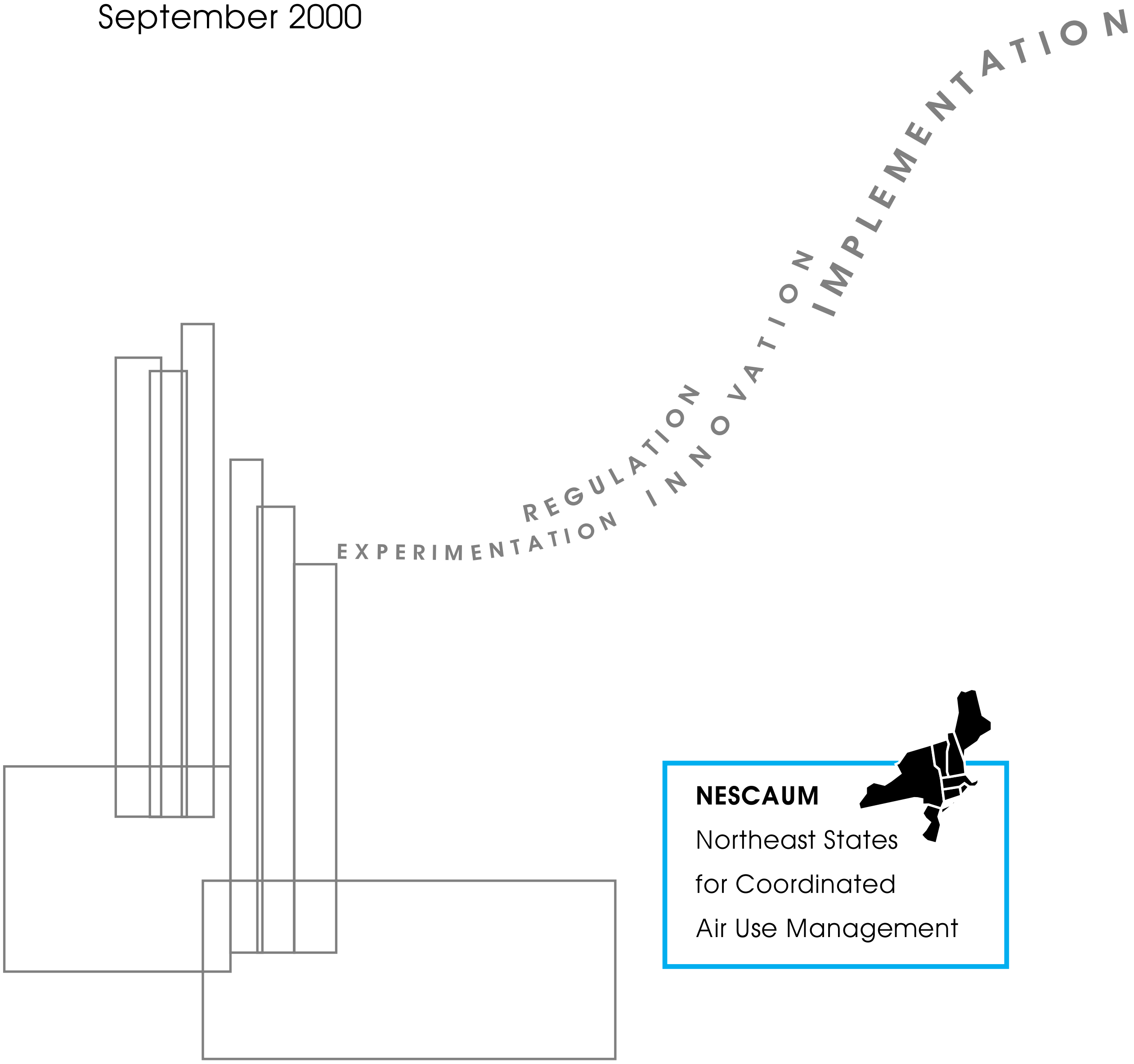
Environmental Regulation and Technology Innovation:

Controlling Mercury Emissions from Coal-Fired Boilers

September 2000



NESCAUM
Northeast States
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Air Use Management

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Executive Summary

Introduction

Mercury is a toxic heavy metal that persists in the environment once it is released into the atmosphere. Concern about high levels of mercury deposition and subsequent bioaccumulation in aquatic ecosystems – a phenomenon that can pose health risks for humans and animals that eat mercury-contaminated fish – has emerged as an important public health and environmental issue in recent years. Increasing attention is being focused on the need to control unregulated mercury emissions from coal-fired power plants, which remain the largest source of this pollutant in the United States.

In a 1998 report to Congress on hazardous air pollutants associated with electricity production, the U.S. Environmental Protection Agency (EPA) identified mercury as the hazardous pollutant of greatest potential concern for public health. EPA is now required to reach a determination by December 15, 2000, as to whether it is appropriate and necessary to regulate hazardous air pollutants (including mercury) from power plants. If the Agency makes an affirmative determination it will be legally required to propose regulations by December 2003 and to issue final regulations by December 2004, with full implementation to occur by the end of 2007. Meanwhile, several Northeastern and Midwestern states are pursuing or considering independent regulatory action to reduce mercury emissions, including emissions from coal-fired boilers, at the state and regional level.

As the question of regulating power plant mercury emissions becomes more urgent, a number of arguments are being raised in opposition. Typically, these emphasize remaining scientific uncertainty about the specific harms caused by power plant emissions and the need for more research.[1] In addition, concerns are being raised about the commercial availability of cost-effective mercury control technologies. Such objections have a familiar ring: they have been raised in one form or another at nearly every major decision point in the history of U.S. environmental regulation.

This report addresses the feasibility and appropriateness of the near-term regulation of power plant mercury emissions given the current status of mercury control technologies. To shed light on this question, the report reviews three case studies in control technology development: the first involving automobile emissions, the second two concerning power plant emissions of oxides of nitrogen (NOx) and sulfur dioxide (SO_2). Together, the case studies suggest that there is a strong link between technological innovation and the existence, timing, and stringency of regulatory drivers. Where strong regulatory drivers exist, substantial technological improvements and steady reductions in control costs almost always follow. This dynamic has occurred even when control options were limited or largely untested at the time regulations were introduced.

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[1] The report does not address the relationship between power plant emissions and human health or other impacts. However, a few observations are worth noting in this regard. First, human activity almost certainly accounts for a substantial increase in global atmospheric mercury concentrations since pre-industrial times. Second, coal combustion is the largest identified source of airborne mercury emissions in the U.S. Third, because mercury is persistent and subject to long-range airborne transport, power plant emissions are likely to contribute to cumulative long-term impacts as well as to elevated downwind deposition. Finally, measured concentrations of methylmercury in some species of fish have already reached unhealthful levels in many parts of the U.S., especially in the Northeast, southern Florida, and the upper Great Lakes. This situation has led to the stringent regulation of other major mercury sources, the issuance of fish consumption advisories in 41 states, and a number of related state and regional initiatives.

At present, several options for controlling mercury emissions from coal-fired boilers are well understood and have been extensively tested, if not yet commercialized. These include pollution prevention options such as fuel switching, co-firing, and coal cleaning, as well as at least two promising add-on control options: activated carbon injection and enhanced wet-scrubbing. Viewed in the context of the case studies summarized below, the present status of mercury controls clearly does not preclude near-term regulation. On the contrary, past experience strongly suggests that the impetus of regulatory drivers is all that is lacking to advance these technologies to full-scale commercial application.

Case Study 1: Automobiles

The evolution of motor vehicle emissions controls over the past three decades provides a striking example of successful technological response to regulatory drivers. Compared to the average car produced in 1965, a new passenger car manufactured in 2005 will have to achieve emission reductions of over 98 percent for volatile organic compounds (VOCs), almost 98 percent for carbon monoxide (CO) and 95 percent for NOx. This will be accomplished through the use of sophisticated controls that have been designed to perform effectively and reliably in literally millions of installations and under a wide range of operating conditions.

This impressive achievement is the direct result of regulation and legislation at the federal level and by the State of California, which to this day retains a significant leadership role as the only state empowered to design and set its own vehicle emissions standards. The automobile example provides a useful introduction not only because the link between regulation and technology was so strong in this case, but also because it nicely illustrates the many non-technological influences that shape policy choices on the one hand and industry responses on the other. As with the more recent issues, such as mercury emissions from coal-fired boilers, key regulatory decisions concerning automobiles were preceded by years of scientific and political debate. Such debates, together with strategic efforts by industry to promote cheaper but less effective controls to the exclusion of more promising technologies, delayed meaningful regulatory action for two full decades after scientists in the early 1950s first identified the automobile as a substantial contributor to photochemical smog.

Following adoption of the landmark 1970 federal Clean Air Act Amendments, technology development accelerated markedly. This and subsequent rounds of regulation led to a host of technological responses, including the development of sophisticated add-on control devices (such as catalytic converters), considerable refinement of the internal combustion process (such as fuel injection and electronic controls), and the introduction of cleaner fuels (such as reformulated gasoline).[2] In several instances, major advances -- including the catalytic converter, early evaporative controls, and fuel injection -- existed in concept and even in practice long before they were adapted and widely implemented by U.S. manufacturers.

Today, the success achieved in reducing motor vehicle emissions stands out as one of the great technological accomplishments of the last half-century of environmental regulation. The path

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[2] The 1970 Amendments were followed by similarly groundbreaking Amendments in 1990, and by several noteworthy California initiatives. A number of factors made substantial regulatory movement possible in each case. Both 1970 and 1990 were years when public interest in the environment reached a high point nationally. In California, the obvious severity of pollution problems in the Los Angeles basin, together with federal air quality attainment deadlines, created the political momentum necessary to sustain tough mandates.

to this success was not without setbacks: early emissions controls tended to deteriorate quickly and were sometimes unpopular with consumers; occasionally, proposed mandates were later perceived as too ambitious and were themselves relaxed or delayed (usually following concerted political efforts by the auto industry). Even in these instances, the strength of the initial regulatory signal served to propel technology forward, and progress toward underlying policy goals continued. Thus, the aggressive federal standards proposed in 1970 were achieved and eventually exceeded in the 1980s and 1990s, and even earlier in California. More recently, California's ambitious sales requirement for zero emission vehicles (ZEVs), though it has yet to be implemented, has prompted major research and development investments that are helping to open the door to a new generation of alternatives to the internal combustion engine, including hybrid-electric, fuel cell, and battery-electric technologies.

Case Study 2: Power Plant NOx

If the history of automobile emissions controls serves to illustrate technological response to strong regulatory drivers, the history of power plant NOx controls provides an example of weak technological response to initially weak drivers in the U.S., followed by a period of rapid technology diffusion in response to changing regulatory signals. Unlike motor vehicle emissions, which are directly regulated by the federal government and California, power plant NOx emissions have been regulated primarily via states' obligation to develop implementation plans for attaining ambient air quality standards for ozone. Because the role of power plant NOx emissions in ozone formation and transport was for several decades under-appreciated,[3] this mechanism has only in the last few years begun to provide strong regulatory drivers for advanced NOx control technologies.

Limited NOx control requirements have existed since the 1970s. These were initially applied only to new or substantially modified facilities under the New Source Performance Standards (NSPS) program, and to facilities that might otherwise cause deterioration in areas with clean air or affect visibility in pristine wilderness areas. The standards applied under these programs, moreover, were explicitly technology following; that is, they were set with reference to control technologies that already had been "adequately demonstrated." For most of the last three decades, regulatory interpretation of this term effectively restricted consideration of available control options to combustion modifications such as low-NOx burners (LNB) and overfire air (OFA) systems. In their early applications, these technologies typically achieved emissions reductions on the order of 30 to 40 percent. Many years of in-use experience subsequently improved the control effectiveness of modern LNB/OFA systems to well over 40 percent -- enough, in many cases, to comply with the more stringent requirements first imposed on existing power plants by the Clean Air Act Amendments of 1990.[4]

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[3] California recognized the importance of controlling NOx emissions for purposes of ozone attainment as early as the 1970s, more than a decade before federal regulators came to the same conclusion (in addition, Los Angeles was the only area to violate the federal ambient NO_2 standard). Hence, the state evinced an early interest in advanced NOx controls and promoted a number of adaptations geared to its largely gas and oil-fired boiler population.

[4] The 1990 Amendments introduced phased emissions limits for certain classes of boilers for purposes of acid rain mitigation, and mandated the installation of Reasonably Available Control Technology (RACT) for existing facilities in ozone non-attainment areas.

Only recently, with the adoption of the Ozone Transport Commission's 1994 NOx Memorandum of Understanding and EPA's subsequent proposal to require similar NOx reductions across 22 eastern states, are more substantial emissions cuts -- on the order of 85 percent from uncontrolled levels -- being contemplated for a large number of existing power plants.[5] These reductions will necessitate the use of control technologies beyond LNB/OFA, including advanced post-combustion technologies such as selective catalytic reduction (SCR). Nevertheless, even these most recent requirements cannot be said to be "technology forcing.” SCR technology has already been extensively demonstrated and applied overseas, notably in Japan and Germany, two countries that imposed stringent emission limits on major NOx sources much earlier than the U.S. Full-scale applications of SCR were taking place in Japan as early as the late 1970s and Germany began aggressively deploying this and other control options by the mid 1980s in response to widespread concerns about forest die-off due to acidification. SCR and other options, like selective non-catalytic reduction (SNCR), natural gas reburn, and combination of approaches, are now being actively tested and demonstrated on existing boilers in the U.S. in response to existing and pending regulatory requirements. As a result, the number of commercial-scale SCR systems for coal-fired boilers in the U.S. is expected to grow from 10 to more than 30 units in the next two to three years.

In sum, the last decade or so of NOx regulation in the U.S. has elicited a strong technological response, though not in the sense that it has spawned new control technologies. Rather, recent developments have demonstrated the rapidity and success with which existing technologies can be adapted and implemented once the regulatory incentives exist to do so. In the relatively short span of the last five years, technologies capable of reducing NOx by as much as 90 percent and even higher have been demonstrated on existing U.S. boilers. This represents more than a doubling of the reduction levels typically achieved over the two decades before 1995 when a relatively limited suite of technologies sufficed to meet existing standards. Similarly deep reductions are likely to become commonplace if control programs already adopted in the Northeast and Southern California -- and proposed for much of the eastern half of the country -- are implemented as planned over the next several years.

Case Study 3: Power Plant SO_2

Efforts to control sulfur dioxide emissions from power plants predate even the earliest attempts to reduce power plant NOx emissions or control automobile exhaust. To this day, the primary add-on control technology used for sulfur control remains flue gas desulfurization (FGD). This basic approach, commonly known as "scrubbing,” was first demonstrated at the Battersea power plant in London as early as 1936. The history of FGD development over the subsequent 65 years is an interesting one. On the one hand, the technology has advanced dramatically to the point where modern FGD systems are highly reliable, capable of achieving routine control levels of over 95 percent, and equipped to generate salable by-products, such as gypsum, instead of waste. On the other hand, actual deployment of FGD systems remains surprisingly limited, given ongoing concerns about acid deposition and newer concerns about the health and visibility impacts associated with fine airborne particles (which in the East tend to have a significant sulfate component). At present, about

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[5] EPA's proposal is authorized under Section 110 of the Clean Air Act, which provides for federal action to reduce interstate transport of ozone. Its implementation has so far been delayed by litigation. Both Phase III of the OTC MOU and EPA's Section 110 action would limit NOx emissions to about 0.15 lb/mmBtu by 2003.

one-third of all coal-fired electric generating capacity in the U.S. is being scrubbed (a total of about 90 GW); worldwide scrubbed capacity is over 250 GW.

Early FGD systems were typically installed on a case-by-case basis in response to local concerns about the corrosive effects of acid emissions on human health, vegetation, and manmade structures. The first U.S. requirements for add-on SO_2 controls, introduced as part of the NSPS program in 1970, applied only to new and substantially modified facilities. Throughout the 1970s and 1980s existing plants were subject to control requirements only to the extent they could be shown to contribute to non-attainment of ambient air quality standards for SO_2. However, early FGD systems had serious reliability problems and hence were often avoided, especially at a time when cheaper compliance alternatives -- notably, building taller smokestacks to disperse emissions and switching to low-sulfur fuel oil -- were available to mitigate local impacts. Thus much of the growth in FGD installations over this time period (the number of systems deployed nationwide grew from 4 units in 1971, to 52 units in 1980 and nearly 150 units in 1990) was concentrated among new or modified facilities subject to NSPS.

Despite the relatively limited U.S. market for FGD retrofits, the reliability and performance of scrubber systems improved steadily. First, those facilities that did implement FGD systems had an interest in making them work properly. Second, federal agencies, suspecting that tall stacks should not and would not be the final answer to SO_2 control, continued to commit substantial public funds to related research and development efforts. Third, the formation in 1974 of the Electric Power Research Institute (EPRI) by the electricity industry in response to ongoing reliability problems in the late 1960s and early 70s provided another important institutional resource for furthering research and development efforts. Fourth, an anticipated boom in FGD demand prompted a number of competing technology vendors to enter the market in the early 1970s. The technology vendors also helped to advance system design and operation. Finally, commercial-scale applications overseas, notably in Europe and Japan, played a substantial role in advancing the capability and reliability of FGD technology.

Growing concern about acid rain and recognition of the impact of long-range transport of sulfates (formed from sulfur dioxide) from tall smokestacks inaugurated a new phase of SO_2 regulation under Title IV of the 1990 Clean Air Act Amendments. Importantly, this legislation also provided for a novel "cap and trade" approach which made it possible for some sources to purchase additional emissions allowances in lieu of installing control technologies, while providing other sources with a market incentive to over-comply if they found it cost effective to do so. To date, this approach has created an active market for emissions allowances and has achieved targeted reductions at lower cost and with less disruption to electric supply systems than first anticipated. However, the current Acid Rain Program also is creating far less demand for FGD retrofits than was originally anticipated. In fact, only 27 new installations have been ascribed to the implementation of Phase I requirements. This outcome is the result of several factors, chief among them the fact that switching to lower-sulfur fuels has emerged as a highly cost-effective and popular option,[6] and the fact that early compliance and banking options effectively increased the pool of available allowances in the early years of the program. Nevertheless, FGD technologies have advanced under the current

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[6] A key factor in this regard is the reduction in transport costs for low-sulfur western coals that followed the earlier deregulation of the U.S. railroad industry.

cap-and-trade approach. Because excess allowances have market value, those facilities that do invest in scrubbers have an incentive to maximize the emissions reductions.

Thus, SO_2 presents an unusual case in the sense that reductions achievable with commercial control technologies have outstripped present regulatory requirements. Put another way, the *quality* of modern FGD technology (which is capable of achieving emissions reductions in excess of 95 percent) is not matched by the *quantity* of FGD installations being stimulated by a program that aims to achieve overall national reductions of only 50 percent. Whether scrubber systems are eventually installed on a greater number of power plants depends on the continued viability of fuel switching (which is in turn sensitive to relative fuel costs) and on whether ongoing concerns about acid deposition, combined with new imperatives to address fine particulate pollution and regional haze, lead to a further tightening of the national SO_2 cap.

Cost Issues

Compliance costs are an important factor in most regulatory decisions; Chapter V of this report reviews the cost histories associated with all three case studies. Once again, a pattern emerges in which early estimates consistently overstate actual compliance costs, often by a factor of two or more. Likely reasons for this pattern include poor or incomplete information, conservative assumptions (occasionally motivated by a political desire to bolster the case against regulation), and a failure to account for the often dramatic effects of technological innovation. Even "independent" estimates tend to exhibit these biases, though typically to a lesser degree, since outside analysts often rely on industry data and use similarly conservative assumptions.

In any case, the evidence from past experience with power plant NOx and SO_2 controls is highly suggestive. Total costs, including both capital and operation and maintenance (O&M) costs, tended to fall dramatically as control technologies passed from the conceptual research and development phase to full-scale demonstration and commercialization. In some cases, this process first occurred overseas so that by the time control technologies such as SCR were imported to the U.S. from countries with more stringent regulations, costs already had undergone a dramatic decline. Thus, in the NOx case, cost estimates declined by as much as 90 percent (on dollars per ton of NOx removed basis) for SCR technology and 75 percent (on a levelized, mills per kWh basis) for SNCR between 1989 and 1998. Cost trends for Phase II of the national Acid Rain Program are similarly striking. In 1989, industry estimated that annual compliance costs would range from $4.7-6.6 billion per year with trading; a year later, EPA put the range at $1.6-5.3 billion per year. By 1997, EPRI's estimate had fallen to $1.5-2.1 billion per year, three to four times lower than the figures widely cited in the congressional debates that shaped the 1990 Amendments. The automobile case, though different in terms of the technology trajectory involved, exhibits similar patterns of cost overstatement. For example, costs estimated by industry in 1994 for achieving various categories of low emission vehicles under California's LEV program were anywhere from five to more than ten times higher than costs estimated by the California Air Resources Board (CARB) in the same year. The CARB's estimates have proven to be quite close to the actual cost.

In sum, past experience suggests that pre-regulatory cost estimates, especially those at the high end of various assessments, must be viewed with caution. In fact, it is usually reasonable to assume that technological innovation and commercialization will reduce actual control costs by an indeterminate but substantial amount.

Status of Mercury Control Technologies

Despite the current lack of mercury control requirements for coal-fired power plants, considerable work has been undertaken and is now underway on identifying and developing potential control technologies. At this point, a number of options appear to be available in the near to medium term. These include "pollution prevention" approaches, such as switching to fuels with essentially zero mercury content (e.g. natural gas), co-firing with low-mercury fuels, and coal cleaning, a technique that can remove some of the mercury from coal prior to combustion. In addition, a number of add-on control devices already in use for other pollutants are being tested for their ability to remove mercury. Of these, early results suggest that significant amounts of mercury can be removed in fabric filters (baghouses) and by wet scrubber systems. Interestingly, certain combinations of existing technologies may prove more effective than stand-alone systems. For example, by promoting the oxidation of elemental mercury to soluble forms, SCR systems installed for purposes of NOx control may substantially enhance the mercury removal achievable by wet scrubber (FGD) systems. Limited bench-scale experiments with these types of technologies have yielded a wide range of mercury removal rates, from as low as a few percent to as high as 75 to 95 percent.

At present, the most promising control technologies specific to mercury involve the use of sorbents (typically, activated carbon, though other, more exotic sorbents are also being considered) which are injected into the flue gas to adsorb mercury and are subsequently collected in particulate control devices. This basic approach has been employed with considerable success to reduce mercury emissions from sources such as municipal and medical waste combustors; its application to coal-fired boilers presents additional challenges because of the relatively lower concentrations of mercury and high volumes of gases present in power plant exhaust. Nevertheless, sorbent-based controls for power plants are near commercialization and have demonstrated mercury removal rates of greater than 90 percent in some pilot-scale applications. In addition, a number of other technologies -- such as corona discharge, fluidized circulating bed, COHPAC[7], and electro-catalytic oxidation -- are being explored. These activities suggest that a variety of control options will likely become available, provided regulatory or other incentives are sufficient to sustain interest in the ongoing development of these and other technologies.

Given the experimental status of most available control options, cost estimates for varying levels of mercury removal are preliminary in nature. Extrapolating from current pilot-scale applications, recent estimates by EPA suggest that the costs of sorbent-based mercury controls are likely to range from 0.17 to 1.76 mills per kWh. On a per kWh basis, this range is comparable to that associated with existing NOx control technologies such as LNB or SCR.[8] In short, the prospects

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[7] EPRI's COHPAC system, which has been demonstrated at PSE&G's Hudson Station in New Jersey, combines electrostatic precipitator (ESP) technology, a pulse-jet baghouse, and sorbent injection to improve the efficiency of both mercury adsorption and subsequent particulate capture.

[8] Given the much smaller quantities of mercury being removed, control costs on a per ton or per pound basis are, of course, substantially higher. This type of comparison is not relevant, however, given the far greater health and environmental hazards of mercury compared to an equivalent amount of conventional pollutants.

for successful near-term regulation of power plant mercury emissions are highly promising in light of the work that already has been done to advance several control options and given the reasonable magnitude of current EPA cost estimates.

Conclusions

This report does not attempt to argue that past regulatory experience provides an exact recipe for reducing mercury emissions from coal-fired power plants. However, there is little reason at this time to suspect that achieving such reductions will pose technological or economic hurdles distinct in kind or degree from those encountered in past technology applications. On the contrary, it is highly probable that the underlying trajectory of technology development common to all of the case studies also applies to the mercury situation. If so, policymakers can be reasonably certain of two things: first that mercury emissions from power plants can be substantially and cost-effectively controlled. Second, that the control technologies capable of accomplishing this will not become commercially available and widely implemented until regulatory or other incentives encourage their introduction.

Beyond these basic observations, the case studies offer a number of other insights relevant to the current mercury debate:

- Research and development efforts are unlikely to be sustained at a vigorous level in the absence of regulatory or other drivers capable of creating a viable market for advanced control technologies.

- The stringency and timing of emissions reduction requirements strongly influences subsequent technology choices. Weak standards can prove inefficient, both in the sense that they promote investment in control technologies that may ultimately be deemed inadequate, and in the sense that they can necessitate politically costly and protracted reiterations of the regulatory process, thereby delaying needed human health and environmental protections.

- A variety of frequently unpredictable factors, such as fuel costs, structural changes, and other policy developments, can profoundly affect future technology choices. Hence, the most successful regulations have avoided picking "winners," instead focusing on establishing well-defined performance goals.

- A combination of aggressive performance requirements and flexible attainment mechanisms has proven highly successful in the past. An important benefit of cap and trade approaches is that they provide incentives for ongoing technology improvement by creating a market for over-compliance at individual facilities.[9]

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[9] Note that over-compliance at individual facilities does not translate into aggregate over-compliance, since excess allowances from one facility become available to others who may then choose to under-comply. As with more traditional approaches, it remains unlikely that industry, on an aggregate basis, would choose to exceed minimum requirements. Emissions fees or taxes can provide dynamic incentives for continued reductions, but these alternative regulatory approaches have not been widely used to date.

- Costs almost always decline substantially once regulatory mandates are introduced and control technologies are commercialized.

To conclude, past experience suggests that further delay in the regulation of mercury emissions from power plants cannot be justified on the basis of concerns about technology availability. On the contrary, delay is likely to stall efforts to advance promising control technologies, efforts that are already yielding promising results. If, on the other hand, emissions limits and deadlines are established soon, available evidence from more than four decades of environmental regulation in the U.S. strongly indicates that the successful commercialization of cost-effective mercury controls will soon follow.

I. Introduction

A. Background and Context

Mercury is a toxic heavy metal that causes serious health effects at high doses and more subtle developmental effects at lower doses. Once released to the atmosphere it persists in the environment, often traveling long distances before being deposited to the earth's surface where it tends to bioaccumulate[10] in aquatic ecosystems. Current concerns about mercury stem from the recognition that mercury levels in certain fish populations have reached levels that may present health risks to exposed individuals and wildlife. Ingestion of contaminated fish constitutes the primary pathway of exposure to mercury and is thought to pose special health risks for the developing fetus and young child. Nationwide, 41 states, including all eight northeastern states, have issued mercury-related fish consumption advisories.

Coal-fired power plants presently account for about one-third of total emissions of mercury from human activities in the United States, or an estimated 50 tons annually.[11] Their contribution relative to other anthropogenic sources in North America is likely to grow with the implementation of stringent control requirements for other major sources, such as municipal waste combustors and medical waste incinerators, and with the already completed phase-out of mercury content in such consumer products as batteries and paint. By contrast, mercury emissions from power plants have remained unregulated to date.

Pressure to change this situation is growing. In 1998, the New England states and eastern Canadian provinces adopted a resolution that established a long-term goal of "virtual elimination" of mercury emissions within the region. This resolution was followed by a nearer term commitment to reduce emissions by 50 percent by 2003. As a result of this and other initiatives, a number of northeastern states have introduced mercury emissions limits for municipal waste combustors and medical waste incinerators that are more stringent than federal standards. In addition, individual states in the Northeast and Midwest are actively exploring regulatory options to reduce emissions from electricity generating boilers. More recently, in July 2000, the New England Governors and Eastern Canadian Premiers adopted a new resolution calling for reductions in mercury emissions of up to 90 percent from coal-fired power plants in the region.

At the federal level, the U.S. Environmental Protection Agency (EPA) is required under Section 112 of the 1990 Clean Air Act Amendments to study mercury emissions and to issue regulations to protect human health and the environment if necessary and to date has issued two major reports on the subject. The EPA's 1997 *Mercury Study Report*

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[10] Mercury bioaccumulates in an organic form called methylmercury. It is this specific type of mercury that is of concern from the standpoint of human exposure.

[11] This total was estimated by EPA in its 1997 Mercury Study Report to Congress, cited later in this Introduction.

to Congress identified significant health and environmental impacts associated with mercury. EPA's subsequent *Study of Hazardous Pollutant Emissions from Electric Utility Steam Generating Units*, completed in 1998, identified mercury as the hazardous pollutant of "greatest potential concern." Prior to issuing these reports, EPA had examined the available scientific literature on mercury health effects and had proposed a "reference dose" for daily methylmercury exposure intended to estimate the threshold below which there was no appreciable risk of harmful health effects. EPA's reference dose has been used as a guideline by numerous states in issuing fish consumption advisories; importantly, its validity was recently confirmed in a National Academy of Sciences report, *Toxicological Effects of Methylmercury*.[12]

Based on these and other studies, EPA is required under the 1990 CAA Amendments, to make a determination about the necessity of initiating regulatory action to reduce power plant emissions of mercury. Having missed the original statutory deadline, EPA is now under court order to make this determination by December 15, 2000. In the meantime, several new assessments of mercury emissions control technology are underway or have recently been completed under the sponsorship of NESCAUM, EPA, and the U.S. Department of Energy.

B. Purpose and Organization of this Report

As the issue of regulating mercury emissions from coal-fired power plants gains urgency, especially in advance of EPA's December 15 decision, industry interests are voicing a number of familiar objections. Many of these objections center on the likely availability and cost of control technologies that have not yet been commercialized, on the limited experience with these technologies, and on the mixed results obtained from research and development efforts to date. In this regard, the current mercury debate – though unique in its particulars – is not dissimilar from many earlier regulatory debates. The purpose of this report, therefore, is to review the status of mercury control technologies in the context of past regulatory experience and by doing so to provide insights relevant to the decisions now being faced by state and federal regulators.

