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Review and assessment of options for enhanced voluntary measures and new or existing international legal instruments

Report presenting the costs and benefits for each of the strategic objectives

Addendum

Note by the secretariat

The annex to the present addendum contains the full text of the report referenced in document UNEP(DTIE)/Hg/OEWG.2/5.

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* UNEP(DTIE)/Hg/OEWG.2/1.

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Annex

UNEP Report

on

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

June 30, 2008

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Executive Summary

Mercury is an important environmental contaminant. This contaminant is toxic, persistent, and long-lived in the atmosphere, and can be transported globally. International action is required to reduce environmental and health risks at local, regional, and global scale.

A new assessment of the emissions of mercury is underway. A draft version of the UNEP report on emissions will be available as a draft at the second meeting of the ad hoc open-ended working group. Information from the UNEP emissions report has been used in the preparation of this report on cost benefit analyses.

This report presents a qualitative assessment of 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 (OEWG 1) that met in Bangkok 12-16 November 2007.

Costs have been assessed as including the economic costs of introducing the necessary equipment or actions to obtain the mercury reduction. Costs are defined as being small, medium and large, based on the highest cost of abatement for a given strategy (emission category).

Benefits of reducing mercury emissions include social, economic, ecological and human health benefits. For ingested mercury, the benefits are estimated to be $12,500 USD per kg of mercury[1]. For inhaled mercury, the benefits are between $1.34 and $1.22 per kg of mercury.

In conducting the cost-benefit analysis, the benefits are assessed on the basis of the impact of the reduction of mercury releases, and are then related to costs. Statements regarding the benefits of activities are base on the assumption that that the benefits are large if they exceed the costs by at least a factor of 2. If the benefits are equal or lower than costs, then it was assumed that the benefits are small. Medium benefits are between the large and small benefits.

While all strategic objectives specified have been assessed, assessment in detail was possible only where information was available. In particular, the costs and benefits of reducing emissions from coal burning have been addressed in some detail in this report.

In assessing ways to reduce anthropogenic mercury emissions, technological and non-technological measures have been assessed. A number of technological measures are available for reducing mercury emissions from anthropogenic sources where mercury is a by-product (e.g. power plants, smelters, cement kilns, other industrial plants), waste disposal and other uses. These measures differ with regard to emission control efficiency, costs, and environmental benefits obtained through their implementation. Very often mercury emissions are substantially reduced by equipment employed to reduce emissions of other pollutants. The best example is the reduction of mercury emissions achieved through the application of desulfurization measures.

The analysis also took account of the range of efficient, non-technological measures and pre-treatment methods are also available for the reduction of mercury releases from various uses of products containing mercury. These measures include ban on use and substitution of products containing mercury, and cleaning of raw materials before their use (e.g. coal cleaning). These

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[1] A conversion figure of 1 USD = 0.64 € has been used throughout this report.

measures also include energy conservation options, such as energy taxes, consumer information, energy management and improvement of efficiency of energy production through a co-generation of electricity and heat in coal-fired power plants.

The costs of reducing mercury emissions in this report are linked to the economic costs of introducing the necessary equipment or introducing other necessary actions to obtain the reduction. These costs include the investment costs and operational and maintenance costs.

A summary of the costs and benefits for each of the strategic objectives are presented in Table 1 below.

Table 1: Costs and benefits of Hg emission reduction for various reduction options

Reduction option		**Costs**	**Benefits**
1	Reduction from coal usage	Medium →Large	Large
2	Artisanal and small – scale gold mining	Small → Large	Small → Large
3	Reduction of Hg trade emissions	Small	Large
4	Reduction from industrial processes	Medium → Large	Medium → Large
5	Reduction of waste generation	Small → Large	Large
6	Promotion of Hg waste collection and treatment	Small → Medium	Large
7	Reduction from waste disposal	Medium → Large	Large
8	Reduction of Hg consumption in VCM and chlor-alkali production	Small → Large	Medium → Large
9	Reduction of Hg use in products	Small	Large
10	Reduction from dental practice	Small → Large	Medium
11	Reduction of supply from mining and extraction	Small → Medium	Large
12	Reduction of supply from decommissioned cells and stockpiles	Small → Medium	Large
13	Prevention of contamination from spreading	Large	Medium → Large
14	Control and remediation of contaminated sites	Small → Medium	Large
15	Increase of knowledge among states	Small → Large	Large
16	Increase of knowledge among users and consumers	Small	Large

It can be seen from this table that costs and benefits vary significantly between strategic objectives.

The final conclusion of the reported work is that there are benefits to investment in reducing mercury emissions and exposure in the future primarily for the sake of improvement of human health and more generally human welfare. Measures with the application of technology, such as implementation of installations to remove mercury from the flue gases in electric power plants, waste incinerators, and smelters are rather expensive (medium to large costs) compared to non-technological measures, such as prevention activity, capacity building, and promotion of mercury-containing waste separation (small to medium costs). Both groups of measures would result in large benefits, and parallel application of these, depending on resources would be appropriate.

Introduction

Mercury is an important environmental contaminant requiring action from policy makers, industry, and the general public. This contaminant is toxic, persistent, and transported long distances in the atmosphere and food chain. Coal combustion is believed to be the main source of mercury emissions to the atmosphere.

During the last decade major progress has been made in the assessment of emissions of mercury from various anthropogenic sources in various parts of the world. This progress has been reviewed by Pacyna et al. (2006) and has been used to assess the past, current and future emissions of mercury. It is estimated that the total anthropogenic emission of Hg in the year 2005 was about 1960 tonnes, distributed along various categories.

The largest emissions of Hg to the global atmosphere occur from combustion of fossil fuels, mainly coal in utility, industrial, and residential boilers. As much as 46.5 % of the total emission of Hg emitted from all anthropogenic sources worldwide in 2005 came from combustion of fossil fuels. Emissions of Hg from coal combustion are between one and two orders of magnitude higher than emissions from oil combustion, depending on the country. Emissions during the artisanal small scale gold production contributed about 17 %, followed by non-ferrous metal manufacturing, including gold (about 10 %), and cement production (about 9 %) (UNEP 2008).

Emission projections for mercury in 2020 were also estimated within this project (UNEP, 2008) and another project GLOCBA-SE prepared for the Nordic Council of Ministers (Pacyna et al.,2008). Three scenarios were developed: Status quo scenario, Extended Emission Control scenario and Maximum Feasible Technological Reduction scenario. The status quo scenario assumes that current patterns, practises and uses that result in mercury emissions to air will continue. Economic activity is assumed to increase, including in those sectors that produce mercury emissions, but emission control practices remain unchanged. The extended emission control scenario assumes economic progress at a rate dependent on the future development of industrial technologies and emission control technologies, i.e. mercury-reducing technology currently generally employed throughout Europe and North America would be implemented elsewhere. It further assumes that emissions control measures currently implemented or committed to in Europe to reduce mercury emission to air or water would be implemented around the world. These include measures adopted under the LRTAP Convention, EU Directives, and also agreements to meet IPCC Kyoto targets on reduction of greenhouse gases causing climate change (which will cause reductions in mercury emissions). The maximum feasible technological reduction scenario assumes implementation of all solutions/ measures leading to the maximum degree of reduction of mercury emissions and its loads discharged to any environment; cost is taken into account but only as a secondary consideration.

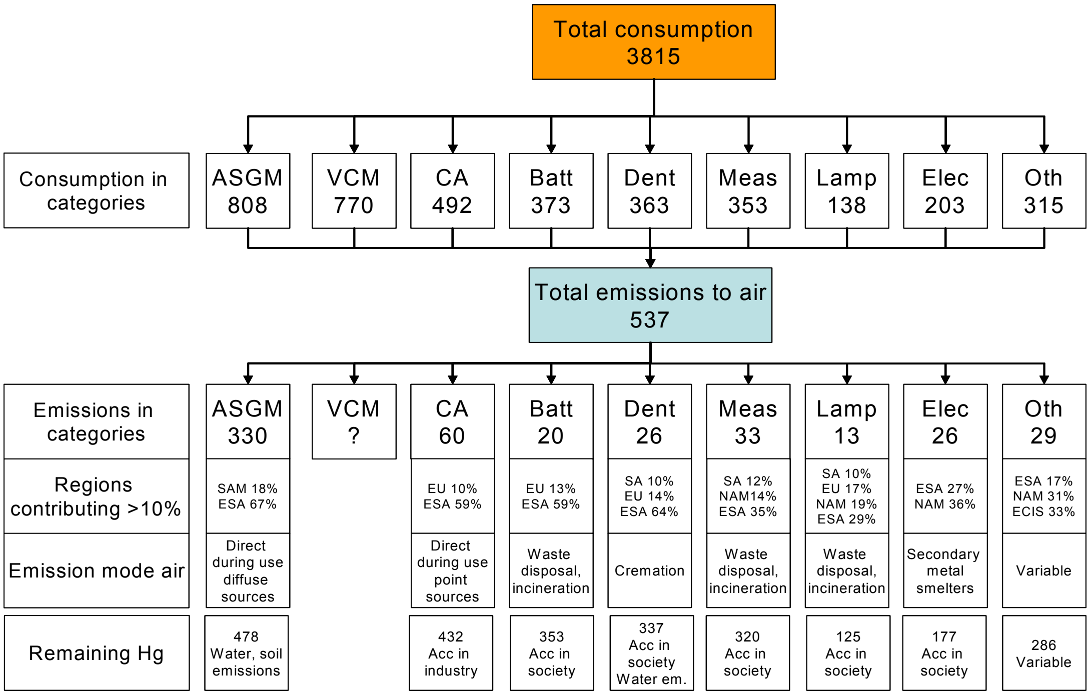
It can be concluded from the scenario estimates that a significant increase of about one third of the 2005 Hg emissions is expected in 2020 in the case that no major change in the efficiency of emission control will be introduced (the status quo scenario). A decrease by one third of the total emissions of mercury in 2005 can be expected in 2020 if the assumptions of the extended emission control scenario are met. As much as a half of the 2005 total emission can be reduced by 2020 if the assumptions of the maximum feasible technological reduction scenario are met. These scenarios are used as the basis for discussion on the costs and benefits of taking action on mercury reduction.

Mercury is intentionally used globally in a variety of industrial applications, products and other applications. Global consumption patterns have recently been assessed in UNEP (2008) where also emissions of mercury from intentional uses were estimated. Intentional uses of mercury were summarised by different geographical regions and by major use category. For purposes of estimating product related emissions, mercury 'consumption' was defined in terms of regional consumption of mercury products rather than overall regional 'demand'. For example, although most measuring and control devices are produced in China (reflected in Chinese 'demand' for mercury), many of them are exported, 'consumed' and disposed of in other countries.

The major mercury applications and intentional use sectors are:

- **Artisanal and small-scale gold mining (ASGM)**. The largest global user of mercury, reportedly continues to increase with the upward trend in the price of gold, and is inextricably linked with issues of poverty and human health
- **Production of vinyl chloride monomer (VCM)**, especially in China, is another area of major concern, especially as it is not yet clear where much of the mercury – estimated to be several hundred tonnes – goes as the catalyst is depleted.
- **Chlor-alkali production (CA).** The chlor-alkali industry is the third major mercury user worldwide. The mercury based technology is being phased out in many regions but continues to be used in others.
- **Batteries** The use of mercury in batteries, while still considerable, continues to decline as many nations have implemented restricting policies. Large quantities of batteries with low mercury content are still produced as are button cell batteries, containing up to 2% mercury.
- **Dental applications**. Some countries have implemented measures to greatly reduce the use of dental amalgams containing mercury and dental use of mercury is declining. However, the speed of decline varies widely, so that mercury use is still significant in most countries.
- **Measuring and control devices**. There is a rather wide selection of mercury containing measuring and control devices, including thermometers, barometers, manometers, etc., still manufactured in various parts of the world, although most international suppliers now offer mercury-free alternatives.
- **Lamps.** Mercury containing (fluorescent tubes, compact fluorescent, HID, etc.) lamps remain the standard for energy-efficient lamps, where ongoing industry efforts to reduce the amount of mercury in each lamp are countered, to some extent, by the ever-increasing number of energy-efficient lamps purchased and installed around the world.
- **Electrical and electronic devices**. Due to the RoHS Directive in Europe, and similar initiatives in Japan, China and California, among others, mercury-free substitutes for mercury switches, relays, etc., are being actively encouraged, and mercury consumption has declined substantially in recent years. At the same time, the US-based Interstate Mercury Education and Reduction Clearinghouse (IMERC) database demonstrates that mercury use in these devices remains significant.
- **Other applications of mercury**. This category has traditionally included the use of mercury and mercury compounds in such diverse applications as pesticides, fungicides, laboratory chemicals, in pharmaceuticals, as a preservative in paints, traditional medicine, cultural and ritual uses, cosmetics, etc. However, there are some further applications that have recently come to light in which the consumption of mercury is also especially significant such as the use of mercury catalysts in the production of polyurethane elastomers and the use of mercury in porosimetry.

In UNEP (2008) emissions of mercury from product categories have been calculated using distribution factors for the mercury consumed in the different products and emission factors to air for releases of mercury from the different paths of the mercury in the products. The general methodology is further described in Kindbom and Munthe (2007). In Figure 1, an overview of intentional mercury consumption and emissions is presented.



Abbreviations
Categories
ASGM: Artisanal and small scale gold mining
VCM: Vinyl chloride monomere production
CA: Chlor alkali production
Batt: Use and disposal of batteries
Dent: Dental amalgam (emissions only related to cremations)
Meas: Measurement and control devices
Lamp: Light sources
Elec: Electronic devices
Oth: Other uses e.g pesticides

Abbreviations
Regions
SAM: South America
SA: South Asia
EU: European Union
ESA: East and southeast Asia
NAM: North America
ECIS: Non EU European countries and CSI

Figure 1. Overview of intentional mercury use and emissions to air. Apart from the consumed and emitted amounts from different categories, information on the main regions where emissions occur, the main emission mode (i.e. source type) and an indication of the fate of the fraction of mercury consumed but not emitted to air ("Remaining Hg"). All figures expressed in tonnes

The overview presented in Fig. 1 provides a guide to the discussion of different management strategies discussed in the following chapters. The main point is that intentional mercury use can result in environmental emissions in various manners.

The costs of reducing Hg emissions is in this project linked to the economic costs of introducing the necessary equipment or introducing other necessary actions to obtain the reduction. In general, the term "cost" is often used when referring to both private cost and social cost, where the social cost is the sum of private- and external costs. When taking into account the social cost, this means that all costs in principle need to be internalised in the product price in order to give products their real price. Who is bearing this cost (the producer or the consumer) is determined by price- and market mechanisms often referred to as "elasticity" by economic theory. The effect of external factors such as pollution controls on production costs may either be absorbed by the producer, or passed on to the consumer, depending on this elasticity of the market

The benefits of reducing the Hg emissions include a spectrum of social, economic, ecological, and human health benefits. For example, mercury exposures through fish consumption (as well as other pathways), ,can cause a range of human health effects including neurological effects, including reductions in IQ (Intelligence Quotient) among children. Dietary methyl-mercury is almost completely absorbed into the blood and distributed to all tissues including the brain; it also readily passes through the placenta to the fetus and fetal brain. One of the measures of benefits is the prevention of IQ loss by reducing exposures. Other benefits to human health could include lower incidence of other types of neurological effects and lower incidence of some types of cardiovascular disease. Ecological benefits include less adverse effects to wildlife, while an economic benefit would be seen from fewer fish consumption advisories with a consequent boost for the recreational and commercial fishing industries. The benefits and costs of fish consumption advisories for mercury were discussed by Jakus et al. (2002).

Critical elements in estimating the cost of methyl-mercury exposure and risk from fish consumption would need to consider the species of fish consumed, the concentrations of methyl-mercury in the fish, the quantity of fish consumed, and how frequently fish is consumed. Those who regularly and frequently consume large amounts of fish -- either marine species that typically have much higher levels of methyl-mercury than the rest of seafood or freshwater fish that have been affected by mercury pollution -- are more highly exposed. Because the developing fetus may be the most sensitive to the effects from methyl-mercury, women of childbearing age are regarded as the population of greatest interest. In the United States, EPA believes that between 1 and 3 percent of women of childbearing age (i.e., between the ages of 15 and 44) eat sufficient amounts of fish to be at risk from methyl-mercury exposure, depending on the methyl-mercury concentrations in the fish. Advisories in the United States have been issued by 39 states and some Tribes.

The societal benefits due to the reduction of damage cost to the society from exposure to Hg pollution (societal cost) on global scale are now being studied within the GLOCBA-SE project (Pacyna et al., 2008). This project uses the results from the EU (European Union) DROPS project (DROPS D5.1 available from Pacyna, 2008). The overall objective of the DROPS project was to provide a full-chain analysis related to impact of health protection measures related to priority pollutants as identified by the EU Environment and Health Action Plan, in order to support the development of cost effective policy measures against pollution related diseases and their wider impacts. Mercury was one of the contaminants studied within the DROPS project. Neurotoxic impacts were found as the main human health end point for mercury. The damage cost data obtained in the DROPS project were estimated for inhalation of Hg polluted air and ingestion of Hg contaminated food, separately. The annual cost of $12,500 per kg of Hg was accepted for the ingestion pathway. In the case of inhalation, the amount of $1.34 /kg of Hg (the case for Poland) was used for the countries in Asia (except Japan), Eastern Europe, Africa and South America, while $2.21/kg of Hg was used for the rest of the world. These values were used in the GLOCBA-SE project to estimate the total damage cost to the society, defined as the societal cost. This cost is related to IQ loss, through loss of earning, loss of education, and opportunity cost while at school. The social benefits associated with IQ increase as a result of emission reductions were assessed taking into account the difference between the damage costs for the scenario with no improvement of Hg emission control (the status quo scenario) and the damage costs for the scenario with improvement of Hg emission control (the extended emission control Scenario. Based on a preliminary assessment, it was concluded that the annual social benefits can be as high as 11 billion US$ for this scenario.

The benefits of reduced mercury releases can be significantly higher for certain subpopulations that are more likely to be affected by fish contamination (e.g., the Native Americans and Asian Americans whose cultures include larger consumption rates of fish compared to the average American). Also, it is important to keep in mind that the cost per tonne reduction and benefit estimates are strongly correlated with baseline air emissions situations in countries, as well as with source and population information. The U.S. or the European Union estimates cannot be transferred and applied to tonnage reductions in another country given variable baseline mercury levels. The population near coastal regions may be higher, and the fish consumption rate may be higher than in the United States or in the European Union. These factors would lower the cost/tonne estimate and raise the benefits.

The reductions of Hg emissions can be obtained within various economic sectors, generating these emissions. UNEP Governing Council Decision 24/3 IV established an ad hoc open-ended working group (OEWG) of Governments, regional economic integration organisations and stakeholder representatives that will review and assess options for enhanced voluntary measures and new or existing international legal instruments to reduce risks from releases of mercury for each of the priorities set out in GC 24/3 IV, paragraph 19. Decision 24/3 IV also requests UNEP Chemicals, in its service as secretariat for the OEWG, "to prepare the analytical and summary reports necessary for [the OEWG's] work." The first meeting of the OEWG (OEWG-1) took place in Bangkok 12-16 November 2007. OEWG-1 adopted a program of inter-sessional work for the Secretariat, including the elements addressed in this Memorandum.

The main objective of the project UNEP-CBA reported here is to prepare a qualitative assessment of 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. It was agreed that this assessment will be general in nature quantifying costs and benefits as small, medium, and large for the following strategic objectives:

- reduce mercury emissions from coal usage,
- reduce mercury emissions from artisanal and small-scale gold mining,
- reduce mercury emissions from industrial processes,
- reduce generation of waste containing mercury,
- reduce emissions to air from incinerators and reduce migration and emission of mercury from landfills
- promote separate collection and treatment of mercury-containing wastes,
- reduce mercury consumption in VCM and chlor-alkali production,
- reduce mercury use in products, incl. Packaging,
- reduce mercury in dental practises

- reduce mercury supply from a hierarchy of sources,
- reduce international trade of mercury and mercury containing products, and
- increase knowledge of and capacity to manage mercury.

The purpose of this report is to provide qualitative assessment of costs and benefits. The costs defined as small, medium and large are related to the highest cost of abatement for a given strategy (emission category).

The benefits are then related to costs. It was assumed in this project that the benefits are large if they exceed the costs by at least a factor of 2. If the benefits are equal or lower than costs, then it was assumed that the benefits are small. Medium benefits are between the large and small benefits.

A recent review of socio-economic consequences of mercury use and pollution is presented in Appendix 1. This review has been published by Swain et al in Ambio, Vol. 36, No. 1 in February 2007 with a co-authorship of Jozef M. Pacyna (Swain et al., 2007). A summary of economic analyses that have been performed on the costs or benefits of reducing mercury emissions or just reducing exposure through fish consumption advisories is presented. This document can be regarded as a major introduction to the reported work.

1 Reduction of mercury emissions from coal usage

1.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Variable, ranging from small (if used as an incremental approach with other pollution reduction measures) to large.

Qualitative Benefit Assessment: Large emission reduction of mercury, both globally and locally with consequent health benefits; Reductions of other air pollutants.

1.2 Mercury emissions from coal combustion

Based on the latest inventories (particularly in UNEP, 2008), coal combustion is the largest emitting anthropogenic mercury source. The coal fired power sector is among the largest contributors to worldwide mercury emissions.

The Hg content in coal and the type and efficiency of emission control equipment are the most important parameters. The Hg content of coal varies from 0.01 to 1.0 ppm with an average of 0.1 ppm.

Various technologies within the same industry may generate different amounts of atmospheric emissions of mercury. It can be generalized for conventional thermal power plants that the plant design, particularly the burner configuration has an impact on the emission quantities. Wet bottom boilers produce the highest emissions among the coal-fired utility boilers, as they need to operate at the temperature above the ash -melting temperature (Pacyna, 1989).

Non-conventional methods of combustion, such as fluidized bed combustion (FBC) were found to generate comparable or slightly lower emissions of mercury and other trace elements than the conventional power plants (Carpenter, 1979; Abel et al., 1981). However, a long residence time of the bed material may result in increased fine particle production and thus more efficient condensation of gaseous mercury. Tests carried out in the former Federal Republic of Germany have shown that the residence time of the bed material can be regulated by changing the operating conditions of a given plant, the reduction of combustion temperature, coal size, moisture content, and bed flow rates (Munzner and Schilling, 1985). A literature review of information on the influence of various FBC techniques on emissions of trace elements has been presented by Sloss and Smith (2000).

The load of the burner affects the emissions of trace elements including mercury in such a way that for low load and full load the emissions are the largest (Bakkum and Veldt, 1986). For a 50 % load the emission rates can be lower by a factor of two.

1.3 Mercury from combustion of fuels other than coal

Major revision of recent data on the Hg content in crude oil indicates the concentration range from 0.01 to 0.5 ppm. It is expected that mercury concentrations in residual oil are higher than those in distillate oils being produced at an earlier stage in an oil refinery. Natural gas may contain small amounts of mercury, but the element should be removed from the raw gas during the recovery of liquid constituents, as well as during the removal of hydrogen sulfide. Therefore, it is believed that mercury emissions during natural gas combustion are insignificant.

The influence of plant design or its size on atmospheric emissions of mercury from oil-fired boilers is not as clear as for the coal-fired boilers. Under similar conditions the emission rates for the two major types of oil-fired boilers: tangential and horizontal units are comparable (Pacyna, 1982).

1.4 Mercury abatement measures and their efficiency

1.4.1 Pre-treatment methods of Hg emission control during coal combustion

Fuel washing and fuel substitution are the major pre-treatment measures to reduce emissions of various pollutants from coal combustion processes, including reduction of mercury.

Coal washing

Commercial coal cleaning (or beneficiation) facilities, particularly in the United States (e.g. NAPAP, 1990) are physical cleaning techniques to reduce the mineral matter and pyritic sulfur content. As a result, the product coal has a higher energy density and less variability (compared to feedstock coal) so that power plant efficiency and reliability are improved. A side benefit to these processes is that emissions of sulfur dioxide, as well as other pollutants including mercury can be reduced. The efficiency of this removal depends on the cleaning process used, type of coal, and the contaminant content of coal. Basic physical coal cleaning techniques have been commercial for over 50 years.

The cleaning of coal takes place in water, in a dense medium, or in a dry medium. Physical cleaning processes are based on either the specific gravity or surface property differences between the coal and its impurities. Jigs, concentrating tables, hydrocyclones, and froth flotation cells are common devices used in current physical coal-cleaning facilities.

The mercury concentrations in the raw coal, the clean coal, and the present reduction achieved by cleaning have been presented by Akers et al. (1993) for coals from various regions in the United States. The removal efficiency ranged from 0 to 60 % with 21 % as average reduction. Kraus et al. (2006) indicate that typically 10 to 50 % of the mercury in coal can be removed by in the cleaning process alone. This efficiency is highly dependent on the type of coal.

Fuel switching

The following options of fuel substitution are often considered in the electric utilities:

- switching from high- to low-sulfur coal burnt in applicable coal-based generation (including switching directly from high-sulfur to low-sulfur supplies, blending high- and low-sulfur coal, cleaning high- and medium-sulfur coal, or a combination of cleaning and blending),
- increasing the use of natural gas, or oil, and
- increasing the use of alternate fuels or importing electricity to meet base load electric-generation requirements.

The two latter methods are the most interesting with respect to the reduction of mercury emissions. The substitution of coal by coal-bed methane to produce heat and electricity would result in decrease of emissions of various air pollutants, including mercury. The following action would be needed in the case of the substitution:

- the modernization of existing utility and industrial heat producing plants,
- the development of new methane burning boilers, and
- the modernization of coal mines with respect to the better exploitation of coal-bed methane.

1.4.2 Primary measures to reduce mercury emissions during coal combustion

Primary measures of emission reduction include solutions where emission reduction occurs at emission generation point, e.g. application of various modifications of combustion process may reduce emissions from a given burner.

Non-conventional combustion technologies

Non-conventional methods of combustion, such as fluidized bed combustion (FBC) were found to generate comparable or slightly lower emissions of mercury and other trace elements compared to the conventional power plants (e.g. Carpenter, 1979, Abel et al., 1981). However, a long residence time of the bed material may result in increased fine particle production and thus more efficient condensation of gaseous mercury. Tests carried out in Germany have shown that the residence time of the bed material can be regulated by changing the operating conditions of a given plant, the reduction of combustion temperature, coal size, moisture content, and bed flow rates (Munzner and Schilling, 1985). A literature review of information on the influence of various FBC techniques on emissions of trace metals, including mercury has been presented by Sloss and Smith (2000).

Low NOx burners

Low NO_X technologies are also likely to reduce mercury emission in the exhaust gases due to the lower operating temperatures. Very limited information on this subject is rather inconclusive. While some sources indicate that the reduction can be achieved, preliminary results of staged combustion in atmospheric fluidized bed combustion (AFBC) units indicated that low NO_X had only little effect on trace element emissions (Smith, 1987).

1.4.3 Secondary measures to reduce mercury emissions from coal combustion

Secondary measures include technological solutions to decrease concentrations of mercury in the flue gas leaving the combustion zone.

Mercury enters the atmosphere from coal combustion in a gaseous form. However, de-dusting installations, such as electrostatic precipitators (ESPs) and fabric filters (FFs) can also remove up to 30 % of Hg from exhaust gases. One should note that ESPs are now commonly used abatement measures in major electric power plants and central heating plants worldwide.

The application of flue gas desulfurization (FGD) has a very important impact on removal of not only sulfur dioxide but also mercury. A number of studies have been carried out to assess the extent of this removal and parameters having major impact on this removal. These studies were reviewed in connection with the preparation of the EU Position Paper on Ambient Air Pollution by Mercury (EC, 2004). It was concluded that the relatively low temperatures found in wet scrubber systems allow many of the more volatile trace elements to condense from the vapour phase and thus to be removed from the flue gases. In general, removal efficiency of FGD installations for mercury ranges from 30 to 50%. It was also concluded that the overall removal of mercury in various spray dry systems varies from about 35 to 90%. The highest removal efficiencies are achieved from spray dry systems fitted with downstream fabric filters.

Higher Hg emission control efficiencies, exceeding 95 %, can be obtained through a combination of FGD and ESP's with "add on" type of equipment including sorbent injection. Sorbent injection generically describes the injection of powdered activated carbon (PAC) or other non-carbon sorbents into the flue gas for mercury control, while mercury oxidation enhancements are intended to improve the mercury capture efficiency of conventional control installations or downstream air pollution control devices by converting elemental mercury to a more reactive oxidized state (e.g. Jones et al., 2006)

Selenium scrubber is a wet media process for removal of fairly large quantities of mercury from the flue gas. The gaseous mercury reacts with activated amorphous selenium, which is circulating in a scrubber with a 20.0 to 40.0 % sulfuric acid. The mercury removal efficiency is between 90.0 and 95.0 %.

Carbon filter bed is another dry media process. Carbon filter bed technology is assumed by the U.S. EPA to remove 80 to 90 % of the mercury in the flue gas.

Lead sulfide process has also been recommended for removal of mercury. The flue gas containing mercury passes through a tower packed with lead sulfide coated balls. The removal efficiency of 99.0 % has been measured.

A detailed review of the current and developing mercury technologies and the control effectiveness that can be achieved from these technologies in the U.S. coal-fired power plants has been presented by IEPA (2006). The study has confirmed that depending on several variables, including coal and boiler type, there are a number of control technologies that will achieve 90+ % removal of mercury. Mercury emissions control technology is a rapidly advancing field, with use of halogenated sorbents that are becoming an affordable and effective option for many applications.

1.4.4 Emission control measures suggested for use within the UN ECE LRTAP Convention

An assessment of technological developments and the best available techniques for the implementation of the heavy metal emission reduction Protocol of the UN ECE Convention on Long-range Transboundary Air Pollution (Kraus et al., 2006) has recently been prepared. It was found out that the ESPs or FFs operated in combination with FGD and sorbent injection techniques are capable of removing between 75% and 90 % of mercury from the flue gases in coal-fired power plants in the additional presence of selective catalytic reduction. The following conclusions were reached with regard to the least costly retrofit options for the control of mercury emissions from units with ESP or FF:

- The modification of dry FGD systems by the use of appropriate sorbents for the capture of mercury and other air toxics is considered to be the easiest retrofit problem to solve;
- Injection of a sorbent upstream of the ESP or FF. Cooling of the stack gas or modifications to the ducting may be needed to keep sorbent requirements at acceptable levels;
- Injection of a sorbent between the ESP and a pulse-jet FF retrofitted downstream of the ESP. This approach will increase capital costs but reduce sorbent costs;
- Installation of a semi-dray circulating fluidized-bed absorber (CFA) upstream of an existing ESP used in conjunction with sorbent injection. It is believed that CFAs can

potentially control mercury emissions at lower costs than those associated with the use of spray dryers.

1.5 Cost of mercury abatement

1.5.1 Incremental cost of Hg abatement

The incremental cost of mercury reduction, i.e. the cost (in US$/kg Hg removed) to achieve a specific reduction is impacted largely by the level of baseline mercury capture exhibited by the existing air pollution control devices (APCD) configuration and coal mercury content. For example, the incremental cost of mercury control will increase when: (1) baseline mercury capture by existing APCD is high; or (2) the coal mercury content is low, because a smaller quantity of mercury is removed from the flue gas for a given level of control. In terms of raw monetary cost, reducing mercury from coal combustion can be quite expensive. The incremental cost of Hg emission reduction varies substantially depending on factors such as the type of coal used, the type of combustion unit, the type of control devices already in place to control other pollutants, the facility configuration, and the percent reductions expected. For example, wet scrubbers installed primarily for mercury have been estimated to cost between US$76,000 and US$174,000 per pound of mercury removed (or between US$168,000 and US$384,000 per kg of mercury removed). This result is very close to the cost of $234,000 per kg of mercury removed, estimated and used in a study of the effectiveness of the UN ECE heavy metals (HM) Protocol and cost of additional measures (Visschedijk et al., 2006).

