67	-
PM and SO₂ control			
CS-ESP + wet FGD	64-74	0-58	21-56
HS-ESP + wet FGD	6-54	0-42	-
FF + dry scrubber	very high		lower
FF + Wet FGD	62-89		
NO_x, PM and SO₂ control			
SCR + spray dryer + FF	94-99	0-47	0-96

CS - Cold Side; HS - Hot Side; ESP - Electrostatic precipitators, FF - Fabric filters, FGD - Flue Gas Desulphurisation, SCR - Selective Catalytic Reduction.

*The ranges are based on a limited set of tests conducted at facilities in the USA.

□ - no data

95. Large efforts have been made in the US to develop technologies for mercury emission control from coal combustion, including extensive full scale tests of the efficiency of control devices for mercury control depending on coal type, type of sorbent and additional additives, temperature and operating conditions. The results of these are summarised in UNEP 2010 (in prep.) and Sloss (2008). Note that mercury controls are already in use at dozens of facilities, as referenced in section 4.3

96. Some recent examples of investigations of mercury removal in air pollution control devices indicate efficiencies in the range of 68 per cent-91 per cent, for thermal power plants in Korea with the highest efficiency for mercury coinciding with high efficiency air pollution control (mainly SCR, CS-ESP and wet FGD) (Kim et al 2010; Pudasainee, 2009).

Mercury specific control in coal combustion facilities

97. In addition to co-benefits from other air pollution control technologies, mercury specific controls can and are being used extensively on a commercial scale in the US.

98. When designing mercury specific control technologies, these are often based on adsorption of mercury on an added sorbent such as activated carbon to enhance the adsorption capacity. The original untreated sorbents were less successful with lower grade coals, due to their higher proportion of elemental mercury in the flue gases which is more difficult to capture. However, sorbents are now available that have been chemically treated e.g. with halogens such as bromine and chlorine. These convert the more difficult to capture elemental mercury into the divalent (oxidised) form that is easier to capture, thereby achieving high mercury reduction across all coal types.

99. Injection of sorbents into the flue gas of coal-fired boilers for mercury control has been applied at boilers in Germany since the 1990's (Wirling, 2000) and has been demonstrated in the United States on several full-scale commercial systems (GAO, 2009, NESCAUM 2010). Sorbents can be added upstream of an existing particulate matter (PM) control device and the mercury is collected in the fly ash fraction. Alternative technologies are where the sorbent is

injected after existing PM control devices, in which case an additional particle control unit is needed to capture the mercury containing sorbent (e.g. the toxic emission control process [TOXECON™] configuration). A third demonstrated configuration for sorbent injection is TOXECON II™ in which sorbent is injected into the middle fields of the existing ESP. The TOXECON™ method, developed and patented by EPRI (Electric Power Research Institute) includes a secondary particle control unit (fabric filter) and has been shown to be able to reduce mercury emissions by over 90 per cent. The majority of the fly ash is thus collected before the mercury control step and has a wider use and marketability. The much smaller ash fraction containing the activated carbon and the mercury is collected in the secondary step

100. Some of the factors that affect the performance of any particular sorbent with regard to mercury capture include (Pavlish et al., 2003 and Srivastava et al., 2006, NESCAUM 2010):

- Physical and chemical properties of the sorbent
- Injection rate and distribution of the sorbent
- Flue gas parameters such as temperature, concentrations of halogen species (e.g., HCl, HBr), and concentration of sulphur trioxide (SO₃)
- Existing air pollution control configuration

101. Most tests summarised in UNEP Process Optimization Guidance Document (POG) (In prep.) and Sloss (2008) were performed with sorbent injection upstream of existing PM controls. Some of the main conclusions are:

- Capture of mercury increases with increasing amount of sorbent added, although an optimal amount was found in some cases where further additions yielded only small effects.
- Lower temperatures (<150 °C) at the ESP inlet favours higher mercury capture.
- Efficient control using untreated activated carbon is dependent on the formation of oxidised mercury in the flue gas which is generally favoured at high concentrations of chloride in the coal. A higher efficiency can be achieved with halogen treated carbon, especially in plants using low rank coal with low chlorine contents.

102. One identified negative aspect of activated carbon injection for mercury control is that the mercury (as well as the added activated carbon) will end up in the fly ashes which may reduce the potential for safe disposal or use of the ash in e.g. construction and cement manufacturing. To avoid this, a second particle removal installation can be made and the activated carbon added after the regular particle control device e.g. as in the TOXECON™ method described above.

103. Additions of bromine to enhance oxidation of mercury in the flue gases have been tested as a pre-combustion additive at levels equivalent to 25 ppm in coal. A mercury emission reduction of 92-97 per cent was consistently observed on a 600 MW unit firing subbituminous coal and equipped with an SCR and wet FGD (Rini and Vosteen, 2009). Similar tests have been performed at 14 units firing coals of low chlorine contents and results show more than 90 per cent of flue gas mercury oxidation for bromine additions equivalent to 25 to 300 ppm in coal (Chang et al., 2008). The use of halogens as pre-combustion additives, especially bromine, has the potential to reduce mercury emissions by over 80 per cent in certain situations (UNEP in prep, 2010).

5.1.4. Costs and efficiencies of control technologies for coal fired power plants

Pre-treatment of coal

104. Coal pre-treatment includes procedures (such as crushing, drying and washing) aimed at increasing the performance and energy output in the combustion process or at reducing emissions of harmful substances. Addition of chemical reagents or coal blending to achieve better performance is sometimes also made. Coal washing is primarily a measure to reduce ash and sulphur contents of the coal but can also decrease the mercury contents. Direct costs of pre-treatment of coal are thus difficult to allocate specifically to mercury control. Costs for coal blending (i.e. mixing with other coal or fuels with lower mercury content) are dependent on availability and market demand of the blending fuel (EPA, 2005).

Improved operation procedures

105. Plant design and operation changes are beneficial to both performance and reliability of the plant and can thus reduce operational and maintenance costs. No data of sufficient quality was found accessible on these costs or reduced costs.

Co-control of mercury in coal combustion facilities

106. An extensive review of control technologies and the associated costs is available from the IEA Clean Coal Centre (Sloss, 2008).

107. One important aspect when estimating costs for mercury emission control are existing conditions in a specific plant. The costs for mercury removal will be very different if the existing conditions involve power plants equipped with modern air pollution control than if very simple emission controls are installed. Optimization of existing emission control installations can potentially reduce emissions of mercury but no general cost estimates are available.

108. The US EPA has developed a manual (CUECost - Coal Utility Environmental Cost manual) for estimating costs of air pollution (including mercury) control in power plants. The method can be used to estimate costs of installation of twelve different technologies for removal of SO₂, NO_x, CO₂ and mercury, either as separate components or as an integrated air pollution control system <http://www.epa.gov/nrmrl/pubs/600r09131/600r09131.html>.

109. Based on an evaluation of control costs performed in the European research project ESPREME, a database of costs and efficiencies for different control options was developed. Annualized costs of selected multi-pollutant reduction technologies per activity and its efficiencies for mercury emission reduction in the coal combustion sector are also published in Pacyna et al. 2010. Annual total costs⁷ (i.e. annual capital- and O&M costs), range from 1.3 USD 2008/MWhe for ESP installation with an estimated mercury removal of >63 per cent to 2-5 – 5 USD 2008/MWhe for more advance air pollution control (particle and sulphur removal in scrubber) with mercury removal efficiencies estimated to >93 per cent. These results represent European conditions and may not be transferable to other regions.

110. Capital cost of abatement equipment at Best Available Technique (BAT) level (particulate control + FGD) contributes less than 5 per cent of the total coal-fired power plant costs (without fuel costs). If the cost of coal is included in the cost of the plant, this contribution is at a level of 3-3.5 per cent. The basis for the estimation is derived from (Rokke,

⁷ Capital costs are assumed from a technology lifetime of **20 years**, including a 4 per cent discount rate.

2006) for the production cost of a new coal plant of 60 USD per MWh, including a fuel cost of 14.10 USD per MWh, together with a total cost of BAT technique of 2.28 USD per MWh.

