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Process Optimization Guidance for Reducing Mercury Emissions from Coal Combustion in Power Plants

Division of Technology, Industry and Economics (DTIE)
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Process Optimization Guidance for Reducing Mercury Emissions from Coal Combustion in Power Plants

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Disclaimer

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SUMMARY

The Process Optimization Guidance (POG) summarizes mercury emission reduction practices applicable to coal-fired power plants. Mercury emissions from combustion of coal in power plants and industrial boilers constituted approximately 26 per cent of the global anthropogenic emissions of mercury in 2005. Therefore, it is important to decrease the amount of mercury emissions from this sector.

The POG is a tool to help determine the approaches to control mercury emissions, in many cases a co-benefit of reducing emissions of other pollutants. It is a tool for individual coal-fired power plants. The POG allows for a preliminary selection of a mercury control strategy for a given power plant. In general, the POG is also applicable to coal-fired large industrial boilers.

The practices and technologies capable of reducing mercury emissions include energy efficiency improvement measures, pre-combustion control measures (e.g., coal blending, coal cleaning), optimizing other (single) air pollutants control technologies to maximize mercury removal, mercury-specific control technologies, and multipollutant control technologies. The practices and technologies are given below:

- Activated carbon injection (ACI) has been demonstrated on a number of full-scale systems and is now a commercial technology. ACI must be used in conjunction with a particulate matter (PM) control device (e.g., electrostatic precipitator [ESP] or fabric filter [FF]). Chemically treated activated carbons are routinely capable of over 90 per cent emission reduction and allow for lower carbon injection rates for the same amount of mercury removal than un-treated activated carbons.

- Improving various areas of operation within an older boiler can reduce mercury emissions by up to about 7 per cent. Many existing plants could be overhauled to improve both efficiency and output while reducing mercury emissions. Conventional coal cleaning may, on the average, remove 30 per cent of mercury; data shows a wide range in mercury removal rates, depending on coal origin. Chemical treatment of coal is capable of mercury emission reduction of up to 70 per cent. Coal selection and blending has the potential to reduce mercury emissions by up to approximately 80 per cent. Use of halogen additives, especially bromine, has the potential to reduce mercury emissions by over 80 per cent.

- Improvement of operational efficiency of ESP or FF can increase their mercury capture up to about 30 per cent and up to about 80 per cent, respectively. Mercury removal of up to about 90 per cent can be expected in wet flue gas desulfurization (FGD) systems. Selective catalytic reduction may increase the amount of oxidized mercury up to about 85 per cent and thus improve mercury capture by wet (FGD).

- Multipollutant control technologies offer the cost advantage of delivering a system capable of controlling several pollutants simultaneously, including mercury. However, these technologies generally require more demonstration experience for full commercial readiness.

The concept for the selection process of mercury control strategy is introduced in the POG as a "Decision Tree." The Decision Tree is a tool to assist the user in a preliminary selection of an optimum mercury control strategy by analyzing other pollutants' (sulfur dioxide [SO_2], oxides of nitrogen [NO_X], PM) control equipment configuration as well as its operation. Based on the results of this analysis, the user may make a preliminary selection of mercury control technology, including mercury-specific technology. Final selection can be confirmed with one of a number of complex predictive models. However, the Decision Tree only considers the type of controls mentioned above. In addition one would need to consider plant operation and pre-combustion measures as described above.

The residues of a mercury control system must be carefully managed to mitigate environmental risks. Mercury captured in FGD remains bound to fly ash particles and FGD gypsum. Some leaching tests for wet FGD sludge and fixated wet FGD sludge tests have demonstrated excessive leaching of mercury under some disposal and use conditions.

General cost trends for controlling mercury from coal-fired power plants are given in the POG. There is a good understanding of the cost of ACI and actual numbers derived from operations in the United States. The cost of co-benefit mercury control is difficult to assess since it is dependent on multiple variables such as coal origin and quality, the extent of refurbishment required for the existing PM controls, or site-specific operating regime of wet FGD. For these reasons, only relative costs are presented in the POG. They should be treated as cost trend indicators. In addition, locally prevalent economic conditions should always be considered when selecting a mercury control option.

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List of Acronyms

ACI	activated carbon injection
BAT/BEP	Best Available Techniques/Best Environmental Practices
CCR	coal combustion residues
COPC	constituent of potential concern
ECO	electro-catalytic oxidation
EPRI	Electric Power Research Institute (USA)
ESP	electrostatic precipitator
ESPc	cold-side electrostatic precipitator
ESPh	hot-side electrostatic precipitator
FF	fabric filters
FGD	flue gas desulfurization
GC	Governing Council
GHG	greenhouse gas
HAP	hazardous air pollutant
ICR	information collection request
ISS	intelligent soot-blowing system
LoTOx	low temperature oxidation
LSFO	limestone forced oxidation wet FGD
MCL	maximum concentration limit
O&M	operating and maintenance
PAC	powdered activated carbon
PEESP	plasma-enhanced ESP
PJFF	pulse-jet fabric filter
PM	particulate matter
POG	Process Optimization Guidance
PRB	Powder River Basin
SCR	selective catalytic reduction
SDA	spray dryer absorber
TCR	Total Capacity Requirement
TOXECON	toxic emission control process
UBC	unburned carbon
UNEP	United Nations Environment Programme
US DOE	United States Department of Energy
US EPA	United States Environmental Protection Agency
WESP	wet electrostatic precipitator

List of Chemical Symbols

Al_2O_3	aluminum oxide
Br_2	molecular bromine
Ca	calcium
CaO	calcium oxide (lime)
$(CaSO_3 \bullet \frac{1}{2}H_2O)$	calcium sulfite hemihydrate
$(CaSO_4 \bullet H_2O)$	calcium sulfate dihydrate
Cl	chlorine
Cl_2	molecular chlorine
CO	carbon monoxide
Fe_2O_3	iron oxide (ferric oxide)
H_2S	hydrogen sulfide
H_2SO_4	sulfuric acid
HBr	hydrogen bromide
HCl	hydrogen chloride
HF	hydrogen fluoride
Hg^{2+}	oxidized form of mercury
Hg^0	elemental form of mercury
Hg^P	mercury bound to particulate matter
$HgCl_2$	mercuric chloride
HgO	mercury oxide
HgS	mercuric sulfide
$HgSO_4$	mercuric sulfate
HNO_3	nitric acid
MgO	magnesium oxide
MnO_2	manganese oxide
NH_3	ammonia
NH_4Cl	ammonium chloride
NO_X	oxides of nitrogen
O	atomic oxygen
O_2	molecular oxygen
O_3	ozone
OH	hydroxyl
S	sulfur
SiO_2	silicon dioxide (silica)
SO_2	sulfur dioxide
SO_3	sulfur trioxide
TiO_2	titanium oxide

List of Units

Ωm	ohm-meter
µm	micrometer
eV	electron volt
GW	gigawatt
kGy	kilogray
kPa	kilopascal
kWh	kilowatt hour
kg/MMm^3	kilograms per million actual cubic meters
mg/kg	milligram per kilogram
MJ/kg	megajoule per kilogram
MW	megawatt
ppm	parts per million

Punctuation

In accordance with English punctuation, dot (.) is used as decimal symbol and comma (,) as digit grouping symbol.

1 INTRODUCTION

At its twenty-fifth session in Nairobi in February 2009, the United Nations Environment Programme Governing Council decided to prepare a global instrument on mercury to be completed prior to the twenty-seventh regular session of the Governing Council, which will take place in 2013.

This Process Optimization Guidance (POG) summarizes practices and technologies capable of providing reduction of mercury emissions from coal-fired power plants. Generally, the POG is also applicable to large coal-fired industrial boilers.

The Process Optimization Guidance (POG) summarizes practices and technologies capable of providing reduction of mercury emissions from coal-fired power plants. These practices and technologies include energy efficiency improvement measures, pre-combustion control/prevention measures (e.g., coal blending, coal cleaning), optimizing other (single) air pollutants (sulfur dioxide [SO_2], oxides of nitrogen [NO_X], particulate matter [PM]) control technologies to maximize mercury removal, and mercury-specific technologies (e.g., activated carbon injection [ACI]). Multipollutant control processes (i.e., processes capable of controlling several pollutants simultaneously, including mercury) could also be used in order to reduce mercury emissions. Switching to alternative fuels and sources of power can also achieve a reduction of mercury emissions.

In general, the POG is also applicable to large coal-fired industrial boilers.

The POG is, as much as possible, a comprehensive study of control options, taking into account the variable mercury content of the coals mined and used in different regions and considering variations in plant type and plant operating conditions. The concept of a Decision Tree, including the steps in the selection process, is introduced. It is intended that the Decision Tree serve as a tool to help determine the most appropriate options to address the needs of a specific coal-fired plant. It is planned that the Decision Tree will eventually be made available in an on-line interactive form.

The POG includes examples of costs for mercury emission control systems. It should be pointed out that these costs may not represent accurate cost data on the various control options, since they will vary significantly with the different challenges each plant and coal combination process poses, and costs can change due to variability in the costs of raw materials, construction materials, and labor rates from country to country. However, the values given will help the reader obtain a general idea of the relative costs of the options available.

1.1 BACKGROUND

The United Nations Environment Programme (UNEP) Governing Council (GC) initiated global assessment of mercury at its twenty-first session held in 2001. Key findings of the global assessment were: (1) mercury is persistent, (2) undergoes long-range transport, and (3) cycles globally. The findings were acknowledged as giving sufficient evidence of adverse global impacts to warrant international action and the GC endorsed the need for global action in 2003. This GC decision resulted in formation of the UNEP Mercury Programme. At its twenty-fifth session, in 2009, the GC decided on a number of matters that will influence the future path of global work on mercury. In particular, the GC decided to prepare a global legally binding instrument on mercury to be completed prior to the twenty-seventh regular session of the GC, which will take place in 2013. The negotiation of the global instrument on mercury has commenced in 2010. To inform the work of the intergovernmental negotiating committee developing the instrument, a study on various types of mercury-emitting sources and current and future trends of mercury emissions, including analyzing and assessing the cost and effectiveness of alternative mercury control technologies and measures (the paragraph 29-study) was called for by the GC. The information in this POG describing various mercury control options from sources utilizing coal combustion for electricity generation has been fed into the paragraph-29 study.

POG was adopted for the name of the document describing various mercury control options from sources utilizing coal combustion for electricity generation in lieu of Best Available Techniques/Best Environmental Practices (BAT/BEP). This is because the POG does not analyze or comment on potential policy options that might define best available control technologies in a legislative context. Instead, the POG focuses on summarizing various mercury control practices and technologies. The POG is an activity of the Mercury Release from Coal Combustion Partnership Area of the UNEP Global Mercury Partnership. POG is one of the three activities carried out under the Mercury Release from Coal Combustion Project (the other two being improvement of information on mercury emissions and promotion of emission reductions through targeted demonstration projects).

1.2 OBJECTIVE AND SCOPE

The objective of the POG is to serve as a tool to help evaluate and select the most appropriate options to address the mercury emission reduction needs in individual coal-fired, electricity generating plants.

Examples of coal usage are given for: China, India, European Union, Russia, South Africa, and the United States. However, the methodology and approaches to controlling mercury emissions described should be largely applicable to all coal-using nations.

The POG reviews strategies for controlling mercury emissions from coal-fired electricity-generating plants. The first strategy considered is to utilize any viable means to improve the efficiency of the plant. This strategy includes plant modernization as well as coal treatment, and it allows for more energy and less emissions being produced from the same volume of coal used. Coal treatment technologies that could be applied prior to combustion and discussed in the POG include conventional coal washing, coal beneficiation for mercury content, coal blending, and coal additives.

A second mercury emission control strategy outlined here is the set of approaches designed to maximize the amount of mercury removal that may be realized as the effect of operation of air pollution control equipment originally designed to reduce the emissions of other pollutants (SO_2, NO_X, PM). Depending on the air pollution control equipment, these approaches could include modernization of electrostatic precipitators (ESP), modification of wet flue gas desulfurization (FGD) scrubber chemistry, alteration of selective catalytic reduction (SCR) operation, or a combination of these.

Mercury-specific removal technologies are also presented in the POG. These technologies should be considered when evaluating overall mercury control strategy, especially where higher levels of mercury emission control are desired beyond what could be achieved through the energy efficiency improvement and co-benefit mercury removal. The POG also presents information on multipollutant control technologies. These are technologies under development that are capable of simultaneously controlling emissions of multiple pollutants, such as PM, SO_2, NO_X, and mercury.

The concept of an evaluation and selection process called the Decision Tree is introduced. The Decision Tree is a tool to assist the user in a preliminary selection of an optimum mercury control strategy (see section 10. Decision Tree).

Residues from coal-fired power plant operation are also described in the POG. Fly ash is the product of coal combustion and is collected by the PM control device. Gypsum is the byproduct of limestone wet FGD process with forced oxidation (LSFO). It can be disposed in a landfill, used for wallboard production, or applied agriculturally. Wet FGD sludge is collected from natural or inhibited oxidation wet FGD. Wet sludge is often called Coal Combustion Residue (CCR). Studies examining the potential for mercury leaching from CCRs to groundwater are discussed in the Post-Control Issues section. Depending on the configuration of the existing air pollution control equipment and subsequent Decision Tree optimization of mercury capture, mercury may be transferred from gas phase (flue gas) onto solid phase (e.g., fly ash, synthetic gypsum) or into a liquid or solid/liquid phase.

Finally, examples of the cost of mercury control are given in the POG. Relative costs of controlling mercury via co-benefit removal and mercury-specific controls are presented.

2 MERCURY EMISSIONS FROM COAL COMBUSTION

Mercury emissions to air from combustion of coal in power plants and industrial boilers constituted about 26 per cent of the global anthropogenic emissions of mercury in 2005. This amounts to about 500 metric tons/year of mercury.

Global anthropogenic mercury emissions have been estimated at 1930 metric tons in 2005 (Pacyna et al., 2010). Of this amount, the largest anthropogenic emissions of mercury (46 per cent of the total) to the global atmosphere occurred from combustion of fossil fuels, mainly coal in utility, industrial, as well as residential boilers, heaters, and stoves (UNEP, 2008). Mercury emissions from combustion of coal in power plants and industrial boilers constituted about 26 per cent (or about 500 metric tons/year) of the global anthropogenic emissions of mercury in 2005. The proportion of global anthropogenic emissions of mercury to air in 2005 from various sources is shown in Figure 1.

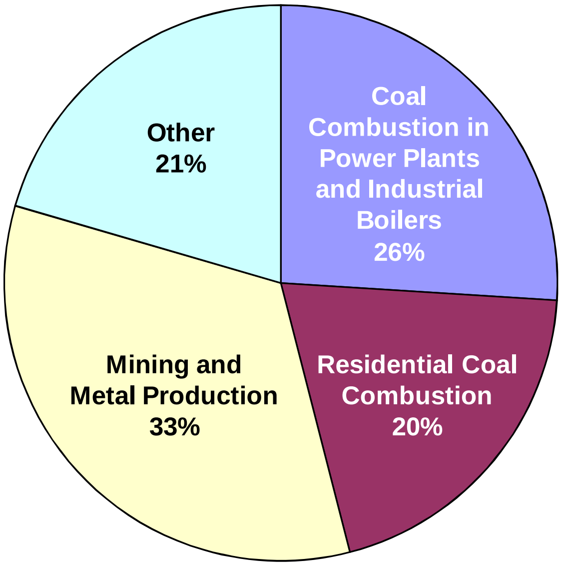

Figure 1. Proportion of global anthropogenic emissions of mercury to air in 2005.

Emissions of mercury from coal-fired plants in Europe (total 25 EU member states) were estimated at around 29 metric tons/year in 2005, having declined from 52 metric tons/year in 1995 (Ritchie et al., 2006). This significant reduction in emissions was a result of a combination of factors including fuel switching (from coal to natural gas), plant efficiency improvements, and co-benefit effects due to the application of SO_2 and NO_X control technologies. The decline is predicted to continue as the legislation applicable to coal-fired power plants in Europe tightens. Total mercury emissions from coal-fired plants in the EU are projected to be as low as 15 metric tons/year by 2020 (Ritchie et al., 2006). There is, as yet, no specific requirement for mercury control at coal-fired power plants in Europe and no widely applied mercury-specific control technologies in place (Sloss, 2008). In the United States, coal-fired utilities contributed about half of the anthropogenic US emissions in 2005 (approximately 48 metric tons/year) (US EPA, 2010). In 2010, the U.S. Environmental Protection Agency (US EPA) is working to propose mercury, and other hazardous air pollutants (HAP), regulation for coal-fired plants. SO_2 and NO_X control technologies have not been applied on all plants in the U.S. and therefore, the co-benefit capture of mercury has not been realized to the full potential. Approximately 190 GW, or about 60 per cent of total coal steam capacity are currently equipped with FGD (or under contract for installation by 2012) in the United States (US EPA, 2010a). The existing FGD capacity is thought to be at least about 60 per cent of generating capacity across the EU and should reach over 80 per cent by the Large Combustion Plant Directive deadline of 2016.

Mercury emissions from coal-fired power plants in South Africa were estimated at 9.8 metric tons/year in 2004 (Dabrowski et al., 2008). The amount of mercury released from the burning of coal for power generation in India was estimated to be 52 metric tons/year in 2001, or 67 per cent of the total 78 metric tons/year released from coal consumption (SENES, 2004). Mercury emissions from coal-fired generation in Russia were estimated at 8 metric tons in 2001 (ACAP, 2005). Mercury emissions from coal-fired power generation in China were estimated at about 141 metric tons in 2005 (Wang et al., 2010). As China is accelerating the deployment of FGD and SCR, the co-benefit removal of mercury could be expected to increase (Wang, et al., 2010).

Some of the countries that are large emitters of mercury from coal (U.S., EU countries) already have control technologies in place that may be capable reducing mercury emissions from the sector. Other countries that are large emitters may have limited amount or no control technologies in place that would be capable of substantial reduction of mercury emissions. Countries initiating mercury control strategies would be advised to maximize the most economic control of mercury through co-benefit control technologies to the extent that the use of these control technologies is necessary to achieve non-mercury goals for air quality or acid deposition. While total mercury control would ultimately require the use of mercury-specific control technologies, significant mercury control can be achieved through the optimization of plant performance and the application of technologies for the control of pollutants such as PM, SO_2, and NO_X that are capable of reducing mercury emissions as a co-benefit effect.

3 MERCURY EMISSION CONTROL STRATEGIES

- **The amount of mercury emissions is related to the amount of coal burned and to the configuration of any existing air pollution control equipment. The first step in the review of strategies for controlling mercury emissions should be to consider any viable means to improve the efficiency of the plant.**
- **The next step in mercury emission control strategies is to maximize the amount of mercury removal that is realized as the effect of operation of air pollution control equipment originally designed to limit the non-mercury emissions (the so-called co-benefit mercury removal).**
- **When higher levels of mercury emission control are desired beyond what can be achieved through the improved efficiency and the co-benefit removal, a plant would have to deploy mercury-specific technology.**

For any given coal-burning power plant and air pollution control equipment configuration, the amount of mercury emissions from the plant is directly related to the amount and type of coal burned to generate a unit of electricity. It follows that if the amount of coal burned to generate a unit of electricity could be reduced, then the overall mercury emissions from a given power plant would also decrease. This could be accomplished by measures undertaken to improve efficiency of a power plant. Therefore, the first step in the review of strategies for controlling mercury emissions should be to consider any viable means to improve the efficiency of the plant. Improvement of the efficiency also provides for reduction of all emitted pollutants including greenhouse gases (GHG) in addition to reduction of mercury emissions. Plant upgrading is therefore a win-win situation with more energy and less emissions being produced from the same volume of coal used (Sloss, 2009).

Significant efficiency improvements may be obtained by the deployment of coal treatment technologies prior to combustion. Coal treatment technologies considered in the context of plant efficiency and mercury removal, include conventional coal washing, coal beneficiation for mercury content, coal blending, and coal additives.

The next step in mercury emission control strategies, beyond efficiency improvement, is the set of approaches designed to maximize the co-benefit removal. This is the amount of mercury removal that is realized as the effect of operation of air pollution control equipment originally designed to limit the non-mercury emissions (here PM, SO_2, or NO_X) and already in place at the power plant. Depending on the available air pollution control equipment, these approaches could include modernization of ESP, modification of wet desulfurization scrubber chemistry, alteration of SCR operation, or a combination of these (Sloss, 2006). When higher levels of mercury emission control are desired beyond what can be achieved through the co-benefit removal, a power plant would have to deploy mercury-specific technology.

This chapter introduces the most common plant efficiency improvement measures, including coal treatment. Following these considerations, this chapter presents the concept and advantages of the co-benefit removal of mercury, and then proceeds to describe the most advanced mercury-specific technologies.

3.1 IMPROVEMENT OF PLANT EFFICIENCY

Improvement of plant efficiency may involve a number of measures designed to reduce the amount of fuel (coal) used and, as a result, to reduce the amount of mercury (and other) emissions. Some of the most commonly applicable measures are dedicated to the improved operation of the boiler. In addition, operating and maintenance (O&M) practices have the potential to improve plant performance, including its efficiency, reliability, and the O&M cost. Deterioration of plant equipment is unavoidable; however, the rate at which this deterioration occurs depends greatly on the O&M practices. Some of the good O&M practices include, for example, steam line maintenance or water treatment. Efficiency measures will be discussed in more detail in the following chapter.