Some years ago, the U.S. Environmental Protection Agency (EPA) had estimated that it would cost between US$67,700 and US$70,000 per pound (or between US$149,300 and US$154,000 per kg) to achieve a 90 percent control level using sorbent injection (US EPA, 2005). Since 1997, Research, Development and Demonstration activities sponsored by the Department of Energy (DOE), vendors and utilities relating to sorbent injection for mercury removal have shown significant advances along with the potential for reductions in overall installation and operational costs. More information on the cost estimates is available from the DOE/NETL’s Phase II Mercury Control Technology Field Testing Program (Jones et al. 2006), particularly with regard to the economic analysis of activated carbon injection method. This analysis was conducted on a plant-specific basis, meaning that the economics are dependent on the actual power plant operating conditions and coal properties observed during full scale field testing at each of the power plants taking part in the Program. Mercury control through activated carbon control was analysed. The 20-year levelized incremental cost of mercury control was found to vary from about US$ 8,400/kg Hg removed to about US$365,000/kg Hg removed.

1.5.2 Hg emission reduction as a co-benefit of reduction of emission of conventional pollutants

At present, it is uncommon for countries to invest in technologies to reduce only mercury from the emissions stream. Instead, countries usually use a multi-pollutant approach, which is much more cost effective. For example, approaches and technologies for controlling conventional air pollutants, including particulate matter, SO_2, and NO_x, typically result in some reduction of mercury emissions as a co-benefit, as mentioned earlier in this chapter. In most countries, mercury controls are contingent upon controls for conventional pollutants, although the degree of the mercury capture by various technologies varies widely. In this context, the incremental cost of adding a mercury reduction effort to a national strategy is much smaller.

Major review of information on the costs of abatement for combustion of coal and other economic sectors was carried out within the EU ESPREME (http://espreme.ier.uni-stuttgart.de) and DROPS (http://drops.nilu.no) projects. The annualized investment and operational costs for installations that are used to remove mercury, including ESPs, FFs, FGD, and "add on" measures just for mercury removal are presented in Table 1. These costs are given in relation to the production of 1 MWhe electricity in utility and large industrial boilers. The information on efficiency of Hg removal using these installations is also included in Table 1.

Table 1: Abatement cost for installations used to reduce Hg emissions from coal combustion processes (in US$/ MWhe) – selected technologies from the EU ESPREME project database (http://espreme.ier.uni-stuttgart.de)

Sector	Emission control technology	Hg reduction (%)	Annual costs (US$ 2008/MWhe)		
			Annual investment costs	Annual operating costs	Annual total costs
Hard and brown coal combustion	dry electrostatic precipitator (ESP) – medium emission control efficiency	24	0,45	0,90	1,35
	fabric filters (FF) – medium emission control efficiency	20	0,46	1,47	1,93
	dry ESP – retrofitted from medium to high control efficiency	32	0,92	0,52	1,44
	FF+wet or dry scrubbers+sorbent injection – state-of-the-art (BAT)	98	0,72	1,80	2,52
	dry ESP + wet or dry scrubber + dry injection -– state-of-the-art	98	2,73	2,40	5,13
	electro-catalytic oxidation – emerging method	80	8,55	11,76	20,31
	Integrated gasification combined cycle (IGCC) – emerging method	90			20,00

1.5.3 Examples of abatement cost estimates

In the United States, EPA promulgated a regulation in 2005 to reduce criteria air pollutant emissions from power plants, the Clean Air Interstate Rule (CAIR). The U.S. EPA calculated the estimated costs and some of the benefits of that regulation. The CAIR rule is primarily aimed at reducing emissions of SOx and NOx from large coal-fired power plants, but as a co-benefit will result in reductions of mercury emissions. The CAIR rule will achieve the majority of its mercury reductions as a co-benefit from controls for SO_2. Applying SO_2 controls (or other multi-pollutant approaches) are more cost-effective at reducing mercury than direct mercury control. EPA also promulgated the Clean Air Mercury Rule (CAMR) which was targeted to specifically further reduce mercury emissions from coal-fired power plants. The co-benefits of CAIR were estimated to reduce mercury emissions to 34.5 metric tonnes in 2010; the specific requirements of CAMR were estimated to further reduce mercury emissions to 13.6 tonnes by 2020. This could cost the U.S. electric power industry about US$ 11.3 billion

1.6 Benefits of Hg emission abatement

Information on the benefits and costs of reduction of mercury emissions from the coal combustion was recently reviewed by NESCAUM (2005). The NESCAUM study describes the results of a comprehensive assessment of the health benefits of reducing mercury emissions from coal-fired power stations in the United States. It has been anticipated that reductions in mercury emissions from coal-fired power plants decrease methyl-mercury concentrations in fish. A model has been developed assuming that equilibrium currently exists between deposited mercury and fish methyl-mercury concentrations and between fish methyl-mercury concentrations and methyl-mercury exposures to individuals who consume these fish. Changes in the quantity of mercury deposited are assumed to lead to leaner and proportional changes in fish methyl-mercury concentrations, assuming that no other factors change. The model accounts for human exposure through commercially and non-commercially harvested fish. Two potential health effects were accounted: cognitive abilities and cardiovascular events. The results from epidemiological studies were used to develop association between methyl-mercury exposures in males and increased risks of myocardial infarction and premature mortality. Using a Willingness-to-Pay (WTP) approach it has been estimated that the value of premature fatality is approximately US$ 6.0 million (in 2000 US$) but it was indicated that this value should be taken with caution. The NESCAUM study (2005) also described the possible benefits of the U.S. power plant mercury emission controls in terms of IQ increases in the annual birth cohorts. The predicted annual benefit associated with IQ increases in the annual birth cohort ranged from US$ 75 million to US$ 288 million, estimated within two scenarios related to different emission projection in the U.S. power plants.

The societal benefits related to the Hg emission on global scale were estimated in by Pacyna et al. (2008). These benefits were estimated as a difference between the damage costs estimated for the SQ scenario of emission reductions and the EXEC scenario. The two thirds of the societal costs due to Hg pollution are associated with the societal costs due to the Hg pollution of the environment by emissions from coal combustion. Based on preliminary results from this study, the annual social benefits associated with the IQ change due to Hg emission reductions from coal combustion worldwide can be estimated to more than $7 billion.

1.7 Summary of cost and benefits for coal combustion

An attempt was made to present the information on types and efficiency of emission reduction technologies for Hg in the coal combustion sector together with the investment and operational costs of these technologies and compared it with societal benefits due to the implementation of these technologies in the year 2020. The information needed for this comparison is presented in this chapter of the report. The results are presented in Table 2.

Table 2: Abatement costs and benefits in the year 2020 due to the reduction of Hg emissions from coal combustion using various emission control technologies, relative to the status quo scenario of pollution.

Efficiency of Hg emission reduction, %	**Abatement cost US$/g Hg abated**	**Societal benefits US$/g Hg abated**
0 - 30 (ESPs or FFs)	100	100
30 - 50 (ESPs or FFs + FGD)	190	320
50 - 99+ (ESPs or FFs + FGD + sorbent injection)	260	540

The benefits were estimated as the difference between the damage costs estimated for the SQ scenario of Hg emissions in the year 2020 (employment of ESPs or FFs only) and respectively the EXEC 2020 emission scenario (application of ESPs or FFs + FGD) and the MFTR 2020 emission scenario (application of ESPs or FFs + FGD + sorbent injection). The damage cost to the society due to exposure to Hg pollution (societal cost) was estimated on the basis of data available from the EU DROPS project (DROPS D5.1 available from Pacyna, 2008). These cost data were obtained in the DROPS project for inhalation of Hg polluted air and ingestion of Hg contaminated food, separately. The cost of $12,500.00 per 1 kg of Hg was accepted for the ingestion pathway.

Only neurotoxic impacts through the IQ loss were considered as the main human health end point for mercury. The total damage cost to the society, defined here as the societal cost, is related to IQ loss, through loss of earning, loss of education, and opportunity cost while at school.

The investment and operational costs were estimated using 1% discount rate. The comparison in Table 2 indicates that reduction of Hg emissions from coal combustion will result in benefits significantly higher than the cost of abatement. These benefits will be higher when the benefits other than the improvement of IQ are added.

2 Reduction of mercury emissions from artisanal and small-scale gold mining

2.1 Overall assessment of costs and benefits

There are a wide range of measures available within artisanal and small scale gold mining (ASGM) to reduce mercury emissions. An overview of the results is therefore presented after the overall assessments for costs and benefits of reduced emissions from ASGM.

Qualitative Cost Assessment: Variable, ranging from small to large.

There are several technical options available for the ASGM. The cost assessments of these measures are linked to the number of individuals affected by the option and the technical requirements of the options. Mercury vapour capture in gold shops is related to small costs thanks to the relatively small amount of gold shops and their immobility. The use of retorts in the mining process is related to medium costs since it affects a larger group of miners and requires educational efforts. To encourage mercury free sluice options is related to medium costs due to the technical requirement of the option.

Market mechanisms such as a decrease in gold prices or increase in mercury prices are associated with large costs since they will affect the economic situation for the miners. Micro credits given to miners that convert their mining activities into mercury free practices (where possible) may result in small costs overall as the credit is paid back.

Other mechanisms such as education of best practice and conversion to other livelihoods may incur large costs since they require much greater institutional efforts.

Qualitative Benefit Assessment: Variable, ranging from small to large

The benefit assessments of technical solutions are linked to the number of individuals affected by the option and whether the solution enables mercury free mining or not. Mercury vapour capture in gold shops is related to medium benefits since there is a large potential for mercury capture, from a relatively limit amount of emission sources. The use of retorts in the mining process has the potential to produce large benefits if broadly utilised, however the use of retorts has been related to small benefits since the adaptations are smaller in size and variable in quality. Retort use relies on individual mining communities being committed to reducing mercury emissions. The benefit from mercury free sluice solutions are related to large benefits since no mercury is required, although this benefit estimate may be reduced by mining situations where it is not feasible to use mercury free sluice solutions.

Market mechanisms such as decreases in gold prices or increases in mercury prices are estimated as giving small benefits since the price difference between gold and mercury is very large. This price difference reduces the motivation to altered mining technologies or reduced mining activities. Micro credits provided to facilitate the move to mercury-free technology may be related to large benefits since they would help in phasing out mercury use, however this may not be universally feasible.

Education is related to small benefits on a global scale since education in itself seems to need to be linked to other options and market mechanisms in order to be effective. It is difficult to quantify the effects of education on community groups, where increased awareness of the hazards of mercury may result in more significant behaviour changes. Conversion to other livelihoods for miners is related to small benefits since other potential miners are likely to take the place of the miners moving to other sources of livelihood.

For all the measures aimed at abating the release of mercury from ASGM activities, the benefits are both local and global in their nature. In relation to other type of emission sources, the benefits from reducing mercury use and release in ASGM has a stronger emphasis on local benefits due to the reduction of high direct exposure to mercury air emissions and water pollution affecting the miners and the local population.

An overview of the cost-benefit of strategies to reduce mercury emissions is presented in Table 3.

Table 3: Overview of the cost-benefit of strategies to reduce mercury emissions

Strategy	*Preliminary Qualitative Cost Assessment*	*Preliminary Qualitative Benefit Assessment*
Technological solutions		
- Mercury vapour capture in gold shops	SMALL	MEDIUM
- Retort use in mining	MEDIUM	SMALL
- Sluice solutions	MEDIUM	LARGE
Market Mechanisms		
- decrease in gold price	LARGE	SMALL
- increase in mercury price	LARGE	SMALL
- micro credit to clean technologies	SMALL*	LARGE*
Other mechanisms		
- Education	LARGE	SMALL
- Conversion to other livelihoods	LARGE	SMALL

* No evaluated experiences on ASGM, but pilot studies are performed

2.2 Small Scale Gold Mining as a source of Hg emissions

The demand for mercury in Small Scale and Artisanal Gold Mining (ASGM) was for the year 2005 estimated to 1000 ton. The major part (650 to 1000 tons) of these 1000 tons is not recycled but rather released via gold mining processes to air and water, thereby causing adverse environmental effects and effects on human health. The environmental and human health effects are more of a local nature in relation to other types of mercury emissions, given the large impact on the gold miners, the local population and their local environment. Mercury pollution from ASGM cause global pollution effects due to the emissions to air from combustion of mercury in the gold mining process, but what is characteristic for ASGM (in contrast to other mercury emission sources) is the high local human exposure of highly concentrated mercury vapour in air and mercury residual in water. This extreme exposure is related to a number of medical conditions not common for other types of "more controlled" mercury emissions. Emissions from ASGM are responsible for approximately one third of all anthropogenic mercury emissions globally. ASGM involves some 10 to 15 million miners and produces roughly 20 to 30 % (500 - 800 tonnes per year) of the global gold production (Telmer 2007).

2.3 Hg abatement efficiency and costs

There are a number of technologies available to reduce the use or release of Hg associated with ASGM. The use of mercury vapour capture technologies in gold shops is estimated as rather efficient since it involves relatively large scale operations and allows for increased income for the users of the technology. The estimated costs are relatively low. A quick estimate is a cost less than US$ 19 / kg reduced mercury emission (not considering education or disposal costs). Telmer (2008) indicate that the installation of vapour capture equipment in a gold shop would cost US$ 35 and capture 90 % of the mercury vapour.

The use of Hg retorts by miners is based on information and education and there are a vast number of miners in need of education, so the use of retorts is estimated as costly, although the unit cost of each retort is low. The efficiency of the measure depends on the application of the retorts.

The use of some sluices can be advocated since experiments indicate a relatively high efficiency in recovering gold (Hylander et al. 2007) under certain circumstances. It is also a mercury free alternative, which further increases the efficiency. The costs associated with using the studied slurry techniques are stated as being more cost efficient than using mercury amalgamation techniques (Hylander 2007). However, investments will be required and the time preferences of local miners will have to be prolonged.

There are other more technology independent options that could decrease the Hg emissions from ASGM. A decrease in gold prices could result in a reduction in gold mining and Hg use. Given a recent price relation between gold and mercury of 1:1000 (ranging between 1:1650 to 1:125, Telmer 2008), it is quite plausible that it would require an extremely large reduction in gold price before gold production via ASGM technologies would become less profitable than alternative income sources for the community. Furthermore, if there were to be a large decrease in gold price, many poor populations would become even poorer.

If the prices for mercury were to increase, the demand should correspondingly decrease. Veiga and Baker (2004) indicate that mercury constitutes 1 - 30 % of the gold production costs for ASGM using mercury amalgamation techniques. All in all, although mercury is very cheap compared to the price of gold, the literature and experience supports that high mercury prices results in reduced mercury losses from ASGM. UNEP (2004) and Hylander (2007), for example, indicate a large demand effect of high mercury prices.

Micro credits have proven themselves as very effective as a tool to reduce poverty in other circumstances in the world (Yunus 2006, Grameen Bank). What this instrument would provide with respect to ASGM is the opportunity to the gold miners to increase their long term thinking when engaging in gold mining. Furthermore, given that this is a loan, implementation costs can be reduced as loans are repaid.

Education has been indicated as inefficient if not followed by increasing mercury prices (EC DG-ENV 2006). However, education of some sort will be needed for the application of any technology or other measure since the current market conditions for miners still encourage the use of mercury amalgamation. Using previous experiences, an estimate is that US$ ~1000 million would be needed to educate 10 million miners in the use of retorts. Mercury would still be used by these miners, although the emissions would be lower.

Conversion to other livelihoods for miners is most likely a very ineffective abatement option. To convert the livelihood of miners will have almost zero benefit since potential miners can fill the gap as long as there are high profits to be made and few alternative occupations in the regions where ASGM take place. A comparison using African conditions indicates that 42 % of the people in sub-Saharan Africa have earnings below US$ 1 / day, while miners earn US$ 3 - 15 per day. Similar estimates can be seen in many parts of the world (Handelsman and Veiga 2006).

2.4 Benefits of Hg emission abatement

Mercury vapour capture technology in gold shops is a relatively potent abatement solution both since the vapour can be condensed and re-sold as mercury and also since the solution is oriented towards the relatively large scale gold shops (large scale as in contrast to single person miners). Some 90 % of the emissions from gold shops can be reduced by this technology.

The reduced emissions from using retorts are uncertain and depend on local conditions.

In a 2-year project community mining groups have been trained and are purchasing and using retorts, each of which costs about $5 when purchased in bulk. 500 miners have been trained and to date, upwards of 80 percent, as self-reported by the miners, are using the retorts. A retort has a maximum potential of capturing 90 % of the mercury vapour A rough estimate, assuming that 10 million miners are using 1000 tonnes of Hg (10 kg / person year) indicates that mercury use could be reduced by around 7 kg mercury per person per year. The costs following this effort are constituted of US$ 5 per retort and US$ 100 per person for education (based on EU experience cited above), which gives a final rough estimate of US$ 15 / kg mercury captured in the retorts during the first year of use. The duration of the training efforts, the retort efficiency and the lifetime of the retort are central parameters for the cost estimates. The costs of training ($100 per person) could be averaged over five years, provided miners were continuing to use the same types of retorts, leading to an approximate cost of $3/kg mercury per year for those five years.

The use of modern sluices is the one of the technologies that is mercury-free. The use of mercury-free technology is more efficient than reducing emissions from mercury, which is the reason to why the benefit estimate is estimated as higher than other ASGM technologies where mercury is used but the emissions are reduced.

Given a recent price relation between gold and mercury of 1:1000, it would require an extremely large price reduction in gold before gold production via ASGM technologies would become less profitable than alternative income sources for the community. What would happen however is that the disposable income would become even smaller for the miners and their community. If the prices of mercury where to increase, the income would become smaller for the miners, resulting in a poorer financial situation.

The implementation of suitably designed micro credits could encourage the use of mercury-free technologies which increases the potential benefit of this abatement option, since no use of mercury has a higher benefit than control of mercury emissions via technical solutions.

Education has been indicated as inefficient if not followed by increasing mercury prices (EC DG-ENV 2006). The benefit estimate is very uncertain and variable, given that almost all of the above mentioned abatement options will require some sort of education to introduce the option. For education efforts aimed at increased awareness and not specified towards any specific abatement option, the benefit estimate is therefore considered as small.

Conversion to other livelihoods for miners can be efficient for the miners considered for this action and their relatives. If other miners will fill the gap of the miners that have been converted into other occupations, this option will have zero effect on the mercury emissions. Given the potential profit to be made from ASGM, this dynamic seems very likely.

3 Reduction of mercury emissions from international trade

3.1 Overall assessment of costs and benefits

The potential effect on costs and benefits are presented for three cases; trade ban from EU; Seller-specified end-use restrictions; and disposal costs mandated via trade restrictions.

Qualitative Cost Assessment: Variable, ranging from small to large.

An export ban from the European Union is estimated as causing small costs since the total economic value of the ~800 tonnes mercury currently traded from EU corresponds to a relatively small economic value. Likewise, the cost estimate for end-use restrictions is small. These costs estimates do not include transactional costs, but an uncertain estimate of corresponding transactional costs indicates that these would constitute ~10% of the disposal costs.

Qualitative Benefit Assessment: Variable, ranging from small to large

An export ban from the European Union is estimated as having medium but variable benefits since reduced export from the European Union will result in reduced end use of mercury. The reduced use will be counteracted to some extent by increased production elsewhere. End-use restrictions are estimated as having medium benefits since they are focused on mercury uses with the worst environmental performance (ASGM).

Table 4 below presents a summary of available cost estimate for different final disposal solutions.

Table 4: Cost estimates for various disposal solutions. Information compiled from EC DG-ENV (2006a) and EC (2006)

Literature Source	Cost estimate	Estimate source
EC DG-ENV (2006)		
- Deep bedrock respitory	~US$ 220 / Hg / year	(SEPA)
- Permanent storage	~US$ 150 / tonne Hg / year	(SRIC)
- Surface storage	~US$ 300 / tonne Hg / year	(US DNSC)
- Disposal in monofill	~US$ 7 000 - 19 000 / tonne Hg / year	(SAIC)
EC (2006a)		
- Costs of storage	~US$ 300 / tonne Hg / year	

It can be seen that the cost estimates linked to disposal in mono-fills represents an outlier in the cost estimates for final disposal, and these results should be considered with some caution.

3.2 International trade as a source of Hg emissions

EC DG-ENV (2006) gives estimates on the direct costs for final storage of mercury that will be a result of the export ban from the European Union. These estimates are given in the Table 5 below.

Table 5: Estimates of cost for final storage of Hg due to export ban from the EU

EU chlor-alkali industry		
Years	2005 – 2010	2011 - 2015
Amount of Hg available	494 t/y to be sold	582 t/y to be stored
Income/costs of storage per tonne	US$ 10 / kg sold	US$ 306 /tonne /year
Total yearly income / cost	US$ 4,9 million / year	~ - US$ 180 000 / year

source: EC DG-ENV 2006

For our CBA purposes, the costs for the mercury producers of restricting trade are in principal made up out of two parts; foregone profits and costs of disposal. From a socioeconomic perspective however, the costs are only based upon the costs of disposal. The foregone profits are excluded from the socioeconomic costs estimates since they constitute restructuring costs. The economy will restructure from mercury trade into production and trade of a suitable substitute. The foregone profits will negatively affect chlor-alkali plants for example, but the foregone profits of the chlor-alkali industry induce increased profits in some other industry and are thereby offset. In this context, it should be mentioned that mercury trade is currently not a major source of income for any firm based in the EU (EC DG-ENV 2006).

It can be seen from Table 5 that the export ban will induce disposal costs corresponding to US$ 0.30 / kg per year for Europe. If putting these costs in comparison to the benefit estimates from the ESPREME study (Friedrich 2008), which presents external cost estimates of mercury to some US$ 12 500 / kg mercury emitted the net-benefit indicated is very large, ~ US$ 12 500 / kg. The benefit/cost ratio would equal 40 000, a very high number. However, the disposal costs are only related to the mercury stored and not to actual emissions to the environment so the comparison is not directly relevant.

In 2005, the global supply of mercury was 3690 tonnes (incl. 400 tonne from mercury stocks); the global demand was 3439 tonnes. For EU-25 the supply was 625 tonnes (incl. 0 tonne stock) and demand was 440 tonnes. For 2004, the import / export flows of 'elemental' mercury to and from EU-25 were 723 tonnes (US$ 3 615 000 - US$ 10 845 000) imported and 824 tonnes (US$ 4 120 000 - US$ 12 360 000) exported. The international market price for mercury has ranged between 5 and 15 US$ / kg mercury. The export flows from EU-25 were much smaller in 2004 (824 tonnes) than earlier (1658 tonnes in 2002, 1110 tonnes in 2003) due to new requirements on end-user specification from one of the main exporters of mercury. This requirement was installed to avoid exported mercury being used in ASGM in developing countries. The global trade patterns for 2004 are presented in Figure 2 below (EC DG-ENV 2006)

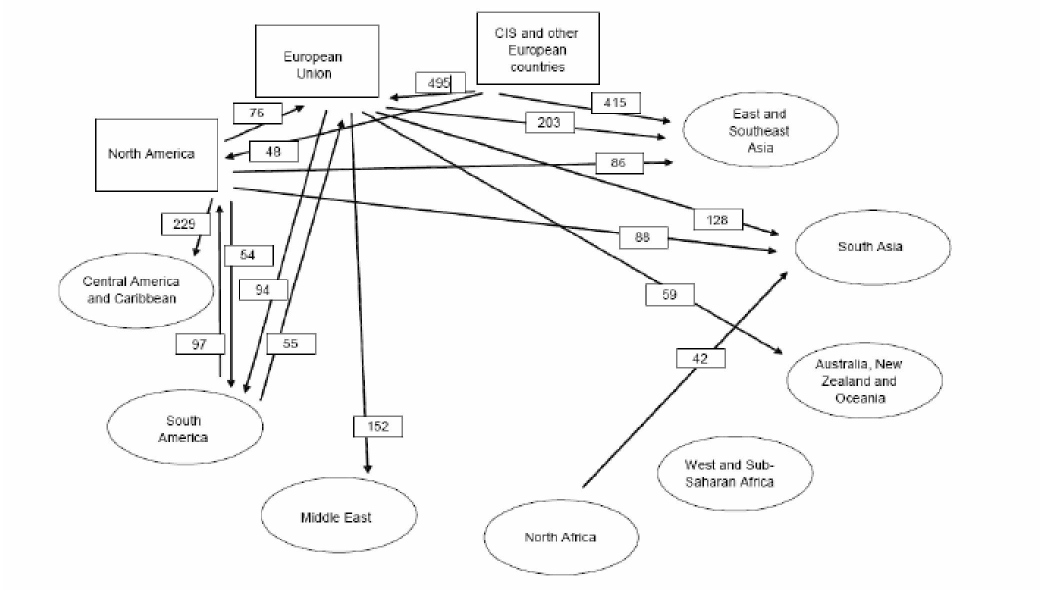


Figure 2: Global trade patterns for 2004 (EC DG-ENV 2006)

The countries, such as Spain, Netherlands, USA, United Kingdom, Germany, Belgium and Australia together exported more than 1325 tonnes of mercury to other than EU-25 in 2003 (EC DG-ENV 2006).

3.3 Hg abatement efficiency and costs

When the EU introduces a mercury export ban (scheduled to be adopted in September 2008 and enter into effect 15 March 2011) there will be an income loss corresponding to $6.25 million / year for specific European firms and disposal costs corresponding to $ 0.16 million /year. The income losses will be offset by income increases in other firms. Other costs would include restructuring costs for firms. Globally there is a potential mercury price increase, but this can be counteracted by the availability of substitutes for mercury and increased domestic production of mercury. However, the net trade and global supply of mercury should be reduced. As an additional remark, EC (2006) estimates that a trade ban will have a neutral economic impact on the current traders of mercury in the EU.

User restriction is a voluntary way used within the EU to avoid traded mercury ending up in e.g. small scale artisanal gold mining, which is usually the case with mercury exported from the EU (EC 2006). One of the major traders in the EU, MAYASA, decided in 2004 to restrict export to avoid exported mercury being used in ASGM in developing countries. The means used for enforcing this restriction is not specified (EC DG-ENV 2006). User restrictions involve a potential mercury price increase, but this can be counteracted by the availability of substitutes for mercury and increased domestic production of mercury. However, the net trade and global supply of mercury should be reduced due to the increased use of substitutes and increase in mercury prices.

Administrative costs related to a trade ban and disposal are very difficult to estimate, but estimates from EC (2006) indicate some 3 - 12 % of the disposal costs, or US$ 0.78 - 5 million for a ten year period.

This section has only addressed the impact of a ban on export by the EU, and only addressed direct costs involved in storage.

3.4 Benefits of Hg emission abatement

The benefit of an export ban is estimated as medium since the European Union currently supplies almost 1/4 of the global mercury traded globally, which indicates a large impact from an export ban. The benefit is also considered as medium since an export ban will reduce final use, which has a larger effect than reducing emissions and leakage from final use. The benefits are considered as variable since it is difficult to divine which trade flows that will be reduced and the environmental impacts vary over regions.

The total amount of mercury not traded according to the example on end-use restrictions in EC DG-ENV (2006) reduced the amount of export from EU by 30 %. The benefit can be large since they occur in hot spots where the negative impacts are very large from using mercury.

The impact of the expansion of such export limitations globally has not been further investigated.

4 Reduction of mercury from emissions from industrial processes, including use as a catalyst, by-production, contamination of component materials, and heat production

4.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Medium to Large
Cost Categories: Capital costs, operating costs

Qualitative Benefit Assessment: Medium to Large mercury emission reduction both globally and locally.

4.2 Industrial processes as a source of Hg emissions

Industrial processes contribute about 25 % to the total emissions of anthropogenic mercury to the atmosphere.

Emissions from non-ferrous and ferrous metal industry are estimated to contribute about 10 % to the total emissions. With regard to the Hg emissions from non-ferrous metal production, their amounts depend mainly on: 1) the content of Hg in non-ferrous metal ores used mostly in primary processes or scrap used in secondary non-ferrous production, 2) the type of industrial technology employed in the production of non-ferrous metals, and 3) the type and efficiency of emission control installations.

The content of Hg in ores varies substantially from one ore field to another (e.g. Pacyna, 1986, UN ECE, 2000) as does the Hg content in scrap. The Hg emissions from primary production using ores in non-ferrous smelters are between one and two orders of magnitude higher that the Hg emissions from secondary smelters with scrap as the main raw material, depending on the country. Pyro-metallurgical processes in primary production of non-ferrous metals, employing high temperature roasting and thermal smelting emit Hg and other raw material impurities mostly to the atmosphere. Non-ferrous metal production with electrolytic extraction is responsible more for risks of water contamination.

Major thermal non-ferrous metal smelters in developed countries employ ESPs and FGDs, working with efficiencies comparable with those for noted for energy production. This information has been obtained by the authors of the report (Pacyna et al. 2001) from:
- Cominco Ltd in Canada,
- Hudson Bay Mining and Smelting Co. Limited in Canada,
- Kennecott Utah Copper Corporation in the United States,
- Huttenwerke Kayser AG in Germany,
- Berzelius Metall GmbH in Germany,
- Norddeutsche Affinerie in Germany, and
- Metaleurop Weser Blei GmbH in Germany.

More details on individual non-ferrous metal works are available from metal Bulletin Books (see also www.icmm.com).

Among various steel making technologies the electric arc (EA) process produces the largest amounts of trace elements and their emission factors are about one order of magnitude higher than those for other techniques, e.g., basic oxygen (BO) and open hearth (OH) processes. The EA furnaces are used primarily to produce special alloy steels or to melt large amounts of scrap for the reuse. The scrap which often contains trace elements, and on some occasions mercury, is processed in electric furnaces at very high temperatures resulting in volatilization of trace elements. This process is similar from the point of view of emission generation to the combustion of coal in power plants. Much less scrap is used in other furnaces, where mostly pig iron (molten blast-furnace metal) is charged. It should be noted, however, that the major source of atmospheric mercury related to the iron and steel industry is the production of metallurgical coke.

The primary sources of mercury emissions from portland cement manufacturing contribute with about 9 % to the total anthropogenic emissions of this element. These emissions are generated in the kiln and preheating/pre-calcining operations. The kiln operations consist of pyro-processing (thermal treatment) of raw materials which are transformed into clinkers. Raw material processing differs somewhat for the wet and dry processes. Mercury is introduced into the kiln with fuels such as coal and oil which are used to provide energy for calcination and sintering. Other fuels, such as shredded municipal garbage, chipped rubber, petroleum coke, and waste solvents are also being used frequently.

Occasionally, building companies mix cement with fly ash from coal combustion in proportion about 3:1 in order to produce concrete. Fly ash may contain mercury through the condensation of gaseous mercury on fine fly ash particles in the flue gas before the collection of fly ash on de-dusting devices, such as ESPs or FFs (Pacyna, 1980). However, it is difficult to assess how much of mercury enters the environment through this pathway.

