Economics of mercury control

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Abstract

With mercury legislation now in place in Canada and under revision in the USA, and action being considered on a global scale, there is a flood of new technologies into the market place – from treatments for enhancing existing control technologies to completely new, mercury-specific, systems. Since there is currently no universal 'best available technique' for mercury removal, the approach at each plant is being determined on a case-by-case basis.

This report summarises the regulatory situation regarding mercury emissions in different countries, the status of mercury control technology development, and the costs of emission reduction. Where possible, the economic evaluation includes any increased costs due to changes in waste disposal options for coal combustion by-products. The report also considers mercury control options during coal processing and preparation.

Acknowledgement

Acronyms and abbreviations

ACI	activated carbon injection
ALAPCO	Association of Local Air Pollution Control Officials, USA
BAT	best available technique or technology
BEP	best environmental practice
CAIR	Clean Air Interstate Rule. USA
CAMR	Clean Air Mercury Rule, USA
CCC	Clean Coal Centre
CEM	continuous emissions monitor
CESP	cold-side electrostatic precipitator
CFBC	circulating fluidised bed combustion
CSI	Clear Skies Initiative, USA
CURS	Center for Urban and Regional Studies, USA
CWS	Canada-Wide Standards
EC	European Commission
ECO	electro catalytic oxidation
EIP	Environmental Integrity Project, USA
ESP	electrostatic precipitator(s)
EU	European Union
FBC	fluidised bed combustion
FF	fabric filter (baghouse)
FGD	flue gas desulphurisation
FPP	Fayette Power Project, USA
GEF	Global Environment Fund
HELCOM	Helsinki Commission
HESP	hot-side electrostatic precipitator
ICAC	Institute of Clean Air Companies, USA
ICR	Information Collection Request, USA
IGCC	integrated gasification combined cycle
IQ	intelligence quota
IPPC	Integrated Pollution Prevention and Control,
EU	European Union
LCPD	Large Combustion Plant Directive, EU
LRTAP	long-range transboundary air pollution
LSFO	limestone forced oxidation
MACT	maximum achievable control technology
Macf	million actual cubic feet
Macm	million actual cubic metres
MEA	multilateral environmental agreements
MEPOP	political initiative on mercury and persistent organic pollutants
mill	$\frac{1}{10}$ US cent
MW	megawatts
MWe	megawatts (electric)
NARAP	North American Regional Action Plan
NDRC	National Development and Reform Commission, China
NETL	National Energy Technology Centre
NPI	National Pollution Inventory, Australia
NRDC	National Resources Defence Council, USA
NSPS	New Source Performance Standards, USA
NWF	National Wildlife Federation, USA
OEWG	open-ended working group (UNEP)
OSPAR	Oslo and Paris Commission
PAC	powdered activated carbon
PCO	photo chemical oxidation
PEESP™	plasma enhanced electrostatic precipitator
POPs	persistent organic pollutants

PRB	Powder River Basin
PRTR	Pollutant Release and Transfer Register, EC
ROM	rough order-of-magnitude
SAICM	Strategic Approach to International Chemicals Management
SCR	selective catalytic reduction
SDA	spray dry absorber
SIP	state implementation plan, USA
SNCR	selective non-catalytic reduction
STAPPA	State and Territorial Air Pollution Program Administrators, USA
UN	United Nations
UNECE	UN Economic Commission for Europe
UNEP	UN Environment Programme
UN FCC	UN Framework Convention on Climate Change
US DOE	US Department of Energy
US EPA	US Environmental Protection Agency
USGS	United States Geological Survey
UV	ultra-violet

Contents

1   Introduction

Mercury is released into the environment through natural processes (such as volcanic activity and weathering of rocks). However, human activities (mining, fuel use, products and processes such as chlor-alkali production) are now assumed to be the main source of mercury release into the environment. Total emissions globally have been estimated at 5000 t/y (US DOE, 2008). Mercury is still used in many products such as batteries, switches, thermometers, pressure gauges, dental amalgam and even face-whitening creams. Mercury use in small-scale gold mining is growing at an alarming rate with potentially over 50 million people worldwide involved in this activity. Alternatives to mercury exist in most industrial and product applications.

Mercury is released to the atmosphere from natural sources and human activities where it can drift for a year or more, spreading with air currents over vast regions of the globe. Mercury circulates between air, water and soil, until it comes to rest in sediments or landfills. On average, about three times more mercury is deposited from the atmosphere now than before the Industrial Revolution 200 years ago. For the last 30 s years, emissions from developing countries and economies in transition have increased, offsetting decreased emissions from developed countries.

Mercury pollution, mostly in the form of methylmercury, has caused severe health effects in several chemical incidents, most notably the incident in Minamata Bay, Japan, which caused severe neurological damage to over 2200 victims. Most exposure to methylmercury in the environment does not occur at anywhere near these levels. Governmental bodies have set daily mercury intake levels that are considered safe. The risk from diet mainly depends on how much contaminated fish is eaten: moderate consumption of fish with low levels of mercury is not a major cause for concern. However, there is sufficient evidence showing the effects of methylmercury on foetal development to justify warning children and women of child-bearing age to be careful about the species of fish they eat. Methylmercury also poses a health threat to predator fish, fish-eating birds and mammals, such as bald eagles, loons, otters, polar bears and seals.

Coal combustion can be a significant source of mercury to the atmosphere in some countries. However, on a global scale the contribution from coal combustion in most developed nations is relatively minor – mercury emissions from human activities in the USA are estimated to account for only around 3% of the global total and only 1% is from US coal-fired power plants. The USA recently promulgated several new pieces of legislation including the Clean Air Mercury Rule (CAMR) and the Energy Policy Act, along with the Global Climate Change and Clear Skies Initiatives (CSI). However, CAMR was overturned in early 2008 and is currently being rewritten. Many individual states within the USA have set their own more stringent regulations and targets most of which still apply, despite the vacation of CAMR. The US Department of Energy (US DOE) has invested heavily in the development of low-cost efficient energy techniques with a target of 90% mercury emissions capture efficiency by 2010 and continues to spend billions of dollars on clean coal technologies. All these regulations and targets mean that there is an unprecedented amount of activity in the development of mercury control technologies in the USA. In Canada, the Canada-Wide Standard (CWS) is also causing rapid movement towards the deployment of new mercury control options.

Although other countries in Europe and Asia are also actively developing mercury control systems, they are doing so without the urgency incurred by national and regional binding legislation and specific mercury reduction targets such as those that apply in North America at the moment. As a result the majority of research and development on new mercury control techniques is taking place in North America.

The United Nations Environment Programme (UNEP) Governing Council is likely to take a decision in February 2009 to further strengthen international action on mercury. The form this action is likely to take, whether a legally binding mechanism or a more flexible voluntary approach, is as yet undecided. A framework is established for voluntary initiatives under the UNEP Global Mercury Partnership; any additional measures initiated by the UNEP Governing Council in 2009 would complement and strengthen the existing, ongoing activity under the Partnership.

The tightening of legislation and the instigation of action plans are likely to result in the further reduction of mercury emissions from developed countries. However, the rapid increase in coal use in countries such as those in Asia may override reductions elsewhere. It is therefore essential that mercury control strategies are made both technically and economically viable in developing countries to ensure that the current upward trend in global mercury emissions is controlled effectively.

This report reviews the economics of mercury control options, from the legislative approaches through to the control technologies themselves. Chapter 2 reviews the legislation and action plans in place around the world that aim to reduce the global mercury burden. International, regional, multilateral, national and state legislation are summarised and discussed, highlighting, where possible, why different strategies are in place in different places. Chapter 3 then discusses the economics of the legislation, the cost benefits and the changes in the cost of mercury control with market factors. Chapter 4 concentrates on evaluating the potential for mercury control as a co-benefit effect due to the installation of control technologies for other pollutants such as SO_2 and NOx. Chapter 5 looks more closely at mercury-specific control technologies for large coal-fired plants. Finally, Chapter 6 looks at how the selection of the most appropriate mercury control strategy in developing countries will be determined by both plant specific factors and greater legal and economic issues.

As discussed in a previous IEA CCC report (Sloss, 2003) legislation on emissions is commonly set in response to a recognised environmental problem. Mercury pollution has been reported in lakes in Northern Europe (especially in Sweden) and in North America. It is these countries that have taken the most remedial action with respect to the mercury problem. There may be significant mercury pollution in other emerging economies. However, pollution monitoring and reporting is not as stringent in these areas and therefore the problem is not being recognised and/or dealt with as efficiently.

Countries such as the USA, Canada and most of Europe arguably have relatively accurate information on mercury emissions. However, the data from Asia, Africa, South America, and Indonesia are sparse. Unfortunately, the lack of data on total coal use, plant type, control technologies and so on, mean that even a best estimate/guess based on generalised emission factors is likely to give an inaccurate result. In order for reduction strategies to be successful, there has to be some means to identify major sources, to determine baseline emissions, and to estimate the potential or observed reductions in emissions due to proposed or applied approaches. In a recent UN Environment Programme meeting concerning global mercury emissions (UNEP ad hoc open-ended working group on mercury, Bangkok 2007), one of the priorities listed was the requirement for more detailed and accurate emission inventories. A guidance document has been produced by UNEP which gives simple and concise instructions on how to prepare a best estimate for a mercury emission inventory (UNEP, 2005). A new study updating the previous global emission inventory for mercury will be available by the end of 2008 and will be presented to the UNEP Governing Council at its 25th session in February 2009.

Further work is also needed on the nature of mercury cycling in the environment, including a better understanding of the environmental effects of this complex element and more accurate information on the cause and effect of the different biological and neurological problems that may occur with increasing concentrations of mercury in the environment. A greater understanding of the true nature of problem will lead to more suitable and applicable solutions. These problems are outside the scope of this report but are the primary concern of the UNEP Global Mercury Partnership (*see below*).

As would be expected, the majority of the legislation and action on mercury control discussed in this Chapter applies in the developed world. The sections to follow briefly review the current and impending legislation on mercury both internationally and nationally.

2.1   International

Agreements between countries to work together to reduce emissions and concentrations of mercury are summarised in the sections to follow.

2.1.1   Multinational and binational agreements

There are a number of international agreements and action plans to co-ordinate action to reduce mercury emissions. These include (Sloss, 2003):

UNECE: The United Nations Economic Commission for Europe (UNECE) has a convention on long-range trans-boundary air pollution (LRTAP). This convention was published in 1998 and covers heavy metals including mercury. The protocol has been signed by Canada, Europe, Russia and the USA. Although the protocol calls for the installation of BAT (best available technique or technology) at new stationary sources, it does not go so far as to define BAT for coal-fired plants nor to specify any reduction strategies.

OSPAR: Oslo and Paris Commission's programme on reduction of land-based pollutants transported to the North Sea.

HELCOM: The Helsinki Commission programme covering the North Sea.

Barcelona Convention: A programme similar to OSPAR and HELCOM covering the Mediterranean Sea.

MEPOP: A EUREKA European political initiative studying the atmospheric cycling of mercury and persistent organic pollutants.

Nordic: Project between Denmark, Finland, Norway and Sweden to reduce mercury emissions.

Arctic: The Arctic Council's Environmental Protection Strategy includes mercury.

NARAP: North American Regional Action Plan between Canada, the United Mexican States and the USA to reduce mercury fluxes.

Binational Toxics: Canada and the USA have a project for cleaning up substances, including mercury, in the Great Lakes Basin Area.

None of these agreements or programmes include guidelines on how the proposed reductions in emissions or concentrations should be achieved other than by recommending 'best practices'. The agreements rely on the individual governments of each signatory country to produce a successful strategy to reduce mercury emissions. They therefore do not necessarily guarantee results. Action is rarely, if ever, taken against countries that are not as successful as others in reducing emissions.

2.1.2 UNEP Mercury Programme

In 2007, within Decision 24/3, the UNEP Governing Council recognised that *current efforts to reduce risks for mercury are not sufficient to address the global challenges posed by mercury*' and concluded that '*further long-term international action is required to reduce risks to human health and the environment...* For this reason, an ad hoc open-ended working group (OEWG) of Governments, regional economic integration organisations and stakeholder representatives was established that would review and assess options for enhanced voluntary measures and new or existing legal instruments in order to make progress in addressing the issue of mercury.

To facilitate the work of the OEWG, UNEP have prepared a study (UNEP 2007b) on options for a global initiative for the reduction of global mercury emissions which considered the different approaches, both legal and voluntary, that could be used to reduce global mercury emissions. Options for enhanced voluntary measures were discussed within the study. The study also outlined how voluntary approaches could include bilateral and multilateral cooperation as well as co-operation on a global scale. The approach could be a comprehensive, over-arching instrument or a number of narrower, discrete interventions which could concentrate on individual source types such as mercury containing products or mercury from coal combustion.

The study noted that voluntary approaches, lacking strong enforcement mechanisms, may be less likely to achieve as much mercury reduction as a legally binding approach. The report then reviewed existing international legal instruments to determine which, if any, could provide a template for a new legally binding mechanism for mercury, such as:
● the control of mercury wastes under the Basel Convention;
● expanding the scope of the Stockholm Convention (POPs – persistent organic pollutants);
● restriction of international trade in mercury under the Rotterdam convention;
● inclusion of mercury in the 'right to know' under the Aarhus Convention's Kiev PRTR (Pollutant Release and Transfer Register) Protocol;
● potential synergies with the UN Framework Convention on Climate Change (UN FCCC) (co-benefits through greenhouse gas mitigation).

There were also two options for new international legal instruments presented:
● mercury control under the Stockholm Convention – a new protocol for a legal instrument that addresses mercury directly;
● free-standing mercury convention – an independent freestanding protocol agreed on by signatory governments.

The effectiveness of an international agreement, voluntary or legally binding, is closely linked to the availability of financial and technical assistance to aid implementation. This is discussed in more detail in Chapter 3.

It was acknowledged that any new, legally binding, global instrument of mercury would not enter into force until 2012 at the earliest. The working group will prepare a final report representing options and any consensus recommendations to the Governing Council at its 25th session in February 2009. Although the options and consensus recommendations were not available as this report went to press, there is an emerging acknowledgement that a mixture of voluntary and legally binding measures are required to address to mercury issue at the international level. As part of Decision 24/3, UNEP was also tasked with strengthening ongoing voluntary mercury partnership activity, including the development of an over-arching framework for a UNEP Global Mercury Partnership, uniting governments and stakeholders around the world to work together. Partnership activity was initiated in 2005 and has been divided into five partnership areas, as follows:
● coal combustion;
● small-scale gold mining;
● chlor-alkali production;
● mercury in products;
● mercury fate and air transport research.

With international discussions on the delivery mechanism(s) for mercury reductions still in an early stage, it appears that the available response measures specific to reducing mercury emissions from coal combustion include:
● the establishment of mercury emission reduction targets and timetables;
● the establishment of mercury emission limits (end of pipe controls);
● improvements in energy efficiency in products and processes for lessen demand from electricity and the need to combust coal in electricity generation;
● improvements in energy conversion efficiency to reduce coal combustion (housekeeping, maintenance, boiler optimisation);
● transition to other energy sources (such as renewables) to reduce coal combustion;
● the pre-treatment of coal prior to combustion (coal washing);
● the use of higher ranking (lower mercury) coals;
● the establishment of mercury-specific BAT (best available technique or technology) standards for emissions control devices to capture mercury in flue gases;
● the use of air pollution control technologies for other criteria pollutants to capture mercury in flue gas;
● promotion of the development and use of mercury specific and cost-effective control techniques;
● promotion of the development and use of cost-effective multi-pollutants (so called 'zero or low emission' control techniques);
● establishment of monitoring and reporting programmes.

The first objective of the coal partnership is to produce a guidance document on BAT/BEP (best available technique/best environmental practice). This document, to be produced in conjunction with the IEA Clean Coal Centre (IEA CCC) would provide a simple summary of various mercury control options at coal-fired plant to allow developing nations to select measures which would be most appropriate and economic for their situation. It is proposed that this document would be completed by the end of 2008.

2.2   Regional

As discussed in a previous IEA CCC report (Sloss, 2003), existing legislation in Europe, especially the Large Combustion Plant Directive (LCPD) for particulates, SO_2 and NOx, has meant that most plants are fitting technologies such as low-NOx burners, SCR (selective catalytic reduction) and FGD (flue gas desulphurisation). This has meant that mercury emissions have also been reduced due to co-benefit effects. Co-benefit effects are those whereby mercury emissions are reduced as a result of the installations of control technologies for other pollutants and these are discussed in more detail in Chapter 4.

Richie and others (2005) have estimated the reduction in mercury emissions from coal-fired plants in Europe as a result of current and impending legislation, as shown in Table 1. Mercury emissions are predicted to decrease significantly between now and 2020, even with the arrival of new accession countries into the EU. As these new countries join they must adopt, within a prescribed timescale, the relevant EU legislation and therefore the co-benefit effects of SO_2 and NOx control within the LCPD are evident.

The European Commission (EC) recognises the significant reduction that has already been achieved in mercury emissions from coal combustion and also that this reduction is likely to increase with the tightening requirements already specified under the LCPD. It is therefore likely that the 'wait and see' approach to reducing mercury emissions could continue for a few more years. However, it is also likely that mercury monitoring could become a requirement on larger plants to obtain more accurate mercury inventory data. BAT for mercury may become a requirement in the future, but this will not be made binding until there is a better understanding of what BAT for mercury would actually comprise.

The new IPPC (Integrated Pollution Prevention and Control) permit scheme will make it easier for the EC to monitor individual plant operation and emissions. The new scheme will provide the regulators with a large amount of plant performance characteristics and data as well as more accurate emission inventories. This information will be useful in determining if mercury specific limits or controls are required and possibly even provide guidance on the most appropriate

means of mercury reduction at each plant. Data from the IPPC will be available at the end of 2008 and any decisions would be made after that. IPPC applies only to plants >50 MW. However, the EU is already considering the possibility of extending the scheme to include smaller plants.

2.3   National

Emission inventories and country specific action plans and legislation were discussed in more detail in previous reports from the IEA CCC (Sloss, 2002, 2003). The sections below focus on legislation and action being taken specifically to reduce emissions of mercury from the coal sector. Where possible, an explanation is given as to why the selected approach to mercury reduction was taken. The countries are ranked in order of those with the most legislation specific to mercury control.

2.3.1   USA

The electricity sector in the USA contributes around 40% of the total national emissions of mercury (Palmer and others, 2007). However, the US EPA believe that only 8% (11 of 144 t) of the mercury deposited from the atmosphere in the USA is actually from electric power plants in the USA, the remainder being from trans-boundary air pollution (McManus and others, 2005).

Mercury emissions in the USA dropped from 199 t/y in 1990 to 101 t/y in 1999, as shown in Figure 1 (US EPA, 2008). During that time, emissions from utility coal boilers only
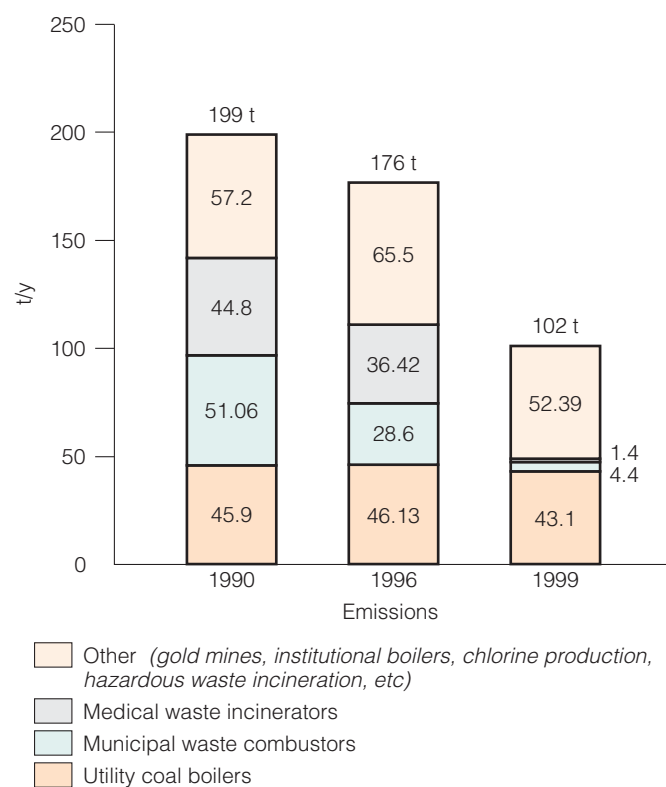
Figure 1   Mercury emissions in the USA have dropped 45% since 1990 (US EPA, 2008)

Table 1	Mercury emissions from coal plants in Europe, t/y (Richie and others, 2005)		
Year	EU-15*	†10	Total
1995	22	30	52
2000	20	14	34
2005	15	14	29
2010	11	13	26
2020	9	6	15

*   EU-15 are the original 15 member countries of the EU
†   the 10 new accession states

dropped from 51 t/y to 48 t/y. So, although the total emissions from coal combustion had decreased slightly, the contribution from coal to the total actually increased from 23% to almost 43% (US EPA, 2007a,b,c). It is therefore not surprising that coal has become the target for specific legislative action.

The USA proposed to reduce mercury emissions with two new rules. The Clean Air Interstate Rule (CAIR) will reduce SO_2 emissions by 70% and NOx by 65% (from 2002 levels) throughout most of the eastern United States via two-phase cap-and-trade programme. In Phase 1 of the plan 75% of the total emissions will be reduced. Although CAIR does not specifically target mercury, it is assumed that mercury emissions will be reduced as a co-benefit since the technologies used to control SO_2 and NOx (such as FGD and SCR) can help reduce mercury emissions. This co-benefit effect is explained in more detail in Chapter 4. CAIR is predicted to cost the USA $2.5–5.2 billion (in 2004 dollars) each year until 2020 (discussed in more detail in Chapter 3).

The Clean Air Mercury Rule (CAMR) was the first national regulatory programme in the world which would have specifically targeted mercury emissions from coal-fired power plants. However, the CAMR has been vacated, as outlined in the following official statement from the US EPA (Mazza, 2008):

> On February 8 2008, the United States Court of Appeals for the DC Circuit issued its ruling on the petitions for review of EPA's March 2005 Clean Air Act §112(n) Revision Rule and Clean Air Mercury Rule (CAMR). The court vacated both rules. EPA's §112(n) Revision Rule concluded that it was not appropriate or necessary to regulate mercury emissions from power plants under §112 of the Clean Air Act. The court held that under the plain language of the Clean Air Act, EPA must make certain findings from the §112c (9) before removing any source category from the §112c list of source categories. Because EPA has not made those findings in the §112(n) Revision Rule, the court found that EPA could not remove power plants from the §112c list, and therefore vacated the rule. The court also vacated CAMR, because coal-fired electric generating units are listed under §112 and therefore regulation of existing sources' mercury emissions under §111 is prohibited.

Although the CAMR has been vacated, it is still important to include discussion of it here since it has been the driving force behind the decisions that have been made at many coal-fired plants over the last years in the US in preparation for compliance. The US EPA is currently formulating a new mercury regulation which is likely to aim for the same level of mercury reduction but through a different legislative strategy.

CAMR aimed to reduce mercury emissions by two thirds by 2018. CAMR proposed to cap emissions of mercury as follows:

cap	target year
38 tons/y (34 t/y)	2010
15 tons/y (13.4 t/y)	2018

Each of the US states plus two tribal groups were to be allocated a cap on total mercury emissions and would trade within themselves to achieve this cap. The budgets were permanent, regardless of any growth in the electric sector. Further, the caps were mandatory and were coupled with 'significant penalties' for non-compliance (US EPA, 2007a,b,c). Within the cap-and-trade programme, CAMR addressed differences in coal type by using allocation 'adjustment factors' for each coal rank as follows:

bituminous	1
subbituminous	1.25
lignite	3

The reason for the adjustment factors for different coal ranks is discussed in more detail in Section 5.1. Individual state mercury budgets for the programme were to be derived by multiplying individual units' adjusted baseline heat-input ratio of the total national baseline heat input by the available mercury allocation (Steele and Schaefer, 2005). The CAMR would have applied to all coal facilities over 25 MW (EERC, 2005a).

The caps under CAMR would not have changed or increased with time so any new plants being brought online would either have needed to negotiate significant allowances and/or have effective mercury control systems. In November 2005 the US DOE knew of 129 proposed new coal-fired plants representing 77 GW of new power (Toole-O'Neil, 2006).

Despite the cancellation of the CAMR cap-and-trade programme, new coal-fired plants (those with construction starting on or after 30th January 2004) will still have to meet stringent new source performance standards (NSPS). The limits for mercury are summarised in Table 2.

CAMR was vacated as a result of action taken by many different organisations who were unhappy in how it was formatted. For example, STAPPA/ALAPCO (State and Territorial Air Pollution Program Administrators and the Association of Local Air Pollution Control Officials) suggested that the mercury reduction targets within CAMR should have been lower and they were concerned that the use of trading and banking could mean that mercury emissions would actually be 24 t/y in 2020 rather than the 15 t/y cap that was proposed (Becker, 2005).

Table 2	New source mercury emission standards, USA (EERC, 2005a)	
Unit	lb/10^6 TWh	ng/J
Bituminous units	21.0	0.00260
Subbituminous units Wet FGD Dry FGD	42.0 78.0	0.00530 0.00980
Lignite units	145.0	0.01830
Coal refuse units	1.4	0.00018
IGCC	20	0.00250

An independent report by an action group called the Environmental Integrity Project (EIP, 2007) suggested that the EPA's proposed legislation was *unlikely to have any measurable benefit in the short term* although long-term results would have been more acceptable. The EIP has used the US DOE and US EPA databases of power plants and emissions to estimate the most 'polluting' plants in the USA. The 486 coal-fired plants tracked in the US EPA's toxic emissions inventory released 48.3 tons (43.1 t) of mercury in 2005. Of these, the largest 274 plants released 43.5 tons (38.8 t). The top 50 plants together emitted 16 tons (14.3 t) of mercury but generated less than 18% of the electricity. The EIP names American Electric Power's Pirkey plant (Texas) and Reliant's Shawville plant (Pennsylvania) as the two 'dirtiest' plants based on mercury emission rates. Based on total mercury emissions, in pounds, TXU's Martin Lake (Texas) ranked highest with 1705 lb/y (773 kg/y). Southern Company's Scherer plant (Georgia) came second at 1662 lb/y (754 kg/y). The EIP report lists each of the top 50 plants on the list and gives details of their annual emission rates and total tons emitted.

Whilst the EIP report points the finger of blame at several plants and utilities in the USA, it also provides an indication of which plants should be targeted first for mercury control. The EIP suggests that, once the technologies for mercury capture are commercialised, these systems could be applied to the most polluting plants first in order to achieve the most efficient and rapid reduction in mercury emissions. However, the economics of mercury control at these plants may not be simple due to the variations in mercury chemistry at each site.

Another common complaint against the CAMR was that the trading of mercury would allow some areas to become cleaner whilst others became more polluted. Stadler (2005) cited information that suggested that in some regions of the USA, 60–80% of the mercury deposition is attributable to US sources. Stadler argued that sources outside the USA are not significant contributors to local deposition and that communities across the country may only see local mercury reductions if their upwind neighbours make improvements at their facilities.

