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Environmental costs of mercury pollution

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Abstract

Mercury (Hg) has been used for millennia in many applications, primarily in artisanal mining and as an electrode in the chlor–alkali industry. It is anthropogenically emitted as a pollutant from coal fired power plants and naturally emitted, primarily from volcanoes. Its unique chemical characteristics enable global atmospheric transport and it is deposited after various processes, ultimately ending up in one of its final sinks, such as incorporated into deep sediment or bioaccumulated, primarily in the marine environment. All forms of Hg have been established as toxic, and there have been no noted biological benefits from the metal. Throughout time, there have been notable incidents of Hg intoxication documented, and the negative health effects have been documented to those chronically or acutely exposed. Today, exposure to Hg is largely diet or occupationally dependent, however, many are exposed to Hg from their amalgam fillings. This paper puts a tentative monetary value on Hg polluted food sources in the Arctic, where local, significant pollution sources are limited, and relates this to costs for strategies avoiding Hg pollution and to remediation costs of contaminated sites in Sweden and Japan. The case studies are compiled to help policy makers and the public to evaluate whether the benefits to the global environment from banning Hg and limiting its initial emission outweigh the benefits from its continued use or lack of control of Hg emissions. The cases we studied are relevant for point pollution sources globally and their remediation costs ranged between 2500 and 1.1 million US\$ kg⁻¹ Hg isolated from the biosphere. Therefore, regulations discontinuing mercury uses combined with extensive flue gas cleaning for all power plants and waste incinerators is cost effective.

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1. Introduction

Since ancient times, man has been aware of the toxic nature of elemental mercury (Hg) and since, its salts and organic compounds. Most are familiar with Hg intoxication, having heard of “mad hatters”, hat makers who used fur treated with mercuric nitrate. In industrialized nations, health concern was, until a few decades ago, focused on occupational exposure, e.g. in mines, even though there were several documented incidents of local Hg intoxication within the past 50 years and evidence of wildlife mortality in the 1960s. This led the Swedes and Finns

to initiate studies on the risks of using Hg, where knowledge was acquired about environmental transformations of inorganic Hg to far more toxic organic forms along, initially, largely unknown transfer pathways with emissions from the large quantities of Hg used in paper and chlor–alkali plants. (Hylander and Meili, 2005 and references therein).

In general, the state of Hg research is relatively immature compared with other trace metals, especially lead. Elemental Hg is unique when compared to other trace metals found in the atmosphere, in that it is approximately 95% in the gaseous elemental form (Slemr et al., 1985; Schroeder and Munthe, 1998), where other metals, e.g. lead, are primarily associated in the atmosphere as aerosols. The characteristics of gaseous metallic Hg, such as low aqueous solubility, mean that it has relatively low reactivity and is stable. Therefore, gaseous elemental mercury has a long atmospheric residence time, enabling global transport. Its vapor pressure allows it to be

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deposited and re-emitted, as does bacteriologic conversions and subsequent emission. All of these factors contribute to its spread throughout the globe to areas where there are very little natural or local man-made inputs, such as the Arctic. This global transport means that all people and environments will pay some type of cost as a Hg receptor, some more than others, due to enhanced deposition, as is the case in the Arctic. Characterizing this cost is a challenge that needs to be met to properly enact legislation regulating production, consumption and emission.

Although illegal pollution occurs, especially in countries with weak administrative and executive capacity and by persons with few moral restrictions, most pollution is legal in the aspect that emitters are generally meeting the national, legal requirements with respect to emission-control. However, the legal requirements are often far from sufficient to safeguard clean production processes and environmentally benign products. One of the reasons the legal requirements are lacking is that damages to the environment are often difficult to quantify, and even more difficult to assign a monetary value. In addition, there is a time lag between emission and effects. As a result, the environment has become a sink for Hg pollution. This is costly to society and causes a variety of costs, so-called externalities, not included in the prices the producers and consumers pay for products and services. Examples are reduced recreational value of polluted areas, culture losses, reduced biodiversity, fish and other food and feed resources not suitable for consumption or become extinct, and health effects from Hg entering the body via contaminated food, soil, water, and air.

Views on the issue of mercury are often politically charged, partially due to the costs involved in emission reduction. The views range from the far right, disregarding the need to reduce Hg emissions (e.g. Kava et al., 2004) to the far left, claiming that Hg must be banned immediately without any exception. In an effort to give policy makers and the public an objective, environmental, economic review of costs associated with mercury emission control and remediation, we have compiled data from several case studies. Our objectives are to identify and quantify environmental costs of Hg pollution and where possible, assign monetary values on Hg pollution. The aim is to present data possible to use in a global perspective by quantifying remediation costs for different case studies and compare these costs with preventive measures and potential effects in the Arctic, having hardly any anthropogenic Hg emission sources. The goal is to provide input to lawmakers and citizens with respect to Hg and its regulation and control. Since Hg has no redeeming biological use, all anthropogenic use and release is considered contaminant.

2. Material and methods

Environmental costs can in a general sense be defined as costs to the environment and health not included in the prices the producers and consumers pay for products and services. The costs may be divided into two types of categories: i) damage (or value-loss) costs, such as contaminated fish, and ii) policy costs incurred in responding to pollution damage plus the costs of changing practices to meet legal obligations (Pretty et al., 2003).

Policy costs needed in responding to pollution damage but still not paid, because damage has still not been remediated, are occasionally termed environmental debt (Jernelöv, 1992). Total environmental costs comprise a wide range of costs, estimated according to different methodologies. Our aim is not to evaluate different methods, but to focus on certain costs caused by Hg pollution, which will be described briefly together with methods for cost calculation used.

Damage costs of commercially exploited resources such as fishery and organized tourism can be based on damage occurred and the actual market value. Damages to ecosystem services, public health, etc. are more difficult to value, partly because we know too little about the importance of ecosystem services and health effects from Hg, alone as well as in combination with other substances. Often, these costs are based on willingness to pay to avoid damage or willingness to accept compensation to tolerate e.g. a deteriorated health, although the latter choice is often not made before the damage has occurred. Policy costs may be based on estimated, or actual, whenever available, costs for remediation, prevention, and control.

We have restricted this study to policy costs for remediation at six industrial sites, to costs for preventing and controlling Hg emissions, and to food damage cost for the Arctic and potential loss of IQ in the Greenland population. To accomplish this, we reviewed literature on global Hg use, emissions, and pollution effects and studied remediation costs of different case studies and costs for measures to avoid mercury pollution. The absence of basic, quantified data on environmental effects from Hg pollution on a global scale as well as in the Arctic, caused that we could calculate a tentative damage cost only to food in the Arctic, excluding potential effects on biodiversity and most aspects of health, cultural loss and other aspects of the total damage costs.

The only health cost calculated here is the one due to loss of intelligence caused by excessive prenatal exposure to methyl Hg. The National Research Council (2000), based on studies in the Faroe Islands and New Zealand, concludes that developmental effects become apparent at levels around $5.8 \mu\text{g Hg L}^{-1}$ cord blood of newborn children (Kjellström, 1989; Grandjean et al., 1997). Above this limit, a linear relationship has been assumed with a loss of 1.5 IQ points for each doubling in cord blood Hg concentration (Trasande et al., 2005). Loss of one IQ point was calculated to correspond to a 2.578% decrease in lifetime expected earnings (average for males and females; Trasande et al., 2005). The value of lifetime earnings discounted to present value used in this study is 897 735 US\$, which is the average for an American child born in 2000 with an assumed 1% annual growth in productivity and a 3% real discount rate as calculated by Max et al. (2000). The annual average number of births in Greenland for 1998–1999 was obtained from United Nations (UN, 2002). Mercury levels in cord blood were obtained from Bjerregaard and Hansen (2000), sampling 178 newborn children and their mothers in western Greenland, where most Greenlanders are living and where the Hg levels in humans are lower than in eastern and northern Greenland (Hansen et al., 1983; Hansen and Pedersen, 1986). Since no table was presented including all original data, we constructed a normal curve. As a conservative approach, this

was based on the geometric mean, being lower than the arithmetic mean.

The remediation cases we present are mainly from Sweden, where about a dozen remediation projects involving Hg polluted sites have been or are being executed. Economic aspects of remediation at Minamata, Japan, the site of the most serious industry pollution from Hg, are also included.