Heat is produced in large and medium size central heating plants, as a co-generation product in large electric power stations, industrial boilers and small residential and commercial furnaces. Large industrial plants generate their own electric power or process steam. The process of generation of emissions of Hg from these plants is similar to the one for emissions from coal and oil combustion in electric utility plants, discussed in Chapter 2 in this report. Similar are also installations to control these emissions. The main difference is brought by the type of boiler employed, which is often stoker-type boiler. The pulverized and cyclone boiler units are generally associated with larger industrial complexes and are similar in design to those used in electric utilities.

Commercial and residential furnaces are mainly used for space heating. Small stoker-type boilers and hand-fired units are still used in many regions of the world. Emission control equipment is not generally used in these small furnaces.

4.3 Hg abatement efficiency and costs

Large non-ferrous smelters use high efficiency air pollution control devices to control particle and sulfur dioxide emissions from roasters, smelting furnaces, and convertors (e.g. Pacyna et al., 1981; Pacyna et al., 2001). ESPs are the most commonly used devices for removal of particles. Control of sulfur dioxide emissions is achieved by absorption to sulfuric acid in the sulfuric acid plants, which are commonly a part of the smelting plants. Mercury is emitted mostly in a gaseous form and therefore, the ESPs are not very effective in the element removal. The element does not end up in sulfuric acid plants and is instead emitted to the atmosphere from the smelter stacks. The amount of these emissions depends on the content of mercury in the ore. This content varies substantially from one ore field to another. Only limited information has been gathered on mercury emission rates from non-ferrous smelters by the U.S.EPA (1993).

Mercury can be emitted to the atmosphere during the production of metallurgical coke, which is used in iron and steel industry. ESPs or FFs and less frequently wet scrubbers are used in the coke production plants to control emissions, particularly those generated during quenching. This process is performed to cool down the coke and to prevent complete combustion of the coke upon exposure to air. Although no data are available for the performance of the ESPs or FFs in coke production plants it is expected that mercury removal is limited (U.S.EPA, 1993).

The U.S. EPA has some experience with quantifying the costs and benefits of reducing mercury emissions from various industrial sources. One such industry is secondary steel production. This category is a significant source of mercury air emissions largely because mercury-containing switches are in the scrap metal (such as cars) used to make steel. In the United States, a program was established in 2006 called the National Vehicle Mercury Switch Recovery Program (NVMSRP). The NVMSRP, along with a few state mercury switch programs, will reduce mercury emissions by about 34 tonnes over the next 15 years, which represents the mercury content in approximately 61 million switches. The program is designed to remove mercury-containing switches from scrap vehicles before they are recycled in secondary steel mills, therefore preventing mercury emissions At this time, the precise cost-effectiveness of this program is unknown, although components of the costs include: outreach and education, design efforts which are typically do not require significant ongoing monetary investment. However, the voluntary effort to remove switches provides an incentive of about US$1.00 per switch. While this may not reflect the actual cost of removing the switch (some states have proposed incentives of up to US$7.00 per switch), it still costs significantly less than installing end-of-pipe controls to capture mercury at the furnace. In December of 2007, EPA also promulgated the Electric Arc Furnace Rule which codifies and builds upon the voluntary program (EPA, 2007).

With regard to chlor-alkali plants, the EPA promulgated an emissions standard based on maximum control technology (MACT) in December of 2003 to limit mercury emissions from this industry. The MACT rule requires controls and emissions limits for process vents and relatively stringent work practice standards or a cell room monitoring program to minimize fugitive emissions from the cell rooms. The total estimated capital cost of the final rule for the nine mercury cell chlor-alkali plants was around US$1.6 million, and the total estimated annual cost is about US$1.4 million per year. Plant-specific annual costs in our estimate range from about US$130,000 for the least-impacted plant to about US$260,000 for the worst-impacted plant. The final rule will reduce mercury air emissions from existing emission points within mercury cell chlor-alkali plants by 675 kg per year, a 74 percent reduction from current levels. The final rule also requires rigorous work practice standards such as periodically washing down work floors and covering waste containers. These requirements will reduce mercury emissions from so called “fugitive sources” throughout the plants. Although EPA is not able to accurately quantify the reductions associated with these work practice standards, these requirements will reduce mercury air emissions industry wide. By any accounting, the costs of implementing the MACT rule are significantly less than facility conversion to non-mercury cell technologies.

Major review of information on the costs of abatement for combustion of coal and other economic sectors was carried out within the EU ESPREME (http://espreme.ier.uni-stuttgart.de) and DROPS (http://drops.nilu.no) projects. The annualized investment and operational costs for installations that are used to remove mercury, including ESPs, FFs, FGD, and “add on” measures just for mercury removal are presented in Table 5. These costs are given in relation to the production of 1 tonne of specific production, indicated as specific activity indicator. The information on efficiency of Hg removal using these installations is also included in Table 6.

Table 6: Abatement cost for installations used to reduce Hg emissions from various industrial processes (in US$ /tonne of specific production- SAI) – selected technologies from the EU ESPREME project database (http://espreme.ier.uni-stuttgart.de)

Sector	**Specific activity indicator (SAI)**	**Emission control technology**	**Hg reduction (%)**	**Annual costs (US$ 2008/SAI)**		
				Annual investment costs	**Annual operating costs**	**Annual total costs**
Sintering	tonne sinter	dry electrostatic precipitator (ESP) – medium efficiency of emission control	5	0,10	0,05	0,15
		dry ESP – optimized	70	0,21	0,20	0,41
		virgin activated carbon injection (SIC)+FF – optimized	80	2,10	1,12	3,22
		calcium hydroxide-impregnated adsorbents (SICa) – emerging method	100	1,05	1,24	2,29
Primary lead	tonne primary lead	dry ESP – medium efficiency of emission control	5	0,06	0,04	0,10
		fabric filters (FF) – state-of-the -art	10	0,12	1,12	1,24
		virgin activated carbon injection (SIC)+FF+FGD – optimized	90	2,48	1,32	3,80
Primary zinc	tonne primary zinc	dry ESP – medium efficiency of emission control	5	0,10	0,06	0,16
		fabric filters – state-of-the-art	10	4,50	1,12	5,62
Primary copper	tonne primary copper	fabric filters – medium efficiency of emission control	5	1,80	13,80	15,60
		fabric filters – state-of-the-art	10	3,87	25,65	29,52
Secondary lead	tonne secondary lead	dry ESP – medium efficiency of emission control	5	0,10	0,06	0,16
		fabric filters – state-of-the-art	10	6,75	1,12	7,87
Secondary zinc	tonne secondary zinc	dry ESP – state-of-the-art	5	0,10	0,06	0,16
		fabric filters – state-of-the-art	10	0,12	1,42	1,54
Secondary copper	tonne secondary copper	dry ESP – state-of-the-art	5	10,89	15,86	26,75
		fabric filters – state-of-the-art	10	6,64	43,97	50,61
Cement production	tonne cement	fabric filters – medium efficiency of emission control	5	0,20	0,22	0,42
		fabric filters – optimized	98	0,39	0,38	0,77
		wet FGD – optimized	90	1,35	0,45	1,80
Coke production	tonne coke	use of raw materials with low HM content – optimized	5	0,00	0,02	0,02
		fabric filters – medium efficiency of emission control	5	0,21	1,65	1,86
		fabric filters – optimized	5	0,46	3,08	3,54
		dry ESP – medium efficiency of emission control	5	0,76	1,11	1,87
		wet FGD – medium efficiency of emission control	30	2,80	1,91	4,71
		wet FGD – optimized	40	3,04	2,79	5,83
		dry ESP – optimized	70	1,40	1,57	2,97
Iron and steel foundring	tonne cast iron	fabric filters – medium efficiency of emission control	5	10,80	82,77	93,57
		dry ESP – medium efficiency of emission control	5	38,10	55,47	93,57
		fabric filters - retrofitted from medium method to state-of-the-art	98	12,46	71,10	83,56
		dry ESP – optimized	70	69,80	78,97	148,77

Sector	Specific activity indicator (SAI)	Emission control technology	Hg reduction (%)	Annual costs (US$ 2008/SAI)		
				Annual investment costs	Annual operating costs	Annual total costs
Pig iron production	tonne cast iron	fabric filters – medium efficiency of emission control	5	0,20	0,75	0,95
		dry ESP – medium efficiency of emission control	5	1,53	2,22	3,75
		dry ESP - retrofitted	72	1,28	0,94	2,22
		dry ESP – optimized	70	2,79	3,16	5,95
Basic oxygen furnace steel	tonne steel	dry ESP – medium efficiency of emission control	5	1,20	3,00	4,20
		wet scrubber Venturi – optimized	8	5,68	0,54	6,22
		dry ESP – optimized	70	4,32	4,50	8,82
Electric arc furnace steel	tonne steel	fabric filters – medium efficiency of emission control	5	0,21	1,65	1,86
		dry ESP – medium efficiency of emission control	5	0,76	1,11	1,87
		fabric filters - retrofitted	98	0,26	1,42	1,68
		dry ESP– optimized	70	1,40	1,57	2,97

Major assessment of costs and environmental effectiveness of options for reducing mercury emissions to air from small scale combustion installations, SCI, (<50 MWth) has been prepared for the European Commission by Pye et al. (2006). It was concluded that:

- One of the most cost-effective options were preventive options (e.g. options prior to combustion to minimize emissions), such as coal washing and fuel switching. Such options require the use of a better quality, cleaner fuel within the same fuel type, or the switching to an alternative fuel with lower emissions. Another preventive option is reduction in energy consumption through energy efficiency;
- Only limited technical abatement options (such as removal of mercury from flue gases after combustion) were identified for SCI.

An assessment of abatement costs for reduction of heavy metals, including mercury within various industries was carried out for the heavy metal emission reduction Protocol of the UN ECE Convention on Long-range Transboundary Air Pollution (Visschedijk et al., 2006). The results of this assessment are similar to the data presented in Table 6.

4.4 Benefits of emission reductions

Information on monetary valuation of environmental and human health benefits related to the reduction of Hg emissions from individual industrial sources is largely missing in the literature.

Societal benefits related to the decrease of the 2005 Hg emissions from industrial sources worldwide until the year 2020 are estimated by Pacyna et al (2008) as a part of an assessment of socio economic costs of continuing the status quo of mercury pollution from all major anthropogenic sources. The social benefits were estimated as a difference between the societal costs (damage costs) related to the emission reductions calculated for the scenario assuming the status quo of environmental pollution between the years 2005 and 2020 and the emission reductions projected in the scenario where application of modern emission control devices is employed.

For the metal industry and cement manufacturing preliminary results suggest an annual damage costs to society due to ingestion of Hg contaminated food in the year 2020 can be as high as US$ 5 600 million along the assumptions defined for the status quo scenario relating to pollution of environment by Hg since 2005 and up to US$ 1 900 million along the assumptions defined for the extended emission control scenario (UNEP, 2008). The damage costs to society due to inhalation of Hg polluted air were estimated insignificant compared to the damage costs due to ingestion. Thus, the societal benefits due to reduction of Hg emissions from metal industry and cement manufacturing in the year 2020 were estimated to be about US$ 3 700 million.

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5 Reduction of generation of wastes that contain mercury

5.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Variable depending on the management technique such as incineration and land filling

Qualitative Benefit Assessment: very high relative to the abatement costs if management is in place

5.2 Hg abatement efficiency and costs

The sources of waste containing Hg are several. These sources may differ from one region to another and the quantity of waste-Hg from different sources may be correlated with life style and the level of economic development in the different countries and regions as well. Since the sources are different and the emissions from these sources are local and region specific, the costs to reduce generation of wastes that contain mercury and the measures as well as their implementation to reduce the emissions from these sources differ depending on whether the source is in a developed country or in a less developed one.

Table 7 brings together some sources of mercury containing waste generated both in developed countries (DC) and in less developed countries (LDC) where most of the sources are related to certain level of economic development.

Table 7: Examples of different sources of waste

Wastes from natural gas purification and transportation - wastes containing mercury
Metal-containing wastes other than those mentioned in batteries and accumulators/wastes from the manufacture, formulation, supply and use (MFSU)/manufacture, formulation, supply and use/of salts and their solutions and metallic oxides/ - wastes containing mercury
End-of-life vehicles from different means of transport (including off-road machinery) and wastes from dismantling of end-of-life vehicles and vehicle maintenance - components containing mercury
Batteries and accumulators - mercury-containing batteries and lamps and electronic devices
Construction and demolition wastes - construction and demolition wastes containing mercury
Wastes from natal care, diagnosis, treatment or prevention of disease in humans – amalgam waste from dental care
Separately collected fractions - fluorescent tubes and other mercury-containing waste

Source: adapted from: http://ec.europa.eu/environment/chemicals/mercury/doc/czech_rep_1.doc

In order to reduce generation of waste that contains Hg in DC different policy instruments including regulations, market based instruments as well as information are used. The reason why these policy instruments are developed is based on the fact that Hg in waste is leading to externalities (both ecological and health) where the damage costs may be very high as in the case of ingestion of fish containing Hg and the implied reduction of IQ in the population of new natal. Hence, the policy instruments have led to the implementation of different measures and waste management of wastes containing Hg such as recycling, land filling and incineration (for more detail on related to the costs of these measures see section 7). However, whilst the use of regulations and market based instruments have led to different technical measures that are moderate in the case of landfills and high in the case of incineration (which has led in some cases to the export of hazardous waste to LDC), the most costs effective measures are non technical being the results of good information highlighting the consequences of Hg emissions on the environment and human health.

In LDC on the other hand, the environment issue became an interesting subject quite recently and many LDC lack well formulated guidelines and policy structures regarding waste in general and waste containing Hg in particular. However, whilst both technical and non technical measures are used in DC, the measures in LDC are most of the time artisanal and in many cases chaotic. Uncontrolled dumping of wastes on outskirts of towns and cities has created overflowing landfills, which are not only impossible to reclaim because of the haphazard manner of dumping, but also because they have serious environmental implications (http://cat.inist.fr/?aModele=afficheN&cpsidt=2384293).

Nevertheless, when it comes to abatement costs when policy instruments are in place these may be high if they are technical and if transaction costs including monitoring are included in the estimations. However, costs to reduce waste containing Hg may be low and cost effective if the policy instruments are based on guidelines and information.

At the global level, the international community is working to strengthen legislation on the use, movement and disposal of toxic and hazardous waste (http://www.marketresearch.com/product/display.asp?productid=1470786). Example of the international work is the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal (in force since 1992). The transaction costs of these works are certainly efficient leading to lower global environmental as well as health damage.

6 Promotion of separate collection and treatment of mercury-containing wastes

6.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: small to medium, at least in developed countries

Qualitative Benefit Assessment: relatively large

6.2 Hg abatement efficiency and costs

Outside the industrial sector which is responsible for high emissions/releases of Hg, several other products include Hg which should be managed appropriately e.g. recycled in order to prevent emissions of this pollutant. Some of the products containing and which can be recycled are shown in Table 8.

Table 8: Some products containing Hg

Fluorescent bulbs:	All fluorescent bulbs typically contain 10-40 milligrams of mercury (.01 - .04 grams of mercury). Nonetheless, these bulbs use up to 50-75% percent less electricity than incandescent bulbs, making them the environmentally preferred choice. Remember to keep fluorescent bulbs out of the trash, avoid breakage and contact a recycling service to remove them.
Thermometers	Mercury thermometers can be identified by the silver colored liquid in the bulb. -Thermometers typically contain 0.5 - 0.7 grams of mercury. Large thermometers can have as much as 3 grams of mercury. Alternatives: Replace with digital thermometers or alcohol (red bulb) thermometers.
Thermostats	Non-electronic thermostats contain an average of 5.25 grams of mercury. Alternatives: Replace with electronic thermostats.
Other sources of mercury	
Button cell batteries (some types) - like those used in watches	
Dental fillings	
-Mercury switches - silent light switches and tilt switches, found in automotive trunk and hood lights, clothes irons, and space heaters	
-Old pesticides, fungicides, paint	
-Electronic devices	
-Different equipment at different public units e.g. drinking water systems	

Source: http://www.wastecap.org/wastecap/commodities/mercury/mercury.htm

The use of electronic devices has proliferated in recent decades both in DC and LDC, and proportionately, the quantity of electronic devices, such as PCs, mobile telephones and entertainment electronics that are disposed of is growing rapidly throughout the world. In 1994, it was estimated that approximately 20 million PCs (about 7 million tonnes) became obsolete. By 2004, this figure increased to over 100 million PCs. Cumulatively, about 500 million PCs reached the end of their service lives between 1994 and 2003. 500 million PCs contain approximately 287 tonnes of mercury (Puckett and Smith, 2002). This fast growing waste stream is accelerating because the global market for PCs is far from saturation leading to proportional increase of electronic waste (Culver, 2005).

To limit emissions of Hg several techniques may be used (as discussed above). Some of these techniques are presented in Table 9. However, where mercury containing products are used, promotion of separate collection and treatment of Hg-containing waste is likely to be most effective in limiting releases of Hg. Whilst promotion may give results in the developed world, this strategy may be more challenging in the developing world where there is often no differentiation between municipal, hazardous and medical waste in terms of applied techniques or achievable emission limits.

Table 9: Some Hg management techniques

Sector	**Best available technology (BAT)**	**Emerging techniques**
Municipal, medical and hazardous waste incineration	-Separate collection and treatment of Hg containing wastes -Substitution of Hg products -Sorbent injection -FGD -Carbon filter beds -Wet scrubber with additives -Selenium filters -Activated carbon injection prior to the ESP or FF -Activated carbon or coke filters -Selective catalytic reduction (SCR) -Co-incineration of waste and recovered fuel in cement kilns -BAT for cement kilns -Co- incineration of waste and recovered fuel in combustion installations -Avoid Hg entering as an elevated component of the secondary fuel -Gasification of the secondary fuel -Injection of activated carbon -BAT for combustion installations	Heavy metal evaporation process Hydro-metallurgical treatment + vitrification Municipal waste incineration PECK combination process

Source: http://www.unece.org/env/lrtap/TaskForce/tfhm/third%20meetingdocs/Summary_BAT_060407.doc

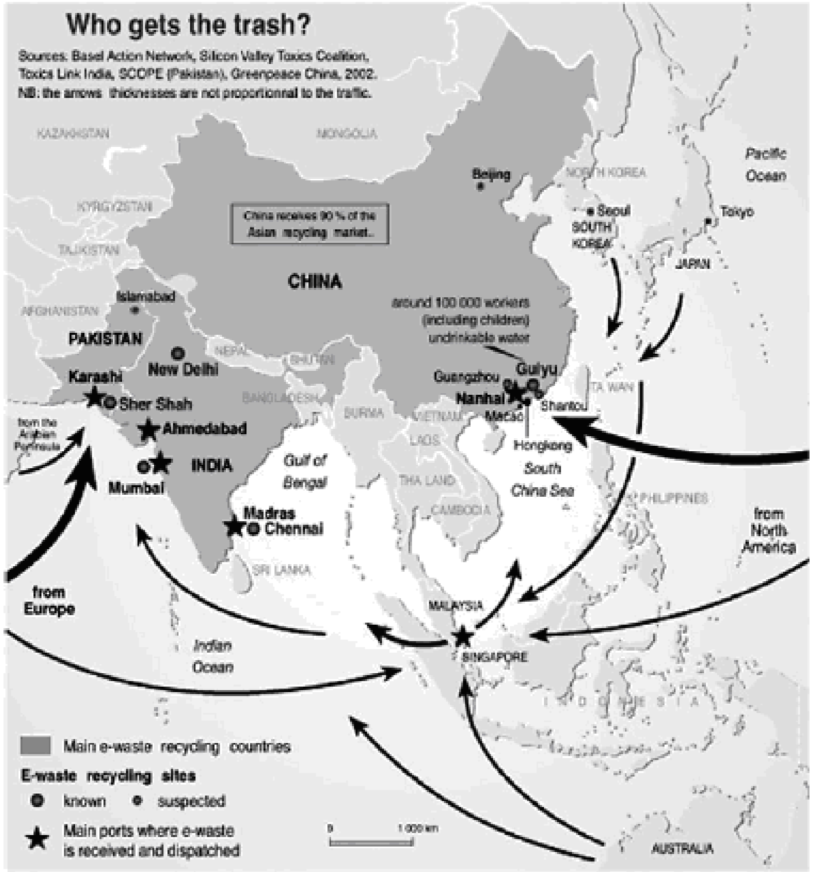
Nevertheless, promotion of Hg management is warranted at levels including households, industries and the public sector. However, except technical measures that may be used to reduce emissions of Hg, other measures such as substitution would be more cost effective. Table 10 shows some examples of substitution measures related to public water system unit in the US, where the costs, depending on the measure, are quite low.

Table 10: Examples of strategies using mercury-free alternatives in public water system units in the US, where abatement costs, depending on the measure, may be quite low.

Switch and relay alternatives	
Mechanical switch (metallic ball, snap switch, microswitch)	Uses a solid such as metallic ball that moves back and forth completing or breaking the circuit. Price for float switch replacement runs from US$ 25 to US$ 250. Price for free-floating float with inverter microswitch ranges from US$ 93 to US$ 175. Price for tilt switch ranges from US$ 1 to US$ 25.
Magnetic dry reed/magnetic switch	Metal reeds are drawn together completing the circuit in the presence of a permanent magnet. Prices for magnetic reed float switch range from US$4 to US$ 600 depending on use and features.
Continuous level transmitters	Use relay switches in a series. Price ranges from US$ 450 to over US$ 1200 depending on length. Allows for continuous data transmission capability.
Sensor alternatives	
Submersible pressure transmitter or transducer	The sensor probe is suspended by cable from the top of the tank and continuously measures pressure based on the water level above the sensor. Suppliers and manufacturers contacted report that these are low maintenance, come in no corrosion (titanium) models, and are easy to install. Prices range from US$ 350 to US$ 800.
Electronic pressure transmitter (nonsubmersible)	The transmitter is connected to plumbing at the bottom of the tank and measures pressure based on the water level above the sensor. Suppliers and manufacturers report that these are no or low maintenance, weather proof, and easy to install. These sensors measure the exact water level as opposed to the presence or absence of water at a certain level. Costs range from US$ 560 to US$ 900.
Ultrasonic, sonic, radar	Sound waves or radar travel down the measurement tube and reflect against the surface of the tank contents before returning back to the receiver. Electronics measure the time and calculate water level. Prices range from US$ 200 to over US$ 1000 depending on features and accessibility.

http://www.mass.gov/dep/water/drinking/mercbmp.pdf

The costs of Hg management may be small to medium in the case of developed countries, for examples costs of collection, transportation and recycling of switches in the US ranges between US$ 0.004 and $1.0. Some lesser developed and developing countries import considerable quantities of electronic waste. The Figure 3 below depicts the main electronic waste traffic routes in Asia.



Who gets the trash?

Sources: Basel Action Network, Silicon Valley Toxics Coalition, Toxics Link India, SCOPE (Pakistan), Greenpeace China, 2002.

NB: the arrows thicknesses are not proportionnal to the traffic.

China receives 90 % of the Asian recycling market.

around 100 000 workers (including children) undrinkable water

Figure 3: Main electronic waste traffic routes in Asia.

The benefits of separate collection and treatment of Hg containing waste are relatively large compared to the costs of abatement, including both technical and substitution measures.

7 Reduction of mercury emissions to air from medical, municipal, and hazardous waste incinerators and reduce migration and emission of mercury from landfills (all done)

7.1 Overall assessment of costs and benefits

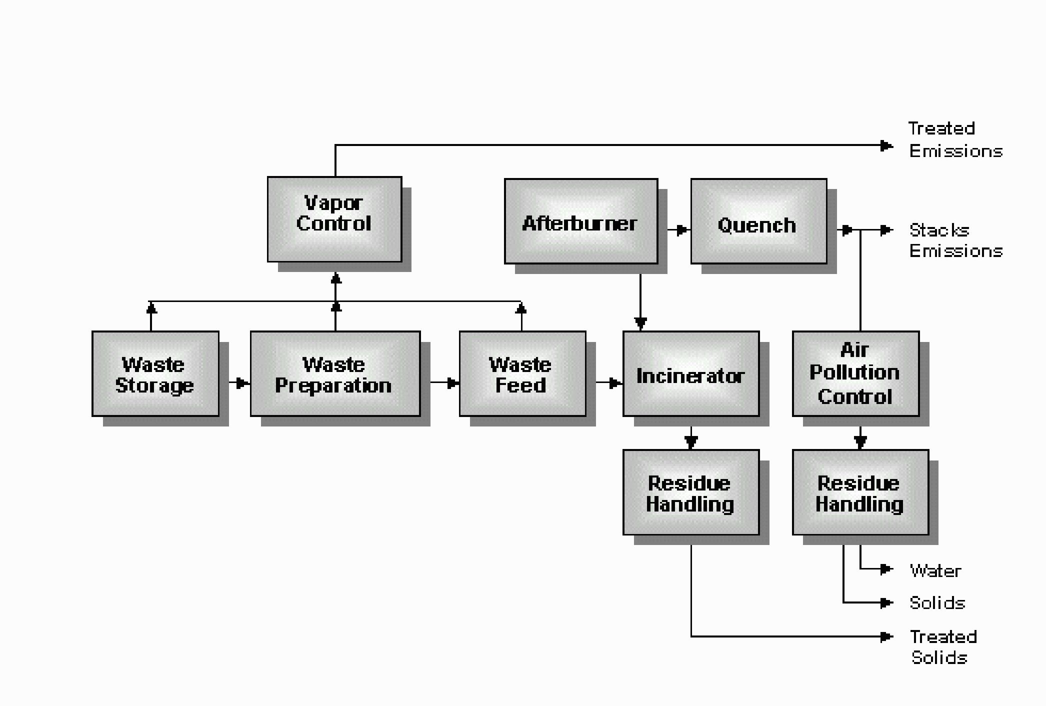
Qualitative Cost Assessment: Large to medium in the case of incineration and land filling, respectively

Qualitative Benefit Assessment: The benefits of incinerating or land filling (although they are high to medium, respectively), they are very large compared to the total costs of these management technologies.

7.2 Hg abatement efficiency and costs

Comparing of the external effects of incineration and land filling different studies revealed that the cancer risk from living near a landfill was about 100 times that of living near an incinerator. Furthermore, criticisms levelled against incineration arise from its history of releasing dioxins and furans. Consequently, for incinerators to provide an effective means of reducing the bulk of municipal waste in general, it is important that they do not emit harmful gases, compounds and particles. This is why the fulfilment of these requirements is correlated with high investment and maintenance costs.

For properly operated incinerators, the destruction and removal efficiency exceed the 99.99% requirement for hazardous waste and can be operated to meet the 99.99% requirement for PCBs and dioxins. Off gases and combustion residuals generally require treatment (http://www.frtr.gov/matrix2/section4/D01-4-23.html). Figure 4 below is a schematic cradle to grave illustration of how waste can be managed using an incinerator.



Source: http://www.frtr.gov/matrix2/section4/D01-4-23.html

Figure 4 : A schematic cradle to grave illustration of how waste can be managed using an incinerator.

When it comes to costs related to incinerator and waste treatment, Table 11 shows an example based on different scenarios and the type of waste in question. The Table represents estimated costs to apply incineration technology at sites of varying size and complexity where not only Hg is managed but other hazardous pollutants as well, e.g. dioxins.

Table 11: Cost of incinerator

Incinerator	Scenario A	Scenario B	Scenario C	Scenario D
	Small site		**Large site**	
	Easy	Difficult	Easy	Difficult
Cost per m^3	$1047	$1540	$914	$1399

Source: http://www.frtr.gov/matrix2/section4/4-23.html

As shown in the Table 11 the costs differ depending on whether the site is small or large. These costs range between $1047 in the case of a small site and $1399 for the difficult case when the site is large. However, for costs specific to mercury, Table 12 shows both investment and operating costs while investing in an incinerator.

Table 12: Investment and operating costs while investing in an incinerator.

Sector	**Emission control technology**	**Hg reduction (%)**	**Annual costs (US$ 2008/ tonne waste)**		
			Annual investment costs	**Annual operating costs**	**Annual total costs**
Waste incineration and cremation processes	wet scrubber (wSC) with alkaline addings – medium efficiency if emission control	20	0,12	0,08	0,20
	waste separation – medium	60	0,60	0,60	1,20
	dry ESP – optimized	70	1,84	6,99	8,83
	ESP+wet scrubber+activated carbon with lime+FF – optimized	99	2,31	2,48	4,79
	two-stage scrubber+wetESP – optimized	90	2,31	1,82	4,13
	virgin activated carbon injection (SIC)+FF – optimized	80	2,19	4,02	6,21
	virgin activated carbon injection (SIC)+venturi scrubber+ESP – optimized	95	5,25	6,15	11,40
	virgin activated carbon injection (SIC)+venturi scrubber with lime milk+caustic soda+FF– optimized	99	5,78	7,08	12,86

Investment and maintenance costs of appropriate landfills are relatively lower. Landfill controls can be implemented to limit mercury release and will also benefit management of many other hazardous wastes. As an example, the costs for the thermal treatment application at Lipari Landfill (Lipari site) in New Jersey included $430 000 in capital cost and $5, 019, 292 in operation and maintenance costs - The unit cost for this application was $67/tonne based on treating 80 000 tonnes of soil. The land fill Lipari was used for disposal of a variety of household, chemical, and other industrial wastes (http://costperformance.org/profile.cfm?ID=137&CaseID=137).

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.

When it comes to the benefits of incinerating or land filling, these are very large. Using the abatement cost e.g., $67/tonne) for land fill and $1047 /tonne for incineration (assuming 1 m^3 = 1 tonne), these costs are much lower than the damage avoided or the benefits reached if these management technologies are in use. Hence, the benefits of incinerating or land filling are very large compared to the total costs of these management technologies.

8 Reduction of mercury consumption in vinyl chloride monomer (VCM) and chlor- alkali production

8.1 Overall assessment of costs and benefits

Cost Assessment: Small if achieved through best practices, possibly high per plant or for countries that have many plants, but small globally. If achieved through conversion, high capital cost. Small long-term cost for chlor-alkali, higher for individual facilities.

Benefit Assessment: Medium to Large for VCM, Large for chlor-alkali.

8.2 Hg in VCM production

The use of mercury as a catalyst in the production of VCM is a major use of and source of mercury emissions in a few countries, primarily in China, largely because it is a coal-based process compatible with the predominant fuel supply in those countries (Maxson, 2006). While alternative processes are available, significantly more information on this sector is necessary to better understand the current operating practices of existing facilities and the mercury catalyst management processes before cost estimates can be made. But given the number of facilities using this process, requiring or encouraging their conversion or upgrading them and enforcing consumption or release limits could be costly in the specific countries in question. Industry-to-industry technical exchange on best management practices for the mercury catalyst and the exploration of alternatives could be less expensive, but information on the cost and technical issues for conversion has not yet been fully analyzed. There is some evidence that points to potential economic incentives for upgrades that could offset some of the cost.

The Natural Resources Defence Council, in collaboration with China's Chemical Registration Center, estimates that in 2004 the VCM sector was the largest user of mercury in China, consuming 700 metric tonnes per year. Multimedia releases from this sector are not well characterized. Also, due to increased demand for PVC, one estimate suggests that mercury use in this sector is projected to reach 1,000 metric tonnes by the end of the decade. Addressing this sector is likely to reduce global mercury risks significantly if the projected demand estimates for PVC are correct.

8.3 Hg in chlor-alkali production

There are three different processes for chlor-alkali production. Two processes: the mercury method and the diaphragm techniques date from the end of the 19th century, while the third process: membrane technique was developed on an industrial scale in the 1970s. Membrane cells release less hazardous substances and are more energy-efficient than the older techniques (e.g. KEMI, 2004).