3.2 COAL TREATMENT

Coal treatment includes conventional coal washing, beneficiation, blending, and coal additives. Conventional coal washing, while primarily targeting the minimization of ash and sulfur content of coal, can also decrease the mercury content of coal. Coal beneficiation includes coal washing and additional treatment designed to decrease the mercury content of coal. The remaining coal treatment technologies (coal blending and coal additives) have been designed to specifically address minimization of mercury air emissions by promoting chemical transformations of mercury in the power plant’s combustion and post-combustion equipment that facilitate mercury removal. They can be used in addition to coal washing (e.g., blending of two streams of washed coal) or as stand-alone approaches (e.g., halide addition into the boiler).

Coal washing is an important aspect of efficiency improvement as it reduces ash content and improves calorific value. Coal washing can lower the ash content from the 40 per cent range to around 20 to 30 per cent or better, depending on the coal mineralogy. A boiler designed for washed coal will be more compact than the one designed for unwashed coal because it can be designed for a higher flue gas velocity. As the ash quantity is lower after burning a washed coal, the amount of erosion will also be reduced. In addition, improvement of coal quality can impact the design of multiple components of a power plant, including furnace, economizer, pulverizers, air heater, and fans. These considerations will be discussed in more detail in the following chapters.

3.3 CO-BENEFIT REMOVAL

In order to define the co-benefit removal of mercury from coal fired flue gas, one needs to consider the forms of mercury typically present and their amenability for control. Three forms of mercury species may typically be present in the coal-fired flue gas: gas-phase oxidized form of mercury (Hg^{2+}), gas-phase elemental mercury (Hg^0), and solid-phase mercury associated with PM (Hg^P) such as fly ash or unburned carbon (UBC). To describe the relative proportion of the three forms of mercury in the flue gas, the term speciation is used. The mercury speciation of mercury in flue gas is a function of coal type and is of importance to the co-benefit mercury control of existing equipment. Generally, more Hg^{2+} occurs in flue gas from burning bituminous coal compared to the flue gas from burning lower-grade coals.

Mechanisms responsible for speciation of mercury can be briefly presented as follows. Mercury that is naturally present in coal, mostly as sulfur-bound compounds, is released during combustion. Mercury content may vary, depending on the coal source and type, but is typically defined as being from 0.01 to 0.3 mg/kg. When the coal is burned, the combustion temperatures (approximately 1,500 °C) vaporize mercury to form gaseous Hg^0. Cooling of the combustion gases and interactions of Hg^0 with other combustion products (in the presence of fly ash) may result in conversion of a fraction of Hg^0 to other forms. An example of mercury species occurring along the flue gas path is shown schematically in Figure 2.

Oxidized mercury compounds may include mercuric chloride ($HgCl_2$), mercuric bromide ($HgBr_2$), mercury oxide (HgO), or mercuric sulfate ($HgSO_4$). Particulate matter-bound (PM-bound) mercury may be either elemental or oxidized. The known mechanisms for oxidation of Hg^0 in the flue gas are homogeneous oxidation (minor contribution) and heterogeneous (e.g., fly ash/UBC-promoted) oxidation (Senior, 2004 and Vosteen et al., 2006b). Additional oxidation of mercury may also occur as a result of the application of post-combustion NO_X controls. Mercury species transformations in the flue gas are shown in Figure 3. The extent of mercury oxidation and resultant mercury speciation affect the amount of co-benefit mercury removal.

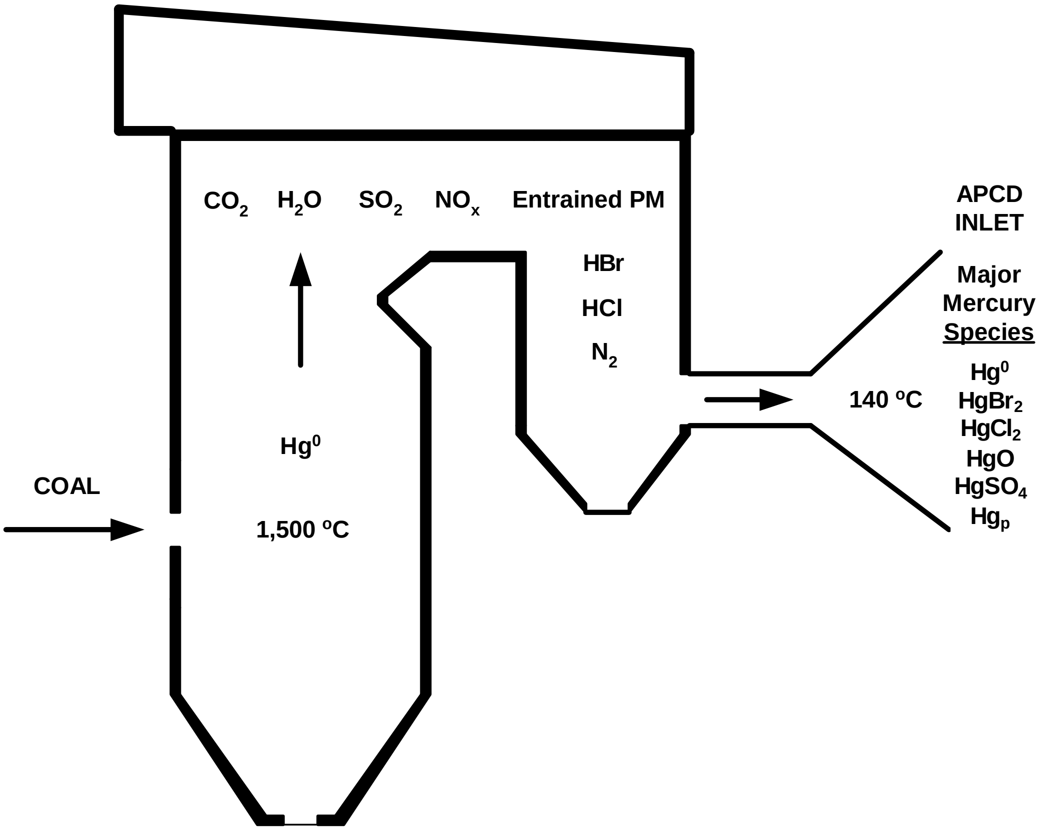

Figure 2. Mercury species in a coal-fired boiler flue gas; APCD = air pollution control device (based on US EPA, 2000).

One of the most complete efforts to-date to understand the potential for co-benefit removal of mercury was the information collection request (ICR) that started in late 1998 in the United States and was mandated by US EPA. As part of the ICR, the US EPA selected a subset of the coal-fired electric utility generating units at which field-source testing was performed to obtain mercury emission data for the air pollutant control devices then installed at these units. The US EPA sent ICRs to the owners and operators of approximately 1,100 units. The test locations were selected by the US EPA to approximate the nationwide distribution of coal-fired steam generating units by type of boiler, coal burned, and air emission controls used. At each of the selected test locations, measurements were made of the mercury content in the inlet and outlet gas stream for the farthest downstream control device used on the unit. The testing followed an US EPA-approved sampling protocol and included three sample runs at each sampling location. Samples of the coal burned during the source test were also collected.

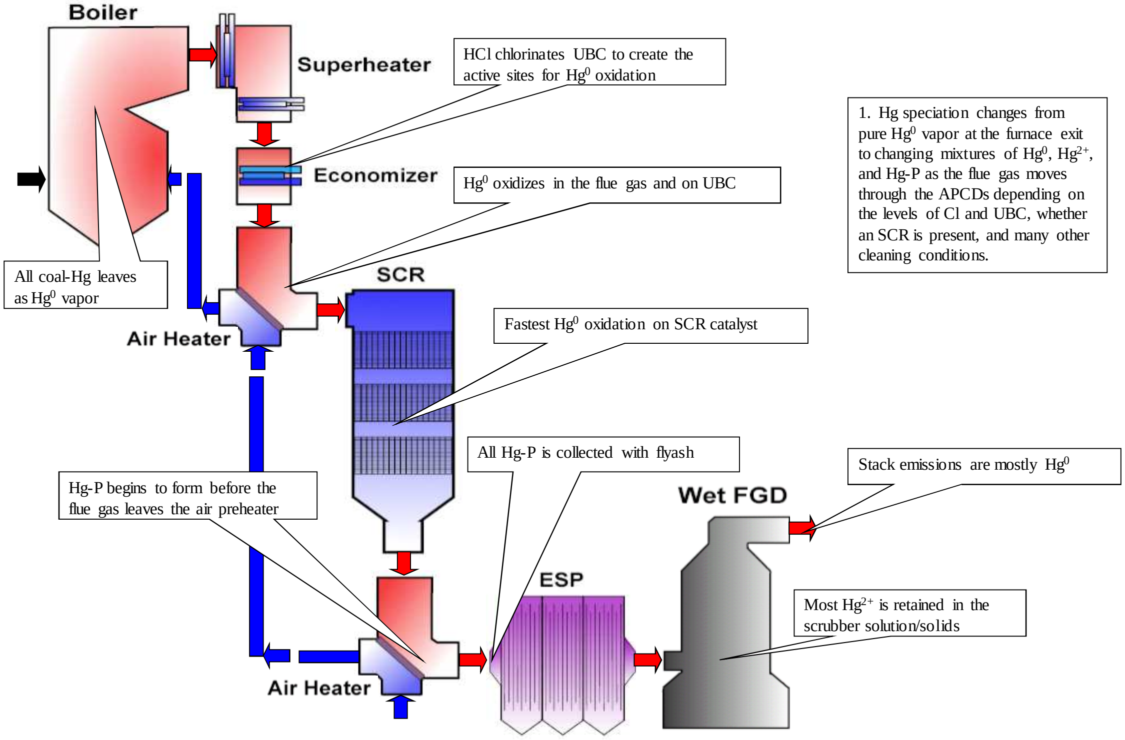

Figure 3. **Mercury species transformations in the flue gas (from Niksa and Fujiwara, 2009)**

Based on data collected through the ICR, Table 1 summarizes the average reduction in total mercury emissions from units burning different coals and equipped with different post-combustion equipment configurations. It should be pointed out that data presented in Table 2 represents a small data set and pertains to air pollution equipment operating on flue gas generated by combustion of US coals. Combustion of non-US coals may result in different mercury captures for the same type of coal, which may be due to differences in composition.

Table 1. Average Mercury Capture by Existing Post-combustion Control Configurations Installed on Coal-fired Units (US EPA, 2002)

Post-combustion Control Strategy	Post-combustion Emission Control Device Configuration	Average Mercury Capture by Control Configuration, per cent		
		Coal Burned in Pulverized-coal-fired Boiler Unit		
		Bituminous Coal	Subbituminous Coal	Lignite
PM Control Only	ESPc	36	9	1
	ESPh	14	7	not tested
	FF	90	72	not tested
	PS	not tested	9	not tested
PM Control and Spray Dryer Absorber	SDA + ESP	not tested	43	not tested
	SDA + FF	98	25	2
	SDA + FF + SCR	98	not tested	not tested
PM Control and Wet FGD System [a]	PS + FGD	12	10	not tested
	ESPc + FGD	81	29	48
	ESPh + FGD	46	20	not tested
	FF + FGD	98	not tested	not tested

[a] Estimated capture across both control devices

Notes:
ESPc = cold-side electrostatic precipitator
ESPh = hot-side electrostatic precipitator
FF = fabric filter
PS = particulate matter scrubber
SDA = spray dryer absorber system
SCR = selective catalytic reduction
FGD = flue gas desulfurization

As can be seen from Table 1, the capture of mercury across the existing equipment can vary significantly based on coal type and specific equipment configurations, as well as other factors (e.g., fly ash properties, UBC). The level of co-benefit removal ranges from negligible to more than 90 per cent. The strongest factor is the type of coal burnt, which in turn affects mercury speciation at the inlet to the control device(s). In general, units burning subbituminous and lignite coals demonstrate significantly lower mercury capture than similarly equipped bituminous-fired units. Units that burn bituminous coals typically have relatively high concentrations of Hg^{2+} at the inlet to the control device(s). Units that burn subbituminous coal or lignite typically have relatively low concentrations of Hg^{2+} and high concentrations of Hg^0 at the inlet to the control device(s). The relatively high concentrations of halogens (e.g., chlorine, bromine) in bituminous coals result in the oxidization of Hg^0 to form Hg^{2+}, primarily thought to be mercuric halides ($HgCl_2$ and HBr_2). By contrast, both subbituminous and lignite coals tend to have lower amounts of halogens and higher amounts of alkaline material (calcium and sodium) than bituminous coals. Chlorine and bromine from the combustion of subbituminous coal and lignite tends to react with the alkaline materials in flue gas, and little if any halogen is left available for the oxidization of Hg^0. Therefore, flue gas from combustion of subbituminous coal and lignite tends to have relatively low concentrations of Hg^{2+}.

Gaseous compounds of Hg^{2+} are water-soluble and can absorb in the aqueous slurry of a wet FGD system. Thus, a major fraction of gas-phase Hg^{2+} vapors may be efficiently removed when a power plant is operated with a wet scrubber for removing SO_2. However, gaseous Hg^0 is insoluble in water and, therefore, does not absorb in such slurries. Early European (Gutberlet et al., 1992) and US Department of Energy (US DOE)-funded study (DeVito and Rosenhoover, 1999) showed that the nominal mercury removal for wet FGD systems on units firing bituminous coals was approximately 55 per cent, with the removal of Hg^{2+} between 80 and 95 per cent. In another study, measured mercury removal on eight bituminous coal-fired boilers with SCR and FGD an average mercury removal (coal to stack) varied from 65 to 97 per cent (Withum et al., 2004). Further, applicability of a wet FGD in controlling mercury may be augmented by an upstream SCR, which has the potential to alter mercury speciation in a way that increases the amount of oxidized mercury in the flue gas (Gutberlet et al., 1992). PM-bound mercury can be captured by ESPs and fabric filters (FF).

3.4 MERCURY-SPECIFIC TECHNOLOGIES

In cases where the amount of mercury capture is desired beyond what can be achieved through the co-benefit removal, mercury-specific control technologies are needed. To date, use of sorbent injection has shown the most promise as a mercury control technology. The sorbent most often used and most thoroughly tested is powdered activated carbon (PAC). In the basic scenario envisioned for sorbent ACI for supplementary mercury control in the 1990's (Wirling et al., 1999), PAC is injected between the air heater and the particulate control device, as shown schematically in Figure 4. When ACI is used as a mercury-specific control, injection upstream of the air heater inlet is preferred because it improves mixing of sorbent in the flue gas and augments sorbent residence time (Martin, 2009). Initial concerns of injecting at the air heater inlet location were evaluated over a 1-year period, and tests showed no significant increase in deposition, corrosion, or erosion (Pavlish et al., 2009).

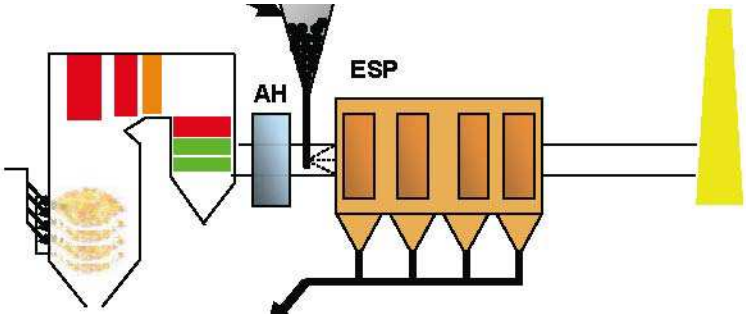

Figure 4. Schematic of ACI (AH: Air Heater) (US DOE, 2005)

After adsorbing the mercury from the combustion flue gas, PAC is captured together with the fly ash in the PM control device. Variations of sorbent injection technology have been developed over the years that include sorbent type and injection location. These, in turn, dictate the extent of hardware modification to accommodate the process into the existing ductwork. Mercury capture by ACI can be significantly enhanced by addition of chemicals to a boiler (Vosteen, 2003).

Several multipollutant technologies are capable of simultaneously removing mercury in addition to removing SO_2, PM, and NO_X from flue gas. These technologies are at varying levels of development and commercial availability. Multipollutant technologies are defined as technologies that, if installed, would themselves be capable of removing SO_2, PM, NO_X, and mercury rather than relying on optimization of air pollution control equipment operation for maximization of co-benefit mercury removal.

Sorbent injection for mercury control and multipollutant technologies will be discussed in more detail in the following chapters.

4 IMPROVEMENT OF PLANT EFFICIENCY

Improving various areas of operation within an older boiler can reduce mercury emissions by up to about 7 per cent. Many existing plants could be overhauled to improve both efficiency and output while reducing mercury emissions.

Improvement of plant efficiency may involve a number of measures designed to conserve fuel (coal) and, as a result, to reduce the amount of mercury emissions per unit of electricity generated. Some of the most commonly applicable measures are shown in Table 2 and include: new burners, improved air preheater and economizer, improved combustion measures, minimization of short cycling, minimization of heat transfer surface deposits, and minimization of air infiltration. Efficiency improvement measures presented in Table 2 are discussed in more detail below.

In addition, O&M practices have a significant impact on plant performance, including its efficiency, reliability, and operating cost. Good O&M practices can significantly slow down the deterioration of heat rate of a power plant. Some of the good O&M practices include, for example, steam line maintenance and water treatment. A well operated and maintained plant will experience less rapid deterioration of heat rate; hence, O&M practices themselves influence coal use and mercury emissions. It is important to distinguish normal maintenance from capital expenditure. Capital expenditure is typically only done once every few years and includes repairs/replacement of major equipment all at one time. Good O&M practices should be an ongoing concern in daily plant operation.

Older burners are typically inefficient. This inefficiency results in incomplete combustion and the need for high excess air. Replacing or retrofitting burners with more efficient ones will yield efficiency gains by improving one or more of the consequences of inefficient combustion (incomplete combustion, excess air, cycling duty). Increasingly, more sophisticated combustion monitoring and controls are an integral part to this retrofit (tuning and optimization). Burner maintenance is very important because inoperable dampers, broken swirlers, or clogged nozzles will make an otherwise good burner perform badly.

Table 2. Commonly Applicable Efficiency Improvement Measures at Power Plants

Measure	Efficiency Improvement (percentage points)[a]	Mercury Reduction Potential (per cent)	Comments
Replace/upgrade burners	Up to 4-5%	Up to ~ 6%	Site-specific considerations (ability to retrofit)
Improved economizer	40°F increase in flue gas temperature equals a ~1% efficiency loss	Relates to efficiency gain in boiler	
Improved air preheater	A 300°F decrease in gas temperature represents about 6% improvement	~ 1% per 40 °F temperature decrease	
Combustion tuning	CO from 1000-2000 to < 200 ppm UBC from 20-30% to 10-15%	up to ~3%	Manual tuning with parametric testing; decreasing UBC can result in lower native Hg capture
Combustion optimization	0.5 – 3.0	up to ~ 4%	Neural network-based
Instrumentation and controls	0.5 – 3.0 (in addition to optimization)	up to ~ 4%	
Minimize short cycling	Up to ~ 4% – 6%	Up to ~ 5% - 7%	Very site specific
Reduce slagging and fouling of heat transfer surfaces	1% to 3%	Up to ~ 4%	Site specific; fuel quality/operating condition have large impact
Reduce air leakages	1.5 – 3%	Up to ~ 4%	Requires routine maintenance procedures

[a] Efficiency improvement given in percentage points rather than per cent improvement

For most coal-fired boilers, using waste heat gas recovery systems to capture and use some of the energy in the flue gas can increase energy efficiency. Heat recovery equipment includes various type of heat exchangers (economizers and air heaters), typically located after the gases have passed through the superheater and steam generating sections of the boiler. Economizers improve boiler efficiency by extracting heat from the flue gases discharged from the final superheater section. Air heaters transfer heat from the flue gas to the incoming combustion air. Pulverized coal-fired furnaces require the use of air heaters to evaporate the moisture in the coal. This heated air also serves to transport the pulverized fuel to the burners.

There are a number of options that can be applied to improve the combustion process and the overall performance of the plant and include the following groups: combustion system tuning, combustion and plant performance optimization, and instrumentation and controls. Tuning of the combustion system requires a visual check to ensure that everything is in good working condition and set according to the manufacturer's recommendations or the optimum settings developed for the particular plant. Simple parametric testing may be required, involving changes in the key parameters such as carbon monoxide (CO) emissions, steam outlet conditions, flue gas outlet temperature, and NO_X emissions. Optimization can be accomplished in a number of different ways. For boilers which change operating conditions frequently, it may be justifiable to install a software-based optimization system. Optimization systems can function with instrumentation and controls available at the plant. However, a digital control system is a requirement to achieve good performance. If maximizing plant efficiency or minimizing emissions is required, certain instrumentation is justifiable as additional instrumentation makes it possible to achieve better performance.

A boiler cycle refers to the complete "off-on-off" sequence, which includes the firing interval, purges, and idle period. Overall boiler efficiency must account for the complete "cycle duty," and not just its efficiency while firing at steady (full) load. Therefore, overall boiler efficiency is the useful heat provided by the boiler, divided by the energy input (chemical input from coal) over the cycle duration. This overall boiler efficiency decreases when "short cycling" occurs. Facilities have several ways to minimize "short cycle" conditions and improve overall efficiency. For example, using automatic controllers that determine the incremental costs (change in steam cost/change in load) for each boiler, and that shift loads accordingly, will maximize efficiency and reduce energy costs.