To that end, this report summarizes three discrete case studies of technological response to environmental regulation. Chapter II reviews past experience with emission regulation of automobile industry, while Chapters III and IV describe the evolution of NO_x and SO_2 controls for coal-fired boilers. Because controversies about cost typically play an important role in the policymaking process, the case studies are followed by a separate discussion in Chapter V of how cost information typically evolves, together with technology, prior to and after the imposition of regulatory requirements. Chapter VI then provides a summary of the current state of knowledge concerning technological options for reducing mercury emissions from coal-fired boilers. The concluding chapter draws lessons from the case studies and connects them to the current mercury debate.

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[12] Issued by the National Research Council in July 2000.

The insights gained from this approach are necessarily qualitative; nevertheless, they are illuminating and point compellingly to the importance of regulatory drivers in advancing pollution control technologies. Time after time, cost-effective emissions controls were not widely commercialized until after regulatory mandates were introduced. Once mandates were established, industry almost always proved capable of meeting them, often at far lower cost than originally anticipated. Regulatory delay not only did not help to advance control technologies, it tended to have the effect of slowing further progress. In sum, past regulatory experience strongly suggests that mandated reductions in mercury emissions from power plants are not only feasible but may be necessary to significantly advance future control technology development.

This report was prepared by the Northeast States for Coordinated Air Use Management (NESCAUM) with the assistance of several consultants with specific expertise in the different areas of environmental regulation discussed here. NESCAUM is a non-profit association of the air quality management agencies of eight northeastern states, including Connecticut, Maine, Massachusetts, New Hampshire, New Jersey, New York, Rhode Island, and Vermont. NESCAUM coordinated a major 1998 assessment of mercury in the northeast states and eastern Canadian provinces as well as a more recent, June 2000 report on the status of mercury control technologies. NESCAUM continues to be involved in state and regional efforts to reduce mercury emissions and provide input to federal regulators.

II. The Regulation of Automobile Emissions: A Case Study

A. Introduction

At first glance the history of automobile emissions controls might appear to have little relevance to current debates about the reduction of mercury emissions from power plants. Obviously, power plants and cars are very different, as are the industries that are affected by their regulation. Nevertheless, this case study provides a useful and compelling introduction to the general topic of technological response to regulatory drivers. This is because, from both a technological and regulatory standpoint, the progress achieved in reducing automobile emissions stands as one of the most important and impressive accomplishments of the last half-century of environmental policymaking in the U.S. A review of how this progress unfolded provides a clear illustration, not only of the direct link between technological innovation and regulation, but of the many non-technological considerations that shape policy decisions on the one hand and industry responses on the other. Thus, the automobile example proves to have important similarities to the two subsequent case studies concerning power plant emissions and introduces a number of themes, patterns, and linkages pertinent to the mercury question as well.

Technologies to reduce automobile emissions have evolved in direct response to three decades of regulatory intervention by the federal government and the State of California. The result has been a dramatic reduction -- by as much as two orders of magnitude -- of typical new car emissions rates for several key pollutants, achieved at a time when significant improvements were also being made to vehicle performance and quality. Moreover, emissions controls did not make cars unaffordable; the automobile industry at the turn of the millennium is as profitable as it has ever been.

This progress was not achieved easily; it entailed years of scientific and political conflict, frequently occurred in fits and starts, and led to a few failed experiments along the way. Nevertheless it is clear that progress of this magnitude would not have occurred without regulatory drivers. Before such drivers were introduced in the early 1970s, American automobile manufacturers made only minimal progress toward controlling emissions; in fact, they were accused by the federal government in 1969 of actively colluding to suppress the development of catalyst technology. When a first serious federal effort at emissions regulation was finally proposed under the Clean Air Act Amendments of 1970, manufacturers reacted by warning of “business catastrophe and massive difficulties,” conceivably leading to “a complete shutdown of the U.S. auto industry”.[13]

As it turned out, the emissions limits proposed in 1970 were deferred several times but were eventually met in the mid-1980s. At no point did these and subsequent regulations stop automobile production and, with the exception of some limited model availability in California for a few years, manufacturers experienced relatively few problems in complying with state and federal emissions requirements.[14] Today’s production vehicles routinely meet standards that are four to ten times more

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[13] The former prediction was made by Earnest Starkman, vice president of General Motors in 1972; the latter by Lee Iacocca, president of Ford Motor Company in 1973 (Mastio, 2000).

[14] Problems were experienced in *maintaining* the emissions performance of vehicles in use; these are discussed elsewhere in this chapter.

stringent than the proposed standards (CARB, 1994a) that had inspired such dire predictions in the early 1970s, and prototype gasoline vehicles have been produced that are cleaner still. As a result, most urban areas of the country – including Los Angeles – have experienced measurable improvements in air quality since the 1970s, though many – again, including Los Angeles – still fail to meet health-based national standards.[15] Given the relentless increase in automobile use that also occurred during this period, it is clear that these gains could not have been achieved without substantial advances in emissions control.

The specific regulatory, technological and political aspects of automobile emissions control over the past fifty years are described in separate sections of this chapter. Throughout, several institutional actors, notably the State of California, the U.S. Congress, and the U.S. Environmental Protection Agency (EPA), play a prominent role. California's unique importance in this evolution is an outgrowth of the unusually severe smog conditions that had begun to afflict the Los Angeles basin by the 1940s. To this day, California retains the unique ability to set and design its own automobile standards and serves as the nation's laboratory for regulatory innovation in the arena of motor vehicle emissions controls. Since the landmark 1970 Clean Air Act Amendments, EPA has also played a critical role as the standard-setting authority for vehicles supplied to most other states in the nation.[16] More recently, northeastern states, some of which have moved to adopt California's standards under authority granted by the Clean Air Act, are playing a more proactive role in shaping the future evolution of automobile emissions regulations.

B. A Brief History of Automobile Emissions Control

Figure II-1 shows composite emissions limits under California and federal law over the period from 1965 to 2005 for nitrogen oxides (NO_x), volatile organic compounds (VOCs)[17], and carbon monoxide (CO).[18] The figure also includes actual in-use emissions from California for 1970 to 1987 model years (in-use data were not available at the federal level). As is evident from the figure, California began regulating automobile emissions somewhat earlier than the federal government and tended to have more stringent requirements until the mid-1980s. Since then, California and federal requirements have tended to converge, although the emissions limits included in the second phase of California's Low Emission Vehicle (LEV) program remain more stringent than the latest federal requirements.

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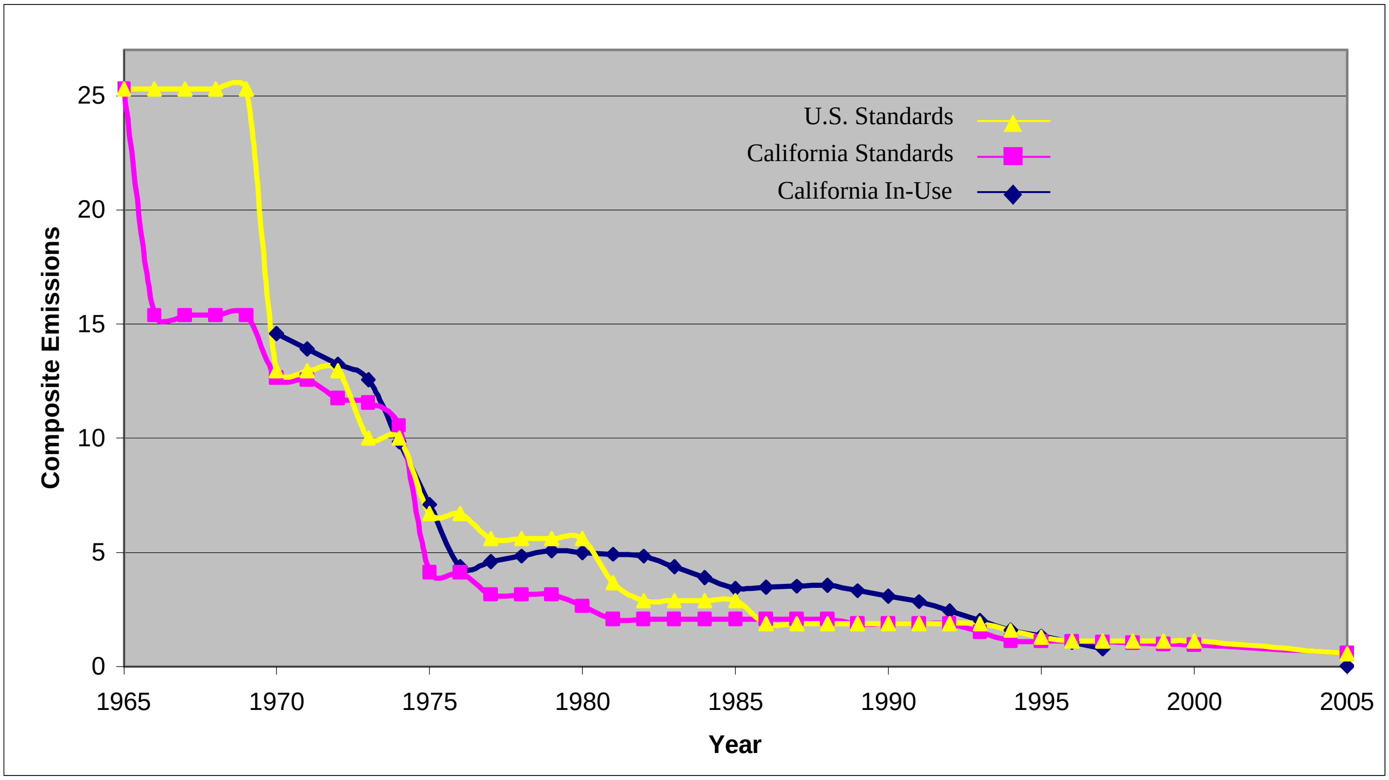
[15] As was pointed out in the early 1990s, vehicle emissions inventories have not declined as dramatically as the improvement in new car emissions performance might have suggested (Calvert, et al. 1993). Reasons for this undoubtedly include in-use deterioration of emissions controls, vehicle tampering, the fact that test cycles used to demonstrate compliance do not reflect real-world driving conditions, and the presence in the fleet of a small number of "super-emitters" that contribute disproportionately to overall emissions.

[16] Federal standards apply to most non-California cars. However, other states can adopt California standards and a few northeastern states, including Vermont, New York, Massachusetts, and Maine, have availed themselves of this option.

[17] Older regulations frequently referred to hydrocarbons in place of VOC, while more recent regulations sometimes reference non-methane organic gases (NMOG). All these terms have somewhat different technical meanings, but can be used interchangeably for purposes of this discussion.

[18] As is discussed later in this chapter, NOx and VOC are precursors to the formation of tropospheric ozone (i.e. photochemical smog), while CO poses direct health risks when inhaled. The values graphed represent the sum of g/mi emissions rates for NOx and VOC, plus the CO g/mi emissions rate divided by 7. CO is discounted to more fairly represent its environmental impact relative to NOx and VOC.

Figure II-1: Automobile Emissions Trends



In providing an overview of this history of emissions control, it is useful to distinguish between three periods, a pre-regulatory period before 1970, a first regulatory period from 1970 to 1987 and a second regulatory period from 1988 to 2005. In each of the regulatory periods, a new round of mandated emissions limits led to dramatic improvements in vehicle control technologies. Before 1970 and in the mid-1980s, when technology-forcing mandates either did not exist or had been met, additional improvement did not occur and vehicle emissions rates leveled off.

Figure II-2a: 1970-1987 Auto Emissions

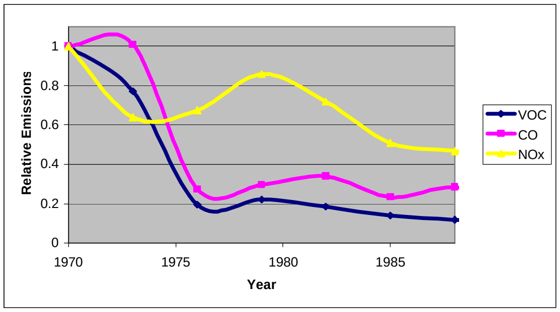
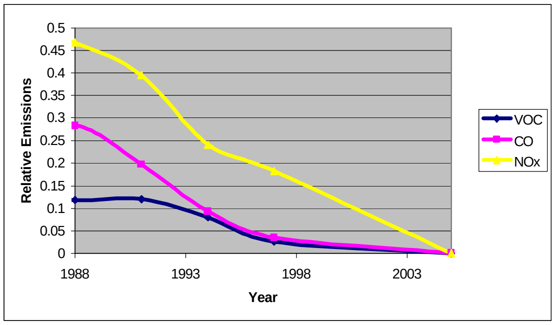


Figure II-2b: 1988-2005 Auto Emissions



B.1 The Pre-Regulatory Period (1940-1969)

The problem of air pollution generally, and of automobile emissions specifically, had of course been identified long before 1970. Prior to 1960, most air quality regulations in California and elsewhere focused on industrial sources and visible types of pollution (such as soot and smoke). Automobiles were first targeted with California's Motor Vehicle Pollution Control Act of 1960, which created a state board to certify pollution control devices for automobiles. That same year, California's Department of Public Health established ambient air standards and articulated the goal of reducing total hydrocarbon emissions from the state's vehicle fleet back to 1940 levels. However, these requirements did not specify emissions limits per se and had no deadlines. A year later, California took the first concrete step toward controlling automobile emissions by mandating the use of positive crankcase ventilation (PCV) technology to reduce evaporative hydrocarbon emissions. Perhaps because the technology involved was both simple and cheap, this step appears to have met with no serious opposition from auto manufacturers, some of whom began to install PCV valves even before they were required. Modest tailpipe standards (see Table II-1) were not adopted by California until 1964; these took effect in 1966 and required CO and VOC emissions to be reduced by roughly half to two-thirds, but left NO_x emissions levels essentially unchanged.

Progress toward emissions standards was similarly gradual at the federal level and until the 1970s consisted largely of following California's lead. Congress first mandated research into automobile pollution in 1962; this was followed by a request for recommendations concerning the need for federal regulation (under the first Clean Air Act in 1963) and later by a mandate for developing regulations "as soon as practicable" (under the Clean Air Act Amendments of 1965). Meanwhile, the federal Motor Vehicle Pollution Control Act of 1965 extended California's modest tailpipe standards to all cars as of 1968 (Krier and Ursin, 1977, p. 175). Importantly, Congress in its 1967 Amendments to the federal Clean Air Act, narrowly preserved California's right to establish independent motor vehicle emissions standards despite a strong industry push to strip the state of that ability.[19]

These initial efforts at automobile regulation were not particularly effective and certainly did not go far enough to offset the explosive growth in car ownership and use that was occurring at the same time. Air quality in Los Angeles and other areas of the U.S. continued to deteriorate, creating political pressures that eventually led, in 1970, to a new set of federal Clean Air Act amendments.

B.2 The First Regulatory Period (1970-1987)

The landmark 1970 Clean Air Act Amendments marked a turning point in air quality regulation generally[20] and automobile emissions control specifically. For the first time, auto manufacturers were faced with clear deadlines and technology-forcing mandates. Specifically, the newly created EPA was charged with reducing automobile emissions by 90 percent relative to a 1970 baseline within a five-year timeframe. As a result of the 1970 Amendments, emissions limits

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[19] As noted previously, other states do not have this ability, although they are allowed to adopt California's program. The main justification for this preemption is the concern that a patchwork of differing state requirements would pose an undue burden on the industry.

[20] In other key provisions, the 1970 Amendments called for the establishment of national, health-based ambient air quality standards and set deadlines for states to submit detailed plans for achieving those standards.

for VOC and CO were established for 1975 and an emissions limit for NO_x was established for 1976. These deadlines were later postponed several times in response to strident industry opposition and a changing political dynamic (described in the next section). As a result, the emissions limits envisioned in the 1970 amendments were not achieved nationwide until the early 1980s for CO and VOC and even later in the case of NO_x.

For most of the first regulatory period, California enforced emissions limits for VOC and NO_x that were more stringent than federal limits (see Table II.1). However, California's CO standard, while more stringent than the federal CO standard for the late 1970s was actually less stringent than the federal CO standard during the 1980s. This difference reflects a difference in policy assumptions and emphases: compared to the federal government, California at the time was more concerned with NO_x reductions and had concluded that relaxing the CO standard was necessary to achieve more stringent NO_x limits. In any case, by the end of the first regulatory period in 1987, both California and the federal government had successfully reduced automobile emissions – by more than a full order of magnitude in the case of VOC and CO and more than four-fold in the case of NO_x.

B.3 The Second Regulatory Period (1988-2005)

After the mid-1980s, emissions rates remained largely unchanged for the remainder of the decade. By late in the decade it was becoming painfully apparent that while air quality had improved relative to the 1970s, Los Angeles and other major urban areas were still far from meeting national health-based standards for ozone. This realization, combined with a resurgence of public interest in environmental issues more generally, contributed to the political momentum for a major overhaul of state and federal air policies. In 1988, California passed its own Clean Air Act which required, among other things, that the California Air Resources Board (CARB) take "whatever actions were necessary, cost effective, and technologically feasible" to reduce vehicle VOC and NO_x emissions by 55 percent and 15 percent, respectively, relative to a 1987 baseline. These reductions were to be achieved by 2001. California's action was followed in 1990 by the adoption in Congress of a sweeping new set of amendments to the nation's Clean Air Act which required automakers to begin reducing tailpipe VOC and NO_x emissions by 35 percent and 60 percent, respectively, starting in 1994. In addition, EPA was authorized to require another 50 percent reduction in emissions after 2003, upon a finding of technological feasibility, necessity, and cost-effectiveness. Using this authority, EPA in 1999 finalized a second tier of federal standards that will take effect in 2005. Compared to the average uncontrolled 1965 car, the VOC emissions of a car certified to federal Tier 2 standards will be more than 98 percent lower, its CO emissions will be almost 98 percent lower, and its NO_x emissions will be 95 percent lower.[21]

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[21] Average 1965 emissions rates were 10 g/mi for VOC, 80 g/mi for CO, and 4.1 g/mi for NOx. Federal Tier II standards for these pollutants will be 0.125 g/mi, 1.7 g/mi, and 0.20 g/mi, respectively.

Table II-1: A Comparison of California and U.S. Auto Standards (gm/mile) 1965-2005[22]

	United States			California			
	VOC	NOx	CO	VOC	NOx	CO	Notes
Pre-Reg	11	4.1	84	11	4.1	84	
1966-1967				6.3	n/s	51	CARB standards were concentrations. Values are estimated as CVS75 values.
1968-1969	6.3	n/s	51	6.3	n/s	51	USEPA standards were based on CARB standards. Values are estimated as CVS75 values.
1970	4.1	n/s	34	4.1	n/s	34	CARB standards were based on a 7 mode test. USEPAl standards were based on CVS72. Values are estimated as CVS75 values.
1971	4.1	n/s	34	4.1	4	34	CARB standards were based on a 7 mode test. U.S. EPA standards were based on CVS72. Values are estimated as CVS75 values.
1972	3	n/s	28	3	4	34	CARB standards were based on a 7 mode test. USEPAl standards were based on CVS72. Values are estimated as CVS75 values.
1973	3	3.1	28	2.9	3	34	CARB standards were based on a 7 mode test. USEPAl standards were based on CVS72. Values are estimated as CVS75 values.
1974	3	3.1	28	2.9	2	34	CARB standards were based on a 7 mode test. USEPAl standards were based on CVS72. Values are estimated as CVS75 values.
1975-1976	1.5	3.1	15	0.9	2	9	All subsequent values are based on CVS75.
1977-1979	1.5	2	15	0.41	1.5	9	All subsequent values are based on CVS75.
1980	0.41	2	15	0.39	1	9	Movement to non-methane VOC
1981	0.41	1	3.4	0.39	.7 or 1	7 or 3.4	Meeting the lower NOx limit allows a higher CO limit. Must warranty 0.7 vehicles for 7yr/75K miles.
1982-1988	0.41	1	3.4	0.39	.4 or .7	7	If .7 NOx selected must meet 7yr/75K miles warranty rather than 5yr/50K warranty.
1989	0.41	1	3.4	0.39	.4 or .7	7	If .7 NOx selected must meet 7yr/75K miles warranty rather than 5yr/50K warranty. A minimum of 50% must meet .4 NOx standard
1990-1992	0.41	1	3.4	0.39	.4 or .7	7	If .7 NOx selected must meet 7yr/75K miles warranty rather than 5yr/50K warranty. A minimum of 90% must meet .4 NOx standard
1993	0.41	1	3.4	0.25	.4 or .7	3.4	If .7 NOx selected must meet 7yr/75K miles warranty rather than 5yr/50K warranty. A minimum of 40% must meet .4 NOx standard
1994	0.25	0.4	3.4	0.25	.4 or .7	3.4	A minimum of 80% must meet .4 NOx standard
1995-1996	0.25	0.4	3.4	0.23	(0.4)	(3.4)	LEV program has no overall NOx and CO standards. NOx and CO estimated from VOC standards and projected fleet mix.
1997	0.25	0.4	3.4	0.2	(0.35)	(3.4)	LEV program has no overall NOx and CO standards. NOx and CO estimated from VOC standards and projected fleet mix.
1998	0.25	0.4	3.4	0.16	(0.30)	(3.3)	LEV program has no overall NOx and CO standards. NOx and CO estimated from VOC standards and projected fleet mix.
1999	0.25	0.4	3.4	0.11	(0.24)	(3.3)	LEV program has no overall NOx and CO standards. NOx and CO estimated from VOC standards and projected fleet mix.
2000	0.25	0.4	3.4	0.07	(0.20)	(3.2)	LEV program has no overall NOx and CO standards. NOx and CO estimated from VOC standards and projected fleet mix.
2001	(.13)	0.3	(3.4)	0.07	(0.20)	(3.2)	USEPA and CARB now both use fleet averaging approach based on VOC standard for CARB and NOx standard for USEPA.
2002	(.075)	0.3	(3.4)	0.07	(0.20)	(3.1)	USEPA and CARB now both use fleet averaging approach based on VOC standard for CARB and NOx standard for USEPA. 5% ZEV requir.
2003	(.075)	0.3	(3.4)	0.06	(0.18)	(2.8)	USEPA and CARB now both use fleet averaging approach based on VOC standard for CARB and NOx standard for USEPA. 10% ZEV requir.
2004	(.075)	0.2	(1.7)	0.06	(0.14)	(2.5)	USEPA and CARB now both use fleet averaging approach based on VOC standard for CARB and NOx standard for USEPA. 10% ZEV requir.
2005	(.075)	0.2	(1.7)	0.06	(0.11)	(2.5)	USEPA and CARB now both use fleet averaging approach based on VOC standard for CARB and NOx standard for USEPA. 10% ZEV requir.

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[22] Shaded, bold numbers indicate the more stringent standard for that pollutant in that year in cases where there were differences between the California and U.S. vehicle emission standards.

California, meanwhile, developed standards for several new categories of vehicles in its effort to enforce progressively stricter standards for the 1990s and beyond. Its Low Emission Vehicle (LEV) program, which is now in its second phase, includes "transitional low emission vehicles" (TLEVs), LEVs, "ultra low emission vehicles" (ULEVs), and, more recently, super ultra low emission vehicles (SULEVs). Automakers are required to meet the California standards by producing a mix of vehicles in these categories.[23] Additionally, California adopted sales mandates for so-called "zero emission vehicles" (ZEV's) in an effort to promote alternatives to the internal combustion engine. When it was adopted in the early 1990s, the ZEV category was understood to mean battery electric vehicles. Originally ZEVs were required to comprise at least 2 percent of all cars sold in California by 1998, at least 5 percent by 2001, and 10 percent by 2003. Because of concerns about the market-readiness and cost of battery technology, the near-term (1998 and 2001) ZEV mandates were subsequently dropped and replaced by technology demonstration agreements with the auto manufacturers. The 10 percent mandate for 2003 still stands, although CARB is currently considering possible modifications. These could include changes to the timing and/or scale of the mandate, as well as increased flexibility to include other advanced technologies, such as hybrid electric and fuel cell vehicles.

C. The Scientific Debate:

Each step in the evolution of automobile regulations and technologies has been accompanied by a debate about the contribution of automobiles to air pollution and about the necessity and feasibility of reducing their emissions. As with other important policy questions, affected industries – in this case primarily auto manufacturers and oil companies – attempted to shape and in some cases prolong the scientific debate in ways that served their own interests. Often this involved sponsoring their own research and enlisting academic experts whose views could help support industry's position within and beyond scientific circles.

A scientific understanding of ozone smog and its causes first began emerge in the late 1940s with the pioneering work of Dr. Arie Haagen-Smit at the California Institute of Technology. Haagen-Smit's early findings were vigorously disputed by the petroleum industry-sponsored Stanford Research Institute (SRI), which argued for years that the causes of ozone formation were uncertain and that automobiles rather than refineries bore primary responsibility for southern California's poor air quality. The resulting scientific debate dominated the early 1950s but by the middle of the decade, Dr. Haagen-Smit and others had succeeded in pinpointing and quantifying specific aspects of the ozone formation process. Meanwhile, available studies increasingly implicated the automobile as a major contributor of smog-forming emissions.[24]

The auto manufacturers had by that time created a joint study program to conduct their own research on the question of vehicle culpability for urban air pollution. In 1953, the Ford Motor Company claimed that vehicle exhaust gases were "dissipated in the atmosphere quickly and do not present an air pollution problem." In the same document, Ford insisted that "the need for a device

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[23] A specified hydrocarbon standard governs the mix of vehicles that will be needed; final NOx levels will depend on the mix of TLEVs, LEVs, ULEVs, etc., used to meet the hydrocarbon standard.

[24] A committee of the California State legislature, investigating a severe pollution episode in Berkeley in 1949, drew an early link between automobiles and air pollution when it noted that a likely cause of the episode was the unusually heavy traffic caused by a local football game (Doyle, 2000).

that will more effectively reduce exhaust vapors has not been established".[25] Opposing scientific claims contributed to a perception of uncertainty that affected policymakers' willingness to act. For example, a Los Angeles official stated in 1955 that while there was evidence "pointing in the direction" of a need to reduce vehicle emissions, their "contribution must be clearly demonstrated before any action can be taken" (Krier and Ursin, 1977, p. 259).

By the late 1950s, the accumulation of scientific evidence was such that there was near unanimous agreement among air pollution experts that motor vehicle exhaust would need to be controlled. At that point, auto manufacturers began to focus on a different argument: namely that Los Angeles presented a peculiar and unique case, and that broad-based controls on vehicles could not be justified on the basis of the Los Angeles situation. Chrysler, for example, in a 1958 report to the U.S. Public Health Service, questioned whether the amount of organic hydrocarbons emitted by vehicles would have "any demonstrable significance in the total air pollution problem" absent the specific combination of meteorological and topographical features found in Los Angeles (Heinen, 1958). Ford, in a 1958 report, suggested that these meteorological and topographical features themselves could be responsible for producing "a significant tonnage of hydrocarbons and oxides of nitrogen" (Chandler, 1958). These arguments were ultimately defused by a growing recognition that urban air quality problems were not limited to Los Angeles, indeed that summertime ozone episodes were becoming more acute in many other areas of the country as well.[26]

After 1970, when the question of whether vehicle exhaust would need to be controlled to reduce ozone smog formation had largely been settled, new debates emerged. These tended to focus on questions of how far and how quickly emissions could be reduced and on the potential effects of control requirements on consumers and the auto industry. Nevertheless, scientific arguments continued to be deployed with each regulatory step to tighten emissions control requirements. For example, California for many years placed greater emphasis on NO_x control than did the federal government; hence its NO_x standard for automobiles was more stringent than the national standard until as late as 1995. Auto manufacturers objected to California's more stringent NO_x requirements on the basis of a long-running controversy (which continues to this day) about the relative efficacy of reducing NO_x emissions relative to VOC emissions as a means of controlling ozone formation.[27] The opinion offered in a 1977 Virginia hearing by the U.S. Chamber of Commerce provides a typical illustration of industry's view of such nuances:

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[25] These claims were made in a letter from the Ford Motor Company to LA County Supervisor Kenneth Hahn in March of 1953 (see Krier and Ursin, 1977, pp. 88-9 and Esposito, 1970, p. 37).

[26] For example, the U.S. Department of Health, Environment, and Welfare estimated in 1961 that as much as 90 percent of the nation's population then lived in areas exposed to poor air quality. (Gordon, 1963)

[27] The controversy stems from the complexity of photochemical reactions in the atmosphere. Under some conditions, NOx can act to break down ozone. The net effect in terms of ambient ozone levels depends on the relative concentrations of NOx and VOC and other factors. More recently, EPA has moved closer to California in its emphasis on the need for NOx reductions, which are now believed to be key to controlling ozone formation in many urbanized areas in the eastern U.S. Consequently, federal tailpipe NOx standards are now in line with California's standard. The fact that Los Angeles did not comply with ambient standards for NO_2 probably also helped considerably in justifying a more stringent California NOx standard in the 1980s.

> "So while our present state of knowledge is painfully inadequate, much of the intelligence which we do possess suggests that much pollution is beyond our control due to meteorological and natural origins while feasible control techniques which decrease primary pollutants may concurrently increase secondary pollutants. With our increasing confusion and frustration comes a lessening of faith in the national approach to control of photochemical oxidants." (Knight, 1977).

Thus, the issue of scientific uncertainty – and the dilemmas and frustrations it poses for policy makers – is one which had to be continually overcome throughout the long evolution of automobile emissions regulation. Nor is the automobile case unique. Time and again it has taken political will, rather than a final resolution of scientific uncertainty, to move environmental policy forward.

D. The Political Process

The public, like the scientific community, tended to focus on large, industrial sources and visible types of pollution (such as soot and smoke) for the first half of the 20th century. By the 1940's, however, the persistence of a yellowish-brown haze over Los Angeles, and the eye irritation and loss of visibility it caused, had emerged as major public concerns and citizens and politicians increasingly suspected that the automobile was partly responsible. Even so, it took several decades for the political process to yield effective policy responses to the problem of automobile emissions. The fact that it took so long undoubtedly had more to do with politics than with science or with the state of available control technologies. Put simply, the obvious options for reducing auto emissions – reducing vehicle travel and building cleaner cars – were fraught with political difficulties. Cars were, and are, ubiquitous, dispersed, and central to most people's daily lives (the more so in a geographically large and mobility-dependent area such as Los Angeles); hence attempts to regulate their use are apt to be extremely unpopular and difficult to enforce. Forcing manufacturers to supply cleaner cars has proved to be a far more effective approach, and one generally supported by the public, however it too faced a substantial political hurdle: the concerted opposition, for many years, of auto manufacturers and often oil companies – both powerful and politically well-connected industries.