For chlor-alkali, the global trend for conversion to non-mercury cell technology or reductions in mercury use and emissions has been established. This industry is declining substantially in the world with chlorine and caustic soda now being produced using more efficient, environmentally friendly, non-mercury processes. As of 2004, there were approximately 150 chlor-alkali plants worldwide that still use mercury cell technology (UNEP, 2007).

There were about 14 U.S. facilities in the mid-1990s using the mercury-cell process (so-called mercury cell chlor-alkali plants: MCCAPs); this year only 5 such facilities will still be in operation. The existing U.S. facilities are subject to a technology based emissions standard (the MACT regulation), which requires controls and emissions limits for process vents and relatively stringent work practice standards or a cell room monitoring program to minimize fugitive emissions from the cell rooms. Mercury use by the U.S. chlor-alkali sector was reduced by 94 percent from 1995 to 2005, from about 160 tonnes in 1995 to 10 tonnes in 2005. Emissions were reduced about 50 percent from 1990 to 2002 (from about 10 tonnes to 5 tonnes), and are expected to decrease to 2.5

tonnes by 2008. These numbers suggest that the benefit of reducing mercury consumption in chlor-alkali production can be quite high, with little opportunity cost given industry trends.

According to Euro Chlor information, there remained at the beginning of 2005 over 50 MCCAPs in Europe that continue to use the mercury process to produce chlorine (Concorde, 2006). Mercury consumption and releases have been greatly reduced from the 500-1,000 tonnes per year estimated in the 1970's. However, the average age of the EU plants is nearly 35 years, and further efforts to reduce mercury releases below present levels may challenge the technical limits of what is possible without converting to a mercury-free process. Unacceptably high Hg emissions before and into the 1980's pressed the member countries of OSPAR (the Oslo and Paris Convention for the protection of the North Sea and North-East Atlantic) to recommend in 1990 that the mercury cell chlor-alkali process should be phased-out by 2010. The European IPPC Bureau, in its 2001 BAT (best available techniques) reference Document on chlor-alkali industry, confirmed that the mercury cell process does not reflect BAT, and the IPPC Directive calls for non-BAT processes to be phased out by mid-2007. The implementation of the 1990 OSPAR Decision and the IPPC Directive as well, are ultimately the responsibility of each of the countries concerned. However, the countries uneven response to the OSPAR and flexible interpretation of the 2007 IPPC deadline reflect the diverse and shifting political and economic priorities of different countries within the EU (Concorde, 2006).

The Swedish Chemicals Inspectorate (KEMI) concludes that the use of mercury in the Swedish chlor-alkali industry should be covered by a general national ban (KEMI, 2004). KEMI also considered that mercury for chlor-alkali production should be granted a time-limited exemption from the ban and be allowed to be marketed and used until 31 December, 2009. It is interesting to note that according to KEMI, a national ban on the use of mercury in the chlor-alkali industry after that date will produce no greater further impacts for the companies affected beyond those which follow from the IPPC Directive.

In other larger EU countries there is no general agreement that a phase-out of the mercury process is needed before 2020 (Concorde, 2006). The production costs of these old plants are low.

Chemical industry has self-imposed a target for 2007 of 1 g Hg/ tonne of chlorine capacity. A discussion is now being carried out that this limit can be lowered until 0.75 g Hg/ tonne of chlorine capacity by 2012. It should be added that the best performing EU MCCAPs report emissions in the range from 0.2 to 0.5 g Hg/ tonne of chlorine capacity, and this lower range of emission is reflected in the BAT reference Document on chlor-alkali production. The phase-out of mercury in the chlor-alkali industry is expected to be fairly straight-line phase-out of remaining mercury cell capacity by 2020 (EC, 2006). The industry has, through a voluntary agreement, committed to phase out the use of mercury until 2020.

Hg emissions from MCCAPs in regions other than North America and Europe seem to be higher. Srivastava (2008) report that the mercury consumption in Indian companies are at least 50 times higher than in the world best companies. This high consumption of mercury results in emission of about 47 g Hg/ tonne caustic soda produced, one of the highest emission factors ever noted for this industry. Srivastava (2008) calls for a serious effort by the Indian chlor-alkali industry in moving towards membrane cell technology.

8.4 Cost and benefits of Hg emission reductions

Recent studies on the costs and benefits of reducing Hg emissions from US coal combustion facilities were used to derive conservatively estimated annual EU health benefit of some $39-47 per 1 gram of MCCAP atmospheric mercury emissions eliminated (Concorde, 2006).

The Concorde (2006) analyses also the costs and benefits (especially energy savings, reduced costs of mercury monitoring and waste disposal, etc) to industry of converting a typical MCCAP to the membrane process. There are various cases of actual conversions that have generated an attractive two- to three-year return on investment. However, it was pointed out that an EU industry investment on average in conversion of the MCCAP process to membrane process may not show an attractive bottom-line return until close to 10 years. The Concorde (2006) study concludes that combining the considerable "bottom-line" benefits of MCCAP conversion with even a conservative estimate of the public health benefits, it can be expected that the overall benefits, even when accumulated over only 5 years, are nearly twice the costs associated with the technology transition. Therefore, the conversion of MCCAPs should be regarded with a high priority when discussing the whole range of public health and other benefits associated with industrial development of chemical industry.

More information on combined benefits and costs of converting European MCCAPs to membrane process is presented in Table 13 below.

The existing MCCAPs use various control techniques to reduce Hg emissions, including: 1) gas stream cooling, 2) mist eliminators, 3) scrubbers, and 4) adsorption on activated carbon or molecular sieves (e.g. US EPA, 1995). Gas stream cooling is often used as the primary mercury control technique or as a preliminary removal step to be followed by a more efficient control device. Mist eliminators can be used to remove mercury droplets, water droplets, or particulate matter from the cooled gas streams. Scrubbers are used to absorb the mercury chemically from both the hydrogen stream and the end box ventilation streams. Sulfur- and iodine-impregnated carbon adsorption systems are commonly used to reduce the mercury levels in the hydrogen gas stream if high removal efficiencies are desired. This method requires pre-treatment of the gas stream by primary or secondary cooling followed by mist eliminators to remove about 90 % of mercury content of the gas stream.

Table 13: Combined benefits and costs of converting European MCCAPs to membrane

Combined benefits and costs (billion euro of 2004)	***Estimated annual benefits & costs***	***During 5 yrs.***		***During 10 yrs.***	
		Discount rate 5%	Discount rate 10%	Discount rate 5%	Discount rate 10%
Present value – **total conversion costs, including:** Investment cost, cleanup, etc.	2.6 one-time	**2.6**	**2.6**	**2.6**	**2.6**
Present value total benefits, including:		**4.9**	**4.4**	**8.4**	**6.9**
Industry benefits					
Health benefits*		1.7	1.5	2.8	2.3
Environmental benefits	various 0 annual	3.2	2.9	5.6	4.6
	significant	not included	not included	not included	not included
Ratio of total benefits/costs		**1.9**	**1.7**	**3.2**	**2.7**
Assumptions for conversion of European MCCAPs to the membrane process: -annual chlorine production capacity ≈ 6 million tonnes -10-15% of capacity will close rather than convert -annual atmospheric mercury emissions ≈ 4-5 g Hg per tonne chlorine capacity ≈25-30 tonnes mercury total -annual health benefits >25 euro per gram of mercury emissions eliminated -annual environmental benefits may be similar to health benefits, but are not quantified here					
Note: * Health benefits are based only on estimates of neuro-developmental impacts – specifically loss of intelligence – of methyl-mercury exposure in the US due to fish consumption, although there is evidence of other health effects as well. The figure of 25 euro per gram of mercury emissions eliminated (multiplied by 25-30 tonnes of mercury emissions eliminated upon full conversion) is a conservative estimate based on two key sources: one assuming human methyl-mercury exposure from consumption of both marine and freshwater fish, and the other assuming exposure from consumption of freshwater fish only.					

Major review of information on the costs of abatement for existing MCCAPs was carried out within the EU ESPREME (http://espreme.ier.uni-stuttgart.de) and DROPS (http://drops.nilu.no) projects. The results for chlorine production are summarized in Table 14 below.

Table 14: Annual investment and operating costs for chlor-alkali industry.

Sector	**Emission control measures**	**Hg reduction (%)**	**Annual costs (US$ 2008/ tonne chlorine)**		
			Annual investment costs	**Annual operating costs**	**Annual total costs**
Chlorine production (mercury cell plants)	good practices during maintenance and repair – optimized	20	0,02	0,02	0,04
	improvements of the mercury cells – state-of-the-art	15	0,06	0,02	0,08
	wet scrubber (WSC) with chlorinated brine or hypochloride addings- state-of-the-art	60	1,65	1,35	3,00
	virgin activated carbon injection (SIC)+FF - optimized	98		4,28	
	technology switching to diaphragm or membrane cells - BAT	100	36,96	0,00	36,96

9 Reduction of mercury use in products, including packaging

9.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Variable, ranging from small to large

In the European literature on the reduction of mercury used studied for this report, the costs related to a reduction of mercury in household measuring products are estimated as small since there are many available substitutes at similar prices. For other products, the availability of substitutes is smaller. For many other regions, the economic and technical situation is less favourable for reduction of mercury use in products. The qualitative cost assessment is therefore ranging between small to large.

Qualitative Benefit Assessment: small

For the European region, for which there available estimates, the total amount of mercury in household measuring products that are feasible to substitute are relatively small. For other products and regions, the potential is lower. Therefore, the qualitative benefit assessment remains small.

9.2 Mercury in products (incl. packaging) as a source of Hg emissions

Mercury in products is relatively scarce in Europe (except for products for dental practices) Therefore, measures aimed at reducing the amount of mercury will have relatively small effects on the use of mercury in society. The environmental impact of reducing the amount of mercury in products and mercury must be treated as relatively large since a removal of mercury in products is an 'up-stream' measure that indirectly affects 'down-stream' emissions such as emissions from incineration of waste, emissions from landfills and leakage to water and soil. Globally, the potential for reducing mercury in products will differ from the European situation. The differences relates to the level of economic and technical development, which in turn affects the local or regional availability of substitutes to mercury (Table 15).

An inventory of the amount of mercury demanded by relevant sectors shows that for EU-25, the amount of mercury demanded for use in products equals 155 tonnes (batteries, measuring & control, lighting, electrical & electronic, other). The other uses of mercury are covered in other chapters of this report.

Table 15: EU-25 and global mercury demand by sector (2005)

Mercury demand	**Global demand [tonnes]**	**EU 25 market demand [tonnes]**
Small-scale	1000	5
Chlor-alkali	619	190
Batteries	**400**	**20**
Dental	270	90
Measuring & control	**150**	**35**
Lighting	**120**	**35**
Electrical & electronic	**140**	**35**
VCM	700	Unknown
Other, laboratory, pharmaceutical etc	**40**	**30**
Total	3439	440

Source: EC 2006a

Also of importance for the emissions of mercury from products is the amount of mercury being recycled from these product categories. An estimate on the recovery of mercury specified in product categories is shown in Table 16 below. However, these estimates are uncertain on an EU level, and even more uncertain on a global scale.

Table 16: EU-25 and global product/process mercury recycling – 2005

EU25 and global product and process mercury recycling - 2005	**Hg in EU-25 waste stream (t)**	**EU-25 Hg recycled or recovered (%)**	**EU-25 Hg recycled or recovered (t)**	**Hg in global waste stream (t)**	**Global Hg recycled or recovered (%)**	**Global Hg recycled or recovered(t)**
SS gold mining	Not applicable	not applicable	Not applicable	Not applicable	not applicable	not applicable
Chlor-alkali	Not applicable	not applicable	32	Not applicable	not applicable	84
Batteries	**40**	**25%**	**10**	**500**	**15%**	**75**
Dental	72	25%	18	200	15%	30
Measuring & control	**42**	**25%**	**11**	**160**	**15%**	**24**
Lighting	**46**	**25%**	**11**	**150**	**15%**	**23**
Electrical & electronic	**42**	**25%**	**11**	**150**	**15%**	**23**
VCM	Unknown	unknown	unknown	700	43%	301
Other, laboratory, pharmaceutical, etc.	**36**	**25%**	**9**	**50**	**15%**	**8**
Total for these categories	278		101	1910		566

Source: EC DG-ENV (2006)

What can be seem is that the recycling rate is somewhere around 25 % in the EU and lower globally. For the products covered in this chapter it is estimated that some 254 tonnes of mercury in the EU-25, and 857 tonnes globally reach the waste stream without being recycled. This cause high potential for emissions from products once they reach the waste stream, which is covered in other chapters of this report.

9.3 Hg abatement efficiency and costs

The European ban of household measuring devices containing mercury, due to be applied by member states from 3 April 2009 will mainly cause costs related to restructuring of firms. In Europe, there are substitutes available at similar prices for all household applications, so the ban would be easy to implement. The availability of substitutes is only one of the factors determining the costs of reducing mercury in products. However, other costs relating to administrative efforts such as legislation and efforts related to phasing out of mercury are very difficult to estimate and also very dependent on the availability of substitutes for mercury in products. The ban relates only to households since it is estimated that mercury in measuring devices for professional use is not possible to substitute given the technologies currently available and the extensive control of these professional devices (EC 2006b). If a ban only covers a certain region, and only production (not use) there is a risk of re-allocation of markets and production, which would decrease the impact of the abatement measure.

The impact assessment for the proposed amendment to the European Council Directive 76/769/EEC (EC 2006b) covers the potential impact from a European product ban on household measuring devices such as thermometers. The general conclusion here is that the use of mercury in measuring and control equipment for households can be reduced from ~55 tonnes to ~28 tonnes per year in EU 15 corresponding to a 50 % reduction of mercury use in this product category. The costs of this ban would come mostly as restructuring costs since there are already substitutes available at similar prices. The costs would mostly affect manufacturers of Hg thermometers, but these costs would be offset by the increased benefits for the manufacturers of non-hg thermometers.

To reduce the use of mercury in batteries is already an ongoing process globally (EC DG-ENV 2006), and costs for continued reduction of mercury use in batteries should therefore be small.

The situation seems to be more difficult for electrical and electronic devices. Efforts are being made to promote mercury-free substitutes, but mercury use remains significant (EC DG-ENV 2006). This indicates that the reduction of mercury use in this product category would be associated with medium or high costs.

For lighting products, the availability of substitutes is smaller than for electrical and electronic devices (EC DG-ENV 2006). This indicates that the reduction of mercury use in lighting equipment is associated with high costs.

When studying the literature it is clear that the feasibility of any ban or restriction on products and packaging containing mercury will depend on what substitutes for mercury are available. Mercury has specific characteristics and it does not seem to be easily substituted in some products, for example some measuring equipments in hospitals.
The question of costs of reducing mercury in products and packaging is directly translatable to the question of whether there are compatible substitutes available at similar prices.

9.4 Benefits of Hg emission abatement

The benefit related to reduced mercury in household measuring products in the example above relates to relatively small amounts of mercury, ~28 tonnes. This reduces the total qualitative benefit assessment of such a measure. Also, the potential for re-allocation of markets and production further reduces the potential of such a measure. What increases the possible benefit however is that it would have measure both in the use part as well as in the waste part of the products' life cycle. In EC (2006b) it is estimated that the consumption of 33 tonnes of mercury for measuring devices would implicate some 8 tonnes of mercury emitted to air via landfills and incineration. A quick estimate would then indicate that reduced use of mercury in household measuring devices in EU-15 could reduce emissions by 6 tonnes annually. If extrapolating this estimate, the global reduction in emissions would be some 16 tonnes if the 150 tonnes of mercury used globally in measuring devices was reduced to 76 tonnes.

Given that the recycle rate for batteries, electrical & electronic devices as well as lighting equipment is estimated as equal to the recycling rate for measuring devices it is estimated that the emissions to air should be equal as well. However, due to the relative lack of substitutes, the potential emission reduction should be small for these product categories. Globally, it is indicated that the situation is less positive given that the recycling rate is lower and that the technical and economic availability of substitutes is low in many parts of the world. The benefit is therefore estimated as small.

10 Reduction of mercury use in dental practice

10.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: low to large for amalgam separator installation for the developed world. These costs would be much higher in the developing countries.

Qualitative Benefit Assessment: If the benefits are mainly related to the ingestion of fish these would in most cases not be greater than the costs.

10.2 Hg abatement costs and benefits

Dental amalgam is a mixture of mercury with an alloy consisting of silver, tin, copper, and zinc particles. This practice has been used in dentistry to restore carious lesions in teeth for about 150 years in the US and worldwide, although there are records of its use as a dental filling material in China as early as the 7th Century (Phillips, 1991). Although dental amalgam is a source of exposure to elemental mercury and may be the source of health hazard, for example migraines, erethism, and multiple sclerosis, it is still the most commonly used material, comprising approximately 60% of all restorations (http://english.pravda.ru/news/science/06-06-2008/105448-dental-0).

As of 2008, the use of dental amalgam has been restricted in Sweden, Norway and Finland mainly for environmental reasons.

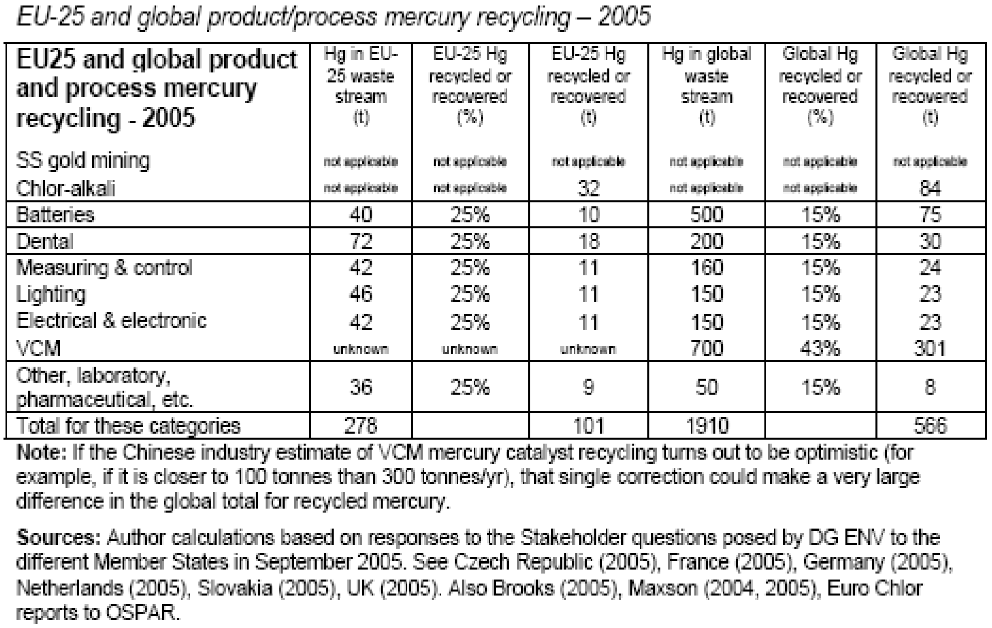
The question of direct impacts of the use of dental amalgam is controversial. Examples of studies showing negative impacts of dental amalgam have been presented e.g. Wojcik et al (2006). On the other hand, the American Dental Association Council on Scientific Affairs has concluded that both amalgam and composite materials are considered safe and effective for tooth restoration and the National Institutes of Health (NIH) has stated that amalgam fillings pose no personal health risk, and that replacement by non-amalgam fillings is not indicated. The US Food and Drug Administration is considering new labelling requirements for dental amalgams, and is also reviewing evidence about safe use, particularly in sensitive subpopulations. (http://www.fda.gov/cdrh/consumer/amalgams.html)

Hence, in the developed world there is no consensus to whether dental amalgam is a source of direct health hazard or not. Given the effects of mercury exposure through the environment, restrictions on the use and safe handling in dentistry to prevent releases of mercury to air and waste water have been imposed in several countries.

The amount of mercury used in dentistry in Europe in year 2000 was 70 metric tonnes, the US was 51 tonnes, while worldwide (include Europe and US) was 272 metric tonnes,. The worldwide demand for mercury in dentistry is predicted to be 250 metric tonnes by the year 2020 as more people worldwide get access to dentistry (Jacobsson-Hunt, 2007).

Table 17 shows that only 15% i.e. 30 tonnes of dental Hg are recycled or recovered while 200 tonnes Hg are accumulated in waste streams at the global level in year 2005 (http://ec.europa.eu/environment/chemicals/mercury/pdf/hg_flows_safe_storage.pdf).

Table 17: Recycling of product/process mercury in the EU and globally in 2005.



EU-25 and global product/process mercury recycling – 2005

EU25 and global product and process mercury recycling - 2005	Hg in EU-25 waste stream (t)	EU-25 Hg recycled or recovered (%)	EU-25 Hg recycled or recovered (t)	Hg in global waste stream (t)	Global Hg recycled or recovered (%)	Global Hg recycled or recovered (t)
SS gold mining	not applicable	not applicable	not applicable	not applicable	not applicable	not applicable
Chlor-alkali	not applicable	not applicable	32	not applicable	not applicable	84
Batteries	40	25%	10	500	15%	75
Dental	72	25%	18	200	15%	30
Measuring & control	42	25%	11	160	15%	24
Lighting	46	25%	11	150	15%	23
Electrical & electronic	42	25%	11	150	15%	23
VCM	unknown	unknown	unknown	700	43%	301
Other, laboratory, pharmaceutical, etc.	36	25%	9	50	15%	8
Total for these categories	278		101	1910		566

Note: If the Chinese industry estimate of VCM mercury catalyst recycling turns out to be optimistic (for example, if it is closer to 100 tonnes than 300 tonnes/yr), that single correction could make a very large difference in the global total for recycled mercury.

Sources: Author calculations based on responses to the Stakeholder questions posed by DG ENV to the different Member States in September 2005. See Czech Republic (2005), France (2005), Germany (2005), Netherlands (2005), Slovakia (2005), UK (2005). Also Brooks (2005), Maxson (2004, 2005), Euro Chlor reports to OSPAR.

In order to reduce the externalities of dental amalgam and depending on whether it is a source of health hazard or not there are two ways of concern:

1. Replace the amalgam filling.

As shown in Table 18 the abatement costs related to replace dental amalgam fillings with mercury free fillings at dentists and to dispose of the Hg safely are in the range of $129000 / kg Hg where the reduction potential is large.

Table 18: Costs for strategies avoiding Hg pollution and their potential to reduce Hg pollution

Activity	**Costs $/kg Hg**	**Reduction potential**	**Place and year**
Increase recycling of chair-side traps in dentistry	240	Medium	Minnesota 1999
Install amalgam separator	33000 - 1300000	Medium/ large	Minnesota 1999
Replace dental amalgam at dentists	129000	Large	Sweden 2004
Remove dental amalgam fillings at death	400	Large	Sweden 2004

Adapted from Hylander et al (2006).

2. Install dental amalgam separator or increase recycling of chair-side traps in dentistry.

As shown in Table 18 increasing recycling of chair-side traps in dentistry is a low cost strategies but the potential of reduction is medium. When it comes to installing amalgam separator where the potentials for reduction vary between medium and large depending on the kind of separator installed, the costs of the strategies range between $33, 000/kg Hg and $1300 000 /kg Hg.

Furthermore, results of cost effectiveness analysis conducted by US EPA gave rise to the following results (http://www.epa.gov/ARD-R5/mercury/meetings/Vandeven.pdf):

- Installation and purchase of separators at an estimated 110 000 to 133 000 clinics will require $111 million to $266 million, industry-wide;
- The operation and maintenance of these amalgam separators will require $78 million to $133 million per year;
- Conservatively assuming a separator has a useful life of 10 years, the combined annual cost is $ 89 million to $160 million per year;
- The annual cost of reducing one tonne of potentially bio-available Hg is $91 million to $282 million per tonne. That is $90625 /kg Hg and $281250/kg Hg , respectively.

Comparing the US EPA results to those presented in the Table 18, the first ones are in the lower range.

Since the benefits or the damage costs are mainly related to ingestion being $12,500/ kg Hg, most of the abatement costs presented here for capturing mercury used in dentistry are higher than the benefits. The use of non-mercury alternatives to dental amalgam for new fillings have a higher costs to consumers (2004 dental fees in the US indicated a $30 extra fee for composite fillings, based mainly on the increased time required). The incremental cost of composite fillings would be decreased if pollution effects were adequately factored in (http://www.mercurypolicy.org/new/documents/FINALReportfromMPPTestimony070708.pdf)

11 Reduction of supply from mining and extraction of virgin mercury and other ores (relates to trade and hierarchy)

11.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: small, medium

The reduction of primary Hg mining is estimated as a relatively inexpensive way of reducing Hg emissions, but the cost distribution should affect the less developed countries the most.

Qualitative Benefit Assessment: large

Since the reduction in Hg supply is characterised as an up-stream abatement option, the qualitative benefit assessment is estimated as large. Although there will be feedback mechanisms reducing the initial effect.

11.2 Hg mining as a source of Hg emissions

The current main Hg mines are in Khaydarkan in Kyrgyzstan (550 tonnes) and China (ca. 200 - 650 tonnes and growing). Previously, the Almadén mine in Spain supplied some 240 tonnes and Algeria supplied an equal amount. Since 2004 however, the Almadén mine is closed since 2004, and the state owned corporation is currently involved mainly in trading of Hg. The mine in Algeria is closed since 2003 when unfavourable conditions made Hg mining too expensive at the site (MBM 2005). While the Almadén mine was still in use, some 10 – 30 tonnes of Hg was directly emitted from the mining of Hg. Primary Hg mining is off course still a large source of potential Hg emissions and adverse environmental effects.

11.3 Hg abatement efficiency and costs

The costs for abatement of Hg emissions via reduction in Hg mining will vary according to local conditions. As an example, the Hg mine in Algeria was put out of use in 2003 due to unprofitability, not environmental reasons (MBM 2005). But mercury is an important metal for some purposes, and easy access to this metal might be considered important for economic growth in some regions, for example China. From the production side, foregone profits if terminating a Hg mine, might be offset by other investment opportunities, but the major part of the costs will be born by Hg buyers who are short of substitutes for mercury.

11.4 Benefits of Hg emission abatement by reduction in Hg mining

The reduction in Hg mining has a number of environmental benefits. The obvious reduction in Hg related effects will be combined by the environmental effects related to reduced mining activities and the following turnover of soil. However, one should be aware that the benefits related to mercury emissions will be offset to some extent by the feedback mechanisms following the price increase induced by the reduction in Hg mining. The extent of these feedback mechanisms is currently unknown. Examples of feedback mechanisms are; increased mining activities in other mines; re-opening of previously closed mines; increased recycling activities etc. It should be noted that increased efforts to recycle mercury will decrease the mercury being discarded as waste.

12 Reduction of mercury supply and management of mercury from decommissioned chlor-alkali cells and existing stockpiles

12.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Variable, depending on whether a storage infrastructure needs to be developed.

Qualitative Benefit Assessment: Large for chlor-alkali, large for products and processes.

Large stocks of Hg exist on the global level. These include government stocks, surplus mercury in chlor alkali industries and stocks at mercury mining facilities (Maxson 2006).

The generation of Hg from recycling and the recovery of Hg from decommissioned chlor-alkali plants have become increasingly significant contributors (10-20% in recent years) to global supplies because recycling has increased and the production of mined Hg has declined. However, in the interest of eliminating surplus Hg supplies from the global market, the European Union (EU) draft regulation for a Hg export ban, presently under discussion in accordance with Actions 5 and 9 of the EU Hg strategy. This ban targets all Hg from decommissioned EU chlor-alkali plants for retirement as of 2011 (EC, 2005). The supporting analysis suggests that recycled and by-product Hg (along with reduced Hg mine production, as necessary) will be more than adequate to meet global Hg demand (Maxson, 2006). It is estimated that the switch to mercury-free technology in the chlor-alkali industry will release around 12,000 tonnes of metallic mercury (EC, 2006a).

The US government, through the Defense National Stockpile Center (DNSC), owns one of the world's larger stocks of Hg, and in the early 1990s began selling it on the international market after declaring it unneeded for future defense needs. A moratorium on sales was declared in 1994 as a result of concerns that marketing Hg may contribute to global environmental contamination. The relative merit of selling versus retiring the Hg was studied (DNSC, 2004), and in February 2006 the US government announced that the stockpile of some 4400 t of Hg would be stored indefinitely in a warehouse.

Global demand of mercury has decreased from around 7 000 tonnes per year in the late eighties to 3 000 - 4000 tonnes in 2005 (Maxson, 2006). The supply to meet this demand is described in Table 19 and indicates that mining and by-product mercury is the main source of mercury to the global trade.

Table 19: Sources of Hg supply (2005)

Sources of Hg supply	Range of Hg supply (Mt)
Mining and by-product Recycled Hg from chlor-alkali wastes	1800-2200 90-140
Recycled Hg-other Hg from (decommissioned) chlor-alkali cells	450-520 600-800
Stocks	0-200
Total	3000 - 3800

Source: http://www.chem.unep.ch/mercury/PM-HgSupplyTradeDemand-Final-Nov2006-PMformat19Jan07.pdf

Another example of decreasing demand is the USA where production exceeded the demand of less than 500 tonnes per year in the late nineties as presented in Figure 5 .

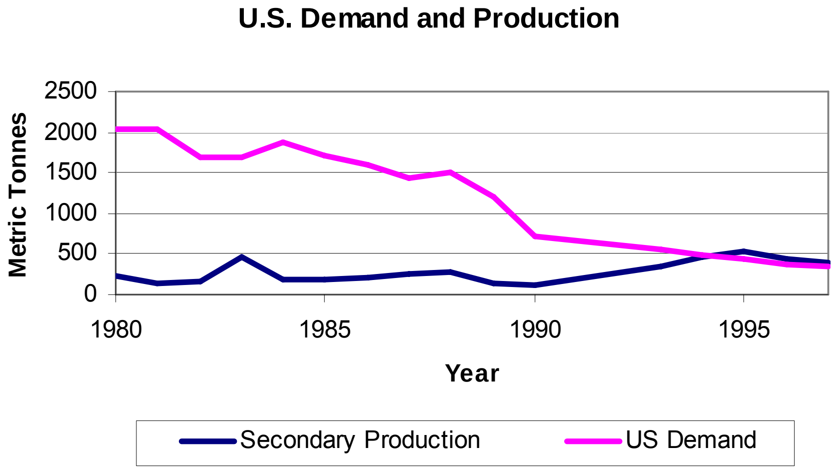


Figure 5: U.S. demand and Production

Source:(http://www.newmoa.org/prevention/mercury/breakingcycle/compendium/Weiler.ppt#256,1,Can the U.S. act alone on mercury?)

The reasons for the decreasing demand have been several including regulations and international agreements. The continued rate of decline in mercury demand will depend primarily upon reductions of use in the product manufacturing sectors (battery, electrical product, and measuring device) as well as in the industrial sectors chlor-alkali and vinyl chloride production. To reduce the consumption in the main consumption sector, small-scale gold mining, represents a major challenge.

A continued reduction of mercury demand driven by either environmental concern or economic realities will require the development of technological and economical instrument for the safe storage and management of surplus mercury.

For all excess mercury stocks, the cost of reducing the supply would be the opportunity cost to the forgone sale plus the cost of long term management of the excess stocks. The projected cost of storing federal mercury stockpiles (US$42 million over the next 40 years) in the United States could be used to project the costs associated with storing U.S. private stockpiles, but the numbers would vary based on the costs in an individual country. One issue to be mindful of in terms of supplies from decommissioned sources and stocks is the timing of making these supplies available on the market: there could be economic, social and environmental costs in introducing large supplies of mercury into the market at one time.