Fourteen individual states were also involved in the appeal against CAMR and together they submitted a petition to the US EPA protesting that the CAMR was not sufficiently stringent. These states were: New Jersey, California, Connecticut, Delaware, Illinois, Maine, Massachusetts, New Hampshire, New Mexico, New York, Pennsylvania, Rhode Island, Vermont and Wisconsin (US EPA, 2007a,b,c).

Under the Clean Air Act Amendments of 1990, individual states may promulgate their own emissions regulations as long as they are more stringent than those set at the national level. As a result, several individual states have set their own mercury legislation for coal-fired facilities. As of January 2008, all large coal facilities in Massachusetts will have an emission limit of 0.0075 lb/GWh (0.0034 kg/GWh) for mercury, equivalent to an 85% mercury removal rate. After January 2012, this limit will tighten to 0.0024 lb/GWh (0.0011 kg/GWh), equivalent to a 95% mercury removal rate.

It is thought that existing and recently installed pollution control equipment for SO_2 and NOx will mean that plants will be able to meet the new limits without additional mercury controls. However, the tighter limit in 2012 is likely to require additional mercury control technologies.

A 90% mercury removal rate (equivalent to a limit of 3 mg/MWh) is already applicable in New Jersey (since December 2007) and in Connecticut (since January 2008; expressed as a limit of 0.6 lb/MBtu). Minnesota, Indiana, Maryland, Montana, New York, Ohio and Virginia have also proposed new legislation specific to mercury reduction from coal-fired plant but, as yet, the legislation is not promulgated (EERC, 2005a).

Those states mentioned above that have set mercury legislation distinct from CAMR are likely to keep their current legislation in its existing format. However, any aspects of state regulations that reference CAMR will need to be amended. The emission standards and monitoring requirements of these states are summarised in Table 3.

The fact that these states are determined not only to meet but to surpass the mercury control requirements suggest that the impetus is there for mercury reductions target to be met. However, Poulson and others (2003) argue that the disparate state programmes in the USA will be costly for the power industries as well as the states' economies. They suggest that only a co-ordinated national approach to multi-pollutant standards will be successful and economic.

It has been reported that utility companies in the USA are 'disappointed' with the court ruling against CAMR, with many sources regarding the original rule as 'reasonable'. The legal arguments may result in significant delay to reductions in mercury from this sector (Energy Central, 2008).

It has been suggested that the EPA has only a few options on how to resolve the issue and replace CAMR. The first option is to set mandatory mercury reductions at coal-fired plants either through MACT requirements, strict emission limits or a combination of both. There is also the option of a broader, multi-pollutant, approach that would cover all emissions from coal combustion, including CO_2 (Energy Central, 2008). The EPA could also take the legal course of de-listing coal-fired plants from the 112(c)(9) procedures that caused the negation of the CAMR. However, if de-listing is not attained then new emission standards or requirements must be promulgated under section 112 (the MACT rules). It is unlikely that any decision will be made on the new rules until after the Administration change in 2009, although a bill has already been proposed that would require the EPA to propose new control standards no later than 1 October 2008 which would include a proposal for a 90% mercury reduction requirement (Levin and Yanca, 2008).

It is possible that the EPA will prepare MACT rules based on a similar proposal that was originally submitted in 2004 as an alternative to the CAMR. The proposed rules set emission limits for existing and new plants which varied with coal type and were based on a lb/Btu or lb/MWh basis, such as those in Table 3 (Levin and Yanca, 2008).

Table 3 **Emission standards and monitoring requirements for US states that promulgated rules independent of CAMR** (Levin and Yanca, 2008)

State	Emission standards	Monitoring requirements
NJ	either 90% removal efficiency of an emission standard of 3 mg/MWh by 15 December 2007	CEMs* or 3 tests/quarter
	agreement to implement multi-pollutant approach by 15 December 2008	
MA	either 85% removal efficiency or an emission standard of 0.0075 lbs/GWh (0.0034 kg/GWh) by 1 October 2008	CEMs, sorbent traps or other devices by 1 January 2008
	either 95% removal efficiency or an emission standard of 0.0025 lbs/GWh (0.0011 kg/GWh) by 1 October 2012	
	reductions include both vapour and particulate phase mercury	
MD	either 80% removal efficiency, or an equivalent emission standard in the order of ounces/trillion Btu heat input, or as measured in lbs/y by 1 January 2010	CEMs by 1 January 2010
	removal efficiency of 90%, or an equivalent emission standard in the order of ounces/trillion Btu heat input, or as measured in lbs/y by 1January 2010	
CT	removal efficiency of 90% or an emission standard of 0.6 lbs/tBtu of heat input by 1 July 2008	Quarterly tests CEMs by 1July 2008
WI	removal efficiency of 40% by 1 January 2010	Annual test
	removal efficiency of 75% by 1 January 2015	
NH	removal efficiency of 80% with scrubber technology by 1 July 2013	Bi-annual test until CEMs 'become available'
DE	removal efficiency of 80% or emission standard of 1.0 lbs/tBtu heat input by 1 January 2009	CEMs required if the emission standard is used. Otherwise 3 stack tests/quarter
	removal efficiency of 90% or emission standard of 0.6 lbs/tBtu heat input by 1 January 2013	
CEMs Continuous emissions monitors		

With the negation of the CAMR, there are now no US federal regulations that currently specifically require mercury monitoring and reduction at coal-fired plants. According to the National Resources Defence Council (NRDC) there are 32 coal-fired plants in 13 US states who are already in construction to install control technologies to comply with the now negated CAMR. This could cause major financial and operational problems for some of these plants until the new EPA regulations are formulated (NRDC, 2008). However, some of these plants are still bound by the regulations outlined in Tables 2 and 3.

Although CAMR is being reformatted, the CAIR stands. This means that there will still be an expected reduction in mercury emissions as a result of co-benefit effects from the installation of new SO$_2$ and NOx control technologies. Mercury monitoring will continue at most US plants under either CAIR or regional regulations (Levin and Yanca, 2008).

Even in the absence of the CAMR, the US DOE is continuing its investment in the development of clean coal technologies, including mercury control. This is discussed in more detail in Section 3.2.

2.3.2 Canada

The coal-fired utilities in Canada are the largest source of

mercury emissions in the country. The CWS consists of two targets (CCME, 2006):

● provincial caps on mercury emissions from existing coal-fired plants with the 2010 provincial caps representing a 60% national capture rate of mercury from coal burned or 70%, including recognition for early action;
● capture rates or emission limits for new plants, based on best available control technology (BAT), effective immediately.

The provincial caps for annual mercury emissions are shown in Table 4. There may be a second phase of the CWS which would explore the capture of 80% or more of mercury from coal burned from 2018 and beyond. It is also noted that the federal government in Canada will *'aggressively pursue further reductions in the global pool of mercury'* (CCME, 2006).

Any new coal-fired facilities to be built in Canada will have to meet stringent limits for either percent mercury capture or emission rate, as shown in Table 5. Of note is that Canada's allowable new source emission rates for mercury are more stringent than that specified in the USA (as shown in Table 2) on an energy equivalent basis.

Saskatchewan had already taken action as early as 2004 towards mercury reduction, and this will help the area reach the caps for years 2010–13. The early actions included early

Table 4	Provincial caps for annual mercury emissions for coal-fired plants in Canada (CCME, 2006)		
Province	Estimated emissions, kg/y	2010 cap, kg/y	Coal-fired units as of 2007
Alberta	1180	590	7 - Sheerness, Battle River, Genesee, Sundance, Keephills, Wabamun, H R Milner
Saskatchewan	710	430	3 - Boundary Dam, Poplar River, Shand
Manitoba	20	20	1 - Brandon
Ontario	495	0	4 - Atikokan, Nanticoke, Thunder Bay, Lambton
New Brunswick	140	25	2 - Belledune, Grand Lake
Nova Scotia	150	65	4 - Lingan, Point Tupper, Trenton, Point Aconi
Total	2695	1130	

Table 5	Mercury emission limits for new coal-fired units in Canada (CCME, 2006)	
Coal type	Capture in coal burned, %*	Emission rate*, kg/TWh
Bituminous	85	3
Subbituminous	75	8
Lignite	75	15
Blends	85	3
* based on BAT economically achievable		

mercury controls at the Poplar River power station (CCME, 2006). Ontario is aiming for goal of zero mercury emissions from coal-fired utilities. The Lakeview coal-fired plant was closed in 2005 and all coal-fired electricity generation is to be phased out by 2010.

Alberta has committed to reaching its 50% reduction target through the implementation of the Clean Air Alliance Electricity Project team recommendations, based on a 90% capacity factor. Table 6 shows a comparison between the

CWS, the USA's now vacated CAMR (*see* Section 2.3.1) and the action being taken in Alberta. The CWS and the original CAMR are similar in many ways, requiring a cap-and-trade approach to a total mercury reduction of a specified value within a specified time-frame. Alberta has established its own regulations over and above the CWS by passing the Mercury Emissions from Coal-fired Power Plants Regulation in March 2006. All coal-fired power plants in Alberta had to submit a proposal by 1April 2007 describing how they would achieve a minimum 70% mercury (based on coal mercury content). Actions being taken by certain utilities in Alberta to comply with this regulation are discussed in more detail in Chapter 6.

2.3.3 Japan

Japanese legislation tends to be in the form of negotiation and consensus building rather than punitive actions. Voluntary measures play a significant role in Japan's pollution control strategy. Guidelines based on BAT are developed for non-regulated pollutants. It is therefore likely that utilities within Japan will negotiate voluntary agreements with the government prior to any legislative action. Based on past sexperience, any legislation or action for mercury would be very tight (Newman-Sutherland and others, 2001). Emission

Table 6	Comparison of US EPA CAMR, CWS, and regulations in Alberta (Omotani, 2007)		
	US EPA CAMR*	CWS†	Alberta
Current emissions	48 t/y	2.70 t/y	1.18 t/y
Future estimated emissions	Phase 1 38 t/y	1.13 t/y	4.43 t/y
	Phase II 15 t/y		
Estimated reduction	Phase I 21%	52%	63%
	Phase II 69%	(58%‡)	
Implementation date	Phase I 2010	2010	2010/2011
	Phase II 2018		
Future plans	None beyond Phase II	80% for 2018 and beyond?	optimisation plan, 80%
Mechanism	Cap-and-trade	provincial caps	% reduction
* US EPA's Clean Air Mercury Rule (*see* Section 2.3.1)			
† Canada Wide Standard			
‡ including recognition for early action			

inventories for Japan are not readily available, especially for non-regulated pollutants such as mercury. However, Japanese coal-fired plants tend to be efficient and well-maintained with ESP, FGD and SCR being routinely installed on all units. It is therefore likely that, due to co-benefit effects, emissions of mercury are already being reduced by 40–70%, maybe more. At the moment, it is not known if Japan is proposing any 'more stringent' mercury specific or multi-pollutant limits or reduction targets so it is not possible to estimate whether mercury emissions in Japan will decline, stabilise or even increase in future.

2.3.4   Australia

Although most of Australia's coal-fired units are not fitted with FGD, the mercury content of Australian coals is generally regarded as low (*see* Section 5.1). Mercury emissions are monitored under the National Pollution Inventory (NPI). The implementation of pollution legislation in Australia is at the State rather than the Federal level. A new Air Toxics NEPM (National Environmental Protection Measure) has been proposed by Environment Australia which includes mercury. This may eventually culminate in a load-based fee for mercury (Newman-Sutherland and others, 2001). However, it is unlikely that this would happen in the immediate future as there are no recorded problems of mercury pollution in Australia and therefore not the same level of concern in the community over mercury emissions that there is in the USA.

2.3.5   Other countries

As discussed in Section 2.2, countries in Europe must comply with the LCPD and IPCC and are therefore finding mercury reduction is happening as a co-benefit effect. Mercury specific legislation at large-scale coal-fired plants in the near future is therefore unlikely, although the EC is keeping a close watching brief on the issue. Some European countries, notably Sweden due to the mercury pollution in Swedish lakes, have their own action plans to reduce mercury. These tend to concentrate on sources such as chlor-alkali plants, waste incinerators and end-product uses of mercury such as dental amalgam, thermometers, batteries and light switches.

When global mercury issues are discussed, emissions from developing countries such as those in Asia are usually of most concern. The rapid growth in coal consumption in these areas will inevitably lead to an overall increase in global emissions of mercury. As mentioned previously, data on mercury emissions from countries such as those in Asia, Africa, Indonesia and South America are sparse. Without accurate data it is not possible to determine accurately the most important sources or the extent of the mercury problem in these areas. As a result, it is not always evident which sources would be the most appropriate for effective mercury control. The UNEP mercury programme has developed a toolkit for the identification and quantification of mercury releases (UNEP, 2005). The toolkit is being field tested in Burkina Faso, Cambodia, Chile, Madagascar, Pakistan, the

Philippines, Syria and Yemen. The inventories for most of these countries are expected to be finalised by early 2008. Once the toolkit has been verified, it is planned that it will be used worldwide to produce global mercury inventory data.

Estimates for mercury emissions from China vary enormously due to the dearth of accurate information on coal mercury contents and total coal burn in the different areas. Wu and others (2006) cite values in published literature as low as 214 t/y and as high as 500 t/y for total mercury emissions. Even estimating emission from coal combustion alone is difficult due to the rapid growth in coal combustion in China in the last decade and the still sparse information on coal burn throughout such a large country. Wu and others estimate that emissions from coal combustion have increased from 202 t/y in 1995 to 257 t/y in 2003. The biggest sector with respect to emissions is the industrial sector with emissions increasing from 105 t/y in 1995 to 124 t/y in 2003. During this period, mercury emissions from coal combustion for power generation have increased almost 6% annually to reach a total of 100 t/y in 2003. The growth rate varies from area to area with areas such as Fujian showing a 14% annual growth rate in total mercury emissions where as Guangxi has a lower rate of <2%. Beijing actually has a negative emission growth rate due to faster penetration of control technologies such as ESP and FGD (Wu and others, 2006).

These problems are common to most if not all developing countries. At the moment, China has greater concerns with the major air pollutants such as particulates, SO_2 and NOx. Mercury is therefore somewhat low on the national agenda and is unlikely to be targeted with specific legislation in the foreseeable future.

In the past, emission limits for sulphur from most plants in China were based on factors such as stack height, plant age, location and so on but are generally much more lenient than those seen in, for example, the EU. However, as part of the 2002 *Policy for prevention and control of SO_2 emission pollution due to coal combustion*s, the use of FGD is being promoted in existing coal-fired power plants and the use of low-sulphur fuel is encouraged at smaller plants. All new plants have to meet standards equivalent to those in most EU countries. High quality ESP and FGD must be fitted to all new units and in most cases some level of NOx control must be included. For existing plants there is a programme to upgrade ESP and to fit FGD. As China installs these systems, mercury could be reduced as a result of co-benefit effects (*see* Chapters 4 and 6).

Coal is the main energy fuel in India where over 560 mines are in operation. India is the sixth largest electricity generating and consuming country in the world and the population and energy demand are increasing rapidly (Mukherjee and others, 2008). According to national data, the mercury content of Indian coals is 'very low' (<0.1 ppm). The high ash content of the coal is reported to capture 'most' of the mercury in the fly ash, with emissions being below the detection limit. In order to verify this, further test work on mercury emissions is being undertaken by the BHEL Tiruchi facility (Sachdev, 2008). Again, as with China, the emphasis

is likely to be initially on more traditional pollutants such as particulates, SO_2 and NOx rather than mercury in the near term. Legislation for SO_2 emissions in India are largely based on minimum stack height and there is currently no legislation for NOx (Sloss, 2003). There is therefore currently little or no co-benefit mercury removal in India and none expected until new legislation or action is introduced.

2.4   Comments

Coal combustion is a significant, if not the major, source of mercury emissions in many countries. Some areas in the developed world, such as the EU, have managed to reduce mercury emissions significantly (around 70% in the last three decades) due to measures taken to reduce emissions of other pollutants such as SO_2 and NOx. These reductions are expected to continue even without mercury-specific legislation due to further co-benefit effects from existing and impending legislation. Mercury-specific legislation has been set in Canada and certain states in the USA which will required the installation of mercury specific control technologies. Although some of this reduction will be achieved through co-benefit effects, mercury specific control technologies are increasing in demand. Despite the vacation of CAMR in the USA, which would have specifically targeted mercury, there is already impetus for mercury control at many plants. What form the replacement regulation for mercury in the USA will take remains to be seen.

China and India are increasing their coal utilisation at a dramatic rate. Legislation for pollutants such as particulates, SO_2 and NOx will take priority over potentially more costly requirements set specifically for mercury. However, some mercury reduction will be achieved as a result of co-benefit effects from such legislation.

3 Evaluating the economics

Before considering the options for mercury control in Chapters 4 and 5 it is important to understand the complexity of evaluating the economics. Controlling mercury emissions globally will incur costs at many levels, from the expenses associated with establishing and enacting relevant legislation and action plans, down to the cost for the control technologies themselves. However, there will also be benefits in terms of improved health and ecosystems. The sections below summarise the different costs and benefits relevant to mercury emissions from coal combustion.

3.1 Cost of legislation

Establishing legislation to reduce emissions requires a significant amount of preparation and interaction and negotiation between individuals, organisations and governments even before the legislation is enacted. Examples of the cost for establishing international agreements within the UN are summarised in Section 3.1.1. The section also includes a discussion of how funding mechanisms within such agreements can assist developing countries. Section 3.1.2 then discusses the costs and benefits of the CAMR in the USA. Although the CAMR rule has been vacated it was the first and most extensive national legislation in the world specifically set to control mercury emissions from coal-fired utilities and therefore the economic analysis of the rule is still of relevance when considering mercury control strategies.

3.1.1 International agreements

Establishment of a multinational agreement, legally binding or otherwise, is not inexpensive. For example, the cost to the United Nations of negotiating the Rotterdam Convention was around US$3.2 million, the Stockholm Convention was just under $6 million; and the non-legally binding Strategic approach to International Chemicals Management (SAICM) was US$6.5 million (UNEP, 2007b).

Costs for legally binding agreements, largely for administrative services within the secretariat, can also be significant, for example (UNEP, 2007b):

		Annual budget (US$) 2006
Climate	Bonn	33,554,365
Ozone	Nairobi	3,323,532
Basel	Geneva	4,121,940
Rotterdam	Geneva/Rome	3,710,224
Stockholm	Geneva	5,275,636

This annual budget does not include the initial negotiation costs for preparatory meetings and so on. For example, the Stockholm Convention required eight major meetings and cost US$6 million (not including hosting costs, participatory costs or UN security costs). Even if mercury is adopted within one of the above conventions, as discussed in Section 2.1.2, it

is likely that any new legislation for mercury could require a new secretariat within UNEP.

The cost to individual countries of any international agreement will be significantly different depending on whether the agreement is legally binding or voluntary. Voluntary approaches allow a significant amount of flexibility which could reduce costs. However, legally binding approaches lead to a 'level playing field' in which specific economic sectors are subject to similar controls, regardless of where the economic sectors are located. This can also discourage 'free-riding', stimulate innovation and foster a faster global transition to alternative processes and techniques (UNEP, 2007b).

The effectiveness of an international agreement, voluntary or legally binding, is closely linked to the availability of financial and technical assistance to aid implementation. Some multilateral environmental agreements (MEAs) contain provisions for mandatory financial mechanisms. For example, the Montreal Protocol is regarded as successful partly because it includes a mandatory multilateral fund. This ensures the funding of the included mechanisms which greatly assists developing countries with compliance. Mandatory financial mechanisms are only found in legally binding instruments. Whilst they may then incur significant costs for donor States, they do tend to determine whether ambitious, legally binding approaches are possible (UNEP, 2007b). A legally-binding agreement would also require that developing nations are given access to technologies for mercury reduction or control which would have a technology-forcing effect. This, in turn, could lead to faster development of more cost-effective mercury control technologies.

Enhanced voluntary measures could include international action through technology transfer and assistance, including partnerships. This would obviously have a significant cost factor. To overcome any potential cost barrier, firms and Governments of developed countries can be assisted in funding environmentally sound technologies through grants, concessionary loans, export credits and guarantees. At the moment, the GEF (Global Environment Fund) may be limited in its ability to support mercury-reduction technology, this may change in the future after the next replenishment (UNEP, 2007b).

3.1.2 CAMR

As discussed in Section 2.3.1, the USA was the first country to set emission legislation which is specifically applicable to emissions from coal-fired utilities. The legislation proved to be highly controversial and was eventually vacated in February 2008. However, before this occurred there were in numerous reviews published of the economics involved. These economic studies have taken into account the benefits of mercury control (such as improvements in health) as well as the costs for the technologies required to reduce emissions

and are therefore still valid despite the demise of CAMR in its current format.

Even without CAMR, the first phase of the mercury reduction required in the USA is expected to be a result of co-benefit effects from the Clean Air Interstate Rule (CAIR). The US EPA have reported that the upper bounds of benefits resulting from the removal of mercury emitted from coal-fired power plants after implementing CAIR is $210 million per year (Griffiths and others, 2007).

The benefits of mercury control under CAIR and CAMR (or its replacement) will be, it is hoped, a significant reduction in mercury emissions with a resulting decrease in mercury concentrations in the environment. In turn, this should lead to improvements in mercury-related health effects such as neurological disorders, learning difficulties and developmental delay. These types of cost-benefit analyses are generally quite complex and are based on assigning a monetary value to human health effects. For example, the studies involving the CAMR assumed that the cost to an individual of each lost IQ (intelligence quota) point is around $3000. It is therefore not surprising that values for the benefits of actions such as the CAMR were open to debate and criticism. For example, Griffiths and others (2007) compared the monetised impact of IQ (intelligence quota) detriments from mercury emissions. These calculations have to take into account an estimate individual dose, at birth (pre-natal). There then has to be an assumed effect and a resultant reduction in lifetime earnings due to this effect. Griffiths and others (2007) provide a comprehensive guide as to how these calculations are performed and why estimates from the US EPA differ from other published estimates.

Stadler (2005) has listed several of the most noted reports on how much mercury pollution costs in the USA:
● the Harvard School of Public Health has stated that a 70% reduction in mercury emissions by 2018 would result in up to $5 billion in health benefits annually. The US EPA estimated a lower value of $50 million annually;
● a report from Mount Sinai estimated that mercury exposure could cost an estimated $8.7 billion annually in lost earnings due to lower intelligence from mercury exposure;
● an EPA commissioned study found that the Southern USA alone could see $2 million annually in public health benefits from cutting mercury emissions across the country.

Hylander and Goodsite (2006) review the environmental costs of mercury pollution such as damage costs in the Arctic and remediation of contaminated sites. Effects on wildlife, ecosystems and recreational fishing would also be included in benefit analyses. A decrease in mercury pollution in surface waters would bolster fishing and tourism industries.

Any new mercury legislation will mean the requirement for the installation and monitoring of control systems which, in return, will lead to new industries and employment opportunities. It is estimated that the manufacture and installation of pollution control equipment could create

300,000 jobs over the next decade in the USA alone (NWF, 2004).

It is the balance of costs versus benefits which determine the economic success of the legislation. The costs of achieving mercury reductions depend on the form as well as the stringency of any regulation. The US EPA estimated that the total cost of complying with CAMR would range from $0.16 to $0.8 billion, and even then the EPA control costs could be overestimated by as much as a factor of four, since the technologies reduce in cost as they become more commercialised (Stadler, 2005). A report to the US Congress in 2004 stated that the total quantifiable benefits of the EPA legislation were estimated at more than $15 billion annually. This is around 16 times the estimated cost of compliance. Stadler (2005) agrees that the benefits greatly outweigh the potential costs.

Palmer and others (2007) used a model based on data from the electricity sector, atmospheric transport and environmental and public endpoints affected by pollution. The model concluded that the economic benefits of CAIR and CAMR would be far greater than the costs, even under cautious assumptions. However, it was stressed that the analysis could not provide economic advice regarding mercury emission targets since the modelling reflected inadequate links between models of emissions, physical science, public health and social science. Although the model could provide details on changes in particulate matter as a result of mercury reductions, they could not provide direct accounting of the benefits of reduced mercury emissions.

CAMR was predicted to save $9 billion (in 2004 dollars) up to 2020 because of the flexibility of the trading scheme (McManus and others, 2005). However, the trading option within CAMR was one of the main arguing points for many organisations who opposed it. Palmer and others' (2007) model could not cover the implications of mercury trading – that is the potential creation of mercury hot spots or local concentrations of mercury that could lead to particular adverse health effects. It is alleged that these situations could substantially offset the cost savings from allowing mercury trading.

Palmer and others (2007) considered four scenarios for SO_2, NOx and mercury control:
● baseline (existing legislation continues);
● CAIR plus EPA mercury (including the mercury cap-and-trade scheme under CAMR) but no seasonal SIP (state implementation plan) for NOx reduction;
● CAIR plus EPA mercury plus SIP NOx reduction (which would have been suspended under CAIR);
● CAIR plus tighter mercury with MACT – includes an additional target of 90% mercury reduction or <0.6 lbs Hg/million Btu (0.28 kg/GJ). This approach has been proposed by several independent advisory committees but has not been adopted by the EPA.

The 'CAIR plus EPA mercury' options represent the same approach as was proposed under CAMR. The results from the model under the different scenarios are summarised in Table 7. Palmer and others (2007) used their model to

Table 7 Overview of electricity price, generation and new capacity for 2010 and 2020 (Palmer and others, 2007)										
	Baseline		CAIR plus EPA mercury cap		EPA mercury cap and seasonal SIP NOx policy		Tighter mercury with MACT		Tighter mercury with trading	
	2010	2020	2010	2020	2010	2020	2010	2020	2010	2020
Average electricity price, 1999 $/MWh	61.9	68.6	62.8	69.3	62.7	69.3	63.2	69.9	67.3	73.7
Coal generation, billion kWh	2326	2618	2271	2556	2257	2536	2283	2538	1960	2206
New capacity, MW*	2226	30,650	2286	28,590	1751	26,860	2047	27,620	3273	33,440
National emissions: tons tonnes	53 47.3	53.5 47.7	30.57 27.29	24.58 21.95	30.57 27.29	24.99 22.31	9.50 8.48	8.17 7.29	9.63 8.60	8.23 7.35
Allowance prices, $/lb ($/kg)			80,930 (36,786)	36,040 (16,382)	77,980 (35,445)	35,760 (16,255)	–	–	721,800 (328,091)	1,429,000 (649,545)
* includes all investments after 1999										

determine the increase in electricity cost to the consumer as a result of the actions required at coal-fired plants to meet the new legislation. The model suggests that the US EPA's CAIR and CAMR would have 'fairly small' impacts on the average price of electricity even if the tighter mercury MACT policy were introduced. The increase in costs from the average energy price in 1999, in $/MWh was less than $10 in each of the scenarios but highest in the scenario which required tighter mercury reduction through trading. The tighter mercury reduction scenario would also result in a significant reduction in coal burn, as shown in the table, and a concomitant increase in the amount of new capacity of utilities required. Impacts on electricity prices are discussed in Section 3.2 *below*. However, if trading were used to achieve 90% mercury reduction equivalent, this would lead to a 10% increase in electricity cost in 2010 and a 5% increase in 2010, relative to the baseline scenario. The estimated emission allowance for mercury in 2020 would be around 36,000 $/lb (16,364 $/kg), which is close to the EPA's forecast of 35,000 $/lb (15,909 $/kg).