Options to avoid emissions into the environment may, according to engineers' definition, be divided into three categories: a) preventive measures, b) primary control measures, and c) secondary control measures. Preventive measures are undertaken to prevent the generation of emissions. This could be fuel substitution and/or fuel washing with regard to the Hg emissions from power plants. At primary control measures, Hg emissions are generated but reduced at the moment of their generation, e.g. selection of various types of industrial technologies with lower combustion temperatures to reduce Hg emissions from power plants. At secondary control measures, Hg emissions are generated but removed later on from exhaust gases by flue gas desulfurization, application of electrostatic precipitators (ESP), carbon beds, etc. Here we present preventive measures and secondary control measures, because primary control measures are less efficient for Hg due to its characteristics such as low melting and boiling temperatures and high vapor pressure.

All monetary values are given in US\$ by converting values in Swedish crowns by the monthly average exchange rate as of September 2004 (1 US\$=7.4484 SEK) based on daily fix [(bid+ask)/2] (Riksbanken, 2004). Current prices have been used if otherwise not stated. In order to permit comparison of remediation costs between different case studies, the remediation cost per kilogram Hg recovered has been calculated. In case recovery of other toxics has been a contributing reason to carry out a remediation, the remediation costs should ideally be divided between the toxics. This partition is not evident, therefore, in those cases we present a figure where all remediation costs have been assigned Hg and mention types and quantities of other toxics secured.

3. Results and discussion

3.1. The Arctic

One of the most actual present day large-scale Hg problem is faced in the Arctic, where ecosystems and local communities are fragile due to high exposure, both from enhanced deposition as explained below, and dietary consumption, which may be synergistically multiplied by other organic pollutants bioaccumulated in the indigenous dietary sources (Grandjean et al., 2003). The Arctic as a region has no known significant natural or man-made sources of Hg, except for mines and smelters at the Kola and Taymyr peninsulas in Russia, annually emitting up to 5.5 metric tons Hg, including emissions from waste incineration and coal combustion in adjacent cities and gold mining in northeastern Siberia (calculated from ACAP, 2005). However, Hg's toxicity has already had measurable effects in the sub Arctic Faroe Islands (Grandjean et al., 1992, 1995, 1997;

Sørensen et al., 1999; Steuerwald et al., 2000), which through its isolation and traditional marine diet, may be compared with other indigenous communities of the far North.

Schroeder et al. (1998) reported on their 1995 discovery of the springtime depletion of tropospheric, gaseous Hg in the high Canadian Arctic, leading to a greater than expected deposition of Hg to the Arctic. It has been dubbed atmospheric Hg depletion episodes, AMDE (Schroeder et al., 2003). Skov et al. (2004) predict that this perennial phenomenon results in that, over the entire Arctic region, approximately 200 metric tons Hg is annually deposited, most of it of anthropogenic origin. This is more than double the amount of Hg earlier thought to be deposited to the Arctic. The USA has the largest anthropogenic Hg emissions of the countries with part of the territory in the Arctic, based on both total and per capita emissions (Norwegian Pollution Control Authority, 2005). Globally, China is the dominating Hg polluter and a considerable part of this Hg together with large Hg emissions in Central and South Europe enters the Arctic (Pacyna and Keeler, 1995; Dastoor and Larocque, 2004).

With the current market price of liquid Hg at approximately 14 500 US\$ per metric ton (Hayes, 2004), an oversimplified calculation shows that the price for these 200 tons of Hg, if sold on the market, would have been approximately 2.9 million US\$. This is much less than the 2000 million US\$ or more to salvage this amount of Hg from coal emissions (Poulson, 1994), which in this aspect appear to not be a cost effective measure.

3.1.1. Damage costs in the Arctic

Another aspect is to look at damages caused by these Hg emissions and related costs. There are no data on quantities of fish and other food, such as marine mammals, rejected in the Arctic because of Hg levels above a limit safe for consumption, but at continued emissions, an increasing quantity of fish and mammals caught will exceed this limit. The WHO guide line of 0.5 mg total Hg kg⁻¹ f.w. is generally used as a limit (Galvão and Corey, 1987), although Canada (for those who consume large amounts of fish), China and Japan has lower limits (0.2–0.4 mg total Hg kg⁻¹ f.w.; UNEP, 2002). Some other countries have a higher limit, 1.0 mg total Hg kg⁻¹ f.w., for piscivorous fish species, a value based on commercial considerations and not on health aspects (UNEP, 2002). However, the 0.5-mg value should be halved to harmonize with the revised provisional tolerable weekly intake (PTWI) for methyl Hg, being reduced from 3.3 to 1.6 µg/kg body weight per week (UN-FAO/WHO-JECFA, 2003). This revision was made by the Joint FAO/WHO Expert Committee on Food Additives at their meeting in June 2003 and it was realized to sufficiently protect the developing fetus, exposed to methyl Hg through contaminated food eaten by the pregnant mother. In areas such as the Arctic, where the daily fish servings exceed 90 g day⁻¹, the safe fish Hg limit is lower than in other areas, resulting in that an important part of the catch is presently unsuitable for consumption (Johansen et al., 2004). Mercury content consistently exceeds guideline limits for subsistence consumption or commercial sale of lake trout and northern pike in the Canadian Shield lakes of the Northwest Territories and northern Quebec, and also burbot

liver from Canadian Arctic, generally, exceed the limit (Braune et al., 1999).

Greenland, with most of its territory in the geographic area of the Arctic, has in average for the last 28-year period (1976–2003) landed 120 000 tons of fish and shellfish, which together with marine mammals captured, have resulted in the production of close to 58 000 tons year⁻¹ of fish, shellfish and products from marine mammals (FAO, 2004). The median export price for the exported quantities in current values has been 0.95 US\$ kg⁻¹, with the minimum value of 0.44 in 2001 and the maximum value of 1.89 in 1994. The median import price for fish and related products to Greenland has been more than twice as high at 2.30 US\$ kg⁻¹, with the minimum value of 0.46 in 1984 and the maximum value of 4.17 in 1998. The value of the annual marine production is 54.8 million US\$, if valued to the Greenland median export price, and 133.4 million US\$, if valued to the median import price (FAO, 2004). The export price is motivated to use when valuating marine products exported, making up 6.0% of total marine production in Greenland (FAO, 2004). However, to value the domestic consumption, we have chosen to use the import price, because fish and mammals caught with excessive levels of pollutants have, in general, to be replaced with imports and not by reduced exports, because of the marginal quantities exported (6%). Based on this, the value of the average annual marine production in Greenland is 128.7 million US\$. We have estimated the cost of Hg polluted marine products to be 24.5% of this value, corresponding to 31.5 million US\$ year⁻¹. This fraction is based on that the Inuits of western Greenland would need to avoid certain components of their traditional food to not exceed health guidelines for pollutants. These food items make up 24–25% of their present diet and by excluding them, their Hg intake would decrease by 44% (Johansen et al., 2004).

In addition to Greenland, more than 20 other countries, having no or a limited part of their territory in the Arctic, are annually capturing about two million (2 074 829) tons fish and shellfish in Arctic waters to an annual value of 1000 million US\$ (obtained by multiplying Greenland median export price with annual average captures for the last 28 years from the geographic Arctic, adjusted for losses from capture to sellable products with the same factor as for marine production in Greenland; data from FAO, 2004). This value is in parity with the cost of flue gas cleaning for anthropogenic Hg reaching the Arctic. Although not all fisheries in the Arctic are acutely threatened by Hg pollution, Inuits of eastern Canadian Arctic have a mean mercury intake calculated to be 122 µg day⁻¹ for women and 166 µg day⁻¹ for men (Zauke et al., 1994; Johansen et al., 2004). Since most of this Hg is in the methyl form, the intake is 5–7 times the revised, tolerable daily intake (UN-FAO/WHO-JECFA, 2003).

In North Greenland, where the highest Hg exposure in Greenlanders is documented, more than 80% of the population exceed the benchmark level of concern for the United States, 58 µg Hg L⁻¹ blood, and 16% exceed the World Health Organization (WHO) minimum toxic blood concentration in non-pregnant adults, 200 µg Hg L⁻¹ blood (Hansen and Pedersen, 1986). Weihe et al. (2002) suggest that observed

neurobehavioral deficits in Inuit children from Qaanaq, NW Greenland, might be related to dietary Hg exposure. Eating habits changing away from the traditional food, partly spurred by consumption advisories, has resulted in a reduced Hg burden of Inuit body tissues since the 1970's and 1980's (Hansen et al., 1983; Tulinius, 1995; Oostdam et al., 1999). Still, hair total Hg concentrations are often an order of magnitude higher nowadays among Inuits than the hair Hg concentrations among inhabitants of sub Arctic Alaska before industrialization (Egeland et al., 1999). Present hair methyl Hg concentrations may diverge even more than total Hg from pre-industrial concentrations (Egeland et al., 1999; Rothschild and Duffy, 2002).