The main risks associated with stocks of mercury are associated with the fate of the mercury if sold and distributed on the global market. The risks as well as the potential management options will depend on the ultimate use of mercury and ultimately the amounts of mercury released to the environment in that application. However, to avoid any risks and to prevent potential future contamination, several countries in the developed world are promoting efficient management of existing stockpiles and Hg containing waste to prevent environmental contamination.

In Sweden for instance, the strategy is that Hg should not be recycled but should be finally disposed of in a safe and environmentally sound way. Using this as a starting-point, the Government has commissioned two enquiries into how such final disposal can be effected. In 2001, the Committee on the terminal storage of mercury proposed that a legal requirement for waste containing mercury to be stored permanently deep in bedrock. Waste owners should cooperate and bear the responsibility for the construction, location, building and management of a deep storage facility. An estimated total of 1 100 – 1 400 tonnes of waste containing mercury is waiting to be put into terminal deep storage (http://www.regeringen.se/). The Swedish EPA has estimated the costs of deep storage of a facility of 1000-2000 tonnes Hg should be in the range of 200-300 million SEK (around 20-30 millions US$) (2001 prices). Hence in average the cost for 1500 tonnes would be 250 millions SEK (around 25 millions US$; 2001 prices). The cost per tonne would be around 170 000 SEK/tonne (or 17000 US$) (http://www.regeringen.se/content/1/c4/26/09/a4b611c4.pdf).

In order to manage, or retire excess mercury several potential alternatives have been identified. Through a contract with the U.S. Environmental Protection Agency's Office of Research and Development, SAIC (Science Applications International Corporation) has been developed a methodology to be used to evaluate, prioritize, and select alternatives in a systematic manner (Vierow 2002). This methodology identifies criteria for such an evaluation, such as environmental performance, catastrophic risks, need for regulatory changes, implementation considerations, and cost. A system to ascribe weightings, or the importance of one criterion over another, is proposed (with the assistance of a commercial software package). Finally, a total of 11 alternatives for the storage and treatment/ disposal of elemental mercury are evaluated according to these criteria. Preliminary results are presented in Table 20 to show how the alternatives compare to one another when evaluated using the methodology (Randall 2002): Assuming that only benefits (non-costs) or only costs are important. The second column ("overall") shows that the landfill options are preferred independent of the treatment technology. The storage options rank next, followed by the treatment technologies combined with mono-fills, bunkers, or mined cavities. The reasons why the landfill options are preferred become apparent when costs are considered. The third column of results shows the rankings if only cost is taken into account. The landfill options are cheapest and this clearly outweighs the relatively unfavorable rankings that result from a focus on the benefits. However, if the costs are not an important factor, then the three storage options occupy the first three places in the "non-costs only" ranking. The last column of shows unfavorable rankings for the operating costs of the storage options. This arises for two reasons: a) if storage continues for a long period, even relatively small per annum costs will add up; and b) storage is not a means for permanent retirement of bulk elemental mercury and the analysts assumed that, sooner or later, a treatment and disposal technology will be adopted, which adds to the cost. This is enough to drive the storage options out of first place in the base-case rankings. However, the analysis would support continued storage for a short period (up to a few decades) followed by a permanent retirement option. This would allow time for the treatment technologies to mature.

Table 20: Summary of results for 11 evaluated alternatives

Alternative	Ranking (as fraction of 1 000)					
	Overall		Non-Costs Only		Costs Only	
	Score	Rank	Score	Rank	Score	Rank
Stabilization/amalgamation followed by disposal in a RCRA- permitted landfill	137	1	99	5	217	1
Selenide treatment followed by disposal in a RCRA- permitted landfill	123	2	66	9	217	1
Storage of elemental mercury in a standard RCRA-permitted storage building	110	3	152	2	126	5
Stabilization/amalgamation followed by disposal in a RCRA- permitted mono-fill	103	4	92	7	135	3
Storage of elemental mercury in a hardened RCRA-permitted storage structure	95	5	173	1	44	6
Selenide treatment followed by disposal in a RCRA- permitted mono-fill	94	6	74	8	135	3
Storage in a mine	81	7	140	3	44	6
Stabilization/amalgamation followed by disposal in an earth-mounded concrete bunker	70	8	108	4	42	8
Stabilization/amalgamation followed by disposal in a mined cavity	63	9	97	6	42	8
Selenide treatment followed by disposal in an earth-mounded concrete bunker	62	10	a*	a	A	a
Selenide treatment followed by disposal in a mined cavity	61	11	A	a	A	a
Number of alternatives evaluated	11	—	9	—	9	—
Total	1 000	—	1 000	—	1 000	—
Average score (total divided by number of alternatives, either 9 or 11)	91	—	111	—	111	—

Note: Shading indicates the highest ranking alternative. *) These options were evaluated for the overall goal but were not evaluated at the lower levels of cost and non-cost items separately, due to the low score from the overall evaluation. RCRA = Resource Conservation and Recovery Act

Hence, the methodology gives some ideas on the possible alternatives as well as the cost effectiveness of each of the alternatives. Furthermore, the methodology is designed to be flexible in order to allow for differences in criteria importance, the addition of other alternatives, and the substitution of better information (both qualitative and quantitative) as it is developed in the future.

13 Prevention of mercury contamination from spreading

13.1 Overall assessment of costs and benefits

- Small spills:

Qualitative Cost Assessment: High costs compared to substitution costs of the used product

-Large spills:

Qualitative Cost Assessment: Very high costs
Qualitative Benefit Assessment: The damage is quite difficult to estimate depending on whether the spill takes place in a developed or a developing country.

13.2 Hg abatement efficiency and costs

When the quantities of Hg spills are large measures are taken both in the developed world and the developing countries although the implementation of the measures is relative. In Europe and at the EU level the Commission Directive 93/112/EC of 10 December 1993 enable professional users to take the necessary measures relating to protection of health and safety at the workplace, and to protect the environment (http://www.reach.sgs.com/cts_directive_93_112_eec.pdf).

In the US, the Solid Waste Disposal Act of 1965, as amended, also known as the Resource Conservation and Recovery Act is the Federal Act that controls the management and disposal of solid and hazardous waste (http://www.epa.gov/osw/laws-reg.htm).

-Small spills

If the spilled quantities are small, in developed countries abatement measures are applied where the spill can be cleaned up by residents or workers following a set of relatively simple procedures. In the USA, for instance, many laboratories follow strict regulations on the use of mercury. There are well-documented procedures, especially in those accidents with broken glass and other cases when mercury is spilled. In all cases special removal kits are prescribed preventing any loss of mercury, which will evaporate afterwards. Moreover, special regulations and services exist to remove and store mercury at save places (http://www.knmi.nl/samenw/geoss/wmo/mercury/). In Canada, material from cleanup of mercury spills must be disposed of according to the provisions of the province's Environmental Management Act and the Hazardous Waste Regulation (http://www2.worksafebc.com/i/posters/2007/WS%2007_01.htm).

In developing countries in general, when small spilled quantities occur no clean up takes place and Hg is still used in many instances such as small scale mining and disinfection, such as following piercings. In Morocco for example, elemental Hg can be bought in a spice shop with left over Hg stored in the house with a high risk of spill.

In the US the cost of mercury spills is a topic of interest because some hospitals gain support for mercury reduction programs by using spill cost avoidance as a justification for change (http://www.epa.gov/osw/laws-reg.htm). In general, the true costs of mercury spills are not well documented and tend to be anecdotal. However, the cost of substitution to products with no Hg may be lower. For example the clean- up costs of 1 broken sphygmomanometer is equivalent to $5000. For that cost, one could buy 30 or 40 non-mercury ones (http://www.epa.gov/osw/laws-reg.htm). Table 21 shows some examples giving insight on the potential cost of cleaning up Hg spill.

Table 21: Small spill of Hg and its clean-up cost

Cost Estimate for Clean-up	**Reference & Description**
Small spill – over $1000 Large spill – around tens of thousands of dollars	http://www.middlecities.org/PDF/mercury_bulletin.pdf "Mercury Contamination Risk Control", Middle Cities Risk Management Trust, Okemos, MI "A typical thermometer contains ½ to 3 grams (.018 to .11 ounces) of mercury. A typical household mercury fever thermometer contains approximately 1 gram of mercury. A typical barometer contains 1 pound (454 grams) of mercury and poses a significant spill risk. The cost of cleaning up a spill will vary by the size of the spill and the degree of exposure to property and people. Small spill clean-ups usually cost over $1000 and large spills can go into the tens of thousands of US$. "
$10 000 for one broken barometer	http://www.pprc.org/pprc/pubs/topics/healthcare.html#mercury Northwest Guide to Pollution Prevention by the Healthcare Sector "A large barometer fell and broke in a 60 square foot office in a Medical Center located in the Puget Sound Region. The barometer was used to calibrate instruments used in treatment of patients. No one knew when the barometer fell and broke in the office. "The following are costs associated with the mitigation of the spilled mercury in this 60 square foot office area: Outside Vendor Cleanup Company – Time, Materials and Labor: $4000 Replacement of Mercury Spill Vacuum: $3200 Medical Follow up (Blood Testing) For Hospital Staff: $260 Mercury Disposal Costs: (Will Vary Per Vendor Used): $1600 Labor Hours Cost for Hospital Personnel Involved Est.: $1000 Total Costs for Spill Mitigation: $10060

Source: Adapted from http://www.sustainablehospitals.org/PDF/IP_spills_cost.pdf

-Large spills

Larger spills or spills in which the contamination has spread are often more expensive and difficult, as these spills involve more sophisticated methods of collection, decontamination, and disposal (http://www.epa.gov/osw/laws-reg.htm). For instance, in June 2000, a Newmont contractor carrying containers of mercury spilled 330 pounds of the chemical over 25 miles of roads and towns in Peru. The mercury was picked up by locals who thought it was valuable. Some of them boiled it on kitchen stoves looking for gold. The spill affected 1 100 people and required a massive, multimillion-dollar cleanup effort by Newmont that included digging up streets and the floors of homes. A later World Bank investigation found that Newmont had stopped using an Environmental Protection Agency-approved container for the mercury; that the mercury had been loaded incorrectly on an open truck; and that company officials initially misrepresented the size and seriousness of the spill, hampering emergency response efforts (http://www.theminingnews.org/news.cfm?newsID=191).

Hence, the developing countries may have regulations when it comes to large spill but they are often not respected and the case in Peru has been a wake-up call.

When it comes to large spilled quantities, the damage costs are difficult to estimate. In the case of Newmont, the mining firm offered up to around $5900 to more than 700 local residents, but over 1 100 others are still engaged in a legal battle (http://en.wikinews.org/wiki/Peruvians_sue_Newmont_Mining_Company_over_mercury_poisoning

14 Control and remediation of contaminated sites

14.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: many techniques can be used for cleaning up contaminated sites and the costs dependent on the used method.

Qualitative Benefit Assessment: Compared to cleaning up costs, the benefits may be very large

14.2 Hg abatement efficiency and costs

Given the unique behaviour of Hg, several techniques exist or are currently being developed for remediation of contaminated sites. Since, the Hg species present in a given environment depend on the initial released form, the thermodynamic stability of this compound and the transformation rate of the released form to a more stable one (Baeyens et al, 1979). These issues must be well understood to effectively design and evaluate appropriate remedial solutions in Hg impacted areas (Hinton et al 2001). However, any measure employed must consider the risk to ecological or human health and have the acceptance of regulators.

Below are some examples of techniques where their costs are qualitatively described (adapted from: Minamata).

-Excavation and ex-situ (i.e. off-site or aboveground) treatment of Hg-contaminated soils is the most frequently employed practice for Hg recovery. Although excavation can be complicated if it extends below the water table or costly if the contamination is distributed over a large area, it is essentially a well-understood practice.

-Thermal Treatment: As the volatility of Hg and its compounds increase with temperature, thermal heating of excavated soil is a potentially effective means for Hg recovery from contaminated soils.

-Hydrometallurgical Treatments: Chemical extraction of Hg from excavated soils can be induced through four primaries mechanisms: desorption of adsorbed species; oxidation of metallic Hg; use of strong complexing agents; and through dissolution of precipitated Hg. Efficiency of any mechanism employed may decrease over time due to re-complexation and re-adsorption and removal of the most soluble compounds at early time.

-In-Situ recovery: Methods for in-situ recovery of Hg are far less established than ex-situ techniques. As well, due to subsurface heterogeneity, more uncertainly generally exists concerning the effectiveness of in-situ processes, and clean-up times tend to be longer than ex-situ treatments. Despite these factors, many in-situ technologies are very promising and – mainly due to the fact that contaminated soil and groundwater remain in the subsurface – may become more cost-effective and practical than excavation and treatment methods for many Hg-contaminated sites.

-Soil Vapour Extraction coupled with Soil Heating: Soil Vapour Extraction uses a vacuum to force air through the unsaturated zone. Currently, soil heating can be costly over expansive areas and difficult to homogeneously heat a soil volume.

- In-situ Leaching and Extraction: Used in conjunction with Pump-and-Treat systems, In-situ leaching and extraction involves the injection of chemicals to enhance Hg solubility in groundwater, thereby reducing clean-up time and improving recovery rates from groundwater. Pump-and-treat is a frequently practiced, cost-effective remedial alternative employed either for removal contaminants from the subsurface and/or hydraulic containment of a contaminant plume.

-Electro-Kinetic Separation: This process involves the generation of an electric field through application of a low-voltage direct current in a soil matrix. Heavy metals, such as Hg, migrate towards electrodes placed in the soil where they accumulate and can subsequently be removed at a lower cost than excavating the entire impacted area.

-Interceptor Systems: Interceptor Systems, such as trenches and drains, are extremely simple and effective at recovering Hg as “free product” (essentially as metallic Hg); however, this treatment is limited by topography and stratigraphy and does not address Hg held in residual saturation.

- Phyto-remediation: Phyto-remediation is a promising albeit unproven technology, wherein plants assimilate and concentrate metals from soils. This technique holds much promise for the cost effect remediation of shallow soils over a fairly widespread area, but issues such as limited access to vegetation by wildlife and time required for clean-up must be addressed.

For the sake of illustrating clean up costs, an example from Sweden may give some insight on the magnitude of these costs. EKA was a chlor-alkali firm which closed in 1928. The total costs to decontaminate the area where the industry was located are estimated to $28 million in 2008. About 90% of the estimated existing 16 tonnes of hg will be removed leading to a cost of around $1 944 /kg Hg. An extra benefit of this measure is a removal of around 850 g dioxin (http://www.nwt.se/ArticlePages/200707/09/20070709211004_437/20070709211004_437.dbp.asp).

Comparing the clean-up cost estimated in the Swedish case with Hg damage related to ingestion of fish i.e. $12 500/kg Hg, the benefits are much higher than the costs.

15 Increase of knowledge and capacity on mercury among states

15.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Small to large
Cost Categories: Research, information sharing

Qualitative Benefit Assessment: Large.

15.2 Increased knowledge on environmental assessment and options to reduce Hg pollution on global scale

Mercury pollution has been widely recognized as a global problem. Therefore there is a need for global action to protect human health and the environment against mercury pollution. The UNEP Governing Council in 2001 commissioned the Global Mercury Assessment, which was completed in 2002. This assessment concluded that policy action on global scale could have significant effect on mercury levels in the environment. Furthermore, the UNEP Governing Council concluded in 2003 that there is enough evidence on significant global adverse impacts from mercury to warrant further international action to reduce the risks to humans and wildlife from the release of mercury. In 2005, ministers and other government representatives from several countries met at the UNEP meeting and addressed the question of establishing of a possible international convention aiming at the reduction of emissions of and exposure to mercury on a global scale. Interesting report on the global assessment process and its history is described by Eckley Selin (2005).

One of the important issues related to establishing of global mercury convention is the understanding of the global mercury problem and its potential solutions by policy makers in individual countries and their political will to agree on reduction of Hg emissions and exposures. Increased knowledge on various options for such reduction is of primary importance towards obtaining such agreement. Therefore, there is considerable benefit to further increase knowledge of mercury contamination, specifically in the areas of inventories, human and environmental exposure, environmental monitoring, and socio-economic impacts.

International programs and conventions play a very important role in building capacity among various countries with regard to their knowledge on sources, environmental transport, effects and emission reduction options for mercury. The UNEP has developed a Toolkit as a guidebook on how to estimate emissions from various emission sources in countries which do not have their own methodologies for emission estimates. Accurate and complete data on emissions is a prerequisite for any further assessment of fate and effects of contaminants, as well as for assessing their future changes. A number of countries used this Toolkit when calculating their national emissions for submission to UNEP (UNEP, 2008).

The most comprehensive international agreement regulating mercury to date is the 1998 Aarhus Protocol on Heavy Metals to the UN ECE Convention on Long-range Transboundary Air Pollution (LRTAP) (www.unece.org), This convention covers the European countries, the United States and Canada. The member countries report their emissions to the European Monitoring and Evaluation Programme (EMEP) (www.emep.int). The Aarhus Protocol establishes the emission reduction limits for Hg and other heavy metals and suggests best available techniques for limiting emissions from various sources. An important tool for improving the capacity on development of emission inventories and their future scenarios for the LRTAP countries, as well as other countries is the Joint EMEP/ CORINAIR Atmospheric Emission Inventory Guidebook (http://reports.eea.eu.int/EMEPCORINAIR3/en/) (UN ECE, 2000).

The Arctic Council with its 8 member countries and 6 permanent participants representing Arctic indigenous groups is another setting providing the opportunity for capacity building and our knowledge on sources and impacts of mercury as a global pollutant. The Arctic Monitoring and Assessment Programme (AMAP) has been involved in development of global emission inventories for mercury (e.g. UNEP, 2008), monitoring the Hg levels in various environmental ecosystems in the Arctic and assessing environmental and human health impacts of this contaminant (e.g. AMAP, 2002)

15.3 Increased knowledge on environmental assessment and options to reduce Hg pollution on regional and national scale

Policy makers in Europe have taken the advantage of improved information on emissions. Following the preparation of a Position Paper on Ambient Air Pollution by Mercury (http://europa.eu.int/comm/environment/air/background.htm#mercury see also EU 2001), the EU adopted the European Mercury Strategy (http://europa.eu.int/comm/environment/chemicals/mercury), the EU Community Strategy Concerning Mercury. The development of this strategy has been accompanied by a number of research projects supported by the European Commission to obtain more knowledge on mercury and to develop tools that can be used by the EU member states and other countries to assess emissions, fate, and impacts of mercury pollution and to propose policies on how to reduce these emissions and impacts. These projects include: MAMCS (Mediterranean Atmospheric Mercury Cycle System: www.eloisegroup.org), MOE (Mercury Over Europe: www.eloisegroup.org), MERCYMS (An Integrated Approach to Assess the Mercury Cycle into the Mediterranean Basin: www.iia-cnr.unical.it/MERCYMS/project.htm), and ESPREME (Estimation of Willingness-to-pay to Reduce Risks of Exposure to Heavy Metals and Cost-benefit Analysis for Reducing Heavy Metals Occurrence in Europe: http://espreme.ier.uni-stuttgart.de). A large data base has been developed and used in various countries on emission control technologies that can be used to reduce emissions from various sources with information on the efficiency of these technologies and their investment and operational costs.

The overall objective of the EU DROPS project (http://drops.nilu.no) was to provide a full-chain analysis related to impact of health protection measures related to priority pollutants, including mercury in order to support the development of cost effective policy measures against pollution related diseases and their wider impacts (Pacyna, 2008). The main achievement of the project is the development and application of methodology for the assessment of costs and benefits from the implementation of measures for the reduction of human exposure to selected contaminants. This methodology consists of models, analytical procedures, and databases. The models and databases developed within the EU projects can be used in countries worldwide after certain adjustments to specific conditions that may be in place when using the European data outside the region.

The body of mercury information developed by the United States is described or referenced in the U.S. EPA’s Roadmap for Mercury of July 2006 (www.epa.gov/mercury). International cooperation and capacity building was fond in the Roadmap as an important tool to help further mercury reduction efforts. For example, the United States participated in a capacity building program to help the government of Burkina Faso develop a more accurate and comprehensive mercury inventory. This work helped to inform local authorities on environmental concerns, setting the stage for the development of and regulations for mercury control.

The South African Mercury Assessment (SAMA) program has been organized in South Africa to improve the knowledge on sources, behaviour and impacts of mercury in the country (Leaner et al., 2008). Another initiative to improve our knowledge on these issues is the South African – Norwegian project on mercury in South Africa (MERSA). Both initiatives can provide a possibility for capacity building on mercury pollution in the whole continent of Africa with benefits to other African countries.

15.4 Increased knowledge as a factor to the development of policy options

Major benefits of an increased knowledge for the development of policy options to reduce pollution were reviewed by Swain et al. (2007). This review is available in the Appendix 1. The following policy options were reviewed: 1) policy options to reduce releases of Hg to the environment and 2) policies to limit exposures to Hg through risk communication. Releases of Hg to the environment can be reduced by policies related to the supply or demand of Hg, implementation of technological controls for reduction of industrial emissions or discharges from waste disposal, or the reduction of quantities of produced goods that result in such releases. In general, policy options used routinely to reduce pollutant emissions from industrial processes include technology requirements, emission performance standards, emission taxes, and cap-and-trade (CAP) approaches. Other policy options such as subsidies and restrictions on the sale and disposal of Hg (and Hg-containing items) could influence Hg releases from small-scale practices such as artisanal gold mining. Application of any of these alternative policies to Hg reduction will have benefits and costs. An economic approach to evaluating different policy options is to balance, at the margin, the benefits and costs of any policy option. Such policies are deemed economically efficient. The knowledge of policy options, their efficiency in Hg reduction and cost of implementation, as well as environmental, and human health benefits is needed for policy making at a national and regional level in order to introduce ecosystem based management of environmental resources in a given country.

In the case of Hg, economic analysis is complicated by the need to track benefits and costs at various geographic scales, from local to global. While the costs associated with the implementation of new processes or control technologies can be estimated in a relatively straightforward manner, the assessment of benefits is complicated by the scientific uncertainties reported in the environmental literature [i.e., the linkages between reducing environmental Hg releases and lower levels of Hg in the atmosphere and in fish (Lindberg et al., 2007; Munthe et al., 2007)] and health science literature [i.e., linkages between reduced levels in the environment, reduced exposures, and health improvement (Munthe et al., 2007). Ideally, economic analyses highlight these uncertainties as well as those introduced in the benefit–cost component of the analysis, and researchers will conduct additional analyses to assess the sensitivity of the results to the assumptions associated with the uncertainties (US EPA, 2000).

In addition to reducing Hg releases, human Hg exposures can also be reduced through risk communication policies, including fish consumption advisories, improved communication of the occupational risks associated with Hg releases during artisanal gold mining, and product labelling. Consumption advisories and the risk communication challenges associated with small-scale gold mining are often in focus. However, the practical implications of advisories on fish consumption have seldom been documented, and most likely vary a great deal depending on the nature of the advice, how it is communicated, and the alternatives available to the community.

Information digests for policy making with regard to the costs and benefits of reducing Hg emissions and exposure can be the way for communication of scientific knowledge to the policy makers at state, regional, and local levels. The development of information digests can be coordinated by the UNEP Chemicals Global Mercury Partnership. The digests should include the guidance on how to prepare economic analysis (such as the U.S. EPA, 2000) and databases with information on efficiency and costs of possible environmental protection measures (such as the EU ESPREME project database http://espreme.ier.uni-stuttgart.de) the EU DROPS project database (http://drops.nilu.no).

Mercury Information Clearinghouse is another example of how the latest information on mercury policy, measurement, baseline levels and emissions, and control can be communicated to policy makers, as well as general public with mercury users and consumers. An example of such information channel can be mercury Information Clearinghouse on Advanced and Developmental Mercury Control Technologies available from the national technical Information Service at the U.S. department of Commerce (U.S. DoC, 2004).

Training workshops can be organized in different parts of the world with regard to the use of existing methods and databases on the assessment of benefits and costs related to the reduction of Hg pollution worldwide. Such workshops can be organized under auspicious of the UNEP Chemicals Global mercury partnership.

16 Increase of knowledge and capacity among individual mercury users and consumers

16.1 Overall assessment of costs and benefits

Qualitative Cost Assessment: Small.
Cost Category: Consumer education.

Qualitative Benefit Assessment: Large.

16.2 Capacity building as an instrument for pollution mitigation

Increased information knowledge and capacity building among individual Hg users and consumers may be seen as a policy instrument to reduce the emissions of Hg and thereby the environmental and health impacts of this pollutant. Except regulations and market based instruments, capacity building information and co-operation are a cost effective instruments to mitigate pollution.

Capacity building can be defined as "*People helping people to build skills to change their own future. Skills can be built a number of levels, including at the level of the individual, organization, community or system"*. Furthermore, a World Bank summary of participatory processes refers to capacity building as the improved ability to make decisions about a project and transfer information between groups. The focus is on building people's capacity to participate in decision-making about a certain subject, as opposed to identifying capacities in a community and strengthening these elements.

- The information instruments: When it comes to informational instruments a distinction is usually made between information strategies for production and information strategies for consumption. Examples of information based strategies that may be introduced by government towards a cleaner production include (UNEP (2001)):

    - *promoting the adoption of targeted, high-profile demonstration projects, to demonstrate the techniques and cost-saving opportunities associated with cleaner production.*

    - *encouraging educational institutions to incorporate preventive environmental management within their curricula, particularly within engineering and business courses*

    - *issuing high profile awards for enterprises that have effectively implemented cleaner production.*

- Since it is often difficult or in some cases impossible for consumers to trace the original causes of environmental problems, it is vital that the authorities also use information instruments to improve consumers' understanding and awareness of these issues. Extensive research and monitoring work must be supported and published and public awareness of environmental issues should be increased through education and special training. Other informative measures such as environmental labelling schemes attempt to control consumption patterns by encouraging consumers to use products and services that are less harmful to the environment (Finland Env. Ad. 2006).

- Voluntary and co-operative regulatory instruments that do not involve the public directly include energy auditing schemes, promotion of energy savings, promotion of technologies, golden carrot programmes (e.g. subsidising development and implementation of energy saving products and technologies) and other 'soft' policy instruments. These programmes can be understood as subsidising development or supply of preferred technologies and subsidies for provision of certain types of costly information to firms.

The benefits of capacity building may be summarized in the following:

-increase recycling
-increased use of substitutes
-clean up of spills e.g., remediation of contaminated sites
-increase of storage of excess Hg

16.3 Communication of risk of Hg pollution to mercury users and consumers

Proper communication of risk of Hg pollution to mercury users and consumers is of vital importance with regard to reducing environmental and human health impacts. The issue of risk communication is related to the non-technological measures to limit emissions and exposure to mercury.

The most known non-technological methods of mercury emission reduction include: energy conservation and pollution prevention solutions. Energy conservation means using less energy to achieve the same level of energy service. Energy services include heat, light, sound, shaft power, and mobility. Decreasing energy production and use will result in the decrease of mercury emissions and provide additional benefits of reducing emissions of sulfur dioxide and other pollutants. A system of credits or vouchers could be developed and presented to the utilities for mercury reduction goals. Demand-side management (DSM) programs should be identified. The DSM refers to actions undertaken by, for example, electric utility to modify customer demand patterns. The DSM programs consist of information dissemination, technologies, or financial incentives.

A few solutions of pollution prevention can be presented for mercury, including:

- materials separation,
- product content bans,
- input taxes on the use of mercury in products, and
- labeling of products.

Material separation deals mostly with the separation of mercury containing materials from the waste streams of MWCs and MWIs. A very small portion of wastes (perhaps less than 1 %) containing very high content of mercury from batteries, fluorescent lights, thermostats and other electrical items needs to be separated from the rest of the wastes, such as paper, plastic, dirt, containing very low concentrations of the element. Several communities in many countries all-over the globe have already implemented household battery separation programs in an effort to reduce mercury in wastes to be incinerated.

Labelling the mercury-containing products would help consumers to select the ones which are mercury-free. This is particularly important for switches and devices that most consumers would not expect to contain mercury.

Consumer education and awareness is also an important aspect of dealing with the public health threats posed by mercury. Considerable benefit has been found in Europe and the United States in consumer awareness programs as awareness provides a critical tool for preventing exposures. Experiences with responding to and cleaning up local mercury spills and other unusual mercury hazards demonstrates that quick communication and effective response can make all the difference between mercury poisoning and a quick, easy clean-up effort. Experience also shows that making sure that contaminated areas are quickly identified and that local residents are aware of how to recognize and report environmental hazards can greatly help to prevent avoidable exposure. Well-established notification procedures go hand in hand with effective clean-up and rehabilitation policies and processes in helping to safeguard the public.

Elemental mercury is put to magico-religious uses, most problematically the sprinkling of mercury on floors of homes in Caribbean and Latino communities. Indoor mercury spills are persistent and release toxic levels of mercury vapour over long periods of time (e.g. Wendroff, 2005). It is claimed that ritualistic mercury contamination should be taken seriously by both the public health and the environmental health communities. Risk communication is an important issue in the matter.

Consumer advisory with regard to the risk to Hg pollution is also very important element of capacity building of users and consumers. Fish consumption advisory was mentioned in the previous chapter. Various reference doses with regard to safe level of methyl-mercury content in fish were proposed by various organizations, such as Food and Agriculture Organization (FAO), the European Commission, Health Canada, the U.S. Food and Drug Administration (FDA), the US EPA, ranging from 0.1 to 0.4 of MeHg per kg of body weight per day. It is very important that consumers are properly advised on the safe level of methyl-mercury in fish. However, one should be aware that this advice is based on solid scientific evidence. It should be remembered that eating fish provides high nutritional value such as vitamins A, F, and C, protein, omega-3 fatty acids, mono-lipids, iron and zinc.

Another example of a risk communication can be the advisory to small scale gold miners and their families (Swain et al., 2007). In the case of small-scale gold mining using Hg amalgamation, the primary toxicological issue is the inhalation of Hg converted to the gas phase during the heating of the amalgam. Heating often takes place inside or near the home. Artisanal workers and their families can be exposed to harmful levels of Hg vapor. Risk communication in the form of advice to avoid the Hg amalgamation technique or to reduce exposure during its use must take into account the limited options available to the gold miners and the widespread poverty and hardship associated with this occupation. Field researchers (e.g., Vega and Hinton, 2002; Spiegel et al., 2006) emphasize that effective risk communication strategies need to be intertwined with strategies targeting improved profitability through better gold recovery methods or reduced losses of Hg, thus reducing the artisanal miner’s production costs. Within each country, the industry is geographically scattered, so the logistical aspects of risk communication are a major challenge. Thus, to be effective, in each region, risk communication strategies may involve training a cadre of small-scale gold miners who can demonstrate and discuss the advantages of improved practices to their fellow miners (Spiegel et al., 2006).

Industries using mercury have also responsibility to take for informing the public on human risk related to their products. For example, the Computer TakeBack Campaign aims to protect the health and well-being of electronics users, workers, and the communities where electronics are produced and discarded by requiring consumer electronics manufacturers and brand owner to take full responsibility for the life cycle of their products (www.computertakeback.com)

17 Concluding remarks

During the recent decade major progress has been made in the assessment of anthropogenic sources of mercury and development of emission inventories on national, continental and even global scale, including development of scenarios addressing mercury emissions until 2020 (UNEP,2008). Key relevant to the consideration of costs and benefits are:

1) what are the abatement costs for Hg emission reductions using various measures in different emission source categories?, and
2) what are the environmental and societal benefits of Hg emission reductions?