Rezek and Campbell (2007) summarise the different mathematical approaches used to estimate the marginal abatement costs for different legislative approaches to controlling emissions of multiple pollutants. The authors then go on to propose a new method based on *inequality constrained generalised maximum entropy estimation.* Estimates for the cost of mercury abatement using the different models ranged from 467 $/ounce (13,076 $/g) to 1089 $/ounce (30,492 $/g). From this, Rezek and Campbell (2007) suggested that the cost savings from using a trading approach to mercury control would be lower than that for SO_2 but higher than for NOx. The original SO_2 trading scheme in the USA was extremely successful and cost effective.

Although it would be interesting to compare the different cost analyses above, this would not be prudent since they are based on different types of analyses and different assumptions. However, it is safe to say that most if not all of the studies concluded that the benefits of mercury control far outweigh the costs. However, the more stringent the

legislation, the greater the increase in electricity costs and the more likelihood of a move away from coal to alternative fuels.

3.2   Cost of technologies

Estimating the cost of mercury control is complex. Assumptions must be made about average mercury capture rates, equipment and operating costs and so on. Srivastava and others (2005) give an excellent, detailed account of how such cost estimates are prepared and the assumptions made. It is simply not possible to take into account the different coal types, combustion conditions and plant characteristics of each plant and so general assumptions must be made on mercury behaviour in a typical coal-fired power plant. For example, for US EPA model plant application estimates, it was assumed that the average mercury concentration in coal is 0.10 mg/kg for eastern bituminous coal and 0.07 mg/kg for subbituminous coal. Srivastava and others (2005) used a complex combination of coals, existing control technologies, plant types and so on to estimate mercury control costs at different plant types burning different coals. These results are discussed in more detail in Chapter 5.

The most-cited cost comparison data is that produced by the US DOE/NETL for their extensive multi-phase study on mercury control options for coal-fired utilities. The economic analyses were performed on a plant-specific basis, dependent on operating conditions and coal properties observed during full-scale testing. The DOE analyses were performed at three different scenarios – low (50%), mid (70%) and high (80–90%) mercury control. This level of control was calculated above the plant-specific baseline mercury removal, and therefore only evaluated the mercury removal directly attributable to activated carbon or similar control technologies (Jones and others, 2007).

The cost estimates were expected to be ±30% accurate. This is similar to the accuracy of the rough-order-of-magnitude (ROM) costs or 'study' level costs acceptable for regulatory development as described by the US EPA. The inaccuracies in

the cost estimates derive from the uncertainty of a number of assumptions regarding the installation and operation of full-scale systems. Inaccuracies in the mercury estimates were due in turn to the inherent inaccuracy of mercury monitors working at low concentrations of mercury. Further inaccuracies arise due to estimates and assumptions made in the cost budget, such as (Jones and others, 2007):

- exclusion of monitoring costs;
- retrofit of equipment is assumed to be uncomplicated;
- retrofit of the activated carbon injection system occurs during a scheduled plant-outage, ie no loss of plant revenue;
- no balance-of-plant impacts due to installation of the system.

Other cost effects, such as loss of revenue from ash sales (due to possible contamination of the fly ash with carbon, depending on the system) were also included. Details of the estimated costs of different control technologies as estimated by the DOE study are given in Chapter 5.

Jones and others (2007) emphasised how, for a given level of mercury control, a single parameter such as coal mercury content can give a wide range of incremental costs. The incremental cost of mercury control is inextricably linked to the specific assumptions used in the development of the particular cost estimate. Therefore any comparison of one estimate with another should be conducted *cautiously and with a clear understanding of the context of the specific application*.

CURS, the Center for Urban and Regional Studies at the University of North Carolina, USA, has prepared CUECost, a spreadsheet model for calculating the costs of control technologies at coal-fired utilities (CURS, 2007). CUECost produces rough-order-of-magnitude (±30% accuracy) of the installed capital and annualised operating costs for air-pollution control systems. The system currently covers the most commonly-available control technologies: FGD (limestone with forced oxidation, lime spray-drying and so on), ESP and baghouses/fabric filters, SCR, SCNR, natural gas reburning and low NOx burners. It also covers mercury control using powdered activated carbon (PAC) injection.

In the CUECost programme, the effects of existing equipment (such as ESP or FGD) on mercury reduction are kept separate from those due to the PAC. The algorithms used in the programme were based on the results of two recently completed full-scale demonstrations as well as algorithms developed from pilot-scale data. Default criteria are included for all input parameters and these are based on a generic 500 MWe coal-fired power plant located in Pennsylvania, USA. Different parameters for larger or smaller plant may be input as necessary. A coal library is also included so that a coal similar to the actual coal burn may be selected if an actual ultimate analysis is not readily available.

CUECost can be downloaded from the US EPA website (www.epa.gov) and is supported by Windows XP or higher operating systems. The system is designed to be user-friendly allowing a step-by-step input of data relating to the following parameters:

- general plant technical input (including boiler operation, coal analysis, excess air);
- economic input (including inflation/discount/escalation);
- indirect cost rates (including engineering, general facilities, contingency);
- fixed cost factors (including maintenance %, operating labour);
- air pollution control system specific technical input (including operating criteria, equipment sparing);
- retrofit factor.

For the mercury-specific control technologies, the system requires data on the type of sorbent used and the operating conditions of the particulate control system used to capture the sorbent. 'Learning rates' are derived from capital and operating and maintenance costs based on historical cost estimates reported in the literature. The learning rates represent the fractional reduction in cost associated with each doubling of cumulative total capacity of the technology. The application of the learning rates allows projections for future cost reductions by 2020. Sorbent injection technologies are expected to incur a 28.72% reduction in capital cost by 2020 as compared to older technologies such as wet FGD (LSFO, 9.9%), dry FGD (LSD; 20.4%) and SCR (7.4%). However, operating and maintenance costs are predicted to remain unchanged to 2020.

The mercury control spreadsheet of CUECost includes co-benefit controls from air pollution control systems for particulates, SO_2 and NOx as well as sorbent-based mercury specific controls. The estimate includes the impact of the 'parasitic load' of the sorbent adding to the particulate mass as well as filter replacement costs. It also includes the impact the sorbent may have on fly ash marketability when the sorbent is captured in the existing particulate control device.

Since there are few data available from commercial operation of sorbent injection, the costs are based on estimates and projections from pilot studies. Table 8 shows the estimated costs of different systems calculated by CUECost based on a 500 MWe plant firing Wyoming Powder River Basin (PRB) coal, with activated carbon injection, a capacity factor of 65% and 80% mercury removal (above baseline). From the data in the table it is evident that the most economic option for this plant would be activated carbon injection with a dry scrubber and a fabric filter. More details of the assumptions made for these estimates can be found in the original document on the US EPA website.

As with any new technology, over time the system improves in design and efficiency and the cost comes down. Technologies being developed for the reduction of mercury emissions from western coals (those for which the mercury chemistry is the most problematic) were reduced in cost by a factor of four or more even before 2004 (Foerter, 2005). Estimates prepared before 2005 by both the US EPA and US DOE put mercury emission reduction costs at 1–3 $/MWh (Foerter, 2005). According to the ICAC, between 2004 and 2005, the cost of activated carbon injection systems dropped by a factor of four (Stadler, 2005).

Figure 2 (NWF, 2005) shows the rapid reduction in estimated costs of mercury removal between 2001 and 2006. Although

Table 8	Estimated cost of activated carbon injection control (ACI) systems in CUECost (CURS, 2007)				
Equipment configuration	Capital, 2005 $/kW	Operating and maintenance, 2005, $/MWh	Hg removed by sorbent injection, lb/y (kg/y)	Control cost, 2005 $/lb Hg removed, ($/kg)
ACI + Cold-side ESP	19.41	4.06	240.7 (109.4)	53,380 (117,436)
ACI + Cold-side ESP + Wet FGD	19.41	4.06	188.9 (85.9)	68,013 (149,629)
ACI + dry scrubber + fabric filter	3.17	0.32	290.7 (132.1)	3,844 (8,457)
Based on: 500 MWe plant, Wyoming PRB coal, active carbon injections, capacity factor of 65%, 80% mercury removal				
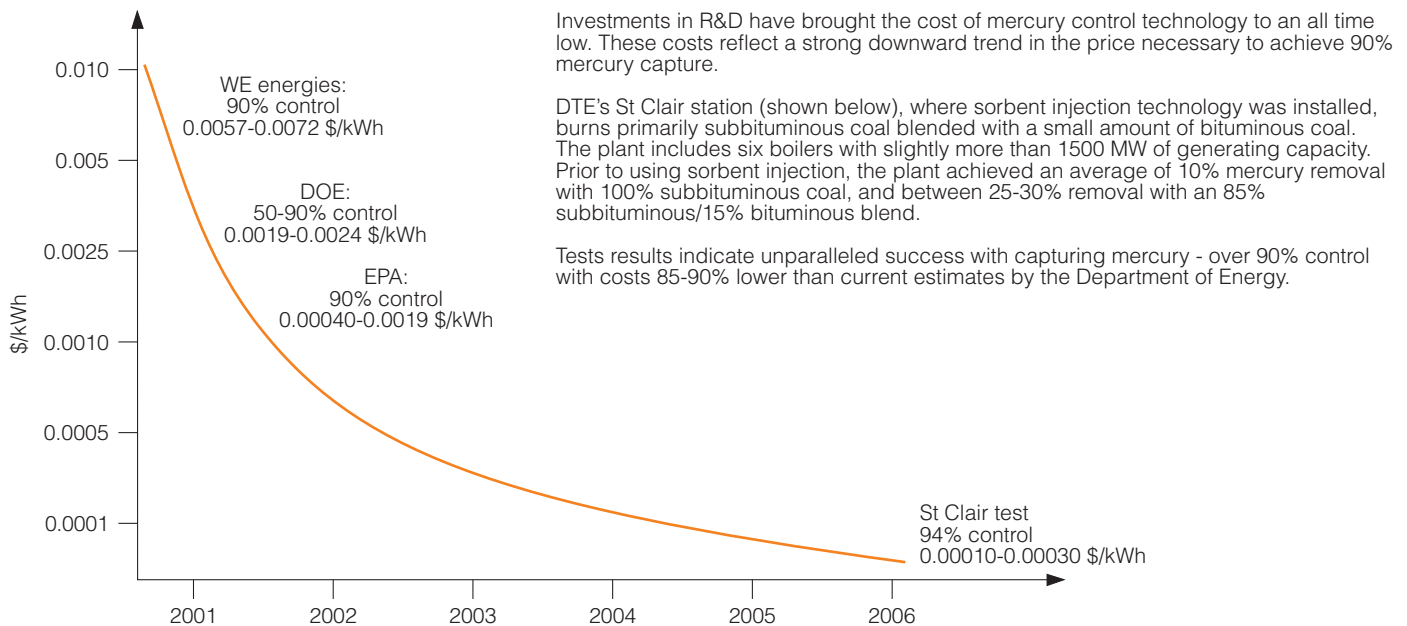
Figure 2   **Reducing the cost of mercury control** (NWF, 2005)

the data used in the graph are somewhat selective of the most successful and economic tests run under the US DOE's mercury programme, they do give a general indication of how quickly costs can be reduced as technologies pass through development and testing stages and into the commercial market place.

The National Wildlife Federation in the USA (NWF, 2004) produces opinion-based documents on the effect of mercury in the environment and have published reports which aim to express the costs and benefits of mercury control in ways that the general public can understand. Although it could be argued that the results may be an over-simplification of the issue, the ball-park figures produced are interesting. Based on results from some of the US DOE's mercury studies at full-scale plants, the NWF estimate that 90% mercury control at 'every' plant in the USA would cost the average household from around $0.70 to a little over $2.00 per month, depending on the state. The study was based on a number of general scenarios which were based on results from successful mercury reduction at plants already fitted with ESP, FGD and/or SCR systems and from activated carbon demonstration studies. It would seem that the study assumes that mercury control at all plants in the USA will be as cheap and effective as that at those few plants studied already.

The NWF (2004) compared their own estimates with those of the Institute of Clean Air Companies (ICAC) and the US DOE and found them comparable. ICAC estimated an additional 0.1–0.3 ¢/kWh to the average retail customer rate of 8 ¢/kWh. This is around a 1.2–3.7% increase. The US DOE estimate for 2003, for 60–90% mercury reduction from subbituminous coals, was 0.127–0.215 ¢/kWh (NWF, 2004). It is unclear whether the NWF study used the same cost basis and reduction target used by the US DOE for their estimate and therefore the results should not be compared directly. It is possible that the NWF data are based on more dated targets and therefore the US DOE estimates would be more likely to be accurate.

Poulson and others (2003) suggest that the installation of new technologies could mean that plants must come offline. This would trigger capacity shortages and connection outages just before implementation dates. This, in turn, could create a market imbalance driving wholesale energy prices up and possibly leading to power outages. Poulson goes on to discuss in great detail the effect tightening emission legislation can have on the economy – increases in energy prices, and even unemployment with household disposable incomes dropping noticeably. Although the paper by Poulson and others (2003) is somewhat dated as it precedes the final decision on CAIR

and CAMR, it does provide the reader with an interesting summary of how emission legislation can impact the economy.

Much of the initial work on control technologies for mercury in the USA was funded by the US DOE. However, the US DOE reduced the budget for the development of mercury control technologies to zero at the end of 2007. The argument was that CAMR and the subsequent market forces would drive the further development of mercury control strategies and that the DOE itself was no longer required to provide further demonstrations (Feeley, 2008). However, with the CAMR now vacated it is possible that funding for mercury, either alone or as part of multi-pollutant control strategies, could re-appear on future agendas.

Even without the CAMR, the US DOE have many programmes related to clean coal development which will reduce emissions of all pollutants, including mercury, from future utilities. These programmes include:
● Clean Coal Power Initiative;
● Power Plant Improvement Initiative;
● Clean Coal Technology Demonstration Program.

All these initiatives are heavily funded ($2 billion over ten years) to ensure the advancement of clean coal technologies. More details on these programmes can be found at www.netl.doe/technologies/coalpower.

Although the main driver behind the development of mercury control systems was the now vacated CAMR in the USA, it was argued that the long-time frame for the required mercury reduction was actually slowing the development of control systems since the first reduction target will be achieved by co-benefits alone. According to Foerter (2005) CAMR would have delayed any commercial market for mercury specific controls by two or three decades. Foerter argues that, as with utilities, the air pollution control companies' priority will be a return on their investment. Large markets provide greater incentives for development of new or advanced technologies and encourage competition between vendors. This, in turn, results in the development of more innovative and cost-effective solutions. Foerter also believes that the CAMR would have unfairly penalised eastern bituminous coals and could have provided sufficient incentive for plants to switch from eastern to western coals.

Bellas and Nentl (2007) have used a marketing-based analysis to study the predicted adoption of mercury control technologies. A previous study on the characteristics of the utilities and plants that were the 'early adopters' of fabric filters in the USA was used as the basis to predict which plants would be most likely to be the first to adopt mercury control systems. It had been shown that, in many cases, larger firms/utilities are likely to be early adopters of new technologies. Larger firms are also likely to have more operating units and are therefore likely to buy more systems than small companies, simply because they can. Previous data had also shown that investor-owned firms are more likely to become early adopters than publicly-owned firms. More risk-averse firms are less likely to become early adopters. However, the size of the firm may be important as plants

containing more boilers should be less risk averse regarding new technology than plants with only one boiler. Plants with only one boiler would be less willing to attach an unproven technology to their only power source. In the same way, utilities that own multiple plants may be able to substitute the required output from one plant with output from another and are therefore at less risk.

Bellas and Nentl (2007) note that, if mercury regulation were to be through a technology-based standard, that is BAT/MACT, then early adoption of one technology or another might establish the standard for the industry. This early arrival onto the marketplace could give one technology a dominant position in the market. It would therefore be a sensible commercial move for companies designing mercury control technologies to identify the utilities/owners who are most likely to be early adopters. This would ensure their product arrived on the market place early and would therefore have a chance to become the dominant technology.

The model run by Bellas and Nentl (2007) based on fabric filter adoption in the USA concluded that smaller, more 'nimble' utilities were the most likely to adopt new technologies but that they would tend to do so in ways which allowed them to protect themselves from failure. This could be by placing the technologies on smaller boilers within a larger plant. Age of the plant was found to be an important factor in the early adoption process, as was the cost of the installed unit. The earliest fabric filters were often installed on the older units. This may be because the low cost fabric filters were attractive to owners of older boilers that needed particulate control.

The Bellas and Nentl (2007) model also suggested that publicly-sowned and co-op utilities were likely to be early adopters, which disagreed somewhat with the reviewed studies which suggested that investor-owned utilities were the early adopters. From their study, Bellas and Nentl (2007) concluded that certain industrial sectors could be targeted for marketing mercury control systems. The most likely early adopters of mercury control technologies would be:
● smaller firms that have smaller capacity units within larger plants;
● firms that are publicly-owned rather than investor-owned.

The study also concluded that cost does matter and is an important consideration in the early adoption process. Cheaper technologies will sell faster. Bellas and Nentl (2007) therefore suggest a price penetration strategy where a low introductory installation cost could be offered to those plants and utilities which fit the early adopter profile defined above. First-to-adopt firms could be attracted with financial incentives such as low interest loans or deep discount.

3.3   Cost of co-benefit effects

Arguably, mercury reduction through co-benefit effects (the installation of technologies such as FGD and SCR which also reduce mercury emissions) can be regarded as minimal or even 'free'. Mercury emissions from Europe have been reduced by 70% since 1970 due to legislation for particulates,

NOx and SO$_2$ (Sloss, 2003). Arguably, this mercury reduction cost the EU nothing. It is certainly not possible to separate the cost of mercury reduction from that of the other pollutants.

Mercury reduction in the short term in the USA is likely to be achieved through co-benefit effects. Again, it is not possible to determine specific costs for the mercury reduction through this co-benefit.

Ideally, some mercury reduction in other countries such as those in the developing world, will be achieved at minimal or no direct cost as a co-benefit from other approaches to pollution control such as increased fuel efficiency, fuel switching or controls for particulates, NOx and SO$_2$. This is discussed in more detail in Chapter 6.

3.4   Comments

Controlling mercury emissions from coal-fired utilities will incur costs. There will be costs associated with establishing relevant legislation and ensuring the legislation is applied. On a global scale, any action taken to curb mercury emissions is likely to be far more effective if funding is made available to ensure that developing countries can afford to adopt costly strategies or technologies. On a national scale, the new legislation expected in the USA to replace CAMR will be costly but is likely to result in health and environmental benefits which should far outweigh the costs.

Estimating the cost of control technologies for coal-fired plant is problematic since much of the current data are based either on pilot-scale studies or short-term full-scale studies. Further, the variation in mercury behaviour at different plants makes it difficult to make generalised assumptions for cost analyses. As the market for mercury control technologies develops further, prices are likely to drop quite dramatically. Market forces will determine which control technologies become the most popular, with early successful demonstrations having the most opportunity to take a larger portion of the marketplace.

Inevitably the cost of controlling mercury emissions will be passed to the consumer. However, estimates suggest that the costs will be minimal. The 'cheapest' approach to reducing mercury is to maximise the 'free' mercury reduction achieved through co-benefit effects of flue gas technologies installed to reduce emissions of other pollutants. This is discussed in more detail in Chapter 4.

4 Existing technologies and co-benefit approaches

Existing technologies for particulates, SO$_2$ and NOx, such as baghouses, FGD and SCR, can often help reduce mercury emissions as a co-benefit. As mentioned in Chapters 2 and 3, the EU has achieved significant mercury reduction through co-benefits of SO$_2$ and NOx legislation and the USA plans to follow this by achieving significant mercury reductions through co-benefit reductions within the CAIR rule. Taking advantage of co-benefit effects could be a cost-effective way of reducing mercury emissions in developing countries.

The UNFCCC, its Kyoto Protocol and the European Emissions Trading Scheme (ETS) are also having co-benefit results for mercury. Many countries are moving away from coal combustion to fuels and technologies which are not so carbon-intense. The decreased use of coal in these areas will clearly also result in a reduction of mercury. Increased energy efficiency also results in less coal use and lower mercury emissions.

A previous report from the IEA CCC (Sloss, 2002) emphasised the complex behaviour of mercury within coal-fired plants. Mercury emissions depend upon the following factors:
- mercury content of the coal;
- chlorine and sulphur content in coal and their resultant interactions with mercury;
- mercury speciation in flue gas;
- distribution of mercury between the solid and gas phase;
- the unburnt carbon content of the coal.

Plant specific factors include:
- the type of boiler and the combustion conditions experienced by the coal;
- distribution of ash between bottom and fly ash;
- nature and efficiency of flue gas emission control devices;
- interaction between fly ash and mercury;
- temperature effects on flue gas interactions;
- conditions of the emission control equipment and flue gas.

Several technologies which have been applied on coal-fired power plants for years for particulate, SO$_2$ or NOx control have proven successful in removing trace elements, including mercury, for no extra cost. The efficacy of these systems for mercury control is variable but in some situations it is possible, at least theoretically, to adjust the operation of these systems to enhance mercury removal.

Figure 3 shows the layout of a theoretical coal-fired plant fitted with the most common flue gas control technologies. Figure 4 shows the average mercury removal across various pollution control devices based on information from the US EPA's Information Collection Request (ICR). Co-benefit mercury removal with different control technologies are also summarised in Table 9. The mercury reduction efficiency varies not only with the different control technologies but also with the different types of coal. 'Native' mercury capture, as described in the table, applies to baseline mercury capture,
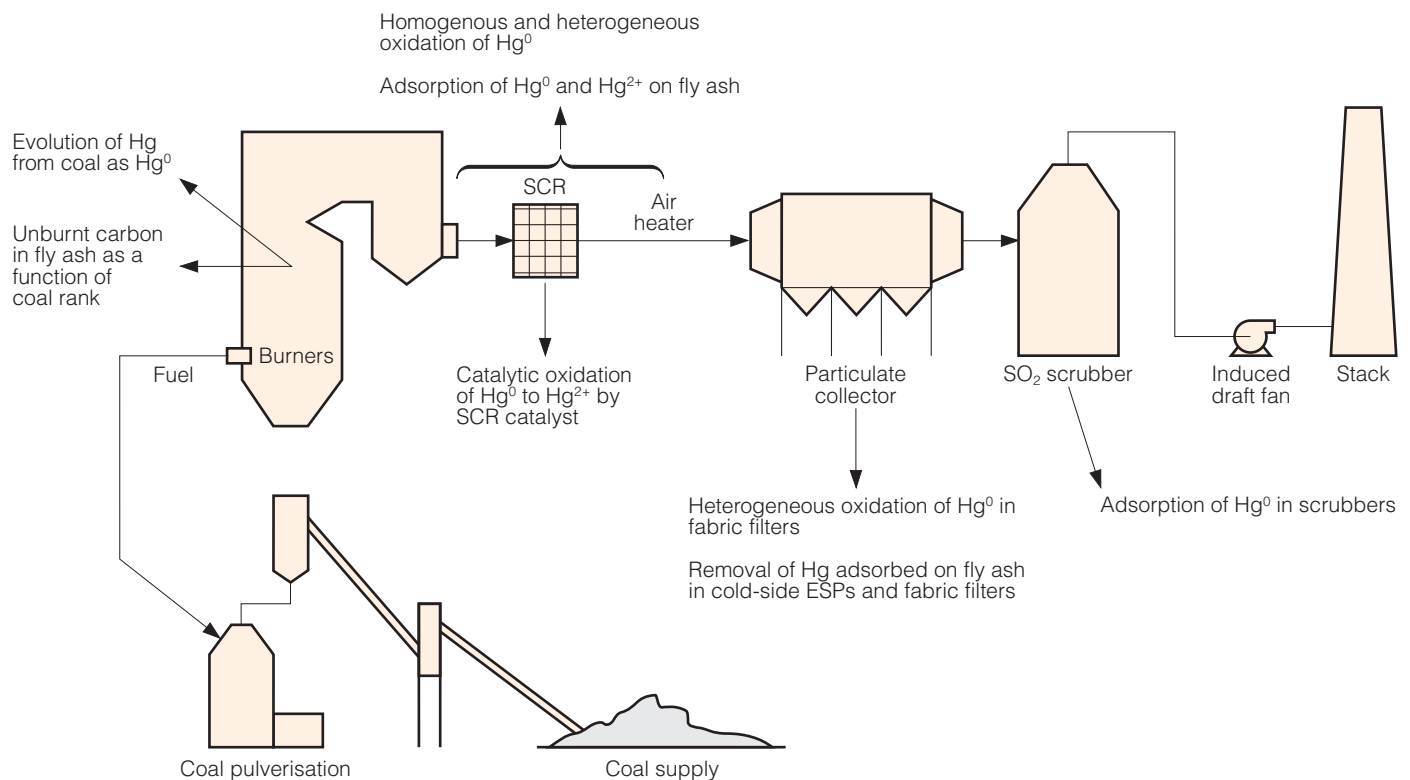
Figure 3   Diagram of a coal-fired power plant illustrating critical pathways for mercury transformations (Kolker and others, 2006)
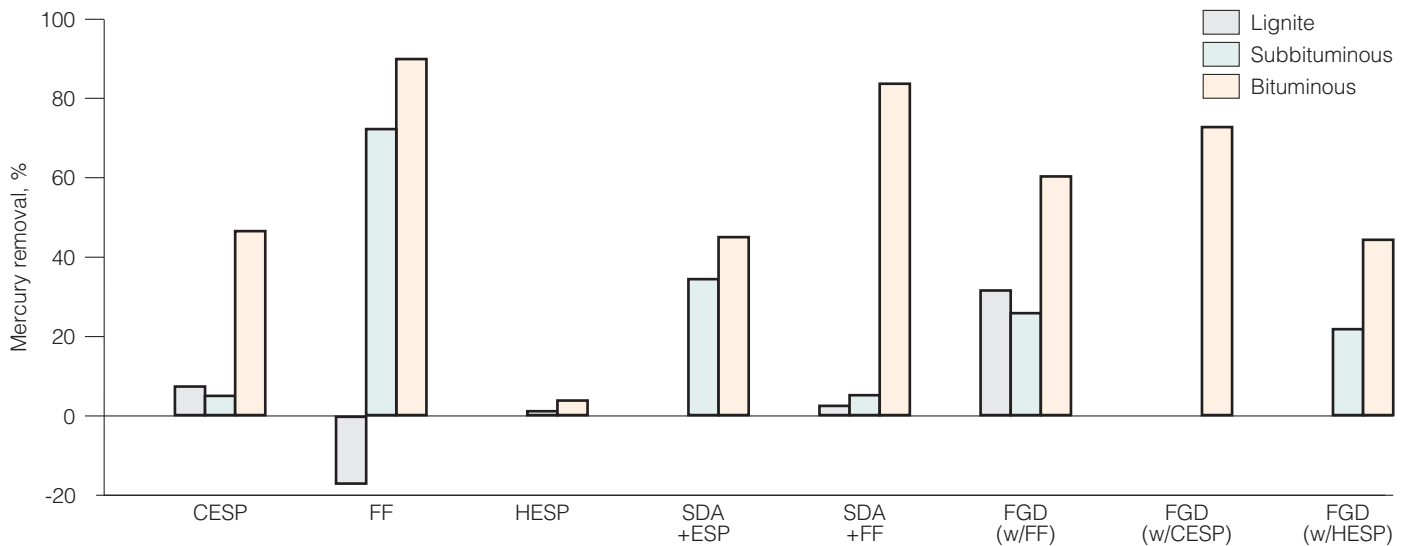
Figure 4   Average mercury removal across air pollution control technologies in coal-fired utility boilers (Kolker and others, 2006)

commonly on the fly ash, that occurs at the plant due to the plant conditions and not as a result of any specific control systems. Mercury capture due to co-benefit effects can range from virtually nil to over 90% (Srivastava and others, 2006). Details of the co-benefit effects are discussed in the Sections below.