Mercury specific control in coal combustion

111. Reduction of mercury emission from coal combustion is an effect of implementation of technologies for controlling conventional air pollutants (particulate matter, SO₂ and NO_x) and specific technologies for heavy metal removal. Examples from US power plants reported in the United States Government Accountability Office (GAO)-10-47 (2009) and NESCAUM (2010) report indicate that mercury control is being achieved at commercial scale at many power plants at relatively low costs using carbon injection, but alternative control measures may be needed in some cases. There are also examples where conventional air pollution control installations are sufficient to reach mercury control of >90 per cent. Some boiler types can also reach this level of mercury removal without additional control. The costs for purchase and installation of carbon (and sorbent) injection systems and monitoring equipment are in the range \$1.2 million to \$6.2 million (year 2008 USD) per power plant, significantly less than for other types of air pollution control for controlling particle matter, sulphur dioxide or nitrogen oxides. In comparison, the average investment (purchase and installation) cost of a wet scrubber for sulphur dioxide control, is more than \$86 million (year 2008 USD) per boiler. For power plants investing in fabric filters in addition to the sorbent injection systems the investment cost was \$12.7 to \$24.5 million (year 2008 USD). The actual amounts of mercury removed are not reported in this study (GAO, 2009). However, the NESCAUM (2010) report includes data on mercury control efficiencies from a number of coal power plants in the US.

112. According to National Energy Technology Laboratory (NETL) research activities sponsored by the US Department of Energy (DOE), the costs of sorbent injection for mercury removal have shown significant advances along with the potential for reductions in overall installation and operational costs. A DOE economic analysis released in 2007 indicates that the cost of mercury control could be drastically lowered compared to original estimates due to a reduction in the injection rate of a sorbent when using more efficient treated sorbents, and even offsetting the higher costs of the treated sorbents. The analysis indicated that a cost of 90-per cent mercury emission control by means of activated carbon injection ranged from about \$30,000 to less than \$10,000 per pound (equal to \$22,000 to \$66,000 /kg) of mercury removed for DOE field testing sites (Feeley, 2008). These DOE tests sites used a chemically-treated (brominated) activated carbon. Although the capital cost of a mercury control system is relatively low, the major expenditure comes from the cost of sorbent itself. Generally, brominated carbon affords much lower injection rates (mass sorbent/flue gas flow) than the untreated carbon to accomplish the same level of mercury removal. Thus, despite the fact that chemically treated carbons are more expensive than untreated ones, the use of chemically treated carbons allows to significantly lower the cost of mercury removal.

113. Fly ash captured from air pollution control devices can potentially be reused for engineering applications and has an economic value. Adding absorbents like activated carbon can affect the quality of the fly ash (and gypsum) and potentially prevent sales. Mercury control costs are therefore also affected by a potential loss of revenue for plants that sell their fly ash for beneficial reuse. A report from the Northeast States for Coordinated Air Use Management (NESCAUM) has recently summarized the DOE tests which indicated an increase of 170-300 per cent on the mercury removal costs can be observed if loss of such ash revenue is taken into account (US DOE, 2006 and NESCAUM, 2010). However, this economic impact has led to technological advances in the form of the development of concrete-friendly sorbents, which should eventually offset this cost.

114. The US DOE has estimated capital costs for activated carbon injection in a 360 MW unit burning coal with low sulphur, low chlorine and high alkalinity) equipped with spray dryer absorber (SDA) plus FFs (US DOE, 2006). For this unit, capital cost (defined as total capital requirement [TCR]) was determined to be \$3.6 /kW (0.03 year 2010 USD/MWhe). Total O&M costs for this example were estimated at \$600,000 /year (0.21 year 2010 USD/MWhe) for 90 per cent removal of mercury. An additional cost of operating an ACI system was the by-product impact or the value of disposal costs plus the unrealized revenue from the sale of ash (due to activated carbon contamination of fly ash). The by-product impact for this 360 MW unit was estimated at USD 1,430,000/year.

115. Additional examples of capital and operating costs for different air pollution control technology configurations are presented in Table 4.

116. In Table 8, costs are shown for adding mercury specific emission control to a hypothetical power plant with selected existing equipment configuration. The cost estimates are shown as presented in Sloss (2008) (80 per cent mercury removal) and based upon the National Energy Technology Laboratory (NETL) reported cost for ACI, summarized in the NESCAUM report. These NETL costs were based upon its Phase II field test programs, presenting several cost scenarios for various coals, configuration, and level of mercury control. For the sorbent technology (ACI), the analyses included scenarios for 50 per cent, 70 per cent, and 90 per cent control for bituminous, subbituminous and lignite coals, and for oxidation technologies (catalysts and additives, such as CaBr₂) the analysis included scenarios for mercury reduction of 73 per cent for sub bituminous and lignite coals (NESCAUM, 2010). Typically, the costs of adding bromine (CaBr₂) is driven by chemical costs, as for the sorbent technologies while catalysts are typically driven by investment/capital and re- generation costs (NESCAUM, 2010).

117. Based on the data in Table 8, hypothetical calculation examples can be made. For a 220 MW power plant with only ESP configuration, introducing ACI technology would remove 180 grams (90 per cent) mercury per tonne of consumed lignite coal containing 0.2 ppm mercury, at a cost of \$0.13-1.20 /MWhe. A 500 MW power plant with ESP+FGD configuration would by adding bromine remove 73 grams (73 per cent) mercury at a cost of \$0.08 /MWhe per tonne sub- bituminous coal containing 0.1 ppm mercury or remove 146 grams mercury per tonne lignite coal containing 0.2 ppm mercury.

Table 8. Examples of capital and operating & maintenance costs, as well as removal efficiencies for different configurations of mercury specific emission controls. Based on data from Sloss (2008)/ Curs (2007) and NESCAUM (2010)

Existing equipment configuration 1=lignite coal 2=sub-bituminous coal	New equipment configuration	Capital cost (year 2010 USD/MWhe)	O&M cost (year 2010 USD/MWhe)	Removal efficiency (per cent) / plant capacity (MW)	Literature
cESP	+ACI	0.15	4.06	80 /	Sloss, 2008
cESP+FGD	+ACI	0.15	4.06	80 /	Sloss, 2008
Dry scrubber+FF	+ACI	0.02	0.32	80 /	Sloss, 2008
ESP ¹	+ACI	0.04	0.09-1.16	90 / 220	NESCAUM, 2010
ESP ²	+ACI	0.06-0.07	0.14-1.06	90 / 240 and 140	NESCAUM, 2010
ESP+wFGD ¹	+ CaBr ₂	0.01	0.07 ⁸	73 / 500	NESCAUM, 2010
ESP+wFGD ²	+ CaBr ₂	0.01	0.07	73 / 500	NESCAUM, 2010
ESP+wFGD ¹	+ Pd catalyst	0.02	Not available	73 / 500	NESCAUM, 2010
ESP+wFGD ²	+ Au catalyst	0.03	Not available	73 / 500	NESCAUM, 2010

5.2. Non-ferrous metal production

118. Non-ferrous metal production includes the production of e.g. zinc, copper, lead and gold. The non-ferrous metals are produced from mined ores, which are treated in several process steps to extract the final product.

5.2.1. Origin of mercury and emissions from non-ferrous metal production

119. Mercury emissions from non-ferrous metal production depend mainly on the content of mercury in the non-ferrous metal ores used and the type of industrial technology, as well as the control technology employed in the production of non-ferrous metals. Mercury occurs as an impurity in many sulphide ores. In these ores, the mercury may take the place of the elements zinc, copper, cadmium, bismuth, lead, and arsenic. In some metal ores it also occurs as elemental mercury or as an alloy with other metals (amalgams). In some of these deposits mercury contents were high enough to allow an aimed production of mercury as a by-product. A global average of 123 mg mercury/kg in zinc ore resulting in a mobilization of 600 t mercury during zinc production has been estimated (Hageman et al 2010 and references therein). Most of the mercury associated with copper is found in massive sulphide deposits.

⁸ O&M costs are based on expert statements for the Para29 study (KNX technology) of 1.6 million USD per year.

The mercury fraction is a function of the zinc concentration and the environmental conditions during formation of the deposit. Mercury is commonly found associated with gold deposits although the amount of mercury in gold ore can vary widely, from less than 0.1 mg/kg to over 100 mg/kg (Hagemann et al, 2010 and references therein).

120. The industrial technology employed will to a large extent determine the fate of the mercury contained in the ore. If high temperature processes (i.e. roasting and sintering) are used in the initial treatment of the ore, the mercury will be released to the gas phase whereas if electrolytic processes are employed, the mercury will remain in the liquid phase (Figure 5).

121. In the high temperature process most of the mercury in the concentrate is expected to be evaporated from the oxidation. The evaporated mercury follows the gas stream which can be cleaned by particle filters and dry and wet electrostatic precipitators (ESPs), or scrubbers, resulting in either dry solid wastes or sludges containing mercury (UNEP toolkit). Sulphuric acid is often co-produced at non-ferrous plants due to most ores being rich in sulphur. The sulphur, as well as the mercury, is released from the concentrate during the initial oxidation in the roasting/sintering step and the mercury containing sulphuric gas stream is passed on to a sulphuric acid plant. Mercury specific removal techniques may be installed to ensure a low enough mercury content of the produced sulphuric acid. Remaining mercury from roasting or sintering is expected to follow the residue to recycling or deposition.