Los Angeles first began to experience severe ozone episodes, sometimes spanning several days, in the early 1940s. Citizens complained, public demonstrations were held, government officials vowed to take action, and the *Los Angeles Times* engaged its own scientific experts on air quality. A mounting consensus that government intervention was needed led to the formation in 1947 of the Los Angeles County Air Pollution Control District[28] and to new regulations aimed at limiting backyard trash burning and other sources of emissions. By 1953, LA County Supervisor Kenneth Hahn began writing to the major car companies, asking for information on available control technologies to reduce automobile emissions. Frustrated by their less than proactive responses,

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[28] The APCD was predecessor to today's South Coast Air Quality Management District (SCAQMD); its creation was significant from a jurisdictional and institutional standpoint because it unified air policy decision-making at the county-level rather than leaving it in the hands of numerous individual southern California cities

Hahn also sought legal counsel concerning the County's ability to require control devices on automobiles (Koster, 1956, p. 40). The response was that any such requirement would be "arbitrary, capricious and void" until "a satisfactory device is perfected and available on the market" (Krier and Ursin, 1977, p. 98). Instead, the County attorney suggested that it might be possible to ban automobiles from the downtown area altogether until control devices were available. However, such a drastic solution could hardly be politically or legally sustained unless the situation was "sufficiently serious".[29]

The counsel's opinion, in this case, reflected a widely held assumption that control technologies needed to be demonstrated before substantial emissions reductions could be required. Unfortunately, this assumption operated as a powerful disincentive to the advancement of control technologies, since automakers knew that any advances would invite regulation. Thus it was probably not surprising that the only emissions controls introduced in the 1960s – PCV technology and a package of air/fuel ratio and timing adjustments – were simple, cheap, and provided only modest emissions reductions.

Meanwhile, air quality problems continued to worsen, creating a mounting sense of frustration among citizens and public officials by the late 1960s. In 1969, California Sate senator Nicholas Petris introduced a bill in the General Assembly that would have prohibited the sale of all internal combustion vehicles in California by 1975 (Doyle, 2000). The Petris bill did not pass, but it did garner sufficient support to make the automakers take notice and to signal a new impatience on the part of policy makers with the status quo. Impatience was also mounting in Washington DC, where similarly 'radical' proposals to ban or dramatically curtail emissions from internal combustion engines were floated at the end of the decade.

These initiatives, combined with the political activism of the Vietnam era and a growing public awareness of environmental issues generally, helped set the stage for a first round of technology-forcing national mandates in 1970. Not coincidentally, 1970 was also the year that Earth Day was first celebrated in Washington D.C. and other cities around the nation. This event, which garnered substantial media interest, heralded the arrival of a national environmental movement that eventually spawned numerous advocacy organizations. Such organizations have played an important role in the evolution of air and environmental policy ever since. Politicians too responded to the surge of public interest. President Nixon, for example, spoke of a new federal research effort to promote alternatives to the internal combustion engine in his 1970 State of the Union address; later that year, lawmakers packed Congressional hearing rooms for lengthy technical debates about pollution, automobiles, and fuels.[30] By the end of 1970, Congress had created a new federal agency devoted to environmental protection (the EPA) and passed sweeping Clean Air Act Amendments that substantially tightened vehicle emissions limits in a relatively short (5 year) timeframe.

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[29] Interestingly, a previous finding by the same legal office found that, in the case of stationary sources, a "law is not invalid solely…because there are no appliances available to prevent the prohibited smoke or fumes". The seeming inconsistency may not be unrelated to the additional political difficulties of regulating a dispersed source with powerful relevance to many individual lives.

[30] Activists at the time speculated that a desire to focus on something other than the Vietnam War contributed to this high level of interest. In any case, observers noted that lawmakers became educated to an unusual degree about the scientific and technological complexities of the issue at hand. (Ayers, 2000)

As soon as the 1970 Amendments passed, the automobile industry petitioned EPA for an extension to the 1975 deadline. Sounding the kinds of grim predictions noted in the introduction to this chapter, manufacturers claimed that the standards were technologically impossible to implement in the prescribed timeframe and would cause massive economic dislocations. Interestingly, California supported automakers' initial request for an extension to the 1975 deadline,[31] testifying in 1972 that: "Since it genuinely appears that the auto manufacturers will not be able to comply with the federal standards within the allotted time, we find ourselves for the first time in 20 years to be almost in agreement with them. This is an incredible turn of events." (Jones, 1975, p. 258) Eventually, a chorus of protest from manufacturers, the oil industry, and other transportation interests – together with the findings of a National Academy of Sciences report (NAS, 1972)[32] that tended to confirm the technological difficulties automakers were experiencing – forced EPA to extend the 1975 implementation deadline.

Manufacturers, however, were seeking more than a year or two reprieve and continued to wage a concerted lobbying and public relations campaign against the emissions standards of the 1970 Amendments. As it turned out, several new political factors proved helpful to the industry's case. For one thing, the 1973 OPEC oil embargo created an instant energy crisis and shifted national attention to the problem of petroleum dependence. At the same time, Japanese automakers, with their smaller, more fuel-efficient products, were capturing a growing share of the American market, especially on the West Coast. The latter development lent a new political urgency to American manufacturers' claims about the intolerable costs of emissions regulation. Importantly, it also caused autoworkers and their powerful unions to align themselves, for the first time, in opposition to proposed environmental regulations. Meanwhile, manufacturers also argued that there was a direct conflict between improving fuel economy and reducing pollutant emissions. They asserted that relaxing emissions standards could reduce fuel consumption by billions of gallons and insisted that they could make no progress on fuel economy without a moratorium on further emissions control requirements.[33] Ultimately, the 1977 Amendments to the Clean Air Act further delayed and partially weakened the standards prescribed in 1970, though not to as great an extent as the industry wished.

A decade later, the political winds had shifted again: the energy crisis was over, the profits of American auto manufacturers had rebounded in the early 1980s, and people were becoming aware of stratospheric ozone depletion, acid rain, and a new problem called global warming. The *Exxon Valdez* oil spill in Alaska and the perceived anti-regulatory bias of the Reagan years had again heightened public sensitivity to environmental issues. All of these factors combined to set the stage, at the federal level, for a new round of sweeping Clean Air Act Amendments in 1990. Meanwhile, California regulators obligated to write an implementation plan for bringing Los Angeles into attainment with national air quality standards faced a seemingly Sisyphean task, sparking tense debates between CARB and South Coast District officials about the extent of future mobile source

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[31] California's concern stemmed from a nuance of the federal legislation, which allowed California to maintain its own emissions limits only if they were more stringent than federal requirements. The emissions standards proposed in the 1970 Amendments were actually more stringent than California's for CO and VOC, in part because California had made a trade-off between controlling for CO and NOx. California was concerned that it risked losing its unique ability to set state-specific standards, as well as its more stringent NOx limits, if the federal government pressed ahead with the 1970 standards.

[32] It should be noted that the NAS report was criticized at the time for relying on data supplied by the auto manufacturers. Nevertheless, its findings carried considerable weight in the context of the political debate.

[33] In fact, there was evidence that cars with catalysts in some cases had superior fuel economy.

controls. This period of intense pressure on California regulatory agencies inspired a number of ambitious proposals that went beyond more stringent tailpipe standards to promote alternative fuels such as methanol and to mandate the introduction of “zero-emissions” vehicles. Meanwhile, Northeast states, facing similar pressures to write attainment plans for their own heavily polluted urban areas, were contemplating the adoption of California’s more stringent auto emissions requirements. Together, the state and federal regulations set in motion in the late 1980s inaugurated a second round of auto emissions reductions as dramatic, in relative terms, as those achieved between 1970 and the early 1980s.

Even a very brief review of the politics of automobile regulation over the last three decades suffices to illustrate the importance of several key factors in the regulatory process. One of the most important factors, clearly, is the support or opposition of affected industries. Vehicle regulations generally faced strong opposition from auto manufacturers and frequently from oil companies as well, but it is worth noting that independent pollution control equipment manufacturers often played an important role in support of emissions limits. Similarly, foreign manufacturers sometimes affected the debate simply by demonstrating that lower emissions could indeed be achieved in mass production vehicles. Overall, of course, industry opposition tended to outweigh industry support, and thus several additional factors proved important to advancing the regulatory process at key junctures. These included a level of public concern and interest sufficient to get the attention of politicians, the accumulation of a critical mass of scientific consensus as to the problem being addressed, the existence of a basic level of knowledge and understanding among legislators and regulators, and finally the leadership of key individuals in the policymaking process.

E. History of Technology Development

Automobile emissions are a function of both fuel and vehicle design, and achieving substantial reductions has necessitated improvements in all aspects of the fuel-engine system. To comply with modern emissions limits, modifications to gasoline have been combined with sophisticated engine controls, three-way catalysts, and other engineering innovations. Interestingly, many of the technologies that have been used to comply with automobile emissions standards were known long before standards required their use. However, it was generally not until standards existed that these technologies were adapted, refined, and mass-produced for use in automobiles. This section describes how the technologies available to reduce auto emissions evolved with – and largely in response to –legislative and regulatory drivers.

The chief automobile pollutants of concern, from the standpoint of the last three decades of environmental regulation, are VOC, NO_x, and CO. Both VOC and NO_x are regulated primarily because they play a critical role in the formation of ground-level or tropospheric ozone.[34] Ozone is a powerful oxidant, known to impair lung function, cause eye irritation, and damage materials, vegetation, and crops. The photochemical conditions conducive to ozone formation require sunlight; hence, ozone smog is primarily a summertime phenomenon. Gasoline, which consists of a complex mixture of hydrocarbons, creates VOC emissions both when it is burned and when it evaporates.

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[34] As distinct from stratospheric ozone which shields the earth’s surface from the harmful effects of ultra-violet radiation. Ironically, both the human-induced formation of tropospheric ozone and the human-induced depletion of stratospheric ozone emerged as significant environmental concerns in the last half of the 20th century.

Emissions of NO_x, by contrast, are formed from nitrogen and oxygen in the air only at the high temperatures present at combustion. Carbon monoxide is a product of incomplete combustion; it can impair motor skills and, at higher levels, interferes with the oxygen-carrying ability of blood. With a few exceptions, carbon monoxide has receded as a pollutant of concern in most parts of the United States. However, ozone smog levels in most major metropolitan areas remain above current health-based standards.[35] Hence, achieving continued VOC and NO_x reductions remains an important priority for air quality regulators throughout the country. In addition, certain constituents of gasoline VOC emissions are known or suspected of being toxic, while NO_x emissions are also implicated in acid deposition, the eutrophication of sensitive water bodies, the formation of fine particles, and visibility impairment (i.e. haze).

The amounts of pollutants formed in the combustion process depend on the air-fuel ratio in the combustion chamber, the timing of spark plug firing, and even the geometry of the engine cylinders. CO and VOC emissions tend to be highest when an automobile engine is operating under fuel-rich conditions (i.e., when the air-fuel ratio is too low for complete combustion). These conditions typically occur at start-up and idle in older vehicles, or at times of acceleration. To minimize CO and VOC emissions, the engine should run with exactly the air-fuel ratio needed for complete combustion or slightly lean (i.e., excess air). Unfortunately, these are also the combustion conditions that tend to maximize NO_x emissions. Given the many different demands put on an automobile engine in the course of normal driving, maintaining optimal combustion conditions for minimizing all three types of pollutants presented a significant engineering challenge in the 1960s and 1970s. In addition, it was necessary to control the evaporation of gasoline from the fuel tank, fuel lines, and engine and during refueling. Uncontrolled evaporative emissions can be substantial and during the 1960s accounted for as much as 40 percent of total vehicle hydrocarbon emissions.

As noted previously, the first widely adopted add-on control technology – positive crankcase ventilation (PCV) – involved a simple device to reduce evaporative emissions from the crankcase. The technology was well known and had been implemented on military vehicles as early as the 1940s. Automakers first introduced it to passenger cars in the early 1960s as California regulators began signaling the need for emissions limits (Krier and Ursin, 1977, pp. 135-139; 146-148).

Meanwhile, several non-automobile manufacturers, seeing a potential California market, began exploring catalyst technologies in the early 1960s and by 1964 the California Motor Vehicle Pollution Control Board was able to certify four experimental new pollution control devices. Three were catalytic mufflers, one was a direct flame afterburner; none had been developed by the ‘Big Three’ (i.e. Ford, Chrysler, and General Motors) which continued to resist the use of such technologies. To forestall catalyst requirements, Chrysler instead began to promote a “Clean Air Package” of measures consisting primarily of carburetor adjustments and timing changes. Ford and General Motors followed suit with air injection modifications. All three companies argued that these modifications obviated the need for add-on controls and were more cost-effective.[36] Faced with their unanimous opposition, the Control Board opted not to require add-on control devices; subsequently, most (though importantly, not all) of the third-party manufacturers that had begun catalyst R&D efforts shelved their programs.

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[35] Non-compliance with the ozone standard would likely worsen under the new 8-hour ozone standard recently proposed by EPA. The revised standard is currently under court review.

[36] Unfortunately, these types of modifications also turned out to be of dubious long-term value, since they were highly susceptible to in-use deterioration and tampering.

During the 1960s, automakers continued to develop evaporative controls and worked to refine the control of air-fuel ratios under different operating conditions. These remained the primary strategies for reducing emissions throughout the 1960s and formed the basis for industry's claims, near the end of the decade, that "We have turned the corner as far as motor vehicle emissions are concerned."[37] Or, as the head of Chrysler's air pollution engineering program put it, "We've done the job, what next?" (Bedingfield, 1970).

The next major advance in emissions control would be the exhaust gas catalyst, a technology that was first conceived at the turn of the century and was already in use at petroleum refineries and coal mines by the 1950s. Basically, this technology consists of a reaction chamber in which exhaust gases are exposed to a catalyst that converts them to more benign chemical forms.[38] Ford Motor Company demonstrated a crude vanadium pentoxide catalytic device for automobiles as early as 1957, but this development was not welcomed by other manufacturers and the technology was quickly shelved. Indeed, for most of the 1960s, automakers showed little interest in advancing catalyst technology. On the contrary, a 1969 anti-trust suit brought by the federal government and the State of California charged the 'Big Three' with active collusion to suppress catalyst technology under cover of the joint research effort they had begun in 1953. The suit was eventually settled by a consent decree that ended manufacturers' ability to collaborate, but also sealed forever the court records and government evidence collected in the case (Aplet and Meade, 1997).

By the early 1970s, the adoption of progressively more stringent standards in California and under the 1970 federal Clean Air Act Amendments had revived interest in catalytic converters. The technology had continued to advance in the late 1960s thanks to the efforts of at least one third-party manufacturer, Englehard Industries. Though automakers continued to find fault with catalysts at EPA's 1972 hearings on the need for an extension of the federal standards, General Motors eventually committed to equipping most of its cars with catalysts by 1975.

An extremely important step that made widespread catalyst use feasible was the mandated phase-out of leaded gasoline that began in 1972. Leaded gasoline severely impaired catalyst effectiveness; in addition its use posed significant public health concerns. The introduction of unleaded gasoline in the 1970s thus provides a first important example of the co-evolution of fuel and automobile requirements in the quest for reduced emissions.[39] Meanwhile, foreign automobile manufacturers were demonstrating that it was possible to meet the standards that had been proposed for 1975 even without catalysts by using alternative engine designs, such as the stratified charge and rotary engine (Grad, et al., 1975).[40]

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[37] GM vice president Harry Barr as cited in Doyle, 2000, p. 25.

[38] Chemically, a catalyst is a substance that promotes chemical reactions without itself being consumed by those reactions. A typical example in automobile applications is the platinum-iridium catalyst.

[39] The addition of lead (in the form of tetra-ethyl lead) to gasoline was initiated by General Motors in 1922 to alleviate problems with knocking in the higher compression engines being introduced at the time. Public health concerns were raised almost from the beginning but a phase-out of lead did not get underway until the 1970s. Lead use continued much longer in Europe and continues to this day in some countries. The story of how lead came to be added to gasoline, despite widespread awareness of its toxicity, is itself an interesting one that says much about interplay of public and private interests in the political arena.

[40] Brock Yates notes that in 1974 Honda "borrowed" a new Vega from General Motors and returned it six weeks later with a new cylinder head that gave the car better performance and enabled it to meet the proposed 1975 emissions standards (Yates, 1983).

The next important technological challenge faced by auto manufacturers was controlling NO_x. The first technology used for this purpose, exhaust gas recirculation (EGR), was introduced in 1973 in response to pending NO_x standards. It essentially consisted of a tube and valve to recirculate some exhaust back to the intake manifold. Like PCV technology it was relatively simple and cheap and is still in use today. However, the more stringent NO_x requirements that began to come into effect in California in the late 1970s and nationally in the early 1980s required a more sophisticated approach. Anticipating these requirements, catalyst manufacturers began developing a three-way catalyst that could perform both the oxidation processes needed to destroy VOC and CO and the opposite reduction processes needed to destroy NO_x. From a technology standpoint, three-way catalysts added substantial complexity to emissions control systems because they required precise control of the oxygen present in the exhaust stream. Nonetheless, Volvo introduced this technology to meet California's stringent new NO_x and VOC standards as early as 1977 and by 1986, when a 1 gram per mile NO_x standard went into effect nationally, three-way catalysts had begun to be widely used.

The development of the three-way catalyst was eventually linked to other technological improvements, including oxygen sensors and sophisticated microprocessor controls. Another important advance, from an emissions control perspective (as well as from a fuel economy and performance perspective) was the replacement of carbureted engine designs with direct fuel injection.[41] Fuel injection was first developed in Germany in the mid-1930s and demonstrated on a Mercedes Benz sports car as early as 1954. Even though European and Japanese cars used fuel injection by the late 1950s and 1960s, American manufacturers did not begin exploring this technology until the late 1970s. By the late 1980s, fuel injection had largely replaced carburetion in American vehicles. More recent advances have enabled manufacturers to improve further on the emissions limits achieved in the 1980s. These advances have included microprocessors capable of adaptive learning, additional engine valves, fuel temperature sensors, more sensitive oxygen sensors and the development of "on board diagnostic" (OBD) systems to identify problems and alert drivers to malfunctioning components.

Increasingly stringent standards have also forced manufacturers to develop strategies for reducing start-up emissions. Because catalysts are optimally effective only at higher temperatures, a disproportionate share of emissions occur when a cold engine is first started. In the early 1990s it was widely assumed that manufacturers would have to find a way of electrically pre-heating catalysts to meet California's LEV standard. This was perceived as a significant challenge, from both a cost and consumer acceptability standpoint. Ultimately, pre-heated catalysts proved not to be necessary to meet the LEV standards. Indeed, by 1995 Honda certified an Accord to California's even more stringent ULEV standards, largely by improving on the catalyst and moving it closer to the engine so it would warm faster – all at an incremental cost of just $200-300. Many manufacturers had assumed that such low emissions rates were unattainable with a gasoline-powered vehicle and had projected much higher costs to build alternative fueled vehicles to meet ULEV standards. More recently, Honda has developed another novel approach in which an absorbing

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[41] Air-fuel ratios can be controlled much more precisely when the fuel is injected directly to the combustion cylinder. In a modern vehicle, an oxygen sensor typically monitors the exhaust stream prior to reaching the catalyst and provides continual feedback to a microprocessor in the engine compartment. The microprocessor in turn makes fine adjustments to the rate of fuel injection in each cylinder, creating a "closed loop" of information feedback and engine control.

medium captures excess CO and VOC emissions at start-up and retains them until the catalyst has reached operating temperatures (Kishi et al, 1999).

Substantial advances, meanwhile, have also occurred in the area of evaporative emissions control. These were achieved, to a significant extent, through modifications at the filling station,[42] as well as through changes in vehicle design. More recently, attention has refocused on reducing and even eliminating evaporative emissions from vehicles. Federal regulations will mandate on-board controls for vapor recovery at refueling after 2000, and Honda has developed an alternative to carbon canisters for this purpose (Kishi et al., 1999). In addition, further modifications have recently been prompted by the discovery that years of vibration can produce tiny leaks in engines, fuel lines, and other components. Such leaks can become a major source of remaining hydrocarbon emissions from modern vehicles, and efforts are being made to reduce or eliminate their occurrence.

In fact, recent efforts to reduce the deterioration of emissions controls have proved almost as important as the development of the controls themselves. The fact that emissions performance tended to decline in older vehicles was perhaps not surprising. But as late as 1990 it appeared that even newer cars could produce surprisingly high emissions – far in excess of the regulatory limits – if some component of the emissions control system failed or was tampered with (Calvert et al, 1993). The introduction of inspection and maintenance (I&M) programs in many non-attainment areas was, of course, one response to this issue. In addition, recent regulations increasingly require that stringent emissions limits be maintained up to 50,000 and even 100,000 miles. As a result, manufacturers have improved the durability of catalysts and other key components (such as oxygen sensors), and developed on-board diagnostic systems aimed at detecting malfunctions. These efforts appear to be reducing typical in-use deterioration rates in the vehicle fleet. For example, a 1998 study by the University of California, Riverside found an emissions control failure rate of about 2 percent per year for 1985 to 1995 automobiles (Barth, 1998). More recent data, though still incomplete, indicate failure rates of 0.5 to 1 percent for post-1995 vehicles (McClintock, 2000).

Besides regulating vehicle emissions, both California and the federal government have mandated changes to gasoline aimed at reducing pollutant emissions. The 1990 Clean Air Act Amendments established the use of federal reformulated gasoline, or RFG, in air quality non-attainment areas. California, meanwhile, maintains its own reformulated fuel requirements. The federal RFG program required refiners to develop a lead-free gasoline that contained reduced levels of benzene (a potent air toxic); did not evaporate as readily as conventional gasoline (thereby reducing hydrocarbon emissions); and contained 2 percent oxygen. The latter requirement was originally intended to help alleviate carbon monoxide emissions but it has recently come under scrutiny because of mounting concern about the water quality impacts of MTBE, the additive which has been extensively used to fulfill the oxygenate mandate. EPA recently took an important next step

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[42] Today, most filling stations employ "Stage I" and "Stage II" vapor recovery to capture gasoline fumes from tanker trucks and individual automobile gas tanks, respectively. Stage II vapor recovery was first tested in San Diego, California in 1972 but it was not introduced elsewhere in the country until well into the 1980s. Because it involves extensive modifications to service stations, Stage II became the subject of an intense controversy pitting automakers against oil companies and state and local regulators against their federal counterparts. The oil companies, seeking to shift costs to automakers, argued for on-board vapor recovery systems. They were supported in this position by EPA, but opposed by state and local air officials who, for once, found themselves aligned with the car companies. States and localities, under pressure to write attainment plans, needed the immediate emissions reductions that Stage II would provide. (By contrast, requirements for on-board systems could only be applied to new cars.) Moreover, they anticipated that Stage II would be unpopular with the public and felt they needed a federal mandate to implement it.

in fuels regulation by proposing substantial further reductions in the sulfur content of gasoline and diesel fuel. Much as the removal of lead was critical to the introduction of catalytic converters, this step is necessary to open the door to further improvements in emissions control technology. Not surprisingly, petroleum refiners and automobile manufacturers have found themselves on opposite sides of this latest regulatory development.

In sum, the technological hurdles to reducing vehicle emissions by as much as two orders of magnitude proved to be formidable, but ultimately surmountable. That surmounting these hurdles necessitated the simultaneous regulation of two large and politically powerful industries – automobile manufacturers and oil companies – only makes this accomplishment more impressive.

F. Cost Issues

Most discussions about the aims and merits of particular regulations come down at some point to a question of cost. How much it will cost to achieve a societal objective, whether the benefits of doing so justify those costs, and – perhaps as importantly – who will bear the costs, are questions found at the heart of almost all political and regulatory debates. In the case of technology-forcing regulations, cost almost always represents an area of uncertainty – and hence an area of controversy – because technologies are involved that have not yet been commercialized. Typically, industry's cost estimates are higher than those of regulators and environmentalists and typically (though not always) control costs prove to be lower than first anticipated.

Certainly, cost arguments have always figured prominently in auto manufacturers' objections to new rounds of emissions regulation. As has already been noted, dire predictions were made in the early 1970s when the 'Big Three' warned that new federal standards, then due to take effect in 1975, would put them out of business altogether. Coming from an industry that accounted for about one-sixth of the gross national product and directly or indirectly employed millions of citizens, this was a powerful claim. In terms of cost per vehicle, a Chrysler ad in the early 1970s claimed it would cost the consumer an extra $1,300 to own and operate a car under the 1975 standards; similarly, Ford hypothesized that the cost of its Pinto might rise from $2,000 to $3,000 rendering it uncompetitive with cheaper Japanese imports. A 1972 report issued by the White House Science Office put the cost of the 1975 standards at a more moderate $755 per vehicle, but still concluded that this expense was too high (Doyle, 2000, p. 77 and 92).[43]

Ultimately, of course, the 1975 standards were delayed and (in the case of NO_x) relaxed; hence, there is no firm data on what it would actually have cost to comply in the original timeframe. However, information compiled by the Automotive Consulting Group (ACG), an industry-sponsored organization, on the cumulative price of regulations up to 1991 provides an interesting point of comparison (ACG, 1993). ACG's figures indicate that the cumulative cost of all regulations going back to 1968 added $1,495 (in 1991 dollars) to the average retail price of an automobile in 1991.[44] This figure is considerably less (in constant dollars) than those being cited in the 1970s and compares to an average increase of approximately $4,500 in inflation-adjusted overall retail car prices over the same time period (again, in 1991 dollars) (Davis and McFarlin, 1996).

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[43] At this time, the Nixon administration was supporting manufacturers' calls for extensions and other modifications to the pending standards.

[44] According to ACG, safety regulations added another $1,087 in incremental costs over this same period.

Industry's tendency to overestimate the cost of meeting emissions requirements was borne out more recently in the context of California's LEV (low emission vehicle) program. Table II-2 compares per vehicle cost estimates at various points in time from the California Air Resources Board (CARB) and various industry sources for California's different categories of low emission vehicles. These projections may be compared against the actual implementation costs shown in the last column to the right.

Table II-2: Estimated Costs of California's LEV Program

Vehicle	CARB '94	CARB '96	ACG '93	AAMA '94	Actual
TLEV	$66	$72	$273	$298-487	$35
LEV	$120	$120	$788	$911-1343	$83
ULEV	$227	$145	$679-1,326	$1,666-4,005	$251

Sources: The CARB (California Air Resources Board) and AAMA (American Automobile Manufacturers Association) figures are taken from Cackette, 1998; the ACG (Automotive Consulting Group) figures are found in ACG, 1993 (see references at the end of this chapter).

A similar tendency to overstate future regulatory costs can be found when one examines past cost projections on the fuels side of the vehicle equation. In 1991, for example, CARB estimated that California reformulated gasoline would cost an additional 12 to 17 cents per gallon (¢/gal); the petroleum industry's worst case projection was 23¢/gal. In 1996, CARB revised its estimate downward to a range of 5-15¢/gal, with a mid-point estimate of 10¢/gal. A year later CARB checked the price of California RFG against the price of gasoline sold in several cities outside the state and concluded that the additional cost was in fact closer to 5.4¢/gal (Cackette, 1998).

Claims of imminent industry shutdown, such as were heard in the 1970s, have not accompanied more recent emissions standards; nevertheless, industry projections of dramatic economic impact and job losses as a direct result of regulatory action have not been uncommon. Such projections are usually based on industry's estimates of the incremental cost to meet new standards. Incremental costs are assumed to result in an equivalent reduction in consumer spending, which when subjected to the multiplier effect of top-down macro-economic models, results in seemingly large negative impacts.

In the early 1990s, for instance, when several northeastern states were moving to adopt California's LEV program and expand the use of reformulated gasoline, industry warned that the region would suffer dramatic job and income losses. A 1991 analysis, sponsored by the American Petroleum Institute (API), estimated that New York State alone would lose 28,000-66,000 jobs in the year 2000 if it adopted California's LEV program and required gasoline similar to California's Phase II RFG (DRI/McGraw Hill, 1991).[45] These losses were on top of the 15,000-25,000 jobs estimated to be lost in the study's base case, which assumed use of federal RFG only where it was already mandated (i.e. in the New York City metropolitan area). In addition, the study estimated that New York's economy would suffer $1.7-4.5 billion in lost wages and salaries; again this was in addition to an estimated $1.1-1.7 billion impact from the base case alone. These results were premised on

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[45] At the time, auto manufacturers were arguing that northeastern states would have to adopt California fuel as well as California cars to get the emissions benefits of the LEV program and to maintain the legal standard of identicality with California's program.

incremental costs for Phase II California RFG of 16-24¢/gal; incremental costs for LEV and ULEV vehicles of $250-$1,100 per car; and incremental costs for ZEVs of $2,700-$7,500 per car. In hindsight, these assumptions were clearly too pessimistic. New York as well as Massachusetts eventually adopted the California program and neither shows signs of suffering the kinds of economic impacts forecast in the 1991 API study. On the other hand, cost and other arguments did win the auto manufacturers a reprieve from California's ZEV mandate; consequently, the 2 percent sales requirement by 1998 was never enforced in California or the Northeast. The ZEV mandate was always the component of California's program which industry found most objectionable; presumably it also accounted for a sizable part of the job and income losses being claimed as a result of that program.

In sum, debates about future costs are subject to inherent uncertainty and therefore tend to be as contentious and subject to interpretation as the scientific debates described earlier in this chapter. An additional complexity arises from the fact that costs are often difficult to allocate among specific pollutants and control technologies. For instance, many control technologies for cars and other pollution sources reduce emissions of more than one pollutant; moreover, they sometimes have other, non-pollutant benefits. The sophisticated microprocessors now common in modern vehicles are used to provide increased passenger comfort and improved power and performance, as well as to reduce emissions. In these cases, it is difficult to allocate the costs of a technology among its various benefits.

G. Setbacks and Problems

Though the story of automobile emissions regulation must be viewed, on the whole, as one of impressive technological success, the progress achieved over three decades of regulation was not without setbacks and unanticipated problems. As with many aspects of the scientific, political, and cost debates, the experience in this case was not atypical of most regulatory processes.