In an attempt to deal with these questions, a qualitative assessment of the potential costs and benefits associated with Hg reductions within major emission source categories has been attempted in this report. This assessment started with the information on socio-economic consequences of mercury use and pollution, integrated and synthesized within the paper published by a group of authors including Swain (the lead author), Jakus, Rice, Lupi, Maxson, Pacyna, Penn, Spiegel and Veiga in the Ambio journal in 2007 (Swain et al., 2007). The paper is included in Appendix 1 with the permission from Ambio and the lead author.

A number of technical and non-technical measures are available for reducing the Hg emissions from: anthropogenic sources where Hg is a by-product (e.g. power plants, smelters, cement kilns, other industrial plants), waste disposal and other various uses. Measures differ with regard to emission control efficiency, costs, and environmental benefits obtained through their implementation. Very often Hg emissions are substantially reduced by equipment employed to reduce emissions of other pollutants. The best example is the reduction of Hg emissions by the desulfurization installations. The same applies to de-NOx installations, and control devices reducing emissions of fine particles. It can be concluded that technical measures for mercury emission reduction are available within the major emission sources categories, such as combustion of coal to produce electricity and heat, manufacturing of non-ferrous metals, iron and steel production, cement industry and waste incineration. These measures vary with respect to the emission control efficiency and cost. Most measures could reduce Hg emissions from the above mentioned sources by up to 90 % without employing any “add on” equipment, such as adding absorbents specific for Hg.

Since the sources of waste containing Hg differ and the emissions from these sources are local and/or region specific, the costs to reduce the generation of wastes differ depending on whether the source is in a developed country or in a less developed one. Preliminary Qualitative Cost Assessment reveals that these costs are variable depending on the management technique, such as incineration and land filling. Whilst the introduction of various emission control measures may give results in the developed world, the outcome of this strategy may not be very positive in the developing world where there is often no differentiation between municipal, hazardous and medical waste in terms of applied techniques or achievable emission limits. Therefore, emphasis in the developing countries should be put on developing adequate policy instruments to mitigate Hg releases.

Efficient, non-technological measures and pre-treatment methods are also available for the reduction of Hg releases from various uses of products containing mercury. These measures include ban on use and substitution of products containing mercury, and cleaning of raw materials before their use (e.g. coal cleaning). These measures also include energy conservation options, such as energy taxes, consumer information, energy management and improvement of efficiency of energy production through a co-generation of electricity and heat in coal-fired power plants. These measures also include prevention options, such as Hg containing wastes and material separation, labelling of Hg containing products, and input taxes on the use of mercury in products.

Capacity building through improvement of knowledge on Hg pollution impacts, emission reduction options, and their costs among authorities responsible for environmental protection in various countries and among individual mercury users and consumers is also a very important issue at present, as explained in the reported work.

The costs of mercury spills are not well documented and tend to be anecdotal. In the developed world costs to clean up small spills are very high compared to both the benefits of cleaning up as well as the costs of substituting the products with potential spill. For large spill the damage costs as well as the abatement costs are quite difficult to estimate depending on whether the spill takes place in a developed or a developing country.

Information is becoming available from the literature on benefits to the environment and society from implementation of various emission control measures for mercury. Environmental and human health consequences associated with Hg pollution have been studied for several decades, starting immediately after the Minamata disease was reported in 1956. The Swedish Medical Board issued bans for sale of fish from certain rivers and lakes due to high concentrations of methyl-mercury in 1967. The US Sport Fishing Institute suggested in 1969 that Hg may be a larger threat than DDT (cited in Eckley Selin, 2005). It became clear that ingestion of methyl-mercury with contaminated food is more dangerous than inhalation of inorganic mercury. Contaminated fish became the main factor in this context. However, there have been several reservations on how to relate the emission of inorganic Hg from various anthropogenic sources to the concentrations of methyl-mercury in fish and then ingestion of methyl-mercury. These reservations have not helped in the development of dose-response functions for Hg, which have only recently been developed.

The development of dose-response functions is a very important step in estimating environmental and human health benefits from reducing the Hg exposure and emissions in the first place. The societal cost-benefit analysis has not been a subject of the reported work. The results of the reported work are meant to contribute to such analysis to describe the environmental and socio-economic impacts of hg emission reductions at local, regional (e.g. continental) and global scale. However, the data reported here were used in another study to a preliminary estimate of societal costs and benefits of Hg reduction on global in the case that there will be a status-quo with Hg pollution in the future (Pacyna et al., 2008). This study concluded that the overall environmental and human health benefits from the reduction of mercury emissions from anthropogenic sources are considerable. The final conclusion of this study was that there are good reasons to invest in reducing Hg emissions and exposure in the future primarily for the sake of improvement of human health and more generally human welfare, also from economic point of view. The study reported here adds that there are measures for Hg emission reduction for which this investment can be done.

The development of dose-response functions has been also a very important factor in the assessment to what extend the emissions of Hg should be reduced. Shall they be phase-out or only reduce? If phase out, do we have substitution for Hg if necessary for the production process? If it is enough to reduce Hg emissions, do we have efficient enough measures? And the final question is, can we afford from economical point of view to implement measures, necessary for Hg emission reductions? The reported work has been challenged to provide at least some contribution to attempt answering these questions. However, more research is needed to provide the policy making with more detailed answers to the above questions. The matter is further complicated by the fact that mercury is a global contaminant with emissions in one region to be deposited in another region. It has been recognized that the current efforts to reduce risks from mercury are not sufficient to address the global challenges posed by mercury (governing Council Decision 24/3). Decisions on the most effective way to address these global challenges will need to be taken, and the consideration of the costs and benefits of taking (or not taking) actions will provide input into that discussion.

Information on costs and benefits of mercury reductions within the strategic objectives discussed in this report is available in Table 22.

Table 22: Costs and benefits of Hg emission reduction for various reduction options

Reduction option		Costs	Benefits
1	Reduction from coal usage	Medium →Large	Large
2	Artisanal and small – scale gold mining	Small → Large	Small → Large
3	Reduction of Hg trade emissions	Small	Large
4	Reduction from industrial processes	Medium → Large	Medium → Large
5	Reduction of waste generation	Small → Large	Large
6	Promotion of Hg waste collection and treatment	Small → Medium	Large
7	Reduction from waste disposal	Medium → Large	Large
8	Reduction of Hg consumption in VCM and chlor-alkali production	Small → Large	Medium → Large
9	Reduction of Hg use in products	Small	Large
10	Reduction from dental practice	Small → Large	Medium
11	Reduction of supply from mining and extraction	Small → Medium	Large
12	Reduction of supply from decommissioned cells and stockpiles	Small → Medium	Large
13	Prevention of contamination from spreading	Large	Medium → Large
14	Control and remediation of contaminated sites	Small → Medium	Large
15	Increase of knowledge among states	Small → Large	Large
16	Increase of knowledge among users and consumers	Small	Large

It can be seen from Table 22 that costs and benefits vary significantly within various Hg reduction options defined within the strategic objectives set out in Annex 1 of the report of the first meeting of the Open Ended Working Group. Measures with the application of technology, such as implementation of installations to remove Hg from the flue gases in electric power plants, waste incinerators, and smelters are rather expensive (medium to large costs) compared to non-technological measures, such as prevention activity, capacity building, and promotion of Hg-containing waste separation (small to medium costs). Both groups of measures would result in large benefits. This indicates that the technological and non-technological solutions for mercury emission and exposure reductions can be carried out in parallel. More emphasis on technological measures can be put in the developed countries, while the process of emission and exposure reduction in the developing countries may start with non-technological solutions. Technological solutions may be introduced in these countries gradually as a follow-up process after non-technological solutions are in place.

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Products containing waste: http://www.wastecap.org/wastecap/commodities/mercury/mercury.htm

Cost of incinator: http://www.frtr.gov/matrix2/section4/4-23.html

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Appendix 1

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Socioeconomic Consequences of Mercury Use and Pollution

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Socioeconomic Consequences of Mercury Use and Pollution

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Abstract

In the past, human activities often resulted in mercury releases to the biosphere with little consideration of undesirable consequences for the health of humans and wildlife. This paper outlines the pathways through which humans and wildlife are exposed to mercury. Fish consumption is the major route of exposure to methylmercury. Humans can also receive toxic doses of mercury through inhalation of elevated concentrations of gaseous elemental mercury. We propose that any effective strategy for reducing mercury exposures requires an examination of the complete life cycle of mercury**.** This paper examines the life cycle of mercury from a global perspective and then identifies several approaches to measuring the benefits of reducing mercury exposure, policy options for reducing Hg emissions, possible exposure reduction mechanisms, and issues associated with mercury risk assessment and communication for different populations.

INTRODUCTION

Mercury (Hg) has historically been used in a wide variety of activities, both in compounds and as a liquid metal, resulting in widespread dispersion and the creation of some heavily contaminated sites. Geological materials that contain high concentrations of Hg are usually found only in areas known as mercuriferous belts, such as along the west coast of the Americas. Even though most natural materials (including coal, oil, and minerals) contain very low levels of Hg, the use of large quantities of these materials releases significant amounts of Hg into the biosphere each year. Because Hg can vaporize at ambient temperatures, the earth’s atmosphere plays a major role in the dispersion of Hg. Human activities, including Hg mining, use, and waste disposal, combined with releases from the refining of other metals and the combustion of fossil fuels, have significantly increased Hg emissions to the atmosphere. Compared to preindustrial times, atmospheric deposition has increased uniformly in remote regions by a factor of about three (± 1x) (1). Deposition increases above three-fold have been documented near emission sources; these depend on stack height, the quantity and chemistry of the emitted Hg, and local atmospheric chemistry (1). Bacteria in aquatic systems convert a small proportion of the deposited Hg to methylmercury (MeHg), which bioaccumulates in fish (inorganic Hg does not bioaccumulate). Aquatic systems vary in the efficiency with which atmospherically deposited Hg is bioaccumulated in fish. For example, the Hg concentration of fish in neighboring lakes can vary by as much as ten-fold, even when atmospheric Hg is similar (2). Nevertheless, in a given aquatic system the production of MeHg is believed to be approximately proportional to atmospheric Hg deposition, so it is likely that historical increases in Hg emissions have increased MeHg concentrations in fish (3).

Hg use and releases are of concern because of two exposure pathways: consumption of MeHg-contaminated fish and inhalation of elevated concentrations of Hg vapor. The most common route of MeHg exposure for humans and wildlife is the consumption of fish from marine and freshwater sources. While low levels of Hg vapor are normally measured in the atmosphere, elevated concentrations can result from a variety of intentional Hg uses and accidents involving Hg, especially indoors where dilution is constrained.

In industrialized societies, although Hg was once a component in many products, it is now viewed as a material for which the risks of use generally outweigh the benefits. Hg-free substitutes are now viable for nearly all uses (energy-efficient lighting remains a notable exception), although their adoption varies greatly across jurisdictions and industrial sectors. The link between Hg emissions to the atmosphere and fish contamination has influenced policy development in industrialized countries since the mid-1990s, variously resulting in mandatory control of Hg emissions from waste incineration and other sources, restrictions on the labeling, sale, and disposal of Hg-containing products, and control of Hg emissions from coal-fired power plants as recently mandated by federal and state regulations in the United States (4, 5) and Canada (6).

Hg is also a commodity, with net flow from industrialized countries to developing countries, where its uses are less constrained. Annual global Hg consumption peaked at about 10 000 tonnes (t) in the 1960s; at that time large amounts were used in electrolytic and chemical processes, pesticides, paints, and batteries. Use gradually decreased to an estimated 3500 t in 2005 (Fig. 1a). In 2000, however, global trade statistics reveal that at least 9000 t of metallic Hg were bought and sold across national borders, confirming that Hg is an actively traded commodity. However, the Hg market is opaque, with considerable uncertainties associated with many of the estimates presented here.

Significant and growing coal combustion, particularly in developing countries, and industrial-scale refining and smelting of metals also are responsible, in part, for the continued elevation of atmospheric Hg levels (10). Although global Hg pollution of the biosphere is likely a major contributor to the widespread MeHg contamination of fish, effective international agreements that address Hg pollution have yet to be developed.

This paper describes how certain human activities mobilize Hg from geological stores into the biosphere and outlines the resulting social and economic consequences. We present possible policies to reduce exposure to Hg and MeHg, and explore issues associated with risk assessment and communication for different populations. We utilize substance flow analysis, which reveals not only the source and fate of a material, but also potential control points. We discuss policy options for addressing control points in the flows of Hg that could ultimately reduce fish contamination while simultaneously reducing the potential of harmful exposure to Hg vapor.

Concern about MeHg in fish

Exposure to MeHg through consumption of contaminated fish by humans has the potential to interfere with normal neurological development and function and may increase the risk of myocardial infarction (heart attack) (11). Negative neurological and reproductive effects of MeHg on fish-eating wildlife—birds (*e.g.*, loons), mammals (*e.g.*, river otter and mink), and fish that eat other fish—are also a concern (12).

Concern about inhalation of Hg vapor

Absorption of liquid Hg through the skin or digestive system is quite limited, but the body retains 80% of inhaled Hg vapor (13). An indoor spill of metallic Hg, even in quantities as small as a gram, can give rise to Hg concentrations in ambient air that approach WHO recommended limits for occupational exposure. Heating metallic Hg and Hg compounds, such as cinnabar and scrap dental amalgam, has generated fatal inhalation doses (14). Although heating Hg is a rare activity in most societies, an estimated 10 to 15 million people earn their living in small-scale gold-mining operations by amalgamating gold with Hg, and then concentrating the gold by heating the amalgam, thereby releasing the Hg in vapor form. This process directly exposes the miners and can expose their families and neighbors, (estimated to total about 50 million people), to elevated Hg levels (15). In 2005, small-scale or artisanal gold mining was estimated to contribute more than 10 percent of annual global anthropogenic Hg loading to the atmosphere (about 300 out of a total 2400 tonnes, Fig. 2).

Concern about Hg vapor extends beyond small-scale mining to include Hg in products. One such application, dental amalgam, is the most common way that people are exposed to elevated Hg vapor, although the levels are not as high as those frequently encountered in artisanal gold mining. Even though Hg levels in autopsy brain samples correlate positively with the number of amalgam fillings, vapor concentrations from amalgam are usually well below those associated with even subtle neurobehavioral effects (13). Otherwise, Hg is usually encapsulated in consumer and commercial products, such as fluorescent lamps and thermometers. However, these devices can break during use or disposal, releasing Hg into indoor air and the environment. Historical uses of Hg left a legacy of many small bottles of liquid Hg. In the United States, Hg spills have caused expensive and disruptive clean-up efforts (16). Although adverse health effects have rarely been documented from Hg spills, we do not consider this as evidence of a low potential for harm. For example, three children were hospitalized in Michigan (USA) as a result of Hg poisoning, including one child who was no longer able to walk. Investigation revealed that exposure occurred after a small vial of Hg was spilled in the children's bedroom approximately two to three months prior to detection of the gross symptoms (17).

GLOBAL COMMERCIAL AND ENVIRONMENTAL PATHWAYS

Although somewhat simplistic, conceptually, it is useful to divide the earth's Hg between two repositories: Hg in the biosphere and Hg stored in geological formations. The biosphere encompasses the portion of Earth and its atmosphere that supports life; geological formations include deep sediment deposits, and buried minerals and fossil fuels. The most significant human impact on Hg is its mobilization from long-term geological storage into the biosphere, where Hg cycles between air, soil, and water.

Assuming only two repositories is simplistic because mobilized Hg can enter sub-compartments that appear to be stable, such as Hg encapsulated in a device such as a thermometer, Hg that has settled into deep ocean waters, and Hg sequestered in deep soils, such as the vast peatlands of Canada and Siberia (18, 19) or the ancient soil of the Amazon Basin (20). However, use of only two repositories is useful from a policy perspective because human activity can mobilize the Hg from apparently stable compartments of the biosphere. Thermometers inevitably break, and the Hg in soils can be mobilized through fire (19), forestry practices (21, 22), agriculture (20), climate change (18, 19, 23), and the creation of reservoirs (24). Additional research is needed to understand the fate of Hg that enters both coastal and deep-water marine systems (1).

The movement of Hg into and out of the biosphere is shown in Fig. 2, based on the global model of Mason and Sheu (25), a model that is representative of the current scientific understanding of global Hg cycling (see review of four models by Seigneur et al. (26)). Although there is a considerable range of estimates in the models for specific components of the global Hg cycle and there is uncertainty regarding the scale of natural emissions and reemission of deposited Hg (1), the four models converge on a relatively tight range for total global emissions (natural and anthropogenic); that is, from 6060 to 6600 t yr^{-1}, and for current anthropogenic emissions alone, from 2000 to 2400 t yr^{-1}. Reemission of previously deposited anthropogenic emissions roughly equals current anthropogenic emissions (1). In general, the global models can be considered uncertain, but useful, frameworks.

Although current anthropogenic emissions in Fig. 2 are constrained to 2400 t yr^{-1} to conform to the Mason and Sheu model, it would be reasonable to assign an uncertainty of ±30% to total anthropogenic emissions (1), with some sectors embodying less uncertainty than average and some sectors more. It has been suggested that Hg emitted from global coal combustion has a ±25% uncertainty; nonferrous metal production ±30%; waste disposal and incineration has uncertainty up to 500%; and that Hg use in artisanal and small-scale gold production is too poorly understood to allow a quantitative uncertainty factor to be assigned (1). A protocol, or "toolkit," has been developed to assist countries that wish to identify and quantify sources of Hg emissions (27). The toolkit, including extensive text and a spreadsheet template, explains how to develop detailed quantification of Hg pathways that are only broadly defined in Fig. 2.

Given the uncertainties in the current understanding of global Hg pathways, Fig. 2 can reasonably be used as background information for discussions of policy options that might be adopted to reduce Hg emissions and, ultimately, exposure from fish consumption and small-scale gold mining.

Each letter in Fig. 2 codes for a compartment in which Hg is stored or from which it is released (see Tables 1 and 2). A two-letter code indicates a path from one compartment, indicated by the first letter, to another compartment represented by the second letter. For instance, VL denotes the pathway of atmospheric Hg to the Earth's surface whereas LV is the path of Hg volatilization from land to the atmosphere, with the annual flux in tonnes next to each arrow. Note that Hg is relatively mobile in the biosphere; in this model, over 80% of the Hg deposited to the oceans is reemitted to the atmosphere. In contrast, only about half of the Hg deposited to land is reemitted within a few years, because Hg can associate strongly with soil.

Pathways with an asterisk (*) are amenable to manipulation to minimize Hg release or exposure at the time this manuscript was written. Therefore, we refer to these pathways as control points. For example, FH* denotes that the exposure of humans (H) to MeHg from contaminated fish (F) may be reduced through selection of fish to consume; it should be noted, however, that in some societies or regions there may be little choice of fish species to consume. The lack of an asterisk indicates that it is unlikely that present social or economic policies can affect that pathway. MP* indicates that the creation of products by manufacturers does not have to include Hg. PV indicates that once Hg is a product component, it is inevitable that some of that Hg will be released to the atmosphere as a result of breakage, no matter what policies, such as recycling, are adopted.

FOSSIL FUEL COMBUSTION: Compartment C

Fossil fuels (coal, oil, and natural gas) contain a wide range of Hg concentrations in their natural state. Some natural gas supplies are high in Hg vapor, but the Hg is routinely removed during refinement before distribution to avoid degradation of aluminum heat exchange surfaces via amalgamation in gas-processing plants (30). Few analyses have been performed on the Hg content of petroleum or oil sands, or on the fate of Hg throughout the oil exploitation and refinery process. While there is a lack of general data on Hg concentrations in oil and oil sands in their natural state, once refined, the Hg emissions from the combustion of these appear to be much lower than from coal (30).

Coal combustion is responsible for about 60% of anthropogenic Hg emissions (pathway CV*, 1500 t yr^{-1}; Fig. 2). A continuous decrease in Hg emissions in Europe and North America since 1980 has resulted from the installation of emission-control equipment, particularly electrostatic precipitators (ESPs), fabric filters, and flue gas desulphurization (FGD) technologies. These existing technologies incidentally capture Hg largely to the extent that coal combustion produces divalent Hg. Metallic Hg vapor (Hg^0) is usually poorly controlled by equipment designed to capture particulates or sulfur. The emitted form of Hg (and its removal) is highly dependent on the coal type and the installed emission control equipment (31).

There are two major types of FGD systems: wet and dry. In general, removal efficiency for Hg in FGD systems ranges from 30 to 85% depending on the proportion of divalent Hg, which is related to the halogen content of the coal. The greatest removal efficiencies with existing technology can be expected when a wet FGD system is installed downstream of a fabric filter (31).

Most utility power plants in developed countries are equipped with either ESPs or fabric filters for particulate control. Removal of Hg by these devices is highly dependent on halogen content and unburned carbon in the flue gases. Generally speaking, fabric filters remove approximately twice as much Hg as ESPs under similar flue gas conditions (5, 31). As part of efforts to control Hg emissions from coal-fired power generation, significant new information is emerging on flue gas chemistry (Hg, chlorine, sulfur, and the effect of nitrous oxide, or NOx, control) that will help to improve Hg control. A wide variety of technologies are being developed (5, 32, 33) and applied (34) to reduce Hg emissions. As with other emission control technologies (e.g., SO_2, NOx, etc.), it is possible that innovations will yield commercially available options that are significantly less costly than initially estimated (35).

ORE REFINING: Compartment O

Ores of metals, especially nonferrous metals such as gold, silver, copper, lead, zinc, and nickel, often contain Hg because the geological processes that concentrate these metals typically also concentrate Hg. In particular, sulfide ores often contain significant concentrations of Hg, because of a high chemical affinity between Hg and sulfur. Cinnabar—mercuric sulfide—has been the main source of Hg as a commodity for thousands of years. Despite recent mine closures in Spain and Algeria, cinnabar mining of 1300-1400 tonnes annually remains the largest source of elemental Hg (Fig. 2; 9) and is increasing in China to meet internal demand. The processing of cinnabar is associated with elevated atmospheric Hg emissions. As global Hg demand declines, however, less expensive sources of Hg—recycling, Hg from closed chlor-alkali plants, and as a byproduct from large-scale gold, zinc, and copper mining—may be expected to supply a greater proportion of Hg demand. The availability of Hg as a byproduct may increase significantly if new regulations restrict Hg emissions from ore refining.

Hg IN MANUFACTURING AND PRODUCTS: Compartments M and P

Sources of Hg to industries include new Hg from current mining (pathway OM) as well as recycled Hg (pathway RM). Hg has been intentionally used in many products and processes, although consumption decreased by ~ 50% from 1990 to 1998 (Fig 1). Annual use of Hg in products in 2004 (Fig. 3) was comprised mostly of dental amalgams (270 t), electrical switches and relays (150 t), measuring/control equipment (160 t), energy-efficient lighting (110 t), and disposable batteries (estimated at 600 t). Smaller quantities also continue to be used in an array of other products. For example, some Hg is used in cosmetics; in the USA, up to 65 ppm is allowed. Hg continues to be used in chlor-alkali (chlorine and caustic soda) production (possibly 700 t in 2004, but declining every year) and as a catalyst for the production of vinyl chloride monomer. The latter remains a significant use in China, India, and Russia (estimated at 250 t yr^{-1} in this paper, although recent information from China puts this number as high as 600 t (37)).

Before 1990, large quantities of Hg were used in ways that dispersed Hg widely in the biosphere; these uses included fungicides used in seed coatings, the paper industry, and latex paints (38). Today, most Hg added to products is not released during normal use, with the exception of Hg-containing skin-lightening soaps and creams; vaccines containing thimerosal (a Hg-containing preservative that is injected into the body during vaccination); dental amalgams; and volatilization from Hg-catalyzed polyurethane products. The largest remaining dissipative use of Hg is in dental amalgams, which can result in direct human exposure during inhalation, occupational exposure in the dental office, releases to wastewater both from dental offices and homes, emissions during incineration of dental wastes, and flue-gas emissions during cremation (pathways PH, PV, PD*, DA*, and DV*).

Electrical switches and relays, measuring/control equipment, energy-efficient lighting, and batteries do not release Hg until disposal, except due to misuse or accidents (pathways PH and PV). It is possible to manufacture these products in such a way as to limit air or water emissions of Hg (pathways MV* and MH*). Therefore, the extent of Hg emissions from Hg-added products is likely a function of the method of disposal (pathways MD* and PD*). Chlor-alkali factories reportedly have greatly decreased their Hg releases in many nations (39, 40). However, considerable uncertainty remains regarding the environmental fate of hundreds of tons of Hg unaccounted for annually by the global chlor-alkali industry, much of it released within developing countries.

Polyvinyl chloride (PVC) is typically manufactured from petrochemical feedstocks. There is, however, a Hg-catalyzed method to make a PVC feedstock, vinyl chloride monomer (VCM), from coal; this method typically results in large Hg consumption. Such PVC production is a substantial contributor to Hg releases (pathways MV*, MH*, but mostly MD*), conservatively estimated at

150 t yr^{-1} (9, 41), but possibly twice that amount (37). However, there remains considerable uncertainty as to what part of those releases go the atmosphere in contrast to other waste streams.

The substantial reduction of global Hg use (Fig. 1a) is due to two factors: (1) substitution of non-Hg products (*e.g.*, paints, batteries, thermometers, and pesticides) and production processes (mainly chlor-alkali) and (2) more efficient use of Hg in production, except, typically, in cases where production has been shifted to developing countries.

SMALL-SCALE GOLD MINING: Compartment S

A combination of high gold prices and persistent poverty contributes to a proliferation of small-scale gold mining that uses Hg amalgamation to concentrate gold (42, 43). While the scale of gold production by artisanal miners is not well defined, it may constitute 20 to 30% of global gold production, ranging from 500 to 800 t of gold each year (44, 45) and occurs in over 50 developing countries (15). Because gold is easy to sell and transport, and its value remains relatively stable in countries with unstable currencies, it constitutes one of the more important extraction economies (42).

Hg is widely used in small-scale gold mining, despite laws prohibiting its use. Hg is combined with gold-containing silt or crushed ore to form a gold-Hg amalgam, simplifying recovery of the gold. Generally the amalgam is then heated with a blowtorch or over an open fire, vaporizing the Hg and leaving the gold behind (pathways SV*, SH*). Miners are estimated to lose, on average, 1 to 2 grams of Hg per gram of gold produced; thus, this process annually releases approximately 1000 t of Hg to the biosphere, of which an estimated 300 t is emitted directly to the atmosphere (Fig. 2, 3). Virtually all of the Hg consumed by this activity is released somehow to the environment. The leading consumer of Hg through this activity is thought to be China (200 to 250 t yr^{-1}), followed by Indonesia (100 to 150 t yr^{-1}), while Brazil, Colombia, Peru, the Philippines, Venezuela, and Zimbabwe each consume an estimated 10 to 30 t yr^{-1} (46-48). The unregulated trading of Hg in developing countries makes it readily available at the mine sites (pathways OS*, RS*).

Hg use in small-scale mining has left a legacy of thousands of polluted sites with impacts extending far beyond localized ecological degradation, often presenting long-term health risks to persons living in mining regions (49). Inhalation of Hg vapor is the primary exposure pathway for miners, gold shop workers, and people living near areas where the gold-mercury amalgam is produced and processed. Miners and community members often breathe air with Hg concentrations above 50 micrograms per cubic meter—50 times the World Health Organization maximum public exposure guideline. Consequently, many miners and others—particularly amalgam burners, who are often women—demonstrate tremors and other symptoms of Hg poisoning (50). Two mining practices may be increasing local Hg exposure due to inadequate information: improper use of retorts to recover Hg may increase exposure to vapor (51) and cyanide leaching after Hg amalgamation may increase Hg bioavailability and fish contamination (15).

UNIDO efforts to reduce Hg releases have yielded new retorting techniques using readily available pipes and kitchen bowls, which allow miners to contain Hg emissions and recycle as much as 95% of the Hg from the vaporization process (52, 53). Such approaches are garnering increased attention, in particular as Hg prices increased four-fold and more between 2002 and 2005 (Fig. 1b). While the international price for one kg of Hg increased from less than \$5 to more than \$20 (all prices in constant 2005 U.S. dollars), the price for a kg reached more than \$100 in the small-scale gold mining sites in Mozambique, Zimbabwe, and Indonesia.

Hg-free alternatives to dissolve gold (cyanide) or to concentrate gold (e.g., gravity separation, magnetic sluices, and coal-oil gold agglomeration methods) are relatively costly and currently inaccessible to most miners in developing countries (54). Other promising Hg-free techniques to extract gold from concentrates include electro-oxidation and alternative methods of leaching, such as the iGoli Process. Widespread adoption of Hg-free gold-mining methods would require substantial time and investment in both technology and social-change efforts (55).

RECYCLING AND RETIREMENT (Compartments R, D, and X_T)

All products eventually enter the waste stream. Hg contained in waste products may be recycled (DR*), incinerated (DV*), left in place, dumped on land, dumped into wastewater (DA*), released through breakage during use or disposal (PV) or "retired" through placement in a warehouse, engineered landfill, or deep bedrock repository (DX_T). Every time Hg moves from one compartment to another, there is some loss to the atmosphere and potential for human exposure via inhalation and eventually via deposition, aquatic methylation, and fish consumption (56-58). In some societies, regulations may protect workers from exposure, incinerators may have Hg-control devices, and diversion of Hg-containing products from the waste stream may be mandated. But in much of the world, Hg exposure and disposal are unconstrained (44). The incentive for firms in industrialized countries to recycle is regulatory in nature rather than economic, because the market value of recovered Hg is usually much lower than the cost of recycling. On the other hand, recycled Hg may be considered a "cheap" Hg source because it is a product of the waste disposal process that has already been paid for.

Recycling

From a sustainability perspective, recycling of materials is generally preferred over landfill disposal because recycling obviates the need for (and costs associated with) landfills as well as the environmental costs associated with extraction of virgin material. However, unless it is integrated into a larger strategy of stable or decreasing Hg supply and demand, recycling Hg may not decrease global Hg pollution levels if the recycled Hg is merely returned to the marketplace. In the latter case, Hg recycling could have the effect of increasing the Hg supply and decreasing the price.

Under current regulations in most developed countries, it is cheaper to recycle waste containing a significant percentage of Hg and to sell the recovered Hg on the open market, than it is to dispose of Hg-bearing waste at a hazardous waste landfill (59). The generation of Hg from recycling and the recovery of Hg from decommissioned chlor-alkali plants have become increasingly significant contributors (10-20% in recent years) to global supplies as recycling has increased and the production of mined Hg has declined. However, in the interest of eliminating surplus Hg supplies from the global market, the European Union (EU) draft regulation for a Hg export ban—presently under discussion in accordance with Actions 5 and 9 of the EU Hg strategy. This ban targets all Hg from decommissioned EU chlor-alkali plants for retirement as of 2011 (60). The supporting analysis suggests that recycled and by-product Hg (along with reduced Hg mine production, as necessary) will be more than adequate to meet global Hg demand (36).

Retirement

The alternative to marketing surplus Hg is the intentional retirement of Hg; that is, permanent storage that removes Hg from commerce and the biosphere. Hg need not be recycled, or purified, before it is retired. Unprocessed wastes or pure Hg could be stored indefinitely in warehouses or engineered land disposal sites, such as lined and capped landfills. There have been few efforts to compare the benefits of recycling and retirement, although Sweden has decided to retire Hg wastes (61).