122. The leaching process as a part of the electrolytic process, results in a mercury containing liquid leach product and a solid residue. Parts of remaining mercury may follow precipitates to further processing from the purification process. No data has been identified on mercury emissions from the electrolysis step (UNEP Toolkit).

123. The smelting process can include secondary materials which in principle could represent a source of mercury. Mercury in a smelter feed will be volatilized and enter the gas stream.

124. Mercury emissions from fuels used in heating processes are regarded to be less significant.

125. The main steps for primary non-ferrous metals production and the mercury flows within the steps can be observed in Figure 5.

5.2.2. Technologies for non-ferrous metal production

126. There are many technologies that are used for primary production of non-ferrous metals. Distinction can be made between the more widely used hydrometallurgical (electrolysis) process, and the pyro-metallurgical (thermal) processes.

127. The pyro-metallurgical process treats minerals and metallurgical ores and concentrates thermally to extract the metal, thus most pyro-metallurgical processes require energy input to sustain the temperature at which the process takes place. The energy is usually provided in the form of fossil fuel combustion.

128. In the hydrometallurgical process, chemistry is used for the recovery of metals from ores, concentrates, and recycled or residual materials.

129. Different technologies and combinations of technologies can be employed in the steps shown in Figure 5. For detailed descriptions of various technologies for non-ferrous metal production, please refer to the EC Draft Reference Document for Non-ferrous metal industries at <http://eippeb.jrc.ec.europa.eu/>.

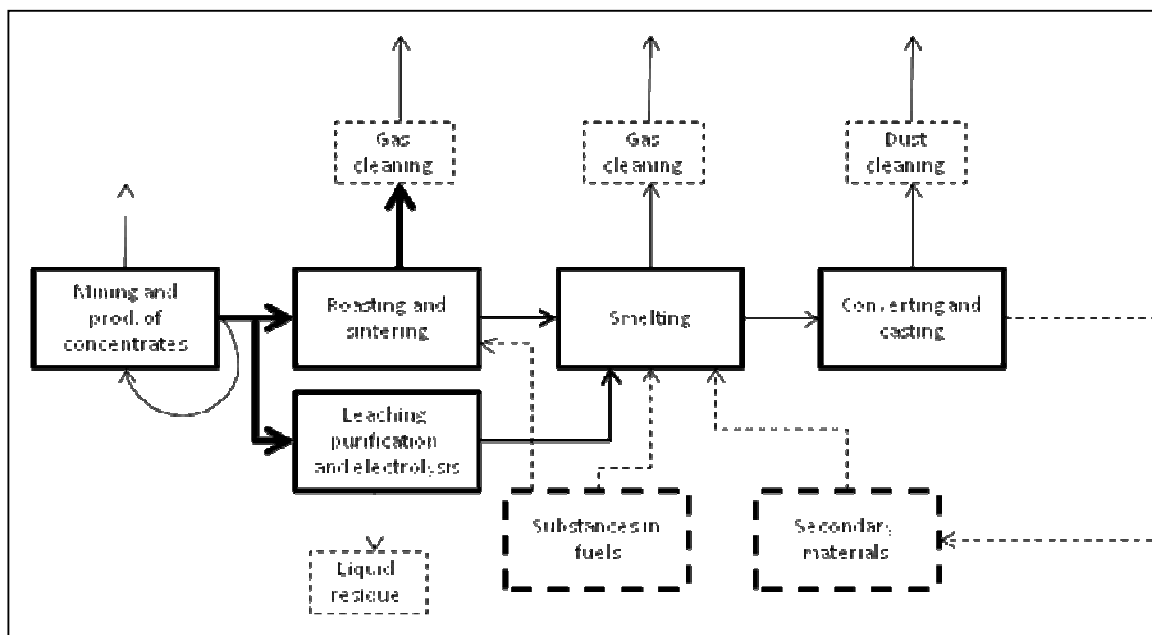


Figure 5. Main steps for non-ferrous metals production and the mercury flows within the steps. Roasting/sintering and Smelting are the high temperature processes where mercury may primarily be emitted to air.

5.2.3. Control measures in non-ferrous metal production

130. Non-ferrous metal ores (e.g. copper, zinc, pyrite, lead and gold) often contain mercury as a trace constituent. General technical air pollutant control technologies described in chapter 4 are used on non-ferrous metal production facilities. The air pollution controls can also, to a varying degree, capture mercury in the flue gases. In addition to co-benefits from common air pollution control technologies, specific mercury removal techniques have been developed for use in non-ferrous metal production. Development of mercury control technologies for non-ferrous metal production facilities has been driven both by ambitions to reduce air emissions but also to prevent mercury from contaminating the sulphuric acid, which can be produced after the roasting of sulphur-containing minerals. Some mercury will be removed from the flue gas via precipitation (in e.g. scrubber or filter units for air pollution control) with selenium or sulphate already present in the ores.

131. Measurements at a hydrometallurgical zinc smelter in China (Wang et al 2010) showed mercury removal efficiencies of air pollution control devices handling waste gas from the roasting furnace. The mercury removal efficiency measured for each step is shown as an average per cent reduction value \pm a standard deviation. After a waste heat boiler, cyclone and ESP, the gas goes through a flue gas cleaning step which includes a wet scrubber using recycled dilute sulphuric acid. This flue gas cleaning equipment had a mercury removal efficiency of about 17.4 ± 0.5 per cent. The next step is an electrostatic demister to remove water vapour, which removed 30.3 ± 10.9 per cent of the mercury. The mercury removal efficiency of the mercury reclaiming tower was 87.9 ± 3.5 per cent. This reclaiming tower uses the Boliden Norzink process take mercury out of the gas before the sulphuric acid plant. The acid plant itself removed 97.4 ± 0.6 per cent of the mercury from the gas entering the acid plant.

132. Hylander and Herbert (2008) in their emission inventory of mercury from non-ferrous metal production assume mercury removal efficiencies of between 95-99 per cent if a sulphuric acid plant is present at a copper, lead or zinc pyrometallurgical smelter, assuming that mercury was prevented from contaminating the sulphuric acid product. If only ESPs and/or scrubbers are installed, the assumptions are of 80 per cent mercury removal if the ESP and scrubbers are efficient, and 40 per cent if less efficient. If there is limited or no sulphur removal devices installed and no mercury specific controls, the assumed removal of mercury is 10 per cent.

133. Several different methods are available for specific removal of mercury from the flue gases in the non-ferrous metal industry. Processes where mercury is converted to a solid chemical compound which can be removed via precipitation, scrubbing or filtering include the Outokumpu and Bolkem processes where the mercury is precipitated as solid sulphate compounds and the Boliden Norzink process where mercury is precipitated in the form of mercurous chloride. Alternative methods include carbon or selenium filters (for low concentration treatment) or selenium scrubbers (UNEP toolkit, 2010). Additional information is available on http://www.sulphuric-acid.com/techmanual/GasCleaning/gcl_hg.htm

134. According to information submitted in response to the questionnaire, there is a range of mercury specific removal technologies used in industrial gold production in the USA. These are in addition to the uses of particulate matter control by bag filter or ESP on most plants, and SO₂ controls with e.g. wet scrubbers on some plants. The mercury specific controls include mercury condenser in combination with various types of carbon adsorption beds or filters, giving a removal efficiency of 93-99.6 per cent. Also scrubbers are installed at some plants, e.g. mercurous chloride or hypochlorite injection scrubbers that achieve high mercury removal efficiencies.

135. With regard to industrial gold production, the Nevada Department of Environmental Protection (NDEP) implemented the Nevada Mercury Control Program (NMCP) in 2006 to achieve reductions in air emissions from precious metals operations (i.e., gold and silver production facilities). Although a number of the industrial gold production facilities in the U.S. already have effective mercury specific controls, this NMCP program will require facilities to use best available mercury air emission control technologies at all gold and silver mine production facilities in Nevada that use any thermal process (e.g., furnaces, roasters, autoclaves, kilns, electrowinning, retorts, etc.).

136. Furthermore, the USA is developing national regulations for mercury emissions from industrial gold production. If the regulations proposed for public comment on April 28, 2010 were fully implemented, along with the Nevada Mercury Program, the US estimates that national mercury emissions from this industry would be reduced by about 94 per cent from 1999 emissions levels, and more than 96 per cent from potential uncontrolled emissions levels. A reduction of mercury emissions from ore pre-treatment processes (e.g. roasting) would be 94 per cent compared to uncontrolled conditions and 64 per cent in comparison to 2007 conditions. From the subsequent gold production steps, including e.g. kilns, retorts and smelting, a reduction of 98-99 per cent in comparison to uncontrolled conditions and approximately 90 per cent in comparison to 2007 conditions is estimated.