Altogether, this indicates that many Inuits are exposed to alarmingly high prenatal and postnatal exposure to methyl Hg. Cord blood Hg levels above 5.8 µg L⁻¹ of born children are associated with loss of intelligence, which causes diminished economic productivity that persists over the entire lifetime of these children (Kjellström, 1989; Grandjean et al., 1997; National Research Council, 2000). This cost of methyl mercury toxicity is calculated to 8700 million US\$ annually (range, 2200–43 800 million US\$ in 2000 US\$) in the USA alone (Trasande et al., 2005).

Nearly 3/4 of the children born in Greenland are estimated to have cord blood Hg levels above 5.8 µg L⁻¹ (Table 1). Thereby the share of children with neurodevelopment deficiencies caused by methyl Hg is markedly larger in Greenland than in the USA, assuming that the sensitivity to methyl Hg damage is comparable in the two populations. However, the cost related to the damage is smaller in Greenland, because of fewer inhabitants there. Nevertheless, the cost is important to calculate and should be added to national figures of costs related to Hg in countries with anthropogenic Hg emissions. Based on data and calculations as presented in the Material and Methods section, the cost in Greenland of lost IQ due to methyl mercury toxicity of children born is estimated to 59.1 million US\$ each year (Table 1). The Greenlanders are bearing this cost without any gain from lower costs e. g. for reduction of Hg emissions as is the case in the USA, because nearly all Hg originates from outside their territory.

Table 1
Cost attributable to loss of intelligence from methyl mercury contaminated food in Greenland

Variable	Segment of births (percentile)						Sum
	27.2	33.5	47.4	74.5	98.2	>98.2	
Cord blood concentration (µg L ⁻¹) ^a	5.8	–11.6	–23.2	–46.4	–92.8	>92.8	
IQ-pints lost within interval	0	0.76	2.26	3.76	5.26	6.72	
Number of children affected ^b	–	61	134	262	229	17	703
Lost income (million US\$) ^c	0	1.0	6.8	21.9	26.8	2.6	59.1

^a Range 2.4–181 µg L⁻¹.

^b 967 children born year⁻¹ in average for 1998–1999 (UN, 2002).

^c Loss of 1 IQ-point is valued to 22 275 US\$ decreased lifetime earnings (see Material and methods section).

Some may argue that using a value of lifetime earnings calculated for the USA is inappropriate, because the value may be lower in Greenland. Certainly, the economic situation varies between countries but the per capita gross domestic product (GDP) demonstrates only a marginal difference between the USA (36 924 US\$) and Denmark (39 497 US\$), administering Greenland (2003 US\$; UN, 2004; no data are available on Greenland separately). In addition, USA is the dominating Hg emitter of the countries having territory in the Arctic and Hg emitted causes in health terms the same damage if a Greenlander is affected as if the victim is USA citizen.

The health effects most often associated with Hg toxicity are neurodevelopment deficiencies in developing fetuses and, at higher exposure, brain and other nervous damages in adults, but cardiovascular diseases may be even more costly (Rae and Graham, 2004; Rice and Hammitt, 2005). However, existing data from the Arctic has not permitted a more thorough study of health costs. For the same reason, also other costs related to Hg deposition in the Arctic have been omitted. Examples are lost recreational values and culture losses, physical stress associated with guidelines for consumption of fish and maritime products, and a possible reduction of biodiversity.

3.1.2. Aspects of mercury and related costs in the Arctic

Conventional economic approaches, such as cost benefit analyses, cannot handle fundamental differences between fishery and energy production via fossil fuel combustion. Fishery is an everlasting, renewable food resource as long as it is not harmed by pollution or excessive landings, while energy production via fossil fuel combustion is a non sustainable activity, extracting and finishing limited resources. This is why evasive efforts to not pay the full environmental costs should be counteracted so that the activities do not cause damages lasting far beyond the time limit of benefits obtained.

Smoke stacks at power plants are tall in order to distribute Hg and other pollutants emitted over a wider area and reduce deposition of them in the vicinity of the plant. A large part of Hg emitted to the atmosphere is transported globally as gaseous elemental mercury (Hg^0) before deposition, thereby shifting the pollution costs away from the ones profiting on no or reduced flue gas cleaning costs. Ethical aspects need to be considered when judging whether this is justified (O'Neill, 2004).

Small amounts of Hg enter the marine food web, are bio-magnified and eventually end up in people. How little? Let us consider that there is about five liters of blood in the adult human body, which with data from the Hansen and Pedersen (1986) survey show that more than 80% of the population in north Greenland exceeded 250 μg Hg per person and 16% exceeded 1000 μg Hg per person. If one applies the lower value as an average value for 80% of the entire population of Greenland of approximately 50 000 (Greenland statistical yearbook, 1992) and applies the 1000 μg limit to represent the average value of the remaining 20%, then the total amount of Hg reaching the top of the food chain in Greenland is 10 g in 80% of the population and 10 g in the other 20% of the population. This is totally 20 g of Hg distributed throughout the bodies of all Greenlanders, or $1/10^7$ of the approximate total annual Arctic deposition.

Increased atmospheric deposition of mercury since pre-industrial times have increased lake and sediment Hg fluxes by a factor of 2.5–3 (Fitzgerald et al., 2005; Semkin et al., 2005). How much of the deposition that actually ends up in the biota is not fully understood. Most Hg deposition is as inorganic Hg but there is also a significant fraction of methyl Hg deposited, suggested to be from a marine source (St. Louis et al., 2005). In freshwater lakes in the Arctic, methylation is driven by atmospheric Hg deposition, but minor amounts of the methyl Hg formed enter the biota as the sink, while the major part is photo-decomposed (Hammerschmidt et al., in press). However, it is clear that other biological compartments, besides humans, are also acting as sinks, and above a certain concentration also paying a cost. This threshold concentration differs depending of type of damage and varies with species and individual susceptibility. For this and other obvious reasons, the evaluation is not easy, but the example illustrates the need to carry out careful, multi-discipline analysis, including non-monetary economic aspects, since only a very small fraction of the deposited Hg ends up in humans, and it is assumed that practically any amount of Hg in humans above normal background exposure is too much.

Mercury concentrations of Arctic, marine biota indicate an increasing trend (Muir et al., 1999; Riget et al., 2004), although the ability of time series available to detect trends is rather poor. Time series are needed with more frequent sampling and starting before the 1970's. One possibility could be to analyze Hg content of baleen kept at museums or found at archeological sites. Baleen is an incrementally-growing tissue of balaenopterian whales, which preserves relatively well over time and might be useful for studies examining long-term changes of metal levels in whales (Hobson et al., 2004). Also more comprehensive Hg time series analyses are needed for terrestrial biota and freshwater fauna to evaluate a possible relation to increasing Hg levels in sediment records of the Arctic (Braune et al., 1999).

Global warming will increase the Hg load in the Arctic due to melting ice and increased weathering as a result of smaller areas with permafrost, while the increased temperature will stimulate Hg methylation, and an increased content of organic matter in water will inhibit photodecomposition and extend its residing time in the water column. Thereby, natural sources, or reemission of originally anthropogenic Hg, may in the near future be a larger contributor to Hg becoming bioavailable in the Arctic than present, anthropogenic Hg emissions (Macdonald et al., 2005). However, it should be observed that the anthropogenic emissions can be controlled by human actions now, while Hg released due to global warming are in practice not controllable any longer, although the rising temperature is caused by anthropogenic emissions of greenhouse gases, mainly carbon dioxide (Paeth et al., 1999).

Atmospheric mercury concentrations have been nearly constant for the last half of the 1990's (Slemr et al., 1977). As the economy of China, globally the dominating Hg polluter, grows it will need to generate more power. Assuming it does this by increasing output of coal combustion power plants or building more plants with its present technology, then Hg

deposition to the Arctic from this source will steadily increase in the future. Several new coal combustion power plants are also planned in the USA to compensate the decreasing energy output from oil, projected from the decrease in discoveries of new oil fields (Alekklett and Campbell, 2003).

Anthropogenic emissions of Hg, even if locally smaller than natural Hg sources, are adding to the natural Hg pool. Mercury background concentrations in biota are the result of equilibriums within different Hg cycles reached throughout millennia. Nowadays, large anthropogenic Hg emissions to air and water have resulted in Hg concentrations in biota increasing as an effect of efforts to reach a new equilibrium in chemical reactions and biological processes. The methylation cycle is poorly understood, especially in marine environments and quantitative links between the source and the effect are presently not available other than on a local or regional basis. However, it is evident that continued large Hg emissions from coal combustion will influence methyl Hg levels in Arctic biota and thus human exposure.