The U.S. government, through the Defense National Stockpile Center (DNSC), owns one of the world's larger stocks of Hg, and in the early 1990s began selling it on the international market after declaring it unneeded for future defense needs. A moratorium on sales was declared in 1994 as a result of concerns that marketing Hg may contribute to global environmental contamination. The relative merit of selling versus retiring the Hg was studied (62), and in February 2006 the U.S.

government announced that the stockpile of some 4400 t of Hg would be stored indefinitely in a warehouse.

Hg IN FISHERIES AND HUMAN COMMUNITIES: Compartments F & H

The consumption of fish is the primary route of exposure to Hg in the form of MeHg (pathway FH*). Patterns of human exposure to MeHg are largely determined by the global distribution of fisheries, the trade in fish, and fish consumption.

Concentrations of MeHg in fish tissues vary by over a factor of ten (63) because of variation in the environmental biogeochemical pathways of Hg and in aquatic foodwebs. Efforts effort to quantify the benefits of reducing anthropogenic Hg pollution are dependent on biogeochemical and foodweb models relating Hg releases to fish concentration. Much of the research on the biogeochemical pathways of MeHg has focused on freshwater ecosystems; in contrast, marine fish make up 92% of the global fish harvest (64). The quantification of reduction benefits would be enhanced if research efforts more closely matched the source of fish that people eat.

The global fish harvest has increased in recent decades and appears to have stabilized at around 130 million t, a figure that includes aquaculture (30 million t) and fish reduced and processed for use in meal employed in both agriculture and aquaculture (20 M t) (64, 65). Based on these estimates, per capita consumption is 24 and 14 kg live weight/year in developed and developing countries, respectively. In some countries (notably in the Western Pacific region), annual fish consumption is appreciably greater, ranging up to 75 kg/person-year. Freshwater fish account for about 5% of the total fish harvest in developed countries and 15% in developing countries.

Over one-third of the global marine harvest of approximately 85 million t enters international trade. Half of the production from developing nations, including tuna and other high-value piscivores, is exported to industrialized nations. The ability to assess patterns of exposure to MeHg is limited by data needed to link Hg concentrations to trade statistics. The quantity of fish produced via aquaculture is an increasing proportion of the global fish production (some 25%), but little is known about MeHg concentrations in these fish.

Fisheries are not easily classified. Commercial, recreational and subsistence fisheries are often closely inter-related. Subsistence fishers often sell part of their catch commercially, a practice that yields cash for goods that must be purchased, such as boats, motors, gasoline, and market foods. Small-scale production of fish appears to involve some 35 million producers and their families, of whom fewer than 5% are from developed nations (65). Subsistence fishing—fishing primarily for local distribution and consumption—remains a significant human exposure pathway in some populations, and can be associated with relatively high levels of chronic MeHg exposure (66, 67). Subsistence fisheries are often held as common property with rules of conduct tending to be informal, local, and unwritten. Quantitative data on production and distribution of fish are rarely available, thus complicating the evaluation of MeHg exposure. The Arctic and sub-Arctic regions present a challenge on several fronts, including the finding that Hg from lower latitudes may be deposited by so-called ‘Hg-depletion events’ (1), even though there are few anthropogenic emission sources. In some Arctic communities, marine mammals are also a significant dietary source of MeHg. In both Arctic and Sub-Arctic settings, alternatives to locally harvested fish and marine mammals may be culturally untenable, unavailable or not affordable (68).

ECONOMIC ANALYSES OF MERCURY USES AND POLLUTION

Economists often discuss a good’s “opportunity cost”; that is, what must be sacrificed to obtain a good or service. For most goods, this is reflected in the price. But when the production of a good involves the release of a pollutant, such as Hg, the price may not include the associated environmental and social costs; these costs are called externalities. A negative production externality implies that individuals are adversely impacted by production in ways that did not get factored into the product price; for anthropogenic Hg releases, this results when individuals are exposed to Hg or MeHg, as described in Fig. 2. Economists have developed a number of methods to measure the costs of these externalities. We briefly review those methods here, focusing on methods to quantify the benefits to human health associated with reduced Hg exposure. Relative to humans, the effects of Hg on wildlife are poorly understood, so with the exception of the effect of consumption advisories on recreational fisheries, the economic value of reducing Hg pollution has seldom been quantified for wildlife and ecosystem functioning. As our review of existing studies shows, the scientific uncertainties even for humans lead to differing assumptions about health outcomes, which are translated into substantial variation in benefit estimates.

An additional consideration is that economic valuation methods do not work well when the implied tradeoff exceeds a person's or community's ability to pay. Economic analysis becomes very challenging if such tradeoffs are not possible, as might be the case when valuing Hg contamination of subsistence fishing communities with no practical dietary substitutes for Hg-contaminated fish.

Few economic studies have been conducted to quantify the benefits of reducing Hg pollution, and all of those have been conducted in the U.S. (Table 3). Among those few studies, there is only one that included wildlife benefits (69). Including the effects of Hg on wildlife and ecosystems is a daunting task; those studies that exclude such benefits will underestimate the full benefits of Hg reduction (70).

Economists employ two general approaches in measuring the human health benefits associated with a policy (71). Benefit-cost analysis (BCA) evaluates changes in health using monetary values, such as the cost of illness (COI) and willingness to pay (WTP) or willingness to accept (WTA) approaches. Cost-effectiveness analysis (CEA) evaluates these changes using summary measures of population health (e.g., disability-adjusted life years (DALYs) and quality-adjusted life years (QALYs), more generally known as health-adjusted life-years (HALYs). BCA collapses all health-related benefits and costs into a single, monetary metric from which economic efficiency can be evaluated. Those policies in which the incremental benefits exceed the incremental costs are beneficial to society. In contrast, CEA does not monetize benefits, instead this approach evaluates potential policies by comparing the ratio of a policy's cost to its health outcome, in cost per HALY. Policies with low cost per HALY ratios are typically preferred to those with high ratios.

Health benefit assessment using benefit-cost analysis

COI methods are used frequently to monetize the health improvement associated with a change in morbidity (illness). These methods measure the direct costs (*e.g.*, treatment costs) and indirect costs (*e.g.*, foregone income) associated with illness and injury, but do not measure losses associated with pain and suffering. For MeHg, COI methods link changes in the exposure of pregnant women to modeled changes in the IQ of their offspring. IQ changes are subsequently linked to changes in future income (throughout adulthood) and supplemental educational costs. Similarly, MeHg exposures can be linked to non-fatal heart attacks, with COI used to estimate the benefits of Hg reduction policies.

WTP methods measure an individual's willingness to exchange wealth for health. Economists consider such methods to be superior to cost-of-illness methods because they include factors such as impairments to quality of life. For example, the value of a statistical life (VSL) measures an average individual's willingness to pay for a small change in the probability of dying sooner (72-75). Consider a Hg-reduction policy that could reduce the probability of death in a population from 2×10^{-5} to 1×10^{-5}, a reduction of one death per 100 000 people. If people are, on average, willing to trade \$60 in exchange for this policy, then the VSL is \$6 million (\$60 divided by 1×10^{-5}). Currently, the most common estimates for VSL used by researchers and policy makers in industrialized countries are between $3 and $7 million; the range arises, in part, due to the different values people hold for different types of mortality risks. VSL is positively related to income: a VSL estimated in a high-income country is generally greater than that estimated in a low-income country.

Table 3 summarizes a number of studies that attempt to monetize the benefits (usually human health benefits) associated with reduced MeHg exposure. The BCA conducted by the USEPA (76) as a component of the Clean Air Mercury Rule (CAMR) examined the costs incurred by U.S. power plants to reduce Hg emissions and the monetized benefits associated with IQ increases in the children born to pregnant women whose MeHg exposures would be reduced as a result of the rule. Other groups also conducted benefit analyses of various Hg emissions reduction proposals under consideration during the development of this regulation (77-81). While the economic valuation models used are similar, assumptions regarding the impact of decreased Hg emissions on the changes in MeHg levels in different types of fish, and the health effects considered, differ markedly. All of the analyses emphasized the numerous uncertainties in evaluating specific policies for Hg reduction, including, (i) changes in Hg deposition rates, (ii) changes in fish MeHg levels, (iii) changes in MeHg intake by humans and the time it takes to observe this change, (iv) changes in IQ due to fetal exposure, and/or changes in all-cause mortality and fatal and non-fatal heart attacks in adults. Some analyses assumed that Hg emissions markedly decreased only U.S. freshwater fish MeHg levels; others assumed that marine fish levels, the primary source of MeHg exposure in the U.S., could also decline as a consequence of decreased emissions. All analyses considered the impact of reduced fetal MeHg exposures on changes in IQ and lifetime earnings, although the slope of the MeHg IQ loss dose-response functions and lifetime earnings estimates vary. Some of these analyses also examined the impact of a toxicity threshold for the fetal neurotoxicity of MeHg that would be consistent with the U.S. Environmental Protection Agency's reference dose. Other studies included the possible economic impacts of decreased myocardial infarctions and premature mortality in adults. (See 11, 76, 80, and 82 for descriptions of the uncertainties in the epidemiologic data regarding these health endpoints.) The U.S. EPA also simulated the time required for freshwater fish levels to change following an emissions reduction. These differences led to a large range of benefit estimates across the studies.

Much of the variability across the study results presented in Table 3 is a direct consequence of the differing assumptions that are made in response to uncertainties in the physical and health sciences of Hg and MeHg. For studies that have focused on nationwide programs, such as the emissions cap or the Clean Air Interstate Rule (CAIR) proposed in the U.S., we can see the impact of differing assumptions by comparing the results on a per capita basis. The EPRI study and that by Gayer and Hahn estimated costs of a 15 tonne cap and trade program at between $15 (Gayer and Hahn midpoint) and $21 (EPRI) per capita. Gayer and Hahn, focusing solely on IQ losses, estimate benefits of a Hg cap at less than $1 per capita. In contrast, the Rice and Hammitt study, examining a similar program but including IQ, heart attack and all-cause mortality effects and assuming that there will be changes in both freshwater and marine fish, estimate benefits at about $16 per capita. The authors noted that heart attack and all-cause mortality effects and marine fish MeHg reductions are much less certain than the other factors included in their analysis, but they did not discount the future benefits. Gayer and Hahn firmly conclude that the benefits of the Hg cap and trade program are less than its costs. Palmer et al., looking at Hg within the context of the CAIR, conclude that benefits (using Rice and Hammitt's estimate of $16) exceed the estimated costs of CAIR ($12 per capita). The primary factor driving the different conclusions is the decision on which health endpoints to include, a decision based on uncertainty in the physical and health sciences.

Economic benefits are based, fundamentally, on a person's WTP for reductions in Hg exposure. Whereas the studies in Table 3 represent a good start to a benefits literature, none of them address the theoretically correct, but difficult to estimate, WTP measures that incorporate uncertainty, such as "option price" (87). For example, consider exposure over time to a given level of Hg in commercially caught seafood. As a result of that exposure, one raises the risk of developing Hg-related health problems, such as AMI, above the baseline risk. Different people receiving the same Hg exposure will react differently: some will have a heart attack and others will not. The option price (OP) measures a person's WTP without resolution of this uncertainty, that is, without finding out if he or she is one of the people whose sensitivity to MeHg increases the likelihood of heart attack. The basic OP model can be augmented to incorporate uncertainty in the health risk itself (e.g., is the best estimate of MeHg-associated increase in AMI risk zero or, is the dose-response slope 0.066 per ppm mercury in hair); it can reflect endogeneity in the risk (e.g., risk estimates can be adjusted based an individual's actions, such as eating more or less fish), and it can include ambiguity about risks (a person may not have a good point estimate of their risk and instead consider their risk to be within a given range). An approach to economic analysis that explicitly incorporates uncertainty seems well suited for the case of Hg, in which policy choices must be made despite the presence of scientific uncertainty in environmental fate, exposures and health effects.

The difficulty with such an approach is that different studies will produce different results. In most cases, a WTP measure such as OP would be elicited via stated preference methods, and WTP estimates will depend on the prior information that respondents have as well as that provided by the analyst. To the degree that researchers differ in the risks and uncertainties explained to respondents, study results will vary and must be evaluated within that context.

Health benefit assessment using cost-effectiveness analysis

The most commonly used HALY measures in CEAs are QALYs and DALYs. Different illnesses are associated with different degrees of severity; for these measures, severity is assessed by a utility weight (71, 73). The utility weights may be based on individuals' preferences for avoiding specific illnesses, or they may be based on expert opinions.

Both Ponce et al. (88) and Cohen et al. (89) examined neurocognitive deficiencies associated with fetal MeHg exposures, assuming that the deficiencies persist throughout the life of the affected individual. The benefit was assessed by multiplying the utility weight (*i.e.*, the decrease in the quality of life that results from cognitive deficits) by the duration of the effect (persisting over the individual's lifetime). For QALYs, this calculation results in the prediction of the quality adjusted life years an affected individual experiences; these are compared with the increased number of quality adjusted life years an individual would experience if a policy to reduce MeHg exposures were put into place. The benefits are expressed as net QALYs gained by the population. For DALYs, the product of the utility weight and duration of the effect results in a prediction of disability-adjusted life years incurred. Population benefits of a policy are then evaluated by the decrease in the number of disability-adjusted life years incurred.

Both QALYS and DALYs are used as the denominators in cost-effectiveness analyses, with the numerator being the cost of the policy. The cost effectiveness of different policies can be compared; those that exhibit the largest gain in QALYs (or decrease in DALYs) for the lowest cost are preferred to those that exhibit smaller gains in QALYs for larger costs.

Benefits assessment in different social contexts

All of the BCAs and CEAs of Hg policies that we found have been conducted in developed countries, with monetary measures and HALYs used to address different economic questions associated with MeHg exposures that result from eating fish. While some of these analyses have attempted to identify subpopulations that benefit from implementation of a policy, most could be improved from additional analyses of the distribution of benefits and costs across the population. Finally, some of these analyses have examined the time frame over which anticipated MeHg exposure decreases and health-consequent benefits might occur. For many, benefits accruing soon after expenses are incurred are preferred to those same benefits accruing in the future (*i.e.*, time value of money). While beyond the scope of this manuscript, both the manner and rate by which future health benefits are discounted is central to many policy analyses and may be particularly important for Hg, given the uncertain environmental response times implied by current research on the biogeochemistry of Hg (3).

In contrast to the number of evaluations in developed countries, we are not aware of a benefits assessment that examines the costs or benefits of reduced Hg releases and MeHg exposures in developing countries, or benefit assessments focused on those engaged in subsistence fishing or small-scale gold mining (Table 4). Such assessments could differ substantially from those in developed regions. Moreover, caution is warranted when applying these economic techniques in situations in subsistence societies where some Hg policies may greatly disrupt the social structure in a society with few alternatives to subsistence fisheries (Table 4; 90, 91), or in regions where small-scale gold mining is prevalent and there is no realistic alternative to using Hg to make a living. Table 4 also demonstrates that economic analyses of the costs and benefits of reducing mercury reductions are lacking for many of the Hg exposure pathways.

Assessments for other effects, including recreational fishing and environmental degradation

Hg contamination may affect the quality of current recreational experiences, decrease future recreational use in this generation and use by future generations, and affect other values that one might hold irrespective of use. In a BCA, values for these impacts may be elicited directly by using stated preference techniques that rely upon choice in hypothetical situations (*e.g.*, choice experiments or contingent valuation) or they may elicit value indirectly by using revealed preference approaches based on the observed choices of people (e.g., the travel cost model). Champ et al. (92) provide an excellent introduction to these methods.

Jakus et al. (83) used observed human behavior to measure changes in commercial and recreational values due to fish and wildlife consumption advisories. For commercial fisheries, the cost of advisories was based on the market demand and supply for contaminated commercial species. Advice suggesting reduced consumption of striped bass means that at least some consumers will consume fewer bass at any given price. This “shift” in commercial demand is used to measure cost in terms of lost market value accruing to consumers and producers. A similar approach was used to measure impacts on recreational angling. Advice advocating restricted consumption of fish or wildlife altered the number of recreational trips or changed the species targeted. The travel cost method was used to calculate the change in the net value of fishing with and without consumption advisories. The method does not account for possible health impacts to those who do not change behavior in response to advisories.

Hagen et al. (69) used a stated preference method, contingent valuation, to value changes in human health as a result of Hg-reduction policies, as well as the effects on recreational anglers and on wildlife. In this study, people were asked to estimate their willingness to pay for a policy designed to reduce environmental Hg levels. Respondents were asked to consider benefits from changes in their health, their family’s health and, possibly, the health of their neighbors and others. As such, this analysis included cultural values, as in the case of a parent who would like his or her children to enjoy eating fish that they have caught as a family, irrespective of the health implications. Values may have been expressed for future generations and broader ecosystem services (so-called “non-use” values), and thus WTP may or may not be tied to one’s direct exposure to Hg.

While all of the above analyses have been conducted in developed countries, both revealed preference and stated preference methods have been used in the developing nations on issues other than Hg. In less-developed regions, the suite of Hg impacts may be similar to those encountered in more-developed countries, but the values that these societies (*e.g.*, subsistence fishing communities) place on these impacts could be very different from the values of those in more-developed countries (Table 4) and should be examined in future studies.

POLICY OPTIONS TO REDUCE Hg POLLUTION

Environmental Hg releases can be reduced by policies that reduce the supply or demand of Hg, implement technological controls (or processes) that reduce mercury releases during the production of goods that result in Hg releases (e.g. gold and electricity), or reduce the quantities of produced goods that result in such releases. In general, policy options used routinely to reduce pollutant emissions from industrial processes include technology requirements, emission performance standards, emission taxes, and cap-and-trade (CAP) approaches. Other policy options such as subsidies and restrictions on the sale and disposal of Hg (and Hg-containing items) could influence Hg releases from small-scale practices such as artisanal gold mining. Application of any of these alternative policies to Hg reduction will have benefits and costs. An economic approach to evaluating different policy options is to balance, at the margin, the benefits and costs of any policy option. Such policies are deemed economically efficient.

In the case of Hg, economic analysis is complicated by the need to track benefits and costs at various geographic scales, from local to global. While the costs associated with the implementation of new processes or control technologies can be estimated in a relatively straightforward manner, the assessment of benefits is complicated by the scientific uncertainties reported in the environmental literature (i.e., the linkages between reducing environmental Hg releases and lower levels of Hg in the atmosphere and in fish; 1, 3) and health science literature (i.e., linkages between reduced levels in the environment, reduced exposures, and health improvement; 3, 11). Ideally, economic analyses highlight these uncertainties as well as those introduced in the benefit-cost component of the analysis and conduct additional analyses to assess the sensitivity of the results to the assumptions associated with the uncertainties (93). Resolution of uncertainty—regardless of the source—will increase the reliability of the economic analyses.

Each of the main policy options has particular strengths and weaknesses. Technology requirements mandate a particular production or control technology, and can have the advantage of offering well-understood pollution reduction. On the other hand, for any specific level of reduction in the release of a given pollutant, economic models suggest that, when compared to other policy options, technology mandates generally have a higher cost than more flexible approaches and may provide less incentive for technological innovation (94, 95). Performance standards offer some flexibility by which firms can reduce costs; for example, firms have an economic incentive to develop less costly control technologies. Ideally, performance standards provide regulators assurance about the level of a pollutant released at each regulated source, an important advantage for pollutants that can have significant local impacts.

Market-based reduction policies can take the form of emission taxes or CAP. Under an emission tax approach, a source may emit any quantity of a pollutant desired but is taxed for each unit released. In CAP, the regulatory authority sets an aggregate emissions level and issues permits (that sum to the target level) to polluters by an auction or simple distribution. Polluters are free to trade permits, with the prevailing price naturally reflecting the incremental cost of control. Economic models suggest that market-based policies spur innovation of new production technologies that are more efficient or less costly than extant technologies (35, 94, 95). If sources of emissions face differing abatement costs, the emissions tax and CAP approaches generally offer lower total costs of control (when summed over all facilities) relative to technology or performance standards. This is because choices about how much to control are made by facility management based on the prices they face rather than by the regulatory authority. A requirement of either the emissions tax or CAP approaches is that emissions be accurately monitored and enforced. For pollutants with local impacts, a key disadvantage of the emissions tax and CAP approaches is the potential for local impacts to persist or increase where facilities deem it uneconomical to reduce pollution. Some have suggested that this problem can be addressed through differential trading rates in a CAP (96), differential taxes, or combining a CAP with a performance standard. Devising a policy to avoid local impacts under a CAP would require considerable knowledge about local factors that control exposure to such a pollutant.

With respect to large point sources of Hg emission to the air, reduction policies have generally relied on performance standards; for example, in the U.S., performance standards were finalized for municipal waste combustors in 1995, for medical waste incinerators in 1997, and for hazardous waste incinerators in 2002 (44). However, in the case of coal-fired electric power plants, in 2005 the U.S. federal government promulgated a CAP (5, 75). At the time of this writing, the regulation of Hg releases from both large and small sources, and strategies to address supply and demand of Hg, is rapidly evolving. This evolution is partly a function of the limits of economic analysis and uncertainties over the environmental fate, exposure, and health effects associated with Hg releases. While an economic approach may assess the relative economic efficiency of any set of potential policies, the choice of any one policy is subject to a variety of factors, only one of which is efficiency. For example, two different policies may be economically efficient, yet reduce very different amounts of pollutant, have different levels of benefits and costs, or have benefits and costs that are distributed quite differently across a population. Strictly speaking, benefit-cost analysis has little to say about a choice between these two policies. Thus, acceptable policies must also satisfy political, social, and cultural criteria.

POLICIES TO LIMIT EXPOSURES TO MERCURY THROUGH RISK COMMUNICATION

In addition to reducing Hg releases, human Hg exposures can also be reduced through risk communication policies, including fish consumption advisories, improved communication of the occupational risks associated with Hg releases during artisanal gold mining, and product labeling. In this section, we focus on consumption advisories and also provide a brief description of the risk communication challenges associated with small-scale gold mining. We note that the impacts of product labeling policies need additional study.

Fish consumption advisories are considered by many policy makers to be an unfortunate and, hopefully, interim public health necessity. In general, advisories are based on an assessment of the human health risks associated with pollutant exposures, including fetal exposures that result from consuming contaminated fish (e.g., 97). The primary policy goal of a fish consumption advisory is to reduce pollutant exposure by reducing the intake of a contaminant (MeHg, in this case), while maintaining recommended fish intake; this is accomplished by recommending consumption of different fish species, smaller fish, or fish from a different fishery (i.e., a different waterbody with less contaminated fish). The risk managers who develop fish consumption advisories consider multiple issues, including what level(s) of fish contamination should trigger the issuance of an advisory, the species of fish involved, and their availability to consumers. Consideration is given to identifying groups or individuals who should follow the advisory (e.g., fishers, women of reproductive age, or those responsible for food preparation). Techniques for communicating with different audiences need to be evaluated. Last, but certainly not least, the languages and concepts used by consumers to convey information should be studied, as well as aspects of local fishing economies, including distribution and sharing practices. The practical implications of advisories on fish consumption have seldom been documented, and most likely vary a great deal depending on the nature of the advice, how it is communicated, and the alternatives available to the community (e.g., 98, 99).

For public health officials, a fundamental tension exists in the development of fish consumption advisories: how to communicate to people that they should avoid eating highly contaminated fish, but, that they should continue to eat fish because of its nutritional value and the associated health benefits. Fish are high in protein and low in saturated fat. Frequent fish consumption has been shown to reduce the risk of heart disease (100). Policy-makers, thus, prefer to consider such advisories as ‘interim’ in the expectation that other measures can be implemented which will effectively reduce the fish tissue MeHg concentrations (3). Knowledge of the impacts of fish consumption advisories on diet is quite limited, and this is an area that requires further research (89).

The contamination of fish by MeHg can be an elusive concept to convey because this toxicant cannot be detected visually in the fish and the fish do not appear to be diseased. Consumers can interpret consumption advisories in various ways; accurate communication of an advisory can be especially difficult when there are linguistic or cultural differences between the risk manager and the affected population (99). Since individuals who communicate advisories may differ in their views of the risks involved and their relation to the nutritional role of fish (e.g. 101), conflicting advice may be received by a fishing community. Those who develop advisories, therefore, require appropriate knowledge of the nature of the fishery itself, as well as fish sharing, preparation, and consumption rates and practices (97, 102, 103).

The use of fish consumption advisories presents additional challenges for remote and isolated communities dependent on subsistence fisheries, including some North American indigenous populations (99). In some subsistence societies, advisories have resulted in complete avoidance rather than reduced consumption of the most contaminated fish (104). The loss or substantial disruption of local fisheries can also have significant public health implications, as well as related cultural impacts (105). Alternative dietary choices available in larger population centers may simply not be available to subsistence populations, and the communities may already face significant diet-related public health problems (such as diabetes). In such settings, health service providers face the task of balancing the message of the consumption advisories against the public health consequences of significant changes in human nutrition (106).

In 2004, the US EPA and FDA released a joint statement (107) that described fish and shellfish as important components of a healthy diet. It noted that “a well-balanced diet that includes a variety of fish and shellfish can contribute to heart health and children's proper growth and development”, although “nearly all fish and shellfish contain traces of Hg....”. Finally, the statement advised women who may become pregnant, pregnant women, nursing mothers, and young children to avoid some types of fish (i.e., shark, swordfish and king mackerel) due to high MeHg levels and to eat up to 2 meals each week of fish or shellfish that are low in MeHg (e.g., shrimp, canned light tuna, salmon, pollock, and catfish). The joint statement also cautioned consumers to check local advisories about the safety of consuming fish caught in local waterbodies, noting that if no advice was available, to eat up to one fish meal of average size per week, but not to consume any other fish during that week. Several investigators have compared the risks associated with MeHg exposures and the nutritional value and health benefits of fish consumption (11, 88, 89, 108). Their findings generally are consistent with the EPA and FDA statement.

In the case of small-scale gold mining using Hg amalgamation, the primary toxicological issue is the inhalation of Hg converted to the gas phase during the heating of the amalgam. Heating often takes place inside or near the home. Artisanal workers and their families can be exposed to harmful levels of Hg vapor. Risk communication in the form of advice to avoid the Hg amalgamation technique or to reduce exposure during its use, must take into account the limited options available to the gold miners and the widespread poverty and hardship associated with this occupation. Field researchers (e.g. 49, 53) emphasize that effective risk communication strategies need to be intertwined with strategies targeting improved profitability through better gold recovery methods or reduced losses of Hg, thus reducing the artisinal miner’s production costs. Within each country, the industry is geographically scattered, so the logistical aspects of risk communication are a major challenge. Thus, to be effective, in each region, risk communication strategies may involve training a cadre small scale gold miners who can demonstrate and discuss the advantages of improved practices to their fellow miners (53).

CONCLUSIONS

Mercury is a naturally occurring element that can affect the health of humans and wildlife. Humans have historically found Hg to be a useful liquid metal for a variety of social and economic purposes, and currently release some 2400 t yr^{-1} of Hg into the atmosphere. The best estimate is that almost 90% of anthropogenic emissions comes from the combustion of fossil fuels in electrical power generation and from the release of Hg in large- and small-scale mining operations, although emissions associated with intentional Hg use are difficult to quantify and may be substantially underestimated. Some Hg emissions to the atmosphere are deposited relatively near the source, but a large portion enters the global atmospheric reservoir.

The most important pathways of human exposure to Hg are through the consumption of MeHg-contaminated fish and inhalation of Hg vapor. Worldwide, the greatest source of inhalation exposure likely is from small-scale gold mining. The release of Hg used in restorative dentistry can be a source of exposure, but the associated risks are not well understood. Exposure to environmental Hg is believed to have a number of potential negative effects on human health, including these: (i) cognitive deficits (e.g., reduced IQ) in children due to fetal exposure and in adults exposed to high concentrations of Hg vapors; and possibly (ii) increases in fatal and non-fatal heart attacks; and (iii) increases in premature death (i.e., some studies link Hg exposures to increased risk of premature mortality regardless of cause).

While the general pathways outlined in this paper are accepted, a great deal of uncertainty remains in the linkages between emissions and human health. Resolution of these uncertainties is important for analysis of Hg reduction policies, but will be difficult to achieve. The key physical and health science questions include the magnitude of the environmental response, the environmental response time (*i.e.*, the length of time between decreased Hg emissions and consequent changes in human health risks), to what degree decreased exposures will reduce the risks of cognitive deficits, heart attack, or premature death and, if so, in which populations. While some exposure pathways are better understood than others, the full extent of health damages in many of these pathways is not known. For example, while fetal exposures are relatively well studied (although uncertainties remain), the dose-response functions for health effects resulting from chronic exposure in adults, such as cardiovascular disease, remain uncertain.

Uncertainty with respect to health effects has implications for choices among many policy alternatives. Both of the primary approaches to economic evaluation, BCA and CEA, require that one be able to link changes in Hg emissions to changes in various health outcomes; uncertainty with respect to policy benefits implies uncertainty in the benefit-cost or cost-effectiveness metrics coming out of the analysis. The few economic studies of Hg policies all express concern about the degree of uncertainties in the physical and health science of Hg exposure, suggesting that a valuation approach that explicitly deals with uncertainty and ambiguity is warranted. Three other aspects of the economic and social analysis are also important for future research. First, all of the economic studies have been conducted in the context of a developed economy, but key aspects of Hg reduction policies affect people living in developing countries or regions (e.g., small-scale mining operations or subsistence fisheries). Second, little research has been done to quantify non-health-related benefits, such as non-use values associated with benefits to future generations. Third, costs (both economic and social) associated with structural changes in local food production systems such as subsistence fishing are not well understood and, therefore, not well quantified in cost-benefit analyses. Given uncertainty with respect to benefits and costs, it is difficult to conclude *a priori* that any one Hg-reduction policy is “better” than any other from an economic point of view, or even if the benefits of such a policy exceed the costs.

Given the variety of sources and reduction options, multiple approaches that reduce Hg releases or exposures are being proposed and considered at multiple levels in governments around the world. Economic and comparative risk analyses need to be conducted to examine these approaches. Because Hg emissions in one part of the planet are transported globally, the efficiencies and risks associated with these approaches need to be evaluated in light of the local, regional, and global impacts. Likewise, international trade also moves elemental Hg globally and is linked to emissions and exposure; better data on country-to-country and in-country commercial flows of Hg (indeed, most countries have a poor understanding of their domestic use of Hg) will contribute to refining existing policies and developing new ones. Decision makers also need to analyze potential interrelationships between policies, so that one policy does not reduce the efficiency of another. To the extent possible, effective and acceptable policies must satisfy political, social, and cultural criteria, as well as economic criteria.

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Table 1. Important global compartments of mercury in commerce and the environment, as diagrammed in Fig. 2.