For example, the exhaust gas recirculation (EGR) technology introduced in the early 1970s had the effect of reducing engine power. Increasing engine size eventually solved this problem, but in the early days of EGR it resulted in numerous owners removing the device and modifying their vehicle to run rich, thereby substantially increasing emissions. Other modifications involving air-fuel mixture and timing were also vulnerable to tampering by consumers interested in improving vehicle performance. Early catalysts, meanwhile, could be quickly 'poisoned' by the use of leaded gasoline, which was still widely available when catalysts were first introduced and generally cheaper than the unleaded alternative.[46] As a first line of defense, vehicles with catalysts were designed to accept only the smaller nozzles used at unleaded gas pumps, however illegal nozzle adapters were sometimes used to circumvent this safeguard. The lead problem eventually disappeared with the full phase-out of leaded gasoline, but similar problems of consumer acceptability later accompanied the introduction of Stage II vapor recovery at many filling stations.

More troubling still was the discovery in the early 1970s that cars being certified to meet existing emissions standards were not necessarily representative of the cars rolling off the assembly line. Because manufacturers provided the emissions data and performed the tests used to certify

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[46] Early catalysts, which became very hot in operation, were also blamed for starting occasional brush fires.

vehicles, test vehicles could be fine-tuned to just meet the standards. Another problem was the inclusion of 'defeat devices' that either overrode emissions control systems under common driving conditions or in some cases invited subsequent tampering. Finally, faulty design affected some control system components, resulting in failure after relatively short periods of use. In response to these types of problems, EPA began performing random assembly line spot-checks in 1977 and continued to conduct tests of vehicles in use. To correct the problems it found, EPA was authorized under the 1970 Clean Air Act Amendments to order manufacturers to recall vehicles that were found to have defective emissions control equipment. Over the last three decades, millions of vehicles have been recalled for such defects and EPA's ability to order recalls has proved crucial to the effective enforcement of automobile emissions regulations.

While automobiles did not always perform to regulatory requirements, the fact that the requirements themselves did not always hold up perhaps qualifies as a different type of setback. As has already been described, the federal emissions limits originally adopted in the1970 Clean Air Act Amendments were substantially relaxed (in the case of NO_x) and/or delayed under overwhelming political pressure. More recently, California's regulators have had to introduce flexibility and lift near-term sales mandates in the context of the state's ZEV program. In this case, the failure of battery technology to advance as quickly as had been hoped necessitated these changes. Thus, while mandates have often proved capable of producing dramatic results, the ability and willingness of regulators to provide flexibility, when appropriate, has also proved to be an important factor in the long-term success of regulatory efforts.

H. Conclusions

As was acknowledged in the Introduction to this chapter, automobile exhaust presents a very different control challenge than power plant mercury emissions. Some features of the power plant situation may make it both easier and more difficult to introduce controls than was the case with cars. On the one hand, automobile controls almost always involved new installations and not retrofits; hence they could be introduced and refined over successive model years.[47] On the other hand, the fact that these technologies needed to be mass produced, applied to literally millions of vehicles, and operated reliably over a range of conditions outside the manufacturers' control undoubtedly presented unique technological challenges. Despite these obvious differences, the pattern of technological response to regulatory drivers that emerges from a review of automobile emissions control is echoed in numerous other examples of environmental regulation, including the two power plant examples that follow (Porter, 1991).

In fact, the automobile case suggests that regulation and technology development typically proceeds along a relatively predictable path. For example, regulation is usually preceded by several years of scientific debate in which industry tends to dispute initial findings and then to argue that scientific uncertainty undermines the case for action. As the scope for scientific argument diminishes, this debate is eventually overtaken by a political and regulatory debate in which the more salient arguments concern the availability of control technologies, cost vs. benefit, the extent and timing of remedies, the distribution of costs among affected interests, and the need for mandates.

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[47] As the subsequent case studies show, the slow turnover of power plants generally necessitates retrofit applications to achieve meaningful levels of overall emissions reductions.

At this point, industry may seek to dissipate regulatory momentum by offering some partial remedies or voluntary actions. In cases where regulation is nevertheless required, and eventually adopted, industry has with few exceptions, risen to the challenge, and not just in the case of automobile emissions. For example, manufacturers argued for two years that it would be impossible to comply with proposed new standards for barbecue lighter fluid in California; once the regulations were adopted, manufacturers complied within 90 days. In some cases, regulatory efforts to promote new technologies provoke sudden improvements in existing ones. In 1989, for instance, a dominant southern California petroleum supplier, the Atlantic-Richfield Company (ARCO), introduced an early version of reformulated gasoline. It was probably no coincidence that California regulators at the time were contemplating a major shift away from gasoline and toward alternative fuels, especially methanol. More recently, efforts in Los Angeles to promote the conversion of government-owned diesel vehicles to natural gas have prompted industry to implement highly efficient particulate traps to reduce diesel emissions.

It is, of course, difficult to say what would have happened in the absence of past regulatory mandates or under a different regulatory approach. The parallel development (and subsequent stagnation) of automobile fuel economy regulations perhaps provides an illuminating comparison. Corporate Average Fuel Economy, or CAFE, standards were first introduced in the 1970s in response to national security concerns generated by that decade's oil crises. These standards resulted in no less than a doubling of the overall fuel efficiency of light duty vehicles over the next decade. As concerns about oil dependence faded, fuel economy standards were left unchanged.[48] Thus, in stark contrast to the continued progress that was achieved in reducing criteria pollutant emissions, there has been no improvement in average automobile efficiency since the mid-1980s. Indeed, fleet fuel economy has actually declined recently thanks to the current popularity of trucks and sport utility vehicles.[49] Combined with current levels of population and travel, the U.S. is now more petroleum dependent than ever.

In short, there can be little doubt that regulation was substantially responsible for the significant gains in automobile emissions control that have been implemented since the 1960s. Beyond what has already been achieved, recent technological advances – including, notably, some of the research and development prompted by California's envelope-pushing ZEV mandate – are laying the groundwork for a next generation of vehicles that are even cleaner, much more efficient, and can eventually run on fuels other than petroleum. Whether and when these vehicles of the future arrive depends in no small part on the continued evolution of state and federal policies, which will increasingly need to respond to environmental concerns beyond ozone formation. Meanwhile, the successful history of automobile emissions control to date provides an instructive, and even inspiring, example of the potential for technology-forcing mandates to provide solutions where they did not seem to exist before. In the case of automobiles, of course, some of the most important technology solutions – such as catalytic converters – existed in experimental form long before regulations were introduced. However, it was not until mandates were adopted that these

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[48] Concerns about global climate change have provided a new justification for revisiting fuel economy regulation, but so far the political consensus for addressing this issue has not matched the formidable opposition to fuel economy regulation that now exists in Congress.

[49] Ford's recent announcement that it will voluntarily improve the fuel economy of its sport utility vehicles and trucks by as much as 25 percent and GM's subsequent assertion that it will match any improvements achieved by Ford, may signal a change from the status quo. If so, it would be interesting to understand the corporate motivations behind these announcements, given the absence of immediate regulatory pressures to improve fuel economy.

technologies were widely adapted and refined to produce the dramatic emissions results common in today’s automobiles.

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III. The Regulation of NO_x Emissions from Coal-Fired Boilers: A Case Study

A. Introduction

Oxides of nitrogen (NO_x), like sulfur dioxide, are a major focus of current environmental regulations applicable to electricity generating boilers. Efforts to limit NO_x emissions date back to the early 1970s, but only in the past decade or so has regulation of this pollutant been sufficiently aggressive as to compel major advances in available control technologies for existing boilers in the U.S. Like both other case studies presented in this report, the history of NO_x controls suggests a strong link between regulation and technology development. Prior to 1990, NO_x requirements were largely limited to new or substantially modified sources; in addition, these requirements were established with express reference to control technologies that had already been "adequately demonstrated." As a result, control levels of just 30 to 40 percent were generally achieved, in most cases by using combustion modification technologies such as low- NO_x burners. Only after the 1990 Clean Air Act Amendments placed a greater emphasis on NO_x control from existing boilers as a means of attaining ambient ozone standards and reducing acid deposition, did more sophisticated post-combustion technologies begin to come into widespread use.

The pattern of NO_x control technology development differs somewhat from that of the other two case studies presented in this report. In the automobile example, successive rounds of technology-forcing regulation by the State of California and the federal government spurred a variety of technological advances by American and foreign manufacturers from the late 1960s through the 1990s. In the SO_2 example, regulations imposed since the 1970s have led to continual improvement of one primary control technology: scrubbers. In the NO_x case, by comparison, relatively weak regulation for the two decades from 1970 to 1990 was followed by the imposition of substantially more stringent requirements in the early 1990s. Even these more recent requirements were not, strictly speaking, "technology forcing"; rather they were based on known technologies that had been developed and demonstrated overseas, first in Japan and then in Germany. However, it was still necessary for American companies to adapt and implement these technologies on a large number of boilers in a relatively short timeframe. To date, this process of technology diffusion has operated with remarkable speed and success. In the past five years, technologies capable of reducing NO_x by as much as 90 percent and even higher have been demonstrated in the U.S. This represents more than doubling of the reduction levels typically achieved in the previous two decades.

At present, NO_x control requirements are driven primarily by concerns over ozone and acid deposition. In addition, airborne NO_x emissions from power plants contribute to fine particle formation and visibility impairment (haze) and have been implicated in the eutrophication of important aquatic ecosystems, such as the Chesapeake Bay. Regulation of NO_x as an acid rain precursor is relatively recent, having been introduced, together with major SO_2 reductions, under Title IV of the Clean Air Act Amendments of 1990. The most stringent current and pending regulations for NO_x, however, are linked to ozone attainment strategies. These too are relatively recent, due to a belated recognition by the U.S. EPA of the importance of NO_x as an ozone precursor. Early ozone reduction strategies placed a far greater emphasis on the control of hydrocarbons. Moreover, the structure of the Clean Air Act itself was premised on a localized view of air pollution; thus, it was not surprising that regulators responsible for remedying air quality violations in urban areas tended to focus on automobiles and other proximate sources rather than on large power plants out in the countryside.

More recently, the need for substantial, broad-based NO_x reductions to combat continued ozone problems, especially in the eastern U.S., has been more widely accepted by scientists and policymakers alike. In 1994, the northeastern Ozone Transport Commission adopted a phased program aimed at achieving NO_x reductions of 75 percent from uncontrolled levels by 2003 for all electricity generating boilers. Four years later, EPA -- using its authority to regulate the interstate transport of ozone under Section 110 of the Clean Air Act -- moved to require similar reductions by 2003 from boilers throughout a larger 22-state region east of the Mississippi. EPA's action is currently being litigated, nevertheless, the further regulation of NO_x emissions from existing boilers throughout the U.S. now appears to be only a matter of time.

Importantly, NO_x control became a priority much earlier in California and overseas. This was partly because Los Angeles was the only area of the country to violate ambient air quality standards for nitrogen dioxide (NO_2). In addition, by the late 1970s California regulators had recognized the importance of reducing NO_x as well as hydrocarbon emissions in order to lower ambient ozone levels. For example, the 1979 ozone attainment plan for the Los Angeles area included NO_x emission reductions of approximately 40 percent in addition to 80% reduction in hydrocarbon emissions. In Germany and Japan, meanwhile, large sources of NO_x were directly regulated and required to meet stringent, technology-forcing NO_x limits by the 1980s. This contrasts with the U.S. situation, where, with the exception of automobile tailpipe standards, regulation of ozone precursors has largely occurred via states' obligation to develop State Implementation Plans (SIPs) for achieving national ambient air quality standards. For the reasons noted above, this mechanism failed to provide strong regulatory drivers for NO_x until well into the 1990s. Thus, much of the advanced control technology now being tested and applied in the U.S. was originally demonstrated in Japan and Germany in the late 1970s and 1980s.

The remainder of this chapter describes the major milestones in NO_x regulation and control technology development over the last three decades. For purposes of convenience, the discussion is separated into two periods: pre-1990 and post-1990. Because many NO_x regulations in the U.S. have been "technology following" – in the sense that they are based on regulators' estimates of what could be achieved with known or experimental control technologies – an overview of the main technology options precedes a detailed discussion of the regulatory and technological advances of each period.

B. Technology Overview

NO_x emissions are a byproduct of fossil fuel combustion and are formed when the nitrogen contained in coal or oil, together with nitrogen in the atmosphere, combines with oxygen at the high temperatures present at combustion. The amount of NO_x emitted from a given boiler depends on a variety of factors and is crucially influenced by the properties of the fuel and by specific parameters of the combustion process. Thus, similar boilers firing different fuels or similar fuels burned in different boilers can yield substantially different NO_x emissions rates. The many factors involved in NO_x formation allow for a variety of approaches to reducing emissions. Broadly speaking, it is possible to distinguish these approaches in terms of combustion modifications vs. post-combustion or "flue gas" controls. Combustion modifications attempt to reduce the amount of NO_x formed in the first place, typically by controlling the introduction and mixing of fuel and air in the boiler. Post-combustion technologies typically involve add-on controls that reduce the amount of NO_x exiting the boiler smokestack.

Before 1990, NO_x control in the U.S. was largely limited to the use of combustion modifications. These chiefly involved boiler tuning or optimization, low- NO_x burners (LNB), and overfire air (OFA). LNB and OFA, in particular, have been widely implemented. As a result, these technologies are available from several competitive vendors – many of which provide designs geared to particular boiler types – and are common at many power plants today.

"Tuning" or optimizing boiler combustion can typically reduce NO_x emissions by 5 to 15 percent. In cases where a boiler is especially "out of tune", higher percentage reductions may be achieved. This approach is usually highly cost-effective, however, achievable emissions reductions are fairly limited. By contrast, low- NO_x burners and overfire air involve dedicated technologies for reducing NO_x formation and can produce substantially lower emissions. Both operate through a process called air staging, in which the quantities of fuel and air introduced in the combustion chamber, and the way they are introduced and mixed, are carefully controlled. Low- NO_x burners and overfire air can be used separately or as a system and are capable of NO_x reductions of about 40 percent from uncontrolled levels. Specifics of boiler configuration (e.g., dry vs. wet-bottom, wall vs. tangential-fired), boiler age, and fuel type (e.g., bituminous vs. sub-bituminous coal) will influence the actual performance achieved.

A more recent approach that falls in the category of combustion modifications is natural gas (or coal) reburn technology. Reburn technology differs from the other control options in this category insofar as the combustion process is used to chemically destroy NO_x shortly after it is formed, as opposed to minimizing its formation. This is accomplished by introducing a second fuel (in most cases, natural gas) to the boiler above the main burner region in a process called "fuel staging." A unique feature of reburn technology compared to other combustion modifications is that the plant operator can adjust the level of achievable NO_x reduction by adjusting the flow of reburn fuel up to a typical maximum of about 20%; hence reburn is known as a "dial in" technology. Site-specific parameters such as initial NO_x levels, boiler size, availability of natural gas, and type and quality of the main combustion fuel are important determinants of the suitability of this approach for any given boiler. Achievable emissions reductions from reburn range from 35 to 60 percent; the technology is commercially available and used in both the U.S. and overseas. Variants of the basic reburn approach have recently been developed. For example, fuel-lean gas reburn (FLGR) has been developed for specific applications where NO_x reductions of 30 to 40 percent are desired; this approach uses less gas than conventional reburn (3 to 7 percent versus 15 to 20 percent on a heat input basis) and has substantially lower capital costs.

SNCR and SCR represent the primary post-combustion NO_x control options commercially available today. They are fundamentally similar in that both use an ammonia-containing reagent to convert the NO_x produced in the boiler to nitrogen and water. SNCR accomplishes this at higher temperatures (1700ºF-2000ºF) in the upper furnace region of the boiler. In contrast, SCR operates at lower temperatures (about 600ºF) by using a catalyst to produce the desired reaction between ammonia and NO_x. In practice, these differences mean that SNCR has lower capital costs and limited NO_x reduction capability (typically 30 to 40 percent but higher in some cases). SCR is more capital intensive but is capable of achieving much greater reductions (up to 90 percent and higher).

Many of the technology options summarized above can be used in combination, though the economics of doing so may not always be favorable. The choice of control technologies, whether alone or in combination, tends to be highly site-specific. However, several technology pairings have become relatively common. For example, SCR or SNCR are frequently used together with LNB/OFA. Another pairing is the use of reburn with SNCR, an approach that benefits from synergies in terms of location and boiler temperature. The combination of reburn and SNCR can yield NO_x reductions of 60 to 70 percent, better than either technology alone.

C. The Pre-1990 Period

C.1Regulatory Milestones

The introduction of New Source Performance Standards (NSPS) in the early 1970s constituted the first major regulation of NO_x emissions from power plants in the U.S. The NSPS program was established under the sweeping Clean Air Act Amendments of 1970; it required the then newly created Environmental Protection Agency (EPA) to develop maximum emissions limits for new and substantially modified sources. NSPS limits were to be based on the best system of emission reduction that EPA determined to be "adequately demonstrated" and were applied to all facilities built or substantially modified after 1971, regardless of their location.[50] In addition, the 1970 Amendments directed EPA to promulgate health-based National Ambient Air Quality Standards (NAAQS) for NO_2, ozone and other pollutants. These requirements eventually became important in the context of NO_x control, first because violations of the federal annual NO_2 standard in Los Angeles helped bolster the case for early NO_x reduction efforts in California, and later as widespread ozone non-attainment prompted a renewed focus on NO_x reductions.

The first NSPS for large fossil fuel steam generating units (boilers) was finalized in 1974. It established an emissions limit of 0.70 pounds per million Btu (lb/mmBtu)[51] of fuel input for most coal units, with higher limits for units that burned lignite coal. EPA promulgated this limit based on the reductions achieved in several tests of LNB technology on tangentially fired boilers and assuming that wall-fired units would perform similarly (EPA 1971). By 1977, the NSPS program had resulted in the successful application of LNB and other combustion modifications to a variety of boilers. Experience with these technologies led to improvements of as much as 30 percent in their control effectiveness and formed the basis for a subsequent revision of the NSPS in 1979. The revised NSPS lowered allowable NO_x emissions limits to 0.50-0.60 lb/mmBtu. The change was based on EPA testing at six boilers and applied to all facilities built or substantially modified after September 1978.

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[50] In other words, all new or substantially modified facilities are subject to NSPS regardless of whether they are located in a non-attainment area or not.

[51] Btu stands for British Thermal Unit. It is a common unit for measuring the input heat content of fossil fuels. Thus, early regulations were specified as an allowable quantity of emissions for a given quantity of fuel input. More recent regulations are sometimes expressed on an output basis, as discussed later in this chapter.

In addition to satisfying NSPS, combustion modifications generally sufficed to meet the BACT (Best Achievable Control Technology) requirements introduced under the Clean Air Act Amendments of 1977 to prevent significant deterioration of air quality in areas with clean air.[52] Nor were NO_x emissions subject to more explicitly technology-forcing regulatory requirements such as LAER (Lowest Achievable Emissions Rate), a standard that applied to new or modified sources located in non-attainment areas. Though non-attainment of the ozone standard was not uncommon in the 1970s, regulators were largely focused on hydrocarbon reductions at this time (except in California) and LAER was not generally applied to new NO_x sources outside California.

Substantial advances in NO_x control technology, meanwhile, were being made overseas by the late 1970s, especially in Japan. These included pilot scale applications of post-combustion controls such as selective catalytic reduction (SCR). Interestingly, EPA explicitly took note of Japanese experiments with SCR in its 1979 promulgation of NSPS revisions. However, the agency concluded that this technology had not yet been "adequately demonstrated" because it had not been implemented on a full-scale coal boiler. Just a year later, however, a first commercial installation of SCR technology was implemented in Japan, which subsequently implemented stringent NO_x limits based on SCR for most boilers.

By the late-1970s a lively debate had begun to emerge about the relative efficacy of controlling NO_x versus hydrocarbons emissions for purposes of ozone attainment in the U.S. California, in particular, became increasingly convinced that NO_x reductions were crucial to grappling with its severe ozone pollution problems in the Los Angeles basin and other areas of the state, and began to send experts to Japan for a first-hand look at SCR technology. Because California's power sector was dominated by natural gas and oil-fired boilers (as opposed to coal-fired boilers), the state's interest in NO_x reduction led to a number of control innovations, including flue gas recirculation (FGR), fuel injection recirculation (FIR), as well as combustion optimization and low- NO_x burner technologies for gas- and oil-fired boilers.

Meanwhile, concern about extensive pollution damage to forests in Germany and other European nations was creating interest in advanced NO_x control technologies from another quarter.[53] In 1984, the German Environment Ministry established a NO_x emissions limit for coal-fired power plants of about 0.12 lb/mmBtu. This limit, which was 76 to 80 percent lower than NSPS in the U.S., was to be met by 1990 at both new *and* existing facilities. Consequently much of the R&D activity on SCR technology shifted to Germany where a number of pilot studies were carried out between 1984 and 1987. German industry relied on these studies, together with lessons learned from SCR experience in Japan, in its rapid adoption of SCR technology during the late 1980s. To date, over 120 SCR systems have been installed on some 30,000 MW of German power plant capacity.

As the decade drew to a close, several issues relevant to NO_x regulation were gaining prominence in the U.S. The first was the continued violation of health-based ozone standards in

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[52] The 1977 Amendments created yet a third category of control requirements known as BART (Best Available Retrofit Technology). BART was introduced under the visibility provisions of the Act and was intended to apply to individual sources that contributed to visibility impairment in so-called "Class I" areas, i.e., national parks and wilderness areas. However, visibility protection measures were never strongly enforced and the BART requirement did not lead to substantial NOx reductions at many existing boilers.

[53] Concerns about forest die-off due to acid rain, or "Waldsterben", galvanized the German public in the mid-1980s and led to the formation of that nation's Green Party.

many urban areas of the country, including broad swaths of the densely populated Northeast. A second issue was the recognition that existing regulations -- which primarily affected new facilities -- had left the great majority of utility boilers throughout the country essentially uncontrolled. A third issue was acid rain, which had gained widespread public attention in the late 1980s. These issues, combined with a broader resurgence of interest in environmental issues more generally,[54] created the conditions for more aggressive regulation of power plant NO_x emissions in the next decade and beyond.

C.2Technology Challenges and Advances (Pre-1990)

The chief NO_x control approach used in the U.S. during the pre-1990 period was LNB technology, which was initially deployed as a result of NSPS requirements for new or substantially modified facilities. Such applications were relatively few in number, especially compared to the population of existing, pre-1971 power plants (these represented a total installed capacity of some 300,000 MW). As it became increasingly obvious in the late 1980s that existing boilers would have to be controlled to effectively reduce overall NO_x emissions, R&D efforts to develop retrofit LNB technologies gained momentum. Compared to implementing LNB for newer boilers, which could be designed to have inherently lower emissions, retrofitting LNB systems to older boilers posed a substantial engineering challenge. The first demonstration of LNB on a facility typical of pre-1971 units occurred in 1986 at West Pennsylvania Power's 626 MW Pleasant Station and was designed to achieve emissions levels of about 0.4 lb/mmBtu (Offen et al. 1987; Vatsky et al. 1987).

By 1989, at least 12 major demonstration projects involving LNB, OFA, and reburn technologies were underway or in the planning stages in the U.S. (EPRI-EPA NO_x Symposium 1989). Much of this activity was sponsored by federal agencies and involved the utility industry's cooperative research organization, the Electric Power Research Institute (EPRI). In addition, non-U.S. manufacturers were making similar advances and reporting on retrofit experiences in Germany, Japan, the United Kingdom, and Italy (Eskinazi et al. 1989). A number of problems with LNB/OFA surfaced during this period. The most significant and common of these problems related to increases in carbon monoxide (CO) emissions and carbon-in-ash (LOI) and to the high temperature corrosion of burner components and air and fuel flow controls.

Meanwhile, post-combustion approaches – notably SCR – were undergoing rapid development overseas in response to stringent regulation in Germany and Japan. Most of these early systems achieved NO_x reductions on the order of 60 to 80 percent. Initial problems with SCR in these countries included catalyst poisoning, plugging, erosion, and the formation and deposition of ammonium sulfates. Technology advances throughout the 1980s resulted in improved catalyst selectivity (which decreased the simultaneous oxidation of sulfur dioxide during the NO_x -ammonia reaction), greater resistance to catalyst poisoning, higher catalyst activity and longer catalyst life. By 1990, the combined number of commercial SCR applications in Japan and Germany had grown to about 200 units (representing about 40,000 MW of capacity) and catalyst costs had come down by a factor of three (from $900 per cubic foot to approximately $300 per cubic foot) (IEA Coal Research 1991; Eskinazi et al. 1989). There was thus a wide base of experience to draw from as the use of SCR began to be contemplated in the U.S. in the late 1980s. It was by then well known, for example, that the deployment of SCR with combustion controls such as LNB could reduce the

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[54] The broader political dynamics of this period are reviewed in Chapter II, which covers automobile regulation.

amount of NO_x entering the SCR system and thereby minimize catalyst volumes and efficiency losses (Necker 1989; Mori 1989). In addition, a number of practical, operational challenges had emerged. These included ensuring uniform distribution and mixing of NO_x and ammonia, monitoring downstream ammonia concentrations, maintaining process control during load swing operation, and disposing of the catalyst (Suyama 1987). Experience had also resulted in a number of cost-saving strategies, such as leaving space in the reactor for future catalyst additions.

The late 1980s also saw early experimentation with two additional control options: SNCR and reburn. An ammonia-based SNCR system had been developed as early as 1977 by Exxon; urea-based technology was subsequently developed and patented by EPRI and eventually sub-licensed to several vendors under the trade name NO_x OUT®. Responding to California's early interest in NO_x reduction, the first full-scale demonstration of SNCR on a gas-fired boiler was undertaken in Southern California in the late 1980s (Mansour et al. 1987; Radak et al. 1991). During this period, a number of pilot demonstrations and evaluation programs were also underway in the U.S. and Japan to test reburn technologies. The basic concept of reburn had been understood since the 1970s. It was generally considered less cost-effective than LNB, the chief technology in use during the 1970s and 1980s, and had therefore received less attention. By the mid-1980s, however, interest in reburn technologies had begun to re-emerge, in large part because this approach provided a potential control option for cyclone-type burners, for which LNB technology was not a suitable alternative. Substantial operational advances in SNCR, reburn, and other control technologies, however, did not come until the 1990s when a number of regulatory drivers at the states and federal level were put in place in the eastern U.S.

D. The Post-1990 Period

D.1 Regulatory Milestones

Major new federal legislation, in the form of the Clean Air Act Amendments of 1990, ushered in a period of more aggressive NO_x regulation in the early 1990s. To address ongoing ozone pollution problems, existing power plants were for the first time required to install controls; this requirement applied only to plants located in non-attainment areas but substantially increased the number of units subject to NO_x controls. In addition, Congress took an important step toward recognizing the regional nature of ozone formation and transport by creating the northeastern Ozone Transport Commission and empowering it to promulgate region-wide control requirements. The other major provisions of the 1990 Amendments relevant to NO_x control involved acid rain mitigation and visibility. In addition to creating a novel, nationwide system of tradable allowances for SO_2, the Act's new acid rain provisions directed EPA to implement annual national NO_x reductions totaling 2 million tons. To accomplish this, EPA was directed to promulgate new standards over a specified timeframe for certain populations of existing boilers. New visibility provisions, meanwhile, did not specify further regulation but did require continued efforts to address regional haze.

The emissions limits resulting from the first round of regulation under the Amendments of 1990 generally remained achievable through a combination of combustion modifications such as LNB and OFA. (As noted in the previous section, considerable research and development had gone into adapting these technologies for retrofit purposes in the late 1980s) These approaches generally sufficed, for example, to meet states' RACT (Reasonably Available Control Technology) limits, which were imposed by the summer of 1995 on all units located in areas whose ozone non-attainment status was deemed moderate or worse. RACT requirements overlapped to a considerable degree with NO_x limits that were imposed under the Acid Rain Program starting in 1996. That program required tangential and wall-fired dry bottom boilers (excluding cell burners) to limit emissions to 0.45-0.50 lb/mmBtu by 1996 for Phase I units and to 0.40-0.46 lb/mmBtu by 2000 for Phase II units. These standards were developed with explicit reference to the reductions achievable with LNB technology. Similarly, the development of emissions limits for other boiler types under the Acid Rain Program (i.e., cell, cyclone, wet bottom, and vertical fired units) was explicitly based on reductions achievable at a cost comparable to the cost of LNB. Limits for this group were later set at 0.68-0.80 lb/mmBtu depending on boiler type; these too were implemented at the beginning of 2000.[55]

The first important steps beyond control levels achievable with LNB/OFA technology were driven by regulatory actions on the part of California and the Ozone Transport Commission (OTC). In California, Los Angeles Basin Rule 1135 was adopted in 1991. It mandated the installation of NO_x controls on power plants by the end of 1997 and specified a company-wide emissions rate of 0.15 lb./MWh to be achieved by the end of 1999. The latter requirement is equivalent to an emissions limit of approximately 0.015 lb/mmBtu on a heat input basis and is 10 times more stringent than the control levels currently being proposed in the eastern U.S. Similarly, Ventura County Rule 59, also adopted in 1991, applied unit-specific NO_x emission limits over the 1992-96 timeframe. Together these requirements imposed reductions of close to 90 percent on gas-fired electricity generating boilers in southern California. Flexibility to achieve these requirements through a cap and trade program was later added with the adoption of the RECLAIM program in 1993.

In 1994, the OTC adopted a Memorandum of Understanding (MOU) aimed at achieving substantial NO_x reductions across the 12-state ozone transport region stretching from northern Virginia to Maine.[56] The MOU committed participating states to develop regulations that would go beyond the RACT requirements already mandated under the Clean Air Act to achieve total NO_x reductions of 55 to 65 percent by 1999 and 65 to 75 percent by 2003.[57] Most OTC member states, including the eight NESCAUM states, are participating in Phase II of the OTC MOU and several have created a NO_x budget program to allow for trading of allowances among sources. The budget applies to all units rated 15 MW and up,[58] as well as to similar-sized industrial boilers, and is enforced for the five-month ozone season from May to September. To date, most units have complied with the Phase II requirements by utilizing LNB, OFA, and in some cases SNCR or SCR.