Stakeholders interested in continued exploitation of nature as a sink for contamination, by virtue of their ambition of maximized instant profit, are often disregarding externalities such as Hg bioaccumulation in biota and, as a consequence, advocating that their dissipative activities should not be hindered as long as it has not been scientifically proven that their emissions are the cause of a specific contamination driven damage. Considering the extraordinary powers of technology and that Hg is toxic and an element and therefore not degradable to harmless elements, an alternative approach, often referred to as the “precautionary principle”, could be that Hg emissions would not be permitted until it eventually has been scientifically proven that they will not cause damages in any aspect.

3.2. Remediation of contaminated sites

The Swedish environmental protection agency has investigated the possibilities to remediate some extensively polluted industry sites in order to obtain the goal to hand over a clean environment to future generations. Mercury has been emitted at several of these sites. Here we present results from prospecting or remediation activities at the Swedish sites Delångersån, a river dewatering into the lakes Kyrksjön and Långsjön, the chlor-alkali plant at Skoghall (northern Vänern), the lakes Svartsjöarna and Turingen, and the bay Örserumsviken of the Baltic Sea. We also present experiences from Minamata, Japan. The presentations are focusing on economic data in relation to pollutants removed, since the economic aspects of remediation are usually not easily accessible, although they, generally, determine whether a remediation project will be executed or not.

3.2.1. Delångersån, Kyrksjön and Långsjön

Forså paper mill has been producing paper pulp and paper from 1868 until 1983 and emitted its effluents into the river Rolfstaån, a part of the river Delångersån, in Hudiksvall municipality, Gävleborgs county in central eastern Sweden, nearly 300 km to the north of Stockholm (Fig. 1; Braff and Johansson, 1996). Large numbers of cellulose fibers were

emitted with the process water and these have built up large fiber banks, contaminated with Hg and chlor organic compounds. An organic Hg fungicide (phenyl mercury acetate) was used as an anti-slime agent in tubes and as a preservative for the pulp from the early 1940s until 1966. As a result, an estimated 2 tons Hg entered the river. The chlor organic compounds originate from recycled paper and transformer oils containing chlorine-based substances (PCB) and possibly also from wood impregnation chemicals such as pentachlor phenol. In a 1990 study, a 3-km stretch of the river contained 640 000 m³ of fiber sediments, containing about 100 kg of Hg and about 10 kg of PCB (Braff and Johansson, 1996). The Hg content varied between 0.01 and 3.2 mg kg⁻¹ dry matter. It was highest at the sediment surface and increased downstream. Methyl Hg made up 0.5–4.8% of the total Hg content. As a result of erosion and gas production, the fibers and pollutants are continuously being transported downstream and influencing ever-larger bottom areas in the river and the lakes Kyrksjön and Långsjön, situated downstream.

In 1966–1970, 1-kg pikes from Kyrksjön had on average 7.1 mg Hg kg⁻¹ f.w. and the maximum values recorded were 14 and 20.4 mg Hg kg⁻¹ f.w. (Braff and Johansson, 1996). Thereby the lakes have the dubious honor of world record for Hg in fresh water fish. The first years after that the Hg emissions were terminated, Hg content in fish decreased rapidly. In 1990, 1-kg pikes from Kyrksjön contained in average just below 2 mg Hg kg⁻¹ f.w. and 0.2–0.3-kg perches from Kyrksjön and Långsjön contained in 1993 about 1 mg Hg kg⁻¹ f.w. (Braff and Johansson, 1996). The decrease rate in fish Hg content has then approached an asymptotic shape and it was concluded that it would take centuries for the fish Hg content to reach background levels (0.5 mg Hg kg⁻¹ f.w. in 1-kg pikes) without any restoration activities. Therefore, a restoration plan was made.

In Rolfstaån, it was estimated that 640 000 m³ of contaminated fiber banks and bottom sediment with about 100 kg Hg and 10 kg PCB needed to be removed from the river and disposed safely on land to efficiently reduce the methylation and release of Hg from the sediment to the water column. The total cost for the remediation was estimated to 11–13 million US\$ nearly a decade ago, corresponding to close to 150 000 US\$ kg⁻¹ Hg secured (Table 2). So far, the remediation has not been executed in Rolfstaån, because of the large costs and unpredictable outcome of remediation of just the river but not the lakes Kyrksjön and Långsjön (M. Palm, Länsstyrelsen Gävleborg, personal communication). Many years have passed since the pollution took place and contaminated fibers have been transported from the river and spread over large areas of the lake bottoms, which therefore now contain about 300 kg Hg. This means that even larger volumes of sediments must be removed for a successful remediation aiming at obtaining fish Hg levels approaching background values in a near future.

3.2.2. Skoghall

Skoghall chlor-alkali plant was built at the northern shore of the lake Vänern, just to the south of Karlstad, in 1918 and operated with Hg cells until 1989 (Fig. 1; Lundgren, 2001). An estimated 100–130 tons Hg has been emitted from the plant,

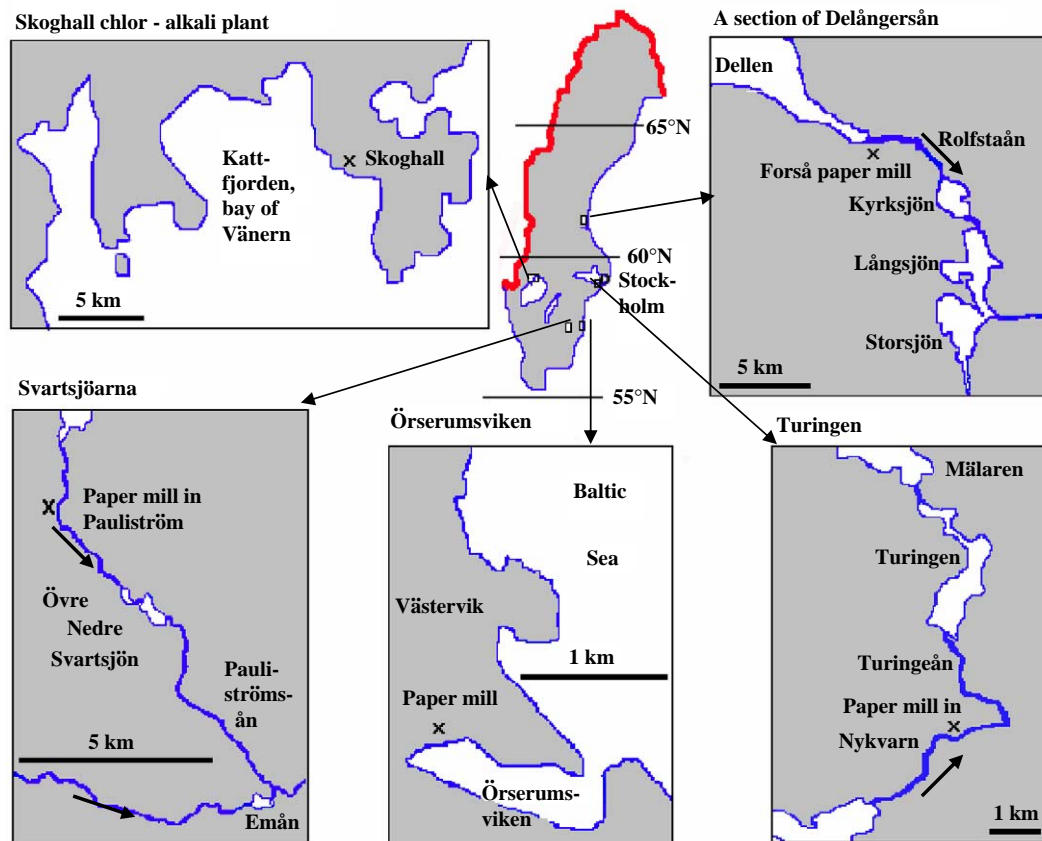


Fig. 1. Sites (rivers, lakes, and a bay of the Baltic Sea) polluted with mercury from a chlor-alkali plant at Skoghall and from paper mills at the other sites. Arrow beside a river indicates flow direction.

about half via water. This has resulted in that fishes, such as pikes, in adjacent parts of Vänern have Hg levels far above the limit for consumption. A successive change of chlor and alkali production for Hg free membrane technology was initiated in 1977 and completed in 1989, when Hg left over was transferred to another plant. The empty, contaminated building and soil is annually emitting about 16 kg Hg to air and 2.5 kg Hg to water (Lundgren, 2001).