137. It can be concluded that there are many different options for mercury removal in the non-ferrous metal industry. However, due to the various non-ferrous metal ores, each deposit of which may have significantly different characteristics, it is virtually impossible to generalise about feasibility, costs, etc. (UNEP, 2006).

5.2.4. Costs and efficiencies of control technologies for non-ferrous metal production.

138. Large non-ferrous smelters use high efficiency air pollution control devices to control particle and SO₂ emissions from roasters, smelting furnaces, and converters. Mercury is emitted mostly in a gaseous form and therefore, the fabric filters or ESPs are not very effective in the element removal. Control of flue gas emissions is achieved by absorption of sulphur dioxide in the sulphuric acid plants, which are commonly a part of the smelting plants. Combination of dry ESP, wet scrubbers, mercury removal and wet ESP is considered as BAT in the EU (EC, 2001a). The amount of remaining emissions depends on the content of mercury in the ore.

139. Out of the mercury removal technologies, specific to non-ferrous metal production, the Boliden-Norzink process is reported to have the lowest capital and operating costs, despite some energy cost of operating the scrubber. (UNEP, 2006, chapter 5.3, production of recycled (non-ferrous) metals). No data on costs were found regarding the Boliden-Norzink process. As noted in section 5.2.3, the sulphuric acid plant itself has the unintentional effect of removing mercury from the gas as well.

140. Annualized costs of selected multi-pollutant reduction technologies per production of 1 tonne of metal production, indicated as specific activity indicator, and its efficiencies for mercury emission reduction were analysed within the ESPREME project (<http://espreme.ier.uni-stuttgart.de>). Emission control installations with lower potential for mercury removal ranging from 5 to 10 per cent (e.g. ESPs) are rather inexpensive, with the annual total cost in the range between 0.1 and 0.2 USD 2008/ tonne of metal produced. Fabric filters with similar Hg emission reduction potential are almost one order of magnitude more expensive than ESPs for both primary lead and primary zinc production. This difference is even bigger for the primary copper production sector reaching the total annual cost from 15.0 to 30.0 USD/tonne of copper produced.

141. Mercury specific measures, such as activated carbon injection used in combination with ESPs, fabric filters and FGDs in primary lead and primary zinc production increase the total annual cost by a factor of about 3 in comparison with ESPs/FGDs or FF/FGDs used alone (Pacyna et al 2010).

142. Detailed information on costs and efficiencies for reducing mercury emissions from gold production is available from the US as a result of the US EPA process of developing a proposed mercury regulation. Assessment of costs and efficiencies for industrial gold production in Nevada has resulted in estimates of emission reductions of 99.7 per cent from controls including gas cooling, PM control and the Boliden Norzinc calomel scrubber. Installation costs were about \$3.5 million and operating costs have been estimated to be \$660,000 per year for controls that have been in place for about 10 years. For other thermal units, in the gold production industry such as kilns, retorts and furnaces, the best controls are sulphur-impregnated carbon filters, which achieve about 93 per cent to 99 per cent reduction. The cost estimate for installation of a carbon adsorption system at one of the facilities (with a volumetric flow rate of 2500 cubic feet per minute) was about \$184,000. The estimated annual labour costs for this carbon adsorber is about \$10,000 per year and the estimated carbon replacement cost is \$54,000 per year. More details are available at the NDEP web site: <http://ndep.nv.gov/baqp/hg/clearinghouse.html>

5.3. Cement production

5.3.1. Origin of mercury and emissions from cement production

143. Mercury is naturally present in raw materials (e.g. limestone) used for cement production, and also in fuels used in the process, leading to mercury entering the cement kiln system.

144. Natural raw materials for cement production like limestone, chalk marl, shale or clay are extracted from quarrying and mining operations (e.g. rock drilling, blasting, excavation, hauling and crushing), before they are crushed in a mill or press and made chemically homogeneous by mixing the raw materials in controlled proportions.

145. The raw output (dry powder or slurry) from the extraction and preparation process is fed into the kiln and treated thermally to produce clinker. The thermal treatment involves 1) drying and preheating, 2) calcinations and 3) sintering. After being cooled, the clinker is milled together with gypsum (CaSO_4) to make cement.

146. Because of its properties, power station ash (fly ash or bottom ash), blast furnace slag, or other process residues (iron slag, paper sludge, silica fume, pyrite ash, and phoshogypsum) can be mixed into the cement. The fly ash used may contain mercury (<0.002-0.8 mg/kg according to Renzoni et al, 2010) but it is difficult to assess how much of mercury enters the environment through this pathway.

147. Regardless of the types of processes and kiln type (e.g. wet- and dry-kiln processes), mercury is introduced into the kiln with raw materials, where the mercury content varies from region to region, and with fuels of varying mercury content. The main output of mercury in the clinker manufacturing process is expected to occur in the kiln where mercury leaves the kiln with the dust and exhaust gas.

148. Very small amounts of mercury that is not emitted to air or captured in emission control equipment can remain in the cement kiln dust or in the final cement product. According to the UNEP toolkit the content in cement is expected to be between 0.02 – 0.1 g mercury per tonne of cement, and Renzoni et al (2010) present average values from studies for the US and Canada of 0.014 g mercury per tonne and for German cements of 0.06 g/ mercury per tonne, with ranges of individual values from below detection limit to values well above the UNEP toolkit range..

149. Since cement production includes energy intensive processes, fuel use consists of a large part of the production costs. Various fuels are used to generate the heat required for the clinker manufacturing process, mainly pulverized coal (black coal and lignite) and petroleum coke, heavy fuel oil and natural gas, but also other, less expensive waste fuels, such as shredded municipal garbage, chipped rubber, and waste solvents are widely used and may contribute to mercury emission from cement production (Pacyna et al., 2010). According to the IPPC (Integrated Pollution Prevention and Control) document on BAT (Best Available Techniques) in the cement manufacturing industries (EC, 2010), fuel consumption in the European cement industry in 2006 was dominated by petroleum coke (39 per cent), coal (19 per cent) or a mixture of both petroleum coke and coal (16 per cent), but they are also using fuel oil (3 per cent), lignite and other solid fuels (5 per cent), gas (1 per cent), as well as different types of waste fuels (18 per cent). According to the US response to the questionnaire, coal (63 per cent) and petroleum coke (21 per cent) are the fuels used in largest amounts, followed by alternative fuels of various kinds (12 per cent), natural gas (3 per cent) and minor uses of other fossil fuels. In Brasil petroleum coke dominates among the fuels, but charcoal and alternative fuels are also used according to responses to the questionnaire.

150. The origin of the mercury emitted from cement production can be both from impurities in raw materials and from the fuels used. The relative contributions vary between plants and between countries and there is no single general truth whether fuels or raw materials contribute most to mercury input to the kiln. A US EPA study (EPA, 2009) on national emission standards for hazardous air pollutants from the Portland cement manufacturing industry, showed that for about 55 per cent of the kilns in the US, non-limestone mercury accounted for greater than 50 per cent of the kiln's mercury emissions (i.e. origins from other raw materials or from fuels). It should be noted, however, that the origins of mercury was highly variable among the investigated kilns. In a study by the Université de Liège it was concluded that in most cases, the major contributors to the total mercury input into the kiln system in Europe are the natural raw materials and not the fuels (Renzoni et al, 2010).

5.3.2. Control measures in cement production

151. Primary measures to reduce mercury emissions in cement production are careful selection and control of substances entering the kiln (EC, 2010), thus selection of raw materials and fuels with low contents of mercury is important. Another primary measure is to ensure efficient operation of the plant, e.g. to reduce fuels or control the temperatures and gas flows at different stages in the process to make use of the potential of controlling the mercury flow through the system. In cement production there are process integrated mechanisms and operating conditions that allow cement kilns to enhance mercury capture and reduce mercury emissions to air (Senior, 2010, Renzoni et al, 2010).

152. For example the mercury adsorption on particles within the dry kiln system can be enhanced by passing exit gases from the preheater through the drying/grinding of raw material in the raw mill. The raw mill can either be on-line (in operation) or off-line (shut down). When the raw mill is on-line, the gases from the preheater pass through the mill before going to the particulate control device (Figure 6). When the mill is off-line, the exhaust gases bypass the raw mill and go directly to the particulate control device after being cooled down.

153. The high dust loading and increased contact time between solids and the gas when the raw mill is on-line allow for more mercury adsorption on solids before the particle control device (Renzoni et al 2010). A project by the Portland Cement Association which included the collection and review of stack emission data (Schreiber et al, 2009) concluded that preheater/precalciner kilns have significantly higher emissions of mercury to air when the raw mill is off, which is due to the missing adsorption capacity of the freshly ground particles in the raw mill. Turning off the raw mill is a regular part of cement plant operations.