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[55] Note that unlike NOx control requirements for ozone attainment, NOx controls under the acid rain are annual – that is, they apply year-round and not just during the May to September ozone season. Sources have a number of options for complying with NOx limits under the acid rain program. They can meet the applicable emissions limit, average their emissions with other facilities within the same company, or petition for an "alternative emission limit" (AEL). The latter is intended for situations where the technology on which the applicable emissions limit is premised fails to perform to expectations due to site-specific conditions. AEL applications must be approved by EPA.

[56] The OTR is comprised of Maine, New Hampshire, Vermont, Massachusetts, Connecticut, Rhode Island, New York, New Jersey, Pennsylvania, Maryland, Delaware, the District of Columbia, and the northern part of Virginia.

[57] Phase I of the OTC NOx program essentially consisted of RACT implementation; the 1999 requirements constituted Phase II, and the 2003 requirements constituted Phase III.

[58] Somewhat less stringent standards apply to units under 25 MW in size.

SCR, in particular, is expected to be more widely used to meet the Phase III requirements of the program, which are essentially based on achieving an emissions rate of 0.15 lb/mmBtu.

More recently, the federal government has attempted to extend NO_x limits similar to those set out by the OTC MOU to a broader region. Following a two-year, multi-stakeholder process known as the "Ozone Transport Assessment Group" (OTAG), EPA proposed new NO_x limits for a 22-state region covering most of the eastern U.S. The proposed limits were based on emissions reductions of approximately 85 percent from uncontrolled levels and implied limiting average emission rates for power plants and other large industrial boilers to approximately 0.15 lb/mmBtu. EPA's action would require states to submit State Implementation Plans for achieving these reductions by 2003 under Section 110 of the Clean Air Act, which provides mechanisms for dealing with interstate ozone transport. States that failed to submit implementation plans would be subject to a federal implementation plan or FIP for achieving the requisite NO_x reductions. EPA's Section 110 SIP call is currently being litigated in the courts. Meanwhile, in a separate but related action, several individual northeastern states have formally petitioned EPA to impose upwind NO_x reductions to mitigate ozone transport under Section 126 of the Clean Air Act.

As EPA seeks to substantially limit emissions from existing boilers throughout the eastern U.S., the Agency is requiring comparable levels of NO_x control (i.e. at about 0.15 lb/mmBtu) for new or substantially modified facilities under the NSPS program. New NO_x NSPS requirements promulgated in 1998 for the first time set "fuel neutral" limits[59] based on electricity output rather than fuel heat input. The latter approach is expected to promote energy efficiency and pollution prevention by encouraging reductions on the basis of useful electric output rather than fuel consumption. The revised NSPS requirement for new units is 1.6 lb./MWh, while modified or reconstructed units will be subject to a more conventional, heat input-based limit of 0.15 lb/mmBtu. Each standard would be enforced on a 30-day rolling average basis. EPA based these recent NSPS revisions on the performance of "best demonstrated technology", including the limited SCR data available from coal-fired boilers in the U.S., as well as European and Japanese units. In addition, EPA considered SNCR data from domestic and foreign units.

Other important regulatory developments in the last decade have included the promulgation of a revised NAAQS for ozone and a new NAAQS for fine particulate matter. The revised ozone NAAQS would have established an 8-hour standard at 80 parts per billion (compared to the current 1-hour standard at 120 parts per billion) and was expected to result in an even greater emphasis on regional control measures. Likewise, a new NAAQS for fine particles could be relevant to the future of NO_x regulation because nitrates contribute to fine particle formation, albeit less so in the East than in the West. Like the Section 110 SIP call, both NAAQS revisions are currently under court review and have not yet taken effect. Finally, in July of 1999, EPA issued new regulations for visibility protection. The new regulations seek to establish a long-term program for addressing regional haze, which could include additional controls for fine particles, SO_2, and NO_x. Visibility SIPs are due from states in the 2006-2008 time frame, depending on attainment status with respect to the fine particle NAAQS.

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[59] In the past, different NSPS requirements applied to different boiler fuels.

D.2 Technology Challenges and Advances: (Post-1990)

The more stringent NO_x regulations introduced since 1990 have prompted further refinement of combustion modification technologies (such as LNB), as well as full-scale implementation of advanced post-combustion control technologies (such as SCR, SNCR, and combined approaches). As discussed in the Section C of this Chapter, some of these technologies were developed and implemented elsewhere (notably in Japan and Germany) over a decade earlier.

In the early 1990s, RACT and Acid Rain requirements led to a rapid acceleration of LNB and OFA installations on existing boilers. By 1996, over 250 coal-fired units had been retrofitted with these technologies and were achieving NO_x emissions rates on the order of 0.45 lb/mmBtu or lower, in accordance with new regulatory requirements (Table III-1). This was lower than the emissions limit specified for new sources in 1977 and reflected substantial improvement in the performance of combustion modification technologies. Relatively few units experienced problems with these installations, and in fact only 10 out of 265 units affected by the Phase I Acid Rain requirements ended up applying for less stringent Alternative Emissions Limits (AELs) (Krolewski et al. 2000). Active competition between a number of domestic and foreign vendors spurred substantial innovation during this period and led to continued improvement in cost-effectiveness (Grusha 1991; Kokkinos 1991).

More recent innovations related to LNB technology include "burner modifications" and "intelligent" controls. The former approach allows for the selective replacement of burner components as an alternative to whole burner replacement and can substantially reduce costs. Intelligent controls are designed to maintain optimum combustion conditions for minimizing emissions and maximizing efficiency. Variants of this approach have been implemented on over 200 coal-fired units in the last five years (Breen et al. 1996; Broderick et al. 1995; Meleck et al. 1997; Sorge et al. 1997; Merlo et al. 1999).

Table III-1: Technology Responses to the Acid Rain Provisions (Title IV) of the 1990 Clean Air Act Amendments

	NO_x Control Technology	**No. of Boiler Applications**	**1998 Average NO_x Emission Rate (lb/mmBtu)**	**NO_x Reduction from 1990 levels**
Dry Bottom, Wall-Fired Units	**LNB**	**66**	**0.45**	**44%**
	LNB with OFA	**21**	**0.47**	**48%**
Tangentially Fired Units	**LNB**	**44**	**0.36**	**43%**
	Separated OFA	**23**	**0.37**	**33%**
	LNB with separated OFA	**23**	**0.36**	**45%**

Source: Krowlewski et al., 2000

Meanwhile, in expectation of more stringent regulations to come, SCR technology was tested in a growing number of pilot projects in the early 1990s. EPRI and the U.S. Department of Energy sponsored pilot projects at units owned by Southern Company Services, the Tennessee Valley Authority, New York State Electric and Gas, Niagara Mohawk, and Pacific Gas & Electric (Eskinazi et al. 1989; Zammit et al. 1995; Teixeira et al. 1995). The stringent NO_x regulations being contemplated in southern California at this time prompted Southern California Edison to undertake demonstration projects utilizing advanced versions of SCR technology at two units in 1991 (Johnson et al. 1991).[60] Two years later, Southern California Edison announced commercial retrofits of SCR technology on an additional four units (Johnson et al. 1993). The company estimated that recent SCR advances had resulted in cost savings ranging from $65 million to $240 million for those four units.

The first full-scale, retrofit application of SCR technology on an existing coal-fired power plant in the U.S. became operational in 1995 at the Merrimack Station in New Hampshire (NESCAUM 1998). This occurred almost two decades after the SCR technology was successfully retrofitted to coal-fired power plants in Japan. The Merrimack system was designed to achieve eventual control levels of 90 percent and is performing at a cost of just $400 per ton of NO_x reduction. Presently, ten SCR systems are in commercial operation, including four retrofits, and several more installations are planned. Consistent with the experience in Germany and Japan, some of the early U.S. installations experienced difficulties with ammonia flow controls and distribution, ammonia 'slip'[61], pressure loss, and increased outages. However, many of these problems have been overcome as plant operators move up the learning curve. More recently, practical SCR advances have targeted operational controls, such as on-line monitors; mitigation of ammonia deposition and plugging of air preheaters; and further process and catalyst optimization; as well as hybrid SCR/SNCR combinations and new approaches to ammonia control (Bondurant et al. 1999; Prietchard et al. 1999; Wallace et al. 1995; Johnson et al. 1999). At the same time, experience with recent retrofit projects is providing valuable new information on cost, achievable emissions rates, and compatibility with low-sulfur western coals, including Powder River Basin coal (Robinson et al.; Cochran et al. 2000). Industry surveys indicate that the market for SCR technology will continue to grow strongly over the next five to ten years (McIlvaine 2000). Growth is likely to be especially strong if broad-based NO_x reduction requirements take effect throughout the eastern U.S. as EPA has proposed.

Pilot demonstrations of SNCR technology also increased in the early 1990s, with first installations on existing coal-fired units being demonstrated at Salem Harbor Station in Massachusetts in 1992 and 1993. These demonstrations revealed a number of problems with the technology and led to further R&D on issues such as temperature control during load swings, ammonia slip, flyash contamination, air preheater fouling from ammonia salts, more responsive system controls, and the formation of N_2O (Kwan et al. 1993; Muzio et al. 1993; Chen et al. 1993; Berg et al. 1997). These efforts resulted in some significant process improvements in the mid-1990s, including advanced controls, optimized injector design, and better matching of combustion conditions to SNCR operation (Afonso et al. 1997; Jones et al. 1995; Staudt et al. 2000). SNCR is

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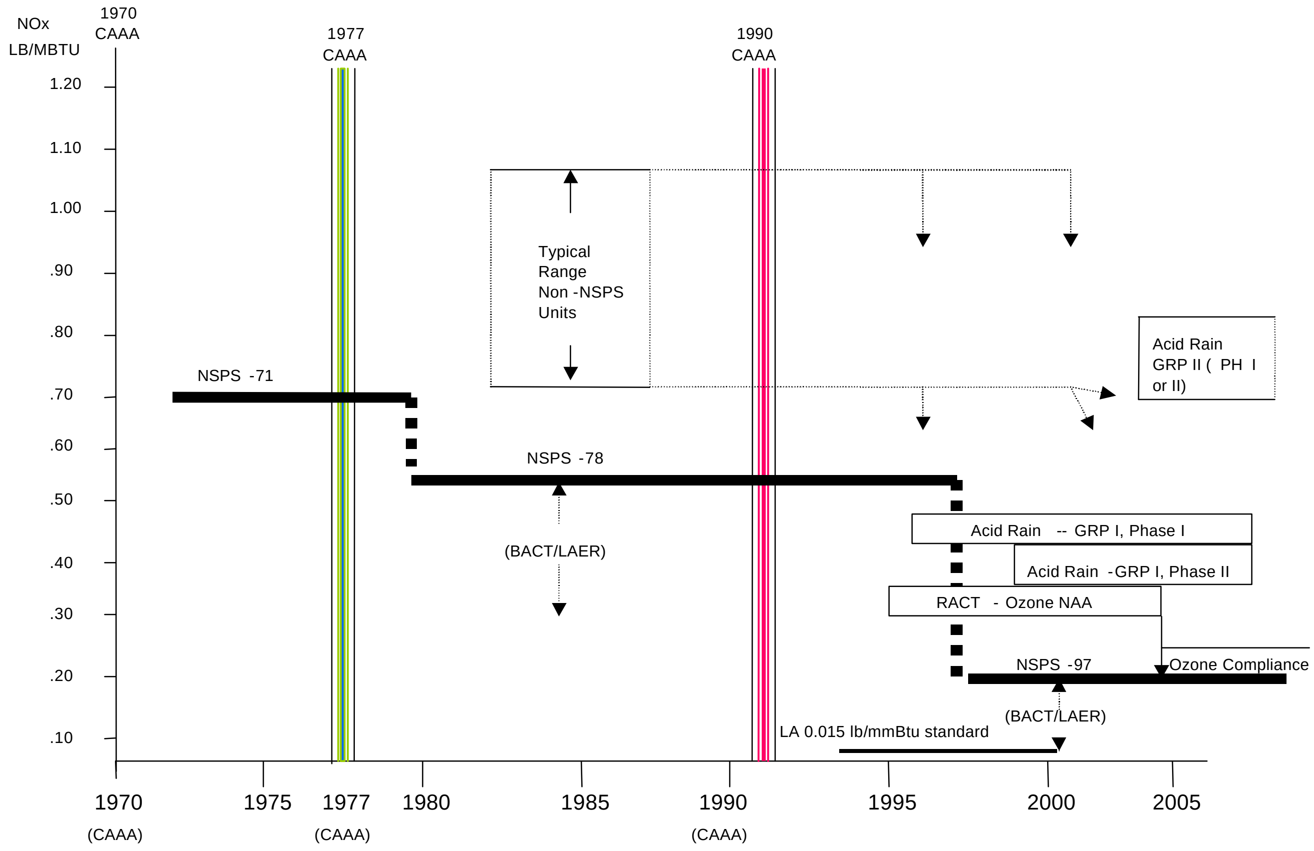
[60] These advanced versions involved the use of catalytic materials on air preheater surfaces and in the duct space between the economizer and air preheater.

[61] Ammonia 'slip' refers to a small but steady release of ammonia to the atmosphere that may occur in SCR and SNCR systems during regular operation.

currently installed on over 20 coal-fired units in the U.S. and over 300 units worldwide (Cummings 2000); it was recently successfully demonstrated under previously untested conditions on a 600 MW coal-fired boiler (Malone et al. 2000).

In addition to SCR and SNCR, several field experiments with reburn technology were beginning to yield results by the early 1990s. For example, a gas reburn system at an Ohio Edison plant and a coal reburn system at a Wisconsin Power & Light plant indicated NO_x reductions ranging from 45 to 55 percent (Brown et al. 1992; Newell et al. 1992). These results were consistent with the results obtained for a gas reburn demonstration on a conventional tangentially-fired boiler owned by Illinois Power (May et al. 1992). Interest in reburn, like SNCR, has been limited by its higher costs (relative to LNB/OFA). However, again like SNCR, it can be readily turned on and off. This feature, together with some more recent innovations have combined to maintain interest in this technology as can be seen from the growing numbers of applications (Folsom et al. 2000; Glickert 2000). More recent advances have included improvements in the efficiency of gas use such as through fuel lean gas reburn (FLGR) and pairings of reburn technology with SNCR. The latter approach has been shown to be capable of achieving NO_x reductions in the 60 to 70 percent range and may provide an alternative to SCR in some applications (Folsom et al. 1995; Glickert 1997; Folsom et al. 1999).

Figure III-1: The Relationship Between Regulations and Implementation of NO_x Control



Throughout the 1990s, experimentation with all of these control technology options yielded performance improvements. Fine-tuning of systems once they are in place and greater operating experience have yielded such improvements even after controls were installed. This process of optimization is evidenced by the reduction in average NO_x emissions rates that occurred between 1996 and 1998. As Figure III-1 reveals, the same technologies implemented under the Acid Rain program in 1996 yielded lower emissions when the more stringent Phase II OTC MOU requirements took effect a few years later.

E. Conclusions

Essentially, the history of NO_x control development differs from other case studies presented in this report inasmuch as the regulatory drivers that first spurred today's advanced technologies largely happened overseas. By the time some states and regions of the U.S. introduced more stringent NO_x emissions limits in the 1990s, SCR technology and other control options were already commercially available. Nevertheless, the speed and relative ease with which a variety of control options have been tested, applied, and adapted to U.S. conditions, still speaks volumes about the ability of industry to act -- and act quickly -- to implement new technologies when compelled to do so by regulation. In addition, the ongoing refinement and optimization of existing control technologies testifies to industry's ability to continually improve performance and lower costs when given the market and regulatory incentives to do so.

The pace of recent activity in the area of NO_x control is in stark contrast to the relatively slow progress that was achieved between 1970 and 1990, when a relatively limited suite of technologies (primarily combustion modifications such as LNB) sufficed to meet existing standards. The fact that for two decades U.S. regulations lagged far behind what was being achieved in other countries was probably not surprising, given that legislative guidelines authorizing the promulgation of standards during this time period were explicitly technology-following. Even now, the typical control levels required in Japan, approximately at 0.1 lb/mmBtu, are more stringent than most NO_x regulations being proposed in the U.S. If the past is any guide, the recent advent of technology-forcing regulations to address long-standing problems of ozone non attainment in the Northeast and elsewhere should continue to produce more effective and diverse options for substantially reducing NO_x emissions from electricity generating boilers. The emissions limits set to be applied to existing sources in the OTR and Southern California, in particular, represent a further substantial increase in stringency from previous standards. These more stringent limits, combined with the compliance flexibility of trading programs, are certain to inspire a broad mix of technological applications and innovations, including, in all likelihood, post-combustion controls that can reduce NO_x emissions by even more than the 85 to 90 percent levels that have been demonstrated to date.

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IV. The Regulation of Sulfur Dioxide Emissions from Coal-Fired Boilers: A Case Study

A. Introduction

Among modern environmental controls, technologies to remove sulfur dioxide (SO_2) from power plant exhaust gases have one of the lengthiest histories. As in the previous case studies, this history demonstrates important linkages between technological innovation and government regulation. A review of the development and deployment of SO_2 control technologies suggests a number of observations relevant to the current debate about mercury regulation. First, the SO_2 experience indicates that private industry will conduct preliminary research based upon the *anticipation* of future emissions control requirements, but is unlikely to commercialize or deploy the technologies that result from this research unless and until such requirements are implemented. Second, the stringency of regulatory requirements strongly influences the control capability of subsequent technologies, especially in cases where a command and control type of approach is used (i.e. each source is required to meet inflexible emissions limits). In contrast, approaches that provide flexibility in meeting emissions reduction requirements can encourage continued efforts to optimize control technology performance at individual facilities by creating market incentives for further reductions.

However, flexibility should not be confused with stringency. Even market-based approaches will not induce overall emissions reductions in excess of regulatory minimums unless they provide incentives for over compliance in aggregate, as well as at individual facilities. A good example is the current national Acid Rain Program, an important feature of which is that improved SO2 control at one facility can be traded off for less control in others – this flexibility creates an incentive to improve performance at individual facilities as much as possible. However, the Acid Rain Program is aimed at achieving a relatively modest 50 percent reduction in U.S. power plant SO_2 emissions from 1990 levels, and switching to low-sulfur fuels is also permitted. Thus, although modern emission control technology is highly reliable, capable of routinely achieving control levels of over 95 percent, and will usually yield salable by-products; to date only 27 additional installations of control technology have occurred nationwide in response to the Acid Rain Program. And since there are no incentives for further reductions in total emissions past 50 percent, emissions levels will remain at this level for the foreseeable future.

The chief SO_2 control technology in use today remains flue gas desulfurization (FGD), commonly known as "scrubbing." An early version of this technology was first installed at the Battersea plant in London in 1936. Although capable of removing up to 95% of SO_2 from the flue gas stream by passing it through a spray of Thames River water mixed with chalk, the Battersea installation was problem plagued, rarely operated a 95% removal rate, and created significant quantities of waste that seriously polluted the river (Slack and Hollinden 1975 pp. 49-50). In comparison, a modern limestone forced oxidation (LSFO) process such as that installed in 1993 at the Bailly Station in Illinois can perform very reliably and produces commercial grade gypsum as a saleable byproduct (Manavi, Styf et al. 1995).

It took many decades to travel from Battersea to Bailly, and the trip was not always smooth. The first U.S. installation of an FGD unit, in 1968 at Union Electric's 140 megawatt (MW) Meramac No. 2 unit in St. Louis, was later abandoned due to operational problems. Now, some thirty years later, about 90 gigawatts (GW) of scrubbed electrical capacity is in operation in the U.S. (accounting for about one-third of the nation's total coal-fired electric generation capacity); worldwide, over 250 GW of scrubbed capacity is expected to be in operation by 2000 (Soud 1994; Srivastava, Singer et al. 2000). Obviously, great progress in SO_2 control technology has been achieved since 1936 and as this chapter reveals, much of it occurred in direct response to government policies.

Controlling SO_2 emissions from power plants presents a non-trivial engineering challenge for several reasons.[62] First, SO_2 constitutes only about 0.2-0.3 percent of the exhaust gas to begin with and removing it involves processing an enormous quantity of hot gas. Thus, a key challenge is providing enough space and time for the SO_2 to make contact and interact with the reagent (e.g. crushed limestone) that chemically removes it. Second, the resultant product is highly corrosive and can easily damage the FGD equipment itself. Third, the chemistry of SO_2 removal is subtle – the fundamental reactions are straightforward, but their speed and controllability at a large scale are affected by a number of parameters that were only fully understood as recently as the 1980s, notably pH and oxidation rate. Fourth, space constraints can present a major problem at power plants that were not originally designed to accommodate pollution control systems; hence siting issues have proved to be difficult to resolve in some hard-to-retrofit cases. Fifth, extended shut downs of power plants can be costly, plant availability and reliability are paramount concerns. This makes the quick installation and reliable operation of emissions control devices crucial.

For convenience and clarity, this case study is divided into four time periods according to the passage of different Amendments to the federal Clean Air Act: (1) pre-1970, (2) 1970-1976, (3) 1977-1989, and, (4) 1990 to the present.[63] Significant technological advances in SO_2 control occurred in *each* of these periods, although the form of these advances varied greatly, as did the reasons for their emergence. A graphic timeline (Figure IV-3) and a tabular chronology (Table IV-2) are presented at the end of this chapter. For each of the periods considered, a description of government actions is followed by a summary of the research and development efforts (R&D) that led to technological progress during that period. A brief look forward and conclusions end this narrative.

B. Pre-1970

B.1 Government Actions

Air pollution is an old problem, but until recently, the pollutants of concern were mostly smoke and soot and the little regulation that existed was developed by local governments (Beaver 1955; Brimblecombe 1987; Grinder 1980; Tarr 1996 pp. 219-283). The concept of chemical air pollution was first recognized in mid 19th century in England, when the Alkali Inspectorate was created to deal with the problem of hydrochloric acid fumes from industrial sources (Brimblecombe 1987 pp. 136-141; Cowling 1982). In the first half of the 20th century, SO_2 pollution was

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[62] See (Cooper, Collins et al. 1997) for a brief overview of the relevant technologies.

[63] See Table IV-2 at the end of this chapter for a brief summary of the relevant legislation, and Figure IV-3 for a graphic illustration of the change in coal-fired power plant standards and emissions over time.

recognized as a problem in some localized areas near urban power plants and smelters (Rees 1955; Weiss, McCaffrey et al. 1999 pp. 245-256). The principal problem associated with these facilities was direct acidic attack on human health, vegetation, and buildings.[64] At the time, government action on air pollution was limited to the courts – complaints could be filed on the basis of the common-law concept of "nuisance"; some of these cases reached as far as the U.S. Supreme Court or international tribunals (206 U.S. 230 and 237 U.S. 474, 477). When relief was granted by the courts, and it often was, the remedy was usually simple, the source of pollution was simply barred from creating the nuisance. The first FGD installations occurred in 1930s in both the U.S. and England as a result of such lawsuits.

Pollution concerns and associated government responses first emerged in their modern form soon after the end of the Second World War with the first detection of photochemical smog in the Los Angeles area, the 1948 air pollution disaster in Donora, Pennsylvania, and similar episodes in Europe (Fensterstock and Fankhauser 1968; Haagen-Smit 1952; Kiester Jr. 1999). Many municipalities and state governments adopted pollution control laws; these had some localized effects but they were unable to stem rapidly worsening air pollution problems in the U.S. during the 1950s and 1960s (Crenson 1971; Esposito 1970; Krier and Ursin 1977).

Congress began to turn its attention to air pollution in 1949, when the first bills specifically funding air pollution research were introduced (Bailey 1998 p. 89). In 1955, Congress passed the Air Pollution Control Act which provided $5 million annually for research and aid to the states. The resulting research began to affect how Congressional leaders (and their staffs) viewed air pollution by providing them with the first clear evidence of both the extent of the problem and the inadequacy of state control arrangements. Highly publicized events in 1962, notably the publication of Rachel Carson's book *Silent Spring* and the London smog disaster, in which 700 people died, heightened political awareness of air pollution problems, as did the first national conferences on air pollution that were held in 1958 and 1962. These conferences also illuminated the tensions between those arguing for national regulation and those arguing for further research (U.S. Public Health Service 1962). Responding to growing constituent concerns, associations representing local governments began to lobby for an enhanced federal role, a step supported by the Kennedy and later the Johnson administrations (Ripley 1969).

The earliest federal legislation to specifically address SO_2 emissions from power plants was the 1963 Clean Air Act. This law established permanent federal research funding, a total of $95 million for 1964-67 (Bailey 1998 p. 107). More importantly, it took a major step toward direct federal control of air pollution by giving the Secretary of the Department of Health, Education, and Welfare (HEW) the authority to take legal action against interstate polluters. This was the first of a series of actions that eventually created a state/federal partnership approach to air quality regulation in which federal mandates played a major role.

During the remainder of the 1960s, interest in pollution control grew, as did the severity and scope of air quality problems. Some cities and states took serious steps to control pollution, many of which directly threatened the economic viability of the coal industry. For instance, in 1965 the New

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[64] Although very similar chemically, these localized effects were quite different from those later described as acid rain. For instance, it was easy to see and smell localized sulfur emissions from uncontrolled facilities at this time. By contrast, acid rain is a long-distance phenomenon that is invisible to the naked eye and produces more subtle, long-term impacts. Acid rain did not become an important issue until the late 1970s.

York City Council restricted the use of coal with high sulfur content and in 1966 four northeastern states announced pollution control plans that similarly threatened coal industry (Anonymous 1966; Jones 1974 p. 165; Knowles 1965). In 1967, the Secretary of HEW published a report recommending reduced reliance on high-sulfur coal in light of the fact that virtually all major American cities were exposed to unhealthy levels of SO_2 (Bailey 1998, p. 129). At that time, the Federal Power Commission identified 12 different state regulations relating to power plant SO_2 emissions, and 21 at the city or county level (Anonymous 1968).

The next major law to affect SO_2 emissions from power plants was the 1967 Air Quality Act. This bill considerably expanded research funding for control technologies (to $125 million), a measure that was strongly supported by the coal industry. In addition, it directed states to set ambient air quality standards; states that did not do so within fifteen months after the law went into effect were faced with potential federal intervention. The 1967 Act designated HEW as the lead agency in all aspects of air pollution control through its National Air Pollution Control Administration (NAPCA), and directed HEW to report on health effects to assist states in setting standards (Portney 1990). Importantly, though some early drafts of the 1967 law contained provisions for federal ambient air quality standards, the final version signed by President Johnson did not (Bailey 1998 p. 133).

In subsequent NAPCA reports, SO_2 emissions, which came predominately from stationary sources, were recognized as the second most important of all air pollutants in the U.S., after automobile emissions (NAPCA 1968). Power plants were selected as one of government's first research priorities because, according to NAPCA:

- they were the largest man-made source of SO_2; accounting for 46 percent of total emissions in 1967,
- they were concentrated in or near urban centers,
- they were growing at a rate of 6 percent per year,
- they were (and are) intimately related to the flow of energy resources, economic development and national well-being, and
- the methods available to control SO_2 emissions were inadequate to the task.

Projections of future SO_2 emissions lent urgency to these findings. NAPCA estimated that power plant SO_2 emissions totaled 15 million tons in 1967, this total was projected to reach an alarming 94.5 million tons in 2000, assuming successful deployment of the fast breeder reactor beginning in the early 1980's (NAPCA 1970).[65] Without a successful breeder program, NAPCA estimated emissions would reach 110 million tons by 2000 and 200 million tons by 2020.

In summary, by the end of the 1960s, several major government actions relevant to the control of SO_2 from power plants had been initiated, many of them involving research funding. Actual implementation of emissions controls remained rare and had occurred in only a few cases as a

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[65] Estimates of future SO2 emissions in the 1960s and 1970s were based on the assumption that coal-fired generation capacity would expand dramatically. Energy forecasting was, and remains, a notoriously difficult exercise.

result of successful lawsuits, despite the fact that some form of state and local pollution regulation existed in most of the U.S. Industry typically supported government research because it tended to delay regulation and was expected to help reduce the cost of future reductions. Toward the very end of this period, a small federal role in the enforcement of air pollution regulations began to emerge. This federal role was to strengthen dramatically in the 1970s.

B.2 Technological Advances

The first pollution control devices were chimneys (or stacks) that directed smoke from cooking and heating fires outside of homes and businesses. These devices did not reduce emissions, they just dispersed them. Through the late 1970s, this approach was one of the primary methods used to mitigate SO_2 pollution in the U.S. Eventually, smokestacks proved to be inadequate for controlling smoke pollution in large urban areas, just as they would prove inadequate for SO_2 control. Switching to low-sulfur fuels was also an option and remains one of the most frequently utilized methods for reducing SO_2. However, fuel switching was also limited by the fact that clean fuels were often scarce or more expensive than dirty coal (Beaver 1955; Brimblecombe 1987 pp. 14-18, 169-175; Tarr 1996 pp. 227-261, 280). Thus, interest in add-on control technologies continued.

Chemical pollution proved more difficult to control than smoke because the hydrochloric acid fumes of some early industrial facilities had such high concentrations that dispersion did not work. Fortunately, a relatively simple solution was available: hydrochloric acid could be washed out of a gas stream simply by passing it through water, a process called scrubbing. (Raufer 1998 pp. xi-xvi). If the resulting acid solution was concentrated enough, it could be sold as a commercial product. William Gossage received a British patent for this technology in 1836. Subsequently, some industrial facilities in England installed towers packed with a solid material over which water was sprayed, but generally only when they were required to do so by local authorities. This approach was very effective in reducing hydrogen chloride emissions (which could be reduced by almost 99 percent), but it emitted SO_2 and other wastes in large quantities. Nonetheless, modern FGD technology eventually emerged from these crude beginnings.