Boiler heat transfer surfaces are exposed to high temperature gases and products of coal combustion, which vary in composition amongst different coal types and operating conditions. Formation of soot and incomplete combustion of carbon all contribute to the potential for surface deposits. Deposits may also be formed on the surface of the tubes. To minimize deposition problems (slagging and fouling), it is important to operate the boiler within the parameters for which it was designed. Also, systems firing ash-laden fuels (such as coal) also include soot blowers that typically use compressed air or steam to periodically remove the deposition from the boiler walls and tubes. More advanced soot-blowing systems, or Intelligent Soot-blowing Systems (ISS) use feedback signals, such as exit gas temperature or heat transfer sensors, to trigger their operation. ISS determines which soot blower needs to be operated and when, depending on the local performance of the heating surfaces.

Air infiltration is an unavoidable concern in boiler systems and ductwork. This occurs as a result of the large temperature difference between the hot combustion gases and ambient air temperature, which creates a negative pressure in the furnace (the so-called "stack effect" or "thermal head"). Indicators of excessive air leakage include: oxygen (O_2) levels measured at the outlet of the boiler, as well as fuel consumption and gas temperatures.

As presented in Table 2, improving various areas of concern within an older boiler can reduce mercury emissions by up to about 7 percentage points. Although most plants will only need improvement in one or two areas, there is the potential for overall mercury reduction to be not insignificant. Many existing plants could be overhauled to improve both efficiency and output while reducing mercury emissions in an economic manner.

5 COAL PREPARATION

This chapter presents coal preparation techniques that can be used to decrease mercury emissions from coal-fired power plant. Coal preparation techniques discussed here include coal washing, beneficiation, blending, and coal additives.

- **Conventional coal cleaning may on the average remove 30 per cent of mercury on mass basis. Data shows a wide range in mercury reduction rates, depending on coal origin.**
- **Chemical treatment of coal, like the K-fuel process is capable of mercury emission reduction of up to 70 per cent.**
- **Coal selection and blending has the potential to reduce mercury emissions by up to approximately 80 per cent.**
- **The use of halogens, especially bromine, has the potential to oxidize mercury by up to 80 per cent.**

5.1 COAL WASHING

Raw coal contains mineral impurities such as rock and clay that are referred to as ash. Where appropriate, this raw coal should be processed (or cleaned) to reduce the ash content; to increase the heating value; and to reduce the sulfur and, potentially, mercury content to ultimately lower emissions when the coal is burned in the utility boiler. In addition, the removal of mineral impurities also reduces O&M costs and slows the deterioration of the boiler system.

Coal washing can lower the ash content from the 40 per cent range to around 20 to 30 per cent, can improve its calorific value, and increase boiler efficiency (Satyamurty, 2007). As the ash quantity is lowered by burning a washed coal, the amount of boiler erosion may also be reduced. In addition, improvement of coal quality can impact the design of multiple components of a power plant, including the furnace itself, economizer, pulverizers, air heater, fans, and particulate control devices.

Conventional coal cleaning methods separate the organic fraction of the as-mined coal from the mineral materials according to the differences in either the density-based or surface-based characteristics of the different materials. Physical coal cleaning typically involves a series of process steps including size reduction and screening, gravity separation of coal from sulfur-bearing mineral impurities, and dewatering followed by drying.

Conventional coal cleaning methods will also remove some of the mercury associated with the incombustible mineral materials. However, they will typically not remove the mercury associated with the organic carbon structure of the coal (US EPA, 2002). One review quotes test data for 26 US bituminous coal samples that indicate a wide range in the amount of mercury removed by coal cleaning (US EPA, 1997). Analysis of five of the coal samples showed no mercury removal associated with conventional coal cleaning whereas the remaining 21 coal samples had mercury reductions ranging from approximately 3 to 64 per cent. The average mercury reduction for all of the samples was approximately 21 per cent.

Another study reported a higher average mercury reduction for 24 samples of bituminous coal (Toole-O'Neil et al., 1999). The average decrease in Hg reduction on an energy basis was 37 per cent (values ranged from 12 to 78 per cent). The average decrease in mercury reduction on a mass basis was 30 per cent for this study. Since the form of occurrence in mercury can vary depending on the location of the coal, to understand quantitatively how much Hg could be removed from a given coal, laboratory and/or pilot-scale testing of coal cleaning methods is needed.

The variation in mercury reductions quoted above might be a function of the type of process used to clean a given coal, coal rank, and the nature of mercury in the coal matrix. Generally, mercury reduction in lower rank coals could be expected to be lower than in bituminous coals because in lower rank coals a greater fraction of the mercury is likely to be bound to organic carbon structure. In coals used for electricity generation, mercury is present in coal bound with pyrite or with the organic fraction of coal (ACAP, 2004). Much of the heavier pyrite can be removed by density-based processes, but not by surface-based processes where the similar surface characteristics of pyrite and the organic matter make separation of the two components difficult (US EPA, 2002). Advanced coal cleaning techniques, such as the ones using naturally occurring microbes and mild chemical processing, were investigated in the past in order to augment mercury removal (Brown et al., 1999). For example, hydrothermal treatment has shown to significantly reduce sulfur and organically-bound mercury in some coals, albeit at a fairly high cost. (Timpe et al, 2001).

In summary, removal of some of the mercury from coal is feasible when conventional coal cleaning methods are utilized. However, the effectiveness of mercury removal from coal during conventional coal cleaning varies widely depending on the source of coal and on the nature of mercury in coal. Because of these factors, testing of coal cleaning methods is needed to understand quantitatively how much Hg could be removed from a given coal.

5.2 COAL BENEFICIATION

Coal beneficiation is capable of improving coal properties beyond what can be achieved with coal washing alone. It includes coal washing as a primary step, but then utilizes additional treatment to reduce the mercury content of coal. An example of coal beneficiation is the

K-Fuel process. The process may also be described as a pre-combustion multipollutant control process, as it results in reduction of mercury emissions in addition to lower PM, SO_2, and NO_X emissions. K-Fuel is a beneficiated coal that is derived from subbituminous coal or lignite coal (Black and Veatch, 2003). The resulting fuel is lower in ash, higher in heating value, and produces lower pollutant emissions than untreated coal. K-Fuel uses a pre-combustion process that improves the quality of the coal—including removing the mercury, moisture, ash, sulfur, and some of the fuel NO_X precursors—before the coal is burned at the power plant. Because these constituents are removed prior to burning the coal at the plant, the need for post-combustion controls may be reduced. K-Fuel technology may be applicable to bituminous coal as well. However, the supplier has focused exclusively on subbituminous and lignite applications in North America because the K-Fuel is a moisture and ash reduction process, and these coals are often characteristic with high moisture and ash content.

The K-Fuel pre-combustion multipollutant reduction technology is a two-step process, illustrated schematically in Figure 5 and described below:

- Physical Separation: A gravity separation process, either wet or dry, is used to remove ash along with other pollutants (sulfur and mercury).
- Thermal Processing, which follows physical separation, employs thermal processing vessels operating under high temperature and pressure (240 °C and 340 kilopascal [kPa]). Figure 6 shows the main components of the Thermal Processing unit.

According to K-Fuel, physical separation studies on a number of low-rank coals exhibited 10 to 30 per cent ash reduction, 10 to 36 per cent sulfur reduction, and 28 to 66 per cent mercury reduction. Total mercury reduction at pilot-plant facilities was reportedly 66 to 67 per cent (Gunderson, 1993 and Vesperman, 1993). NO_X reductions of 40 to 46 per cent have been experienced for K-Fuel (Alderman, 2003). Another demonstration completed in 2006 and using subbituminous coal for K-Fuel production revealed an SO_2 emission reduction of 38 to 40 per cent and NO_X emission reduction of 10 to 22 per cent. The K-fuel process delivered a mercury emission reduction of up to 70 per cent (KFx, 2006).

Because of limited operational experience, presently, there is no adequate information to estimate the price of the processed coal. In the future, the price for K-Fuel may be based on the price of competing coal on a heating value basis plus credits for environmental benefits (SO_2, NO_X and mercury emission reductions).

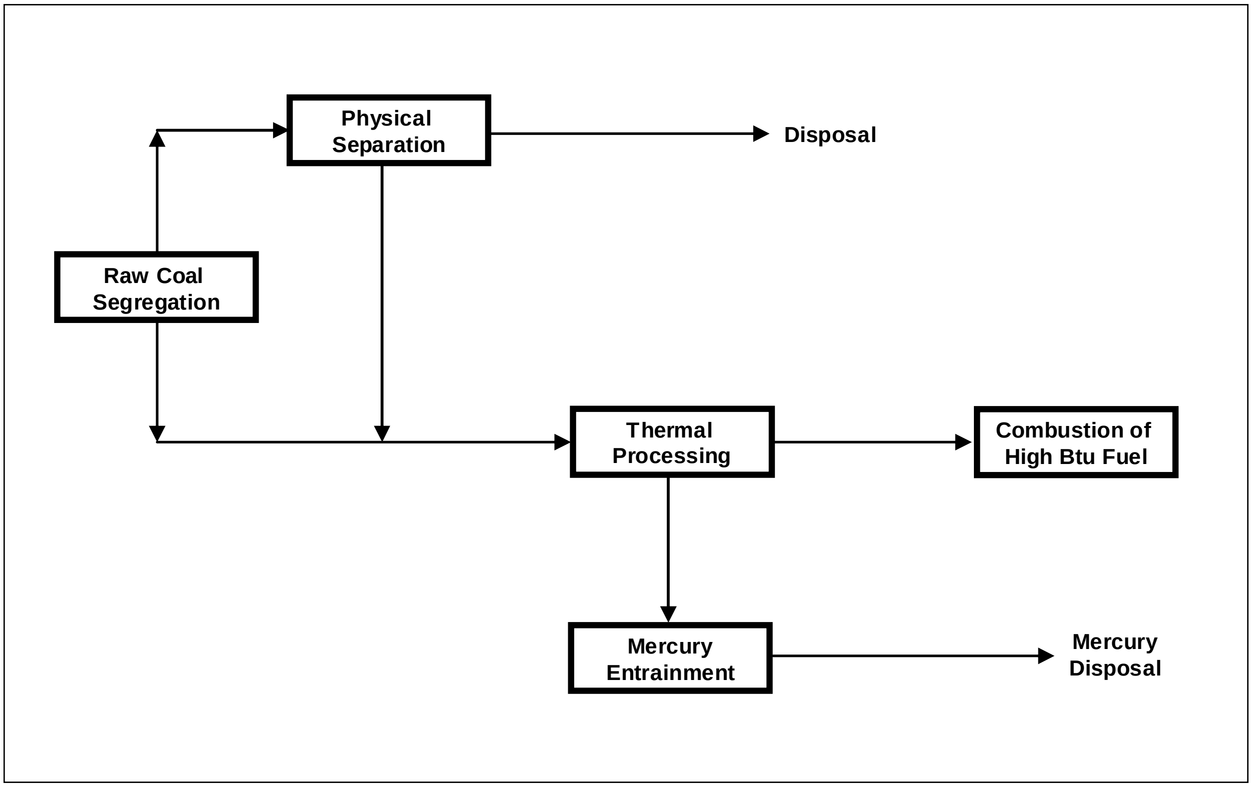

Figure 5. Schematic of K-Fuel technology.

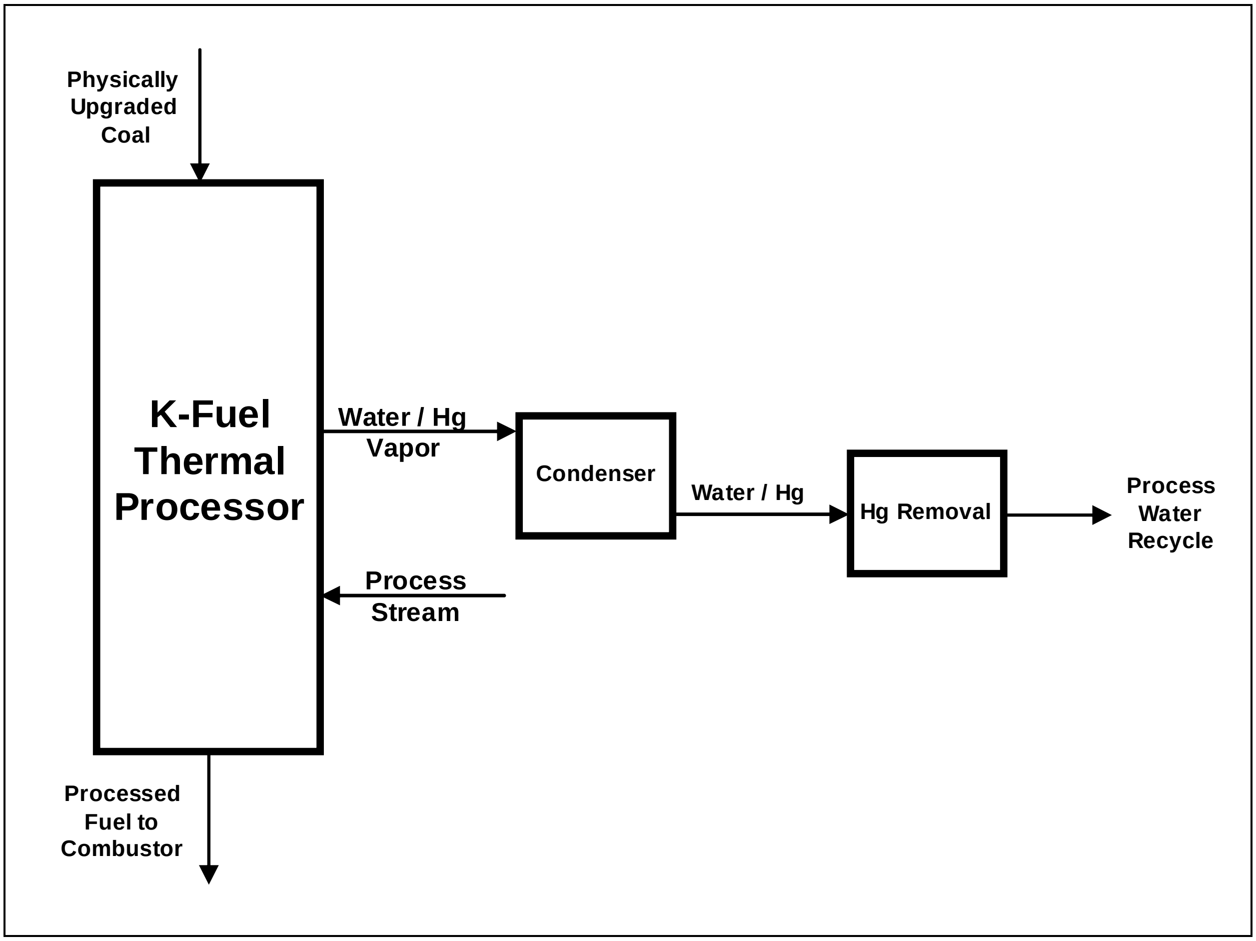

Figure 6. **K-Fuel thermal processing unit.**

5.3 COAL BLENDING/SWITCHING

Coal blending/switching at power plant is quite commonly used as power plants attempt to cost-effectively meet SO_2 emission limits. For example, in the U.S. some plants blend low-sulfur Powder River Basin (PRB) subbituminous coal with bituminous coals (typically with high sulfur content) to reduce SO_2 emissions without FGD installation. As a side effect of this SO_2 emission control strategy, mercury speciation may be altered (less oxidized mercury, more elemental mercury), thus compromising mercury capture in a downstream FGD (if one is available). Bituminous coals typically produce higher fraction of oxidized mercury in the flue gas than do subbituminous coal. Since oxidized mercury is water-soluble it is more readily captured in FGD systems. Consequently, the mercury capture efficiency of FGD systems depends largely on the fraction of oxidized mercury at the FGD inlet (Miller et al., 2006). Therefore, in the context of mercury control, the objective of coal blending is to increase halogen concentrations by mixing relatively high halogen content coal with low halogen coal that might be used at the plant. Figure 7 below shows the trend of increasing mercury capture in dry scrubber system (dry scrubber plus FF) with increasing fraction of western bituminous coal (high halogen content) in a western bituminous/subbituminous coal mixture. As can be seen, coal blending has the potential of increasing the mercury capture by up to approximately 80 per cent.

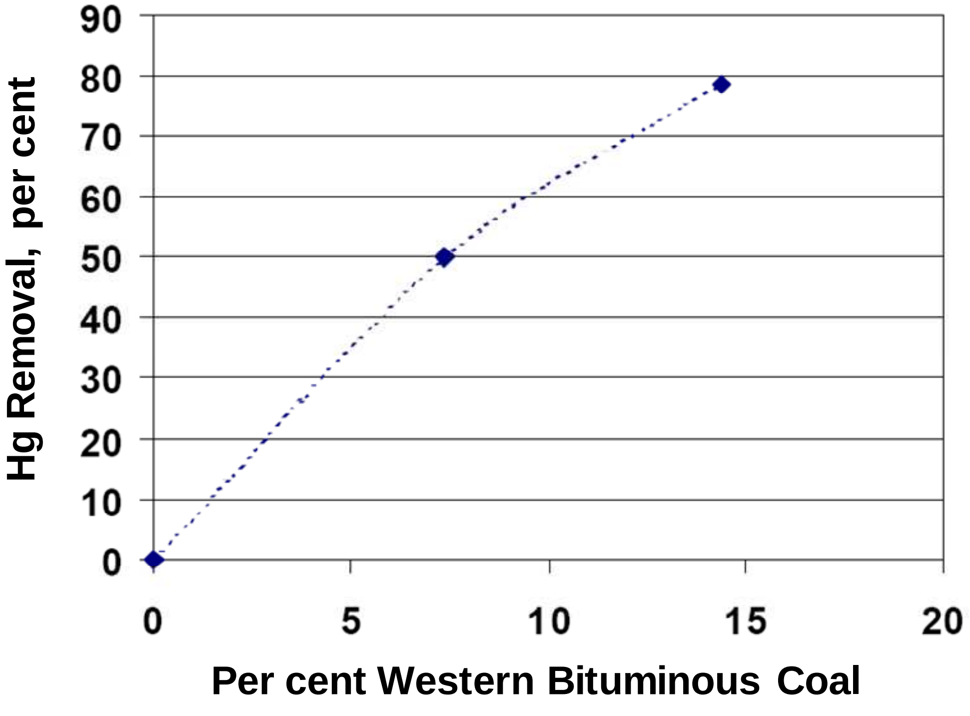

Figure 7. **Effect of coal blending on mercury capture in FGD (Bustard et al., 2005). Western bituminous coal: 0.07µg/g mercury, 106 µg/g chlorine, 1.4 µg/g bromine, and 0.93 per cent sulfur. Subbituminous coal: 0.077 µg/g mercury, 8 µg/g chlorine, 0.6 µg/g bromine, and 0.32 per cent sulfur.**

The trend of increasing mercury capture with increasing fraction of bituminous coal in a coal blend may be augmented if an SCR is installed upstream of an FGD. A field study at a large utility plant firing a 60 per cent subbituminous and 40 per cent bituminous blend at two identical boilers (one with SCR and the other without SCR) demonstrated an increase in oxidized mercury fraction from 63 per cent without SCR to 97 per cent with SCR. The unblended subbituminous coal in a unit without SCR would have achieved between 0 and 40 per cent oxidized mercury (ICAC, 2010). This dataset shows that the fraction of oxidized mercury increases with the addition of bituminous coal to subbituminous coal and that this effect is augmented by the SCR. The extent of oxidation was intriguing because normally there is little or no oxidation of mercury in the SCR operating on a PRB coal flue gas. PRB coals are characteristic with significantly lower chlorine and sulfur content and higher calcium content than bituminous coals. Previously, full-scale field tests conducted in three bituminous coal-fired plants showed increases in mercury oxidation across the SCR catalysts up to over 90 per cent. The resultant mercury removal in downstream wet scrubbers was 84 to 92 per cent with SCR operation compared to 43 to 51 per cent without SCR operation. However, plants firing subbituminous coals showed little change in mercury speciation across the SCR reactors (Laudal, 2002). Thus, coal blending of coal with characteristics similar to those of subbituminous and bituminous coal above may potentially be a cost effective approach for increasing mercury oxidation for plants firing low sulfur (and low chlorine, high calcium) coal. The effect may be more pronounced in plants equipped with SCR systems. An example of typical subbituminous and bituminous coal properties is given below in Table 3. These properties will vary depending on coal origin.

A comprehensive study of the effects of coal blending on mercury speciation in the presence of SCR examined the oxidation of mercury using blends ranging from 10 to 40 per cent bituminous coal mixed in with subbituminous PRB coal (Serre et al., 2008). The per cent oxidized mercury for the SCR inlet and SCR outlet is shown in Figure 8. As expected, the SCR inlet/outlet oxidized mercury concentration was higher for the 100 per cent bituminous coal firing (27 per cent/84 per cent) than for the 100 per cent subbituminous coal firing (6 per cent/3 per cent). As shown in Figure 8, the increase of mercury oxidation across the SCR was larger for a corresponding increase in the per cent bituminous coal. For example, for the 65/35 subbituminous/bituminous blend the amount of increase was 49 per cent (from 13 to 62 per cent). However, the increase was 14 per cent (from 6 to 20 per cent) for the 79/21 blend.