4.1   Particulate control systems

There are two main types of particulate control systems on coal-fired plants – ESP (electrostatic precipitators) and baghouses (also known as fabric filters). Approximately 75% of existing coal-fired plants in the USA are equipped with ESP for particulate control. The remainder are fitted with baghouses, particulate scrubbers or other particulate control devices.

On average, cold-side ESP systems (downstream of the air pre-heater, 135–175°C) capture around 30% of the mercury in the coal, the capture rate ranging from zero to over 60% depending on the coal. According to Kolker and others (2006), the mercury emissions from plants fitted only with ESP systems (that is no FGD or SCR) show a direct relationship between coal mercury content and mercury emissions. In these situations, the selection of lower mercury coals would guarantee a reduction in mercury emissions, although this would be limited to an average of 36% of the mercury content of the coal feed. The capture efficiency is higher for bituminous coals than low rank coals due to differences in mercury chemistry between the coal types (see Section 5.1).

Temperature can play an important role in the capture of mercury in particular control systems. The cooler the flue gas, the more likely oxidised mercury will condense or adsorb onto fly ash particles. Hot-side ESP systems (upstream of the air pre-heater, closer to the boiler, 300–400°C) show lower mercury capture (average 3%) than cold-side systems (average around 30%). The longer residence time and cooler temperature in the cold-side systems is more conducive to

mercury adsorption onto fly ash than the shorter, hotter conditions in the hot-side ESPs. High flue-gas cooling rates between the air pre-heater inlet and the air pollution control device inlet can enhance reaction rates associated with oxidation by species such as chlorine (Kolker and others, 2006).

Work at Consol, PA, USA, in 1999 demonstrated that fly ash alone can remove a 'substantial portion' of the flue gas mercury when the gas is cooled below typical exhaust temperatures (from 300°F down to 220°F; 150°C down to 105°C). When the flue gas was cooled to 200°F (93°C), 80–90% of the mercury was removed. However, reducing the temperature in this area of the power plant can cause back-end corrosion as a result of SO_3 condensation. Therefore this system needs to be run in conjunction with a sulphur control technique such as the injection of alkaline sorbent. Alkaline sorbent injection can reduce SO_3 concentrations by around 80%. Since alkaline sorbent is significantly less expensive than activated carbon, this process has several economic advantages over carbon use. Further, a reduction of 90°F (32°C) in flue gas temperature is equivalent to a 2% increase in efficiency. This equates to a 2% decrease in fuel use and pollution production. Since a typical 600 MWe coal-fired power plant fires 1.4 Mt/y of coal, a 2% fuel saving is equivalent to 27 t of coal (>$600,000 in fuel costs). This will reduce SO_2, NOx and CO_2 emissions by 2% (Feeley and others, 2003).

Wet ESPs installed after FGD scrubbers to catch sub-micron particles have also been shown to remove mercury effectively, including around 40% of the emerging elemental mercury (Kolker and others, 2006).

The presence of unburnt carbon in ash enhances mercury capture by adsorbing oxidised mercury. Studies at a Western Kentucky power plant firing Eastern (Western Kentucky) bituminous coals have shown that mercury capture can increase with increasing fly ash carbon content and decreasing gas temperature (Newman-Sutherland and others, 2001). Baghouses can be more effective for mercury control

Table 9   Pollution controls and their connection to mercury capture (Srivastava and others, 2006)

System	Total capacity, %†			Coal type*	Native capture, %	Range of capture‡	No of units tested	Effect on mercury capture
	Current	2010	2020					
PM controls								
CS-ESP	36.6	25.4	15.6	B S L	29 3 0	0–63 0–18 0–2	4 3 1	good capture of particulates or sorbent-bound mercury; better native capture for bituminous coals than low-rank coals
HS-ESP	6.2	3.9	3.2	B S L	11 0 §	0–48 0–27	3 4	low native capaure; probably requires specially formulated sorbents for high-temperature mercury capture
FF	3.9	3.6	2.4	B S L	89 73 §	84–93 53–87	2 2	good co-benefit capture; Hg^0 may be oxidised across the FF
PM and SO$_2$ controls								
CS-ESP + wet FGD	13.7	11.6	10.5	B S L	69 16 42	64–74 0–58 21–56	2 3 2	good native capture for bituminous coals because of presence of soluble Hg^{2+} in the flue gas; relatively poor capture for low-rank coals
HS-ESP + wet FGD	2.9	3.9	3.3	B S L	39 8 §	6–54 0–42	3 3	moderate native capture for bituminous coals; poor native capture for low-rank coals
FF + dry scrubber	2.9	2.7	2.9	¶				very high native capture expected for bituminous coals, less for low-rank coals
FF + wet FGD	1.6	1.7	1.6	B S L	75 § §	62–89	2	good co-benefit capture for bituminous coals; Hg^0 may be oxidised across the FF and captured in the wet scrubber
NOx and PM controls								
SCR + CS-ESP	15.1	11.8	7.2	¶				good capture of particulate- or sorbent-bound mercury and better native capture for bituminous coals than low-rank coals are expected
SCR + HS-ESP	2.0	1.1	0.6	¶				low native capture is expected
NOx, PM and SO$_2$ controls								
SCR + CS-ESP + wet FGD	9.1	21.0	31.2	native capture >90% (based on two 2 tests at the Dominion Resources Mount Storm power plant, Unit 2)¶				good capture of particulate- or sorbent-bound mercury; better native capture for bituminous coals than low-rank coals; SCR will tend to enhance capture for bituminous coals by oxidising Hg^0 to the Hg^{2+} form
SCR + spray dryer + FF	0.7	0.9	1.4	B S L	97 23 17	94–99 0–47 0–96	2 2 2	very high native capture for bituminous coals, less for low-rank coals; SCR may enhance capture by oxidising Hg^0 to Hg^{2+} form
SCR + HS-ESP + wet FGD	0.2	2.3	3.2	¶				poor capture of particulate-bound mercury in general and total mercury for low-rank coals; SCR may enhance capture bituminous coals by oxidising Hg^0 to the Hg^{2+} form
SCR + FF + wet FGD	0.3	0.3	6.3	¶				high level of mercury capture would be expected for all coals. SCR may enhance capture for bituminous coals by oxidising Hg^0 to the more soluble Hg^{2+} form

* B = bituminous; S = subbituminous; L = lignite
† CAIR projections; totals do not sum to 100%; less-common control configurations not included here
‡ from EPA's 1999 ICR on mercury emissions for coal-fired power plants
§ none tested
¶ no ICR data available

than ESP, especially with bituminous coals, as the filter cake on the baghouse acts as a fixed-bed reactor for unburnt carbon to enhance mercury capture. Native capture for mercury in baghouse systems ranges from low to over 90%, again depending on the coal and combustion conditions (Srivastava and others, 2006).

Sjostrom and others (2001) noted that baghouses gave good (average 70–84%) mercury removal for bituminous and subbituminous coals but poor (average 0%) removal for lignite coals. According to data from the ICR, baghouses are the only particulate control devices to remove any appreciable amount of elemental mercury but this only occurs at coal chlorine contents above 200 ppm.

Under the UNEP Mercury Partnership the US EPA has partnered with India to share expertise on low cost approaches to improve ESP performance in order to achieve greater particulate control including mercury capture. Experts also conducted some stack testing in 2006 and provided training for mercury monitoring that can be used to estimate emissions levels, including evaluating co-benefit mercury capture (UNEP, 2007a,b).

Under a similar partnership the United States have co-operated on a project to transfer a low-cost technology to improve the performance of ESPs at coal-fired plants and other industrial facilities in Russia, Kazakhstan and Ukraine. Around 700 ESP are installed on plants in these countries and they are generally observed to be operating at below the design value efficiencies. The project demonstrated that adjustments such as alignments of plates ($15,000), gas distribution correction ($20,000) and flue gas conditioning ($950,000) can dramatically improve ESP performance. The selection of the modifications/adjustments required was determined using a diagnostic model. Translation of the model results into Russian, along with training manuals and a demonstration also proved very cost-effective. In 2004 a reduction of 28,820 t of particulate emissions had been achieved at seven power plants. The cost-benefit ratio was calculated at around 3–11 $/t of particulate matter. Compared with world standards of 125–150 $/t captured in an ESP on a 500–300 MW unit, 65% capacity factor, the results are impressively inexpensive (Jozewicz, 2008). Although there was no calculation of the reduction in mercury achieved, it is likely to be correlated to the reduction of particulate matter.

Partnerships such as these could be an extremely cost-effective way of reducing mercury emissions at some plants in developing countries. The transfer of expertise will ensure that ESP and baghouses are run efficiently, maximising mercury capture, whilst ensuring that negative plant effects are not incurred as a result. This type of investment, of time and expertise, could be a one-off cost for each plant, with the running costs of the plant remaining unchanged.

4.2   Systems for SO$_2$ removal

The LCPD and IPPC legislation in the EU will effectively require FGD or equivalent on all large coal-fired utilities beyond 2016. The capacity of plants in the USA fitted with

scrubbers (both wet FGD and dry scrubbers) is expected to increase from around 100 GW in 2005 to over 200 GW by 2020 under the CAIR/CSI. There should be a concomitant reduction in mercury emissions as a co-benefit (US EPA, 2007b).

Wet flue gas desulphurisation (FGD) and similar wet scrubbing systems can typically remove 75–99% of the oxidised mercury present in flue gases. Total mercury removal efficiencies average around 55% (US DOE, 2001a). Meij and others (2001) reviewed data on the effect of FGD and found that studies in the USA, Canada, Japan, Germany, Austria and Denmark all agree that wet FGD systems remove at least 50% of the mercury.

The capture of mercury in FGD systems is dependent on its oxidation state and therefore anything which enhances mercury oxidation will enhance mercury capture in the FGD, including other pollutants such a chlorine. The majority of mercury from US bituminous coals is in the oxidised form and therefore can be removed in an FGD system. Mercury from US subbituminous and lignite coals tends to be mostly in the elemental form and is therefore insoluble and less easy to capture in an FGD system. Coal selection and even coal blending will enhance mercury capture in FGD systems. Coal-blending is discussed in more detail in Section 5.1. Kolker and others (2006) suggest that coal selection or blending to achieve optimal chlorine contents of >500 mg/kg could be considered. Elevated chlorine concentrations often correlate with higher concentrations of oxidised mercury.

The US EPA suggested in their original document on mercury control options (US EPA, 2007b) that research and development efforts should be conducted with the objective of making available oxidising catalysts and reagents by 2015 to enhance mercury oxidation and capture in FGD systems. There have been problems reported with mercury re-emission from wet scrubbers. Research is focusing on evaluating and controlling this problem (US EPA, 2007b). Additives such as sodium hydrosulphide (NaHS) can be added to prevent this effect (Srivastava and others, 2006).

Spray dry scrubber systems can remove both oxidised and elemental mercury with total removal efficiencies of as high as 90%, when coupled with a baghouse (US DOE, 2001a). Data from the ICR suggest that spray dry scrubber systems remove between 0% and 99% of the mercury with an average removal of 38% (CATM, 2001). It has been noticed that mercury capture in spray dryer systems in conjunction with baghouses is lower with low rank coals. The spray dryer system can scrub the halogen species from the flue gases which reduces mercury oxidation downstream (Srivastava and others, 2006).

4.3   Systems for NOx removal

Mercury can be captured by unburnt carbon in fly ash. Low NOx burners or low NOx combustion systems can cause an increase in unburnt carbon (5–30 wt% as loss on ignition) and mercury can concentrate on the carbon rich fraction of the fly ash and thus be captured more efficiently in particulate control systems (Kolker and others, 2006).

Lissianski and others (2005) have evaluated the potential for the optimisation of reburning for both NOx and mercury control. GE Environmental Services (GE ES) have patented a process for staging of the combustion system (low NOx burners, overfire air and coal reburning) to improve the reactivity of the ash for mercury adsorption. The approach comprises CO/O_2 sensors, coal dampers and air flow control actuators. The optimisation of the system within a narrow range of combustion conditions and the optimisation of the LOI (loss on ignition; unburnt carbon) to maximise mercury removal whilst minimising the negative effect of high-carbon fly ash and avoiding high CO emissions. This approach is recommended to optimise 'natural' mercury removal in the fly ash and can be used in conjunction with activated carbon to 'polish' further mercury from the flue gas. By enhancing the mercury capture in natural fly ash, the amount of activated carbon required is reduced.

The GE ES process has been tested at the Green Station, a 255 MWe wall-fired boiler with cold-side ESP and wet scrubbers firing bituminous coal. Pilot-scale data suggested that 80% mercury removal in coal reburning could be achieved for LOI in the range of 8–12% and ESP temperatures below 150°C. The mercury removal efficiency was temperature sensitive with greater removal efficiencies at lower temperatures (for example, around 80% at 127°C to 40–60% at 155°C). At full scale, 80% mercury control was achieved with 10–11% LOI and the ESP in the temperature range of 132–140°C (Lissianski and others, 2005). The reburning system clearly has the potential to reduce NOx and mercury emissions simultaneously whilst reducing the cost of any further activated carbon treatment. However, loss of ash sales and reduced boiler efficiency are likely with such high unburnt carbon contents.

Selective catalytic reduction (SCR) for NOx removal does not, in itself, reduce mercury emissions. However, an SCR can enhance mercury oxidation and therefore, if placed upstream of an FGD device, can aid mercury removal in the FGD. In the USA currently around 100 GW of capacity are fitted with SCR technologies and this is expected to increase to 200 GW by 2020 under the CSI. This should result in a significant increase in the amount of mercury being captured as a co-benefit (US EPA, 2007b).

The range of mercury oxidation in SCR systems is 30–98% with an average of 72% for bituminous coal. The rate for subbituminous coal is reported to be 'much lower' (Kolker and others, 2006). No data are available for lignite (Srivastava and others, 2006). It is possible that the SCR oxidation involves chlorine and that explains the higher oxidation rate for bituminous coals (Kolker and others, 2006).

Straube and others (2008) have carried out bench-scale studies investigating the relationship between mercury oxidation and the HCl concentration of the flue gases around the SCR. The mercury adsorbtion was also correlated to the V_2O_5 content of the deNOx catalyst

The oxidation of mercury by SCR catalysts may be affected by the following (US EPA, 2007b):
- space velocity of the catalyst;

- temperature of the reaction;
- concentration of ammonia;
- age of the catalyst;
- concentration of chlorine in the gas stream.

Although manipulation of one or more of these factors may enhance mercury removal, the catalyst is specifically designed for NOx removal. Any co-benefit mercury removal would need to occur without a significant reduction in NOx removal. Co-benefits will disappear if SCR catalysts lose their mercury oxidising capacity faster than their NOx reduction capacity (Offen and others, 2005).

4.4 Maximising co-benefit approaches

As discussed in the sections above, mercury removal occurs to some extent in pollution control systems designed for particulate, SO_2 and NOx removal. The first part of the USA's CAMR aimed to cap mercury emissions at 38 t/y by 2010. This target should still be achieved despite the vacation of CAMR simply by co-benefit reductions expected under the CAIR. Table 10 shows the research and development goals for mercury control with different technologies, both with and without activated carbon, for the 2010 and 2015 target dates.

However, this co-benefit approach is seen by some (*see* Section 2.3.1) as a way of delaying taking any specific action on mercury. Most of the arguments seem to be based on the principle that *Hg-specific control technology will be commercially available after 2010*. This means that in 2010 coal-fired plants will be able to buy control systems and cut their mercury emissions by 70–90% (Stadler, 2005). It is unlikely that the technologies available will be suitable for use on all plants. To apply an inappropriate technology hastily could result in a very expensive mistake. The co-benefit approach is a convenient way of starting the emission reduction process whilst allowing more time for mercury BAT to be developed to suit the plants which need it most.

Co-benefits are not 'free' as such. However, since the costs are charged to SO_2 and NOx reduction protocols, it is hard to establish any specific co-benefit cost. For example, PSEG Fossil is spending $340 million over the next ten years (post 2005) to reduce NOx, SO_2 and mercury at the Hudson and Mercer Generating stations in New Jersey. Assigning a cost for reduction of each of these pollutants individually is not possible (EERC, 2005a).

The main points of concern with maximising co-benefits, according to the US EPA (2007c) are:
- mercury removal for bituminous coals is greater than for other coal types;
- mercury removal in a baghouse is significantly greater than in an ESP (cold- or hot-side) for both bituminous and subbituminous coals;
- the average mercury removal in spray dry scrubber systems (with baghouse) can be >95% for bituminous coals but only around 25% for subbituminous coals;
- the capture efficiency at different plants is highly variable.

Table 10 Research and development goals for projected cost-effective mercury removal capability (%) for key coal type/control technology combinations (US EPA, 2007b)

Control technology	Existing capacity, MW in 2003	Projected Hg removal capability in 2010 using carbon injection*		Projected Hg removal capability in 2010 by enhanced multi-pollutant controls*		Projected Hg removal capability in 2015 by optimising multi-pollutant controls*	
		bituminous	low-rank	bituminous	low-rank	bituminous	low-rank
ESP only	153,133	70	70	NA	NA	NA	NA
ESP + retrofit fabric filter	2,591	90	90	NA	NA	NA	NA
only fabric filter	11,018	90	90	NA	NA	NA	NA
ESP/fabric filter + dry FGD	8,919	NA	NA	90†	60–70†	90–95†	90–95†
ESP/fabric filter + wet FGD	48,318	NA	NA	90‡	70–80‡	90–95‡	90–95‡
ESP/fabric filter + wet or dry FGD + SCR	22,586	NA	NA	90	70–80†	90–95†	90–95†

* based on aggressive implementation of activated carbon and enhanced pollution control technologies
† assumes additional means to ensure Hg oxidation or innovative sorbents will be used
‡ assumes that means to oxidise Hg will be used as needed
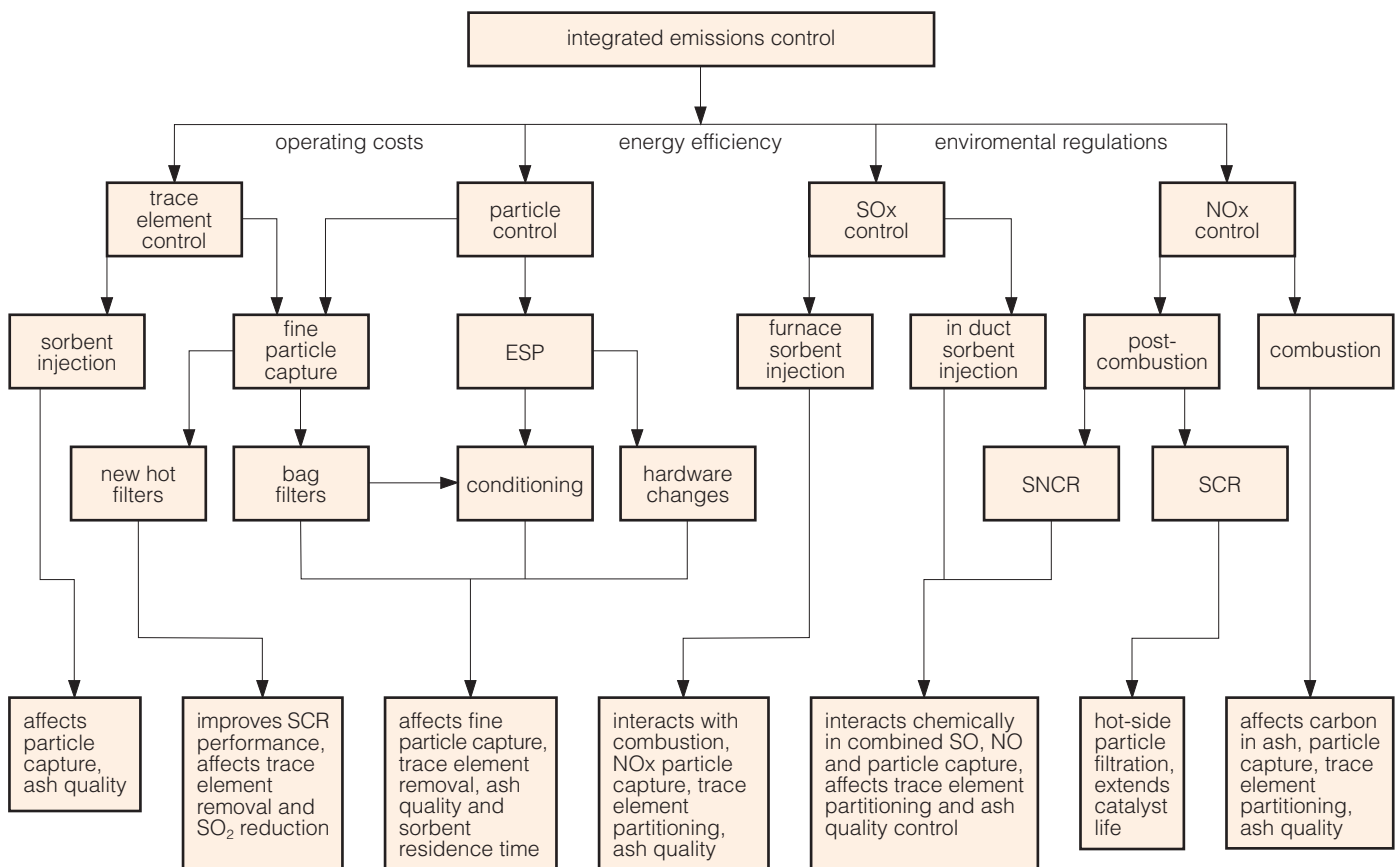
Figure 5 Interactive processes in the control of airborne pollutants from coal combustion (CERC, 2004)

Srivastava and others (2006) emphasise that, although interactions between coals and control technologies can be enhanced to reduce mercury emissions, the science is not completely understood. Field and pilot testing are required at each plant to ensure that the predicted mercury chemistry is actually correct.

Although the Hg reductions may be 'free' extra costs may be incurred with changes in the fly ash, gypsum which occurs as a result. Loss of ash sales will mean a loss of revenue and additional waste disposal costs.

Figure 5 shows the format of integrated emissions control at a

simplified coal-fired plant. Although mercury would automatically be included in the 'trace element' section of the flow chart, the co-benefits discussed throughout this chapter can be achieved through almost every section of the plant. However, the complexity of mercury behaviour and its secondary position in the priorities for most control technologies means that maximising mercury reduction through a plant is a complicated process. As a result, computer models have been developed which help plant operators to model potential improvements in plant operation. For example, MERCURATOR™ is a commercially-available modelling programme based on reaction mechanisms of mercury within coal combustion (Niksa, 2008). The operator supplies plant specific data such as fuel properties, furnace operating conditions, air pollution control characteristics and so on. The model then allows the operator to :

● optimise plant operating conditions and economics to reduce mercury emissions;
● predict mercury emissions based on coal properties and plant characteristics;
● manage plant mercury emissions in a reliable and cost-effective manner;
● expedite the design of mercury control technologies by optimising sorbent characteristics, injection rate, oxidising agent injection conditions and air pollution control device operating conditions.

Similar models are available from other suppliers, notably Reaction Engineering (Senior, 2008).

4.5 Co-benefit approaches in advanced coal combustion systems

This report concentrates on emissions from pulverised coal combustion systems. However, for completion, mercury emissions from other coal combustion are summarised briefly here, the information taken largely from a previous IEA CCC report (Sloss, 2002).

The fluidising velocity of FBC (fluidised bed combustion) systems may affect emissions of mercury. Higher fluidising velocities correspond to shorter residence times inside the boiler and therefore lower mercury removal from the flue gas. Also, high fluidising velocities provide a better mixing of materials in the freeboard space which can improve the mercury capture in fly ash. However, these effects of changes in fluidising velocities only influence mercury emissions slightly compared with temperature effects. Mercury emissions were found to decrease *significantly* with an increase in the secondary/primary air ratio. This may be due to an increased capture of mercury in the fly ash, especially in the unburnt carbon. FBC systems with fabric filter systems gave the highest mercury removal efficiencies in the US EPA's ICR study, ranging from 66% to 99% and averaging 86%. This is thought to be due, in part, to the high carbon content of the FBC ash. The circulating ash in CFBC (circulating FBC) systems has been shown to enhance the removal of mercury.

Although there seems to be nothing published with respect to maximising mercury control in FBC systems, it would seem that maximising unburnt carbons and reducing temperatures may actually be easier and therefore more economic than in pulverised coal fired systems. More work is required in this area.

Although integrated gasification combined cycle (IGCC) plants are not designed with mercury control as a high priority, having the gas in a concentrated, pressurised state could provide an advantage for cleaning technologies. Mercury is present in the syngas largely in elemental form which could be captured by adsorption. A previous report by IEA CCC (Henderson, 2005) suggests that it is likely that there would be a ten-fold saving in the cost of mercury removal in cold gas clean-up systems in IGCC compared to pulverised coal fired plants. Tests at the Polk County IGCC plant, USA, showed the effectiveness of activated carbon impregnated with sulphur for mercury control. However, it would be simpler in process terms to remove the mercury from the crude gas before filtration at around 300°C. At the NUON IGCC plant, Buggenum, 'most' of the mercury is captured in the fly ash (Henderson, 2005).

According to a desk reference document from the US DOE NETL (Klara, 2007), IGCC technologies would aim for a mercury reduction of 90% compared to the input coal. Mercury emissions from IGCC systems in the US (GEE, E-gas and Shell) were estimated at around half of those emitted from subcritical and supercritical boiler systems. Future IGCC plants to be built in the US would be likely to be based on EPRI's 'Coal fleet user design basis for coal-based IGCC plants' specification and would therefore include activated carbon beds for >95% mercury control, especially if the flue gas were to be processed for carbon capture and storage. In the cost analyses for advanced pulverised coal combustion systems, it was assumed that 90% mercury reduction could be achieved by co-benefit approaches.

4.6 Comments

Co-benefit approaches, the ability of particulate, SO_2 and NOx controls to also reduce mercury, have already significantly reduced mercury emissions in the EU and are expected to do so in the near future in the USA.

Since these flue gas treatment systems are not primarily designed to reduce mercury emissions, the amount of mercury captured is variable. These systems can be adjusted to enhance mercury capture, for example:
● lowering temperatures in flue gas and ESP systems;
● increasing unburnt carbon to enhance mercury capture in baghouses;
● ensuring continued oxidation potential in SCR systems.

However, many of these adjustments can cause detrimental effects to the operation of the control system or to areas of the plant downstream. Adjustments to enhance mercury control in these systems will therefore only be considered when the whole balance of plant effects are taken into account. These are likely to be plant-specific and would require expert operation.

In order to ensure that mercury emissions are reduced not only in the developed world, but also in those countries such as China where emissions are increasing, co-benefit approaches may be the most economically sound approach. In order to maximise this, the transfer of information and expertise, if not technologies themselves, would go a long way to reducing mercury emissions in an economic manner.