The principal technology used to address air pollution, however, remained the tall smokestack (Beaver 1955). Although now usually considered inappropriate, at the time dispersion through the use of tall stacks was considered entirely reasonable by many government officials and other experts, as was the relocation of polluting activities to unpopulated areas. Indeed, considerable engineering went into the design of stacks that would effectively diffuse pollution (Ross 1980). A report issued by the group that undertook the first permanent SO_2 research efforts was typical:

> “[T]he Report . . . represents the whole philosophy of thinking and action on air pollution, namely, that the pollution suffered by mankind is a balance between the amount emitted into the atmosphere and the amount removed from near the ground by atmospheric turbulence and blown away harmlessly by the wind. Life in urban areas is tolerable only on account of natural dispersion of pollution, and the utilization to the full of these natural processes, by suitable chimney design and siting, and by proper town planning, must take its place alongside the reduction of emissions, as an equal partner, in the national effort towards clean air.” (Ministry of Technology 1967 p. 5)

Interest in actually reducing SO_2 emissions was nevertheless sufficient to cause patents for FGD technologies to be issued by the end of the 19th century in England (Newall 1955). Early tests were conducted in England just after the turn of the century and the London Power Co. produced a study of control problems in 1930 (Slack and Hollinden 1975 pp. 49-52). When Battersea, a new power plant, was being constructed in London in 1929, the public demanded that its emissions be cleaned up. Because water alone does not absorb SO_2 (as it does hydrochloric acid fumes), it was necessary to build a modified version of the packed tower scrubbers invented in the 19th century (Rees 1955). The major innovation introduced at this time was to add chalk (a form of limestone) to Thames River water and spray this mixture into the packed tower; the effluent was then dumped back in the river. The first system operated between 1933 and 1940, when it was shut down due to the war. After the war, Battersea was expanded and its FGD unit was modified (creating a "second-generation" system) to remove some of the solids from the effluent before it was discharged back to the river. Later, a similar installation was added to the Bankside power plant, also in London, and both systems continued operation through 1970.

The Battersea process was successful, but imperfect. When operating properly, it removed over 95 percent of the SO_2 from the exhaust gas, however it was highly unreliable so the effective rate of emissions control might well have been 50 percent or less. In addition, the Battersea process used enormous quantities of water (20 to 30 tons for each ton of coal burned and was expensive, operating costs added 12 to 15 percent to the cost of coal. It was also difficult to maintain, since, as Rees (1955) notes, "even in this simple process at this late stage of development, satisfactory materials of construction have not been found for all uses." The Battersea process could be used in only a few places, since it needed brackish water to work; moreover, between them, the Battersea and Bankside units exhausted the capacity of the Thames to absorb effluent without undue harm. In addition, these systems sometimes exacerbated local pollution impacts by cooling the exhaust gases which then tended to sink to ground level (hot exhaust gases typically rise and are dispersed). Thus, these early systems were often bypassed to allow for dispersion, which only added to the emphasis on tall stacks. Despite these problems, the basic Battersea process (now called wet lime/limestone scrubbing) came to dominate FGD systems worldwide, albeit in heavily modified form.

Ammonia-based FGD systems had also been invented in the 19th century and were patented in 1931 (Ministry of Technology 1967 p. 16). The first unit was not installed until 1936, when a smelter at Trail, British Columbia was required by an international court to reduce its SO_2 emissions (Hein, Phillips et al. 1955; Strauss 1971 p. 123-135). An ammonia-based FGD was first installed on a power plant in 1939, at the Fulham station in London; a second such system was installed in 1957 in Nottingham.

A central problem for all SO_2 control processes was the disposal of waste products, which were often produced in great quantities. (For instance, the Battersea FGD system produced about 40 tons of calcium sulfate sludge each day.) Many efforts were made to develop an FGD process that was both inexpensive and yielded a saleable product. It was possible to derive sulfuric acid (or sometimes sulfur) from smelter exhaust gases, because they had relatively high SO_2 concentrations. However, even untreated power plant exhaust gas has *less* SO_2 in it than scrubbed smelter exhaust, making smelter-type systems unsuitable for power plants. By the mid-60s, a Japanese company had begun to resolve the problem by devising an FGD system that could produce saleable gypsum (Strauss 1971 p. 127). Ultimately, such water-based systems came to dominate the market, comprising 80 percent of installations worldwide (Soud 1994).

Bench and pilot scale work continued at modest levels in the U.S., Europe and Japan until the 1960's when SO_2 control research accelerated due to the government initiatives discussed above. Many different organizations participated in these efforts, including the U.K. Central Electricity Generating Board, U.K. Ministry of Technology, Chubu Electric Power (Japan), Tennessee Valley Authority, Combustion Engineering, Imperial Chemical Industries, U.S. Bureau of Mines, Monsanto, Research-Cottrell, Pennsylvania Electric Company, Mitsubishi, Mobil Oil, Volkswagen, Metropolitan Edison, Tampa Electric, Tokyo Institute of Technology, Detroit Edison, Babcock & Wilcox (Strauss 1971). At the time, SO_2 removal efficiencies in the 70 to 99 percent range were reported for most processes under investigation, with most results in the upper half of that range. Usually reliability was not reported, so that these results probably reflect peak rather than average system performance.

An important feature of this research was the great variety of solutions that were being investigated: fuel switching (to both low-sulfur coal and to natural gas), the increased use of nuclear power for electricity generation, dispersion by the use of tall stacks, and FGD. The variety of FGD systems being investigated was impressive with NAPCA claiming "60 to 70 in various stages of development" (NAPCA 1969 p. 56). A clear reason for this interest was the emergence of local and state SO_2 control policies across the country at this time.

To implement the 1967 Air Quality Act, NAPCA established an ambitious five-year R&D plan for the control of sulfur oxide emissions (NAPCA 1968). An important feature of this program was that it investigated a wide range of control technologies, processes, and techniques. Key elements of the program were flue gas treatment, stack meteorology, fuel desulfurization, fuel conversion (gasification/liquefaction of coal), new combustion processes, industrial process control, low-sulfur fuel availability surveys, instrumentation development, and program development, analysis and evaluation.

A combination of state and local regulatory efforts and federal funding also led to a number of FGD installations during the 1960s. A combination dry limestone injection, wet lime scrubbing system was researched in the U.S. by Universal Oil Products at Wisconsin Electric Co in 1963 and 1964, and by Combustion Engineering (CE) at Detroit Edison. Both were pilot scale applications, but CE's experience later led to three full-scale installations in the U.S. The first (in 1968) was at Union Electric's Meramec station in St. Louis. It was installed in anticipation of SO_2 emission limits to be promulgated by Missouri regulators and was guaranteed by CE to achieve removal efficiencies of 80 percent for SO_2 and 98 percent for fly ash. Similar CE systems were installed in two units at Kansas Power & Light's Lawrence Station in 1969 and 1971. However, all of these installations proved troublesome and experienced serious plugging in the boiler and air heater. After four years, CE ceased marketing this type of system, but anecdotal stories of these difficulties echoed for a long time.

Considerable technological development was also occurring outside the U.S. Systems similar to the troubled CE FGD units were installed in the U.S.S.R. (in 1964) and in Japan (in 1966), but apparently operated with fewer problems (Slack and Hollinden 1975 p. 50). In the late 1960s, many Japanese companies began to install FGD units of several types, often on industrial boilers (Slack and Hollinden 1975 pp. 53, 101-106).

C. 1970-1976

After the January 1969 Santa Barbara oil spill and reports that Cleveland's Cuyahoga River was on fire later that summer, polls showed that the public considered pollution to be the second most important problem facing the nation (Andrews 1999 pp. 201-254; Hayes 1998 pp. 32-39, 52-65; Jones 1974 pp. 137-146, 152). Nothing had slowed emissions trends, which set new records every year from 1965-70 (U.S. Environmental Protection Agency 2000). It seemed to many that existing state, local, and federal laws -- with their emphasis on research at the federal level and regulation at the state and local levels -- were failing (Davies III 1970 pp. 120-144). As a result, government responses to air pollution took a different course in the U.S. after 1970, and in doing so dramatically altered the pace of technological change.

C.1 Government Actions

The 1970 Clean Air Act Amendments created major new powers for the federal government, specifically giving the newly-created EPA responsibility and authority to set National Ambient Air Quality Standards (NAAQS) and to regulate *new* sources. States remained responsible for the regulation of *existing* sources. It is hard to overstate the popular support for this law: it was introduced by a President (Nixon) who had no previous interest in the environment but was desperate to find a popular issue to divert attention from the Vietnam War, and it was passed in Congress by a vote of 474-1 (Andrews 1999 pp. 227-229; Bailey 1998 pp. 144-155). The 1970 Amendments also represented a radical change in U.S. legislative style insofar as they contained very specific, "agency-forcing" provisions; including explicit schedules for EPA action and national air quality improvement (Ackerman and Hassler 1981; Marcus 1980). These included a very tight deadline for attainment of primary SO_2 ambient standard (i.e., by May 31, 1975), thus signaling a potentially large increase in the market for SO_2 control technologies in the U.S.

The primary NAAQS standards are designed to protect human health and welfare, and EPA was expressly prohibited from taking into consideration economic or technical feasibility when setting them. Within nine months of the promulgation of a NAAQS, each state was required to submit to EPA a State Implementation Plan (SIP) describing how it would attain and maintain the NAAQS. SIPs were required to demonstrate attainment "as expeditiously as possible, but … in no case later than three years from the date of the approval of such plan." Secondary standards were to be achieved within "a reasonable time." Once the EPA Administrator approved a SIP, it became enforceable as a matter of both state and federal law (Findley and Farber 1992 p. 103).

The 1970 Amendments also addressed new sources by directing the EPA to set nationally consistent New Source Performance Standards (NSPS) for several categories of stationary sources, including fossil-fired steam electric generators. These standards were to be set in a manner that would take advantage of the "best system of emission reduction which (taking into account the costs of achieving such reduction), the Administrator determines has been adequately demonstrated." (Ackerman and Hassler 1981, p. 11) In December 1971, EPA proposed an SO_2 NSPS for new and substantially modified electric generators at an allowable emission rate of 1.2 pounds per million Btu (lb/mmBtu) of heat input. This standard was based on EPA's determination that the ability of scrubber technology to eliminate at least 70 percent of a coal burner's SO_2 emissions had been adequately demonstrated, and was based on an uncontrolled emissions rate for eastern coal of about 4 lb/mmBtu.

The 1971 SO_2 NSPS was controversial. For one thing it required an actual reduction in emissions and did not allow for dispersion through the use of tall stacks.[66] Low-sulfur coal was very expensive and was not thought to be suitable for boilers designed for high-sulfur coal. Natural gas was even more expensive (Ellerman 1996). Coal cleaning, the only other alternative to scrubbing, was widely understood to be inadequate for meeting the new standard. Thus, the 1971 NSPS appeared to require that all new power plants either use low-sulfur coal or scrubbers. At the time, the electric power sector was building many new plants and planning even more – partly due to enormous growth projections and partly due to the Northeastern Blackout of 1969, which had elevated the industry's concerns about reliability and supply.

By 1970, almost 40 years of development and experience with FGD technologies existed, beginning with the initial work at Battersea. FGD units had been installed on a handful of power plants, smelters and industrial plants. However, it was widely understood that these applications were not good models for SO_2 control on a modern power plant. The old British scrubbers were not transferable to other facilities, while smelter and industrial facility exhausts were characterized by much higher SO_2 concentrations. Furthermore, the literature on these early applications indicates that they did not operate very reliably and there are frequent references to bypassing the scrubbers altogether. The industry placed a premium on reliability since electricity needed to be generated on demand; thus plant owners feared that unreliable scrubbers would have a negative impact on availability and overall system reliability. Of course, they were also worried about costs. Thus, despite considerable research and several pilot-scale test facilities, only four scrubbers had been installed on commercial power plants in the U.S. by 1970.[67] All were Combustion Engineering designs and all experienced considerable operating difficulties, several of them had already been abandoned. By the end of 1971, three more commercial-scale systems had been added, but there was very little operating experience with them. The design removal rates of these FGD devices were 50 to 70 percent, though it is not clear that actual emissions were reduced by this much due to poor FGD reliability.

There is debate as to whether EPA's 1971 NSPS was "technology forcing." Though a handful of commercial-scale systems existed, reliability problems remained unresolved (reliability typically ranged from 30 to 60 percent) and none were routinely operating at more than 50 percent control effectiveness. For an industry under great pressure to reduce emissions, expand, and avoid further blackouts all at the same time, this level of performance was just not acceptable. At the same time, other potential control technologies (e.g. coal gasification) were widely acknowledged as many years from commercialization (EPA Office of Research and Development 1976). In this sense, the NSPS standard was technology-forcing. However, it is important to note that several state SO_2 control programs were already in place before the EPA set the NSPS standard and several of these aimed for similar control levels (EPA Hearing Panel 1974; Erdman 1975; Pernick and Knight 1975; Slack and Hollinden 1975).

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[66] Importantly, tall stacks remained an option for *existing* sources regulated by the SIP process.

[67] This data, and subsequent data used to construct graphs of scrubber installations comes from a U.S. Department of Energy source, specifically EIA Form 767 from 1998 plus (Srivastava, Singer et al. 2000) and (Soud 1994). The raw data is adjusted for entry errors and differences in reporting.

A 1970 report by the National Research Council's Ad Hoc Panel on Control of SO_2 held that, "contrary to widely held belief, commercially proven technologies for control of sulfur oxides from combustion processes do not exist." The panel did predict however that the limestone injection should be commercially demonstrated in 1 to 3 years, whereas sulfur recovery processes (which had been the subject of the greatest R&D efforts to date) were still 3 to 10 years away. As late as 1976, EPA Administrator Russell Train would admit: "We recognize that FGD technology is not yet a mature technology with all problems resolved to everyone's satisfaction" (EPA Office of Research and Development 1976).[68]

From 1971 to 1976, state efforts were inconsistent, some states failed to enforce regulations altogether while others developed regulations more stringent than NSPS (EPA Hearing Panel 1974 pp. 8, 11-13; EPA Office of Research and Development 1976 pp. 16-18). Two important responses to state SO_2 control requirements in the 1970s were a significant shift to low-sulfur fuel oil and the construction of tall stacks to promote the dispersion of emissions (Ackerman and Hassler 1981, p. 35; EPA Hearing Panel 1974 pp. 1, 2, 11, 13; Williams, Justus et al. 1974). The former was relatively easy and inexpensive given the very low (and declining) price of petroleum at the time, so that by 1973, 21 percent of all fossil-fueled electricity generation in the U.S. was oil-fired (Energy Information Administration 1999 p. 213). This effect was particularly noticeable in the Northeastern states, where, as a result of state air pollution control efforts, coal shipments fell from 70 percent of all utility fuel burned in 1964 to 15 percent in 1973 (Alm and Curham 1984). The OPEC oil embargo of 1973 and subsequent petroleum price increases thus created a major complication for the electric sector's efforts to comply with SO_2 emission requirements. At that point, low-sulfur fuels of all types suddenly became "scarce" (Commerce Technical Advisory Board Panel on Sulfur Oxide Control Technology 1975; EPA Hearing Panel 1974; EPA Office of Research and Development 1976), a perception that applied to low-sulfur coal as well as petroleum products. Although very large reserves of low-sulfur coal existed west of the Mississippi River, low-sulfur coal was expensive and thought to be incompatible with Midwestern and eastern boilers.

Diluting emissions through the use of very tall stacks was feasible insofar as the only objective at the time was attainment of ambient SO_2 standards (acid rain was not yet recognized as a problem). Since nonattainment of the SO_2 NAAQS was generally limited to large urban areas, dilution was thought to be a wholly appropriate and adequate solution. Consequently several states, primarily in the Midwest and Southwest relied heavily on tall smokestacks in their SIPs (Williams, Justus et al. 1974).

EPA officials, meanwhile, remained much more interested in add-on controls such as scrubbers, because of the problems with low-sulfur western coal discussed above. Although research into several alternatives to FGD technology (such as chemical coal cleaning, fluidized bed combustion, solvent refined coal, and low-Btu gasification) was already underway these options were not considered by Agency officials to be even potentially competitive with FGD until the early 1980s. EPA considered physical coal cleaning and the blending of low- and high-sulfur coals appropriate only for plants facing modest reduction requirements. Finally, although EPA officials very reluctantly accepted tall stacks as a control method in the early 70s, they began to reject this approach outright by the middle of the decade as new findings on long-range sulfate transport

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[68] Of course, Mr. Train also noted that the electric power industry did not have much incentive to resolve such issues and had generally not been aggressive in doing so.

emerged (EPA Hearing Panel 1974; EPA Office of Research and Development 1976). However, it took some time for legislation and regulation to catch up.

In support of its position favoring FGD, the EPA engaged in multiple R&D activities during this time period. First, starting in 1967 and lasting throughout the 1970-76 period, the EPA and its predecessors began funding the Tennessee Valley Authority (TVA) Office of Agricultural and Chemical Development to prepare cost estimates of various FGD processes.[69] Second, the EPA established the very influential Shawnee test facility in 1972, in cooperation with TVA. This facility, which was equipped with three 10 MW boilers, provided valuable operating data on scrubbing, beginning with lime/limestone systems (EPA Hearing Panel 1974 p. 13). Third, in 1973 the EPA began its financial commitment to a program of biannual SO_2 Control Symposia, which continues to this day. Fourth, in March 1974, the EPA funded PEDCo-Environmental Specialists, Inc. to evaluate, on a bimonthly basis, the status of FGD technology in the U.S. (Devitt, Isaacs et al. 1976). These FGD evaluations continued into the late 1980s. Fifth, the EPA engaged in cooperative research and demonstration activities with utility/vendor teams, and in 1975 signed a Memorandum of Understanding with the newly formed Electric Power Research Institute (EPRI) to "facilitate sharing of technical information and cooperation of RD&D projects."[70] Finally, the federal Non-nuclear Energy Research and Development Act, enacted in December 1974, authorized EPA's energy/environmental control technology program which proved to be particularly important in the late 1970s (Zimmerman, Forrest et al. 1980).

C.2 Technological Advances

The early 1970s saw the rapid evolution of SO_2 control technologies for coal-fired power plants, along with a sharp increase in the number of commercial installations. In a survey article, Strauss provides a good summary of how the future of SO_2 controls was viewed just as the 1970 Clean Air Act Amendments were being developed (Strauss 1971). At that time SO_2 emissions were expected to increase to as much as 60 million tons annually by 1985 without any controls (pp. 96-98). Even with controls, annual emissions of 25-45 million tons were predicted. The assumptions used to generate these estimates are illuminating: both envision increasingly tight standards (ranging from 80 to 90 percent control) but for *new* facilities only. Neither scenario included any retrofits of existing power plants.[71] Notably, predicted declines in SO_2 emissions after 1970 were attributed to the "successive introduction of nuclear power plants as a replacement for fossil fuel-fired plants."

As already noted, only four commercial scrubber units were operating in the U.S. when the 1971 NSPS was promulgated (Meremac, two at Lawrence, and Hawthorne). The newest two had been operating for about a month and the oldest was to be permanently shut down later that year. The five succeeding years saw the total number of commercial scrubber units grow by over five times. Increasingly this growth was due to NSPS requirements: though most systems were initially installed on existing units, retrofits accounted for a smaller percentage of total installations after 1975. (Retrofitting an existing power plant with FGD was generally 25-30 percent more expensive than installing FGD on new facilities).

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[69] TVA had begun conducting air pollution research in the mid 1950s (Hein, Phillips et al. 1955).

[70] Comments by Steve Gage; Office of Energy, Minerals, and Industry; EPA (EPA Office of Research and Development 1976 p. 9-12)

[71] At the time, it was widely believed that the existing coal-fired power plants would be retired over the subsequent several decades, further emphasizing the importance of new plants over existing (Ellerman 1998b).

The dominant type of FGD technology installed from 1970-76 was a lime/limestone system, although utilities had begun to investigate less expensive dry scrubbing processes wherein a limestone slurry is sprayed into the hot post-combustion exhaust gas stream. In addition, a few commercial regenerable processes were tried in the early 1970s (Ponder and Christman 1975). According to survey of FGD technologies by PEDCo-Environmental Specialists, Inc. in 1976, removal efficiencies ranged from 40 to 90 percent during this period and controls had been installed on units of varying size (30MW-800MW+) and with coals of different sulfur content (Devitt, Isaacs et al. 1976).

BOX: Typical Experiences With First-Generation FGD Units in the U.S.

Duquesne Light Company's Phillips Station, Pittsburgh, PA

This 387 MW plant installed a venturi wet lime scrubber which was started up in July 1973. The FGD operated for three months before it shut down to resolve problems with equipment erosion, acid condensate in the stack, and fan stress. The unit was restarted in March 1974. Ductwork changes to accommodate the FGD disrupted operation of the electro-static precipitator which overloaded the scrubber with solids and filled the disposal pond in two weeks. Fan stress, corrosion (pH dropped to 2.0 in some parts of the system), and severe erosion continued. Solids build-up in the absorber vessels, breakdown of the lime slaker, excessive water blowdown, noise complaints and other problems plagued the system. Repairs and modifications were field engineered in cooperation with the vendor (Pernick and Knight 1975).

Potomac Electric's Dickerson Station, Gaithersburg, MD

This 190 MW plant installed a Chemico-Basic Mag-Ox an FGD designed to scrub half of the flue gas which began operation in September 1973. In the first four months of operation the FGD was available 27% of the time vs. a boiler availability of 95%. The reagent preparation systems experienced on-going problems with plugging in the mixing tank and suction lines. The system operated intermittently from April to August 1974. From August to December the FGD had an availability of 34-64%, largely due to corrosion and failure in carbon steel piping. The system was shut down in February 1975 for extended maintenance (Erdman 1975).

Boston Edison Mystic Station, Everett, MA

This 150 MW oil-fired unit burned high-sulfur fuel and was part of a test program in 1973-4. It used a regenerable magnesium oxide process and sold SO_2 to a sulfur plant. A substantial part of the of the cost of the FGD system was paid for by the EPA. Scrubber availability was very low the first year (50%) but increased in the second year (the last four months of data averaged 69%). The most significant problems were plugging of some equipment and the erosion/corrosion of pumps, valves, and piping. As a result of this project, "Boston Edison considers that the technical feasibility of magnesium oxide scrubbing and regeneration on an oil fired generating unit has been demonstrated" (Irving 1975).

Louisville Gas & Electric's Cane Run Station, Louisville, KY

An American Air Filter wet carbide lime FGD was installed on a 178 MW unit burning 4% sulfur coal. The FGD was started up in August 1976 and encountered serious problems: a pressure drop across FGD and ductwork that limited output to 150 MW; stack corrosion required relining and gas reheat; demister and spray header corrosion required modification. Unit was ready for compliance testing in August 1977 and ran satisfactorily after that. Repairs and modifications were done on site with good support from vendor (Ness 1977). LG&E was cited in the 1993 compliance hearings as one of the few companies that had made a serious commitment to FGD and had hired qualified chemical process operators.

The RD&D being conducted by various vendors and plant operators in the 1970-76 period focused in large part on reliability issues. The text box above summarizes the major reliability concerns of this period. In addition to reliability, "nearly all representatives of EPA, utilities, and FGD system vendors stated or implied that sludge disposal was a significant problem having potential environmental implications" (EPA Hearing Panel 1974 p. 51).

Of course, not all FGD systems performed poorly, even at start-up. The 1973 Compliance Hearings noted the success of Louisville Gas &Electric's Paddy's Run Station, Japan Synthetic Rubber's Chiba Wellman-Lord system, and Mitsui Aluminum's Miike Station (EPA Hearing Panel 1974 pp. 5,6). The Japanese experience is noteworthy because it included successful, early (1971 and 1972) full-scale retrofit installations. One 156 MW unit used a wet carbide lime process and operated with near 100 percent availability without any of the scaling, plugging and corrosion commonly reported at some U.S. installations.

The key difference seems to have been accurate control of process chemistry, particularly pH. As EPRI later reported in a 1979 paper, attention to the details was key: "the reliability of pH sensors in the full-scale scrubbers which have been evaluated cannot be overemphasized as one of the most important features of full-scale systems which operated successfully on a long term basis" (Jones, Hargrove et al. 1979). Japanese firms were clearly determined to make FGD systems work well and recognized that controlling the process chemistry was necessary to doing so, a similar level of commitment and interest was not displayed by most U.S. power companies at the time. More generally, Japanese industrial management in the 1970s was considered far ahead of U.S. practices. Industry leaders, government officials, and researchers from the U.S. began to travel to Japan to learn how "lean production" and "total quality management" could be applied to U.S. manufacturing. This issue was widely recognized in the automobile context (Bloomfield 1991; Cusamano 1985; NRC Automobile Panel 1984; Rae 1984); anecdotal evidence suggests that there may have been similar differences between the electric power industries in the two countries, especially given the relatively poor management typical of regulated monopolies.

Problems can also be traced to the shortcomings of U.S. R&D efforts during this period. An important new development in this regard was the formation of EPRI in response to supply reliability concerns.[72] Although focused primarily on developing new power generation and distribution technologies, EPRI was launched with an environmental division that quickly began to focus on FGD technologies. EPRI's role eventually proved to be crucial. The application of a rigorous, organized scientific approach to resolving scrubber problems was necessary to understand the root causes of poor FGD performance. Moreover, EPRI's leaders signaled an important shift in industry thinking when, at the 1976 Symposium on FGD they issued two challenges. First, the recalcitrant power industry should assume responsibility to make scrubber systems work with the same priority assigned to other power station systems. In other words, plant operators were being

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[72] Unfortunately, some of the provisions in the CAA may have had a role in pushing private industry out of the R&D market, "The current set of legislative mandates to EPA … does not take full advantage of self-interest by instituting incentives for private parties to perform research, especially on pollution control technology…. Some legislation may even have the effect of discouraging private research initiative. As a consequence, the government is forced to conduct research that might be more efficiently performed in the private sector. … The validity of research conducted by EPA to support its decision-making will always be suspect merely because the agency is … in the adversary process of regulation and standard setting" (National Research Council 1977 p. 14). Nonetheless, EPA, DOE, EPRI, power companies, and control equipment makers were able to at least partially overcome this problem.

challenged to make environmental performance an important organizational goal and to stop depending on vendor guarantees under fixed-cost contracts. Second, vendors themselves were challenged to develop a standardized hybrid system that incorporated the best components from the many scrubber designs then available (Nannen and Yeager 1976).

To back this challenge up, EPRI later announced a five-year, $10 million R&D program emphasizing the development, evaluation, and reporting of SO_2 control technologies (Morasky and Dalton 1977). EPRI's plan included characterization of individual full-scale lime/limestone FGD units selected to represent state-of-the-art installations. These units would be fully characterized; all liquid, gas, and solid streams would be monitored, and all engineering designs, decisions and compromises would be reviewed and evaluated. Importantly, responsibility for process innovation and design would be shared by the power companies and FGD vendors, whereas it had historically been the responsibility of the vendors alone. A key objective of the research program was to develop the design and operating support basis by the first quarter of 1978 to allow the electric power industry to confidently purchase large-scale, closed-loop, lime/limestone scrubber and associated byproduct disposal systems for high-sulfur coal combustion.

According to PEDCo-Environmental Associates, Inc., by 1976, the reliability of units had improved to the point that the average availability of scrubber units ranged "from about 80-95 percent depending upon the system and the averaging period" (Devitt, Isaacs et al. 1976 p.24). Other technological improvements by 1976 were in the area of limestone utilization and waste disposal. The TVA Office of Agricultural and Chemical Development reported in 1976 on recent lime and limestone technological developments (McGlamery, Faucett et al. 1976 p. 88). One important trend was that companies began to install redundant FGD units as insurance against reliability problems. This tended to increase capital costs on a dollars per MW basis, but was sometimes economic from the standpoint of increasing availability. Other major changes included:

- the use of spray towers in place of mobile-bed scrubbing devices,
- installation of series hold tanks, magnesium addition and chloride ion control,
- pH and stoichiometry adjustments for increasing limestone utilization,
- mist eliminator wash cycles and hot air injection or recycle reheat schemes to increase operating reliability, and,
- sludge oxidation for more effective disposal.

The period from 1970-1976 saw significant improvement in FGD technology, both in terms of the technology itself and insofar as it was recognized as an integral part of the power station rather than an unwanted add-on that could be by-passed as necessary. At the beginning of this period, utilities relied primarily on vendor design and performance guarantees, and installed and maintained the equipment in an uneven manner. Vendors each had their own proprietary design based on in-house R&D, engineering judgement, and experience. There was often poor understanding of and ability to control the fundamental process chemistry or the nature of stress on material in the FGD unit. Not surprisingly, there were successful experiences and horror stories. However, the most disconcerting feature of the technology at that time was the inability to predict the reliability of any

FGD system in advance. This was partly due to the large number of different systems that had been developed, and the wide variety in material and equipment choices that had been made in their design. Thus, what was learned by solving a problem at one facility often could not be applied to others. In essence, power companies did not have knowledge necessary to purchase and operate satisfactory FGD systems. For these reasons, perhaps the most important R&D development during the 1970's was the increased funding levels and industry commitment brought about by the formation of EPRI.

In sum, by late 1976/early 1977, many problems with SO_2 controls remained, a large variety of processes were in various stages of development and commercialization, a few reliable FGD installations were already in operation, and an organized research effort to improve the state of the art was planned.

D. 1977-1989

D.1 Government Actions

In January 1975, President Ford began a process that led to the first major revisions of the Clean Air Act by submitting the proposed Energy Independence Act to Congress. This bill would have extended deadlines for meeting the NAAQS (which had been previously extended because of the earlier energy crisis) and prevented EPA from setting air standards more stringent than the primary or secondary NAAQS. In subsequent congressional hearings, both environmentalists and power company representatives called for reform of the 1970 Clean Air Act Amendments. Environmentalists were concerned about the deadline extensions and the inability of various regions to meet the primary, health-based ambient air quality standards.[73]

In addition, environmental concerns began to shift to more complex and larger-scale phenomena such as stratospheric ozone depletion, global climate change, and the transboundary air pollution problem of acid rain (Clark and Dickson 2000). The ability of SO_2 emissions to acidify susceptible locations several hundred miles away had been first discovered by scientists in Europe in the late 1960s (Bolin 1972; Cowling and Nilsson 1995; Oden 1967). Researchers in the U.S. initially reported similar effects in the late 1970s (Galloway and Cowling 1978). Vigorous debates about this phenomenon ensued over the next several decades, and major research efforts were launched in both Europe and North America (Boehmer-Christiansen and Skea 1991; Forster 1993; National Acid Precipitation Assessment Program 1991b; Office of Research and Development 1989; Organisation for Economic Cooperation and Development 1979; Regens 1993). During this period, private industry demanded convincing proof that emissions from coal-fired power plants caused specific damages, generating calls for ever more research. At the same time, power companies were concerned about the difficulty of reducing emissions in light of "a lack of 'clean' fuels and effective control technology" (Bailey 1998 p. 184-199).