The above oxidation data were plotted as a function of coal chlorine content and are shown in Figure 9. Clearly, a higher degree of mercury oxidation was obtained with the higher chlorine content. When approximately 40 per cent of bituminous coal was added to the blend (337 ppm chlorine), the extent of mercury oxidation approached that measured for the pure bituminous coal firing (an approximately 50 percentage point increase of oxidized mercury concentration). Further increasing the chlorine content to over 900 ppm only netted a limited increase of mercury oxidation. The graph shown in Figure 9 may be useful to estimate expected levels of oxidized mercury at the SCR outlet as a function of coal chlorine content.

Table 3. Comparison of Typical Properties of Subbituminous and Bituminous Coals

Content	Subbituminous Coal, wt per cent [c]	Bituminous Coal, wt per cent [d]
Moisture [a]	30.24	12.00
Carbon [a]	48.18	55.35
Hydrogen [a]	3.31	4.00
Nitrogen [a]	0.70	1.08
Bromine [a]	0.0006	0.02
Chlorine [a]	0.003	0.100
Sulfur [a]	0.37	4.00
Ash [a]	5.32	16.00
Oxygen [a]	11.87	7.47
Mercury, ppm	0.1	0.1
SiO_2 [b]	35.51	50.82
Al_2O_3 [b]	17.11	19.06
TiO_2 [b]	1.26	0.83
Fe_2O_3 [b]	6.07	20.00
CaO [b]	26.67	3.43
MgO [b]	5.30	3.07
Na_2O [b]	1.68	0.60
K_2O [b]	2.87	0.37
P_2O_5 [b]	0.97	0.17
SO_3 [b]	1.56	1.22
HHV, MJ/kg	19.14	23.49

[a] ultimate analysis, as received, wt per cent
[b] ASTM coal ash analysis, wt per cent
[c] Wyoming PRB coal
[d] Illinois No. 6 coal

Notes:
HHV = higher heating value

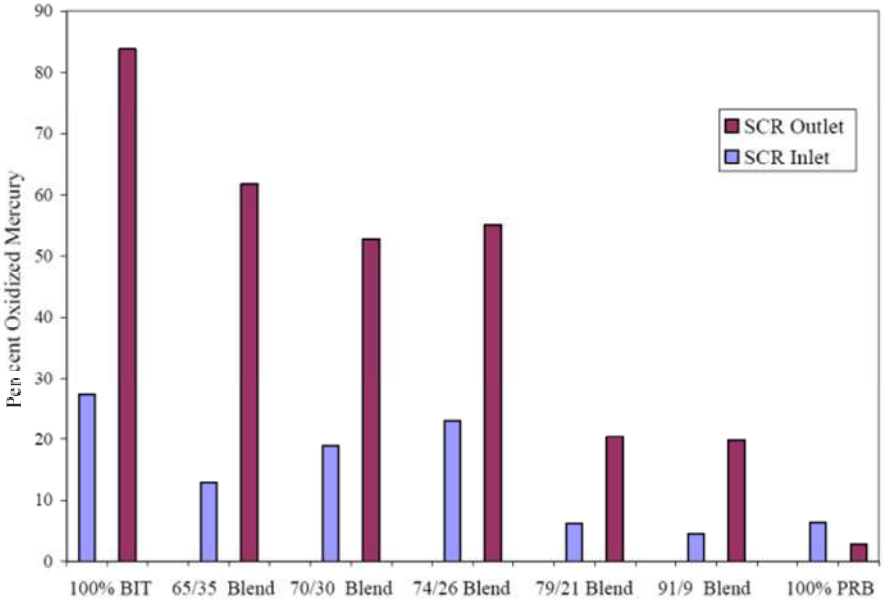

Figure 8. **Per cent oxidized mercury for different subbituminous/bituminous coal blends (from Serre et al., 2008).**

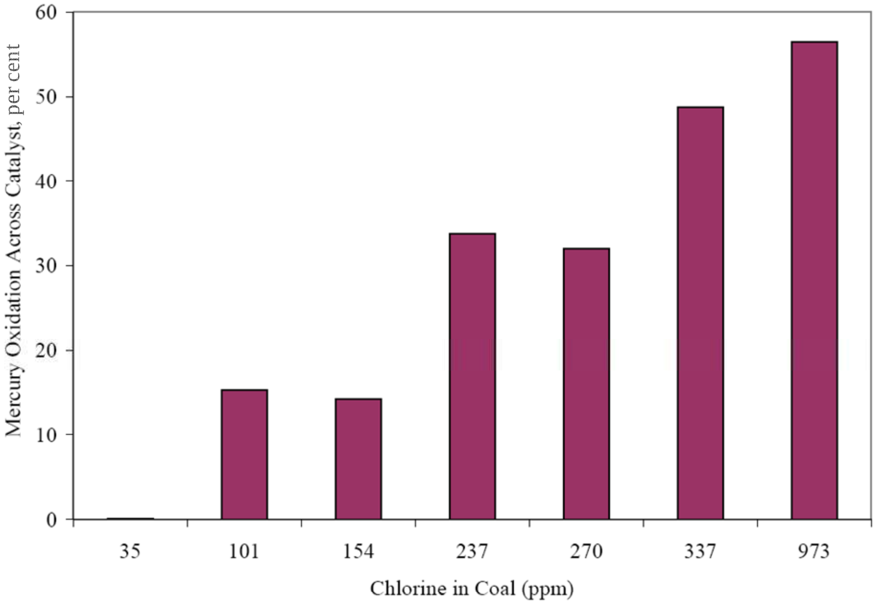

Figure 9. **Mercury oxidation across the SCR as a function of coal chlorine content (from Serre et al., 2008).**

The characteristics of different coal types play a major role in determining the speciation of mercury. This, in turn, can dramatically affect the amount of mercury captured in existing pollution control devices (FGD and SCR systems). As discussed in this section, coal selection and blending has the potential to reduce mercury emissions by over 80 per cent.

Another form of coal blending may be co-firing of biomass and/or waste materials with coal. The use of biomass as a co-fuel for burning alongside coal is increasing as biomass may be considered to be carbon-neutral in some situations. This means that less fossil fuel needs to be burned for the same amount of electricity produced. Replacing 5 per cent of coal with biomass on an energy basis worldwide would produce 40 GW of power and an emission reduction of 300,000,000 metric tons/year CO_2 (Koppejan and Baxter, 2005). However, whether biomass combustion is accepted as carbon-neutral varies with national policy in different countries. The co-firing of biomass at pulverized coal-fired plants is being encouraged in several EU countries as part of the move towards greener energy. However, there is no single EU policy on co-combustion and so individual member states are making their own legislation and action plans, most of which are based on financial incentives such as feed-in-tariffs and premiums. In 2004, co-firing of biomass with coal was being employed at 135 plants world-wide with most of the plants located in the USA, Germany, and Finland (Leckner, 2007).

The volatilization of trace elements from biomass is different to that from coal. However, mercury will volatilize in both cases and, once volatilized, its behavior in the combustion zone will be the same. However, the presence of other species in the flue gas, such as alkaline earth metals and volatile matter, will affect the behavior of mercury and the balance of these may change when biomass is included as an additional fuel source.

The trace element concentrations of biomass materials can be highly variable and there is also the possibility of significant variation between the characteristics of different batches of biomass from different sources, since biomass materials often reflect differences in trace elements in local soils and waters in the area of production. It is therefore likely that many plants switching to co-firing biomass materials will be required, under the operating permit or relevant requirements for the plant, to demonstrate that the new fuel material will not cause an increase in emissions of trace elements.

The element of most concern with respect to emissions from coal and biomass combustion at the moment is mercury. A full-scale demonstration rig at the EU TOMORED project in Germany showed the increase in mercury input to the boiler with the co-firing of 10 per cent sewage sludge, although the increase was only from 0.12 mg/kg with coal to 0.13 mg/kg with the sewage sludge/coal mix. There was no increase in mercury in the bottom or ESP ash or gypsum but an increase in mercury in the wastewater (from 0.001 to 0.002 mg/l) and an increase in emissions from the stack (2.2 $\mu g/m^3$ to 3.1 $\mu g/m^3$) (Thorwarth, 2006). Conversely the co-combustion of straw at the same plant showed increased Hg oxidation and removal across the ESP to the extent that it suggested that the co-firing of straw with coal could actually reduce mercury emissions to the air (Thorwarth and Scheffnecht, 2006).

A range of biomass materials are being been fired at the Drax Power Station in the United Kingdom and new emission standards became applicable. Although most biomass materials did not cause a problem with respect to elevated mercury emissions, the limit of 0.1 mg/kg for mercury could have been reached or even exceeded if the plant has chosen to fire only milled palm nut (DPL, 2009; Ghent, 2009).

The chlorine content of biomass can have an effect on the behavior of trace elements, especially mercury, during combustion. The mercury content of biomass materials are highly variable and in materials such as sewage sludge the concentrations can be significantly elevated. However, it would seem that co-firing these materials with coal does not lead to greater emissions of mercury but rather higher concentrations in the ash.

Cao and others (2008) studied mercury emissions during the co-firing of subbituminous coal (Powder River Basin) and biomass (chicken waste, wood, coffee residues and tobacco stalks) in a laboratory scale FBC system. The mercury content of the coal was around 0.12 ppm compared to concentrations of 0.01 ppm or below for all of the biomass materials. When firing the coal alone, around 38 per cent of the mercury was captured in the existing pollution control system (quartz filters). Co-firing high chlorine fuels such as chicken waste (22,340 ppm, by weight) could reduce mercury emissions by over 80 per cent whereas low chlorine fuels such as wood pellets (132 ppm) only reduced mercury emissions by 50 per cent. However, although tobacco stalks had a high chlorine content (4,237 ppm) the co-combustion of this biomass material in the FBC system did not reduce mercury emissions as significantly as might have been expected for such a high chlorine content. So, although the mercury emissions were strongly correlated to the gaseous chlorine concentration, they were not necessarily correlated to the chlorine content of the fuels (Sloss, 2010). It would seem that the addition of biomass as an additional fuel to coal combustion systems can help reduce emissions of mercury, often significantly. Therefore, co-firing biomass with coal could be considered as an approach to decreasing emissions of mercury from existing plants. However, this would need to be evaluated on a case by case basis.

5.4 COAL ADDITIVES

The amount of mercury capture generally increases as the amount of halogens in coal increase. Consequently, to promote increased capture for coals that have low halogen concentrations, additional halogens are often added such as bromine or chlorine salts. Alternatively, hydrogen chloride (HCl) or ammonium chloride (NH_4Cl) may be added. Halogen additives promote formation of oxidized and particulate-bound mercury, which are more easily captured in downstream devices. Halogen additives may be particularly useful in improving mercury removal for units firing low-halogen coals. The additives may be sprayed on coal or added as solids to a coal stream either upstream of the coal pulverizer or injected into the boiler.

Injection of halogen compounds into the boiler or direct addition to coal prior to combustion may promote corrosion in the boiler. To reduce the potential for corrosion, halogens may be delivered into the SCR process. Tests with HCl injection were conducted at two pilot scale facilities, each equipped with an SCR. One test facility was a 5 MW slip stream on a unit burning bituminous coal. For this location the oxidation of mercury increased from about 80 per cent to over 95 per cent when HCl was injected. The second test facility was a 5 MW slip stream on a unit burning subbituminous coal. For this location the oxidation of mercury increased from about 15 per cent to over 80 per cent when HCl was injected. No amounts of injected HCl were given for either facility (Honjo et al., 2008).

Bromine is thought to have an advantage over chlorine as its Deacon-type reactions are more favorable and it is consumed by SO_2 to a lower degree than chlorine (Buschman et al., 2005). The overall result is that much more molecular bromine (Br_2) than molecular chlorine (Cl_2) is available for interaction with mercury (Vosteen et al., 2003 and Vosteen et al., 2006). A heterogeneous pathway is thought to be important for oxidation of mercury with bromine under coal-fired flue gas conditions (Pavlish et al., 2003 and Senior et al., 2008). This is despite the fact that typically chlorine content in coal is much higher than that of bromine (Vosteen 2006a and Rini and Vosteen, 2008). Full-scale tests were conducted using an KNX^{TM} additive (52 wt per cent water solution of calcium bromide) as a pre-combustion additive at a 25 ppm in coal equivalent level. A mercury emission reduction of 92 to 97 per cent was consistently observed on a 600 MW unit firing subbituminous coal and equipped with an SCR and wet FGD (Rini and Vosteen, 2009). Bromide salts can promote the oxidation of mercury even if only small amounts are added. This was confirmed in an extensive testing program by the Electric Power Research Institute (EPRI). Full-scale tests conducted at 14 units firing low chlorine coals demonstrated more than 90 per cent of flue gas mercury oxidation for bromide additions equivalent to 25 to 300 ppm in coal (Chang et al., 2008).

The comparison of performance of bromine-based and chlorine-based additives at coal-fired boilers is shown in Figure 10 and gives the per cent reduction of baseline elemental mercury as a function of halogen addition rate (Vosteen and Lindau, 2006). As can be seen, for any amount of halogen addition, bromine was much more effective in decreasing the amount of baseline elemental mercury than chlorine. Baseline elemental mercury reduction of 80 per cent could be achieved by adding less than 200 ppm of bromine-based additive. An order of magnitude more of chlorine-based additive was needed to achieve the same level of baseline elemental mercury reduction.

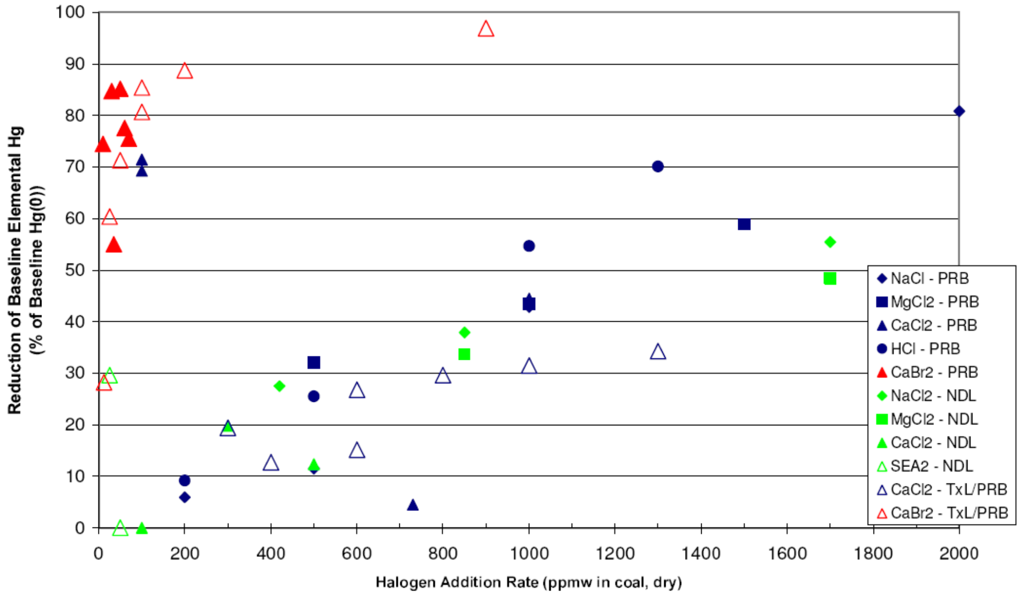

Figure 10. Comparison of the performance of bromine- and chlorine-based additives (from Vosteen and Lindau, 2006).

Some balance-of-plant considerations which are still being investigated and related to the use of halogen compounds include corrosion potential in the boiler and wet FGD, application of fly ash for concrete use, leachability from fly ash, and effect on mercury capture in and re-emission from wet FGD. However, the use of halogens, especially bromine, has the potential to oxidize elemental mercury by up to about 80 per cent.

6 IMPROVEMENT OF PARTICULATE MATTER CONTROL

As described in previous chapters, substantial mercury removal was measured for ESPs and FFs during the ICR. These data are compiled in Table 4 for PM control devices. In addition to a strong effect of coal type on measured mercury removal, there was also significant variability in the extent of measured mercury removal for a given PM control. For example, while the average mercury removal for ESP operating on a bituminous coal was 36 per cent, the range of measured removals varied from 0 to 63 per cent.

- **Capture of mercury in ESP systems can on the average be expected to be up to about 36 per cent. Mercury capture can be significantly greater in FF systems, at up to 90 per cent.**
- **Maximizing mercury capture in ESP or FF systems can be achieved by a number of means including coal switching/blending, temperature adjustments and improvements in the systems' operating efficiencies.**

Table 4. Mercury Removal by PM Control Devices, per cent

Coal Type/ PM Control	Bituminous	Subbituminous	Lignite
ESPc only	36 (0 – 63)	9 (0 – 18)	1% (0 – 2)
ESPh only	14 (0 – 48)	7 (0 – 27)	none tested
FF only	90 (84 – 93)	72 (53 – 87)	none tested

Note: Mercury removal given as average (US EPA, 2002) and range measured (Srivastava et al., 2006)

The range of measured mercury removals, especially for the ESP, may be an indicator of the potential for improvement of mercury capture resulting from increased efficiency of a PM collector or due to modification of flue gas and fly ash properties that promote formation of PM-bound mercury. It is important to understand the PM collection performance of the device since this in turn affects the device's capability to reduce mercury emissions.

ESPs are typically designed to achieve greater than 99 per cent PM collection efficiencies. An ESP operates by imparting an electrical charge to incoming particles and then attracting the particles to oppositely charged metal plates for collection. The particles collected on the plates are removed by periodically rapping the plates, after which they fall into a collection hopper. The effectiveness of an ESP depends largely on the electrical resistivity of the particles being collected. Above and below an optimum value that exists for a given ash, particles become less effectively charged and collected. The PM collection efficiency of an ESP is also a function of sulfur content of coal, which affects the resistivity of fly ash. Coal that contains a moderate to high amount of sulfur, typically a bituminous coal, produces an easily collected fly ash. Lower sulfur coal, such as subbituminous coal, produces a higher resistivity fly ash that is more difficult to collect. Resistivity of the fly ash can be changed by decreasing temperature in the ESP or by conditioning the particles upstream of the ESP with SO_3, sulfuric acid (H_2SO_4), water, sodium, or ammonia (NH_3). For a given coal fly ash, the effectiveness of PM collection efficiency by an ESP is a function of particle size. Particles larger than about 1 to 8 micrometer (μm) are typically collected with efficiencies from 95 to 99.9 per cent. However, particles near the 0.3 μm size are in a poor charging region that reduces collection efficiency to 80 to 95 per cent (Lawless, 1996).

An ESP can be used at one of two locations in a boiler system: the so-called "cold-side" ESP (ESPc) and the so-called "hot-side" ESP (ESPh). An ESP installed downstream of the air heater (flue gas temperature between 130 and 180 °C) is an ESPc. An ESP installed upstream of the air heater (flue gas temperature between 315 to 400 °C) is an ESPh. ESPh allows taking advantage of the lower fly-ash resistivity at higher temperatures. This is particularly important for units burning low-sulfur coal resulting in fly ash with higher electrical resistivity.

FFs are the best commercial PM control technology for high-efficiency collection of fine particles, especially submicron particles. FFs collect fly ash by passing the gases through a porous fabric material, which traps fly ash on its surface forming the so-called filter cake. Gas flow through a FF becomes excessively restricted if the layer of filter cake on the bags becomes too thick. To facilitate the flow of gas, the filter cake is removed periodically. Depending on the filter cake removal method, two basic types of FF are reverse-air and pulse-jet. The selection of filter cake removal method is dependent on whether the fly ash particles are collected on the inside or outside of the bag. For designs in which the dust is collected on the inside of the bags, the dust is most often removed by blowing air through the bag from the opposite side (reverse-air FF). For designs with the dust collected on the outside of the bags, a jet of compressed air is used to displace the filter cake from the bag (pulse-jet FF). For both designs, the displaced dust particles fall into a hopper underneath. Typically, other than routine replacement of worn out bags in the FF, less maintenance is required for the FF than for the ESP.

Other devices used to control PM emissions include wet PM scrubbers and mechanical collectors. These devices provide lower collection efficiencies than those discussed for ESPs or FFs. Further, collection of fine particles (submicron particles) is lower for wet PM scrubbers or mechanical collectors than for ESPs or FFs.

6.1 MERCURY REMOVAL

Fundamental modeling of mercury removal in ESP indicate that mass-transfer limitations, even under idealized conditions, may restrict the potential for mercury capture by PM collected on electrodes in an ESP (Clack, 2006). ESPs generally remove only PM-bound mercury in the process of collecting PM. PM-bound mercury is preferentially bound to UBC. Mercury adsorption capacity of inorganic fractions (fly ash) is typically low compared to the UBC present in fly ash. A relationship between the amount of UBC and mercury removal across ESPc has been observed for bituminous coal fly ash (Senior and Johnson, 2008). This trend is shown in Figure 11 below, showing the percentage capture (percentage of mercury incoming to the ESP) as a function of the amount of UBC. In Figure 11, UBC is expressed at the measured loss on ignition (LOI). As can be seen, between 20 and 40 per cent mercury capture can be expected in an ESP capturing fly ash containing about 5 per cent UBC. For higher UBC contents, mercury capture of as much as 80 per cent could be seen, likely a function of halogens present. Higher concentration of mercury in ESP fly ash was observed when bromide was added to boiler, compared to the no-addition case (Vosteen et al., 2003).