154. Volatilised mercury condenses at relatively low temperature (120-150°C) on raw material particles in the kiln system (UNEP Toolkit, 2010). With a low enough temperature of the flue gas, mercury adsorbed on particles will be collected together with the particles in a particle collection device. It is common that the dust collected in the particle control device (often called cement kiln dust, CKD) is recycled back into the production process (Senior, 2010).

155. One way of reducing mercury emissions to air is periodic purging of CKD to remove mercury adsorbed on the particles from the system. The efficiency of this measure depends, in part, on the quantity of dust removed from the system, and on the temperature prevailing in the air pollution control device (Renzoni et al 2010).

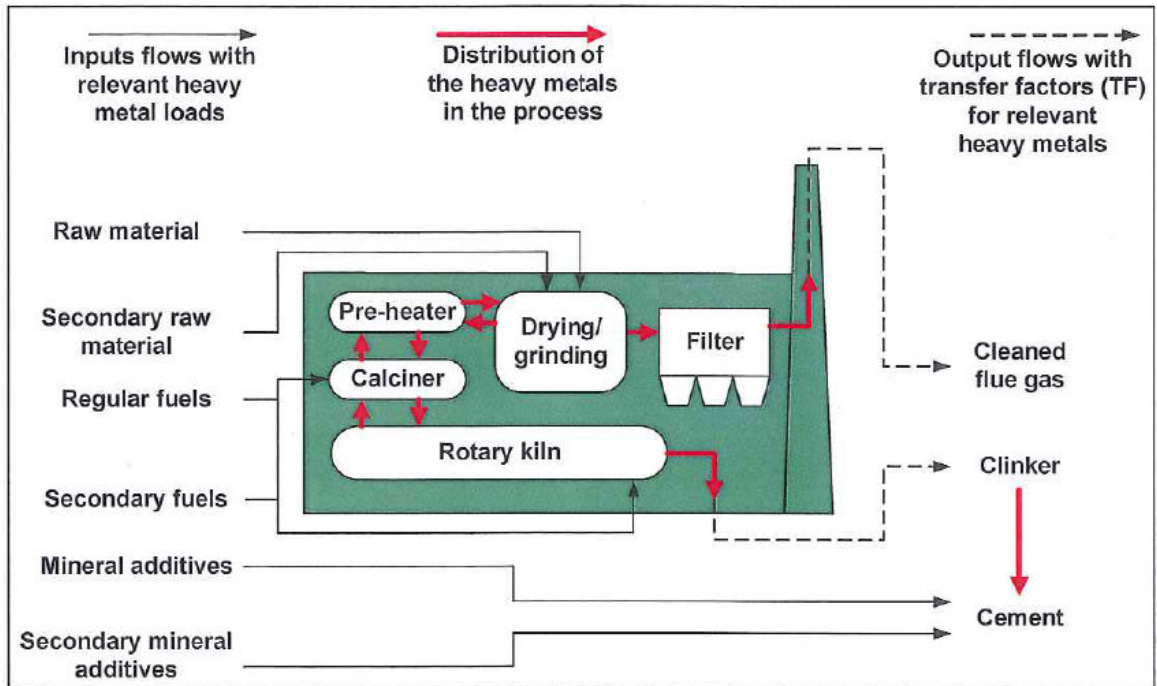


Figure 6. Pathways of metals in cement kilns with preheater in a dry process (EC, 2010 and references therein).

156. From submitted information from the countries (see Annex I for information on the countries providing information) it is obvious that the most common air pollution control equipment on cement plants in those countries are various types of particle control devices. According to the information submitted, particle controls such as ESP or bagfilter/fabric filters are common on cement plants (Brasil, Cyprus, Iceland, UK, USA). Also SNCR are reported for a number of the plants in the UK and some in the US. In the US, some plants are also equipped with wet or dry scrubbers for SO₂ removal. From submitted information there was no indication on any mercury-specific controls installed on any cement plant with one exception. U.S. informed of one cement kiln which is currently operating an activated carbon injection system specifically for mercury control.

157. The use of abatement technology such as fabric filters or ESPs is common on cement plants, but information on removal efficiencies of mercury specifically from cement kilns is scarce. It can, in general, be expected that mercury removal efficiencies in kilns may potentially be higher or are comparable to mercury removal efficiencies with similar devices employed in e.g. power plants, as long as the Cement Kiln Dust (CKD) is actively purged and disposed of.

158. According to Renzoni et al, (2010) emission removal techniques which have been tested in the cement industry include adsorption on activated coke (Polvitec) and activated carbon injection. Activated carbon injection is more complicated than on coal-fired power plants. Since it is common that the collected dust is recycled from the fabric filters as an integral part of the raw material processing system in the cement plant, the injection of activated carbon which would end up in the collected CKD is not desirable. The temperature of

this system would have to be carefully controlled to less than 200°C to ensure proper mercury adsorption and reduce the risk of activated carbon fires in the fabric filter or solids handling system. One option, if other measures fail, is to install an ACI system and an additional dedusting device downstream of the main kiln filter (Renzoni et al 2010).

5.3.3. Costs and efficiencies of control technologies for cement production.

159. One important aspect when estimating costs for mercury emission control are the assumptions of baseline conditions. The differences in costs for mercury removal will be very different if the baseline conditions are cement plants equipped with modern air pollution control or if very simple emission controls are installed. Some controls, such as electrostatic filters may be expected to have low efficiency for mercury removal due to high shares of Hg₀ in gases.

160. Annualized costs of selected multi-pollutant reduction technologies per production of 1 tonne of cement and its efficiencies for mercury emission reduction are presented below.

161. Based on information available from the BREFs on Cement, Lime and Magnesium Oxide production (EC, 2010), annualized costs⁹ (with capital cost in brackets) of selected multi-pollutant reduction technologies per production of 1 tonne of cement¹⁰ would on average be about 0.56 (0.34) 2010-USD and 0.69 (0.32) 2010-USD for the use of ESPs and FFs respectively. Solely introducing dry or wet scrubbers would lead to an average cost of 1.94 (1.12) and 2.63 (0.81) 2010-USD. Higher removal efficiencies (up to 90 per cent) can be achieved if combining FF and scrubbers at an average cost of 2.63 (1.13) 2010-USD.

162. The U.S. EPA cost analysis for installing activated carbon injection (ACI) to control mercury at a cement kiln includes a polishing baghouse. Most cement kiln dust is recycled back the kiln. Injection of activated carbon upstream of the primary baghouse could result in additional carbon in the cement kiln dust, making it unsuitable for recycling to the kiln if the process is not carefully controlled. The costs for mercury specific control in cement production units were estimated using costs that were originally developed for electric utility boilers. The capital cost factors ranged from \$1.81 to \$3.00/ tonne of clinker; the average is \$2.41/tonne of clinker. The total annualized costs ranged from \$0.96 to \$1.13/tonne of clinker; the average is \$1.41/tonne of clinker. A new 1.2 million tonne per year preheater/precalciner kiln would incur a capital cost of \$2.9 million and an annualized cost of \$1.25 million per year (2005 U.S. dollars). Due to the higher cost of a scrubber compared to ACI, the U.S. EPA expects that scrubbers would be installed for mercury control only when the scrubber also is needed for control of other pollutants. US EPA also estimates that ACI can achieve 90 per cent reduction in mercury emissions at cement plants (EPA, 2010).

⁹ Assuming 20- years technology lifetime with 4 per cent discounting.

¹⁰ Cement plant capacity of 3.000 t clinker/day.

5.4. Waste incineration

5.4.1. Origin of mercury and emissions from waste incineration

163. The main influences on the total emission of mercury to air from waste incineration is the mercury content in the waste, the waste burning capacity of the incinerator, the type of incinerator (mass burn excess air or modular starved air), the way in which it is operated (e.g. whether it includes heat recovery) and the degree of abatement fitted to the plant. Avoiding mercury entering the waste stream or pre-treatment of the waste such as separating and removing mercury-containing material before the waste is introduced into the incinerator is the most important primary measure to reduce mercury emissions to air. Mercury can be present in household waste in highly variable concentrations in different countries, mainly depending on the occurrence of mercury in household products and the existence of systems for collection of mercury or if the products are disposed on in the regular waste stream. Common product groups which may contain mercury are batteries, thermometers and fluorescent light sources. Mercury control in waste incineration is thus closely related to the issue of mercury in e.g. household products and medical devices, a fraction of which may end up in waste streams. Substitution of mercury containing products or introduction of effective systems for collection and safe disposal of mercury are thus alternative measures to avoid mercury emissions from waste incineration.