Plans have been made to demolish the building and build a landfill for building scraps and stored wastes from the operation and contaminated soil, thereby reducing annual Hg emissions to an estimated one kg to the air and 0.5 kg to the water (Lundgren,

2001). However, the plans have still not been executed, because the authorities have not approved them. The lack of knowledge, laws, and instructions on how to handle this type of polluted industry site is a problem, delaying its remedy. The company wants to reduce the immediate emissions for a comparably low monetary cost, while the authorities, ideally, prefer remediation to original conditions and, as a minimum, want the emissions to be monitored also after the waste has been put in a landfill. The landfill is planned to contain 10 000 m³ Hg contaminated material with an estimated total of 5400 kg Hg. No detailed budget has been made for the project, but the total costs have been estimated to 16 000 US\$, corresponding to 3000 US\$ kg⁻¹

Table 2

Costs to remediate mercury contaminated sites

Country	Site	Hg emitted (ton)	Hg to secure/secured (kg)	Total cost (million SEK)	Total cost (million US\$)	Cost (US\$ kg ⁻¹ Hg secured)	Year of cost calculated	Observations
Japan	Minamata	190–225	95 000–115 000		517	4500–5400	1993	785 000 m ³ sediment dredged
Sweden	Rolfstaån	2	100 ^a	80–100	11–13	105 000–135 000	1995	640 000 m ³ fiber
	Skoghall	100–130	5400	120	16	3000	2002	10 000 m ³ building scrap and soil
	Svartsjöarna		15–150	100–120	15–16	98 000–1 100 000	2004	260 000 m ³ fiber
	Turingen		350	66	9	25 000	2004	225 000 m ³ sediment
	Örserumsviken		750	115	15	20 000	2002	1400 kg PCB, 170 000 tons sediment

^a Another 300 kg in lake sediments of Kyrksjön and Långsjön.

Hg secured (Table 2). The low cost per unit Hg secured compared to other remediation projects is caused by a highly concentrated Hg containing waste, because only the most severely polluted soil from the industry site has been planned to be put in the landfill. If all soil at the site should be put in a landfill or cleaned from Hg by retorting or some other process, the cost will escalate. The cost would be much higher and totally unrealistic, if the goal is set to restore soils surrounding the site to background levels, being orders of magnitudes lower than the present level. The costs do not include remediation of the sediment contaminated in Vänern, although there is a plan for isolating the sediment of Anholmsviken, the bay of Vänern receiving the effluents from the chlor-alkali plant.

3.2.3. Svartsjöarna

Svartsjöarna, named after its black water, are two lakes in Hultsfred municipality, the province of Småland, southern Sweden (Fig. 1). They were contaminated with Hg from a paper mill located in Pauliström. As in Rolfstaån, large numbers of fibers were emitted here, too, forming fiber banks of 260 000 m³. The amounts of Hg left in the sediments is comparably small, 10–100 kg in the upper lake, Övre Svartsjön, and possibly half that amount in the lake downstream, Nedre Svartsjön (Olof Regnell, Lund University, personal communication). Thereby the remediation cost per unit Hg secured is high, up to 1.1 million US\$ kg⁻¹ (Table 2). However, the environmental impact of Hg present is large because of chemical characteristics of the lake water, having a high concentration of dissolved organic matter. Such black waters are common in the boreal forest zone and known for increasing the fish Hg concentrations (Meili, 1991; Meili, 1997). The prospective for reduced fish Hg concentrations after remediation are good and the lakes are attractive for out-door activities. Starting in 2005, the lakes will be remediated for a planned cost of 15–16 million US\$, of which the former and present owners of the paper mill will contribute with about 20%, with the rest covered by government and municipal funds (Holmen et al., 2004).

3.2.4. Turingen

The lake Turingen is situated about 40 km to the southwest of Stockholm and drains into Mälaren, the main drinking water source of Stockholm (Fig. 1). A paper mill in the nearby municipality of Nykvarn used Hg fungicides between 1946 and 1966 (Projekt Turingen, 2003). An estimated 450 kg of Hg has contaminated Turingen and the river leading to the lake from the paper mill. The amount of Hg used in the plant is not known, but is expected to be many times larger, because most of the Hg used left the plant with wallpapers and other paper products sold.

The river and lake were remedied in 1995 and 1999–2003, respectively, at a cost of nearly 9 million US\$, corresponding to 25 000 US\$ kg⁻¹ Hg (Table 2; Bergman, 2004). The works included digging a partly new canal for the river beside two ponds, where most of the 100 kg Hg found in the river system was accumulated. The eastern part of the inlet bay of the lake was dredged and contaminated sediment and reed roots were

placed in the western part of the bay, covered by a synthetic fabric and a 0.4-m layer of clean, fine sand. Finally, 80% of the lake bottom was covered with an aluminum gel by applying AlCl₃ and NaOH with special equipment. The idea is that the gel of artificial aluminum polymers will reduce Hg water pollution by reducing the exchange between contaminated sediment and lake water and that the gel will reduce bioturbation and other sediment mixing processes between contaminated sediment and non-contaminated sediment formed on top of the gel. The formation of new sediment above the gel is crucial for long-term burial of the contaminated sediment, so the technology is not feasible in waters with a low sedimentation rate. The main advantage of the method is a lower cost than removing the sediment and placing it in a landfill. The long-term effect of this new technology is not known, but a 5-year assessment program commenced in 2004.

3.2.5. Örserumsviken

Örserumsviken, a bay of the Baltic Sea, just south of Västervik township, housed Westervik paper mill, which started its operation in 1915 and closed down in 1980 (Fig. 1; Jansson, 2003). Process water was let out into the bay, initially untreated, later, inadequately treated. As a result, the inner part of the bay filled up with fibers contaminated with Hg and PCB. Phenyl Hg acetate was used in the plant from the beginning of the 1950s until 1966. PCB originated from self-copying paper. In 1972, PCB was prohibited to use in open systems, but it continued to be emitted from the mill for many years, entering the production via recycled paper from archives (Meili, 2002).

Before closing down the plant in 1980, 220 000 m³ of Hg contaminated fiber sediments from 15 ha of the 37 ha wide bay bottom was dredged and deposited on land. This as a step to remediate Örserumsviken, where swimming had been prohibited for health reasons and fishing prohibited because of high Hg content. However, Hg content in pike did not decrease after the remediation in 1978–1979. In 1986–1987, the fish were found to also contain PCB. A third environmental pollutant, poly aromatic hydrocarbons (PAH), was discovered in 1997 (Meili, 2002).

A second remediation was therefore carried out in Örserumsviken in 2001–2003, because the first one had been incomplete and not reduced the pollution to desired levels. In addition, the problem with PCB and PAH had not been considered the first time. The second remediation has now been completed, including dredging the major area of the bay and securing the open-air landfill from the earlier remediation. The total costs amount to 15 million US\$, corresponding to 20 000 US\$ kg⁻¹ Hg secured (Table 2). Assigning the whole remediation cost on the quantity of Hg secured may seem illogical, because avoiding environmental effects of PCB was just as important a reason for the remediation, as securing Hg. Therefore, possibly only half of the costs should be assigned to secure Hg. This results in a cost of 10 000 US\$ kg⁻¹ Hg secured. However, the remediation costs in 1978–1979 were not included. They were 8.3 million SEK (1.1 million US\$) in 1979, today corresponding to about 4 million US\$ index-corrected value (SCB, 2004).

3.2.6. Minamata

Minamata, a fishing village in the south of Japan on the Minamata bay, is the site of one of humanity's most tragic cases of industrial pollution. In the 1940s, it was observed that rotten oysters became increasingly apparent and dead fish were found floating in the bay. In 1951, seabirds were dropping from the air. In 1953, the village cats could not walk steadily and the unknown disease afflicting the village was called the dancing cat disease. One year later, the deadly "dance" was recognizable in humans, who were mainly nourished by marine products from the bay and the name was changed to Minamata disease, which also turned out to be the diagnosis of a man getting ill already in 1942 (Environment Agency of Japan, 1994; Mari Susa, Kumamoto University, personal communication of a journalist's chronology). Already at an early stage, the cause was thought to be discharge from Chisso, a company established in Minamata a few decades earlier. In 1959, it was scientifically proven that organic Hg caused Minamata disease and that the Hg originated from the effluents of Chisso. Chisso used Hg salts (HgSO_4 and HgCl_2) in two production processes: acetaldehyde (an intermediate in plastics production) and vinyl chloride (another plastic). The wastewater contained both inorganic Hg as well as methyl Hg, originating mainly from the acetaldehyde process.