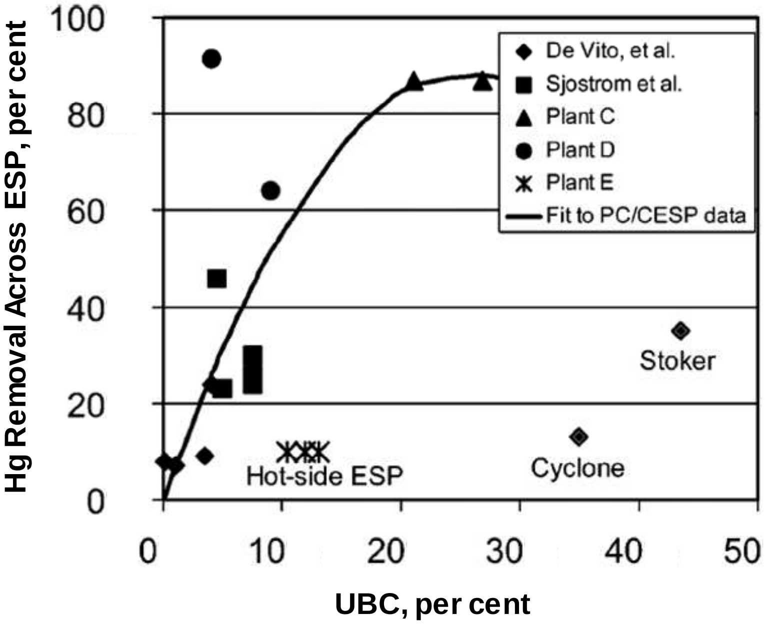

Figure 11. Mercury capture across ESP as a function of the amount of UBC (from Senior and Johnson, 2005).

In addition to the amount of UBC, UBC properties such as surface area, particle size, porosity, and composition may also affect the amount of mercury captured in the ESP (Lu et al., 2007). The study found that while UBC content in fly ash decreased with decreasing particle size, the mercury content of UBC generally increased with decreasing particle size. In addition, the particle size of UBC was found to be the major factor impacting mercury adsorption. Thus, an increased efficiency of an ESP and resultant increased capture of fine fly ash and fine UBC will likely cause a decrease in mercury emissions. However, it should be noted that most of the mass of UBC is in very large particles.

Other important factors governing the amount of mercury that is captured on fly ash (and subsequently removed from the flue gas) are: the temperature of the ESP, the use of SO_3 as a flue gas conditioning agent, and the type of coal. Typically, higher mercury capture is observed in ESPs installed on boilers burning coals with higher halogen content and producing higher levels of UBC in the flue gas. Both of these parameters promote formation of oxidized mercury (Hg^{2+}) and PM-bound mercury, which are easier to capture in the ESP than Hg^0. It follows that if the performance of the ESP can be improved, an additional amount of mercury could be removed from flue gas. The amount of this additionally removed mercury would be a function of the amount of additional PM removed by the ESP. "Low-cost" approaches such as, for example, accurate alignment of plates, adjustment of rapping pattern, elimination of in-leakages, etc., can be used to improve the PM collection efficiency of ESPs (Zykov et al., 2004). A recent example of a successful ESP performance upgrade project has been reported for a 1300 MW plant (Deye and Layman, 2008).

To understand the potential for improved mercury capture via the improvement of ESP efficiency a simplified analysis was conducted. The estimate of the amount of mercury that could be removed from the flue gas as a result of the improved performance of a fleet of ESPs is presented below. Several assumptions were made to calculate the amount of the mercury removed:

- Efficiency improvements implemented for the ESP fleet (as expressed by coal usage) installed on the electricity generating capacity utilizing 40 million metric tons of coal annually
- Range of mercury content in coal between 0.01 and 1.1 ppm
- 80 per cent of mercury content in coal emitted
- 42 per cent ash content of coal
- Ash partition – 80 per cent fly ash and 20 per cent bottom ash
- Existing fleet of ESPs operating at 98 per cent PM capture efficiency on the average
- Range of ESP efficiency improvement from 0.5 to 1.5 percentage points

Utilizing the assumptions described above, the amount of incremental mercury emission reduction was calculated for various levels of ESP performance improvement and for varying mercury content of coal, assuming transfer of mercury from gas phase to solid phase. The results are shown in Figure 12.

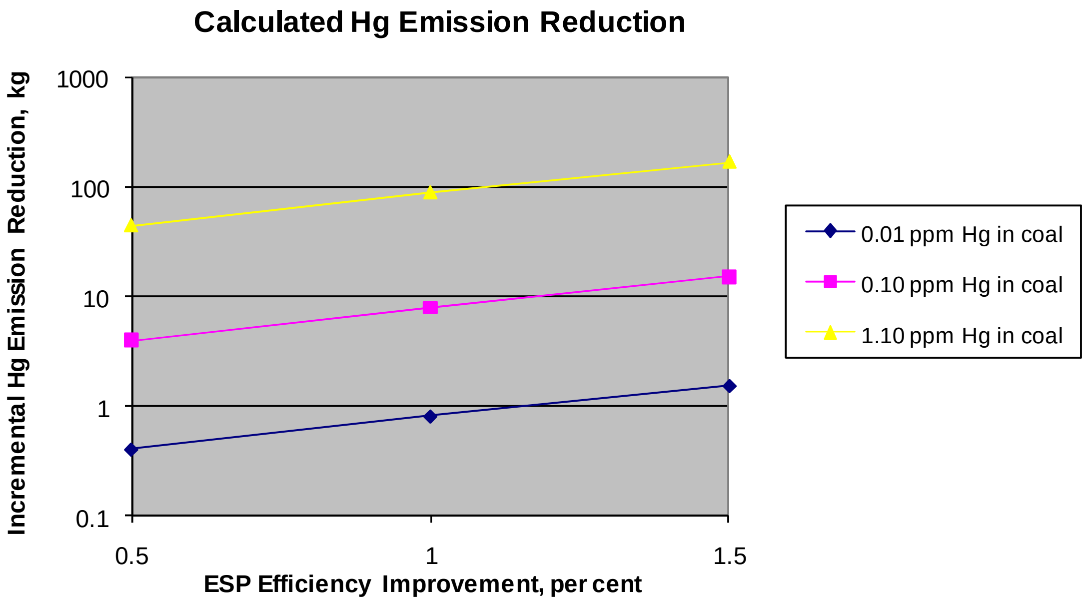

Figure 12. Incremental mercury reduction achieved as a result of ESP efficiency improvement.

As can be seen, about 10 kg of incremental mercury emission reduction could be achieved on an annual basis for the example given above if the efficiency of the ESP fleet was improved by 1 per cent, assuming 0.1 ppm of mercury in coal burned. This amount of incremental mercury removal is likely conservative because the calculations do not account for the increased amount of fine PM removed from the flue gas as the efficiency of the ESP increases. For example, the calculations assume the same incremental amount of mercury removed when ESP efficiency is increased from 98 to 98.5 per cent as when the efficiency is increased from 98.5 to 99 per cent. However, as the amount of fine PM captured increases, the amount of incremental mercury removed is also expected to increase. Since a larger amount of fine PM is expected to be removed when ESP efficiency is improved from 98.5 to 99 per cent than from 98 to 98.5 per cent, more incremental mercury would be removed from the flue gas in the former case. The above results are presented to illustrate the trend of increased mercury capture as a result of the improved efficiency of an ESP. The actual amount of mercury removed by the ESP may be site-specific and be a function of the ESP design, type of coal burned at the power plant, and plant operating parameters (amount of UBC in fly ash).

Higher removals of mercury are generally observed in FFs than in ESPs. FFs are more effective in removing fine PM (most importantly, submicron PM) than an ESP and they tend to remove more of the gas-phase mercury than ESPs. In addition to longer contact time, better contact is provided in an FF (gas penetrates through the filter cake) than in an ESP (gas passes over the surface of the cake). For example, consistent with earlier ICR effort, a study comparing the capture of mercury in ESPs and FFs in coal-fired power plants in China revealed between 4 and 20 per cent capture in ESPs and between 20 and 80 per cent capture in FFs (Wang et al., 2008).

Although actual values will vary with coal type and plant operating conditions, the capture of mercury in ESP systems can be expected to be on the average anywhere up to 36 per cent (range measured 0 to 63 per cent as per Table 4). Capture can be significantly greater in FF systems, at up to 90 per cent. Maximizing mercury capture in these systems can be achieved by a number of means including coal switching/blending, halogen addition, temperature reductions, and improvements in the systems' operating efficiencies.

Mercury capture in wet PM scrubbers would be expected to be significant, as from the point of mercury control, a wet PM scrubber system has chemistry similar to that of wet FGD. Therefore, oxidized mercury would be expected to be captured in a wet PM scrubber. The mercury removal by wet PM scrubber could be augmented by coal switching/blending or the addition of oxidants or halogens. Mechanical collectors (such as cyclones) would be expected to only be capable of low mercury removal due to their limited capability to separate submicron particles from the flue gas.

7 CO-BENEFIT MERCURY REMOVAL BY SO_2 AND NO_X CONTROLS

- **The most important step to maximize the amount of co-benefit mercury capture is to maximize the amount of oxidized mercury.**
- **Mercury removal of up to about 90% can be expected in wet FGD systems.**
- **SCR may increase the amount of oxidized mercury up to about 85 per cent and thus improve capture by wet FGD.**
- **Reduction of oxidized mercury (re-emission) may occur in wet FGD. It can be prevented by correcting scrubber chemistry.**

Co-benefit mercury removal by non-mercury air pollution control equipment is most often accomplished in two fundamental modes: removal of oxidized mercury (Hg^{2+}) in a wet FGD scrubber and removal of PM-bound mercury (Hg^P) in PM control device (ESP or FF). Co-benefit removal of mercury can also be accomplished in spray dryer absorber. Accordingly, the amount of the co-benefit removal may be augmented by the increase of the fraction of oxidized mercury in the total mercury flue gas content or by the improvement of PM control effectiveness (Sloss, 2009). The former can be accomplished by the addition of chemical compounds (oxidizing agents) or by oxidation of mercury over catalysts. The catalyst may be placed in the flue gas for the sole purpose of mercury oxidation or may be installed for another purpose (e.g., for the control of NO_X emissions) and thus provide the co-benefit. Depending on the configuration of pollution control equipment, varying amounts of mercury removal could be accomplished. General trends of co-benefit mercury removal for different configurations of existing air pollution control equipment are shown in Table 5.

As can be seen from Table 5, other than optimizing the performance of ESP itself, the most important step one could take to maximize the amount of co-benefit mercury capture is to maximize the amount of Hg^{2+}. This section will present methods utilized for the purpose of increasing, and in some cases preserving, the amount of co-benefit capture.

Table 5. Trends of Co-benefit Mercury Removal

Existing Control Equipment	Qualitative Mercury Capture
ESPc only	Good capture of particulate- or sorbent-bound; better co-benefit capture for low-sulfur bituminous coals than low rank coals.
ESPh only	Low co-benefit capture; may require specially formulated sorbents for high temperature mercury capture.
FF only	Good co-benefit capture of Hg^{2+}; Hg^0 may be oxidized across the FF.
ESPc + Wet FGD	Generally, good co-benefit capture for bituminous coals due to presence of soluble Hg^{2+} in the flue gas. Relatively poor capture for low rank coals. Hg^0 re-emission may decrease the amount of co-benefit.
ESPh + Wet FGD	Moderate co-benefit capture for bituminous coals; poor co-benefit capture for low rank coals.
SDA + FF	Very high co-benefit capture expected for bituminous coals; somewhat less co-benefit capture expected for low rank coals.
FF + Wet FGD	Good co-benefit capture for bituminous coal, comparable co-benefit capture for low rank coals; Hg^0 may be oxidized across the FF and captured in the wet scrubber.
SCR + ESPc	Good capture of particulate- or sorbent-bound Hg; better co-benefit capture for bituminous coals than low rank coals.
SCR + ESPh	Low co-benefit capture.
SCR + ESPc + Wet FGD	Good capture of particulate- or sorbent-bound Hg; better co-benefit capture for bituminous coals than low rank coals. SCR enhances capture for bituminous coals by oxidizing Hg^0 to Hg^{2+} form.
SCR + SDA + FF	Very high co-benefit capture for bituminous coals, less for low rank coals. SCR enhances capture by oxidizing Hg^0 to Hg^{2+} form, given availability of chlorine in the flue gas.
SCR + ESPh + Wet FGD	Generally, poor capture of particulate-bound mercury and total mercury for low rank coals. SCR enhances capture for bituminous coals by oxidizing Hg^0 to Hg^{2+} form, given availability of chlorine in the flue gas.
SCR + FF + Wet FGD	Generally, high level of mercury capture for all coals. SCR enhances capture for bituminous coals by oxidizing Hg^0 to Hg^{2+} form given availability of chlorine in the flue gas.

Note: Information from Srivastava et al., 2006, was used in compilation of this table.

7.1 CO-BENEFIT MERCURY CAPTURE BY WET FGD

In plants with an existing wet FGD, the most attractive strategy for mercury control is to increase (and in some cases to preserve) the amount of co-benefit mercury capture. Operation of a wet FGD requires that a PM control device be installed upstream of the wet FGD scrubber (Srivastava and Jozewicz, 2001). With PM controls installed, the control of Hg^P takes place. As mentioned before, gaseous compounds of Hg^{2+} are generally water-soluble, and thus wet FGD systems are expected to capture them efficiently (Reddinger et al., 1997; DeVito and Rossenhoover, 1999). However, gaseous Hg^0 is insoluble in water and therefore does not absorb in FGD slurries. Data from actual facilities has shown that capture of Hg^{2+} in excess of 90 per cent can be expected in calcium-based wet FGD systems, though there are cases where significantly less capture has been measured as a result of unfavorable scrubber equilibrium chemistry (Niksa and Fujiwara, 2004). It has also been shown that under some conditions, Hg^{2+} may be reduced in wet FGD to Hg^0, which could then be re-emitted (Laudal et al., 2003 and Nolan et al., 2003). Thus, in the case of wet FGD the optimization of co-benefit strategy sometimes means preserving the amount of Hg^{2+} in the system in order to prevent re-emission of mercury. Mercury re-emission may take place when oxidized mercury is absorbed by the wet FGD slurry, reduced to elemental mercury, and then transferred to gas phase to exit the scrubber. The net effect of re-emission is the limitation of mercury removal by a wet FGD. The occurrence and the extent of mercury re-emission from wet FGDs depend on its chemistry.

Wet FGD chemistry in respect to the removal of mercury can be outlined as follows. When gaseous compounds of Hg^{2+} are absorbed in the liquid slurry of a wet FGD system, the dissolved species are believed to react with dissolved sulfides from the flue gas, such as hydrogen sulfide (H_2S), to form mercuric sulfide (HgS); the HgS precipitates from the liquid solution as sludge. In the absence of sufficient concentration of sulfides in the liquid solution, a competing reaction with sulfites that reduces dissolved Hg^{2+} to Hg^0 is believed to take place. Once this reduction occurs, the newly formed Hg^0 is transferred to the flue gas and increases the concentration of Hg^0 in the flue gas passing through the wet FGD (McDermott, 1999). Hg^{2+} reduction and subsequent Hg^0 re-emission may be more significant in magnesium-enhanced lime scrubbers. These scrubbers operate with the much higher sulfite concentration compared to limestone systems (Renninger et al., 2004). In some cases, the reduction of Hg^{2+} to Hg^0, and subsequent re-emission, has been abated with the help of sulfide-donating liquid reagent (US EPA, 2002). In addition, transition metals in the slurry (originating from fly ash in the flue gas) are believed to play an active role in the conversion reaction, since they can act as catalysts and/or reactants for reducing oxidized species. There also appears to be increased potential for re-emissions of mercury in wet FGDs with appreciable mercury concentrations in the liquor phase (Chang et al., 2008).

7.2 CO-BENEFIT OF NO_X CONTROL

SCR technology has been designed to reduce NO_X through a catalytically enhanced reaction of NO_X with NH_3 reducing NO_X to water and nitrogen. This reaction takes place on the surface of a catalyst, which is placed in a reactor vessel. The reactor ensures that the flue gas is uniformly distributed over the catalyst; it also determines the flue gas velocity. Typical catalyst materials are vanadium-pentoxide and tungsten oxide on a "coated" substrate structure that may take forms such as plate or honeycomb. Under certain conditions, SCR catalysts have been shown to change mercury speciation by promoting the oxidation of Hg^0 to Hg^{2+}, particularly for bituminous coal. Early field measurements conducted in Europe indicated that an SCR reactor installed for NO_X emission control promotes the formation of oxidized mercury species (Gutberlet et al., 1992). Field studies conducted later in the U.S. also confirmed an increase in Hg^{2+} species across the SCR reactors (Laudal et al., 2002). It should be pointed out that under steady operating conditions the SCR itself does not remove mercury. Instead, by increasing the amount of Hg^{2+}, the SCR could improve mercury capture in the downstream wet FGD systems, resulting in the co-benefit removal of mercury (Chu, 2004).

Bench-scale tests that followed the above initial field observations suggested that HCl is an important exhaust gas constituent that is necessary for providing the chlorine for oxidation of Hg^0 to HgCl across the SCR catalyst (Lee et al., 2003). The extent of oxidation of Hg^0 by SCR catalyst and subsequent removal of oxidized mercury in a wet FGD may be affected by the following (Pavlish et al., 2003 and Winberg et al., 2004):

- Halogen content of the coal
- Amount of catalyst used to treat the gas stream
- Temperature of the SCR reaction
- Concentration of NH_3 and its distribution in the flue gas
- Age of the catalyst

Since the operational parameters of the SCR such as temperature, concentration of NH_3 in the flue gas, catalyst bed size, and catalyst age are dictated by the NO_X control strategy, the parameter that shows the most promise for the optimization of mercury removal is the chlorine content of the coal. As discussed before in the coal blending section, oxidation of Hg^0 to Hg^{2+} is greater for bituminous coals (85 to 90 per cent of mercury in the oxidized form) than for subbituminous coals. The results of thermochemical equilibrium calculations of mercury species concentration demonstrated that oxidation of Hg^0 to Hg^{2+} with SCR when firing subbituminous coal was limited by equilibrium rather than by kinetics (Senior, 2004 and Senior and Liniewile, 2004). It follows that, other than altering NO_X control parameters of the SCR, an improvement in oxidation of Hg^0 to Hg^{2+} with SCR on boilers firing low-rank coals is not possible without a change in flue gas chemical composition (e.g., higher active chlorine contents of the flue gas) or without lowering catalyst temperature. Thus, the maximum co-benefit of SCR may be achieved by an appropriate coal-blending. Alternatively, bromide addition can be used (Vosteen et al., 2006).

8 MERCURY-SPECIFIC CONTROL

Injection of sorbents into the flue gas of coal-fired boilers for mercury control has been applied at boilers in Germany since the 1990's (Wirling, 2000) and has been demonstrated in the United States on several full-scale systems (GAO, 2009). In 2010, ACI is considered to be commercially available technology with over 60 GW under contract and over 12 GW operational (ICAC, 2010a). Typically, the powdered sorbent is injected upstream of the existing PM control device. Alternatively, sorbent may be injected downstream of an existing ESP and a retrofit FF is then added (the toxic emission control process [TOXECON^{TM}] configuration). The third demonstrated configuration for sorbent injection is TOXECON II^{TM} in which sorbent is injected into the middle fields of the existing ESP.

- **ACI has been demonstrated on a number of full-scale systems and in 2010 is a commercial technology with 60 GW under contract and over 12 GW operational.**
- **Chemically treated activated carbons allow for higher mercury removal rates and for lower carbon injection rates for the same amount of mercury removal than un-treated activated carbons.**
- **Installation of an additional PJFF allows preservation of fly ash quality while conducting the ACI.**

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

- Physical and chemical properties of the sorbent
- Injection rate of the sorbent
- Flue gas parameters such as temperature, concentrations of halogen species (e.g., HCl, HBr), and concentration of sulfur trioxide (SO_3)
- Existing air pollution control configuration

US DOE, in cooperation with the US EPA and EPRI, along with a number of research organizations and utilities conducted an almost two decades-long program devoted to mercury emission control. The initial field testing (Phase I) investigated untreated ACI and the potential to improve the wet FGD capture. Phase II yielded full-scale testing of chemically-treated ACI. Phase II also included testing of configurations designed to preserve fly ash quality following sorbent injection (TOXECON^{TM} and TOXECON II^{TM}) and evaluations of chemical additives as well as Hg^0 oxidation catalysts for wet FGD application. Phase III was conducted to assess potential balance-of-plant impacts associated with operation of mercury control. Together, Phase I, II, and III results are likely the most complete database available on the approaches to control mercury emissions from coal combustion for electricity generation.