After a lengthy process, a new set of Clean Air Act Amendments were finally signed into law by President Carter in 1977. Major provisions of this legislation included deadline extensions, mechanisms to allow for the construction of new sources in non-attainment areas provided they

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[73] EPA Administrator Russell Train announced on May 30, 1975 that thirty-four of the nation's 247 air quality control regions would be unsuccessful in meeting primary NAAQS for SO_2 (Bailey 1998, p. 184).

installed “Best Available Control Technology” (BACT), and policies effectively prohibiting the use of tall stacks as a control measure in many applications.

Particularly relevant to the electric power industry, but largely unnoticed, was Section 111 of the Amendments, which directed EPA to implement, within one year, a new NSPS for SO_2 emissions. This requirement contained both a rate-based emissions limit and a requirement to use FGD, an approach called “universal scrubbing”(Findley and Farber 1992 p. 105). The provision was the result of lobbying by “a bizarre coalition of environmentalists and dirty-coal producers [that] tried to exploit congressional ignorance to serve their own, mutually incompatible purposes” (Ackerman and Hassler 1981 pp. 27, 35-38). Environmentalists wished to reduce SO_2 emissions as much as possible, while eastern coal producers wished to preserve the market for high-sulfur coal.[74] If scrubbers were required, this would effectively preclude switching to low-sulfur western coal as a compliance option. Most people in the power industry were unaware of this provision which created both surprise and alarm in the industry when it was discovered (Ackerman and Hassler 1981 pp. 48-54; Bailey 1998 p. 198).

Intra- and inter-agency conflict marked the development of final regulations for the new SO_2 NSPS required under the 1977 Amendments (Ackerman and Hassler 1981 pp. 79-103). Participating in this debate was a new entity, the U.S. Department of Energy (DOE), which had been established in 1977 to take responsibility for coordinating a comprehensive national energy plan.[75] DOE and the power industry were strongly opposed to universal scrubbing due to its cost and wanted to be able to use less expensive fuel switching options. Numerous proposals were suggested, some of which would have made it impossible to use most of the coal produced in Ohio, Illinois, Indiana, northern West Virginia, and western Kentucky. As might be expected, legislators from these states, with Senate Majority Leader Robert Byrd of West Virginia at the fore, fought vigorously against such proposals.

The solution to the NSPS problem eventually emerged from FGD research that indicated that “dry scrubbers” (also called spray dryers) were much cheaper than the wet FGD technologies used so far. However, they could only achieve removal efficiencies of 70 percent or less. In 1979, EPA finally issued a new SO_2 NSPS for coal-fired boilers.[76] It was set at 1.2 lb/mmBtu and a 90 percent reduction or 0.6 lb/mmBtu and 70 percent reduction (Alm and Curham 1984, p. 108). This regulation made the practice of fuel switching to lower sulfur coals alone insufficient for compliance with NSPS since a substantial reduction in emissions leaving the boiler (i.e. FGD) was required even if the uncontrolled emissions were relatively low.

After the 1977 Amendments, no further changes were made to the Clean Air Act until 1990. Moreover, something of a retrenchment in national environmental policy occurred during the 1980s. The causes of this retrenchment included a slackening of public interest in environmental issues after 1978,the anti-regulatory position of the Reagan administration, the departure from Congress of key

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[74] Ackerman and Hassler (1981) further claim that the scrubbers had a “symbolic value” for environmentalists and for the EPA, which also drove this decision.

[75] The Department of Energy Organization Act brought together into a cabinet level department such federal government energy-related organizations the Energy Research and Development Administration, the Federal Energy Administration, and the Federal Power Commission (Zimmerman, Forrest et al. 1980, p. 3-23).

[76] As a note, in the next section we will argue that at this date, FGD was understood well enough that this was *not* an instance of technology forcing, although further progress in FGD design and operation would be made.

environmental leaders in the early 1970s (Senator Muskie and Representative Rodgers, in particular), disagreements between newly-powerful members of Congress (especially Senators Byrd and Mitchell, and Representatives Dingell and Waxman), the election of a Republican majority to the Senate in 1980, and the publication of an influential study that highlighted the costs of air pollution control (Bailey 1998 pp. 207-218; Cook and Davidson 1985 pp. 50-54; Kraft 1991; National Commission on Air Quality 1981; Vig 1994). At the same time, the pressures of the oil crises had tended to cause both Congress and the EPA to back off from enforcing environmental laws, as several high-level witnesses later testified in Congressional hearings (Bailey 1998 pp. 210-220). Subsequently, analysts who focused on congressional action during this period tended to view the Reagan years as relatively unimportant in the sense that there was no roll-back of environmental legislation (and indeed even some new environmental legislation). Other analysts, focusing on the role of the executive branch, concluded that EPA's ability to function effectively was severely compromised during this period by budget cutbacks, ideologically-selected leadership, and new bureaucratic obstacles (Bailey 1998 p. 208, Vig 1994 pp. 76-79, Kraft 1991 pp. 25-28, 35-36).

Nevertheless, the retrenchment was not complete and regulatory signals continued to drive technological innovation in SO_2 control. While environmentalists at the time were frustrated by a lack of progress towards environmental goals, the Reagan Administration was similarly frustrated by its inability to roll back environmental regulations to a significant extent. In the Gorsuch and Watt scandals, and more subtly (but perhaps more importantly) in the Administration's failure to deliver promised legislation to reform the Clean Air Act, the anti-environmental positions of President Reagan and his advisors proved to be out of step with public and Congressional opinion (Stockman 1980; Tobin 1984). Indeed, though little legislation passed, dozens of clean air bills were introduced in every session of Congress in the 1980s, most of them focused on controlling acid rain. Together these efforts provided the momentum for the Clean Air Act Amendments of 1990 (Bailey 1998 pp. 216-227). Finally, the public was joining environmental groups in record numbers during the 1980s and polling results indicated a steady rise in support for environmental protection over the course of the decade.

Further impetus for increased SO_2 controls came from growing international pressure to reduce acid rain (Alm 1997; Boehmer-Christiansen and Skea 1991; Levy 1994; Organisation for Economic Cooperation and Development 1979; Shaw 1993; Wettestad 1997). Germany played a leadership role after the discovery of extensive air pollution damage to that nation’s forests in the early 1980s, a discovery that led to a very rapid rise in environmental concerns among German citizens. In 1983, stringent regulations were quickly adopted and over a period of four years Germany installed FGD systems on 35,000 MW of generating capacity. More than 90 percent of the installed units used wet lime/limestone processes that produced saleable, wallboard quality gypsum. Austria and Holland followed West Germany’s lead. In 1988, the UK’s electric power industry placed an order for over 4,000 MW of FGD systems. It would not have been difficult to conclude that further control of SO_2 in the U.S. was only on hold.

Other Congressional actions in the 1980s would have supported this view, especially the creation of DOE's Clean Coal Technology Demonstration Program (CCT Program) (DOE Office of Fossil Energy 2000). This program began in 1985 as a modification of the largely failed, Carter-era program to develop synfuels (gasoline substitutes). One of its original (and ongoing) goals was to stimulate technological advances in the area of SO_2 control. When a joint commission of U.S. and Canadian negotiators later recommended such a research effort, the CCT program was given a powerful boost. At the time, the chief environmental issue motivating the CCT Program was acid rain (and the potential for related regulation), so both SO_2 and NOx emissions controls were investigated. To date, this program has resulted in $5.4 billion of research and development investment, two-thirds of which have come from industry (although less than 15 percent of the total has gone to emissions control technologies).

Congress took several other steps during the 1977-89 period that strongly increased the perceived need for SO_2 control technologies. One was the cancellation of the fast breeder reactor program. This step, in addition to the widespread cancellation of conventional nuclear reactors after the 1979 Three Mile Island meltdown changed industry's view of the future. With limits to the further expansion of nuclear energy (once projected to significantly replace coal), there would be a greater need for coal-fired power plants. This helped prompt efforts to extend the life of older power plants (Ellerman 1998b). Another Congressional step was the passage of the Industrial Fuel Use Act of 1978 which largely precluded the use of natural gas as an industrial or electric power plant fuel, thereby delaying market penetration of this essentially sulfur-free fuel. Had the market been left alone, the current shift to natural gas might well have started sooner and recent declines in SO_2 emissions might have begun earlier. A third Congressional action was the launch of an extensive research program on acidification known as the National Acidic Precipitation Assessment Program (NAPAP). This effort continued through the entire 1977-89 period and even somewhat beyond (Cowling 1992; National Acid Precipitation Assessment Program 1991a; National Acid Precipitation Assessment Program 1991b; Oversight Review Board 1991). NAPAP's findings were highly controversial, especially in the mid-80s, but they increasingly pointed to the need for SO_2 emissions control in order to deal with acidification (Cowling 1988). Thus, for all these reasons, progress on the scientific, technological, and political fronts occurred throughout the Reagan years.[77]

D.2 Technological Advance

The pace of FGD installations increased in the late 1970s and continued to be strong until about 1986, as seen in Figure IV-1. The two curves plot the number of installations and the total installed capacity (measured in MW of generating capacity), and both follow the same trajectory, suggesting that, on average, scrubbers did not grow appreciably in size. By the end of 1980, 52 of the 380 coal-burning facilities operating in the United States had FGD systems.[78] The capacity of plants with such systems totaled 31 GW or 13.2 percent of all U.S. coal-fired capacity.

However, Figure IV-2 shows that an important change occurred in terms of which facilities were being equipped with FGD. Specifically, retrofits become much less common after 1977 than they had been before. Almost all the additional scrubbing capacity seen in Figure IV-1 was thus driven by the NSPS “full scrubbing” requirement. Most of these FGD units were wet lime/limestone systems or spray dryers, technologies developed in the preceding decade which were becoming increasingly reliable. According to an EPRI report in 1977:

> “Flue gas desulfurization is at an intermediate point of maturity. It has been around since the 1930’s but only active since the late 1960’s.

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[77] Discussion of events from 1988 and 1989 that led up to the passage of the Clean Air Act Amendments of 1990 is deferred until the section below on the post-1990 time period.

[78] Many facilities have more than one boiler and some have more than one FGD installation, so comparisons are difficult to make.

> First-generation processes are being commercially applied on a large scale, but there has been little process optimization and there is still development interest in second- and third-generation processes that can reduce costs, improve performance, and give better products" (Rochelle 1977 p. 1-11).

The research needs discussed in this report were very much in line with the research program EPRI had previously announced: chemistry control, systems integration, and advanced system design (including forced oxidation and the use of chemical additives, both of which have since become commonplace). The report also pointed out in greater detail the need for organized research as opposed to disconnected, ad hoc attempts to incrementally improve scrubber technology (Rochelle 1977 pp. 5-7). An important goal remained the improvement of scrubber reliability through the minimization of component corrosion.

In general during the 1977-89 period, EPRI continued to focus on individual FGD subsystems (including reheaters, demisters, contactor-absorbers, recycle tanks, and dewatering devices) to ensure that they were properly designed and could provide acceptable individual and system reliability. EPRI also assumed responsibility for the continued operation of TVA's Shawnee Test Facility, an effort in which EPA also participated. Some new FGD processes were also investigated at this time.

Industry reports on R&D related to SO_2 control during the early 1980s are illuminating. First, most technical papers of the time list the potential for acid rain-motivated legislation as the primary impetus for the research. Continued needs to reduce costs and to comply with state and NSPS requirements are also often cited. Second, EPA and DOE supported several important R&D projects through this time period. Third, the use of organic acid additives to greatly increase SO_2 removal was first investigated in the early 1980s (Delleney, Hargrove et al. 1984). Fourth, a better understanding of the importance of coal chemistry was gained, and the factors that influenced reliability came to be better understood as well. Reliability of FGD systems increased, and 80-90 percent availability could be achieved in lime and limestone based scrubbers (Black & Veatch Consulting Engineers 1982; Doctor 1983; Roop and Pflug 1984). Finally, it was already clear that fuel switching to lower sulfur coals was a much more important means of reducing SO_2 emissions from power plants than any other method, a fact hinted at by Figure IV-2 (Hollinden, Maxwell et al. 1984).

Most FGD units installed during the 1980s were second generation technologies. They represented great advances in the implementation of processes first developed in the 1960s and 1970s, including the use of chemical additives, plus improvements in system design and construction. In general, the challenges articulated by EPRI in 1977 had been met five years later. The primary result of these efforts was to increase FGD reliability, with less improvement in costs and performance. After 1982, there are few complaints of reliability problems and at least some FGD technologies (e.g. lime/limestone scrubbers and spray dryers) were fully commercialized by that time.

Figure IV-1: Cumulative U.S. FGD Installations

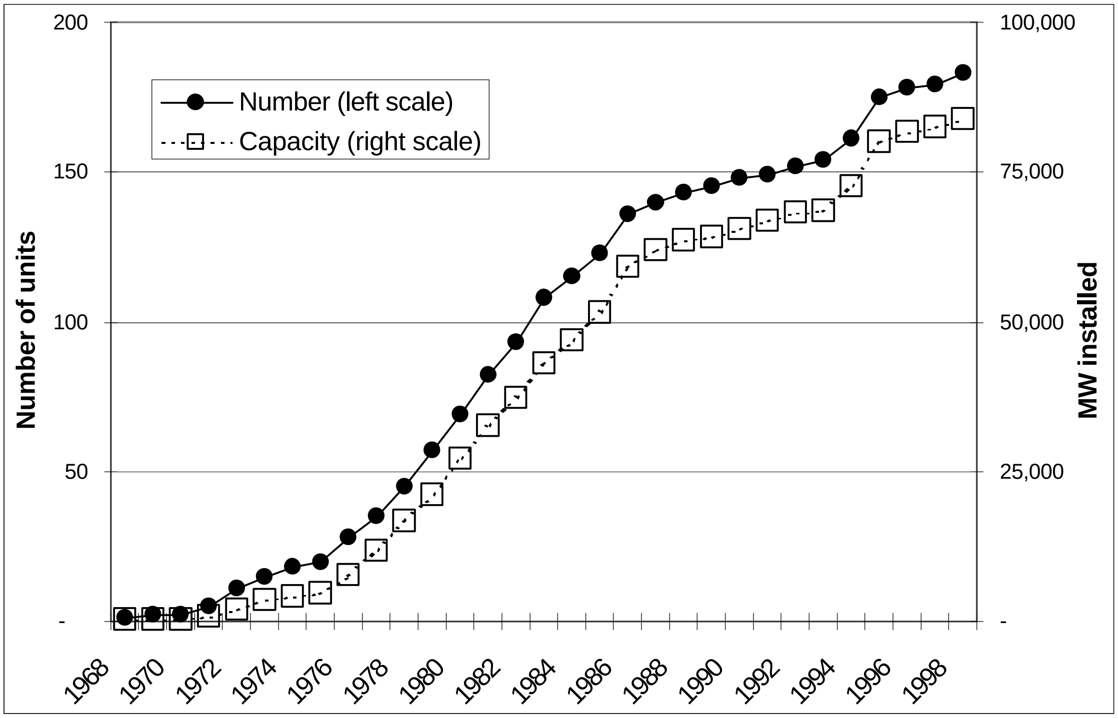
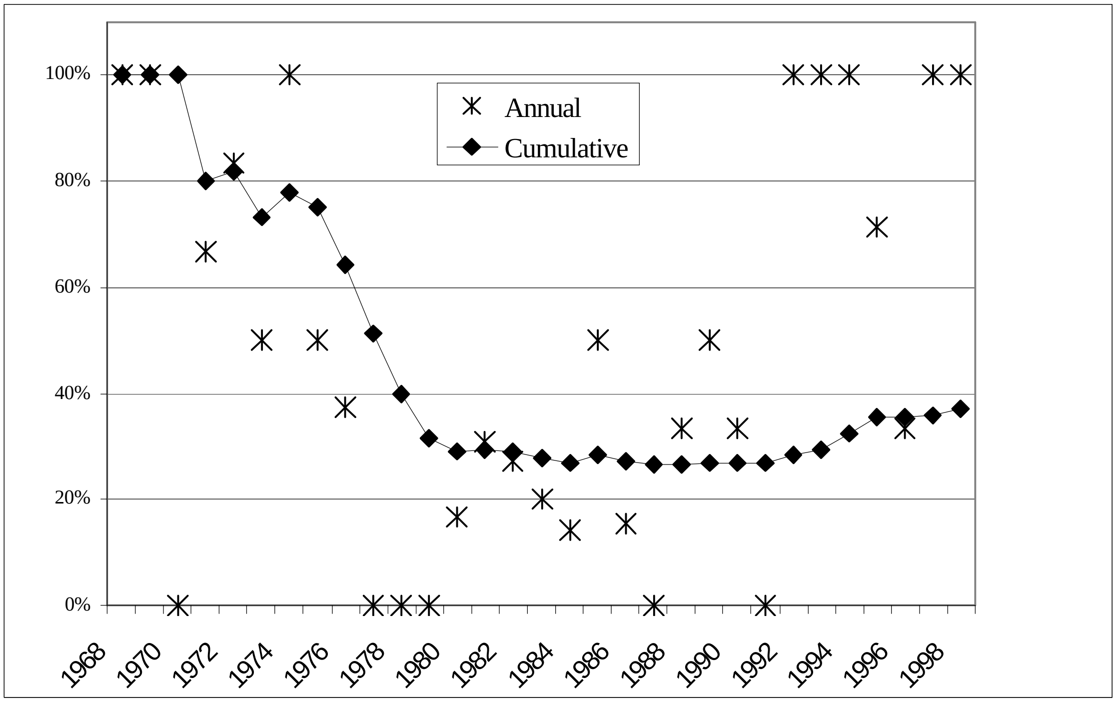


Figure IV-2: Retrofits as a Percentage of all U.S. FGD Installations



By the end of 1989 about half of all the environmental control projects that the CCP Program has supported since its inception (twenty) were already underway (DOE Office of Fossil Energy 2000 p. 3-6) These projects served as the basis for the third generation of FGD technologies, which would become commercialized during the 1990s. It is worth noting that the development of this third generation was considerably better thought-out and better funded, since by the time it was begun several key institutions had developed significant research capabilities and budgets, including EPA, DOE, and EPRI. However, the costs for this development effort were also considerably higher than were the costs of the previous generations of FGD technology R&D, perhaps 3 or 4 times as much.

The power industry reluctantly learned to live with FGD during the 1977-1989 period, but continued to view it as an unnecessarily expensive outcome of overreaching legislative and regulatory processes. FGD installations in the U.S., as a percent of installed coal capacity, leveled off by the end of 1980's at 25 percent, but many more were installed in Germany. Considerable technological progress had been made by that time, new units were reliable and their costs were declining. Moreover, national SO_2 emissions had declined by about a quarter from 1970 levels to about 16 million tons by the late 1980s and almost all of the U.S. had attained ambient air quality standard for SO_2. However, power plant emissions had declined less than 10 percent, and emissions from coal-fired plants specifically had declined by only 3.7 percent (U.S. Environmental Protection Agency 2000 p. 3-12). Thus, coal-fired power plants accounted for almost two-thirds of total U.S. SO_2 emissions. Most of these emissions were being released from tall stacks which promoted their long-range transport in the atmosphere, a situation that would eventually create pressure for further legislative action on acid rain.

E. 1990 and beyond

E.1 Government Actions

Throughout the 1980s, the scientific evidence that power plant SO_2 emissions caused acidification grew and public concern about the problem of acid rain grew even more (Clark and Dickson 2000). Much of the relevant research came from the NAPAP program, but some important findings emerged in Europe as well where concerns about acidification had first been raised and where SO_2 control policies were already in place (Cowling 1982; Levy 1994; Organisation for Economic Cooperation and Development 1979; United Nations 1996; Wettestad 1997).

The year 1988 was particularly important since several events occurred which were to break the gridlock in air pollution control legislation of the previous decade. Most important was the decision by Vice President Bush to push for acid rain legislation as part of his Presidential election strategy. In a 1988 campaign speech, Bush claimed to be an environmentalist in the image of Teddy Roosevelt and, noting that 'the time for study alone has passed,' promised legislation to cut millions of tons of SO_2 emissions by 2000 (Bailey 1998 pp229-230; Vig 1994 pp. 80-81). When Bush entered office he did so with a public mandate for air pollution legislation, an accomplishment that he believed could help him emerge as a leader from Reagan's shadow. Once George Mitchell of Maine replaced Robert Byrd of West Virginia as Senate Majority leader after the 1988 election, the leadership on both ends of Pennsylvania Avenue was pulling in the same direction with respect to acid rain legislation (Bailey 1998 pp. 227-228).

In addition, the media had begun to give more and more attention to environmental issues in 1988. The long, hot summer of that year helped raise the visibility of the climate change issue, which in turn brought attention to many other air pollution problems (Clark and Dickson 2000). When the *Exxon Valdez* oil spill that same year helped push pro-environmental poll results to overwhelming levels, there could be little doubt that acid rain would finally be dealt with (Bailey 1998 pp. 227-228). However considerable negotiations still lay ahead

The 1990 Clean Air Act Amendments (1990 CAAA) have been the subject of numerous books and papers, partly because they provide a classic example of the modern legislative process, complete with organized interest groups, secret negotiations and complex scientific arguments (Bryner 1995; Cohen 1995; Joskow and Schmalensee 1998; Keohane, Revesz et al. 1997). Title IV of this landmark legislation pertained most directly to power plant SO_2 emissions; these were to be regulated under a national Acid Rain Program using a novel approach to regulation that included both a cap on emissions and a market-based allowance trading system (Schmalensee, Joskow et al. 1998).

The cap-and-trade system introduced under Title IV was very different from the regulatory mechanisms that had previously been used to control power plant SO_2.[79] Under this program regulated firms must hold allowances equal to their total emissions. Each source was given an initial allocation based on a reduction from its historic emissions proportionate to the overall cap. Firms then had the option to: 1) control emissions to exactly match their allocation, 2) buy additional allowances in place of reducing emissions, 3) "overcontrol," and bank allowances for use in future years (when fewer allowances will be allocated), or, 4) overcontrol and then sell the excess allowances created as a result. The regulatory changes introduced under the 1990 Amendments were enormous: existing as well as new power plants were to be regulated, total emissions were to be capped (whereas previously, individual facilities were only capped as to their emissions *rate*), and firms were given almost complete flexibility in terms of how they could comply.

The new law also addressed the need for accurate measurement of emissions and accounting of allowances under the cap-and-trade program (White and Mitnick 1992). This produced an impetus for the development of emissions monitoring technologies. Importantly, regulated firms view emissions monitoring much differently in a cap-and-trade program than they do under other sorts of regulations. With a cap limiting total emissions and an active market in allowances, it was in the interest of firms to measure their emissions carefully, especially since the estimation techniques specified by EPA for estimating emissions in case of a failure of the monitoring system were very conservative. Thus, electric utilities demanded accurate, reliable monitoring SO_2 emissions monitoring systems from vendors.

The Acid Rain Program has two phases. Under Phase I (1995 through 1999) 261 electric generating units (called Table A units) at 110 plants in 21 Eastern and Midwestern states (representing 17% of U.S. generating capacity) were regulated at an effective aggregate rate of 2.5 lb./MBTU.[80] One of the most important effects of a cap-and-trade program is that emissions begin to mean money, either the cost of buying allowances, or the cost of forgoing allowance sales. In

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[79] The underlying health based ambient quality standards remained in place, but since acid rain control meant lower emissions than those required to meet the ambient standards, they faded in importance.

[80] That is, the number of allowances each firm receives is calculated by multiplying their fuel consumption for a baseline year by 2.5 lb/mmBtu.

addition, because plants do not have to meet any particular standard, provided they hold the requisite number of allowances, firms have much more flexibility in choosing their compliance strategy.

Although the basic cap-and-trade structure of the Acid Rain Program was fairly simple, it contained a number of rather complicated features designed to make it work better and/or make it politically feasible. First, it was possible to buy allowances, either at an auction (held annually from 1993 through 1999), or directly from the EPA. Second, unregulated units could participate in Phase I (i.e. “opt-in”) under several different provisions (Markey and Moorhead 1991; Schmalensee, Joskow et al. 1998; Zipper and Gilroy 1998). Third, extra allowances were granted to various entities based on several different factors. Importantly, 200,000 allowances were given in each year of Phase I to Midwestern states to offset the higher costs faced by these high-emission states (Bryner 1995 p. 166; Keohane, Revesz et al. 1997). In addition, firms could obtain extra allowances by engaging in voluntary emission reduction activities of various kinds, including demonstrating advanced FGD technologies. The allowances available through this program constituted a sizeable subsidy, $450 million at current allowance prices, and two or three times that value at the allowance prices forecast in the early 1990s. Among other things, these provisions encouraged firms to reduce emissions before the beginning of the regulatory period in 1995. These flexibility mechanisms were fairly widely used: for instance advanced FGD systems were demonstrated at 19 plants.

Not only was the Acid Rain Program much more flexible than previous command-and-control style regulation, it entailed many more uncertainties as well. The central uncertainty was whether any emission allowances would be traded at all, and if so at what price. This source of uncertainty was exacerbated by the fact that firms were required to submit compliance plans to EPA by February 15, 1993. This deadline fell before EPA’s final rules were proposed, before the first allowance auction was held, and before continuous emissions monitors were required to be installed at Phase I facilities (Burtraw 1996 p. 82). Thus, firms had to make many decisions with highly incomplete information. In EPRI workshops held in 1992, 60 percent of utility respondents cited “uncertainties” as their greatest concern about the 1990 CAAA (Rittenhouse 1992). Further, state actions to protect domestic coal industries further complicated things. In at least five states (Kentucky, Illinois, Indiana, Ohio, and Pennsylvania) attempts were made to restrict fuel switching or allowance trading to protect the local coal industry (Ellerman and Montero 1998). More recently, states like New York, have acted to restrict trading to address perceived adverse environmental impacts at the state or local level (Winebrake, Farrell et al. 1995). All of these actions were either withdrawn, or struck down by the courts, but they did cause some firms to install FGD systems as a hedge against the uncertainty associated with the trading program.

Further, market-based approaches, while generally popular with industry for the flexibility they afford, also create more compliance decisions for regulated firms. These compliance decisions begin to look much like investment decisions; firms can choose from a rich menu of choices and can select a portfolio of options with a specific risk/reward structure (and other features) best suited to their individual situation. Thus, both the amount and the nature of uncertainty faced by many power companies have changed. Compliance risks are increasingly borne by shareholders (rather than customers) and are managed with new financial tools (rather than through legal proceedings). Importantly, increases in fuel price (such as may be associated with switching to low-sulfur coal) can no longer be passed through automatically to customers, but become part of the cost basis for the electricity generator. In short, numerous uncertainties were important in the early SO_2 emissions market. These included the willingness of firms to sell allowances and the treatment of allowance purchases and sales by public utility commissions (Gildea 1992; Goodrich 1991; Mitnick, Brown et al. 1992; Solomon and Rose 1992; Wald 1993).

One other crucial public policy development needs to be mentioned: power sector restructuring. This term describes a set of government activities of various sorts undertaken since 1978, when the Public Utilities Regulatory Policy Act (PURPA) began to change the structure of the electric power industry from the regulated monopoly arrangement established in 1935 by earlier regulation (Bohi 1997; Borenstein and Bushnell 2000; Brennan, Palmer et al. 1996; Joskow 1997). Restructuring (sometimes called deregulation) essentially involves establishing competitive markets for power generation while retaining regulated monopolies for power distribution, with transmission left somewhere in between. The effects of restructuring on the environmental regulation of the power sector only became significant in the latter half of the 1990s (Farrell 2000). One of its most important effects has been to greatly increase uncertainty in both input (fuel) and output (power) markets as the latter have moved away from traditional long-term contracts, fixed rate-of-return regulation, and monopoly franchises.

Restructuring had two strong effects on patterns of technological change with respect to SO_2 controls. The first was to dramatically change the meaning of, and elevate the importance of, compliance costs for power generators who for the first time had to compete with one another. The second effect was a rapid reduction in funding for EPRI, since competitive firms had little incentive for cooperative research (Morgan and Tierney 1998). In sum, after 1990 there were new pressures for power plants to reduce SO_2 emissions, but these pressures were structurally different than they had been under monopoly regulation.

It is currently unclear how Phase II of the Acid Rain Program will unfold. During this phase the number of regulated units will increases to include all units of 25 MW or more, for a total of about 2500 units at 1000 power plants. Further, the overall emissions cap will be tightened to the equivalent of an average emissions rate of 1.2 lb/mmBtu. Interestingly, this is the same value proposed for the SO_2 NSPS by the first EPA Administrator in 1971. This rate would now be applied to all plants, existing as well as new, albeit through a flexible trading approach.[81] Meanwhile, despite the fact that current control levels are, in aggregate, not more stringent than those applied to new sources in the 1970s, it is nevertheless probably true that changes in the structure and regulation of the electric power industry have created even greater incentives for innovation in the 1990s.

E.2 Technological Advances

Given the changes introduced by the 1990 CAAA, and the very different economic pressures faced by the industry since, it is not surprising that the pattern of technological change during this period was very different from what had gone before. It is necessary to understand the response of firms to these incentives in order to understand technological innovation during this period.

Perhaps the most important factor affecting technological responses to the Acid Rain Program was uncertainty regarding the ultimate availability and price of allowances, and the number of extra units that might use some of the special features of the law and opt into Phase I by installing advanced FGD units. Early estimates of SO_2 allowance prices ranged from about four hundred

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[81] Ruckelshaus may have had more of a precedent-setting effect than he thought!

dollars to over a thousand dollars per ton, so that when early bilateral trades and the first EPA auction results were announced in 1992 and 1993 revealing allowance prices below $300/ton, some observers were shocked. Subsequent price declines to below $200/ton and then below $100/ton over the next few years were even more surprising. Eventually good explanations emerged (Montero and Ellerman 1998; Smith, Platt et al. 1998), chief among them the lower costs and greater technological feasibility of using very low-sulfur western coals (such as Powder River Basin coal) in Midwestern and even eastern power plants. The unexpectedly low cost of western coal was in part due to the deregulation of the railroads in 1980, which in turn reduced transport costs. At the same time, technological innovation enabled the use of western coals in eastern boilers. Western coals often had chemical and physical properties – especially ash melting temperatures – that were very different from those of the eastern coals these boilers were originally designed for.