Despite the evident causes of Minamata disease, Chisso, supported by the Japanese government, continued to emit Hg into the sea until 1968, when wastewater from the vinyl chloride production was directed to a special pond for the three years prior to the production being halted in 1971 (Kudo and Turner, 1999). The acetaldehyde factory was closed in 1968, because the method had become outdated (Environment Agency of Japan, 1994). However, acetaldehyde is still produced using the same technology and with similar toxic emissions in China, showing that shortsighted economy still outweighs human and environmental health (Watts, 2001).

More than one hundred persons died as a direct result of methyl Hg poisoning in Minamata and tens of thousands were diagnosed with brain damage and damage to other parts of the nervous system, characteristic of methyl Hg poisoning. Therefore, mercury poisoning contributed to the death of many more persons. The symptoms ranged from deafness, blindness, paralysis and convulsions to unsteady walking, restricted field of vision and numbness (Tsubaki and Takahashi, 1986; Takeuchi and Eto, 1999). The economic compensation to victims poisoned with methyl Hg in Minamata has amounted to nearly 1500 million US\$ in current values from the 1950s until October 2004, when the Japanese Supreme Court settled the last Minamata disease lawsuit and found, in agreement with the District Court and High Court previously, that Chisso and the central and Kumamoto prefectural governments shared responsibility for failing to control the disease (Environment Agency of Japan, 1994; Pollack, 1997; Murayama, 2004). The governments were ordered to pay a total of 71.5 million yen (652 900 US\$) in compensation to 37 plaintiffs and Chisso was ordered to pay 240 million yen (2.2 million US\$) in compensation in this lawsuit (Murayama, 2004). Once the responsibility of the governments had been settled in a lawsuit, additional economic compensation became available to victims (Arita, 2005).

From 1932 until 1968, an estimated 190–225 tons Hg were emitted with the wastewater into the ocean from Chisso plants in Minamata (Kudo and Turner, 1999). Compensation to Minamata disease victims has therefore amounted to 6300–7500 US\$ kg^{-1} Hg emitted. Good health is invaluable and not possible to buy, once one has been poisoned by methyl Hg. However, only where there are large-scale monetary costs, as opposed to human costs, has the industry taken notice. This was experienced in Minamata and the same attitude is demonstrated nowadays, when industries with Hg stockpiles, notably the American and European chlor-alkali industry, dispose their stock to usages, which harm the health of the final buyers; such as gold miners using the amalgamation method. When considering the costs for depositing excess Hg in an environmentally sound Hg repository, it is as comparison, valuable to have some figures on health costs caused by Hg handled improperly. The calculations presented are not an attempt to quantify the total health costs due to Hg pollution in Minamata, but only to present some figures obtained from available data. The total health costs are not possible to estimate until the full extent of the effects caused by Hg emissions from Chisso have been revealed. The health effects were not restricted to Minamata Bay and around the Minamata River, but were extended to the major part of the population of about 200 000 persons living around the inland Sea of Shiranui at that time (Fig. 2; Ninomiya et al., 2005). In the coordinate efforts of Chisso and the Japanese government to hide the extent of methyl Hg poisoning, only 2265 persons were authorized as Minamata disease patients and eligible for full compensation, while another 12 000 persons received limited compensation (Environment Agency of Japan, 1994; Watts, 2001). Also the latter were being damaged by methyl Hg from the factory but they were lacking one or more criteria prescribed for Minamata disease, criteria set up by economic and political reasons rather than out of objective, clinical criteria. Instead of launching a survey of the real extent of Minamata disease, the

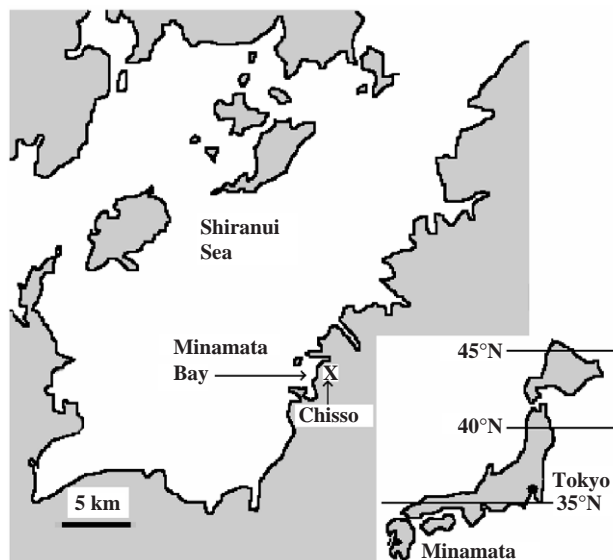


Fig. 2. Waters polluted with mercury from Chisso chemical plant in Minamata, Japan.

Japanese government is trying to conciliate certain victims with economic benefits (Arita, 2005). An approach which may hinder the determination of the full extent of effects at Minamata.

Minamata disease was the direct cause of 101 deaths and a contributing factor to another 800 deaths (Tsubaki and Takahashi, 1986; Watts, 2001). A human life is valued at 2.19 million US\$ (16.3 million SEK) when studying the willingness to pay for safer roads in Sweden (SR, 2005). Using this value for the deaths in Minamata results in totally 221 million US\$ for the 101 deaths and another 1752 million US\$ for the other 800 deaths. This corresponds to about 10 000 US\$ kg⁻¹ Hg emitted (980–1160 US\$ kg⁻¹ Hg for the 101 deaths and 7780–9220 US\$ kg⁻¹ Hg for the other 800 deaths), which is more than the compensation paid to Minamata disease victims.

Commercial fishing was also severely affected by the effluents from Chisso, but the compensation paid to fishery has been relatively moderate, at about 90 million US\$, corresponding to about 400 US\$ kg⁻¹ Hg emitted (Environment Agency of Japan, 1994), while the Minamata Disease Municipal Museum (2000) presents a somewhat lower figure (about 80 million US\$). The comparatively low value is an effect of the authorities' late ban on fishing and no ban on fishing in all waters where the fish were found to have unsanitary Hg levels (Environment Agency of Japan, 1994; Ninomiya et al., 2005). As a result, many more persons were poisoned than would have had been the case with a more accurate response from the authorities on the high Hg levels in fish. In short, by reducing monetary losses for commercial fishing, health costs in the region were drastically increased.

In 1971, the planning for remediation of Minamata bay started (Kudo and Turner, 1999). A plan was set in 1975 and work commenced in 1977, but was suspended until 1980, when it was continued and finally completed in 1990. A steel and concrete wall was built, dividing the bay in two parts. All sediment contaminated with Hg above 8.75 mg kg⁻¹ (and part of sediment with 5.0 to 8.75 mg kg⁻¹) outside the wall was dredged and pumped to the inside part of the bay for burial below fabric and a layer of virgin soil (Hosokawa, 1993; Kudo and Turner, 1999). An estimated 10–30% of emitted Hg had been redistributed into the sea outside the bay before dredging was initiated in 1980 (Kudo and Turner, 1999). At dredging, sediment polluted with Hg levels below 5–8.75 mg kg⁻¹ was allowed to remain in the basin (Hosokawa, 1993). Therefore, we have estimated that only 50% of Hg emitted was secured by dredging and stabilization in the inner part of the bay. With this assumption, the remediation cost ranged between 4500 and 5400 US\$ kg⁻¹ Hg secured in Minamata Bay (Table 2). Of this sum, about 500 US\$ kg⁻¹ Hg are assigned to the Marushima/Hyakken Waterways and Marushima Port projects (Minamata Disease Municipal Museum, 2000).

3.3. Measures to avoid mercury pollution

In connection with remediation costs, it is valuable to study the costs for measures to avoid Hg emissions and related pollution. The actions may for practical reasons be divided between measures against pollution from intentional use of Hg,

such as Hg pollution as a result of Hg used in dentistry and industry, and pollution from activities where Hg is not intentionally used, such as combustion of fossil fuels, where Hg is emitted because of its presence as a microelement in coal, oil, and gas.

3.3.1. Pollution from intentional use of Hg

Pollution caused by continued intentional use of Hg is easy to stop, because interrupted usage will result in terminated emissions and terminated pollution. The only major, intentional use of Hg presently lacking a technically and economically viable, Hg free alternative is the use of Hg in fluorescent tubes and other low energy lighting sources (Hylander and Meili, 2005). However, Hg free low energy lamps have been developed and are expected to enter the market in the near future.

Historical, intentional use of Hg has resulted in large stockpiles of Hg in society, which will result in continued pollution even after a total stop of using Hg in processes or in new products. Mercury used as electrodes in chlor-alkali plants is globally the largest stock of Hg and has been the second largest polluter of Hg used intentionally. The Hg cells in Sweden (400 000 kg) are due to be replaced by Hg free technology before 2010 and accompanied by a safe disposal of their Hg stock (Hylander and Meili, 2005). Mercury cells are well suited to be phased out at a faster pace globally, considering the comparatively low conversion costs per kilogram Hg (Table 3).