8.1 SORBENT INJECTION UPSTREAM OF PM CONTROL

This section presents the information on sorbent injection upstream of the existing PM controls. Figure 13 is a summary of US DOE Phase I tests conducted with untreated ACI at four power plants. This set of data has provided the first systematic insight into the performance and limitations of ACI. As can be seen, injection of sorbent in increasing amounts (mass of sorbent/unit of gas volume) generally tends to increase mercury removal efficiency. However, at the Pleasant Prairie Power Plant, which was burning subbituminous coal (the remaining three plants used low-sulfur bituminous coal), a removal of approximately 60 per cent was reached with modest sorbent addition. Additional carbon injection yielded only small incremental mercury removal. This is the result of a combination of lower levels of chlorine in subbituminous coal and neutralization of halogen species by high levels of sodium and calcium in the subbituminous coal fly ash. As a result, there is little free chlorine in the flue gas stream for mercury oxidation. Mercury oxidation (with chlorination of the surface as the initial step) is necessary for capture of Hg^0 by untreated ACI and in general, the efficiency of mercury capture with untreated ACI increases with the amount of Hg^{2+} in flue gas. The amount of Hg^{2+} in flue gas is directly influenced by the amount of chlorine present in the flue gas. Thus, mercury capture with untreated ACI may be limited in plants firing low-rank coal.

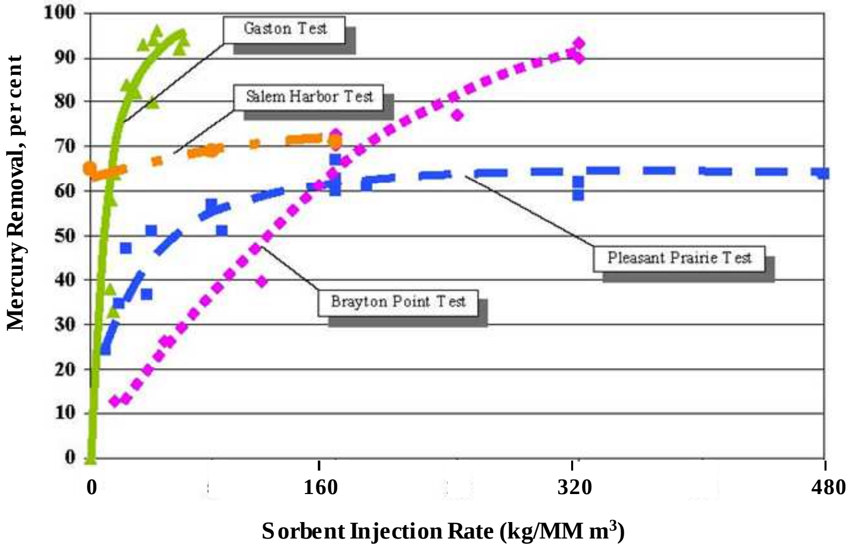

Figure 13. Summary of Phase I tests with untreated ACI (from US DOE 2005).

Another result of Phase I testing was the confirmation of the temperature effect. From earlier bench scale tests it was known that temperature impacted the adsorption capacity of untreated ACI. When flue gas temperature at the injection point upstream of the existing PM control device was around 150 °C, untreated ACI has been shown to work effectively. However, at temperatures approaching 170 °C or more, the effectiveness of untreated ACI dropped off rapidly (Durham, 2003). This is because relatively weak physical bonds on the surface of untreated activated carbon get ruptured at higher temperatures resulting in lower sorption capacity.

Based on the findings above, Phase I concluded that untreated ACI was generally effective for mercury capture on low-sulfur bituminous coal applications, but less effective for the subbituminous coal applications and applications at higher flue gas temperature, such as in ESPh.

As mentioned before, chlorination of the carbon surface is the first step in the mercury oxidation and capture process (Olson et al., 2008). Often, the effectiveness of untreated ACI may be limited due to inadequate free chlorine in the flue gas. To overcome this set of limiting conditions associated with untreated ACI for mercury control in power plant applications, treated ACI sorbents have been developed (Nelson, 2004, Nelson et al., 2004, Pavlish et al., 2008). The most often used and most thoroughly demonstrated treatment/enhancement that was used to improve the performance of ACI was bromine and/or chlorine. However, iodated carbon sorbents have initially been successfully tested at bench scale (Ghorishi et al., 2002).

In terms of mercury removal performance, treated activated carbon sorbents are capable of offering several potential benefits over untreated activated carbon: (1) they expand the application of sorbent injection to situations where untreated activated carbon may not be effective; (2) their use avoids installation of a downstream FF, thereby improving cost-effectiveness of mercury capture; (3) can be operated at lower injection rates, which leads to fewer plant impacts and a lower carbon content in the captured fly ash; (4) results in better performance with subbituminous and lignite coals; and (5) may be a relatively inexpensive and attractive control technology option for countries implementing mercury control strategy as it does not involve the capital intensive FF installation.

During Phase II of the mercury test program, a major improvement in performance of mercury control was observed during full scale field tests of treated AC injected upstream of the existing PM device (Feeley et al., 2008). This trend is shown in Figure 14, which compares mercury removal with untreated AC and treated AC on applications for plants burning lignite or subbituminous coal. The results shown in Figure 14 were the culmination of the development efforts for treated ACI. As can be seen, improved mercury capture efficiency with treated ACI was possible at power plants burning lower-rank coals and these improved capture efficiencies were attainable at relatively low injection rates. The treated ACI achieved in excess of 90 per cent mercury capture at an injection rate of 50 kg per million actual cubic meters (kg/MMm^3) or less (Feeley et al., 2008). Higher injection rates were required to achieve 90 per cent mercury capture during Phase I when untreated ACI was used; in some cases it was not possible to achieve 90 per cent capture.

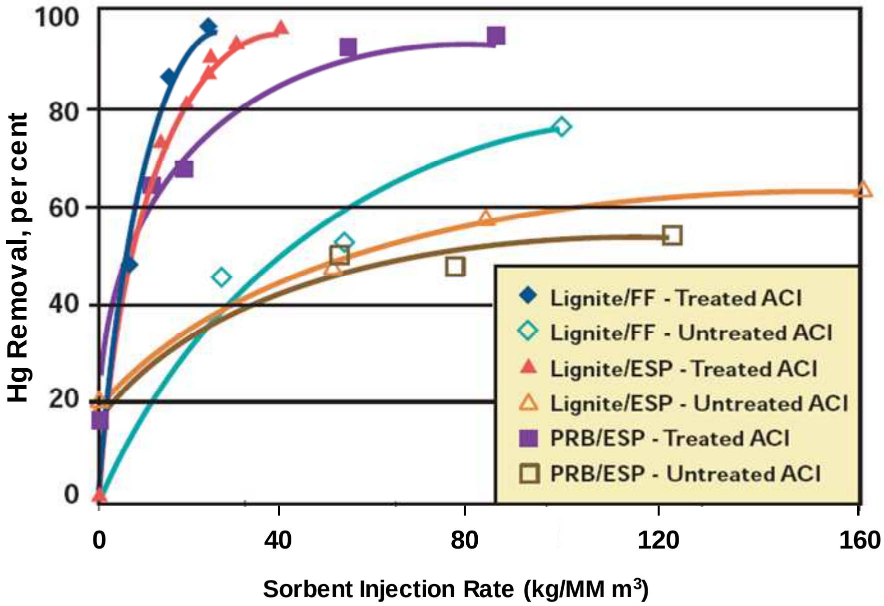

Figure 14. Comparison of untreated ACI and treated ACI performance for mercury emission control (Feeley et al., 2008).

ACI has been demonstrated in numerous applications and as of 2010 there are 135 ACI systems under contract in North America with combined capacity of over 60 GW (ICAC, 2010a). Despite the ACI being thoroughly demonstrated on multiple and diverse applications, there are some remaining potential issues that include:

- Impact on downstream PM collector and other balance-of-plant concerns
- Fly ash marketability for concrete manufacture
- Negative effect of SO_3 on mercury capture

With typical ACI rates, up to about a 4 per cent increase in total loading of the PM control device could be expected (Srivastava et al., 2006). Also, increasing rates of arcing were observed as a result of ACI (Dombrowski et al., 2004). Arcing can degrade ESP performance, and increase outlet PM concentrations, thus triggering permit issues for a plant. As far as the effect of ACI on operation of FF, FF's cleaning frequency would be expected to increase because of the increased particulate loading in the flue gas. This was confirmed during the field tests, which also noted increased opacity immediately after each FF cleaning cycle (ADA-ES, 2003). Another field test showed a slight increase in cleaning frequency as well as an increase in pressure drop across the FF proportional to the ACI rate (Pavlish et al., 2008).

As discussed previously, there is also concern about the impacts of PAC injection on the marketability of fly ash for use as a replacement for portland cement in concrete. A typical ACI system is located upstream of a PM control device and this leads to mixing of the sorbent and fly ash. This mixing can negatively affect utilization of fly ash in concrete production (as a substitute for portland cement). Concrete production is particularly sensitive to carbon content as well as the surface area of the carbon present in the fly ash. Carbon sorbents were developed that could potentially allow coal-fired power plants to continue marketing fly ash for concrete production (Nelson et al., 2006). Other non-carbon sorbents were also tested during Phase III (Kang et al., 2007). Generally, these sorbents were designed to preserve fly ash quality while still allowing sorbent injection rates capable of delivering up to about 85 per cent mercury removal. Recent field tests at a power station firing a blend of lignite and subbituminous coal were aimed at minimizing ACI fly ash impacts by utilizing the combination of addition of bromide to a boiler and injection of brominated PAC (Dombrowski et al., 2009). Other balance-of-plant considerations which are related to the use of halogenated sorbents include corrosion potential in the boiler and wet FGD, application of fly ash mixed with halogenated sorbents for concrete production, leaching from fly ash, and effect on mercury capture in and re-emission from wet FGD.

Field testing has shown that SO_3 in the flue gas, even at low concentrations, can interfere with the performance of ACI. It appears that SO_3 competes with mercury for adsorption sites on the sorbent surface thereby limiting its performance. This phenomenon may be particularly relevant to ACI applications at plants firing high-sulfur coal. One possible solution to address the SO_3 interference issue is combined injection of Hg sorbents and alkaline materials. Alkaline materials that were considered and tested for this application during Phase III include magnesium oxide (MgO) and sodium sesquicarbonate (trona). Based on the results of Phase III testing, trona injection is thought to enhance ACI performance to a greater degree than MgO (Feeley et al., 2008). However, the sodium content of trona may negatively affect fly ash use for concrete production.

8.2 TOXECON^{TM}

As described above, mixing of fly ash with spent sorbent may decrease the marketability of fly ash for concrete production. To remedy this potential issue, the toxic emission control process (TOXECON^{TM}) was developed and patented by EPRI. In TOXECON^{TM}, ACI is conducted downstream of the existing particulate control device and upstream of an added pulse-jet fabric filter (PJFF). The TOXECON^{TM}, shown schematically in Figure 15, makes it possible to separate the ash collected in the existing PM control device from the spent sorbent. The long-term feasibility of using TOXECON^{TM} to reduce mercury emissions was evaluated with lignite/subbituminous coal blends and lignite. TOXECON^{TM} configuration achieved greater than 90 per cent of total mercury removal at ACI rates of 58 kg/MMm^3 with treated ACI (brominated carbon) (Feeley et al., 2008).

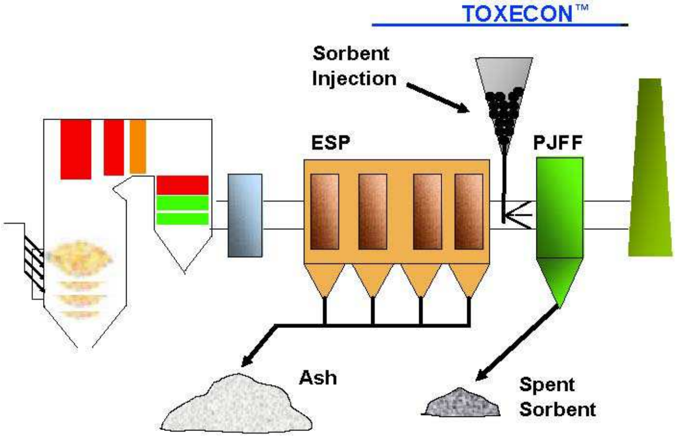

Figure 15. TOXECON^{TM} process configuration (US DOE, 2005).

While TOXECON^{TM} achieves separation of fly ash and sorbent, it can accomplish this at the expense of an additional PJFF. In the TOXECON II^{TM} technology ACI is conducted directly into the downstream collecting fields of a cold-side ESP (ESPc), as shown schematically in Figure 16. Because the majority of the fly ash mass is collected in the upstream collecting fields of an ESP, only a small portion of ash ends up intermixed with spent activated carbon sorbent. This spent activated carbon sorbent intermixed with small portions of fly ash is disposed of as hazardous waste. TOXECON II^{TM} may be applied with cold-side ESPs as well as with ESPh and selected sorbents. Full scale tests of TOXECON II^{TM} demonstrated 90 per cent total mercury removal with 260 kg/MMm^{3} of treated ACI in a facility burning subbituminous coal (Campbell, 2007).

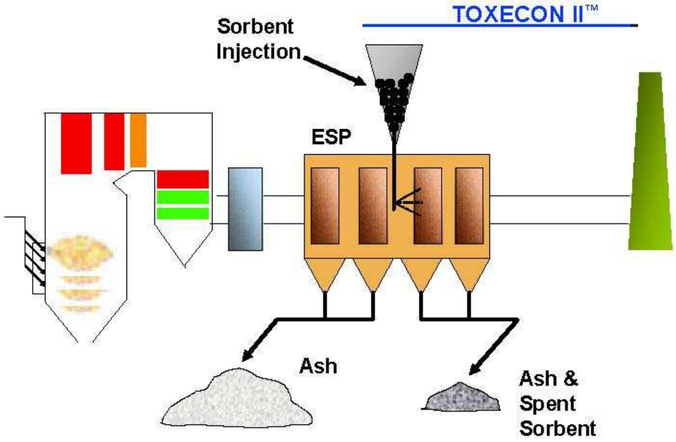

Figure 16. TOXECON II^{TM} process configuration (US DOE, 2005).

9 MULTIPOLLUTANT CONTROL

Multipollutant control technologies offer the cost advantage of delivering a system capable of controlling several pollutants simultaneously, including mercury. However, these technologies continue to be under development and may require more demonstration experience for commercial readiness.

Multipollutant control technologies promise the cost advantage of delivering a system capable of controlling several pollutants simultaneously rather than installing a separate system to address each pollutant separately. The multipollutant control technologies that are currently under development and promise to simultaneously control more than one pollutant from coal-burning power plants are shown in Table 6. These technologies will be discussed in more detail in this section.

Table 6. Multipollutant Technologies Currently Under Development

Technology	Status [a]	Mercury Emissions Reductions, per cent	Other Pollutant Control Capabilities, per cent	Potential Applicability
E-Beam	D/C	98 [b]	SO_2: up to 95 NO_X: 90	New and retrofit
EnviroScrub / Pahlman	B/P	Up to 67	SO_2: 99+ NO_X: 93-97	New and retrofit
Electrocatalytic oxidation	D	90	SO_2: 98 NO_X: 90	New and retrofit
LoTOx [c]	D/C	Up to 90	SO_2: 95 NO_X: 70-95	New and retrofit
PEESP [d]	B/P	Up to 98	SO_2: 90+ (with wet FGD)	New and retrofit
K-Fuel [e]	D/C	Up to 70	SO_2: Up to 30 NO_X: Up to 45	Mostly PRB or lignite

[a] Status: B = bench scale; P = pilot stage; C = commercial; D = demonstration

[b] Hg oxidation. Pilot scale studies by Kim et al., 2008

[c] LoTOx = Low Temperature Oxidation

[d] PEESP = Plasma-enhanced ESP

[e] Described in Coal Preparation section

E-Beam

The electron beam (E-Beam) process is placed downstream of the ESP and is capable of removing simultaneously SO_2 and NO_X. In addition, 98 per cent of elemental mercury can be oxidized (Kim et al., 2008). The process involves cooling of the flue gas, injection of ammonia, and then irradiation by high-energy electrons (Chmielewski et al., 1999). Flue gas is cooled to 60-70 °C in an evaporative spray cooler operated with a dry bottom. In addition, gaseous ammonia is injected into the flue gas either upstream or downstream of the spray cooler. The schematic of the process is shown in Figure 17.

The main component of the E-Beam process is a chamber where the flue gas is irradiated by a beam of high-energy electrons, while water is added to counteract the temperature rise. The irradiation also generates hydroxyl radicals and oxygen atoms, which oxidize SO_2, NO_X, and Hg. The oxidized SO_2 and NO_X species mix with water in the flue gas to form sulfuric acid and nitric acid, which are neutralized by the ammonia. The by-products of the E-Beam process are powder ammonium sulfate and ammonium sulfate-nitrate, which are collected downstream of the E-Beam chamber by an ESP or a FF, and can be used as fertilizer after processing into a granular product. The by-product particles are small and sticky and should be handled accordingly in either an ESP or FF.

The technology is in an early commercialization stage with at least five demonstration plants completed. Demonstration was conducted at Indianapolis Power & Light's Stout station in an 8 MW_e slipstream in 1986. Similarly, the process was tested in 1992 at Chubu Electric Power Company's Nishi-Nagoya plant in Nagoya, Japan on a slipstream (12,000 Nm^3/h) of 200 MW_e boiler. The emission reductions achieved were: 92 per cent SO_2 and 60 per cent NO_X. Another pilot (20,000 Nm^3/h) testing was conducted at Kawęczyn, Poland, where new engineering approaches to the accelerator design were tested and mercury removal was measured. Commercial application on a 90 MW_e unit in Szczecin, Poland has been in operation since 2002. As of early 2009, another commercial application at Sviloza, Bulgaria 200 MW_e plant has been in the design stage. Future market acceptance of this technology will depend on whether it is less expensive than the combination of individual SO_2 and NO_X controls for the levels of SO_2 and NO_X emissions required. Another major factor will be the possibility to sell the fertilizer by-product and the price that can be obtained for it. For example, Szczecin plant sells the fertilizer by-product to a fertilizer manufacturer. No information was given on the final fate of the mercury or the stability of the mercury in the final fertilizer by-product. A concept has been proposed for the fertilizer production companies to provide the ammonia needed to the power plant and receive the "upgraded" solid nitrogen granular fertilizer. Finally, improved design of electron beam accelerators to allow for the reduction of their size and to provide for increased long-term reliability would greatly enhance market acceptance of the process.

The E-Beam process is capable of achieving SO_2 removals of 95 per cent or greater and NO_X removals of about 90 per cent. High SO_2 removals require a minimal irradiation dose, generally much lower than the one dose necessary for NO_X removal. Once the minimum irradiation dose is achieved, the primary factors affecting SO_2 removal are flue gas temperature and ammonia stoichiometry. The dosage required for 90 per cent SO_2 removal is a minimum of 2 kGy. The removal of NO_X depends primarily on the dosage; temperature and SO_2 concentration are of secondary importance. Higher NO_X removals require higher radiation dosages. A dose of about 2-7 kGy is required to achieve 50 per cent NO_X removal, and 80 per cent NO_X removal requires at least 10 kGy (Frank and Hirano, 1988). Higher NO_X removals are obtained at higher temperatures, contrary to SO_2 removal. Higher SO_2 concentrations also improve NO_X removal, making the process better suited for high-sulfur applications.

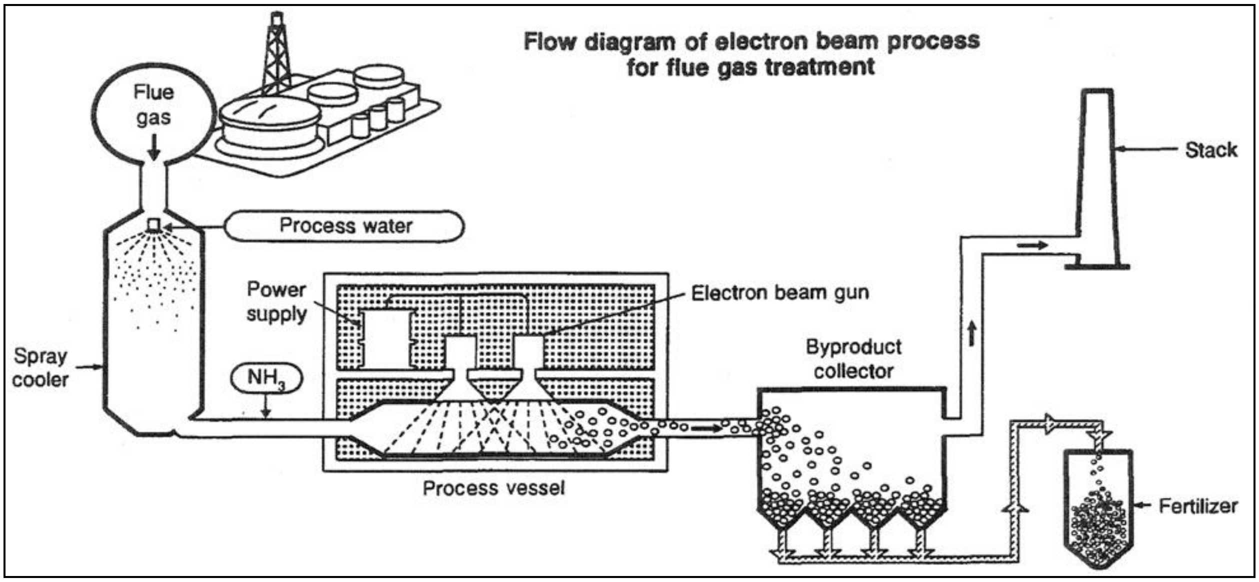

Figure 17. E-Beam schematic (from Frank and Markovic, 1994).