5 Mercury-specific techniques and technologies

Many of the technologies discussed in this chapter are multi-pollutant control techniques or technologies, that is they will help control emissions of other pollutants from coal combustion systems. However, since they are largely being developed in direct response to the requirements for mercury control in North America, they will be considered as mercury-specific technologies for the purpose of this chapter.

The US DOE had a near-term goal of developing mercury control technologies that achieve 50–70% mercury capture at less than three-quarters of the baseline cost estimate of 50,000–70,000 $/lb (110,000–154,185 $/kg) of mercury removed. The aim was for the technology to be available for commercial demonstration on bituminous coal fired plants by 2005 and on lignite and subbituminous plants by 2007. Much of this is achieved by enhancing co-benefits, as discussed in Chapter 4. This goal was achieved in 2007. The longer-term goal for the US DOE is to develop advanced mercury control technologies to achieve >90% mercury capture. These technologies are to be available for commercial demonstration by 2010 (Jones and others, 2007). This goal is now being re-evaluated in the light of the decision to vacate the CAMR. However, mercury control in the USA is still ongoing. The initial mercury reduction will be achieved, as planned, through co-benefit effects of the CAIR. Despite the recent vacation of the CAMR, new legislation on mercury will be promulgated to replace it which will either be in the form of a MACT requirement, a stringent emission limit or rate, or a combination of both approaches. Either way, the development of mercury-specific control technologies remains a priority in North America.

The US DOE has funded an large amount of research into mercury control strategies. The extensive field testing programme is reviewed by Feeley and Jones (2008) and the interested reader is referred to this document for more detailed information.

The mercury control figures cited are assumed to be a combination of baseline/background mercury removal and enhanced mercury removal due to the technology itself, unless otherwise stated.

5.1 Pre-combustion measures

Figure 6 shows the mercury content of the fuel burned in the plants that would be affected by the CAMR rule in the USA (or its replacement) compared with the mercury allocation for each state and tribe. For many states, especially PA, OH and TX, meeting the mercury reduction requirement will be a significant challenge since the allocation of mercury allowance would have been significantly lower than the actual average mercury content of the large amount of coal burned in these areas. Toole-O'Neil (2006) suggests that pre-combustion mercury reduction will be required in many states in addition to flue gas technologies, especially in those states that plan to increase the number of coal-fired plants online.
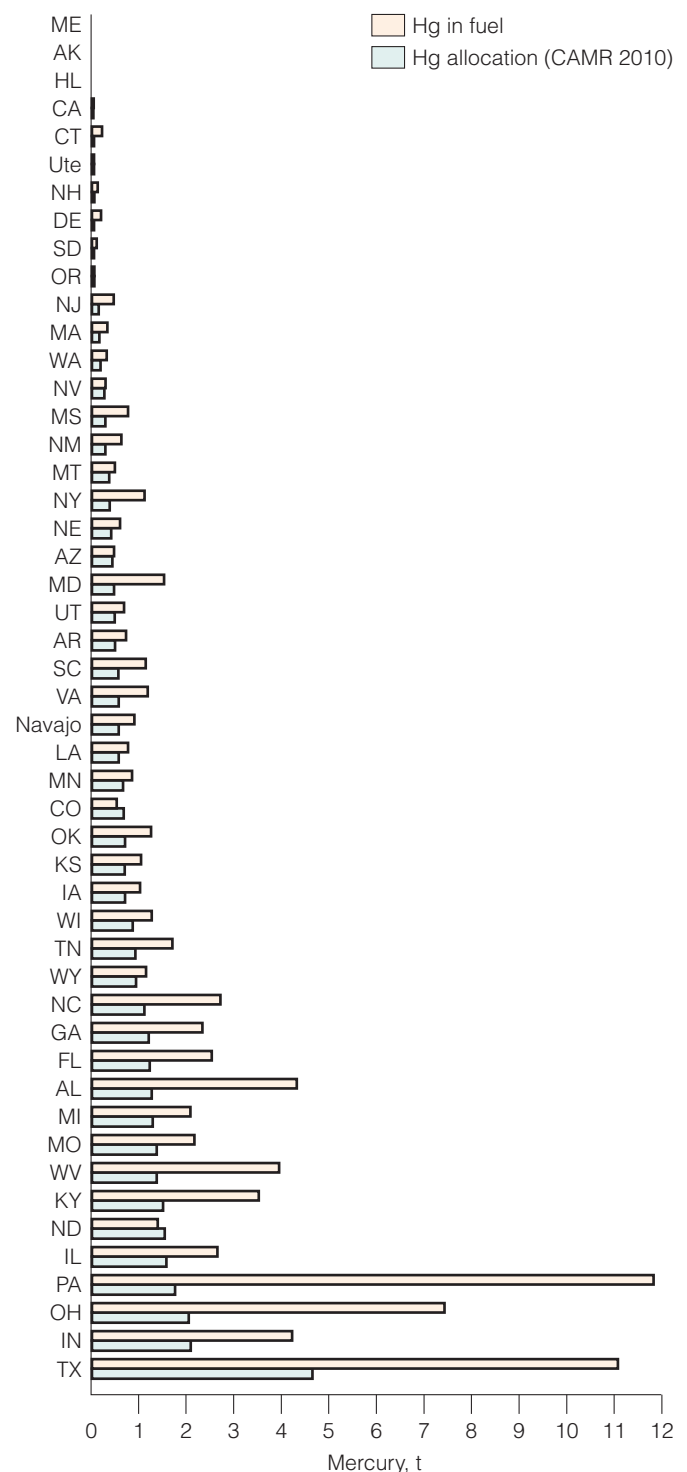
Figure 6 **Mercury allocation budgets (2010) compared with coal mercury content in CAMR affected utilities** (Toole-O'Neil, 2006)

These strategies could include:
- coal selection;
- coal cleaning – new technologies or enhancement of existing treatment;
- coal blending with natural cleaner coals or with upgraded coals;

- cofiring with lower mercury fuels such as natural gas or biomass.

These options are discussed in the sections below.

5.1.1 Coal selection

The mercury concentrations of coals vary greatly and, even though average mercury contents are often published for different coal types and ranks, actual coal mercury contents can vary considerably even from seam to seam. Although US coals have mercury concentrations ranging from 60 to over 900 ppb, the average mercury concentration is between 100 and 160 ppb (0.1–0.16 µg/g; Newman-Sutherland and others, 2001). The US Information Collection Request (ICR) for mercury gave an average of 0.10 µg/g dry coal or 3.5 kg Hg/PJ for US coals. Data from the COALQUAL data set from the US Geological Survey (USGS) gave higher values at 0.17 µg/g dry coal or 5.8 kg Hg/PJ. The COALQUAL data reflect the mercury content of in-ground US coals (that is coals not necessarily representative of the coals actually mined and used) and the ICR data reflect the mercury content of commercial coal during 1999. However, according to Quick and others (2003) commercial US coals contain less mercury than previously suggested and the mercury content has actually been declining during the 1990s. Srivastava and others (2005) note that for recent US EPA mercury control technology studies, the concentration of mercury in eastern bituminous coals is assumed to be 0.10 mg/kg and 0.07 mg/kg for subbituminous coal.

The mercury concentrations of Australian coals are considerably lower than US coals at between 16 and 76 ppb mercury (Newman-Sutherland and others, 2001). South American coals are also reported to have similarly low mercury contents (Mukherjee and others, 2008).

The chemistry of mercury within coal is complex. It is thought to be present in two primary forms – associated with pyrite and other sulphides and associated with organic matter. The mode of occurrence of mercury in coal does not directly affect the speciation of mercury in the combustion flue gas (Kolker and others, 2006). It is beyond the scope of this report to discuss coal chemistry but it is important to note that the association of mercury within coal can have an important effect on the effect of coal cleaning on mercury removal.

Coal chlorine content can also have a significant effect on mercury chemistry. The chlorination of mercury (Hg^0) with HCl or Cl_2 to form mercuric chloride ($HgCl_2$) is a major route for mercury oxidation during combustion and may be the dominant mechanism in mercury flue gas-fly ash interactions. The lower mercury removal efficiencies associated with some western US coals appears to correlate with the low chlorine:mercury ratio. Coal chlorine affects mercury capture in FGD systems and the selection or blending of coals to an optimum of >500 mg/kg chlorine could enhance mercury capture (Kolker and others, 2006). Although the addition of HCl and NaCl may enhance the formation of mercuric chloride, chlorine gas is a much more effective oxidising agent. Oxidising agents are discussed in more detail in Section 5.5.

The presence of sulphur in coal is also important. Coals with high sulphur to chlorine ratios inhibit the formation of Cl_2 and subsequently the formation of oxidised mercury. Selection of low sulphur coal could reduce emissions of mercury from units fitted with cold-side ESP systems or cold-side ESPs plus FGD due to the removal of the sulphur which may block mercury adsorption sites on the unburnt carbon in the ash (Kolker and others, (2006).

The sulphur content of US coal declined during the 1990s and this suggests mercury content would also have declined. There has also been an increase in the production of coal from states with relatively low mercury concentrations (Quick and others, 2003).

There is a strong correlation between coal rank and mercury emissions. Boilers firing bituminous coals tend to achieve higher mercury capture than those firing subbituminous or lignite (Srivastava and others, 2006). This fact is accepted by the US EPA and built in to the now-vacated CAMR. The original CAMR rule addressed differences in coal type by using mercury allocation 'adjustment factors' for each coal rank as follows:

- bituminous   1
- subbituminous   1.25
- lignite   3

There is, however, still controversy over whether the rankings are accurate, despite significant amounts of data in support of the ranking (Steele and Schaefer, 2005). US subbituminous coals and also western bituminous coals, contain lower levels of mercury. Subbituminous coals also contain lower levels of chlorine. Therefore US subbituminous coals may emit less mercury but emit more in the elemental form which is harder to capture. It is therefore important to consider coals not only for their total mercury content but also for the way the mercury behaves during combustion. Foerter (2005) believed that the CAMR would have unfairly penalised eastern bituminous coals and could have provided sufficient incentive for plants to switch from eastern to western coals.

Although it is recognised that the higher rank coals in the USA give rise to lower mercury emissions, the general trend in the USA is still towards the use of lower-rank coals as these generally contain lower sulphur. Whilst the use of low-sulphur fuels is predicted to increase from around 550 tons (491 t) in 2001 to over 800 tons (714 t) by 2025, the use of medium and high-sulphur coals is expected to remain relatively stable at around 450 tons/y (402 t/y) and 150 tons/y (134 t/y) respectively (US EPA, 2007b).

It may be possible to source coal according to its mercury content or mercury emission characteristics. However, coal selection due to mercury content would only occur after other important parameters had been met, such as ash content, sulphur content and so on. If all the characteristics between available coals are similar then the mercury content could be added as a characteristic of choice. Although lower mercury coals should not cost any more or less than other coals, if these coals were suddenly to be in demand then prices may rise accordingly.

Johnson and others (2007) have expanded 'green purchasing' efforts at coal plants to include selection based on mercury content. The potential for mercury reduction has been evaluated for the Fayette Power Project (FPP), Texas, consisting of three large coal-fired power plants. Most of the coal fired in the region comes from the Gillette Coal Field in the Powder River Basin, where there are currently 18 mines in operation. The price of coal from these fields, as with others, varies with the heating value of the coal and also with the sulphur content. Since the three plants in the FPP are in the process of installing FGD systems, the sulphur content of the coal will soon be less of a factor with the selection of coal. It is therefore possible that the plants could switch to buying lower mercury coals instead. Data from several databases have shown that Dry Fork, Rawhide and Wyodak mines all tend to have mercury contents below 0.055 ppm. An equal mix of coals from all three mines would give a mercury concentration of 0.049 ppm which is 41% lower than that of the coal the FPP purchased in 2001. However, Johnson and others (2007) emphasise that there are some discrepancies and disagreements with the reported data on coal mercury concentrations due to both natural variation and measurement error. Although two mines, Antelope and North Antelope/Rochelle, have the lowest average mercury concentrations (around 0.040 ppm) they are not considered potential suppliers for FPP since they also have low sulphur concentrations (0.33% and 0.22% respectively) and would therefore be more expensive than the other coals. It was noted that, although significant mercury reductions could be achieved by changing coal supplies, further and continued testing would be required to ensure that the coal mercury content was consistent throughout the coal. It was suggested that between 10–50% mercury reduction could be achieved by coal switching alone and could be cost-effective.

Quick and others (2003) note that selective mining of low mercury coals could be a practical way of achieving 'significant' reductions in mercury. In the same way that the USA has achieved reductions in sulphur emissions by firing lower sulphur coals, mercury emissions in the USA could also be reduced in a similar manner. This option could be important for other countries too.

Kolker and others (2006) agree that, in situations where the mercury content of the coal beds are structurally or stratigraphically controlled, selective mining could be a useful approach – by avoiding the most mercury-rich zones. However, Kolker and others (2006) suggest that this level of selection would require sampling and analysis at a level of detail beyond that which is ordinarily required by coal providers. The economics of this approach would depend on the mining situation as well as the specifications of utility coal contracts.

In Guizhou Province, China, mercury pollution from domestic coal combustion is evident in at least one village. Many elderly villagers exhibit loss of vision, and this has been attributed to an abundance of mercury minerals. Mercury concentrations in the coals burned have been reported to be as high as 55 ppm, over 200 times the average in US coals (Finkelman, 2007). It is unclear why this coal is so concentrated with mercury. It should be borne in mind that the domestic combustion of coal could be a significant source of mercury to the environment in some areas. It has also been suggested that there is a problem with the theft and use of washery waste coals (Wu and others, 2006). Significant reductions in mercury emissions could be achieved in such locations with very simple changes in coal supply, assuming alternative supplies are available.

Figure 7 shows the mercury content of raw coals as mined in China. The national average is around 0.19 mg/kg. However, the variability within individual coals is large. Coals from Xinjiang and Qinghai have concentrations below 0.05 mg/kg whereas those from Guizhou are over ten times more concentrated at 0.5 mg/mg (Wu and others, 2006). It is therefore possible that emissions of mercury from Chinese coal-fired plants could be cut considerably by simply switching coals. However, in practice, this may not always be economically or even physically possible due to the location of the mines and the plants and the distribution and sales network already established in China. Potential for changing coal supply would need to be determined on a case-by-case basis.
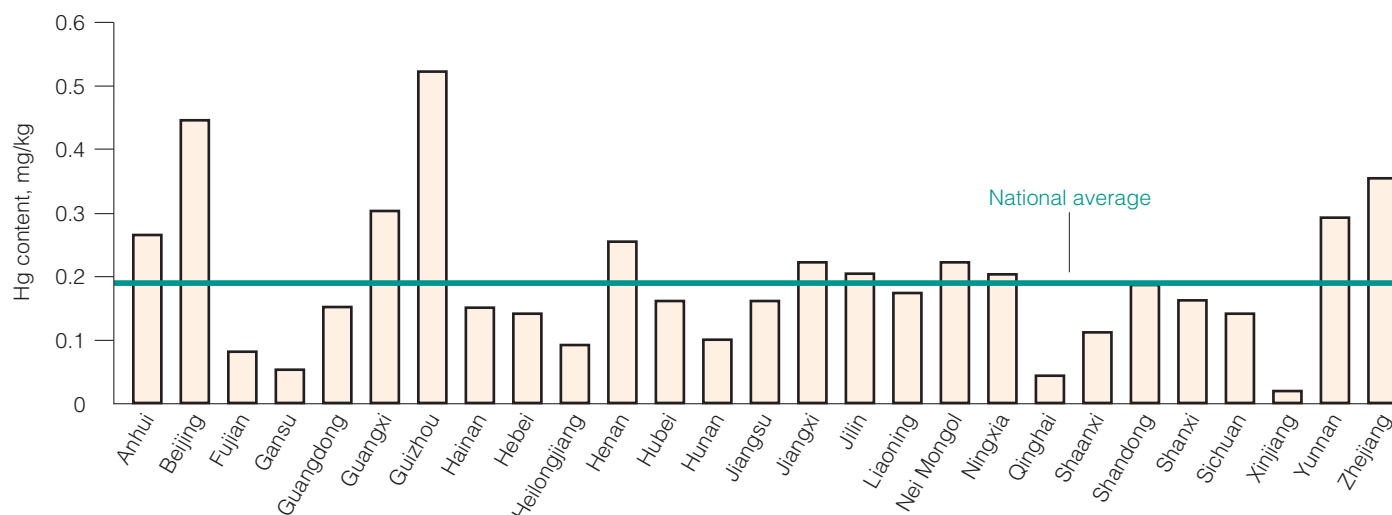

Figure 7   Mercury content of raw coal as mined in China by Province (Wu and others, 2006)

5.1.2 Coal cleaning

Coal preparation, such as washing to reduce sulphur in bituminous coals, can reduce the mercury concentration relative to in-ground concentrations. The removal varies according to the fraction of mercury in the sulphides and the efficiency of sulphide removal (Kolker and others, 2006). Mercury removal by coal cleaning often correlates with the removal of coarse-grained pyrite. This, in turn, correlates with low rank coals such as lignite and subbituminous coals (Mukherjee and others, 2008).

Other coal cleaning processes such as froth flotation, selective agglomeration, cyclones and chemical approaches, are also aimed at sulphur removal and therefore any mercury reduction is a co-benefit. Mercury reduction in these processes ranges from 10% to as much as 70% with an average of 30% on an energy equivalent basis. Again, the mercury reduction is highly correlated to the pyrite concentration – rejection of the coarse pyritic fraction by pulverisers at the coal-fired plant reduces mercury input to the boiler regardless of whether the coal has been cleaned (Kolker and others, 2006).

Physical cleaning of mid-western bituminous coals can achieve up to 26% mercury removal, although pilot-scale processes claim to achieve up to 47% removal. More advanced physical coal cleaning methods are reported to achieve up to 64% mercury removal. Advanced coal cleaning methods such as froth flotation can achieve 1–51% (26% average) over and above conventional cleaning. Selective agglomeration can increase this to 63–82%, with an average of 68% (Pavlish and others, 2003).

Although coal washing/treatments may produce relatively high reduction rates for mercury with some coals, they are not adequate for use as a reliable method for mercury reduction on all coals. The success rate will be site/coal specific (Dronen and others, 2004).

The cost of mercury removal by conventional coal cleaning is regarded as zero since this is performed for other reasons. Additional chemical cleaning of Northern Appalachian coals for mercury removal has been estimated at 17,000 $/lb mercury (2003 estimate; 37,778 $/kg), achieving 58% reduction potential (heat basis). Mild chemical treatment of Southern Appalachian and Eastern Interior coals, achieving around 40% mercury reduction, was estimated at 25,000 $/lb and 33,000 $/lb (55,555–73,333 $/kg) respectively (Pavlish and others, 2003).

Other advanced and largely developmental treatments are available for removing pollutants such as mercury including biological treatments, chemical pre-treatments and air-fluidised systems (Dronen and others, 2004; Honaker and others, 2007). However, little or no information is published on the cost of these treatments.

It is not possible to predict the current baseline rate of mercury removal by coal washing (which is only performed for general coal cleaning and sulphur removal). Neither is it possible to determine to what extent increasing coal washing activities would lead to a reduction in mercury removal. However, it could be assumed that at least a low rate of mercury removal is achieved in most situations (perhaps up to 20% or 30%).

Many of the coals used in China have mercury concentrations which are well above the average. For example, in Guizhou the average mercury concentration in coal is 0.53 mg/kg (*see* also Figure 7). Tests on the coals in this region suggest that the mercury exists mainly in pyrite. If this is the case, then traditional physical coal cleaning would remove a large amount of mercury from the coal. However, in 2000, less than 8% of the coal in the area was subjected to coal cleaning. There is therefore potential to reduce mercury emissions significantly in the Guizhou area by coal cleaning alone (Feng and others, 2002).

In order for any mercury removal by coal treatment to be a guaranteed reduction in mercury emissions, the mercury washed out from the coal would have to be either collected or treated to ensure that it did not end up being released into the environment. Coal washery wastes would also need to be disposed of or used in a controlled manner.

5.1.3 Coal blending/cofiring

As mentioned in Section 5.1.1, bituminous coals tend to allow greater mercury capture in flue gas cleaning systems because of their higher chlorine content and tendency to produce more unburnt carbon. Blending small amounts of bituminous coals with subbituminous coals and lignite may provide some mercury control at some plants. For example, the 360 MWe Holcomb station in the USA fires Powder River Basin coal and is fitted with spray dry scrubbers and fabric filters for SO_2 and particulate control. It was shown that the addition of 'small amounts' of western bituminous coals to the fuel mix increased the vapour phase mercury capture from below 25% to nearly 80%. However, the effectiveness of this approach is likely to be site-specific and would need further evaluation. It is also possible that blending coals could change boiler slagging and fouling characteristics or the performance of the air pollution control system, potentially increasing operation and maintenance costs (Srivastava and others, 2006; US EPA, 2005). Further, the availability of different coals could add cost, depending on factors such as location and suppliers.

Offen and others (2005) note that a 40:60 mix of Eastern bituminous coal with Powder River Basin coal showed oxidation rates across an SCR equivalent to those observed at plants firing 100% Eastern bituminous coal. They also note that an EPRI study demonstrated at a 15:85 mix can have a 'significant' effect on mercury removal.

As discussed in Section 4.2, the oxidation state of mercury determines the capture efficiency in FGD systems. Kolker and others (2006) suggest the selection or blending of coal to achieve optimal chlorine content of >500 mg/kg.

Biomass tends to have a lower concentration of mercury than coal and therefore cocombustion of the two can lead to a

reduction in total mercury emissions from the plant. However, the cocombustion of biomass can cause problems in the plant, such as increased slagging and fouling and a reduction in boiler efficiency, but, more importantly, may actually cause chemical and physical changes to the ash that hinder mercury absorption. Since biomass is so variable, it is important to determine on a case-by-case basis whether it actually enhances mercury capture. Even if mercury capture is enhanced, cofiring biomass is unlikely to be considered a major mercury reduction strategy for a plant.

Zhuang and Miller (2006) have studied the effect of cofiring tyre-derived fuel and western subbituminous lignite. With 100% coal firing there was only around 17% oxidised mercury whereas this increased to almost 48% when 5% (mass basis) tyre-derived fuel was added.

Cofiring with other fuels, such as natural gas or biomass, is commonly limited to below 20% of the blend. Although the costs for alternative fuels may be low, there can be associated plant problems such as requirements for specialist handling or separation and flame stability can be an issue. Sourcing fuels such as tyre-derived fuel may also be an issue (Pavlish and others, 2003).

5.2   Combustion modifications

Theoretically one of the simplest ways to reduce emissions of almost any pollutant is to increase the efficiency of combustion. This reduces the amount of fuel required and therefore decreases the emissions of any species associated with the coal. Increasing efficiency is listed as one of the main priorities under the UNEP Partnership for reducing mercury emissions from coal combustion along with fuel switching to fuels containing less mercury (UNEP, 2007a,b). These techniques could be important strategies for mercury control in the future, especially if adopted into national energy strategies. The rest of this section concentrates on reducing mercury from existing pulverised coal combustion systems.

Romero and others (2006) have studied the modification of boiler operating conditions on mercury control at two full-scale pulverised coal fired boilers in the USA. It was shown that a combination of combustion control settings can be found which produces a major impact on the overall mercury emission whilst minimising impact on unit constraints such as NOx emissions and unit performance. Excess O_2, overfire air register settings, and mill out-service configuration were found to affect mercury emissions, as well as ESP power levels. Lower excess O_2 conditions reduce the availability of air for char burn-out causing increased combustion in the ash. Modification of overfire air registers also achieved the same effect. Any of these modifications causing an increase in unburnt carbon also caused an increase in mercury capture. Increasing LOI from 17% to 22% reduced mercury emissions from 82% to 94%. Romero and others (2006) noted that such combustion modifications for mercury control typically resulted in increased NOx emissions and small penalties in unit heat rate. However, in combination with activated carbon, boiler modification may be a cost-effective method of mercury control. Romero and others

(2006) acknowledged that the data set was small and that further study was needed.

EPRI have also noted a strong correlation between LOI. With Powder River Basin coals, even a small amount of unburnt carbon can increase mercury capture across a baghouse (Offen and others, 2005). Salem Harbor plant in the USA has 'very high' carbon-in-ash (15–30%) which helps achieve over 80% mercury removal in the cold-side ESP system. It is therefore possible to optimise the trade-off between higher fly ash carbon and improved mercury capture at the plant by adjusting the combustion conditions or fuel. Since unburnt carbon is an indication of decreased combustion efficiency, there is also a trade-off in overall plant performance. This approach could be used at any plant with cold-side ESP or baghouses but would be more applicable to those with dry bottom boilers. As mentioned in Section 4.1, hot-side ESP are not so efficient at capturing mercury on carbon. Increased carbon-in-ash could lead to loss of ash sales which would add to plant costs (US EPA, 2005; *see* Section 5.3.3).

GE Energy, USA, have been working on mercury control using combustion staging for a number of years. The combustion conditions in the boiler can be modified using overfire or reburn air and even with optimisation of the basic fuel and air controls. The modifications can be tailored to the plant configuration, depending on coal type, particulate control device, and any controls for SO_2 and NOx. Pilot-scale tests were performed at a 300 kW boiler simulator. As with other studies, this project emphasised the correlation between unburnt carbon and mercury capture. Higher overfire air conditions resulted in greater mercury removal. A combination of fuel blending and combustion modification was shown to improve mercury capture significantly in a cost-effective manner (Lissianski and others, 2005)

5.3   Activated carbon injection

It is clear when looking at the ongoing activity in the USA that the mercury control option receiving the most attention/investment is activated carbon. Sorbents such as activated carbon adsorb mercury depending on several factors including:
- the surface area of the sorbent;
- the pore volume;
- mercury concentration;
- the oxidation state of the mercury (oxidised mercury is adsorbed far more easily than elemental mercury);
- flue gas temperature;
- flue gas composition (presence of competing or reactive species such as chlorine or sulphur).

For PAC (powdered activated carbon) the costs include not only the sorbent itself but also the transportation costs, installation costs for injection ports, storage silos, feeder system and injection lances, amongst others. The capture of mercury on activated carbon is also dependent on the use of appropriate particulate capture technologies. Some plants may already have suitable particulate control systems installed, others may not. There are therefore three scenarios for evaluating the cost of mercury control with sorbents such as

PAC (EERC, 2005b):
- only activated carbon is used;
- activated carbon is used along with an additive;
- installation of a fabric filter is required to cope with the additional particulate load of the activated carbon.

The final scenario is by far the most costly. However, one advantage of the installation of a new fabric filter for dedicated activated carbon capture, in addition to more efficient mercury capture, is that the quality of the fly ash does not change significantly and income from ash sales may not be lost. The following sections discuss the individual costs associated with installing activated carbon systems at full-scale plants.

5.3.1 Installation/retrofit costs

It has been estimated that, for a 500 MW plant, the capital cost for PAC injection equipment would be around 3–4 US$/kW (based on 2003 estimates). Smaller systems can be more expensive at up to 8 US$/kW. An additive injection system would add less than US$100,000 to the total capital cost. If a new fabric filter system is required then the cost escalates. Capital cost for a new fabric filter is around 55–70 US$/kW, depending on the size of the plant. This can be broken down into several cost items as follows (EERC, 2005b):
- major equipment (35%);
- auxiliary or accessory equipment (15%);
- field installation (20%);
- project management and engineering;
- freight, taxes, subcontractor (17%);
- start-up cost, working capital and other capitalised costs (15–20%).