Mercury in dental fillings of the population is the second largest stock of Hg (about 40 000 kg) in Sweden. Besides these stocks, large amounts of Hg are in use in measuring and electrical equipments and in laboratory chemicals in most industrialized countries. Ongoing and potential pollution from these stocks are large in quantitative terms and manifested as many small emission sources over large geographic areas. This is perhaps most evident when looking at dental amalgam, which is carried by 74% of the grown up population in Sweden and results in a continuous release of about 100 kg Hg per year to the wastewater via every day chewing (Skare and Engqvist, 1994; Keml, 2004). This demonstrates that it is not possible to stop pollution from this Hg stock. In addition, technologies to reduce the emissions are costly and more difficult than replacing Hg as a dental filling material.

Replacement of Hg containing products in use for Hg free ones is a possible strategy to prevent pollution from other parts of the Hg stock in society. This strategy has been successfully employed, as demonstrated by the more than 345 000 potential pollution sources in the form of Hg thermometers having been replaced by Hg free thermometers in Sweden (Rein and Hylander, 2000). Where the historical usage and localization of Hg containing products is poorly known, employment of elder persons, earlier engaged in installing Hg containing equipment in e.g. real estates, is a possible course of action for tracking and document the localization of Hg containing products. Results from Sweden and estimates from Minnesota indicate that the costs to remove potential pollution sources, such as Hg containing equipments and laboratory chemicals,

Table 3
Costs for strategies avoiding Hg pollution and their potential to reduce Hg pollution, expressed in the classes: small, medium, and large

Activity	Place and year	Cost ^a (US\$ kg ⁻¹ Hg)	Reduction potential	Reference
Return of Hg thermometers	Sweden, 1992–1996	950–1200 ^b	Large	Rein and Hylander, 2000
Replace mercury-containing items	Minnesota, estimated 1999	20–2000 ^c	Large	Jackson et al., 2000
Collect Hg and Hg compounds in school labs	Sweden, 1995–1999	70–400 ^b	Small	Rein and Hylander, 2000
Collect metallic Hg in school laboratories	Minnesota, estimated 1999	20 ^c	Large	Jackson et al., 2000
Collect Hg compounds in school laboratories	Minnesota, estimated 1999	1400 ^c	Small	Jackson et al., 2000
Replacing Hg cells at chlor–alkali plants	USEPA, estimated 1996	10 100 ^d	Large	USEPA, 1997
Increase recycling of chairside traps in dentistry	Minnesota, estimated 1999	240	Medium	Jackson et al., 2000
Install amalgam separators	Minnesota, estimated 1999	33 000–1 300 000	Medium/Large	Jackson et al., 2000
Replace dental amalgam fillings at dentists	Sweden, estimated 2004	129 000	Large	This study
Remove dental amalgam fillings at death	Sweden, estimated 2004	400	Large	This study
Flue gas cleaning with carbon at crematoria	Sweden, estimated 2004	170 000–340 000	Medium/Large	This study
Flue gas cleaning with carbon at crematoria	UK, estimated 2004	29 000	Medium/Large	This study, BBC News, 2005
Medical waste incinerators with scrubber	USEPA, estimated 1996	4400–8800	Medium/Large	USEPA, 1997
Carbon injection into flue gases at waste incinerators	USEPA, estimated 1996	465–1900	Medium/Large	USEPA, 1997
Combined technologies at waste incineration	Uppsala, Sweden, 2004	40 000	Large	This study
Coal cleaning, conventional, chemical or both	Minnesota, estimated 1999	100 000–128 000	Large	Jackson et al., 2000
Carbon injection into flue gases at power plants	USEPA, estimated 1996	31 000–49 000 ^c	Large	USEPA, 1997
–“–	US Dep. Energy, estimated 1996	149 000–154 000 ^c	Large	Brown et al., 2000
–“–	Minnesota, estimated 1999	20 000–725 000	Large	Jackson et al., 2000
Combined technologies at power plants	USEPA, estimated 1996	11 000–61 000 ^c	Large	USEPA, 1997
–“–	US Dep. Energy, estimated 1996	56 000–85 000 ^c	Large	Brown et al., 2000
Wind as replacement for energy from coal	Minnesota, estimated 1999	1 200 000–2 000 000	Large	Jackson et al., 2000

^a Values in a range reflect differences across facilities of different sizes or at different recovery rates e.g. 90% or >95% of Hg recovered from flue gases, or other site-specific conditions.

^b Cost calculated per kilogram Hg collected and includes costs for information, reimbursement for thermometers, and additional costs for collecting, transport and deposition, while costs for additional working time of shop assistants, municipal officials, etc. are excluded.

^c Total cost per unit of Hg not emitted.

^d Capital and electrical costs. Indirectly reduced Hg emissions caused by lower consumption of electricity from Hg emitting power plants have not been included. The costs increase if pollution occurred earlier needs extensive remediation.

^e 90% reduction in mercury emissions. The EPA figures are based on a lower flue gas temperature when carbon is injected, thereby using the sorption capacity better, resulting in that only 2–34% active carbon is used compared to the DOE estimates.

range between 20 and 2000 US\$ kg⁻¹ Hg (Table 3), which is from four times the registered Hg commodity price at the actual time to 400 times higher. It should be noted that the Swedish figures are the real costs to remove Hg from products in use, while the figures from Minnesota are estimated costs to reduce estimated emissions of Hg via the actual strategy.

A low cost per kilogram Hg removed/not emitted does not necessarily indicate that this strategy should get the highest priority if the economic resources for avoiding Hg emissions are limited. Other factors should also be considered. Such factors are potential damage of Hg emitted and, for the Swedish figures, potential emission from the actual Hg stock. These considerations have been summarized as the reduction potential of the actual strategy, defined as the amount of Hg release that can be prevented or collected (Table 3). The reduction potential of the studied replacement strategies is often large. An exception, when looking at immediate effects, is collecting Hg and Hg compounds, especially in Swedish school and university laboratories. This is because Hg was generally not used in the laboratory exercises at time of collection, but had been left from earlier periods, when the health and environmental effects of Hg were less recognized. In case the Swedish waste handling instruction would be followed at any future disposal of these compounds, the Hg would be safely handled and disposed as harmful waste in the future, while actual emissions from the compounds on the laboratory shelves were small. Nevertheless,

collecting Hg in schools and universities got high priority in Sweden for reasons below.

Firstly, the pupils and students got involved in the collection campaigns, thereby resulting in increased awareness among the generation growing up, about mercury and environmental pollutants in general. This awareness was also transmitted to their parents and to the society in general, partly because large mass media attention about the employment of Hg tracker dogs. These dogs found Hg also where nobody knew there was any mercury, such as spills and Hg stored in irrelevant places without correct documentation. The risk that Hg may not be properly handled in the future, if not collected in a special campaign, was another reason that collecting Hg was organized in national school and university campaigns. The risk of improper handling of Hg waste is mainly not caused by nonchalance or illegality, but it is rather due to human limitation to be correctly informed about proper handling of thousands of chemicals in our daily lives, where for example, just one Hg compound, phenyl mercury acetate (C₈H₈HgO₂), is traded under at least 75 different names (The Registry of Toxic Effects of Chemical Substances as referred in Maxson, 2004).

Mercury emissions from earlier dental amalgam fillings will continue for several decades after a change to Hg free filling materials. Installing amalgam separators at dental clinics and advanced flue gas cleaning at crematories will reduce the major part of the emissions from these sources. The related investment

and running costs for these installations should be included in the cost for using dental amalgam. Presently, this is not the case in any country worldwide. As a consequence, amalgam fillings are considered to be economic while they de facto are more expensive than most, possibly all, other fillings when including environmental costs (Table 3).

Mercury emissions to wastewater from dental clinics can be reduced by installing amalgam separators at different cost levels (Table 2). The less expensive ones are working according to the sedimentary principle and can recover about 85% of Hg emitted to wastewater from dental chairs, while an additional device, recovering suspended and dissolved Hg may result in that more than 99% of the Hg emitted is recovered (Hylander et al., in press). A technical break-through is underway, so the high investment costs for amalgam separators resulting in high costs per kilogram Hg recovered are expected to decrease markedly in the near future and is presently markedly lower in Sweden than the estimates from Minnesota as presented in Table 3.