The energy requirement for E-Beam depends greatly on the NO_X reduction required. When significant NO_X reduction is not required, the auxiliary power for the E-Beam process may range from 2 to 3 per cent of the total plant output. For example, in a demonstration plant about 2 per cent of the plant energy was used for an SO_2 removal of 80 per cent and NO_X removal of 10 per cent. When NO_X reduction required was above 60 per cent, the auxiliary power requirement might reach 5 per cent (Izutsu and Okabe, 1997).

Enviroscrub/Pahlman

EnviroScrub's Pahlman process is a closed-loop dry sorbent system comprised of two discrete steps. One step involves capturing the target NO_X, SO_X, mercury, and PM using Pahlmanite dry mineral sorbent compounds (Interpol, 2001). The other step involves the regeneration of the spent or partially spent sorbent compounds for reuse and separation and isolation of useful byproducts such as nitrates and sulfates for use in fertilizers and industrial chemicals.

The Pahlmanite sorbents are low-density oxides of manganese (MnO_2) in the form of fine black powder (Power, 2002). The sorbent is injected in a reactor, which operates at temperatures between ambient and 160 °C. According to the supplier, different types of reactors are suitable including fluidized bed, baghouse, and cyclone. In addition to SO_2 and NO_X, mercury reacts with the sorbent, which promotes oxidation to HgO followed by sorption by MnO_2. The sorbent is regenerated in a wet chemical process and subsequently dried and returned to the contact reactor for reuse. The technology is in the pilot-scale stage and a trailer-mounted pilot plant is available, which has been tested at flue gas slipstreams of 0.45 Nm^3/s. Mercury removal results are only available from testing at Boswell Energy Center, which indicated mercury removal of up to 67 per cent (Modern Power Systems, 2002). The technology is still at an early development stage and requires further demonstration and techno-economic assessment to develop a more comprehensive picture of its cost-effectiveness.

Electro-catalytic Oxidation

The Electro-catalytic Oxidation (ECO) process shown in Figure 18 treats flue gas in three steps to achieve multipollutant removal (McLarnon and Jones, 2000). First, a majority of the ash in the flue gas stream is removed in a conventional ESP. Following the ESP, a gas converter (barrier discharge reactor) oxidizes the gaseous pollutants to higher oxides. For example, nitric oxide is reacted to form nitric acid (HNO_3) and NO_2, SO_2 is converted to sulfuric acid (H_2SO_4), and mercury is oxidized to mercuric oxide (HgO). Products of the oxidation process are then captured in a wet electrostatic precipitator (WESP) that also collects fine PM. Liquid effluent from the WESP may be treated to remove collected ash and then delivered to a system to produce concentrated sulfuric and nitric acids for sale. The ECO system is designed for retrofit into the last fields of an existing ESP. If the ESP does not have adequate space to fit the ECO system, some or all components could be built downstream of the ESP. In the latter case, the downtime of the plant is reduced, but additional space (footprint) is needed.

Oxidation of gaseous pollutants in the barrier discharge reactor is the key component of the ECO process. Oxidation is accomplished through generation of a non-thermal discharge or plasma. In a dielectric barrier discharge, energetic electrons are produced throughout the reactor without heating the gas stream to high temperatures, requiring considerably less energy than plasma discharges. Dielectric barrier discharges can be operated over a wide range of temperatures and pressures and have been widely used for commercial ozone (O_3) generation.

To form a barrier discharge, a dielectric insulating material is placed between two discharge electrodes. Typically, the material has a high dielectric strength and high dielectric constant (e.g., glass or ceramic) and covers one of the two electrodes. High voltage applied to the electrodes causes the gas in the gap to break down. Presence of the dielectric barrier prevents this breakdown from forming an arc with its resulting energy consumption. Instead, breakdown is in an array of thin filament current pulses, or "microdischarges."

Typical duration of a microdischarge is of the order of a few nanoseconds, and electron energies range from 1 to 10 electron volts (eV).

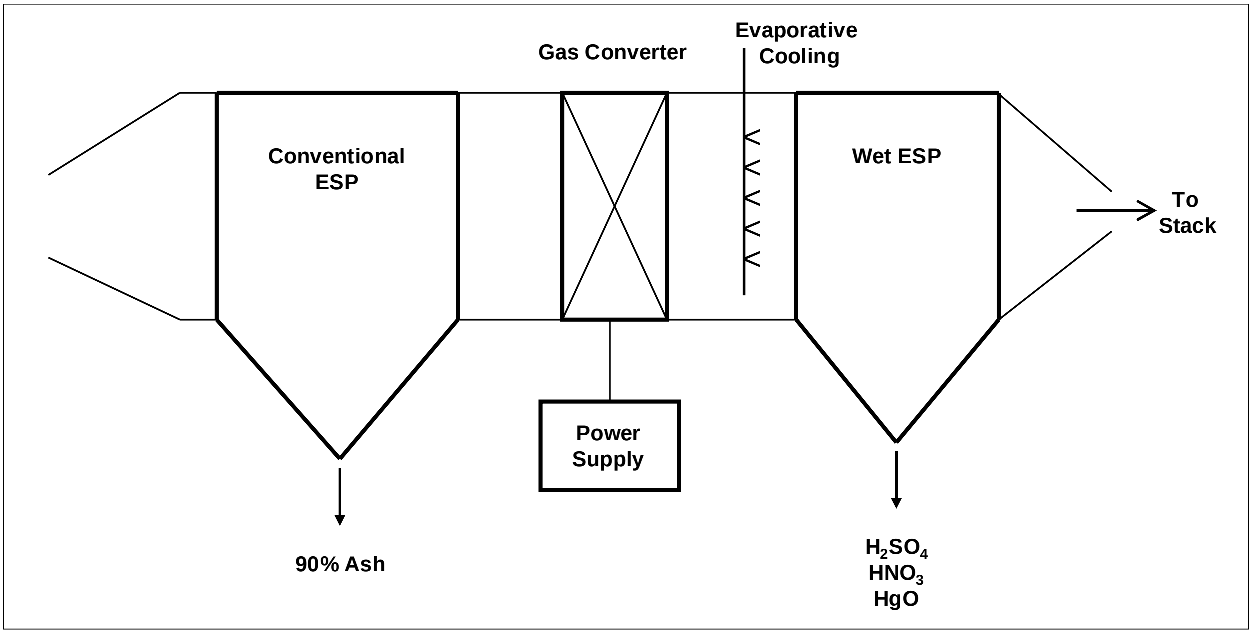

Figure 18. ECO process schematic.

The electron energies formed in the microdischarge are ideal for generating gas-phase radicals, such as hydroxyl and atomic oxygen (O) through collision of electrons with water and oxygen molecules present in the flue gas stream. The above reactions leading to radical formation and the subsequent oxidation reactions can occur at a low temperature of 65 to 150 °C. Aerosols formed by the oxidation reactions, including HgO, HNO_3, and H_2SO_4, exit the barrier discharge reactor in the flue gas stream. At this point the gas enters a WESP where collection of the aerosols, fine PM, and other compounds is accomplished.

The byproducts of the ECO process are raw sulfur and nitric and sulfuric acids, which can be used in the industry for fertilizer and gypsum production. The extent to which these byproducts would be actually used depends on economics (supply and demand of competing products) in the local market (around the power plant).

The technology was originally tested at laboratory scale, followed by pilot scale tests at a 156 MW unit, where a slipstream equivalent to approximately 1 MW was used. The technology achieved 98 per cent SO_2 reduction, 90 per cent NO_X reduction, and 90 per cent Hg reduction (McLarnon and Jones, 2000a). Following pilot scale tests, a commercial demonstration unit was installed and tested at a slipstream equivalent to 50 MW. The plant burned a variety of fuels including bituminous coal with 2 to 4 per cent sulfur. During the 50 MW demonstration, monitoring equipment consistently showed 98 per cent removal of SO_2, 90 per cent removal of NO_X, and 80 to 90 per cent removal of mercury (Boyle, 2004). These results were achieved with 337 ppm NO_X in the inlet of the ECO system, approximately 40 per cent higher than a similar installation with low-NO_X burners.

The ECO process requires a considerable amount of auxiliary power consumption, approximately 3 per cent of the plant's output is needed for the dielectric barrier discharge reactor. The power required for the dielectric barrier discharge reactor is largely determined by the amount of NO oxidation needed and the gas flow. To increase the amount of NO_X removed by the ECO process, it is necessary to increase reactor power. Therefore, to achieve a low outlet NO_X level while minimizing power demand, it is best to start with a low NO_X level from the boiler. As a result, one would typically use an ECO system in combination with low NO_X burners or other devices to minimize NO_X into the ECO reactor. Other power demands include fan power and another for auxiliary loads for the absorber and fertilizer plants.

The overall attractiveness of the process largely depends on local market conditions for ammonium fertilizer, which is a byproduct of ECO process. The ammonium sulfate and ammonium nitrate fertilizer are widely traded commodity chemicals, and their value will depend largely on market conditions at the time and transport costs (Staudt and Jozewicz, 2003).

LoTOx

Low temperature oxidation (LoTOx) involves injection of ozone (O_3) into the flue gas upstream of a wet FGD into a reactor (or even an exhaust duct) to oxidize NO_X to higher oxides of nitrogen such as N_2O_5, and mercury to HgO (Ferrell, 2000). Subsequently, these compounds are removed in a wet FGD because they are water-soluble. As Figure 19 shows, the LoTOx system consists of an integrated ozone generation/injection unit feeding into the LoTOx reactor.

Ozone is produced in-situ and on demand by passing oxygen through a conventional industrial ozone generation system, in response to the amount of NO_X present in the flue gas generated by the combustion or process source. Upon injection of the ozone in the flue gas (typically below 150 °C), oxidation occurs (Ferrell et al., 2002).

The selection of wet FGD type (lime, limestone, or ammonia) does not impact the performance of the LoTOx process because the solubility of N_2O_5 is significantly higher than that of SO_2. Theoretically, there is the potential for oxidation of SO_2 to SO_3; however, as proven in field testing, the reaction rates are very low compared to the predominant NO_X reactions.

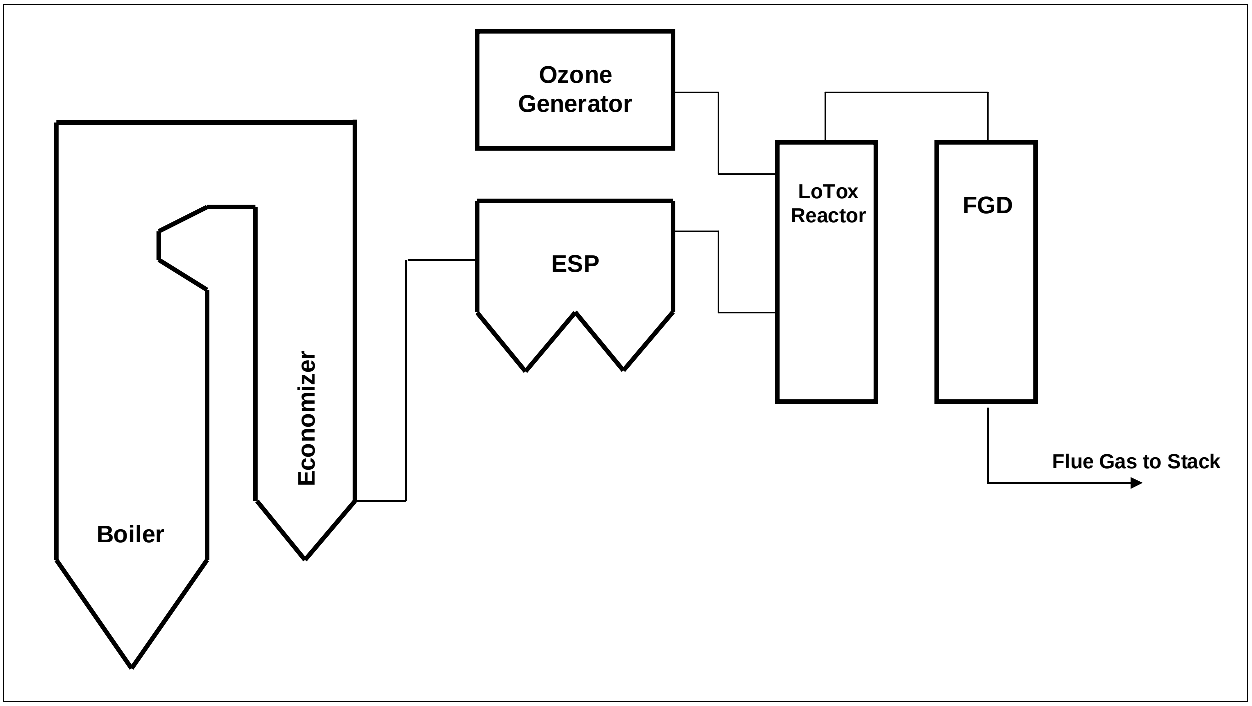

Figure 19. Schematic diagram of LoTOx system.

The technology has been demonstrated up to a scale of 25 MW at a boiler capable of burning high-sulfur bituminous coal (Goss, 2002). Installation of LoTOx by itself for control of NO_X emission is normally recommended when inlet NO_X is below about 130 g/GJ. Above 130 g/GJ, LoTOx is recommended as part of an integrated control approach operating in series with an alternate control process (e.g., a low NO_X burner) that is capable of only moderate NO_X removal efficiency. By combining the two technologies, users may be able to avoid installing an SCR system, which is expected to have higher capital investment and operating costs than LoTOx.

LoTOx is expected to enhance the mercury removal of FGD for all coals, especially subbituminous coal and lignite (Ellison, 2003), as shown below in Table 7.

According to the supplier, LoTOx enhances the SO_2 removal efficiency of the FGD by approximately 5 per cent (depending on the FGD design) and has no impact on SO_3 emissions. Auxiliary power requirements for a 500-MW plant are projected to be approximately 5.0 to 12.5 MW or 1 to 2.5 per cent of the gross power output. The technology has not been thoroughly demonstrated at a utility-scale plant.

Table 7. LoTOx Mercury Removal Data for Various Coal Types

Coal Type	Typical Hg^{2+} as per cent of Total Hg	Hg Removal with FGD Alone, per cent	Hg removal with LoTOx and FGD, per cent
Bituminous	70-85	76	94
Subbituminous	15-45	33	92
Lignite	10-30	19	91

PEESP

Plasma-Enhanced ESP (PEESP) technology oxidizes vapor phase Hg^0 into its oxidized form and then removes it within the WESP process. As discussed before, WESPs have demonstrated capability to remove multiple pollutants. For example, a pilot-scale hybrid ESP (addition of a wet ESP field in a dry ESP without PEESP) demonstrated the following removals: 95 per cent PM, 20 per cent SO_2, 35 per cent HCl, 45 per cent hydrogen fluoride (HF), and 50 per cent oxidized mercury (Altman et al., 2003). In another WESP, a 40 per cent removal of Hg^0 and greater than 70 per cent removal of PM and oxidized mercury were achieved. In addition, greater than 90 per cent removal of $PM_{2.5}$ (PM less than 2.5 microns in diameter) and SO_3 mist was accomplished with the ESP after an FGD (Montgomery et al., 2002).

PEESP involves injection of a reagent gas mixture, through a corona discharge needle that is attached to the central electrode within an electrostatic field. Injection into the area surrounding the sharp discharge point results in generation of hydroxyl radicals, ozone, and other reactive compounds. These react with Hg^0 vapor to promote formation of PM-bound mercury. These negatively charged particles are attracted to the positively charged collecting electrode. The PM-bound mercury and other absorbed pollutants are removed during the wash-down cycle of the WESP. PEESP can be incorporated in an existing WESP by modifying the central electrode to inject the reagent gas. The technology has been tested at bench scale and mercury removal efficiencies of up to 83 per cent were achieved (Montgomery et al., 2003). The supplier projects that up to 90 per cent total Hg removal can be achieved at pilot and full scale.

O&M impacts are not known but are expected to be minimal since the PEESP technology is a passive device retrofitted within a WESP. The PEESP technology seeks to enhance oxidation of Hg^0 to improve total mercury removal. The technology is still at an early development stage and requires further demonstration and techno-economic assessment. Additionally, further investigation is needed regarding the chemistry within the WESP to keep the oxidized mercury from degassing back to Hg^0 (re-emission).

10 DECISION TREE

The Decision Tree is a concept of a selection process to assist the user in a preliminary selection of an optimum mercury control strategy by analyzing other pollutants (SO_2, NO_X, PM) control equipment configuration as well as its operation.

When the management of a coal-fired electricity-generating plant considers mercury control, its operators and technical staff need to analyze numerous factors that will determine which technology or combination of technologies is best suited to address the needs of the plant. As discussed previously, mercury speciation occurring through gas cleaning processes has a decisive effect on a plant's mercury emissions and the performance of any mercury-specific control technology. Generally, Hg^P is almost entirely collected in FFs and reasonably well collected in ESPs. Hg^{2+} is soluble and, given favorable scrubber chemistry, may be efficiently removed in wet FGD scrubbers. Hg^0 is insoluble and mostly passes through air pollution control equipment that may be installed in a power plant. Thus, the mercury emission control strategy should be to first minimize the amount of Hg^0 present in the flue gas by converting it to either Hg^{2+} or Hg^P. Any remaining Hg^0 may be then removed with mercury-specific control equipment, most notably with ACI. Therefore, for design purposes, it is important to understand the amount of Hg^0 escaping the existing air pollution control equipment configuration in order to determine the need for and to be able to properly size the ACI system.

Proprietary models exist that can predict mercury chemistry and fate in the available residence time (e.g., Niksa and Fujiwara, 2009). There are also numerical models that have been published and thus are not proprietary (e.g., the published model that provides the ability to assess configuration and operational parameters' effects of ESPs on mercury capture) (Clack, 2009). These fundamental models are capable of accounting for differences in power plant operations and air pollution equipment configurations, as well as their operating conditions, and should be used to improve the understanding of mercury emissions from a power plant. However, a preliminary selection of an optimum mercury control strategy may be arrived at by analyzing other pollutants control equipment configuration as well as its operation. Optimum mercury control strategy is defined as maximization of co-benefit mercury removal by optimizing operation of other pollutants control technologies. If desired levels of mercury controls are not achieved, augmentation of mercury removal by ACI is possible. This concept of a selection process is here called the Decision Tree and is shown schematically in Figure 20.

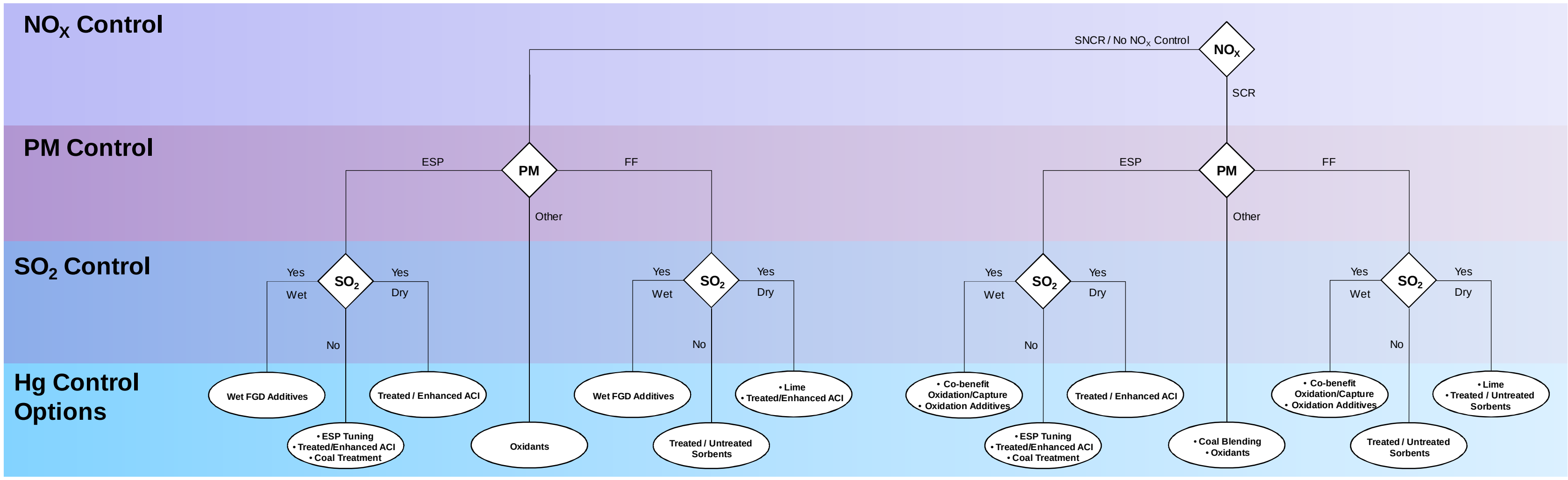

Figure 20. Decision Tree for enhancing mercury control options.

Note: The Decision Tree represents different configurations of control equipment typically found at power plants to control PM, NO_X, and SO_2. Each pathway describes a particular technology configuration, with decision points involving each of these pollutants, and leads to a suggested optimization approach. Based on each level of mercury control desired, the addition of mercury-specific controls (e.g., ACI) should be considered.