There may also be additional cost due to requirements to install foundations, modify duct-work and so on which may double the quoted vendor costs, depending on the requirements of individual facilities.

Pavlish and others (2005) quote capital costs of installing activated carbon/sorbent injection systems as follows:

Plants		
	>360 MWe	3–4 $/kW
	200–360 MWe	4–6 $/kW
	<200 MWe	6–9 $/kW

Installing a new fabric filter/baghouse was 55–70 $/kW regardless of plant size.

The installation of a new fabric filter will take at least 3–4 months and anything up to two years, depending on the plant. The timescale of the retrofitting of co-benefit technologies quoted in a US EPA study was based on the following timescale assumptions (US EPA, 2007b):
- ACI (activated carbon injection) on an existing ESP or baghouse – one year;
- ACI and a retrofit fabric filter (eg COHPAC) – two years;
- new SCR/FGD/ESP or baghouse/Hg system – three to four years;
- existing SCR or FGD system to enhance Hg control – one year.

Table 11 Operating costs for activated carbon injection systems (on a 250 MW plant) followed by either ESP or COHPAC fabric filter for bituminous coals (IJC, 2005)

	ESP	COHPAC
Mercury removal, %	70	90
PAC injection rate, lb/Macf (kg/Macm)	10 (160)	3 (48)
PAC injection cost, $	790,000	790,000
Activated carbon cost, $/y	2,562,000	796,000

The activated carbon costs varied with the quantity required but was estimated at 50 $/kW at 80% capacity for firing bituminous coal

This may require downtime in plant operation which will decrease revenue and increase absolute costs. Construction times may be lengthened depending on the availability of equipment and skilled staff. According to the report by EERC (2005b) a predicted shortage in skilled labour (boiler-makers, pipe fitters and electricians) could delay the installation of multi-pollutant control systems in Canada between 2005 and 2013.

The actual cost of mercury control with activated carbon will also depend on the particulate control system used. Table 11 shows the operating costs for ESP and COHPAC (advanced hybrid particle collection) fabric filters. The estimates are for a 250 MWe plant with an 80% capacity for firing bituminous coal and assumed the cost of the COHPAC system would be around 50 $/kW ($12.5 million). Despite the higher initial installation cost, the COHPAC system achieved a greater mercury removal rate with lower PAC injection and therefore lower running costs. The ESP system required lower capital but higher operating costs whilst achieving lower mercury removals compared to the COHPAC system (IJC, 2005).

5.3.2 Operating and maintenance costs

Regardless of which sorbent or particulate capture system is used, there are associated operating and maintenance costs with sorbent injection systems (CCME, 2005):
- the sorbent itself (*see also* Section 5.3.3);
- activated carbon disposal;
- power;
- operating labour;
- equipment maintenance;
- water (for spraying additives, if used);
- cost of money (inflation and interest).

Estimates for operating and maintenance costs at Canadian power plants based on ACI use are summarised in Table 12. The costs vary significantly from plant to plant. Two plants, one a CFBC system (Point Aconi) and the other with installed wet FGD (Belldune), are managing to achieve significant

Table 12 Additional operating and maintenance costs for Canadian power plants as a result of adding ACI (CCME, 2005)			
Power station	Net capacity, MW	Units	Total O&M costs for all units, million US$/y
Alberta			
Battle River	675	3	8.95–12.41
Sheerness	766	2	6.44–8.00
Genesee	1182	3	6.46–12.61
Keephills	766	2	4.44–8.66
Sundance	2020	6	11.50–22.40
Manitoba			
Brandon	95	1	0.30–0.59
New Brunswick			
Belldune	450	1	0*
Nova Scotia			
Lingan	600	4	1.32–4.92
Point Aconi	165	1	0†
Point Tupper	150	1	0.60–1.16
Trenton	310	2	1.23–2.37
Saskatchewan			
Boundary Dam	814	6	5.11–19.86
Poplar River	562	2	3.38–13.08
Shand	279	1	1.71–6.60

* Belldune is already achieving 70% mercury control with its wet FGD system
† Point Aconi is already achieving 60–90% mercury control, it is a CFBC system

mercury reduction through co-benefit and can avoid the additional cost of mercury control completely, for the moment. The original report, available at http://www.ccme.ca/assets/pdf/hg_eerc_rpt_es_e.pdf gives more detail on the different coal types and air pollution systems at each plant and the grades the most promising options for mercury control according to the specific characteristics at each plant.

5.3.3 Sorbent costs

In a report evaluating mercury control cost in Canada, EERC (2005b) estimated that the cost of the sorbent is 93–94% of the total operating and maintenance cost. Therefore the cheaper the sorbent and the less sorbent required, the better. Srivastava and others (2005) agree stating that the capital costs of activated carbon injection based technologies make up a relatively minor fraction of the total annual costs unless a fabric filter is also required. The major cost is the sorbent itself.

Activated carbon costs vary with type (halogenated or not halogenated) and supplier. Jones and others (2007) listed the costs of carbons from several different suppliers and they ranged from 0.39 \$/lb (0.87 \$/kg) for Super HOK (a conventional activated carbon supplied by RWE Rheinbraun) to 1.35 \$/lb (3 \$/kg) for Mer-Clean chemically treated sorbents supplied by Alstom. However, the price becomes less relevant if more is required to achieve the same result.

The US DOE has carried out extensive work on activated carbons at full-scale plants using different sorbents. In the final report Jones and others (2007) list the purchase cost of the activated carbons tested:

Holcomb Unit 1	DARCO Hg-LH (brominated AC)	0.95 \$/lb	2.11 \$/kg
Meramec Unit 2	DARCO Hg-LH	0.95 \$/lb	2.11 \$/kg
Yates Unit 1	untreated sorbent (Super HOK)	0.39 \$/lb	0.87 \$/kg
Leland Olds Unit 1	untreated DARCO	0.54 \$/lb	1.2 \$/kg
Stanton Unit 10	DARCO Hg-LH	0.95 \$/lb	2.11 \$/kg
St Clair Unit 1	brominated PAC	0.85 \$/lb	1.89 \$/kg

Estimating the cost of sorbents is not simple. The cost can be estimated by a bottom-up approach for a given quantity of sorbent, adding overhead/administrative costs and an estimated profit margin. Critical assumptions include the source of material supply, site location and quantity of product demanded. Costs for sorbent production have been estimated based on several pricing assumptions including (Apogee Scientific, 2004):
● basic production costs – raw material and manufacture (labour, power, consumables, etc);
● marketing costs (assumed to be 1.5–4.5% of basic production costs);
● general and administrative costs (around 5% of basic production cost);
● capital recovery (at an interest rate of 6%);
● reasonable profit.

The estimates did not include research and development costs and ongoing expansion efforts. Based on these assumptions, a price comparison was produced, as show in Table 13. As emphasised earlier, these prices are now likely to be out of date since publication in 2004. However, they do give a feel for the relative prices of different sorbents and how the price may drop with factors such as quantity purchased. Costs will have to be taken into account for shipping and transportation of the sorbent to the power station. This will clearly vary with volume/weight and distance.

The price of activated carbon and related sorbents is not fixed, even when cost budgets have been analysed. Many different sorbents are available and their cost will vary with market forces. Suppliers may offer discounts to utilities based on guaranteed sales over a fixed period. Further competition and maturation of the market could mean that prices may decline

Table 13 Sorbent pricing comparison (Apogee Scientific, 2004)			
Sorbent	Quantity	Price range	
		$/lb	$/kg
Commercial activated carbons	≥900 lb	0.30–0.45	0.66–0.99
Lignite activated carbon	12,500 t/y	<0.20	<0.44
CS80 (commercial sorbent)	2800 t/y	<0.18	<0.40
	30,000 t/y	<0.15	<0.33
Corn-char fly ash	22,000 t/y	<0.25	<0.55

in the near term. Stadler (2005) cites the ICAC (Institute of Clean Air Companies) in saying that the cost of activated carbon injection systems had dropped fourfold in 2004-05. At this time, the current market is one in which supply exceeds demand. However, if the US EPA sets stringent legislation for mercury control in the near future, buyers will find themselves in a weak position with demand out-stripping supply capacity which could push prices up until supply and demand level out (Apogee Scientific, 2004). It is therefore difficult to list sorbent costings in a report such as this without the data becoming invalid and out of date very quickly.

5.3.4   By-product issues

Ash sales can bring revenue to many power plants. Plants in the USA typically receive 4–7 US$/t **for** ash. If the ash becomes unsaleable then the revenue is lost and the plant must bear additional costs for ash disposal, at around 14–16 US$/t. The loss of ash sales therefore causes a total addition expense of 18–23 US$/t of ash (EERC, 2005b).

In a study by Srivastava and others (2005), loss of sales revenue and increased ash disposal costs were assumed to be as follows:

low-sulphur bituminous coals	0.37 mills/kWh
high-sulphur bituminous coals	0.93 mills/kWh
subbituminous coals	1.01 mills/kWh

Costs are higher for higher sulphur bituminous coals and subbituminous coals as these coals produce more ash and gypsum than low-sulphur bituminous coals.

There is a considerable amount of work under way to develop sorbents which will not affect the performance of fly ash in uses such as cement and concrete. The economic effects of loss in ash sales are included in many of the cost estimates discussed in Section 5.3.4. Trace elements, including mercury, and their affect on ash use is discussed in more detail in a recent IEA CCC report (Sloss, 2007) and the interested reader is referred to the original document for more information.

Examples of the US DOE's estimate of how by-product issues affect the total long-term costs of mercury control at different plants are given in Section 5.3.5 *below*. The impact of changes in fly ash on sales and costs will depend on how the

altered fly ash is captured. If activated carbon or sorbent is used with an existing ESP or baghouse then the ash will be altered. If, however, an additional 'polishing' system is added, then there is the possibility that the altered fly ash is collected separately to the normal fly ash. This would significantly reduce cost impacts in many cases.

Sorbents such as MinPlus are being developed which give good mercury capture efficiency without affecting the quality of the ash. Re-usable sorbents (such as several of those discussed in Section 5.4 *below*) could keep costs down, especially disposal costs.

5.3.5   Total costs

Jones and others (2007) emphasise that these total cost economic analyses represent 'snapshots' in time based on various assumptions and conditions. The economics are therefore plant- and condition-specific and are based on relatively small data sets. The costs of mercury control using activated carbon could vary significantly at different locations. The projects discussed below are examples of the most cited material in the published literature but by no means represent all the cost analyses that have been performed to date. They are not intended to be compared directly. Further, by the time this report is published, some of the data reviewed could be considered dated since the field of sorbents for mercury control is so dynamic. This section is therefore intended to give an overview of the total cost considerations and not intended as an indication of actual expected costs.

Table 14 shows the cost of activated carbon injection systems for different coal types. These are the results from the extensive studies performed by the US DOE NETL at numerous full-scale plants throughout the USA. The interested reader is referred to the original document by Jones and others (2007) for more information as the report is extensive and detailed. For clarification, the plant details are as follows:

Yates Unit 1 – low-sulphur bituminous coal-fired plant fitted with a cold-side ESP and a wet FGD system. Baseline mercury removal prior to the installation of mercury-specific controls was 50%.

Monroe Unit 4 – firing a blend of low-sulphur bituminous coal and PRB coal. The plant was fitted with a cold-side ESP and an SCR system and had a baseline mercury removal rate of 25%.

Lee Unit 1 – low-sulphur bituminous coal-fired plant fitted with cold-side ESP and a SO_3 flue gas recirculating system. Baseline mercury removal was 20%.

Portland Unit 1 – medium-sulphur bituminous coal-fired plant fitted with cold-side ESP with 30% baseline mercury capture.

Holcomb Unit 1 – subbituminous PRB coal-fired plant fitted with a spray dryer absorber system and a fabric filter. Baseline mercury removal was 37%.

Plant	By-product impacts*	ACI, kg/m³	COE increase, mills/kWh	$/kg Hg removed	ACI, kg/m³	COE increase, mills/kWh
Table 14 20-year levelised cost of mercury control for coal-fired plants in the USA (Jones and others, 2007)						
a) Bituminous units		**50%**			**70%**	
Yates Unit 1	without	61.84	0.98	121,440	144.23	1.72
(Super HOK)	with		2.92	363,000		3.66
Monroe Unit 4	without	23.45	0.38	37,840	54.29	0.75
(DARCO® Hg)	with		1.62	160,820		1.99
Lee Unit 1	without	32.25	1.14	157,080	77.58	1.95
(B-PAC™)	with		2.85	393,800		3.66
Portland Unit 1	without	9.48	0.45	29,480	22.33	0.69
(Mer-Clean™ 8-21)	with		1.60	105,380		1.84
b) PRB units		**50%**			**70%**	
Holcomb Unit 1	without	1.77	0.15	9,636	4.34	0.18
(DARCO® Hg-LH)	with		0.86	56,320		0.89
St. Clair Unit 1	without	4.18	0.39	37,840	9.64	0.52
(B-PAC™)	with		1.36	133,100		1.49
Meramec Unit 2	without	4.34	0.38	26,840	9.96	0.48
(DARCO® Hg-LH)	with		1.74	123,420		1.84
Dave Johnston Unit 3	without	0.96	0.26	16,368	2.25	0.30
(Mer-Clean™ 8)	with		1.55	96,800		1.59
Stanton Unit 1	without	6.59	0.39	36,740	15.26	0.54
(B-PAC™)	with		1.07	99,880		1.22
c) ND lignite units		**50%**			**70%**	
Leland Olds Unit 1	without	34.53	0.74	40,260	80.95	1.21
(DARCO® Hg & CaCl₂)	with		3.37	183,920		3.84
Stanton Unit 10	without	7.87	0.85	44,660	18.47	1.05
(DARCO® Hg-LH)	with		2.58	135,300		2.78
Leland Olds Unit 1	without	2.89	0.32	17,380	6.75	0.42
(Mer-Clean™ 8)	with		2.95	161,040		3.05

*   Table 14a displays economic data for 80% ACI mercury removal at Monroe and Lee, and 90% ACI mercury removal at Portland
†   Table 14c displays economic data for 80% ACI mercury removal at Lelands Olds and Stanton 10, and 90% ACI mercury removal via Mer-Clean™ 8

St Clair Unit 1 – firing a blend of bituminous coal and PRB coal. The plant was fitted with a cold-side ESP and has a baseline mercury removal of 25%.

Meramec Unit 2 – PRB fired plant fitted with cold-side ESPs and had a 32% baseline mercury removal rate.

Dave Johnston Unit 3 – PRB fired plant fitted with cold-side ESPs and had a 12% baseline mercury removal rate.

Stanton Unit 1 – PRB fired plant fitted with cold-side ESPs and had a 15% baseline mercury removal rate.

Lelands Olds Unit 1 – firing North Dakota lignite and fitted with a cold-side ESP, this plant had a 18% baseline mercury removal rate.

Stanton Unit 10 – firing North Dakota lignite and fitted with a cold-side ESP, this plant did not appear to have a baseline mercury removal rate above 0%.

The results in Table 14 show the costs for different plant firing different fuels and using different activated carbons. The results are not meant to compare costs between plants or between activated carbons since each plant has its own requirements and challenges. However, the results do give an indication of the general range of costs for plants and how these can increase if 80–90% mercury control is required. The impact of the loss of ash sales is also shown to be significant at all plants, in some cases quite dramatically so.

Although somewhat dated, the data shown in Tables 15 and 16 show the comparative costs of different levels of mercury control using ACI for bituminous and subbituminous coals at plants fitted with ESP and no FGD. Although the cost estimates are likely to be inaccurate due to developments in the technology, the tables still serve to indicate that the costs for subbituminous coals are significantly higher than for bituminous coals, due to the difference in the coal chemistry (*see* Section 5.1). The tables also indicate that ACI with a fabric filter is less expensive than ACI alone. Once a fabric filter is installed, the cost difference between 70% and 90% reduction is less significant (NWF, 2004).

As discussed previously (Section 5.1) lower rank coals tend to

$/kg Hg removed	ACI, kg/m³	COE increase, mills/kWh	$/kg Hg removed
80–90%*			
152,900	N/A		
325,600			
52,800	92.84	1.20	74,360
140,580		2.45	151,360
191,840	132.83	2.95	226,600
360,800		4.67	358,600
32,780	85.77	1.94	71,060
87,120		3.09	113,300
90%			
8,602	16.54	0.37	13,398
41,800		1.08	39,380
35,860	37.10	1.16	62,700
103,840		2.13	115,500
24,420	38.55	0.99	39,160
93,280		2.35	92,620
13,134	8.83	0.46	15,818
70,620		1.75	60,500
36,300	58.63	1.29	67,100
81,180		1.97	102,080
80–90%†			
47,300	138.93	1.81	54,780
150,040		4.44	134,640
39,380	34.98	1.30	38,060
104,060		3.03	88,220
16,280	26.34	0.91	27,720
119,020		3.54	107,580

injection at Leland Olds

produce less oxidised mercury and therefore mercury control at plants firing these fuels can be more problematic. Pavlish (2007a,b) has studied mercury control at the lignite-fired SaskPower Poplar River Power Station in Canada. The plant is fitted with ESP. An economic analysis was carried out to evaluate the cost of mercury control. The cost of the system, divided into costs for activated carbon, the initial bag set and the fabric filter material, is shown in Figure 8. The efficiency of the baghouse depends upon the surface area available, that is the air-to-cloth ratio. With a higher air-to-cloth ratio, the installed capital cost can be significantly reduced. However, there can be problems with shortened bag-life for air-to-cloth ratios above 6 ft/min (1.83 m/min) which can significantly affect the cost. The balance of plant costs, including foundations, ash handling, ducting, booster fan and so on, amounted to just over $12 million. Over and above this around another $5 million would be needed for project management, construction services and other associated owner/management requirements.

Figure 9 (a and b) shows the estimated costs of sorbent injection upstream of an ESP versus upstream of a baghouse (fabric filter). The cost varies with the level of mercury removal required and the type of coal being burned. Halogenated activated carbons can provide up to 90% mercury removal at <1.00 $/MWh in an ESP system. Costs for non-halogenated activated carbons are greater because they require a higher injection rate. The choice between halogenated or non-halogenated activated carbon is less important with a baghouse (Figure 9b) and the costs are predicted to be almost half that with an ESP system (Srivastava and others, 2006).

Hoffman and Ratafia-Brown (2003) also estimated the cost of activated carbon for different levels of mercury control with and without a COHPAC system for both bituminous and subbituminous coals. The results are shown in Tables 17 and 18. The data are relatively dated and the expertise in both COHPAC and PAC use have evolved since then. However, the tables serve to demonstrate the initial investment required for a COHPAC system and the overall economy due to the reduction in the amount of PAC required for the highest level of mercury control.

Since the demand for activated carbon systems is expected to be relatively high, several specialised technologies have been commercially developed specifically for the developing market. For example, TOXECON™ uses carbon injection after the ESP ahead of a COHPAC bag filtration unit. The economics of the TOXECON™ unit have been reviewed by Carlton and others (2005) and are summarised in Table 19. It is clear that the control costs for eastern bituminous coals will be lower due to their tendency towards easier mercury capture (*see* Section 5.1). The equipment costs are amortised using a capital recovery factor of 0.15, corresponding to an 8% discount rate and 15-year equipment life. The estimate does not include the consideration of changes in ash sales/disposal costs. The results indicate that TOXECON™ is most economically attractive when high mercury removal rates are needed (>80%) (Carlton and others, 2005). TOXECON™ capital costs were projected at 45–55 $/kW for installations with 'moderate retrofit difficulties' (Offen and others, 2006).

TOXECON™ II has been developed for sorbent injection in a cold-side ESP before the last collecting fields. This protects much of the ash (in the first collecting fields) for sale while avoiding the cost of installing a baghouse. Mercury removal of 50–70% has been reported at a plant firing North Dakota lignite with activated carbon injection at 2–5 lb/Macf (32–80 kg/million m³). Longer-term studies were planned. However, the capital costs for TOXECON™ II are predicted to be similar to those for conventional activated carbon injection at about 2–3 $/kW (Offen and others, 2006).

The US DOE has carried out extensive work on activated carbons at full-scale plants using different sorbents. Six plants have provided significant information on the efficiency and cost of mercury capture. Significantly more detailed information can be found on these studies in numerous reports and papers published by the US DOE. Only data relevant to an economic evaluation of these sorbents are discussed here. The efficiency of mercury removal at the different plants varied considerably from the low quantities (<2 lb/Macf (<32 kg/million m³) required for 90% mercury removal at Holcomb to high quantities (>5 lb/Macf;

Table 15   Estimated annualised costs for mercury control for bituminous coal fired plants with cold-side ESP and no sulphur control (Staudt and Jozewicz, 2003)

Retrofit option	Hg control efficiency, %	Annual costs, ¢/kWh	
		975 MWe	100 MWe
Activated carbon injection	90	0.2451 ($0.0025)	0.2639 ($0.0026)
	80	0.1381	0.1497
	70	0.0974	0.1057
Activated carbon injection + polishing fabric filter	90	0.1233	0.1751
	80	0.1171	0.1682
	70	0.1144	0.1650

Table 16   Estimated annualised costs for mercury control for subbituminous coal fired plants with cold-side ESP and no sulphur control (Staudt and Jozewicz, 2003)
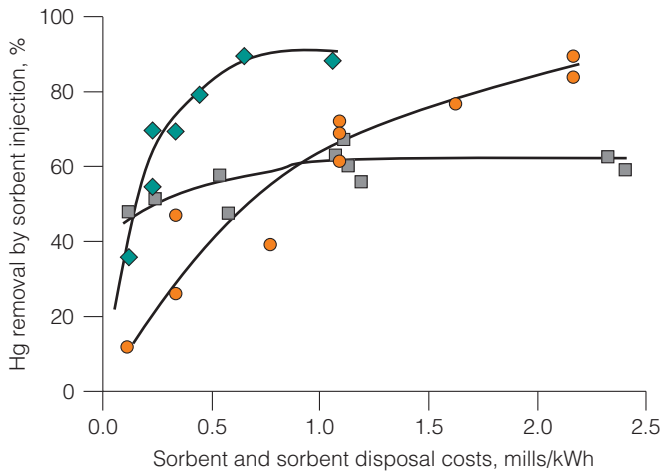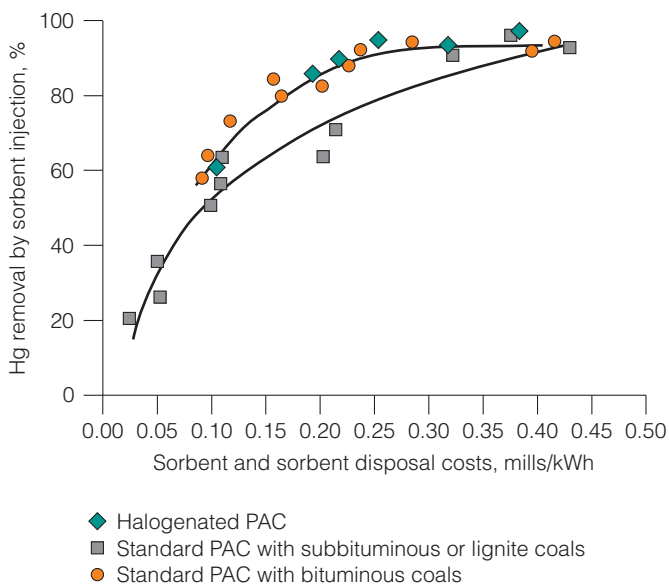
Retrofit option	Hg control efficiency,%	Annual costs, ¢/kWh	
		975 MWe	100 MWe
Activated carbon injection	90	2.0924 ($0.021)	2.1756 ($0.022)
	80	2.0924	2.1756
	70	0.1907	0.2015
Activated carbon injection + polishing fabric filter	90	0.1369	0.1903
	80	0.1236	0.1753
	70	0.1176	0.1685
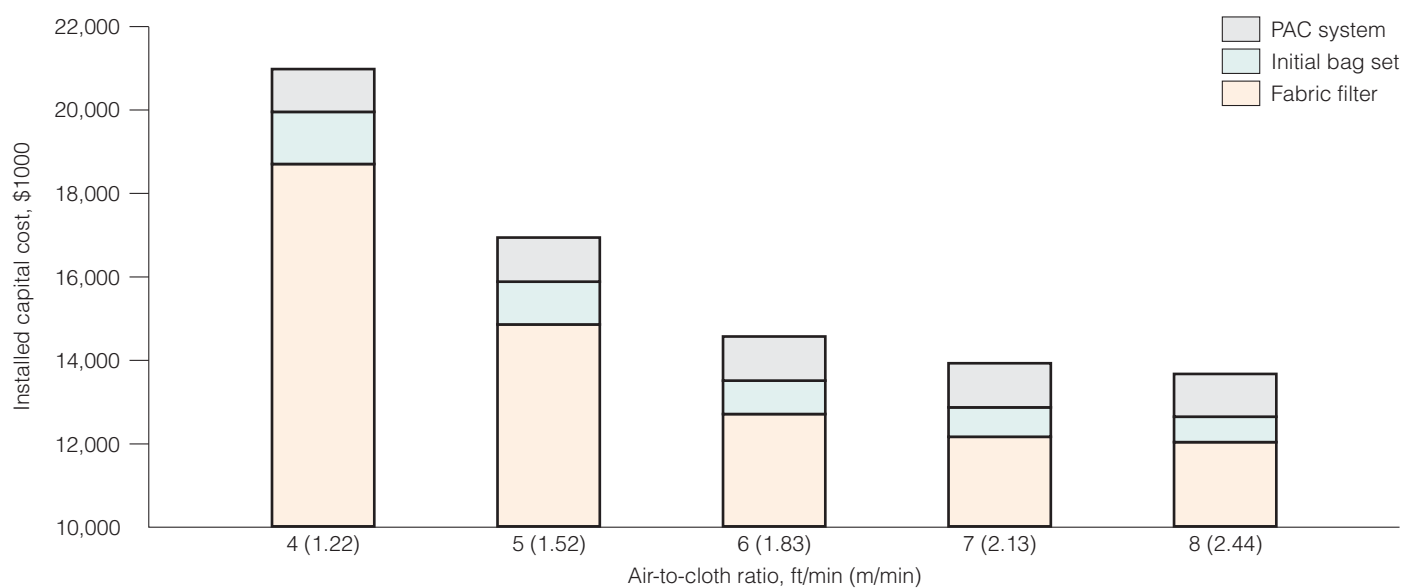
Figure 8   Installed costs for fabric filter and activated carbon system at Poplar River lignite plant (Pavlish, 2007a)

a) cold-side ESP

b) baghouse

◆ Halogenated PAC
▨ Standard PAC with subbituminous or lignite coals
● Standard PAC with bituminous coals

Figure 9   Estimated cost of sorbent injection upstream (Srivastava and others, 2006)

32–80 kg/million m³) required to achieve only 50% removal at Yates.

As expected, the estimate of the cost of activated carbon control at these plants had to factor in such variables as the cost of waste disposal, power consumption, operating and maintenance, labour and spare parts. The price of an activated carbon storage and injection system was included, although it is assumed that the plants already have a suitable particulate control system to capture the sorbent with the fly ash.