It should be observed that emissions of Hg to wastewater from everyday chewing cannot be avoided in any other way than replacing amalgam fillings with Hg free fillings. Although technically possible, this is expensive at 129 000 US\$ kg⁻¹ Hg, calculated as the average cost to replace an amalgam filling containing 0.6 g Hg with a composite filling at Uppsala county council's healthcare service (Folktandvården i Uppsala Län, 2004). We have in this cost not considered any possible health benefits associated with removing the fillings of persons with certain autoimmune and allergic diseases (Shimazu-Kohdera, 2000; Prochazkova et al., 2004).

Amalgam fillings not replaced before death will cause emissions to air, soil, and water upon cremation or burial. Mercury emissions from crematoria are one of the largest air pollution sources in many countries and several crematoria are being equipped with flue gas cleaning technology with associated, high costs (Table 3). The technology cannot recover all Hg in the flue gases, and a certain pollution level of Hg will still occur, contrary to if the fillings are removed before cremation. Removing amalgam fillings in an aesthetically correct and environmentally friendly way is less intruding or offensive than for example, the routine post-mortem examination and removal of a pacemaker in a person who utilized one. No investment is needed in infrastructure and the costs are restricted to working time spent. In addition to being an economically advantageous way to eliminate all Hg emissions from dental amalgam at both cremation and burial, the technology is easy to learn and can be used in all countries, regardless of economic and technical conditions.

3.3.2. Pollution of Hg from combustion

Pollution of Hg from combustion of fossil fuels is the dominating source of Hg pollution. Coal combustion is currently the main source of these emissions, worldwide contributing between 750 (Pirrone et al., 1996) and 1500 tons per year (Pacyna and Pacyna, 2002), thereby responsible for 3/4 of all current anthropogenic Hg emissions to the global atmosphere. Also oil and nature gas contain Hg emitted at combustion if not removed beforehand as generally done for gas, in the Nether-

lands yielding 5–10 tons Hg annually (UNEP, 2002). The mercury pollution from combustion of fossil fuels has problems in common with incineration of wastes (also called waste combustion) and cremation, where the Hg emissions are mainly the result of intentional use of Hg. These problems are a distant distribution of certain species of Hg emitted and that different species (particle bound, Hg^{II+}, and Hg⁰) of Hg are emitted. Especially gaseous, elemental Hg (Hg⁰) is transported far after emission, in average three turns around the globe before deposition (UNEP, 2002). Since a large portion of Hg emitted from combustion creates problems outside local and national boundaries, the interest is limited from the polluters to combat Hg emissions from combustion. At least by other means than using electrostatic precipitators to collect ash particles with particle bound Hg, which otherwise will deposit locally and result in blackening as the ash particles deposit.

The costs for flue gas cleaning is comparably high compared to other pollution reducing strategies (Table 3). One reason is the relatively low concentration of Hg combined with large amounts of material handled. Another is that a combination of different techniques is needed to efficiently capture the different Hg species in flue gases. Installing cleaning systems in an old plant is more expensive than if it is included initially when built, especially if a scrubber is to be installed, and may give sub-optimal cleaning efficiency for some pollutants other than Hg. Therefore, all new generation sources using fossil fuels and waste incinerators should be equipped with efficient flue gas cleaning systems at the time of construction. Although this will increase the investment capital cost by about one third, this is the most cost effective way to reduce global pollution of not only Hg, but of other pollutants from these sources. In addition to the control of mercury emissions there are other benefits presented below.

A new waste incinerator, with capacity to incinerate 175 000 tons solid waste per year and using the heat produced in the district heating network and supplying steam to nearby industries, has been demonstrated in Sweden. It was completed in Uppsala in 2004 for about 135 million US\$, of which 40 million US\$ was spent on flue gas cleaning equipment (Sollenberg, 2004, personal communication). Assuming 20 years of operation, 0.3 g Hg ton⁻¹ waste and 96% removing efficiency (Table 4), 1000 kg of Hg will be recovered with an investment cost of 40 000 US\$ kg⁻¹ Hg. In addition, several other metals, arsenic, sulfur, and chlorine will also be recovered and there will be practically no emissions of dioxins (Table 4, Hylander et al., 2003). In order to simplify the calculations, these pollutants are not bearing the investment cost in this example, but have been assumed to bear maintenance and other running costs. The running costs include chemicals used for cleaning wastewater from the scrubber and landfill costs for used active carbon.

Combining three devices (electrostatic precipitator or ESP, scrubber/condenser, and Filsorption unit with carbon injection) results in higher investment costs than using a single or two devices, but is the only approach available to capture different Hg species (USEPA, 1997; Hylander et al., 2003). In addition to a higher recovery rate for several pollutants, the amount of

Table 4
Recovered and emitted metals and arsenic from waste incineration at Uppsala 2001–2003^a

Element	Content (g ton ⁻¹ waste wet weight)	Recovered ^b in bottom ash (%)	Emitted ^b from flue gases (%)	To air (%)	To water (%)
Mercury	0.27	1.32	96.2	2.41	0.078
Lead	341	83.05	16.94	0.004	0.002
Cadmium	506	29.44	70.53	0.013	0.017
Chrome	23	83.44	16.59	0.019	0.002
Nickel	18	92.64	7.32	0.018	0.014
Zinc	636	55.54	44.45	0.006	0.004
Cobalt	3.8	88.31	11.66	0.004	0.030
Arsenic	2.1	50.83	49.09	0.079	0.005
Copper	432	96.08	3.84	0.001	0.001

^a For description on sampling and analytical procedures, see Hylander, Sollenberg, and Westas, 2003.

^b The sum may slightly deviate from 100% because only two-three decimals are presented.

carbon used in the cleaning is drastically reduced when combining these devices. The operating as well as landfill costs are thereby reduced (Table 3) and positively accompanied with reduced risk of future Hg losses from landfills. The advantage of having a less voluminous Hg containing waste is also an important advantage when considering the case of tooth extraction rather than flue gas cleaning at crematoria.

The Hg content of household waste incinerated at Uppsala is low in an international context (Hylander et al., 2003) and incinerating waste with inadequate separation of Hg containing items, such as batteries, energy saving lamps, and fluorescent tubes, will result in a higher content of Hg in the waste and subsequently lower cleaning cost per unit Hg removed, typically 5000–10 000 US\$ kg⁻¹ Hg (Poulson, 1994) or 500–2000 US\$ kg⁻¹ Hg, if a lower recovery rate and larger emissions are accepted than in the strict emission control at Uppsala (Table 3; USEPA, 1997). These figures may seem more appealing than the costs to remove Hg from power plants in operation (Table 3). The reason for the higher costs for removing Hg from power plants compared to waste incineration is partly a result of comparatively low Hg concentrations in coal. Another is the lower content of hydrogen chloride (HCl), resulting in the formation of mercuric chloride (HgCl₂), which is easily removed from flue gases of waste incinerators. Also, the high content of sulfur in coal and the subsequent formation of sulfur dioxide (SO₂) in the flue gas hampers Hg removal because it can act as a reducing agent, converting oxidized mercury to elemental mercury. These problems may be reduced by co-firing coal with other fuels containing more chloride and less sulfur.

Other pollutants, removed simultaneously as Hg with the same equipment, have not been considered in the presented power plant examples. These pollutants, ranging from other toxic elements and compounds to acidifying agents such as oxides of sulfur and nitrogen, need to be removed to safeguard our environment and human health. Large amounts of mercury, cadmium, and arsenic are leaving the flue gases if no cleaning is performed, while copper, nickel, cobalt, chrome, and lead are mainly left in the bottom ash (Table 4). The cleaning costs and

safe disposal of the ashes should be considered when projecting new, as well as when operating existing power plants and waste incinerators. The costs should be added to the energy cost or waste handling charge to avoid that we as consumers are adversely affecting the living conditions of future generations.

4. Conclusions

Efforts to reduce human and environmental exposure to Hg must be prioritized because of the adverse health and environmental effects. In a world of increasing population, such a nutritive food resource as fish and other seafood should not be withdrawn from human consumption due to anthropogenic pollution. Avoiding pollution is far more cost effective than remediation of polluted sites. Prevention combined with control measures is the only option to avoid further deterioration and a needed recovery of large and especially vulnerable regions such as the Arctic. Remediation costs in the presented case studies range between 2500 and 1.1 million US\$ kg⁻¹ Hg isolated from the biosphere depending on local circumstances such as quantities to secure, nature of pollution, media, geography, technology chosen etc. In general, remediation costs are lower the sooner remediation takes place after the pollution has occurred. To prevent pollution, regulations on discontinued use of Hg is cost effective and should be combined with extensive flue gas cleaning for all power plants and waste incinerators on a global basis.

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