5.4.2. Technologies for waste incineration

164. It should be noted that large-scale waste incineration, which is the focus of this section, is not a common procedure in many countries. Other forms of waste management or lack of controlled waste handling systems will also give rise to mercury emissions via e.g. degassing from landfill deposits or uncontrolled burning of waste. The information presented here is thus only relevant to those countries where large scale waste incineration is an integrated part of the waste management system.

165. There are many different furnace designs and combustion techniques in use for municipal solid waste (MSW) incineration. In a typical incinerator, refuse is fed into the combustion chamber and placed on a grate that moves the waste through the combustor, mixing the waste thoroughly with the hot air to ensure effective combustion. Many incinerator designs have two combustion chambers. Air is supplied to the primary chamber through the waste (primary air). The incomplete combustion products (CO and organic compounds) pass into the secondary combustion chamber where additional air (secondary air) is added and combustion is completed.

166. There are three key classes of MSW incineration technology, which depend on the quantity and form of the waste burned. These are mass burn units, modular combustors and fluidised bed combustors (EMEP/EEA, 2009). In mass burn units, the MSW is incinerated without any pre-processing other than the removal of items too large to go through the feed system and removal of hazardous items, e.g. compressed gas cylinders. Modular combustors are similar to mass burn combustors as they burn waste that has not been pre-processed, but they are generally smaller, ranging in size from 4 to 130 tonnes waste/day. Fluidised bed combustors where waste is incinerated in a bed of inert material, (e.g., sand or ash) on a grate or distribution plate, which is fluidised with air. For the combustion of MSW in fluidized bed combustion (FBC), the waste has to be treated (e.g. shredded) in order to obtain a suitable size. (EC, 2006a)

167. Hazardous waste, which may have high mercury content, is usually burned either in special technology incinerators or in rotary kiln type furnaces. Special technology incinerators include very low technology drum type, grate type, or muffle type furnaces. Also, other technologies (such as supercritical water oxidation, and electric arc vitrification), which treat hazardous waste, can be included in this group (although they are not necessarily classified as “incineration”). Hazardous waste is in some countries incinerated at cement plants and light-weight aggregate kilns. In some countries medical waste is incinerated in hazardous waste incinerators or in municipal waste incinerators suited for the purpose (UNEP, 2005).

5.4.3. Control measures in waste incineration

168. Since all mercury emitted to air from waste incineration originates from mercury in the waste, the most important primary measure is to reduce the input of mercury to the incinerator. This can be accomplished by sorting out mercury containing products from the waste stream prior to incineration.

169. In waste incineration, the removal efficiency of the controls may be enhanced by adsorbing the mercury vapours from the combustion chamber onto acid gas adsorbent material or other adsorbents and then removing the particle-phase mercury. To achieve high mercury control, reducing flue gas temperature at the inlet to the control device to 175 °C (or lower) is beneficial. Typically, newer municipal waste combustion systems use a combination of gas cooling and duct sorbent injection (DSI) or spray dryer (SD) systems upstream of the particle removal device to reduce temperatures and provide a mechanism for acid gas control (UNEP, 2006).

170. Under incineration conditions at temperatures above 850°C and O₂ content of 8-10 per cent vol, the prevailing mercury species will be mercury chlorides (I and II) and elemental mercury (Velzen et al. 2002). The thermodynamically calculated chemical equilibrium for mercury in a typical flue gas containing HCl and SO₂ shows that the major product between 300 and 700°C is HgCl₂, whereas above 700°C elemental mercury is the dominant species. A summary of mercury removal efficiencies for different flue gas cleaning equipment in incinerators is shown in Table 9 (Velzen et al., 2002). These control measures are not specific for mercury and should be viewed as examples of general emission control technologies. For the estimation it is assumed that the HgCl₂/Hg(0) ratio is between 70/30 and 80/20. 'Special absorbents' (or adsorbents) added may be absorbents impregnated with sulphur or sulphur compounds or active carbon based adsorbents, which increase the sorption of mercury on particles.

Table 9. Mercury removal efficiencies of flue gas cleaning systems for waste incinerators (UNEP toolkit)

Equipment	Temp.	HgCl₂	Hg(0)	Overall**	Reference
	(°C)	per cent	per cent	per cent	
Electrostatic precipitator (ESP)	180	0-10	0-4	0-8	Velzen et al 2002
Wet scrubbers	65-70	70-80	0-10	55-65	Velzen et al 2002
Wet scrubbers with conditioning agent		90-95	20-30	76-82	Velzen et al 2002
Spray absorbers+ FF (limestone)	130	50-60	30-35	44-52	Velzen et al 2002
Spray absorbers+ FF (special absorbents added)*		90-95	80-90	87-94	Velzen et al 2002
Entrained flow absorbers + FF (special absorbents added)*	130	90-95	80-90	87-94	Velzen et al 2002
Circulating fluidized bed + FF (special absorbents added)*	130	90-99	80-95	87-98	Velzen et al 2002
ESP or FF + carbon injection				50->90	Pirrone et al 2001
ESP or FF + polishing wet scrubber				85	Pirrone et al 2001

* Special absorbents may be absorbents impregnated with sulphur or sulphur compounds or active carbon based absorbents, which increase the sorption of mercury on particles.

171. As shown in the table, simple electrostatic precipitators sometimes only have very low mercury removal efficiencies. Wet scrubbers or spray absorbents using limestone for acid gas removal has efficiencies of 55-65 per cent and 44-52 per cent, respectively. For obtaining high removal efficiency, more than 90 per cent, the addition of special absorbents/adsorbents, most often activated carbon, is a requisite.

172. Takaoka et al (2002) report an increased emission reduction for mercury of 20 to 30 per cent when introducing ACI in a pilot scale MSWI equipped with a bag filter (BF). Decreased flue gas temperature increases the Hg reduction ratio.

173. In the Republic of Korea, stringent regulatory requirements for dioxin and acid gases are in place and many air pollution control devices are installed in MWIs with a variety of configurations. The average mercury removal is around 85 per cent (Kim et al, 2010). Some facilities also inject activated carbon where mercury removal efficiency is higher. In industrial waste incineration removal efficiencies of 50 per cent-92 per cent were measured (less removal with dry type air pollution control devices and higher with wet). For HMIWI (hazardous, medical infectuous waste incineration) removal efficiency of around 89 per cent was presented, but removal efficiencies were dependant on the mercury content in the input waste.

174. From the responses to the questionnaire it can be concluded that on waste incineration plants it is quite common to have mercury-specific controls in addition to common air pollution devices. The mercury-specific technologies mentioned are for example activated carbon injection or scrubbers of various kinds.

5.4.4. Costs and efficiencies of control technologies for waste incineration.

175. For mercury removal adsorption using carbon based reagents is generally required to achieve the BAT level of emission ($< 0,05 \text{ mg/Nm}^3$ in case of non-continuous sampling) (EC, 2006a). The precise abatement performance depends on the levels and distribution of mercury in the waste. Some waste streams have very highly variable mercury content so additional waste pre-treatment is required.

176. One important aspect when estimating costs for mercury emission control are the assumptions of baseline conditions. The differences in costs for mercury removal will be very different if the baseline conditions are plants equipped with modern air pollution control or if very simple emission controls are installed.

177. Based on work performed in the EU funded project ESPREME, annualized costs of selected multi-pollutant reduction technologies per 1 tonne of municipal wastes and its efficiencies for co-control of mercury emissions have been derived. The information is derived for European conditions and may thus not be relevant elsewhere. For basic particle emission control (ESP or FF), total annual costs of 0.18 USD per tonne waste were estimated. For combinations of particle control with scrubbers, optimized ESPs and activated carbon injections estimated annual costs are in the range 4 – 12 USD per tonne waste. The co-control of mercury in these examples range from 5-10 per cent in the first case and between 80 and 99 per cent in the second example.

178. Average costs of emission control techniques for medical waste incineration (pre-dedusting with an ESP, followed by lime and activated carbon injection and finally a fabric filter) were estimated by Visschedijk et al. (2006) to be around 25 USD per tonne annual incinerated waste.

179. Since the costs related to appropriate incineration and land filling are high and medium respectively, in developed countries, these settings are economically hard to manage in least developed countries. Based on these high costs, opportunities to substitute mercury-free alternatives may be the most preferable option (UNEP, 2008).

180. In the responses to questionnaires, the USA reported costs for ACI on a typical large Municipal Waste Combustor (MWC) unit of 730 tonnes of waste/day, of \$150,000 capital and \$91,000/yr. annual operating costs (in 1987 U.S. dollars), derived during the development of regulations in the early 1990s. These numbers are roughly equivalent to 0.7 USD/year (annualised costs over 20 years and a 4 per cent discount in 2010 USD). These costs do not include costs for spray dryer/fabric filter system, which would add an additional $\$12 \times 10^6$ capital cost and $\$3.6 \times 10^6$ annual cost, if they were not already applied for controlling pollutants other than mercury.