Code	Mnemonic	Definition
A	Aquatic system	Hg in wetlands, lakes, rivers, and oceans. Hg introduced to aquatic systems may become MeHg, which may be bioaccumulated by **F**ish.
C	Coal and other fossil fuel combustion	Hg mobilized by the processing and combustion of the fossil fuels coal, oil, and natural gas ($\mathbf{X_C}$).
D	Disposal	Hg in discarded **P**roducts or process wastes from chlor-alkali or VCM plants.
F	Fish	Hg in fish, virtually all of which is in the form of MeHg, which is produced by natural bacteria in **A**quatic systems.
H	Humans	Hg absorbed by humans following exposure, generally through **F**ish consumption or inhalation of **V**apor.
L	Land	Hg in soil, mostly derived from atmospheric deposition of **V**apor, but can be elevated from mine waste, Hg waste disposal, or geologically rare mineral deposits containing Hg.
M	Manufacturing	Hg used in the manufacture of Hg-containing **P**roducts, or in processes that use Hg to make Hg-free products (e.g., chlor-alkali and vinyl chloride monomer processes).
O	Ore refining	Hg mobilized by the processing and refining of nonfuel mineral resources $\mathbf{X_O}$**.**
P	Products	Hg contained in products, including thermometers, switches, fluorescent lamps, batteries, fungicides, preservatives, seed-coatings, pharmaceuticals, etc.
R	Recycling	Hg that is extracted from discarded products or wastes, purified, and put into commerce or retired.
S	Small-scale gold mining	Hg utilized by independent, artisanal, miners to concentrate geological gold through amalgamation.
V	Vapor	Hg vapor in indoor and outdoor air.
W	Wildlife	Hg absorbed by fish-eating wildlife, such as seal, whale, otter, mink, osprey, eagle, kingfisher, and loon.
X	Out of the biosphere	Hg in the "X" compartments are not part of the Hg cycling in the biosphere and therefore do not harm humans or wildlife. "X" Hg may be mobilized at some point in the future, but for practical purposes is permanently stored unless humans intervene.
$\mathbf{X_B}$	Buried	Hg, formerly in the biosphere, that has been buried in the sediments of oceans, lakes, and river deltas.
$\mathbf{X_C}$	Coal and other fossil fuel deposits	Hg in buried fossil fuel deposits such as coal, oil, and gas, that may be extracted and burned.
$\mathbf{X_G}$	Geological	Hg in geological materials that release Hg vapor to the atmosphere through natural processes.
$\mathbf{X_O}$	Ores	Hg in non-fuel geological resources that are subject to mining and refining, including minerals containing Hg, gold, zinc, nickel, tin, copper, silver, lead, and iron. All geological materials contain some Hg, even limestone that is heated to make lime.
$\mathbf{X_T}$	retiremenT	Hg permanently stored, or "retired" by humans in warehouses, engineered landfills, or deep bedrock repositories.

Table 2. Important global pathways of mercury in commerce and the environment (diagrammed in Fig. 2). An asterisk (*) on the pathway code indicates that the path is amenable to manipulation by society's policies. The estimates of annual anthropogenic flow are constrained by setting total anthropogenic emissions at 2400 t yr^{-1} (25).

Code	From	To	Comments	Annual Flow (tonnes)	Information Source
AF	Aquatic system	Fish	Only a portion of Hg entering an aquatic system bioaccumulates in fish, partly because only a portion is converted to MeHg.		
AV	Aquatic Systems	Vapor	The oceans are estimated to re-emit over 80% of atmospheric deposition; there is little binding capacity in ocean water, and photochemistry produces Hg^0, which has low solubility in water.	2600	25
AX_B	Aquatic system	Sediment burial	Some of the Hg entering an aquatic system associates with particles that settle to the bottom and is buried permanently by new sediment. Mason and Sheu conclude that Hg is building up in deep ocean water and only a small proportion is buried.	200	25
CV*	Fossil fuel combustion	Vapor	Emitted Hg vapor may be elemental or divalent, which greatly affects distance traveled before deposition to earth.	1500	28
CX_T*	Fossil fuel combustion	Retirement	Hg vaporized during combustion may associate with ash or can be caught separately. Coal ash is often used in construction, which may not retire the associated Hg as permanently as landfilling.	700	Estimate
DA*	Disposal	Aquatic system	Some Hg is discharged directly into surface water, or indirectly through treatment plants. In treatment plants, most Hg associates with solids, which if not discharged are incinerated or land-applied.		
DR*	Disposal	Recycling	This category includes Hg from closed chlor-alkali plants, which is saleable as is. The Hg in product waste can be purified at relatively large costs per kg Hg (100 to 1000 US$ per kg), which is then sold for less than $20 per kg. Retirement of recycled Hg may be a better economic choice for societies.	1200	Estimate
DV*	Disposal	Vapor	Disposal to the solid waste stream may include incineration, which vaporizes all Hg, of which some may be caught by pollution control equipment and landfilled.	110	28
DX_T*	Disposal	Retirement	Manufacturing waste Hg may be retired in landfills as sludge. Hg-products might be landfilled after breakage and spilling of Hg.		
FH*	Fish	Humans	Humans absorb most of the methylmercury (MeHg) consumed in a fish meal. The MeHg is in the protein, not the fat, of fish.	25 (as MeHg)	Estimate, assuming 0.2 ppm
FW	Fish	Wildlife	Fish-eating wildlife are particularly vulnerable to elevated Hg in fish.		

LA*	Land	Aquatic system	Of the Hg deposited from the atmosphere to land, a variable proportion (approximately 5 to 20%) is delivered to lakes and rivers draining the land. Human alteration of the landscape (e.g., agriculture and urban development) can affect the transport to aquatic systems.	200	25
LV*	Land	Vapor	The land re-emits about half of the Hg deposited from the atmosphere. Human alteration of the landscape (e.g. climate change, fire, agriculture) can change the rate of re-emission to the atmosphere.	1600	25
MD*	Manufacturing	Disposal	Any manufacturing process that employs Hg will have Hg waste.	830	Estimate
MH*	Manufacturing	Human vapor exposure	Any manufacturing process that employs Hg will produce Hg vapor that potentially exposes the workers.		
MP*	Manufacturing	Products	Products containing Hg are still manufactured even though cost-effective Hg-free substitutes exist for almost all uses, probably because of economic and technological inertia.	1070	Estimate
MV*	Manufacturing	Vapor	Manufacturing both makes Hg-containing products and uses Hg in processes that emit Hg (chlor-alkali and VCM).	120	Estimate
OA*	Ore refining	Aquatic systems	The processing of Hg-containing ores sometimes produce wastes or tailings that enrich aquatic systems with Hg.		
OM*	Ore refining	Manufacturing	Almost all Hg mines in the world are now closed, as the world's Hg demand is now met by by-product Hg from non-Hg ores, recycling, and the closure of chlor-alkali plants.	1600	Estimate
OS*	Ore refining	Small-scale gold mining	By-product Hg is one source for gold mining.	500	Estimate
OV*	Ore refining	Vapor	Heating ores will vaporize any Hg present, either emitting it to the atmosphere or incidentally catching it with air pollution control devices, unless special Hg control efforts are made.	330	28
PD*	Products	Disposal	Hg-containing products will eventually reach the end of their useful life and will either break or be disposed of.	1020	Estimate
PH	Products	Human vapor exposure	When Hg-containing products break and spill, they create the risk of human exposure and time-consuming clean ups.		
PV	Products	Vapor	Hg spills from broken products contribute to the atmospheric burden of Hg.	40	Estimate
RM	Recycling	Manufacturing	Recycled Hg can be used by many manufacturers.	700	Estimate
RS*	Recycling	Small-scale gold mining	Hg used in mining does not have to be of a high purity, so recycled Hg, especially from chlor-alkali plants, can be used.	500	Estimate
RX_T	Recycling	Retirement	Hg need not be purified to be retired, although there may be some advantages in handling and containment. 4400 t retired in 2006 (62)		

SA*	Small-scale gold mining	Aquatic systems	Hg may be used directly in flowing water to concentrate gold, contaminating water and its sediments.	700	Estimate
SH*	Small-scale gold mining	Human vapor exposure	The gold-Hg amalgam is heated to concentrate the gold, exposing miners and their families to Hg vapor.		
SV*	Small-scale gold mining	Vapor	Hg vaporized during heating of Hg-gold amalgam adds significantly to the global atmospheric burden of Hg.	300	29
VA*	Vapor	Aquatic Systems	All atmospheric Hg eventually deposits to earth, with oceans receiving almost half.	3100	25
VL*	Vapor	Land	Hg deposition rates to the continents has increased by about a factor of 3 over pre-industrial rates.	3500	25
X_CC*	Fossil Fuel Deposits	Combustion	Although the Hg concentration in fossil fuel is usually low, much is burned, and all Hg vaporizes during burning, but some binds to particles and is captured. Pre-treatment of fuel has the potential to remove Hg prior to combustion.	3000	Estimate
X_CX_T *	Fossil Fuel Deposits	Retirement	Some of the Hg in fossil fuel is separated from fuel prior to combustion through coal cleaning or natural gas treatment. Oil refineries are poorly understood where the fate of Hg is concerned.	700	Estimate
X_GV	Geology	Vapor	Hg is naturally released from geological deposits, naturally enriched soils, and volcanoes. Lindberg et al. (1) point out that this path is poorly known and may be greater than 1500 t yr^{-1}.	100	25
X_OO	Ore resources	Ore refining	Hg is in relatively high concentrations in sulfide ores of gold, silver, copper, lead, and zinc, and in lower concentrations in non-sulfide ores. Heat and other processes release the Hg, which enters the biosphere unless efforts are made to capture it.	>2500	Estimate

Table 3. Summary of economic analyses that have been performed on the costs or benefits of reducing mercury emissions or just reducing exposure through fish consumption advisories (e.g., 83). All of the studies concern the United States.

Study	**Scenario**	**Health Endpoints or Other Endpoints**	**Benefits Measurement Tools**	**Costs or Benefits in 2004 $US (Entire U.S. unless noted)**
EPRI (77)	Utility sector cap of 15 tons by 2018 or MACT by 2008 (about 24 tons emitted).	IQ change in fraction of population above MeHg RfD.	Benefits not monetized.	Cost of cap: $6 thousand million. Cost of MACT: $19.3 thousand million.
Gayer and Hahn (78)	Utility sector cap of 15 tons by 2020 or MACT by 2008.	IQ	Parental willingness to pay for IQ increases through chelation therapy.	Cost of cap: $3.4 - $5.5 thousand million Benefits of cap: 60 – 150 million Cost of MACT: $15.4 – 20.7 thousand million Benefit of MACT: $82 - $142 million.
Hagen et al. (69)	50% reduction from all sources in Minnesota (U.S.).	Unspecified health effects, recreational fishing, effects on wildlife	CVM	Benefits to Minnesota residents: $255 million per year (1998 Minnesota population = 4.7 million).
Jakus et al. (83)	Issue mercury-related advisories on the Maryland portion of Chesapeake Bay (U.S.)	IQ, AMI, ACM Recreational fishing Commercial fishing	COI VSL TCM	Benefits: Avoided illness $15.4 million per year for consumers of Maryland-Chesapeake Bay fish. Lost Recreation/Commercial value: $9.1 million
Lutter et al. (84)	60%-90% reduction in power plant emissions (U.S.)	Neurological deficiency	Benefits not monetized.	Total cost: $1.2 - $1.9 thousand million $120 000 - $190 000 per case averted
Palmer et al. (79)	Hg cap under CAIR for power plant emissions (U.S.)	IQ, AMI, ACM	COI VSL	Cost: $3.4 thousand million Benefit: Same as Rice and Hammitt
Rae and Graham (85)	30%, 51%, and 100% reductions in power plant emissions (Southeastern U.S.)	IQ, Non-fatal AMI Hypertension ACM	COI VSL	Benefit: $619 – 2 102 million per year for four states in SE United States, only.
Rice & Hammitt (80)	Cap of 26 tons and 15 tons **later ?** on power plant emissions (U.S.)	IQ, alternatively assume with and without a threshold, non-fatal AMI, ACM	COI VSL	Benefit: $3.8 - $5.7 thousand million
Trasande et al. (81)	Evaluate costs of current emissions from all sources	IQ	COI	$9.5 thousand million per birth cohort $1.4 thousand million due to power plants
USEPA (76)	Cap of 38 tons in 2010, 15 tons in 2018.	IQ	COI	Benefits: $0.25 to $1.56 million
US EPA (86)	Hg cap under CAIR and CAMR for power plant emissions (U.S.)	IQ	COI	Costs: $750 million per year by 2020. Benefits: Less than $168 million per year.

Abbreviations: ACM=All Cause Mortality AMI=Acute Myocardial Infarction COI=Cost of Illness CVM=Contingent Valuation Method IQ=Intelligence Quotient change MACT=Maximum Achievable Control Technology TCM=Travel cost Method VSL=Value of a Statistical Life

Table 4. Mercury exposure pathways and the extent of knowledge pertinent to cost-benefit analysis of reduction options. Benefit and cost methods that have been applied are in bold (references in parentheses); other possible approaches are listed. In many cases a necessary prerequisite is to quantify any connection between Hg releases and any endpoint.

Exposure Pathway	**Measurement Tools (for Benefits of Reducing Exposure)**	**Source of Elevated Hg**	**Hg Exposure Reduction Options**	**Measurement Tools (for Costs of Reducing Exposure)**
FH—Commercial Fishing	**COI VSL** (76, 78, 80, 81, 83, 85, 86)	VA—atmospheric deposition	a. Reduce Hg emissions from coal, ore refining, & small scale gold mining via: i. Incentives to develop control technology. ii. Incentives to transfer control technology globally. b. Purchase and consume low-Hg fish c. Consume non-fish source of protein	a. **COT** (77) i. ii. b. **LMV (83)** c. COS
FH—Recreational Fishing	**COI VSL (83)**	VA—atmospheric deposition	a. Reduce Hg emissions b. Consume low-Hg fish, release high-Hg fish. c. Fish low-Hg waters (e.g. different lake)	a. **COT (78)** b. **LRV (83)** c. **LRV (83)**
FH—Subsistence Fishing	COI, VSL	VA—atmospheric deposition	a. Reduce Hg emissions b. Reduce fishing & find alternate source of protein. c. Disseminate culturally appropriate fish consumption advice. d. Consume low-Hg fish, sell high-Hg fish.	a. COT b. QOL c. QOL d. QOL
FH—Subsistence Fishing	COI VSL	DA—Disposal of Products and Wastes to Aquatic Systems	a. Reduce Hg discharge to fishery b. Reduce fishing & find alternate source of protein c. Create incentives for switching to Hg-free products & processes for chlor-alkali and vinyl chloride monomer plants. d. Disseminate culturally appropriate consumption advice. e. Consume low-Hg fish, sell high-Hg fish.	a. COT b. QOL c. COS d. QOL e. QOL
FH—Subsistence Fishing	COI VSL	OA—Ore Refining discharges	a. Reduce Hg discharge b. Reduce fishing & find alternate source of protein. d. Disseminate culturally appropriate fish consumption advice. e. Consume low-Hg fish, sell high-Hg fish.	a. COT b. QOL c. QOL d. QOL
FH—Subsistence Fishing and Recreational Fishing	COI VSL	Reservoir creation or operation	a. Evaluate impacts before reservoir creation. b. Create and operate to minimize Hg in fish. c. Reduce fishing & find alternate source of protein. d. Disseminate culturally appropriate fish consumption advice. e. Consume low-Hg fish, sell high-Hg fish. f. Consume low-Hg fish, release high-Hg fish.	a. COT b. COT COS c. QOL d. COT e. QOL f. **LRV (83)**
FH—Subsistence Fishing	COI VSL	SA—Small-Scale Gold Mining discharges	a. Provide incentives for efficient or reduced Hg use through: i. Reduce international supply of Hg through retirement, etc. ii. Transfer of culturally acceptable technology. iii. Build community capacity to reduce Hg-related problems. b. Economic development for opportunities other than gold mining. c. Reduce fishing & find alternate source of protein.	a. COS COT i. **COHR (62)** ii. COT iii. COT b. COED c. COS QOL

FW—Wildlife consumption of fish	**SPM (69)**	VA—atmospheric deposition	a. Reduce Hg emissions.	COT
FW—Wildlife consumption of fish	SPM	Reservoir creation or operation	a. Evaluate impacts before reservoir creation. b. Operate to minimize Hg in fish.	a. COT b. COT COS
FW—Wildlife consumption of fish	SPM	OA—Ore Refining	Reduce Hg discharge	COT
FW—Wildlife consumption of fish	SPM	DA—Disposal of Products, Wastes to Aquatic Systems	Reduce Hg discharge to fishery	COT
FW—Wildlife consumption of fish	SPM	SA—Small-Scale Gold Mining	a. Provide incentives for efficient or reduced Hg use through: i. Reduce international supply of Hg through retirement, etc. ii. Transfer of culturally acceptable technology. iii. Build community capacity to reduce Hg-related problems. b. Economic development for opportunities other than gold mining.	a. COS COT i. COHR ii. COT iii. COT b. COED
SH—Inhalation exposure	COI, VSL	SV—inhalation while concentrating gold	a. Provide incentives for efficient or reduced Hg use through: i. Reduce international supply of Hg through retirement, etc. ii. Transfer of culturally acceptable technology. iii. Build community capacity to reduce Hg-related problems. b. Economic development for opportunities other than gold mining.	a. COS COT i. COHR ii. COT iii. COT b. COED
MH—Inhalation exposure	COI, VSL COI, VSL	MV—inhalation while manufacturing with Hg	a. Provide incentives for Hg-free processes. b. Develop inexpensive Hg vapor monitoring.	COS COT
PH—Inhalation exposure	COI	PV—Hg vapor from Hg-containing product use and breakage.	a. Choose Hg-free products (e.g. dental fillings, thermometers, medical devices, pharmaceuticals, barometers, paints, thermostats, lamps, etc.) b. Create incentives for switching to Hg-free product production.	a. COS b. COS

Abbreviations:

COED=Cost of Economic Development. COI=Cost of Illness (including both MeHg and loss of protein and PUFA) COS=Cost of Substitute
COHR=Cost of Hg Retirement COT=Cost of Technology, including the costs of research, and development and dissemination of fish consumption advice.
LMV=Lost Market Value LRV=Lost Recreational Value PUFA=Polyunsaturated Fatty Acids QOL=Quality of Life SPM=Stated Preference Method TCM=Travel Cost Method VSL=Value of a Statistical Life (including both MeHg and loss of protein and PUFA)

Figure Legends

Figure 1. a. Historical mercury production and consumption. Consumption has exceeded production since about 1990, with demand met by large supplies from government stockpiles and closed chlor-alkali plants, plus contributions from recycling (7). b. Mercury and gold prices in the U.S. from 1900 to 2005, adjusted to constant 2005 U.S. dollars. The correlation between gold and mercury prices begins after 1971 with the breakdown of a fixed exchange rate system (7, 8, 9).

Figure 2. Important global pathways of mercury in commerce and the environment. Width of pathway line is proportional to the flow per year. See Table 1 for explanation of abbreviations and the text for discussion of the compartments and Table 2 for discussion of the pathways.

Figure 3. Estimates of global mercury consumption for 2004 (9, 36)

Figure 1.

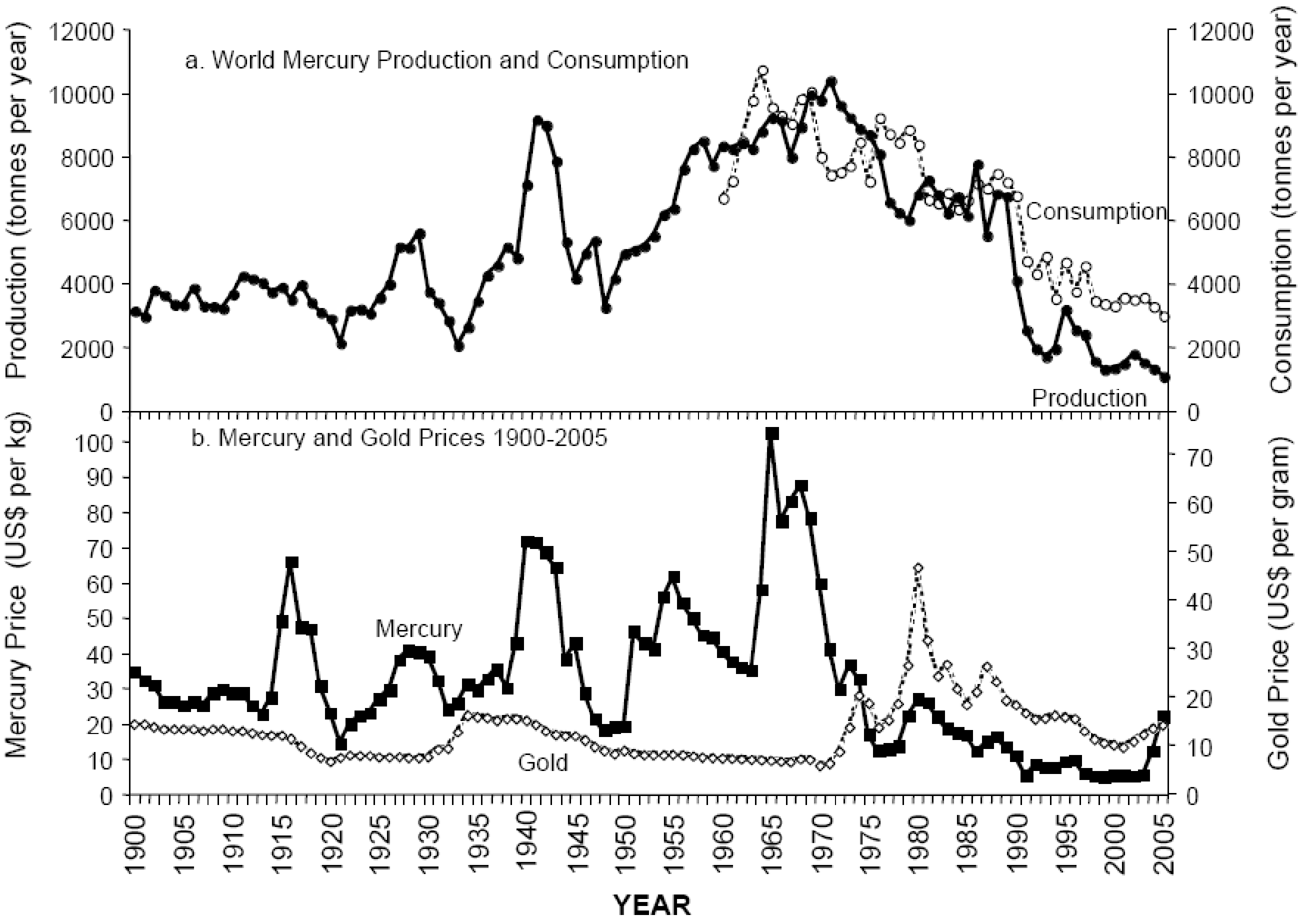

a. World Mercury Production and Consumption
Production (tonnes per year)
Consumption (tonnes per year)
12000
10000
8000
6000
4000
2000
0
Consumption
Production
b. Mercury and Gold Prices 1900-2005
Mercury Price (US$ per kg)
Gold Price (US$ per gram)
100
90
80
70
60
50
40
30
20
10
0
Mercury
Gold
1900
1905
1910
1915
1920
1925
1930
1935
1940
1945
1950
1955
1960
1965
1970
1975
1980
1985
1990
1995
2000
2005
YEAR

Figure 2.

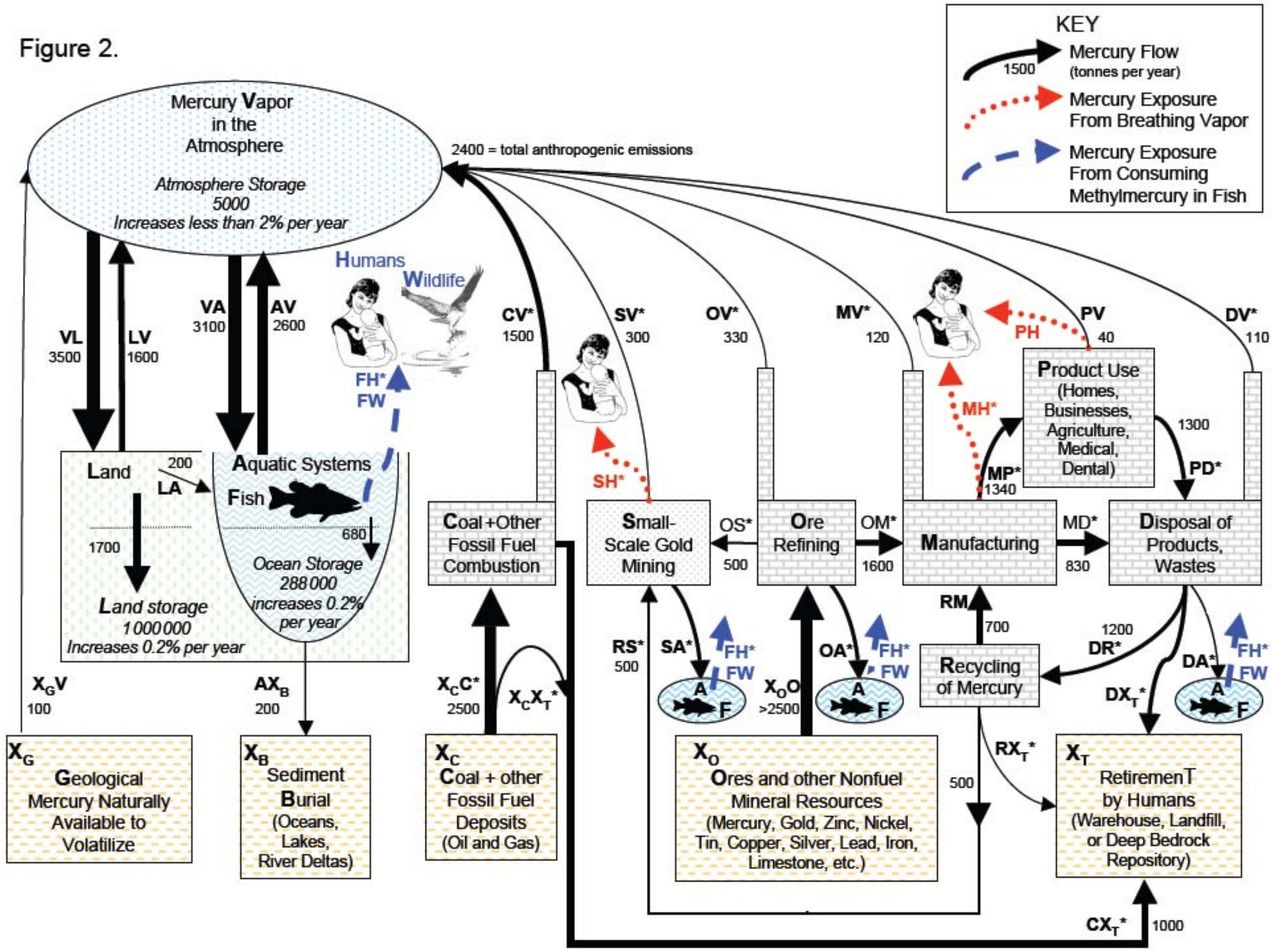
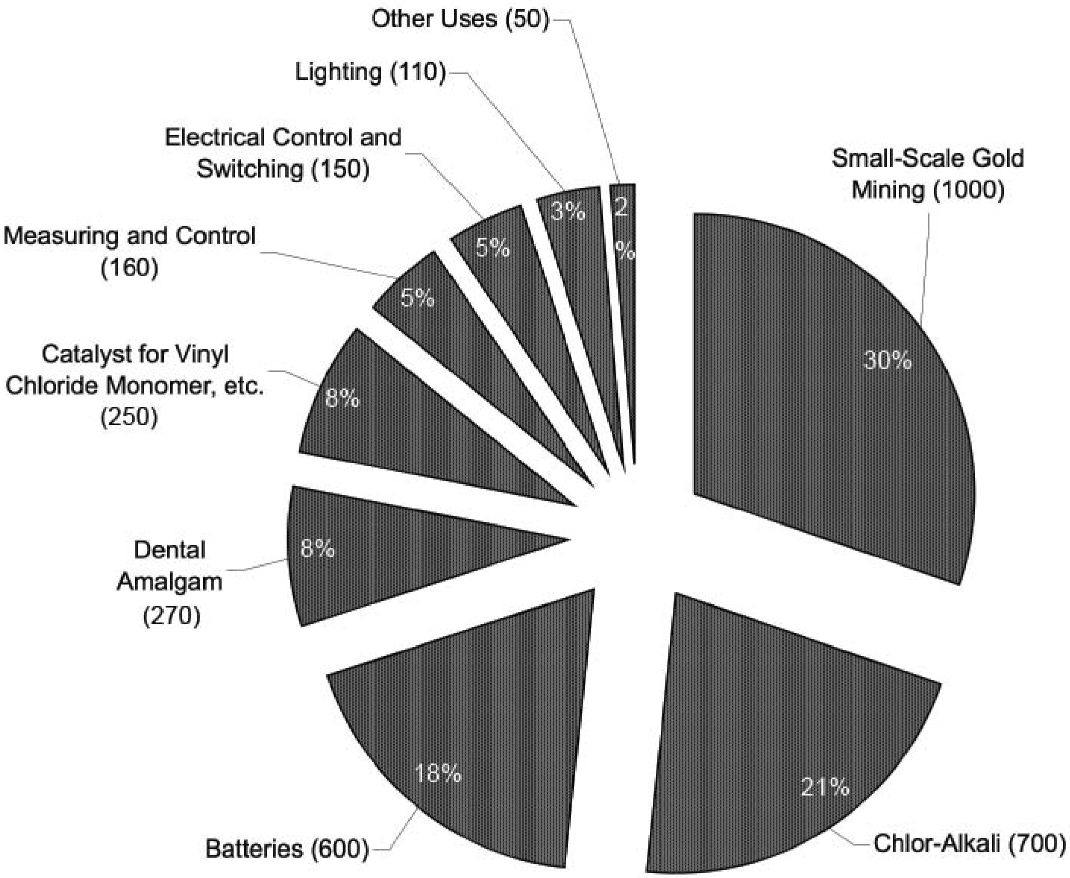

KEY
1500
Mercury Flow (tonnes per year)
Mercury Exposure From Breathing Vapor
Mercury Exposure From Consuming Methylmercury in Fish
Mercury Vapor in the Atmosphere
Atmosphere Storage 5000
Increases less than 2% per year
2400 = total anthropogenic emissions
VL 3500
LV 1600
VA 3100
AV 2600
Humans
Wildlife
FH* FW
CV* 1500
SV* 300
OV* 330
MV* 120
PV 40
DV* 110
PH
MH*
SH*
Land
200
LA
1700
Land storage 1 000 000
Increases 0.2% per year
Aquatic Systems
Fish
680
Ocean Storage 288 000
increases 0.2% per year
Coal +Other Fossil Fuel Combustion
Small-Scale Gold Mining
OS* 500
Ore Refining
OM* 1600
Manufacturing
MD* 830
Disposal of Products, Wastes
Product Use (Homes, Businesses, Agriculture, Medical, Dental)
MP* 1340
1300 PD*
RM 700
Recycling of Mercury
1200 DR*
DA*
DX_T*
RS* 500
SA*
OA*
A F
FH* FW
X_GV 100
AX_B 200
X_CC* 2500
X_CX_T*
X_OO >2500
RX_T*
500
X_G Geological Mercury Naturally Available to Volatilize
X_B Sediment Burial (Oceans, Lakes, River Deltas)
X_C Coal + other Fossil Fuel Deposits (Oil and Gas)
X_O Ores and other Nonfuel Mineral Resources (Mercury, Gold, Zinc, Nickel, Tin, Copper, Silver, Lead, Iron, Limestone, etc.)
X_T RetiremenT by Humans (Warehouse, Landfill, or Deep Bedrock Repository)
CX_T* 1000

Figure 3.


Other Uses (50)
2%
Lighting (110)
3%
Electrical Control and Switching (150)
5%
Measuring and Control (160)
5%
Catalyst for Vinyl Chloride Monomer, etc. (250)
8%
Dental Amalgam (270)
8%
Batteries (600)
18%
Chlor-Alkali (700)
21%
Small-Scale Gold Mining (1000)
30%