Jones and others (2007) factored all these costs into an estimate for mercury control at the six different plants. The results are shown in Table 20. Values for 90% removal at three plants were not available as mercury removal was not possible at this efficiency using the sorbents available. The values in Table 21 do not include the added effect of by-product impacts. Not only does the cost estimate have to include the disposal costs for the waste ash (estimated at 17 $/ton; 15 $/t) but also the lost revenue from fly ash sales to the construction industry (estimated at 18 $/ton; 16 $/t) giving a total of

35 $/ton (31 $/t). This is assumed to be the worse case scenario with respect to addition costs. With this taken into account, the costs are recalculated in Table 21.

The differences in costs are significant. In some cases the sorbent costs accounts for 75% of the mercury control costs and in such cases, any change in the price of sorbent will affect the economics greatly. Further, the amount of mercury capture will depend, as always, on the mercury content of the coal and the background mercury capture through existing flue gas controls and co-benefit effects. Figure 10 shows the effect on the cost of 70% mercury capture of changes in the baseline mercury capture. The cost of sorbent control of mercury can therefore be reduced by enhancing co-benefit effects at the plant. The interested reader is referred to the excellent original report by Jones and others (2007) for further details on the extensive work carried out by the US DOE on the economics of sorbent use at the different plants included in the case studies.

Mercury emissions controls are to be installed, subject to state regulatory approval, at the King and Sherco Unit 3 plants in Minnesota, USA before 2010. The systems will be based on sorbent injection (unspecified sorbent) into existing pollution control systems. Each system is expected to cost $4.5 million to install and $3.8–5.5 million per year to operate and maintain. This is predicted to increase the average energy rate for residential customers in the area by around 16 ¢/month (Businesswire, 2007).

5.4   Other sorbents

For inclusion in a study of sorbents for mercury control, funded by the US DOE, the sorbent manufacturers were required to provide evidence that the cost for removing mercury (per lb mercury removed) would be at least 25% less than that of a standard activated carbon (including production, transportation, handling, feeding and waste handling costs). The manufacturer also had to guarantee that if necessary at least 100,000 t/y of the sorbent would be available to supply to the utility market by 2010 (Apogee Scientific, 2004).

The Apogee Scientific study (2004) demonstrated how variable the mercury capture with different sorbents at different plants could be. For example, sorbents tested at the Midwest's Powerton Station (2x450 MW) achieved 35–85% mercury removal. The variation at We Energies' Valley Power plant was even greater, from virtually no mercury removal to over 95%.

There are many different sorbents coming on to the marketplace. For example, the Southern Research Institute has developed a sorbent consisting of a proprietary oxidant and either hydrated lime ($Ca(OH)_2$) or a silica-modified calcium ($CaSiO_3$) (Feeley and others, 2003).

O'Dowd and others (2004) studied a number of novel sorbents such as fly-ash derived sorbents, 'Nucon' (a sulphur-promoted activated carbon) and granulated alumina. No data were available on the economics of these sorbents.

Table 17   Preliminary costs of mercury control with activated carbon injection for bituminous coal fired plant (Hoffman and Ratafia-Brown, 2003)

	Activated carbon injection for 500 MW bituminous coal fired plant equipped with cold-side ESP		
Mercury removal, %	50	70	90 (with COHPAC)
Sorbent feed rate, lb/Macf (kg/Macm)	2.3 (37)	8.9 (143)	2.4 (39)
Capital cost, thousand $	980	980	28,267
Capital cost, $/kW	1.97	1.97	56.53
Annual O&M @ 80% capacity factor			
Sorbent, thousand $/y	813	3,208	869
Sorbent disposal, thousand $/y	14	55	15
Other, thousand $/y	104	144	2427
Total O&M, thousand $/y	931	3407	3311
Lost ash sale penalty*, thousand $/y	6660	6660	0

*   penalty includes lost sales revenue (18 $/ton:20 $/t) and ash disposal cost (17 $/ton:18 $/t)

Cost and performance assumptions:
– current dollar ($2003) basis, ±30%
– performance based on results of ADA-ES full-scale activated carbon injection testing, except 90% with COHPAC option based on EPRI pilot plant testing
– capital cost for sorbent injection is assumed a 'per installation' cost and is not scaled with sorbent dosing rate
– mercury removal assumes 36% baseline removal across ESP for bituminous coal without carbon injection
– assumes ESP capacity adequate to handle activated carbon loading
– delivered activated carbon cost @0.50 $/lb (1.1 $/kg)
– waste disposal cost @17 $/ton (18 $/t)
– lost ash sales revenue @18 $/ton (20 $/t)
– lost ash sales penalty @35 $/ton (39 $/t) assumes current sale of 100% of fly ash
– 'other' operation and maintenance (O&M) includes: auxiliary power, operating labour and equipment maintenance. COHPAC O&M includes filter bag replacement based on a 5-year life

MerCAP – EPRI's patented Mercury capture by Adsorption Process uses a gold coating on a solid support. The system is periodically heated to regenerate the sorbent capacity and recover the captured mercury. Studies have shown that MerCAP only performed well in the low SO$_2$ environment downstream of an FGD or spray dryer system. MerCap can then act as a flue gas polishing technique without affecting the quality of the ash (Offen and others, 2005).

The sorbents discussed above generally show a reduced mercury capture efficiency at elevated temperatures (>200°C). Granite and Pennline (2006) have patented the use of metal sorbents for mercury control in high temperature systems. The regenerable sorbents will be primarily for use in coal gasifiers, coal-fired electricity generating plants and ore smelters (Granite, 2007). Previous studies have shown palladium to be a potentially effective sorbent for mercury capture. Sorbents of Pt/alumina and Pd/alumina have been shown to capture mercury from synthetic fuel gas feeds at temperatures of 204–371°C. The economics of this system rely heavily on the ease of sorbent regeneration. NETL and Johnson Matthey have initiated a two-year collaborative project to develop and commercialise the sorbents for use in IGCC systems and this will include a detailed cost-analysis of the sorbents (Granite and others, 2006; Granite, 2007).

5.5   Oxidation

As discussed previously, oxidation of mercury significantly improves the capture efficiency. Plants firing coals such as US western coals may need a boost to increase the oxidation of mercury. The simplest way to alter the oxidation state of mercury during coal combustion is through coal switching and blending, as discussed in Section 5.1. For example, blends with small quantities of bituminous coal containing higher levels of chlorine than the baseline subbituminous coals can result in greater baseline mercury capture at a plant. However, for higher levels of mercury capture, stronger oxidation techniques are necessary. Pavlish and others (2005) have reviewed the potential for oxidative treaents at full-scale plants in Canada. Initial studies with halogen injection have proven encouraging with potential costs being around a quarter of those for solid sorbents such as activated carbon.

Since it is known that chlorine in the combustion zone can cause mercury oxidation, many studies have concentrated on the use of chlorine injection or addition for mercury reduction. The injection of chlorine into the boiler could have negative effects on the furnace and downstream, for example by increasing corrosion. However, despite this, chlorine could still be an economic method of mercury control. Chlorine

Table 18 Preliminary costs of mercury control with activated carbon injection for subbituminous coal fired plant (Hoffman and Ratafia-Brown, 2003)

	Activated carbon injection for 500 MW subbituminous coal fired plant equipped with cold-side ESP		
Mercury removal, %	50	60	90 (with COHPAC)
Sorbent feed rate, lb/Macf (kg/Macm)	3.3 (53)	11.9 (191)	3.0
Capital cost, thousand $	984	984	28,719
Capital cost, $/kW	1.97	1.97	57.44
Annual O&M @ 80% capacity factor			
Sorbent, thousand $/y	1369	4930	1246
Sorbent disposal, thousand $/y	23	84	21
Other, thousand $/y	108	151	2596
Total O&M, thousand $/y	1501	5165	3863
Lost ash sale penalty*, thousand $/y	3413	3413	0

*   penalty includes lost sales revenue (18 $/ton:20 $/t) and ash disposal cost (17 $/ton:18 $/t)

Cost and performance assumptions:
– current dollar ($2003) basis, ±30%
– performance based on results of ADA-ES full-scale activated carbon injection testing, except 90% with COHPAC option based on EPRI pilot plant testing
– capital cost for sorbent injection is assumed a 'per installation' cost and is not scaled with sorbent dosing rate
– mercury removal assumes 0% baseline removal across ESP for bituminous coal without carbon injection
– assumes ESP capacity adequate to handle activated carbon loading
– delivered activated carbon cost @0.50 /lb (1.1 $/kg)
– waste disposal cost @17 $/ton (18 $/t)
– lost ash sales revenue @18 $/ton (20 $/t)
– lost ash sales penalty @35 $/ton (39 $/t) assumes current sale of 100% of fly ash
– 'other' operation and maintenance (O&M) includes: auxiliary power, operating labour and equipment maintenance. COHPAC O&M includes filter bag replacement based on a 5-year life

Table 19 Estimated annual activated carbon injection amounts and costs for vapour phase mercury control (500 MWe plant) (Carlton and others, 2005)

Device	Vapour phase Hg removal, %	Estimated annual cost							
		Western coals (WC)				Low sulphur eastern bituminous coals (EB)			
		Capital, million $/y	O&M, million $/y	Total		Capital, million $/y	O&M, million $/y	Total	
				million $/y	levelised, mills/kWh			million $/y	levelised, mills/kWh
ESP	30	0.13	0.8	0.93	0.32	0.13	0.8	0.93	0.32
ESP	50	0.13	1.25	1.38	0.48	0.13	1.7	1.83	0.63
ESP	70	0.13	4.5	4.63	1.60	0.13	4.8	4.93	1.70
ESP	90	0.13	N/A	N/A	N/A	0.13	11.2	11.33	3.91
TOXECON	50	5.07	2.5	7.57	2.61	5.07	2.5	7.57	2.61
TOXECON	70	5.07	3.0	8.07	2.78	5.07	2.6	7.67	2.65
TOXECON	90	5.07	3.4	8.47	2.92	5.07	3.0	8.07	2.78

Assuming 6 ft/m (3 cm/s) air-to-cloth ratio TOXECON fabric filter, 1.3 retrofit factor, no ash sales or other balance of plant impacts
N/A No projections could be made as it is uncertain whether 9% mercury control is achievable at reasonable activated carbon injection rates

addition to the coal or injection into the flue gas has been tested at several plants in the USA including Laskin 2 (firing Powder River Basin coal, fitted with a particle scrubber) and Stanton 10 (firing North Dakota lignite, fitted with spray dry scrubbing and a fabric filter). Although the mercury oxidation did increase, there was also an increase in the opacity of the plant plume and a pressure drop across the fabric filter at the Stanton 10 plant (US EPA, 2005).

**Table 20 20-year levelised cost of mercury control without by-product impacts** (Jones and others, 2007)

Plant	Sorbent	50% mercury removal		70% mercury removal		90% mercury removal	
		COE*, mills/kWh	$/kg Hg removed	COE, mills/kWh	$/kg Hg removed	COE, mills/kWh	$/kg Hg removed
Holcombe	Darco Hg-LH	0.14	9,284	0.18	8,382	0.37	13,332
St Clair	Brominated PAC	0.36	35,640	0.48	33,440	1.06	57,640
Meramec	Darco Hg-LH	0.37	25,960	0.47	23,760	0.99	38,940
Stanton	Darco Hg-LH	0.82	42900	1.02	38,280		
Leland Olds	Darco + CaCl$_2$	0.83	45,320	1.25	48,840		
Plant Yates	Super HOK	0.97	120,120	1.72	152,900		

COE   incremental increase in the cost of electricity
mills   1/10 US cent

Table 21 20-year levelised cost of mercury control with by-product impacts (Jones and others, 2007)

Plant	Sorbent	50% mercury removal		70% mercury removal		90% mercury removal	
		COE*, mills/kWh	$/kg Hg removed	COE, mills/kWh	$/kg Hg removed	COE, mills/kWh	$/kg Hg removed
Holcombe	Darco Hg-LH	0.86	56,540	0.90	42,240	1.09	39,600
St Clair	Brominated PAC	1.36	132,220	1.47	102,520	2.05	111,320
Meramec	Darco Hg-LH	1,75	124,080	1.85	93,940	2.37	93,500
Stanton	Darco Hg-LH	2.57	134,860	2.77	104,060		
Leland Olds	Darco + CaCl$_2$	3.50	191,180	3.92	153,120		
Plant Yates	Super HOK	2.94	365,200	3.69	327,800		

COE   incremental increase in the cost of electricity
mills   1/10 US cent
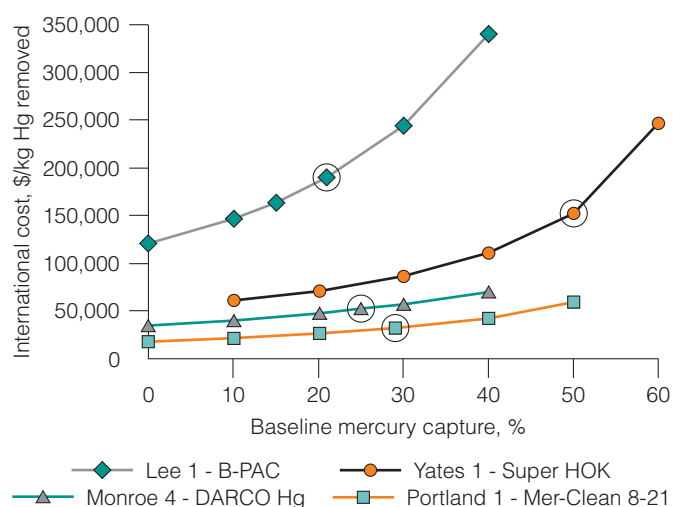
Figure 10 Effect on the levelised cost of 70% mercury control of varying baseline mercury capture (Jones and others, 2007)

Babcock and Wilcox and McDermott Technology Inc carried out full-scale tests of a proprietary liquid reagent to enhance mercury capture in wet FGD systems. The two-year study was carried out at the 60 MWe Endicott Station in Michigan and the 1300 MWe Zimmer Station in Ohio ended in 2002. The results were favourable at the Endcott plant but not at the Zimmer plant, possibly due to the low liquid-to-gas ration in the magnesium enhanced lime wet FGD system (Feeley and others, 2003).

Another halogen, bromine, has been shown to be much more efficient for oxidation of mercury. Bromine can be added to activated carbons to enhance mercury capture. It can also be injected into the coal and/or into the flue gas. Bromine addition has been used to reduce mercury emissions from waste incineration plants in Europe since 2001. Vosteen Consultancy (Vosteen, 2008) have patented bromine injection technology for coal-fired plants. The bromine can be added to the coal prior to combustion as CaBr$_2$. This can be done by either spraying the additive onto the coal on the conveyor during silo charging or on the coal stream from the coal feeders to the mills. If the plant is already fitted with SCR

then high mercury oxidation rates can be achieved at low bromine injection rates. The technology has already been tested as part of the US DOE projects at several sites in the USA including the Holcomb, Meramec and Laramie River stations.

The bromine based oxidation technology has been commercialised via Alstom as 'KNX'. The buyer must pay a 'one-time licence fee' to adopt the patented process. After that, the only remaining cost is that of the bromine additive (Vosteen, 2008). The cost of bromine injection has been estimated at around 400,000 $/y (at 25 ppm in the coal) which is only 20% of the cost of activated carbon.

Berry (2007) presents results from demonstrations of the efficiency of bromine oxidation on mercury capture. Testing was performed at the 4x700 MWe Miller plant in Alabama, USA. The plant is already fitted with SCR for NOx control and wet FGD is planned. To install a TOXECON™ system at this plant would cost US$88–123 million. In addition, there would be operating and maintenance costs of $2 million per year for activated carbon, $1.1 million per year for filter/bag replacement costs and 10 $/ton (or 30,000 $/y) on solid waste disposal costs. There would also be a reduction in plant efficiency for running the TOXECON™ system. Testing at the plant with bromine injection ($CaBr_2$) with the coal at Unit 4 showed significant mercury oxidation at 'very low' bromine injection rates. The chemical costs of $CaBr_2$ are one fifth of the cost of activated carbon (only 400,000 $/y as opposed to an estimated $2 million per year). Further testing is planned. However, studies have shown that the use of Br can affect the 28-day strength of concrete containing fly ash from this process. This would add disposal costs to the plant in addition to lost revenue. Further studies are underway to evaluate the extent of this problem.

Chem-mod™ is an oxidising liquid sorbent for mercury control combined with a powder sorbent for SO_2 and other heavy metal capture. Around 90% mercury removal has been shown in short-term tests with different grades of bituminous and subbituminous coals. The system is claimed to be cheaper than scrubber systems. For a 200 MW plant the Chem-mod™ system would cost $2–8 million compared to $60–80 million for a scrubber (Patel-Predd, 2006).

Ellison (2005) lists a range of available oxidising agents including oxygen, ozone, hydrogen peroxide, chlorine hypochlorite, chlorine dioxide, potassium permanganate and ferrous iron. The costs for these chemicals are around 0.6–1.80 $/lb (1.33–4.00 $/kg).

In addition to oxidising chemicals and solutions, solid catalysts are also being developed for mercury oxidation. Presto and Granite (2006) reviewed oxidation catalysts, of which there are three types:
- SCR catalysts, which can oxidise mercury as a co-benefit (*see* Section 4.3);
- carbon-based catalysts – either on unburnt carbon or injected activated carbons (*see above*);
- metal and metal oxide catalysts, such as palladium. The cost of precious metals such as palladium for mercury oxidation should not prove prohibitive since they can be

effective at low mass loading. For example, a mass loading of 1% iridium catalyst (99% alumina) was sufficient to oxidise 70% of the mercury in a simulated flue gas. Further, the catalyst can be regenerated by heating and purging with either N_2 or CO_2. Preliminary cost analyses show that palladium catalysts coupled with FGD offer a 63% cost saving over activated carbon/COHPAC for an overall 80% mercury removal. The cost saving drops to 9% for 90% mercury removal. Further testing is under way.

Hrdlicka and others (2007) have studied the use of fabric filter coatings to enhance mercury oxidation. Materials such as palladium and titanium dioxide have shown promising results. Testing is ongoing at pilot scale and no cost estimates are yet available.

Photochemical oxidation (PCO) uses 254 nm ultra-violet (UV) light to oxidise mercury in the flue gas (Granite, 2007). Powerspan's ECO (electro catalytic oxidation) is a similar process employing electro-catalytic oxidation followed by a scrubber and a condensing wet ESP. ECO is already being tested on the 50 MWe R E Burger Plant in Ohio (Ellison, 2005). Electron beam technology is also being developed for multi-pollutant control. Estimates for the capital cost of such technology (E Beam) is <200 $/kW, providing simultaneous removal of SO_2 and NOx. This is reported to be lower than the cost of FGD and SCR individually. In addition, the technology can reduce mercury emissions. The E Beam process is a dry technology, thus it does not add to waste disposal issues. The reagent ammonia product is converted to a high value fertiliser stock. The mercury capture efficiency of the E Beam process varies with flue gas chemistry but can be enhanced, if necessary, with the addition of chemical oxidising agents (Ellison, 2005). PEESP™ is a patented Plasma Enhanced ESP technology which is still under development (Nalbandian, 2006).

5.6   Other techniques

As mentioned in Sections 4.5 and 4.5, unburnt carbon in ash can be a useful 'natural' sorbent for mercury control in some systems. The Thief Process is a patented carbon sorbent based process which takes advantage of this effect. The sorbent is prepared at the plant by extracting a mixture of partially combusted coal and gas from the boiler and then reinjecting this downstream. This means that there is no cost for sorbent other than the loss of fuel and any loss in combustion efficiency. The effective cost of the Thief Process was estimated based on the annualised operating costs associated with the thermal heat rate penalties and parasitic power requirements. Whilst activated carbons are cited to range from 500 $/ton to 3000 $/ton (446–2676 $/t), the Thief Process sorbents are significantly cheaper at 90–200 $/ton (80–178 $/t), around 80% less. However, operating and maintenance costs for the Thief Process are higher than for activated carbon systems because it is a less mature technology. There is also a potential loss in ash sales due to the increased unburnt carbon in the ash. A comparison of the cost of an activated carbon system versus the Thief process for a theoretical plant are shown in Table 22. The Thief

Table 22 Economic evaluation of ACI and Thief Process (Granite, 2007)		
	ACI	Thief process – base case
Total control capital cost, $		
Equipment cost, $	462,800	200,00
Freight, $	incl	10,000
Taxes	34,968	21,600
Field materials, $	120,000	160,000
Field labour, $	85,000	300,000
Indirect field costs, $		21,000
Subtotal, $	702,768	712,600
Retrofit factor – Thief 1.20, $		142,520
Bare Installed Retrofit Cost (BIRC), $	702,768	855,120
Engineering and home office fees (ACI -10%, Thief - 20% of BIRC, $	70,277	171,024
Process contingency (ACI - 5%, Thief - 15% BIRC), $	35,138	128,268
General facilities costs (5% BIRC), $	35,138	42,756
Project contingency (15% BIRC and indirect costs), $	126,498	179,575
Total, $	969,820	1,376,743
Total, $/kW	1.94	2.75
Pre-production/shakedown costs – shakedown ACI 2 weeks, Thief 1 month		
Fixed operating cost, $	4951	17,603
Variable operating cost, $	53,866	23,091
2% total capital cost, $	19,396	27,535
Total capital requirement (TCR), $	1,047,673	1,444,972
Total capital requirement (TCR), $/kW	2.10	2.89
Fixed O&M		
Operating labour, 45 $/h	70,200	140,400
Maintenance and materials (5% BIRC)	35,138	42,756
Admin and support labour (20% of operating labur)	14,040	28,080
Total	119,378	211,236
Variable O&M costs		
Sorbent (ACI - 1000 $/t, Thief - 186 $/t)	1,365,000	253,890
Incremental power (0.05 $/kW)	12,300	N/A
Waste disposal (17 $/t)	23,206	23,206
Total	1,400,506	277,096

Levelised cost summary - constant $	ACI			Thief Process - base case		
20 years	$	mills/kWh	$/kg Hg	$	mills/kWh	$/kg Hg
Fixed charges	174,123	0.050	2284	240,154	0.069	3150
Fixed O&M	119,378	0.034	1566	211,236	0.060	2772
Variable O&M	1,400,506	0.400	18,372	277,096	0.079	3634
Total	1,694,008	0.483	22,222	728,487	0.208	9557

Process has only been tested at pilot scale of up to 0.5 MWe. However, even at this developmental stage, it would appear that the Thief Process offers significant cost savings compared to activated carbon injection (O'Dowd and others, 2006; Granite and others, 2006; Granite, 2007).

5.7 Selection

As can be seen from the examples included throughout Chapters 4 and 5, there are numerous options for controlling mercury emissions from coal-fired plants. Ideally, the most economic method of mercury control is to maximise mercury capture in existing flue gas technologies – that is, maximising co-benefit effects. However, for many plants, especially those firing lower grade coals, mercury specific control technologies will be required. Even then, variations in coal chemistry, plant operation and so on mean that 'one size does not fit all' when it comes to selecting mercury control equipment.

At the moment, there is no BAT/MACT defined for mercury in any legislation. This may change in the future, especially with the obligatory rewrite of the CAMR in the USA (*see* Section 3.1.2). However, for the moment, the technology to be used at each plant must be developed on a case-by-case basis.

The US DOE/NETL have an extensive list showing the deployment of mercury control technologies in the USA (Feeley, 2008). The majority of plants have opted for activated carbon-based systems with or without bromine addition. Mercury control is planned for a total of almost 30 GW of capacity as of September 2007 which represents a total of 10% of the total US capacity. Within this, eight units using activated carbon systems are fully operational (~2750 MW). 70 units, including one unit in Canada, are taking action including:
- 50 units (21 GW) firing subbituminous coal;
- 14 units (5 GW) firing bituminous coal;
- 3 units (2 GW) firing lignite coal;
- 3 units (2 GW) firing coal blends.

It would therefore seem that, by default, activated carbon is defining itself as the BAT/MACT for mercury in the USA. Since, as mentioned previously, the US DOE is no longer funding further research into mercury control technologies, it is possible that activated carbon will remain the most common approach. Plants adopting mercury control technologies in future are likely to look at the current statistics and opt for activated carbon because of the amount of experience obtained so far. Any new techniques may need to invest in marketing and promotion to break into the marketplace.

To some extent, the selection may be simplified by studying cost comparisons such as that produced by Srivastava and others (2005). The results are summarised in Table 23. The table is NOT intended to give guidelines on the best technologies for each plant or coal type but rather to give an indication of the most common plant/coal configurations and the estimated costs at each. As discussed before, the costs cited are to be taken as relative rather than exact due to changes in costs over time. If anything, Table 23 emphasises the need for expert advice on the most appropriate technology for each plant on a case-by-case basis.

It must also be remembered that many of the systems

discussed in this report and available in the market place are still largely developmental. The US DOE stated in 2006 that *there remain a number of critical technical and cost issues that need to be resolved through additional research before these technologies can be considered commercially available for all US coals and the different coal-fired power plant configurations in operation in the United States*. The US DOE's field testing programme has been limited to testing at 28 coal-fired units which represents only 2.3% of the 1165 units in operation in the USA (NETL, 2006). According to Stadler (2005) the US EPA predict that only 4% of the 430 coal-fired plants in the USA will have installed mercury specific controls by 2020 (NETL, 2006).

As discussed in Chapter 2, the Canadian CWS sets reduction targets for each province and it is up to the provincial authorities to work with the utilities to determine where and how reductions will be made. Whereas Ontario has chosen the extreme route of phasing out all coal-fired units, areas such as Alberta are considering several different control technologies for each plant.

Pavlish and others (2005) at the Energy and Environmental Research Centre (EERC) in North Dakota, USA, have carried out a study of the technology options available to the different coal-fired plants in Canada, including activated carbons, other sorbents, and oxidation technologies. It was concluded that most of the plants would benefit from most of the technologies considered. However, one of the priorities for establishing suitable options for each plant was the availability of the technology by 2009. This meant that the best approaches were:
- fuel switching (however, not appropriate for many plants);
- conventional coal cleaning (may not be economic at some plants considering the low coal Hg content at most plants);
- activated carbon;
- activated carbon with a newly fitted baghouse;
- scrubbers/FGD (not economic as an option for mercury control alone).

Other technologies such as oxidation techniques showed potential but could take longer to demonstrate and would be considered depending on individual commercial deployment time-lines.

The EERC study (Pavlish and others, 2005) concluded that, similarly to the situation in the USA, activated carbon was the most promising near-term option to achieve >50% mercury reduction. The estimated cost range for various plants in Alberta, Manitoba, New Brunswick, Nova Scotia and Saskatchewan are shown in Table 24. These estimates are first-year costs and are not levelised. They do not account for losses/costs due to changes in fly ash use either. The report stressed that changes in ash use could increase costs by a factor of 2 to 4. The loss of any CO_2 offsets in trading schemes is not included.