181. For Hospital, Medical and Infectious Waste Incinerators (HMIWI), US EPA reported capital costs for the ACI system range from approximately \$3,800 to \$12,000. Annual costs for the ACI system range from approximately \$5,400/yr to \$56,300/yr. Both capital and annual costs are in 2007 U.S. dollars. Estimated capital costs for the activated carbon injection system (ACI) applied to a commercial or industrial solid waste incineration (IWI) unit range from approximately \$5,600 to \$156,000. Annual costs for the ACI system range from approximately \$2,900/yr. to \$3.2 million/yr. These costs for ACI applied to IWI include the incineration of industrial solid waste at cement kilns. Both capital and annual costs are in 2008 U.S. dollars.

182. Additional information on the origin of the costs in the above examples can be found in various documents prepared as a part of the US rulemaking for waste incineration sector (see US Submission of information to UNEP).

6. Conclusions

183. The purpose of this report is to provide a summary of mercury emissions to air, control options in selected sectors and their efficiencies and costs. It is prepared with the intention of supporting the on-going negotiation process to prepare a global legally binding instrument on mercury.

184. The study included preparation and distribution of questionnaires for the collection of information on technical characteristics of the selected sectors in different countries, current information on emissions and emission control as well as plans for future development of the sectors. The study focuses on describing potential emission control approaches and updating emission inventories.

6.1. Emissions

185. Mercury emissions from the selected sectors coal combustion in power plants and industrial boilers, cement production, non-ferrous metal production and waste incineration contributed 51 per cent to the total global emissions to air of mercury in 2005.

186. A review of recent national emission inventory reports and research papers on mercury emissions concluded that in most cases the new emissions estimates were in the range of emission values reported in the UNEP/AMAP (2008) study.

187. An evaluation of global emission trends including a harmonisation of data and improved completeness of inventory indicate that global emissions have varied from 1967 tonnes in 1990, a slight decrease to 1814 and 1819 tonnes in 1995 and 2000, respectively, to 1921 tonnes in 2005.

188. No major change in total global mercury emissions can be seen in this evaluation. However, of note is that a significant shift in emissions between regions has been observed. While Asian emissions increased in the period 1990 to 2005, emissions in North America and Europe decreased during the same period.

189. The accuracy of all current global emission estimates are limited by the lack of detailed facility-specific information and adequate representative monitoring data in many countries.

6.2. Emission control options

190. Mercury emissions from the four selected sectors are dependent on a number of factors including mercury content and other compositional characteristics of fuels and raw materials, the technical configuration of the source facility and existing emission control equipment for air pollutants (such as particle removal with ESP or FF and desulphurization in scrubbers) and mercury specific controls. These factors are highly variable in different regions of the world, as well as within regions, which makes identification of generally applicable control options challenging. For identification of suitable control options for a specific plant/facility, a detailed knowledge of the current technical status of this plant/facility is needed. For development of regional or national strategies for reducing mercury emissions, the economic status of the region also needs to be taken into account.

6.2.1. Basic conclusions on emission control in all the four selected sectors

191. A range of technologies to reduce mercury emissions exist and are in use in various countries for each of the sectors examined. Many of these are in place to reduce other air pollutants (e.g., particles, sulphur dioxide, nitrogen oxides, or other toxic pollutants).

192. Existing equipment to reduce such emissions can also reduce mercury emissions. However, the level of mercury emission reduction can vary significantly, depending on the characteristics of the input materials and details of the controls. This aspect needs to be taken into account both when assessing current mercury emissions, when considering the need for additional measures and in developing future scenarios and strategies for mercury emission control.

193. General measures aimed at improved performance and increased energy efficiency will lead to reduced consumption of fuel and raw material and thus indirectly to reduced mercury emissions.

194. Estimating costs for control of mercury emissions requires knowledge of the current status of the emission sources including technological descriptions of current and planned air pollution control.

6.2.2. Coal fired power plants and industrial boilers

195. Pre-combustion measures (e.g. coal washing, coal blending) can reduce mercury emissions by up to around 70 per cent, with an average of 30 per cent.

196. Pre-combustion addition of halogens (esp. bromine) has the potential to improve mercury removal by enhancing oxidation of mercury in the flue gas (especially in the presence of an SCR) and thus increasing the removal efficiency in downstream particulate matter control and flue gas desulphurisation equipment.

197. Activated Carbon Injection, when used at commercial scale in conjunction with a particle control device, e.g., ESP or fabric filter, can produce significant reduction of mercury emissions. Reductions of more than 90 per cent have been seen. Chemically treated carbons (e.g., brominated carbons) have been shown to be more effective than conventional, untreated activated carbon when treating flue gases containing higher amounts of elemental mercury vapour.

198. Detailed assessment of costs and efficiencies of mercury emissions control requires detailed information on the technical characteristics of the sources as well as on the availability of coal for blending and potential additives. Substantial data is available from field applications in the US.

199. Air pollution control devices may shift mercury from the flue gas to residues such as fly ash or FGD scrubber sludge. Assessments of total control costs should include potential increases in costs from management of these residues due to any potential impacts from changes in the composition of the waste, including increased mercury contents.

6.2.3. Cement production

200. Mercury emitted from cement production originates from coal and other fuels used and raw material such as limestone and other additives. Mercury concentrations are

highly variable in fuels and raw materials and significant emission reductions can be achieved by switching to lower mercury fuels and raw materials.

201. Since cement kiln dust (CKD) removed from stack gases in cement production to a large extent is re-circulated in the process, additions of activated carbon cannot be directly combined with existing particle control equipment if the temperature is not carefully controlled.

202. Adsorption of mercury to CKD in cement production is highly dependent on the operating conditions in the plant, where lower temperatures promote adsorption. Removal of mercury from the process can be achieved by removal of dust from the particle control device, if the temperature of the exhaust gases is low.

203. End of pipe control of mercury emissions in cement production can be achieved with similar control technologies as for coal fired power plants and significant mercury removal may be expected if flue gas desulphurization is introduced in order primarily to reduce acid gases. If activated carbon injection is employed to remove mercury, careful temperature control is needed and an additional particle removal step, where the collected particles are not recirculated may be necessary.

6.2.4. Non-ferrous metal production

204. Non-ferrous metal production is a highly complex industrial process with different configurations depending on which metals are extracted, the characteristics of the ore and which basic process is used. The mercury content of the ore can vary widely.

205. In pyrometallurgical processing of ores, the first step is roasting where sulphur in the ore is converted to sulphur dioxide gas. Any mercury in the ore will be released together with the gas.

206. Large non-ferrous smelters are often equipped with high efficiency air pollution control devices to control particle and SO₂ emissions from roasters, smelting furnaces, and convertors. Mercury is emitted mostly in a gaseous form and therefore, the ESPs are not very effective in the element removal. Control of flue gas emissions is achieved by absorption of sulphur dioxide in the sulphuric acid plants, which are commonly a part of the smelting plants.

207. Additional mercury specific removal (about 90-95 per cent) can be achieved using e.g. the Boliden Norzink technique where mercury is adsorbed in a solution of calomel (mercurous chloride) prior to the sulphuric acid plant.

208. For other thermal units in the gold production industry such as kilns, retorts and furnaces, the best mercury-specific controls are sulphur-impregnated carbon filters, which achieve efficient mercury emission reduction efficiencies in the range of 93 - >99 per cent.

6.2.5. Waste incineration

209. Since all mercury emitted to air from waste incineration originates from mercury in the waste, the most important primary measure is to reduce the input of mercury to the incinerator. This can be accomplished by sorting out mercury containing products from the waste stream prior to incineration.

210. Most modern municipal waste incinerators are equipped with advanced emission control for different pollutants. Some of these devices can be optimised for mercury e.g. by controlling temperature.

211. Mercury specific control can be achieved by carbon injection or scrubbers or both. Examples of costs for Activated Carbon Injection were reported from US EPA from several waste combustion rule processes.

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Annex I Summary of responses to the questionnaire

Responses were officially submitted from: Austria, Australia, Barbados, Brasil, Canada, China, Cyprus, European Community, Iceland, India, Lithuania, Poland, Russia, Seychelles, South Africa, Turkey, United Kingdom (England and Wales) and the USA.

The responses were of varying detail and coverage. Below, submitted data and some additional information from the responses are summarised. A few countries also submitted information indicating future developments of the respective sectors in terms of capacity and/or technological development and future development of air pollutant and mercury control status.