The Decision Tree is a concept of a selection process to assist the user in a preliminary selection of an optimum mercury control strategy by analyzing other pollutants (SO_2, NO_X, PM) control equipment configuration as well as its operation. Final selection can be confirmed with one of a number of complex predictive models. Based on each level of mercury control desired, the addition of mercury-specific controls (e.g., ACI) should be considered. In addition to analyzing control technologies for the above pollutants it is important to consider other practices and technologies as well. The Decision Tree does not consider GHG or HAPs, as coal-fired power plants around the world do not currently address these emissions commercially. It does not either analyze pre-combustion measures such as coal blending and coal washing.

As can be seen in Figure 20, the Decision Tree leads the user through different configurations of control equipment typically found at power plants to control PM, NO_X, and SO_2. Each pathway describes a particular technology configuration, with decision points involving each of these pollutants, and leads to a suggested optimization approach. The decision sequence following this initial decision point is illustrated in Figures 20 and 21 for plants with SCR and for plants without SCR, respectively. For the example shown in Figure 21, the power plant would have SCR, ESP, and wet FGD installed (indicated by the green line). In this case, the optimum strategy would be to maximize the SCR and wet FGD co-benefit mercury removal as discussed in previous chapters. For a slightly different case of a plant with SCR and ESP but without wet FGD, the control strategy would become dependent on ESP improvements and coal blending, and depending on the desired level of mercury control, may be augmented by ACI.

For plants without SCR control technology installed, the Decision Tree sequence illustrated in Figure 22 would apply. As seen in Figure 22, the plant would have only PM control equipment installed for its emission control. If the PM control equipment was an ESP (left branch of pink line in Figure 22), the control strategy would become dependent on ESP improvements such as coal blending to increase the amount of Hg^{2+}. Depending on the Hg^0 and Hg^{2+} ratio in the flue gas, ACI could also be needed. For the case of PM control installed other than ESP or FF, such as a wet PM scrubber, the optimum strategy would involve altering the chemistry of the wet PM scrubber’s hold tank (e.g., by adding oxidants to increase the amount of Hg^{2+}). This is indicated by the other branch of the pink line in Figure 22.

Table 8 below shows Decision Tree optimization for selected existing control equipment configurations. Optimization examples shown in Table 8 have been derived following the Decision Tree schematics shown in Figures 20, 21, and 22.

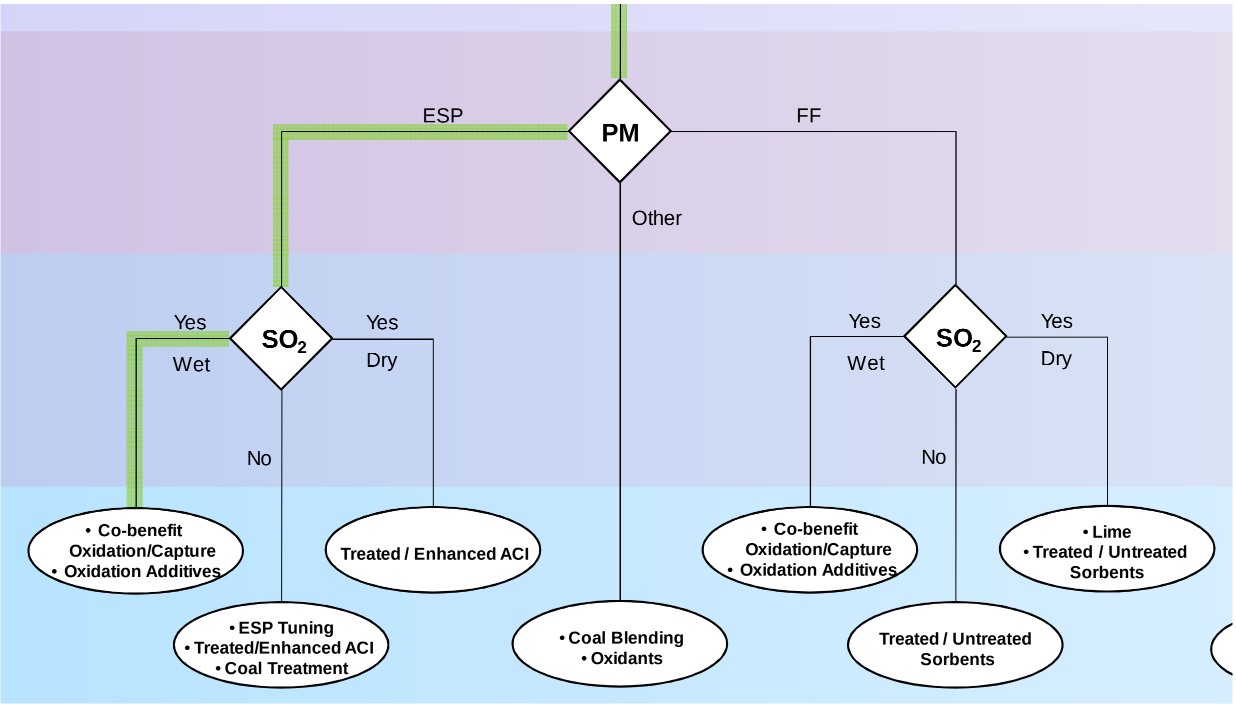

Figure 21. Decision Tree sequence for example plants with SCR.

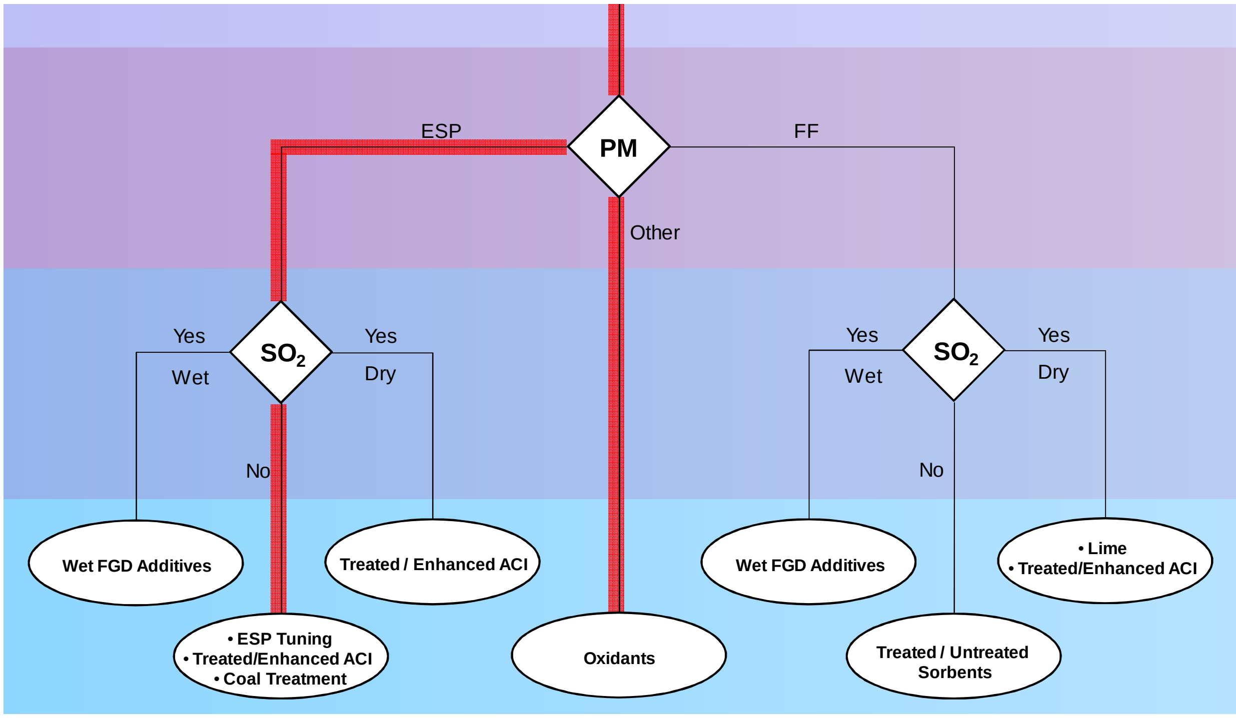

Figure 22 Decision Tree sequence for example plants without SCR.

Table 8. Decision Tree Optimization for Selected Control Equipment Configuration

Existing Control Equipment Configuration	Qualitative Mercury Capture	Decision Tree Optimization
ESP only	Good capture of Hg^P	- Fine tune ESP operation - Increase the amount of PM-bound mercury in flue gas (e.g., by coal blending or addition of oxidants) - May require ACI
FF only	Good capture of all mercury forms	- Optimize bag cleaning cycle - May require ACI
ESP + Wet FGD	Poor to excellent overall capture depending on coal type	- Optimize the amount of Hg^{2+} in flue gas (e.g., by coal blending or addition of oxidants) - Prevent/control Hg^0 re-emission
SDA + FF	Good to excellent capture of all mercury forms	- Optimize bag cleaning cycle - ACI may be required for low rank coal
FF + Wet FGD	Good to excellent capture	- Optimize Hg^0 oxidation by FF - Prevent/control Hg^0 re-emission - Optimize SO_2 control
SCR + ESP	Good capture of Hg^P	- Optimize the amount of Hg^{2+} in flue gas (e.g., by coal blending or addition of oxidants) - Fine tune ESP operation
SCR + ESP + Wet FGD	Good capture of Hg^P Poor to excellent overall capture depending on coal type	- Optimize the amount of Hg^{2+} in flue gas (e.g., by coal blending or addition of oxidants) - Prevent/control Hg^0 re-emission - Fine tune ESP operation - Optimize SO_2 control
SCR + SDA + FF	Excellent overall capture	- Optimize the amount of Hg^{2+} in flue gas (e.g., by coal blending) - Optimize bag cleaning cycle - ACI may be required for low rank coal

Note: Mercury-specific controls (ACI) are suggested as an option for several configurations in this table. In addition, ACI should be considered as an option whenever optimization approaches do not achieve desired mercury emission reductions.

11 POST-CONTROL ISSUES

- **Mercury captured in CCRs appears to remain bound to fly ash particles and FGD gypsum for range of conditions materials encounter during land disposal.**
- **However, mercury appears to leach at levels of potential concern for wet FGD sludge and as managed scrubber sludge. Therefore, these types of CCRs potentially may present environmental risks under some disposal or use conditions.**

Depending on the configuration of the existing air pollution control equipment and subsequent Decision Tree optimization of mercury capture, mercury may be transferred from gas phase (flue gas) to solid phase (e.g., fly ash, synthetic gypsum) or into a liquid or solid/liquid phase (e.g., wet FGD sludge, fixated FGD sludge). These different media are residues from coal-fired power plant operation and are generated in various processes. Fly ash is the product of coal combustion and is collected by the ESP or FF. Gypsum is the byproduct of LSFO. In LSFO, nearly all of the byproduct is calcium sulfate dihydrate ($CaSO_4 \bullet H_2O$), also called synthetic gypsum. The resulting synthetic gypsum can be disposed in a landfill, used for wallboard production, or as feedstock in making cement and concrete. Wet FGD sludge is collected from natural or inhibited oxidation wet FGD. In an inhibited oxidation system, nearly all of the byproduct is calcium sulfite hemihydrate ($CaSO_3 \bullet ½H_2O$). In a natural oxidation system, the byproduct is a mixture of $CaSO_3 \bullet ½H_2O$ and $CaSO_4 \bullet H_2O$. FGD sludge is typically blended with fly ash and lime prior to land disposal. Collectively, these materials are often called coal combustion residues (CCR). Because of the increased content of mercury in the CCRs as the result of mercury capture, there is a concern about the potential for mercury release (mercury leaching) and cross media transfers of mercury and other constituents of potential concern (COPC) resulting from land disposal or use of CCRs.

The potential mercury release route for CCRs that is of most concern is leaching to groundwater because it could negatively affect drinking water quality. Also of concern is the release of mercury to surface waters and potential for its bioaccumulation. Because of the nature of these concerns, leaching study results are typically compared to water quality standards. For example, in the case of the United States, these standards are maximum concentration limit (MCL) for drinking water, toxicity characteristics or threshold for hazardous waste determination, and drinking water equivalent level for determination of non-carcinogenic endpoints of toxicity. It should be noted that leaching results represent an estimate of the maximum release potential under conditions tested. However, they do not describe the amount of mercury that may reach an aquifer.

US EPA conducts the broadest known study on the subject and initial results have been reported (US EPA, 2006; US EPA, 2008; US EPA 2009). The study, conducted as part of US EPA's "Mercury Roadmap", has investigated 73 CCRs (thirty-four fly ashes, twenty gypsum samples, seven scrubber sludge samples, eight scrubber sludge samples blended with fly ash and lime, and four wastewater treatment filter cakes). The primary focus of the study was on mercury, but arsenic, selenium, and other COPCs were also evaluated. CCR samples used in the study were representative of likely configurations of wet FGD and of coal ranks typically used at power plants throughout the world. Each CCR sample was evaluated to determine alkalinity, solubility and release as a function of pH according to method SR002.1 (Kosson et al., 2002). The leach testing protocols used have been drafted for inclusion into US EPA's SW846 as official US EPA methods for more routine use. The analysis that was conducted consisted of 11 parallel extractions of particle size reduced material, at different pH values ranging from pH 2-13, and at a liquid-to-solid ratio of 10 ml extractant/g dry sample. In reporting results from this research, only values for pH ranging from 5.4 to 12.4 were used based on available field data for CCR landfills and surface impoundments (US EPA, 2009).

The study concluded that captured mercury does not appear to leach above the MCL for the fly ashes and FGD gypsum samples that were evaluated. However, there may be potential concern for Hg leaching for the scrubber sludge or as managed sludge (scrubber sludge mixed with fly ash and lime). Therefore, these types of CCRs may potentially present environmental risks under some disposal or use conditions. It follows, that management conditions (i.e., CCRs managed in lined facilities) could be used to mitigate the mercury leaching potential from wet FGD sludge and fixated wet FGD sludge. Leaching of heavy metals was routinely observed. Generally, leaching of individual constituents varied over several orders of magnitude, depending on the conditions of the management scenario as expressed by pH and liquid-to-solid ratio. Leaching concentrations in the study did not correlate with total metal content, and total content was not a good indicator of leaching (US EPA, 2009).

12 EXAMPLES OF CONTROL COSTS

Examples of mercury emission control costs are given in this chapter. These examples of costs are given in US dollars (US$) and were derived from activities carried out in the United States. When interpreting these costs, caution should be exercised when arriving at costs in different countries. One of the approaches that may be used when comparing costs in different countries is the so-called purchasing power parity (PPP) approach, which utilizes the long-term equilibrium exchange rate of currencies in two countries to equalize their purchasing power.

Approach	**Capital Cost**	**Incremental O&M Cost**
Energy efficiency improvement	Moderate	Low
Coal treatment - Pre-combustion	High	Moderate
Coal blending	Low	Low
Coal additives	Low	Low
ESP upgrade	Low	Low
Co-benefit maximization - FGD	Low	Low
Co-benefit maximization - SCR	Moderate	Moderate
ACI	Low	Moderate to High
TOXECON	High	Moderate

As discussed before, mercury emission control can be accomplished as a co-benefit removal by the equipment already in place that might have been installed for a different purpose other than controlling mercury emissions. Alternatively, mercury control may be accomplished by mercury-specific technology such as ACI. The extent to which control approaches rely on co-benefit removal versus the application of mercury-specific control technology depends on the overall effectiveness of co-control and the level of mercury emission control desired. Evaluating the cost of mercury-specific controls such as ACI is more straightforward to assign the costs to because the entire cost is dedicated to accomplishing mercury removal. Defining the cost of mercury removal accomplished as a co-benefit is more complex because cost apportionment needs to be considered between mercury control cost and the cost of controlling other pollutants such as SO_2 or NO_X (Sloss, 2008). The cost of removing mercury with ACI will be discussed in more detail below.

There are three cost components resulting from the application of mercury-specific emission control technology such as ACI: capital cost, variable operating cost, and fixed operating cost. For ACI, the variable cost is estimated to be a major component (US EPA, 2005). The major components of the variable cost are sorbent cost and disposal cost. Activated carbon delivered price is typically between 0.9 and 2.1 US$/kg and disposal costs may be estimated at US$19/metric ton (US DOE, 2006). In addition, lost revenue from fly ash sales due to activated carbon contamination could be estimated at US$20/metric ton. The estimate of lost revenue due to activated carbon contamination is a conservative scenario since "concrete-friendly" activated carbons have been developed (Nelson et al., 2006). Fixed operating costs for ACI systems are relatively low because of the relatively low level of complexity of ACI operation compared to other air pollution control equipment such as wet FGD or SCR. Detailed economic analysis of ACI systems intended for operation in the United States has been conducted by the US DOE (Jones et al., 2007).

The capital cost of an ACI system will be illustrated by an example for a 360 MW unit burning subbituminous coal (low sulfur, low chlorine, high alkalinity) and with SDA/FF existing configuration providing 37 per cent co-benefit mercury capture (US DOE, 2006). For this unit, capital cost (defined as total capital requirement [TCR]) was determined to be US$3.6/kW and was arrived at as shown below in Table 9.

Table 9. Example of Capital Cost for ACI on a 360 MW Unit

Cost Component	Amount, US$
Equipment Cost	711,116
Site Integration (materials and labor),	51,884
Taxes (6%)	45,780
Installation	124,000
General Facilities (10%)	93,278
Engineering Fees (10%)	93,278
Project Contingency (15%)	139,917
Process Contingency (5%)	46,639
Total Capital Requirement (TCR)	1,305,892
TCR, $/kW	3.6

Source: US DOE, 2006

Total O&M costs for the example presented in Table 9 were estimated at US$600,000/year for 90 per cent removal of mercury. An additional cost of operating an ACI system was the byproduct impact or the value of disposal costs plus the unrealized revenue from the sale of ash (due to activated carbon contamination of fly ash). The byproduct impact for this 360 MW unit was estimated at US$1,430,000/year.

For comparison, early US EPA estimates indicated that the capital cost for a baseline ACI system would be in the range of about US$5/kW (US EPA, 2003). However, these capital costs would be expected to increase substantially if the addition of a PJFF or other major PM control device was necessary. For example, US DOE estimated that a 500 MW plant operating at 80 per cent capacity factor and aiming at 70 per cent reduction in mercury emissions would expend US$984,000 in capital costs and US$3.4 million in annual O&M costs (GAO, 2005). If this plant were to install a supplemental PJFF option, capital costs would increase to about US$28.3 million, while annual O&M costs would decrease to about US$2.6 million (GAO, 2005). On the average, PJFF capital cost was estimated at US$15.8 million per boiler (US$ 12.7 to 24.5 million range) (GAO, 2009). As mentioned before, the cost of mercury removal depends on the availability of equipment installed to reduce emissions of other pollutants. On the average, the cost of an ACI system (including monitoring) was US$3.6 million per boiler and it ranged from US$1.2 to 6.2 million (GAO, 2009).

The POG assumes that decisions to install FGD and SCR are driven by the desire to control SO_2 and NO_X emissions. While they both also provide co-benefit mercury emission reduction, no effort has been made to allocate these costs summarized below between mercury and these other pollutants The average cost (in 2008 US$) of wet FGD was reported to be US$86.4 per boiler (US$32.6 to 137.1 million range) while the average cost of SCR was US$66.1 million per boiler (US$12.7 to 127.1 million range) (GAO, 2009).

The costs of FGD and SCR per MW of capacity decline with increasing plant size according to the economy of scale. For example, the costs of FGD are (in 2006 US$), US$301/kW for a 300 MW plant, US$230/kW for a 500 MW plant, and US$190/kW for a 700 MW plant. The costs of SCR are US$124/kW for a 300 MW plant, US$108/kW for a 500 MW plant, and US$98/kW for a 700 MW plant. These cost data were derived for FGD units assumed to remove 95 per cent of the SO_2 and SCR units assumed to remove 90 per cent of the NO_X (IER, 2009).

In general, the cost of optimizing co-benefit mercury control is difficult to assess since it is dependent on multiple variables such as coal origin and quality, the extent of refurbishment required for the existing PM controls (in case of ESP), or site-specific operating regime of wet FGD. Because of these complications, no attempt was made to predict actual cost for different approaches to controlling mercury. Instead, relative costs were arrived at for approaches discussed in the POG. These relative costs, shown below in Table 10, should only be treated as trend indications and should not be construed as universally applicable guidelines to the selection of cost-effective approaches to mercury emission control from coal-fired electricity generating power plants that might be located throughout different countries. Locally prevalent economic conditions should always be considered when selecting a mercury control option.

Table 10. Relative Cost of Mercury Control

Approach	Capital Cost	Incremental O&M Cost	Comments
Energy efficiency improvement	Moderate	Low	Relatively low mercury capture but highly effective for improved boiler operation. Capital intensive options (e.g., re-powering) excluded.
Coal treatment – Pre-combustion	High	Moderate	Washing less expensive than chemical treatment.
Coal blending	Low	Low	May require adjustment and/or refurbishment of pulverizers.
Coal additives	Low	Low	No added fixed O&M cost, often proprietary.
ESP upgrade	Low	Low	No added fixed O&M cost once modifications done.
Co-benefit maximization - FGD	Low	Low	Potential for re-emission of mercury should be mitigated.
Co-benefit maximization - SCR	Moderate	Moderate	May require different catalyst, may require coal blending.
ACI	Low	Moderate to High [a]	Preservation of ash quality an issue.
TOXECON	High	Moderate	Partial offset of cost by the retained value of fly ash.

[a] Costs may vary with plant and reduction requirements.

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