The CWS allows for early actions such as reductions in emissions prior to the start date to be included in each Province's total reduction. For example, early action by

Table 23 Preliminary estimates of costs (2003 constant $) of mercury controls to achieve between 80 and 90% reduction of mercury across existing and, if needed, additional controls (Srivastava and others, 2005)

Coal type	S%	Boiler size range, MW	Existing control configuration†	Additional controls‡	Cost estimates of additional controls, mills/kWh
Bituminous	3	300–975	CS-ESP + wet FGD	PAC + PJFF + CEMS PAC + CEMS	1.144–1.430
Bituminous	3	300–975	SCR + CS-ESP + wet FGD	CEMS	0.003–0.004
Bituminous	3	300–975	FF + wet FGD	CEMS	0.003–0.004
Bituminous	3	300–975	SCR + FF + wet FGD	CEMS	0.003–0.004
Bituminous	3	300–975	HS-ESP + wet FGD	PAC + PJFF + CEMS	1.149–1.437
Bituminous	3	100	SD + CS-ESP	PAC + PJFF + CEMS	1.749–3.096
Bituminous	3	100	SD + FF	PAC + CEMS	0.005–0.370
Bituminous	0.6	100–975	CS-ESP	PAC + PJFF + CEMS	1.171–1.751
Bituminous	0.6	100–975	FF	PAC + CEMS	0.003–0.510
Bituminous	0.6	100–975	HS-ESP	PAC + PJFF + CEMS	1.205–1.804
PRB*	0.5	100–975	CS-ESP	PAC + PJFF + CEMS	1.236–1.903
PRB*	0.5	100–075	FF	PAC + CEMS	1.122–1.266
PRB*	0.5	100–975	HS-ESP	PAC + PJFF + CEMS	1.273–1.960

*   PRB Powder River Basin coal
†   CS-ESP cold-side ESP; HS-ESP hot-side ESP; FF fabric filter; PS   particle scrubber; SD spray dryer;
     SCR selective catalyst reduction;
‡   PAC powdered activated carbon; CEMS continuous emission monitoring system; PJFF pulse-jet fabric filter

Saskatchewan between 2004 and 2009 will be used to meet its provincial caps for the years 2010-13. Early actions so far have included a mercury switch collection programme and early mercury controls at the Poplar River Power Station (CCME, 2006).

The province of Alberta has required that all plants submit details of how they will each achieve 70% mercury reduction. Transalta owns three coal-fired units in Alberta and part-owns a further two. The company has carried out extensive studies to determine how best each plant can achieve the required mercury control (Omotani, 2007). All the plants have ESP and burn western bituminous coal with low sulphur, low chlorine, low mercury and high ash contents. Studies have shown that the majority of mercury in the flue gas at the plants is elemental and therefore difficult to capture, with 25% or less being caught in the ESP system.

Preliminary studies in 2005 showed the following (Omotani, 2007):
● treated/enhanced sorbents performed better than activated carbons;
● coal cleaning reduced the coal mercury content by 40%;
● enhanced sorbents performed well with the cleaned coal whereas standard activated carbons only achieved 10% reduction in flue gas mercury;
● injection of sorbents and enhancements upstream of fabric filters improved mercury capture at lower injection rates but capital and operating costs for fabric filters would be significant and would not reduce overall mercury removal costs.

The effects of combustion conditions on 'natural' mercury removal in the ash were also considered. The 'native' mercury removal rate at the plant varied from 15% to 30% with an

Table 24 Summary of estimated technology cost ranges for plants in Canada (Pavlish and others, 2005)

Coal type	Activated carbon + ESP		ESP + activated carbon + fabric filter	
	First year		First year	
	1000 US$/y	mills/kWh	1000 US$/y	mills/kWh
Bituminous (including blends)	750–1480	0.63–1.23	2400–9000	1.98–2.68
Subbituminous	950–4700	0.74–1.54	2600–9100	2.25–2.89
Lignite	450–7000	0.85–3.29	5000–6800	2.31–3.09

average of 19%. By increasing the 'natural' mercury capture (co-benefit effect) the requirement for activated carbon was reduced by 20–30%.

Once the plant has had the new fabric filter system, booster fan and activated carbon injection system installed, there will be ongoing costs for operation and maintenance. These have been estimated as follows:

Sorbent cost (delivered)	1.21 $/kg
Maintenance labour rate	30 $/h
Fly ash disposal cost	6 $/t
Energy cost	0.035 $/kWh
Overhead:	30% of operating and maintenance (including spare parts)
Filter bag cost	75 $/bag
Bag change-out rate	4 bags/h (during bag replacement)

The operating life of the project was estimated at 20 years. With all these costs and variables taken into account, it was estimated that the total cost of mercury control would be between $41,800 and $37,400 kg/Hg removal, depending on the air-to-cloth ratio used in the baghouse.

5.8 Comments

Knowledge of coal quality, coal chemistry, pollution control device behaviour and so on can promote optimisation of mercury capture as a co-benefit. However, this would need to be done on a unit-by-unit, coal-by-coal basis and would involve expert consultants. Computer modelling might well provide guidance on how individual plants can enhance their mercury capture with the systems they already have in place. By enhancing co-benefit/baseline mercury capture in a plant, the requirement for additional, more costly, methods of mercury control can be reduced.

For many plants the amount of mercury reduction achievable through co-benefit effects may not be enough to meet legislative demands. In these cases, more costly mercury specific control techniques and technologies are required. These include techniques such as activated carbons or sorbent injection, oxidation methods and electrochemical methods. Mercury control with activated carbon is dependent on several factors including:
- sorbent costs;
- impacts on ash sales and disposal costs;
- plant-specific variables such as coal mercury content and baseline mercury capture.

The marketplace for mercury control technologies is still young. At the moment the techniques are not yet required on most plants. The technologies are being developed ahead of the more stringent reduction targets expected in the new US EPA mercury specific legislation and the CWS in Canada. Since many of these techniques are still at the developmental stage in pilot studies, the cost analyses are somewhat premature. Even with a reduction in US DOE funding for these mercury-specific projects, the market will now develop in accordance with demand.

In the immediate future in North America, technologies and sorbents may be relatively inexpensive as manufacturers attempt to establish themselves in a young marketplace. However, as time goes on and demand increases, prices may well increase. A sudden increase in demand for sorbents and control systems may also result in a shortage of expertise, equipment, installation staff and so on which could also affect the economics and success of projects.

For the most part, the technologies have an equal chance to find a niche in the market, although early demonstrations may prove more effective. However, should legislation change to specify BAT/MACT then the market will change dramatically in accordance with the technology defined as BAT or MACT. At the moment, it is commonly assumed that there is no single BAT/MACT for mercury on coal-fired plants. Should the new US EPA CAMR-replacement result in a requirement for MACT then the US EPA will face a tough challenge to select the most appropriate technologies and the market could change dramatically.

6   Mercury control in emerging nations

Is it well established that mercury is a global problem. Further, although emissions from developed nations are being reduced, emissions from developing countries and emerging nations continue to rise. Figure 11 shows the mercury emissions from different global areas and Figure 12 shows the relative emissions from US power plants compared to emissions from all other countries. Bearing in mind that emissions from the EU and Canada are of a similar order of magnitude to those from the US, there is a strong argument that mercury control in other areas of the world need to be tackled. The UNEP Global Mercury Partnership, as discussed
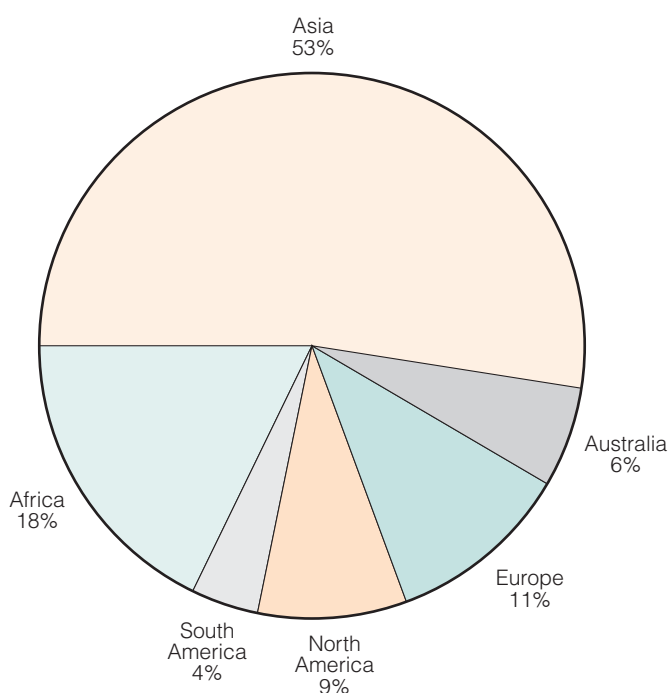
Figure 11   Mercury emissions from different global regions (US EPA, 2008)
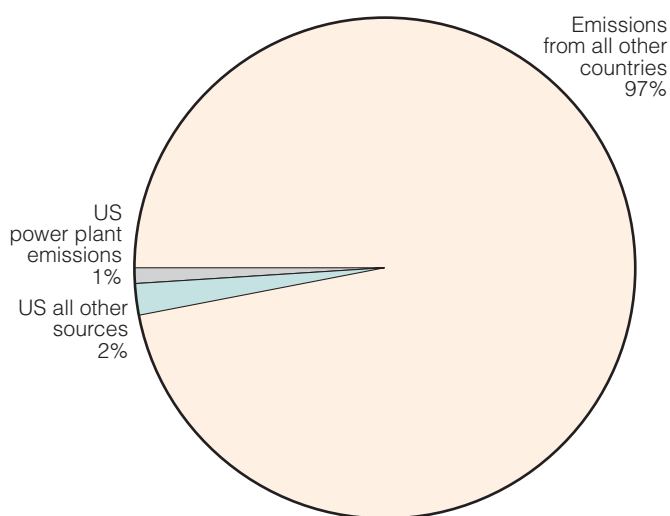
Figure 12   Mercury emissions are a global problem (US EPA, 2008)

in Chapter 2, aims to work together with developing nations and emerging economies to promote the most cost-effective and practical means for mercury control in these areas.

As discussed in Chapters 4 and 5, site-specific differences in plant characteristics and fuel mean that there is no single option (BAT/MACT) for mercury control. The cost of mercury control can be minimised if the plant operator is free to choose the most appropriate technique or technology from a number of different mercury control options. The problem with this approach in emerging nations is that:

● there is often little or no legislative requirement or environmental momentum to reduce mercury emissions;

● there is often little or no experience in maximising co-benefit effects or selecting control techniques and technologies for mercury;

● there is often little or no funding for mercury control.

This chapter looks at the opportunities to reduce mercury emissions efficiently and economically at coal-fired plants in developing countries and emerging nations.

Ideally, existing coal facilities could be replaced by cleaner more renewable sources of energy or more efficient coal combustion systems. However, in most developing countries, power is a priority, coal is cheap and plentiful and there is experience with coal combustion.

As coal will continue to dominate as the fuel source in places such as China and India, mercury control options would be required at existing plants as well as in future capacity. The majority of the information discussed in this chapter relates to emissions from coal combustion in China. This simply reflects the amount of literature available which concentrates on this region and is not intended to imply that China is the only country with such issues. However, the situation in China may reflect, to some extent, the situation in other developing countries. The political and legislative situation in China is very different from that in Europe and North America and this will lead to very different problems and challenges when considering how best to reduce mercury emissions. A separate report from the IEA CCC (Minchener, 2007) gives an excellent summary of the challenges facing the coal sector in China. Much of the information included below, unless otherwise stated, comes from Minchener's report.

6.1   Coal options

Although some coals in China may have a slightly higher mercury concentration than US coals (*see* Section 5.1), it is the lack of emission controls on utilities rather than the coal mercury concentration which is primarily responsible for the per-tonne mercury emission rate in China being three times that for the USA utilities (Kolker and others, 2006).

China is installing new coal-fired boilers at an unprecedented rate to supply energy to a rapidly-growing population. As

mentioned in Section 5.1.2, coal cleaning could be a simple and economic way to reduce mercury emissions in many regions such as Guizhou.

According to Minchener (2007) the majority of coal that will be mined in the future will come from two regions – either Shanxi, Inner Mongolia and Hebei, or Shaanxi, Gansu Ningxia, Qinghai and Xinjiang Provinces. At the moment there are three main categories of mines:

- key state-owned mines that commonly have relatively advanced coal extraction techniques. Output from these mines goes to other state-owned enterprises;
- state-owned mines that tend to be smaller and only partially mechanised but with a similar customer base;
- township and village mines that mostly use manual extraction techniques. In the past the output of these mines went mainly to local customers. However, the situation has now changed and they are also important suppliers to the power and non-power industrial sectors.

As discussed in Section 5.1, Chinese coals have very different concentrations of mercury and, theoretically, emission reductions could be achieved by coal switching. In practice, this is unlikely to happen in the near future due to the lack of legislative requirement and likely excessive cost.

6.2   Maximising co-benefit effects

As the developing world takes steps towards reducing pollutant emissions, some mercury reduction will be achieved quite economically. For example, typical plant efficiencies in countries such as China are reported at around 29% as compared with 36–38% in OECD countries (average for sub and supercritical, old and new units) (Minchener, 2007). Improvements in plant efficiency would reduce emissions of all pollutants simultaneously.

There appears to be some mismatch between the pollution control targets within China and those actually being achieved. As with other developing nations, China has been characterised with a low rate of adoption of clean coal technologies in the past for several reasons including (Newman-Sutherland and others, 2001):

- strong competition for limited capital;
- water shortages;
- limited demand for better quality coal;
- lack of a fair pricing regime to reflect the value of coal quality;
- a belief that there must be an economic as well as a social/environmental benefit for the implementation of a technology transfer project.

The success of mercury control in developing countries such as China will depend not only on the establishment of reduction strategies but also the policing of these strategies to ensure that they are enacted. One official from the Chinese Environmental Protection Agency is reported as saying: *The economy has grown at an unexpectedly rapid rate in some local areas, mostly at the cost of the environment, and the supervision departments at the grass-roots level are barely functioning* (Edie, 2007). This would imply that supplying

power is superseding environmental legislation and that emission limits may be going unchecked. The success of the predicted co-benefits will be totally dependent on the successful implementation of SO_2 and NOx control technologies.

Despite this, China is making significant moves towards reducing pollutant emission. China's near-term approach to address pollution control issues whilst allowing massive expansion in coal use is as follows (Minchener, 2007):

- the introduction of large pulverised coal-fired units with supercritical and, in the future ultra-supercritical steam conditions for high efficiencies, together with improved ESPs, FGD and some form of NOx control. As discussed in previous chapters, these are the types of control systems that could provide co-benefit mercury reduction at little or no extra cost;
- retrofitting and upgrading of existing coal-fired power plants that currently operate with low efficiencies and minimal environmental controls. The Government has declared that, through its National Development and Reform Commission (NDRC), it will 'rehabilitate' medium size (200–300 MWe) power plants to improve efficiency with retrofit of environmental control systems.

However, since power demand now exceeds supply in China, these medium-sized plants must remain operational and so there has been no drive to take them out of service for the extensive modification required.

In 1998, the 'Two control zones' policy in China defined two areas for remedial measures (Minchener, 2007):

- acid rain control zone, where rainwater had a pH value less than 4.5 (around 9% of the total land in SW and SE China);
- SO_2 control zone, where the average concentration of SO_2 exceeded the national standard of 0.06 mg/m^3 and the daily average exceeded 0.2 mg/m^3 (around 3% of the land).

In 1995, the SO_2 emissions from the power plants within these two zones accounted for almost 50% of the country's total emissions. Therefore targeting these areas would provide the fastest and most economic means of reducing emissions.

Economic approaches are being taken to promote FGD installations in the two control zones (Minchener, 2007):

- FGD installation will be supported by 'favourable funding' (loan) conditions;
- the cost of FGD will be reduced through technology transfer and market forces;
- thermal plants with FGD installed will receive a greater price for electricity supplied to the grid;
- the SO_2 tax will be increased and a low SO_2 emission limit established.

With these measures, China could achieve significant SO_2 reductions and significant mercury reductions as a co-benefit. The success of these measures could prove an extremely useful example to other developing nations as to what can be achieved.

Table 25   Projected mercury emissions and potential reductions in the Chinese power sector (Wu and others, 2006)

Year	2003	2010*	2020*	
			Scenario1†	Scenario 2†
Baseline mercury emissions, t	91	95	207.2	207.2
Projected mercury emissions, t		55.6	105.5	92.9
Potential total mercury reductions, t		61.2	101.6	114.3
Reduction by technology				
by FGD, t	1.4	55.6	56.7	30.5
by activated carbon injection			3.3	6.6
by FGD+activated carbon			28	41.7
by FGD+SCR, t			8	20.4
by FGD+activated carbon+SCR, t			5.6	15
* baseline values for 2010 and 2020 are for particulate controls only				
† Scenarios 1+2				

Control technologies	Scenario 1		Scenario 2	
	Region 1	Region 2	Region 1	Region 2
ESP + activated carbon, %	0	5	0	10
ESP + FGD + activated carbon, %	20	10	30	15
ESP + FGD + SCR, %	5	5	15	10
ESP + FGD + activated carbon + SCR, %	5	0	10	5
Note: Region 1 developed provinces such as Beijing and provinces with high emissions, such as Ghizou; Region 2 other provinces				

By the end of 2006, 30% of Chinese coal-fired plants had FGD systems installed and this had increased to 36% by mid-2007. Further, several small, inefficient power plants have reportedly been closed down in an effort to reduce national emissions of the major pollutants (SO$_2$ and NOx) (Edie, 2007). However, according to Wu and others (2006) the installation rate of FGD in China is still only 3% of the total coal-fired generating capacity. Despite this, Wu and others predict an annual growth rate in coal-based electricity in China of 7.5% between 2003 and 2010 and 3.8% between 2010 and 2020. Based on the Chinese Control Plan for Acid Rain and SO$_2$ Pollution Control Zone, Wu and others (2006) predict that the penetration of FGD installation would reach 58% in 2010 and 67% in 2020, nationwide.

Wu and others (2006) suggest that, by 2010, the penetration of FGD technologies in China could keep mercury emissions close to those in 2003, that is by offsetting any increase in emissions due to increased coal use. However, by 2020 only a relatively high rate of advanced control technology use (such as activated carbon, FGD and SCR) would be able to stop a growth in mercury emissions, as shown in Table 25. Scenario 1 predicts a lower adoption of mercury control than Scenario 2. Both scenarios predict that developed provinces, such as Beijing, and those with high mercury emissions, such as Ghizou, will adopt a more stringent approach to mercury control.

Little or no data are available for mercury emissions in other developing economies and economies in transition such as India, Korea, Africa. It must be assumed that mercury is not on the legislative agenda in these areas. It is also likely that actions to reduce emissions of primary pollutants such as SO$_2$ and NOx are significantly less than those in Europe and North America and even less than those in China. Co-benefit effects must therefore be assumed to be minimal. There is therefore potential for significant and economic mercury reductions to be achieved in these countries with the adoption of standard pollution control technologies. The steps being proposed in China could go a long way to demonstrating what could be achieved elsewhere.

6.3   Mercury specific controls

For the most part, it must be assumed that developing nations and economies in transition will achieve the majority of mercury emissions in the short term through co-benefit routes. Mercury is unlikely to be targeted for control due to the potentially large costs that would be associated with doing so. Mercury specific controls in these areas, in the near- to mid-term future, are most like to be as a result of a globally legally binding mechanism such as that outlined in Section 2.1.2.

Despite there being no legally binding reason, it is possible that China could take steps towards specific mercury control options. Wu and others (2006) are optimistic that mercury control technologies such as activated carbon could be commercially available by 2010 and may well be adopted in some Chinese plants after that time. They suggest that a combination of ESP and/or FGD and activated carbon could be installed on up to 35% of plants in the more developed areas of China (such as Beijing) by 2010. Wu and others predict that activated carbon, alone or in conjunction with FGD and/or SCR, could be used on up to 30% of plants in some regions of China by 2020 (*see* Table 25).

Whether these predictions are correct remains to be seen. It will be especially important to identify where funding for such controls will be obtained. However, if the predictions are correct, then this would be a significant step to reduce emissions from China and would be an excellent example to other developing nations as to what could be achieved.

6.4   Facilitating mercury reduction

The choice of mercury control approaches at the remaining plants in the USA, Canada and perhaps other countries in the future, remains to be seen. Ideally, an interactive flow chart similar to Figure 5 in Chapter 4 could be produced which would help individual power plants, especially those in developing countries, establish the most economic and effective means of controlling mercury emissions at each individual plant. There is an ongoing project with UNEP to try and put this idea into practice.

Much could be achieved by the simple transfer of expertise and knowledge to developing countries. China, Canada, Japan, the USA and UNEP jointly held a workshop in Beijing in November 2005 on measurement and control of mercury from coal-fired power plants. The workshop increased awareness of the magnitude of mercury emissions from this sector, examined limited data currently available on the level of mercury exposure in China, and provided information on control approaches (funded through Mercury Trust Fund, and also bilaterally funded through Canada, Japan, USA). Further workshops of this type are planned.

The Governments of Canada and China, and Tsinghua University have co-operated on a study to compare the current China Mercury Emission Inventory with the UNEP mercury emissions toolkit, examine the status of coal washing technology and mercury removal in China, and examine coal combustion-related mercury emissions from small-scale use in residential, commercial, and industrial sectors. Further projects are planned to use established coal modelling techniques to study the potential for mercury control through coal washing and coal switching at a small number of coal-fired facilities in China.

Investment in China could go a long way to demonstrating what can be achieved. Other emerging economies such as India will also need support to achieve mercury control.

6.5   Comments

At the moment, as a country China is the largest emitter of mercury from coal combustion and therefore it is not surprising that most of the data in the literature concentrates on how emissions from China could be controlled. Despite problems with matching actual pollutant reductions with target reductions in the past, it would seem that China is taking important steps towards reducing emissions of SO_2. The regions where almost half of the SO_2 emissions arise have been identified and control measures will be concentrated in these areas. Economic measures such as technology transfer, reduced loan rates, preferential energy tariffs and modified emission fines could go a long way to ensuring the sulphur reduction targets are met. This could mean a 30–80% reduction of mercury emissions at these plants as a co-benefit effect.

The success of these measures in China could act as an example to developing countries and economies in transition of how multi-emission control strategies and maximising co-benefit reductions for several pollutants can be cost-effective.

7   Conclusions

Mercury is a global pollutant. Some countries, such as the USA and Canada, are taking legislative steps to ensure that mercury emissions from coal combustion are reduced. Other countries, such as Japan and most of the EU, have already achieved significant mercury reductions due to the co-benefit effects of pollutant control systems for particulates, SO_2 and NOx. These countries are keeping a watching brief on the mercury issue and may set mercury specific legislation in the future if they feel it is warranted. However, for the upward trend in global concentrations of mercury to be controlled, action will also need to be taken in other countries. Based on the information reviewed throughout this report, it would seem that there are several options that, either alone or in combination, could reduce the global mercury burden. These include:

- Legislative action: USA and Canada are the only countries that have set or are in the process of setting legislation specifically for mercury control. This approach will be costly but is likely to result in health and environment benefits which should far outweigh the costs. Developing countries are unlikely to place mercury high on their national agendas in the foreseeable future; security of fuel supply, providing power to growing populations and dealing with the more 'major' pollutants, such as SO_2 and NOx will take priority. International action, such as that proposed by the UN Environment Programme, could provide the impetus for action much sooner and could also provide economic resources to ensure compliance.

- Improvements in the understanding of the mercury problem: more accurate emission inventories will identify areas of greatest emissions and therefore of greatest concern. Targeting the most polluting sources first could result in the greatest mercury reduction for the least investment. More accurate emission data will also facilitate better regulation and monitoring of the success of reduction strategies.

- Changes in fuel use: switching from mercury-containing fuels, such as coal, to fuels that do not contain mercury, such as gas or nuclear power would reduce mercury emissions. However, coal is a cheap and abundant fuel in many areas and for many countries, coal will remain the dominant fuel source for years to come. In these areas switching coals to higher rank coals and those with lower mercury and/or higher chlorine content could reduce emissions significantly. Coal cleaning, coal blending and the cofiring of other fuels such as biomass or refuse could also reduce mercury emissions.

- Improvement in combustion conditions: increasing plant efficiency reduces fuel requirement and therefore reduces emissions of all pollutants, including mercury. Conversely, altering combustion conditions slightly to increase unburnt carbon can also reduce mercury emissions since the unburnt carbon acts as a sorbent for mercury capture.

- Installing controls for particulates, SO_2 and NOx: control systems such as ESP, baghouses, FGD and SCR can significantly reduce mercury emissions as a co-benefit effect and no extra cost to the plant. It is also possible to adjust the operation of these systems (such as position, type, temperature, the use of additives and so on) to maximise mercury capture. Adjustments to enhance mercury control in these systems must take into account the whole balance-of-plant effects. These are likely to be plant-specific and would require expert operation. In order to maximise co-benefit effects, the transfer of information and expertise, if not technologies themselves, would go a long way to reducing mercury emissions in an economic manner. Computer modelling might well provide guidance on how individual plants can enhance their mercury capture with the systems they already have in place.

- Mercury-specific control technologies: at the moment, it is commonly assumed that there is no single BAT/MACT for mercury on coal-fired plants. The variability in mercury chemistry and behaviour in coal combustion systems means that it is not possible to determine a best available technique or technology which would be suitable for all plants. Site-specific differences in plant type, fuel, operation and so on, and also in the cost of reduction technologies, mean that the cost of compliance for each plant can be minimised if the operator is free to choose the most appropriate control option for his plant. Prior to the vacation of CAMR, it appeared that the market in the USA would be dominated by activated carbon-based processes. However, the delay on the new US legislation may allow more time for mercury process control economics to be further defined and optimised for competing technology options such as alternative sorbents, chemical or photochemical oxidation. These now have an enhanced opportunity to prove themselves in the market on a case-by-case basis.

The adoption of mercury control in different areas of the world will depend largely upon the economics. Countries can only achieve reductions in emissions if they can afford to do so. The economics of mercury control are complex. Co-benefit effects are hard to evaluate with respect to the specific costs for the reduction in mercury achieved. For the most part, initial mercury reductions being achieved through control technologies such as FGD and SCR are seen as 'free', but only if these technologies were being installed anyway. The cost of mercury-specific control technologies is hard to summarise. The market is growing in response to the emerging legislative requirements. Many of the technologies emerging onto the market place are either in the developmental stage or not long out of it. There are therefore likely to reduce in price as they become more established. However, most estimates agree that the benefits of mercury control far outweigh the costs and that the additional costs passed to the consumer through utility bills is likely to be relatively minimal. One of the most important financial considerations at many plants will be to ensure that any steps taken to reduce mercury emissions, such as the use of sorbents, does not result in the loss of revenue from fly ash sales and incur new costs for hazardous waste disposal.

The actions being taken to reduce mercury in areas such as Europe and North America will achieve significant mercury reduction. They will also help identify the most economic methods of mercury control. In order that total global mercury emissions are reduced, the lessons learned in the developed world must be passed to those in the developing world. There is the potential for significant reductions in mercury from areas such as China to be achieved with the simple transfer of expertise and knowledge. Maximising the co-benefits of established control technologies for SO_2 and NOx will allow these nations to reduce mercury emissions at little or no extra cost. This will mean that a reduction in mercury emissions in these areas could start in the near-term. The lessons learned from the development of more costly mercury-specific control technologies in North America could then be passed on in the longer term when the marketplace is established and the costs of these systems have come down. It is in this area that the UNEP Mercury Partnership areas could make a significant contribution to the improvement in emission inventories and the alignment of reduction strategies worldwide.

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