Coal combustion

A=aggregated information, I= information for individual plants

Source characterization for coal fired power plants and industrial boilers summary of submitted responses								
	Plant or unit information (No)	Capacity	Boiler type	Coal consumption	Hg in coal	Installed control technologies	Removal Efficiency	Emissions of Hg
Brasil	>7	A	A	A		A		
Canada	51							A
China	6242	A		A	A	A	A	
India	396	A	A	A		A	A (PM)	
Lithuania	58	A		A		A		
Poland	296							
Russia	120	I	I	I	I	I	I	I
UK	16							I
USA (electrical utilities)	1282 units 522 plants	I	I	I		I	I	I
USA (boilers/process heaters)	596 units 294 plants	I	I	I	I	I	I (PM)	I

PM=referring to particulate matter

Non-ferrous metal production

Source characterization non-ferrous metal production summary of submitted responses								
	Plant information (No)	Process information	Consumption of fuel/reducing agent	Hg contents in fuel	Metal production	Installed control technologies	Removal Efficiency	Emissions of Hg
Brasil	>10 companies	A	A		A			
Canada	12							A
China	8200	A	A		A			
Iceland	1	I	I	I	I	I	I (PM)	I
Poland	17+54							
Russia								A
UK	12							I
USA	28	I			I	I	I	I

PM=referring to particulate matter

Cement production

Source characterization cement production summary of submitted responses								
	Plant information (No)	Cement production	Fuels/consumption	Hg in fuels/raw material	Raw material consumption	Installed control technologies	Removal Efficiency	Emissions of Hg
Barbados	1	I	I		I			
Brasil	70	A	A		A	A		
Canada	17							A
China		A	A					
Cyprus	2	I	I		I	I		I
Iceland	1	I	I	I	I	I	I (PM)	I
Poland	53							
Russia								A
South Africa	11	I			I	I	I (PM)	
UK	12	I	I			I		I
USA	184	I	A	A	A	I		I

PM=referring to particulate matter

Waste incineration

Source characterization waste incineration plants summary of submitted responses							
	Plant information (No +waste type)	Amounts waste incinerated	Waste type	Hg in waste	Installed control technologies	Removal Efficiency	Emissions of Hg
Austria	2	I	I		I		I
Brasil	22 (IW, HZ)	A	A		A		
Canada							A
China		A					
Iceland	1	I	I		I		I
Lithuania	1 (ME)	I	I	I	I	I	I
Poland	1						
Turkey	1	I	I		I	I (PM)	I
UK	33		I				I
USA	96 (HZ)		I		I		I
USA	167 units/66 plants (HO)	A	I	A	A		I
USA	57 (ME)	I	I		I		I

PM=referring to particulate matter

IW=industrial waste, ME= medical waste, HZ= hazardous waste, HO=household waste

Annex II: Guidance for national assessment of atmospheric emission reduction strategies for mercury

This Annex provides general guidance for development of national strategies. For detailed assessment of alternative control technologies for reducing mercury emissions, local knowledge of the industrial sectors, economy and future plans for national legislation is necessary. In Figure 7 a generalized description of a proposed procedure for national assessment is presented in the form of a flow chart.

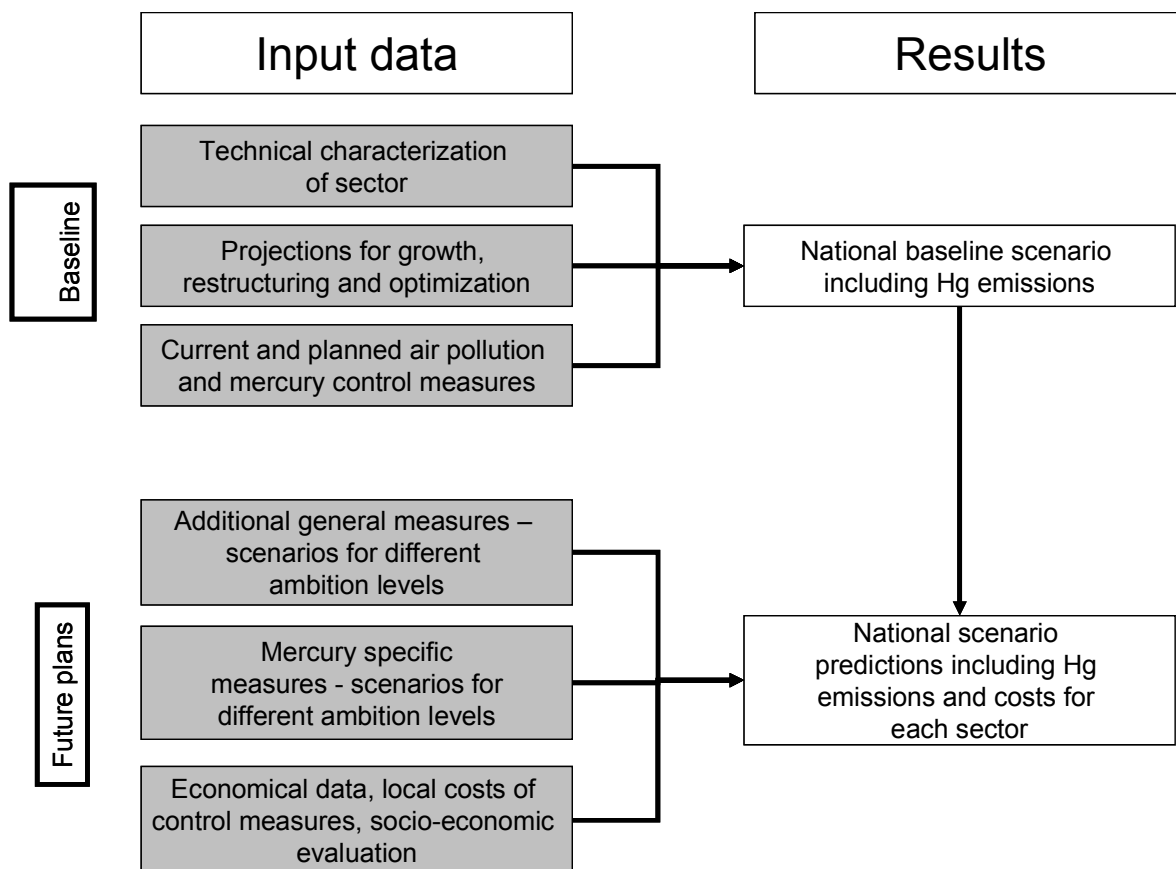


Figure 7. Schematic description of procedure to derive national strategies for mercury emission control to air

The necessary input information in this stepwise procedure for the preparation of the scenarios is outlined in detail in the questionnaires prepared for this study (available on the UNEP website) and can be summarised in the following way.

Preparation of baseline

The purpose of this step is to assess the current mercury emissions and changes in societal drivers for industry and energy (e.g. economic growth, legislation). The basis of the assessment is a characterization of the current technological status and preparation of a mercury emissions inventory for the selected sectors. A mercury emissions inventory can be based on either measurements or be estimated from activity data (energy, industrial production and consumption of raw materials) and emission factors, or a combination of both. Knowledge of mercury contents of fuels and raw materials allows an assessment of total mercury input to an industrial or energy process which, in combination with knowledge of technological status and current status of emission control can be used to estimate emissions.

Included in this step is also to assess how foreseen changes in societal drivers and implementation of existing and future national or international legislation will change industrial activities and mercury emissions. The results of this step are both a current mercury emissions inventory and an assessment of how this will change if no specific actions to reduce mercury emissions are taken.

Scenarios for additional emission control

The main component of his second step is to derive potentially applicable control strategies for reducing mercury emissions. The selection of these control strategies should be based on the information on technical characteristics derived in the baseline evaluation i.e. to be able to select measures that are technically and economically feasible in relation to the current status. Scenarios for different degrees of implementation of control measures or different degrees of modernization can be developed representing different ambition levels. Finally, the costs of implementation should be evaluated. Operating costs and to some extent investment costs will vary substantially with economic status, availability of services and consumables and other local factors. A country based assessment is thus required.

Additional information

As mentioned earlier in this report, the information received from the questionnaires distributed within this study did not result in sufficient information to prepare global scenarios. A few countries have performed their own studies using procedures similar to the one outlined above and which can provide practical guidance to others. Comprehensive information is available from documents prepared as a part of the legislative process to control mercury emissions from different sectors in the USA. One example is from the sector large scale gold mining where assessment of emission reduction potentials and associated costs has been prepared. Information on this process is summarised in chapter 5 in this report. More information can be found in the USA submissions on non-ferrous metal production to UNEP and references therein. The USA submission also includes detailed information on the other selected sectors.

Another example submitted to UNEP Chemicals is the report “Reducing mercury emissions from coal combustion in the energy sector” prepared by Tsinghua University, Beijing, China for Ministry of Environment Protection of China and UNEP Chemicals MEPC, in prep (2010) (also available on the UNEP website). In this report, basic information on the coal sector, mercury emissions, co-control of mercury in air pollution control installations and emission scenarios are presented and discussed.