Since the Acid Rain Program has been implemented, firms have become much more sophisticated in their approach to emissions trading programs. Ellerman shows how the industry moved from a highly fragmented response in the early years to an extensive reliance on the allowance market, which ended up producing a more efficient and coordinated response (Ellerman 1998a). Further, firms are now beginning to use derivatives to hedge risks in the price of emissions allowances (Cantor Fitzgerald 1999; Natsource 1999). This is particularly interesting because it represents a type of innovation that, while driven by environmental regulations, is financial in nature, not technological. The key issue is not that such innovations help reduce pollution, but that they help firms reduce their costs and better manage risk, two priorities that were the focus of innovation in FGD technology through and after the early 1980s.

Early predictions about the Phase I FGD market had scrubber vendors initially anticipating 35-40 scrubber contracts between 1995 and 1999, and some expressed concerns about the capacity of FGD manufacturers in the U.S. to meet this demand (Burtraw 1996 p. 90; Munton 1998 p.28). However, the new market for FGD systems has so far been much smaller than anticipated, with only 27 scrubbers being installed to comply with the Acid Rain Program to date. In total, 41 scrubbers were installed on 24,000 MW of generating capacity in recent years, but eleven of these retrofit installations (Navajo, Cherokee, Hayden, San Juan, and Jim Bridger Stations) occurred in the West in response to lawsuits relating to power plant emissions and visibility degradation, while two additional scrubbers were installed for other reasons.

Table IV-1 shows the compliance options chosen by Phase I units as of 1995; FGD unit installations were chosen by only 10 percent of Table A units, although they were responsible for one-third of the emissions reductions achieved between 1990 and 1995. The majority response, both in terms of the number of plants adopting it and in terms of the emissions reductions achieved, was switching to lower-sulfur coal or blending low- and high-sulfur coals. The longstanding concerns of eastern coal companies were finally realized (to some degree) as much of the low-sulfur fuel came from the Powder River Basin in Wyoming. Interestingly, the appeal of fuel switching/blending was slow to register with some Phase I-affected firms. A number of respondents to a survey of affected utilities in the summer of 1996 indicated that they had, in fact, reversed initial decisions to scrub. Two-thirds of those who had changed their plans pointed to low-sulfur coal costs and one-third pointed to low allowance prices as the reason for this reversal, results that are supported by the trade press in the early 1990s (Greenberger 1992; Kuehn 1993; Rittenhouse 1992; Schmalensee, Joskow et al. 1998; Smock 1991).

Table IV-1 also shows how successful efforts to improve FGD technologies in the 1980s had been. A number of firms did end up choosing FGD instead of fuel switching, a result that contrasts sharply with the 1970s and 1980s, when FGD was installed *only* when it was required. By the early 1990s, reliable, moderate-cost FGD technologies were readily available and had taken their place in an integrated response to the emission reduction requirements of Phase I of the Acid Rain Program.

Table IV-1: Compliance Strategies of Units Affected in Phase I of Title IV of the 1990 CAA, as of 1995.

Compliance Strategy	Number of Units	Emissions Reduction, 1990-95 (Million tons)
Table A Units		
Fuel switching/blending	162	2.550
Obtaining allowances	39	0.100
Installing FGD Equipment	27	1.410
Using Previous Controls	25	0.130
Retiring Facilities	7	0.030
Boiler Repowering	1	0.007
Total Table A	261	4.230
Substituting and Compensating Units	182	0.420
Total Phase I	443	4.650

Source: Zipper and Gilroy 1998

Figures IV-1 and IV-2 show the curves for cumulative FGD installation and the percentage of installations represented by retrofits. The effects of the 1990 CAAA are clear in both cases. In the early 1990's, relatively few FGD units are installed, yielding a slight upward curve for this period in Figure IV-1, and most installations are a result of NSPS requirements and hence involve new facilities, not retrofits. Thus, the curve in Figure IV-2 is very flat from 1980 to 1992. After 1992, however, things begin to change as the voluntary provisions of the Acid Rain Program begin to take effect. Retrofits suddenly become the dominant type of FGD installation and in 1995 there is a noticeable jump in the number and capacity of FGD units installed.

An important development that is not reflected in these figures is a change in the type of FGD process being used. The old NSPS requirement of the 1977 CAA set a standard of 70 percent emissions control from low-sulfur coal, which could be met by using spray dryers (i.e. dry scrubbers), which are somewhat less expensive than scrubbers. Thus, spray dryer technology advanced significantly during the 1980s and many such units were installed. Under the 1990 law, firms found it in their interest to reduce emissions as much as possible with every FGD installation (this way they could delay, or avoid entirely, FGD installations on other power plants and/or generate excess allowances for sale). Thus, wet scrubbers became the dominant FGD technology in the 1990s (Torrens and Platt 1994). (Note that though the capital and operating costs of dry scrubbers are lower, their removal costs on a dollars per ton basis are higher.)

Direct government R&D efforts declined somewhat in the 1990s compared to earlier periods, but the incentives provided by the emissions trading program undoubtedly helped counteract this effect. The DOE CCT Program remained stable during this period, but EPA and EPRI efforts diminished significantly. There were two factors behind this. First, the potential for further research to improve performance of by-then reliable and highly effective FGD systems was reduced. Second, EPRI's ability to fund RD&D shrank with growing competition in the electricity industry due to restructuring.

Those developments that did occur in FGD technology during the 1990-99 period typically enhanced cost-effectiveness, as measured by capital costs, effectiveness in freeing up emission allowances, and operating costs. Both capital and operating costs for scrubbers fell considerably (see Chapter V for more details). One important reason for this was lessening concern about scrubber reliability. Not only had FGD technology itself become highly reliable by mid-80s, allowance purchases could provide an additional safety net in case of reliability problems. Thus, costly design options such as spare absorber modules were dropped in the 1990s. Additional capital cost savings resulted from a trend toward fewer, larger-capacity modules, increased flue gas velocity in the absorber, elimination of flue gas reheat components, and reduction of reagent preparation costs.

Operating and maintenance costs were also reduced due to a number of innovations. Improvements included utilization of a concurrent flow pattern to reduce the pressure drop of the system and new materials of construction such as alloys, clad carbon steel, and fiberglass to provide corrosion resistance at reduced cost. Energy efficiency improvements included operation without gas reheat, wastewater evaporation systems, and heat exchangers that used waste heat from stack gases to increase power plant efficiency. Labor costs were improved through improvements in instrumentation and controls; meanwhile, operating costs could be offset by the sale of commercial-grade gypsum from wet limestone processes. Data in EPRI's FGD Optimization Workbook illustrate this sort of decline in O&M costs over time. For wet limestone systems put in service since 1994, O&M costs generally range from $0.5 to $1.5/MWh. O&M costs for older systems (1980 and earlier) range as high as $4.0/MWh.

Incentives to achieve higher removal efficiencies also stimulated research. Thus, routine control effectiveness was improved to over 95 percent through such measures as the incorporation of additives including dibasic acid, formic acid, and magnesium compounds in scrubber designs and improved gas-liquid contact throughout the scrubber system due to improved hydraulics and ultrafine limestone grind.

The proceedings from FGD technical conferences of the 1990s (held biannually) are noticeably different from previous years.[82] They contain numerous discussions of well-engineered, low-cost, highly reliable systems that were developed on time, under budget, and worked well almost immediately upon startup. The majority of these installations use lime or limestone, but other processes are being used at full scale as well. Gone entirely are discussions of unreliable or failed systems. Finally, emissions of hazardous air pollutants, including mercury are discussed in many of the papers.

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[82] One interesting change is that previously-separate DOE-EPRI and EPA conferences on various pollutants were merged into one "Mega-Symposium" starting in 1997. This institutionalized an integrated view of power plant design and emissions control, although sessions on "multi-pollutant" approaches date to the mid-80s.

F. Looking Forward

The future of SO_2 control technologies remains somewhat unclear, but some things can be said with confidence (Platt 1997). For the short term, the most important feature of recent response to the Acid Rain Program is the substantial level of "overcontrol" in Phase I, which has resulted in a significant build-up of banked allowances. Over ten million tons of allowances were carried over into Phase II, which will reduce the need for more scrubber installations. Thus, despite the simultaneous expansion and increased stringency of the program, only ten FGD new installations have been announced recently, for a total of 4,700 MW (Big Bend 1&2, Dallman 1&2, Hawthorn 5, Homer City 1-3, and Mt. Storm 1&2).

Fuel switching will remain an important means of complying with the Phase II emissions cap, possibly limiting in the long run how many more FGD units are retrofitted on existing coal-fired boilers. Proposals for a tighter standard have been made (such as halving the national cap to about 4.5 million tons annually), which would tend to increase the number of FGD installations substantially. Another potentially important development is the extent of retirement or repowering of coal-fired boilers. New power plants are almost all combined-cycle gas turbines, which means they will have almost no SO_2 emissions (Chambers 1999; Swanekamp 1999). If these plants replace existing coal units, the supply of allowances (which will continue to be issued annually for retired plants) available to remaining plants will increase substantially. This could have an indirect impact on mercury emissions because new FGD systems can play an important role in controlling mercury emissions, but only if they are built.

Finally, any further improvement in FGD technologies is likely to be relatively marginal compared to the innovations that occurred since the early 1970s. Small improvements in capital and operating cost are likely, but siting constraints may limit the former and the maturity of the technology may limit the latter. Without the imposition of a tighter standard for SO_2 emissions, new FGD installations may occur only at new facilities, offering some opportunities for the better integration of controls for multiple pollutants. But new coal-fired power plants may not be common in the future.

G. Conclusions

Although lengthy and complex, the history of SO_2 control is consistent with key features of the previous case studies and strongly suggests a link between regulation and control technology innovation and implementation. The lessons learned in this case study are likely to apply to the case of controlling emissions from coal-fired boilers, especially since it involves the same sources and similar control approaches as those involved in SO_2 control. Specifically:

1. Private industry will conduct preliminary research if emissions control requirements are expected in the future, but environmental technologies will not be commercialized or deployed without public policy requirements, usually backed up by court decisions and enforcement actions. This has happened repeatedly over many decades.

2. The stringency of the emissions reduction requirement strongly determines the emissions control capability of the technologies that are subsequently developed, especially if the

requirement is inflexible and applies uniformly across all sources. (note that the stringency of the requirement is distinct from flexibility in meeting the requirement.)

3. Technology-forcing laws and regulations can be effective in stimulating the development of new technologies, but they do not always achieve environmental objectives. For instance, the ambient SO_2 standard was met in the U.S. largely through the use of tall stacks, not FGD.

4. Flexibility matters a great deal – policies that allow private industry to make choices about how to respond to emission requirements can yield significant innovation in many different dimensions. The emissions trading system of the U.S. Acid Rain Program is a good example; although the Phase I requirement was much *less* stringent and much *more* flexible than the technology-forcing requirements of the 1970s, it caused new improvements in FGD technologies that have resulted in control effectiveness in excess of 95% and stimulated innovation in the use of low-sulfur fuels, as well as encouraging more systemic approaches to SO_2 control.

5. Energy policies and fuel pricing have enormous effects on the progress of air pollution control, and on technological innovation.

6. Both the capital and operating costs of pollution controls continue to decline as technology evolves and as user and vendor industries increasingly become more innovative to meet established regulatory standards.

7. Well-planned, coordinated research is very important to the development of new pollution control technologies and techniques. Federal research funding was crucial for SO_2 control technologies in the 1970s and 1980s.

Figure IV-3: Change in Emission Standards and Annual Emissions of SO_2 from Coal-Fired Power Plants

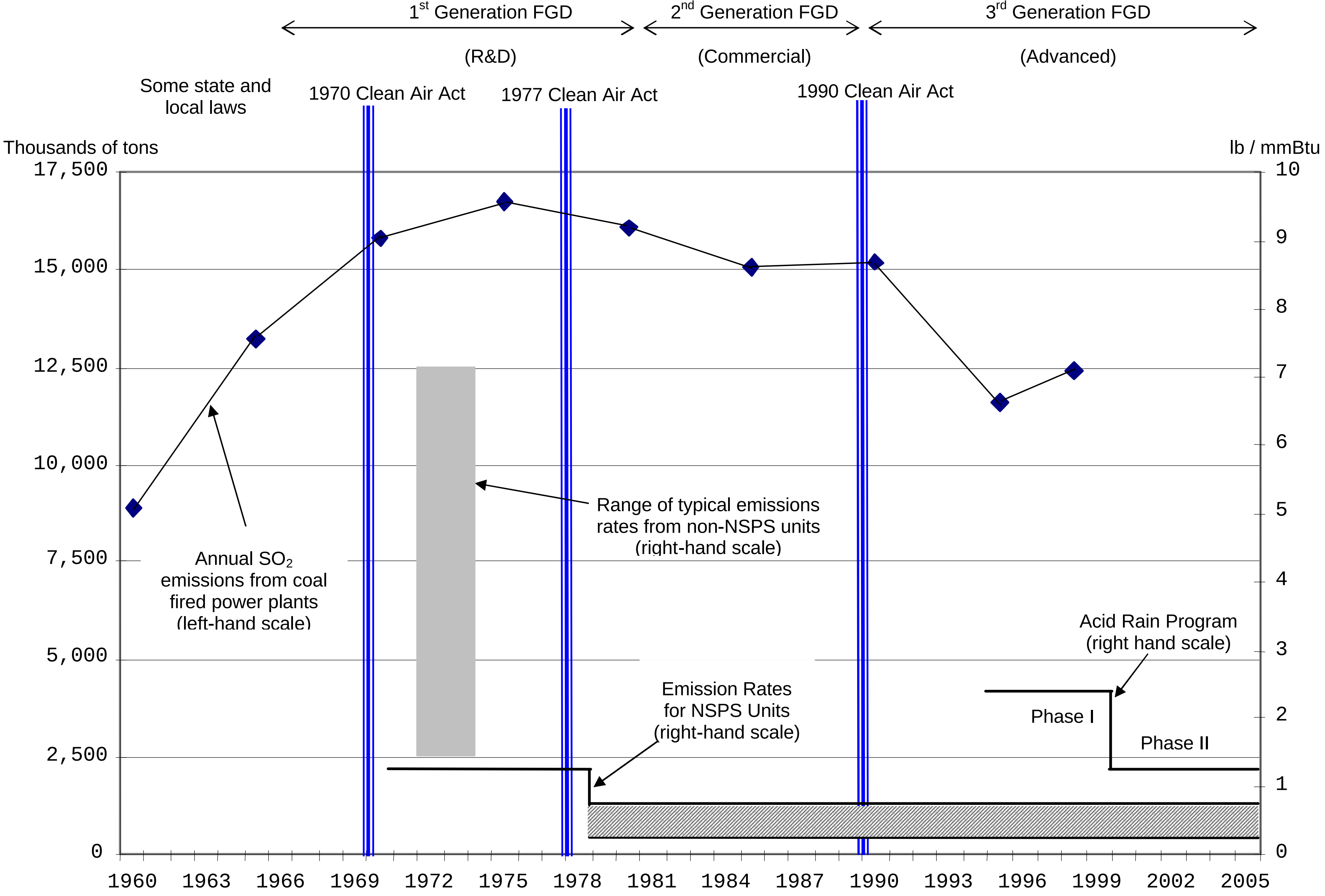

Table IV-2: U.S. Laws Affecting Air Pollution

Year	Name	Major Provisions
1955	Air Pollution Control Act	• Initial authorization for federal research, training, and demonstration projects related to air pollution. Extended in 1959, 1962, and 1963.
1963	Clean Air Act	• Established permanent funding for federal research and state planning • Gave enforcement powers to the federal government through "enforcement conferences" aimed at interstate issues.
1965	Motor Vehicle Pollution Control Act	• Gave Department of Health, Education and Welfare (HEW) authority to set automobile emissions standards as soon as practicable
1965	Clean Air Act Amendments	• Directed HEW to set automobile emissions standards
1967	Air Quality Act	• Authorized HEW to oversee state air quality practices • Created State Implementation Plan (SIP) process and provided funding • Required states to create Air Quality Control Regions • Directed HEW's National Air Pollution Control Administration (NAPCA) to investigate adverse health effects of air pollutants.
1970	Clean Air Act Amendments	• Sharply expanded federal role: newly formed Environmental Protection Agency (EPA) directed to set and enforce national ambient air quality standards • Directed EPA to regulate new stationary sources and required states to regulate existing sources and attain federal standards by 1975. • Set national standards for automoble emissions (90% reduction by 1976)
1974	Energy Supply and Environmental Coordination Act	• Delayed attainment dates for federal ambient air quality and auto emission standards until 1979.
1978	Power Plant and Industrial Fuel Use Act	• Prohibited the use of liquid and gaseous fuels in electric power plants, delaying the deployment of alternatives to coal.
1978	Public Utilities Regulatory Policy Act	• Began restructuring of the electricity industry by requiring electric utilities to purchase power from renewable and other non-utility generators.
1977	Clean Air Act Amendments	• New Source Review (NSR) requirements for SO_2 emissions from power plants – effectively requiring scrubbers on all coal-fired plants. • Created NSR requirements for new stationary VOC sources to use Lowest Achievable Emissions Rate (LAER) technologies • Created Reasonably Available Control Technology (RACT) requirements for existing VOCs sources • Established Clean Air Act Scientific Advisory Committee (CASAC) • Mandated 5-year reviews of NAAQS • Delayed automobile emissions standards • Required vehicle inspection and maintenance programs
1990	Clean Air Act Amendments	• Designation of non-attainment areas as "extreme," "severe," "serious," and "moderate," etc. and a corresponding hierarchy of NSR requirements for sources of VOCs and NOx • Required many prescriptive elements in SIPs for non-attainment areas (e.g. employer commute option, vehicle inspection & maintenance, reformulated gasoline) • Created national Acid Rain Program: cap-and-trade SO_2 control program. • Expanded the types of sources covered by VOC RACT and established NOx RACT requirements for the first time • Required that state transportation plans conform to air quality plans
1992	Energy Policy Act	• Encouraged the development and use of alternative fuel vehicles.

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V. History of Costs for the Three Case Studies

Because cost issues typically play a prominent role in regulatory debates, this chapter reviews the evolution of technology costs for each of the foregoing case studies. Two primary results relevant to the current mercury debate emerge from such a review. First, decreasing costs in each of the case studies demonstrate that well-defined performance standards encourage cost-saving innovations. Second, pre-regulatory cost estimates for major air quality regulations have been uniformly overstated, often grossly so. Although these overestimates are partially caused by a failure to account for innovation, other factors, such as conservative assumptions and poor information, also have an effect. Thus, pre-regulatory estimates, particularly those on the high end, can usually be considered to reflect worst case scenarios and do not necessarily form a reliable basis for policy decisions. In many cases, it is therefore appropriate to consider that some reasonable (and in some cases, substantial) downward adjustment in projected compliance costs is needed to account for the likely effects of innovation and commercialization.

A. Decreasing Costs as Evidence of Technological Development

A clear theme that emerges from the case studies is that regulation spurs innovation, while, conversely, the lack of regulation (or weak regulation) can actually retard innovation. Regulation provides two incentives for technological development: compliance and cost-minimization. Clearly, compliance pressures create an initial incentive to develop necessary control technologies. Once these technologies are available, firms tend to improve on them in an effort to reduce costs. Although it is difficult to separate the effects of innovation from the effects of conservative bias and poor information in accounting for the evolution of cost estimates over time, a sense of the combined magnitude of these effects may be gained by reviewing cost trends for the specific control technologies discussed in the case studies. Since it is more difficult to distinguish between the roles of innovation and conservative assumptions in the automobile case, this chapter begins by reviewing costs for power plant NO_x and SO_2 controls.

A.1 Methodology for Cost Comparison for Coal-Fired Boilers

Numerous studies of power plant control costs have been undertaken over the last 30 years. Since all of these studies vary in approach, several criteria were used to ensure a meaningful and robust analysis. First, only engineering estimates based on full-scale tests were used, not pilot studies or modeling exercises. Second, only studies that used similar assumptions about plant characteristics (boiler size, type of boiler etc.) have been evaluated. Finally, care has been taken to choose studies that were rigorous, well documented, and representative of the technical literature.

Most studies provide information for capital costs and operation and maintenance (O&M) costs to provide an estimate of total costs. Both cost components are sometimes combined and expressed in terms of levelized annual costs. In some cases, a separate analysis of capital and O&M costs can help to isolate the specific effects of technological innovation, an approach that may be instructive when considering potential mercury control costs.

Costs, Technological Innovation, and The SO_2 Case Study

The SO_2 case study provides an excellent history of how costs can decrease over time through regulation-driven innovations. Such innovations may include fuel shifts, changes in engineering practices, and modifications to the control technology itself. Scrubber controls (including wet scrubbers) evolved through three generations of technology development: research and development (R&D), commercialization, and advanced technology. Because data from the first generation are unreliable, this section focuses on cost reductions achieved in the second and third generations of scrubber technology.

Operating and Maintenance (O&M) Costs

As noted in Chapter IV, power companies, technology vendors, and other researchers focused R&D efforts on improving the operational efficiency of scrubbers. Thus, process chemistry control, the handling of scrubber waste, and the co-production of saleable byproducts (instead of waste) have all undergone considerable improvement. Not surprisingly, several studies show that O&M costs have decreased significantly over the last 25 years. The most comprehensive study (Blythe, Horton et al. 1999) covers the current O&M costs of 28 installations from 1977 to 1996. The data indicate a significant downward trend, with newer, wet limestone systems averaging $0.5-1.0/mWh, versus costs for earlier installations that averaged approximately $2.0/mWh. The data are supported by a combination of studies done by EPRI (Keeth, Ireland et al. 1986; Keeth, Ireland et al. 1991) and MIT (Ellerman, Schmalensee et al. 1997). These studies, summarized in Table V-1, indicate that between 1982 and 1991, as the second generation of scrubbers matured, total O&M costs fell by 53 percent. This was followed by another 59 percent decrease between 1991 and 1997, as the third generation of advanced scrubbers were installed. Recent papers from EPRI's annual MEGA Symposium indicate that this trend is continuing (Blythe, Horton et al. 1999; Ogi, Miyata et al. 1999; Smolenski and Murphy 1999). As more recent innovations are incorporated in upgrades to older scrubber systems, even existing installations should continue to improve their cost-effectiveness (Jozewicz and Singer 1999).

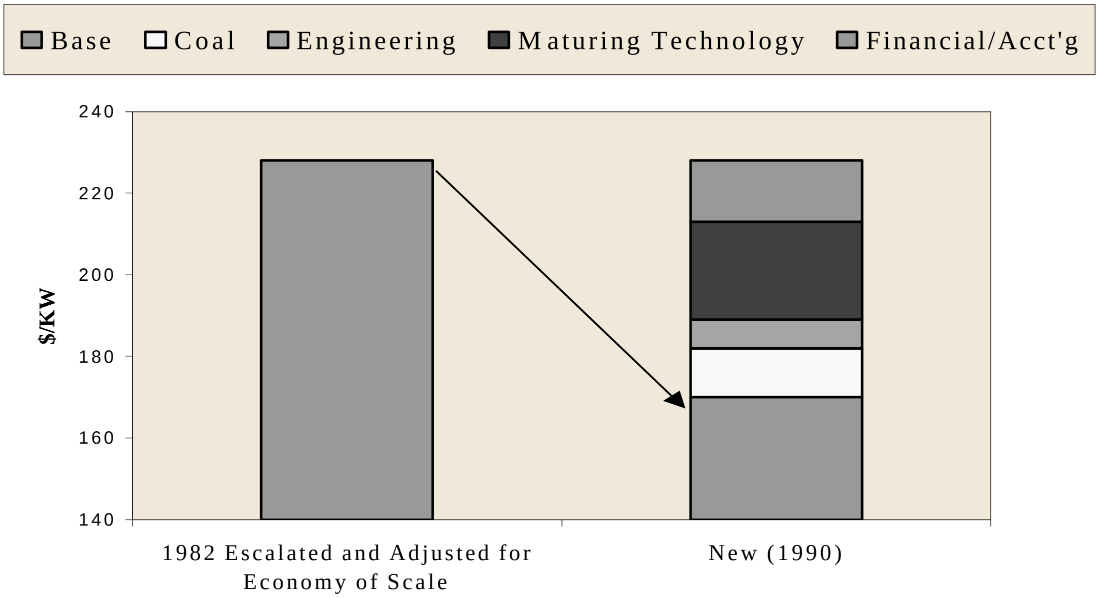
Table V-1: Changes in Limestone Forced Oxidation O&M Costs (1999 Dollars)

	Fixed O&M (mills/kWh)	Variable O&M (mills/kWh)	Total O&M (mills/kWh)	**Cost Decrease**
EPRI 1982-1986	6.3	11.0	17.3	
EPRI 1991	4.0	4.2	8.2	53%
MIT 1997	2.05	1.29	3.34	59% (Cumulative 80%)

Capital Costs

Trends in capital cost are more difficult to track over time, as they are sensitive to many different facility-specific factors and the data do not show clear trends over time. Studies that cover the period between first and second generation of scrubbers (1977-1980) show both cost increases and decreases (Laseke, Melia et al. 1982; McGlamery, O'Brien et al. 1980). A possible explanation for this mixed pattern is that efforts to increase scrubber reliability and removal efficiencies—which were considered high priorities during this time period —sometimes resulted in increased capital costs[83].

Figure V-1: Capital Costs of Wet LSFO from 1982 to 1990



Source: Adapted from Keeth et. al., 1990

Improvements made in the second generation of scrubber technology provide a more consistent picture of decreasing capital costs. Figure V-1 is taken from a 1990 EPRI study and shows capital cost declines (shown as "base" in the figure) for wet limestone scrubbers with forced oxidation (Wet LSFO) of approximately 25 percent (Keeth, Ireland et al. 1990).[84] Over 70 percent of this decrease is attributable to innovation inspired by regulatory incentives: including fuel switching to lower-sulfur coal, improvements in engineering techniques, and maturing technology. This comparison is particularly robust, since it was based on a series of studies over time by the same authors, who explicitly controlled for the confounding factors that make such comparisons difficult (Keeth, Ireland et al. 1986; Keeth, Ireland et al. 1990).

Both phases of the Title IV Acid Rain Program have provided strong incentives for reducing the costs of scrubber technology. Thus, capital costs continued to decline with the third generation of scrubbers, which were increasingly used to retrofit older boilers. Early results from Phase I

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[83] They would, however, decrease costs on a dollars per ton basis, since increased reliability and higher removal efficiencies allow a scrubber to remove more SO_2 at a fixed capital cost.
[84] This is the technology used on the majority of installations.

appeared to indicate that retrofit capital costs had bottomed out, as the data exhibited either stagnant costs (Ellerman, Schmalensee et al. 1997) or modest declines (Burtraw 1996). Since retrofits are constrained by existing infrastructure, this result is perhaps not surprising. However, the latest studies indicate that the capital costs of more recent, Phase II retrofits have decreased significantly with capital costs approximately 25 to 30 percent lower than for Phase I retrofits, and some vendors claiming reductions of as much as 50 percent (Srivastava 2000; Boward and Brinkmann 1998). These recent declines are largely attributable to design improvements such as decreasing the need for absorber modules, a step that can reduce capital costs by up to 35 percent (Jozewicz and Singer 1999). This is evidenced by a recent retrofit installation at the Big Bend station in Florida, which reported retrofit costs below $100/kW—over 50 percent cheaper than average Phase I installations. This dramatic cost reduction resulted from two innovations that grew out of experience at the same facility during Phase I implementation: the use of one scrubber to serve two boilers and a reduction in the number of absorber modules from four to one (Smolenski and Murphy 1999).

Costs, Technological Innovation, and The NO$_x$ Case Study

Whereas the previous review of SO_2 costs focuses on the period after 1971, when NSPS requirements first began to require scrubber systems for new facilities, a review of cost trajectories for advanced NO_x technologies—specifically SCR and SNCR—illustrates the cost changes that can occur within a relatively short time period just prior to and soon after regulation is introduced. As described in Chapter III, strong regulatory drivers in Japan and Germany prompted early development of these technologies overseas; hence they were commercially available to a significant extent by the time more stringent NO_x regulations were introduced in the U.S. in the 1990s.

The data shown in the tables below cover the period from 1982-1998 and include the earliest cost estimates associated with the application of advanced NO_x control technologies in the U.S. as well as the post-regulatory cost estimates based on actual case studies of boilers retrofitted with SCR and SNCR. Both SCR and SNCR cost estimates (Tables V-2 and V-3, respectively) exhibit a similar pattern. Early estimates made at a time when these technologies were still relatively new are very high. Between 1982 and 1989, estimated costs declined somewhat, partially due to increased R&D in the U.S., but to a greater extent due to rapid adaptation and employment in Germany and Japan over the same period (the declines are particularly dramatic for SCR). Estimated costs fell further in the 1990s when further regulatory incentives emerged in the U.S. The difference between early, pre-regulatory estimates and current costs is substantial: costs declined by 65 to 90 percent for SCR (on a dollars per ton of NO_x removed basis) and by approximately 65 percent for SNCR (on a levelized cost basis). Although part of this differential is attributable to overly conservative assumptions in the pre-regulatory period, in-use experience with these technologies and the innovations it inspired clearly played a major role.

Table V-2: SCR Early Engineering Costs Estimates Versus Current Costs (1999 Dollars; Based on a 500 MW, Wall-Fired Boiler)

Study	Capital Costs ($/kW)	Cumulative % Decrease	$/ton	Cumulative % Decrease
EPRI 1985[1]	90-155[4]		2,800-11,290	
EPRI 1989[2]	125	None	2,500-5,000	4-55
NESCAUM 1998[3]	50-75	40-60	1,000-1,100	64-90

[1] (Miller, EPRI Coal Combustion Systems Division et al. 1985) [2] (Eskinazi, Cichanowicz et al. 1989) [3] (NESCAUM 1998) [4] Retrofit factor of 1.24

Table V-3: SNCR Early Engineering Cost Estimates Versus Current Cost (1999 Dollars)

Study	Capital Costs ($/kW)	Cumulative % Decrease	Levelized Cost (mills/kWh)	Cumulative % Decrease
EPRI 1982[1]	29-35		3.5-3.75	
EPRI 1989[2]	6-19	45-80	4-5	Increase
NESCAUM 1998[3]	15	48-57	1.25	69-75

[1]New, 500 MW (EPA Office of Air Quality Planning & Standards 1983); [2]Retrofit, 500 MW (Eskinazi, Cichanowicz et al. 1989); [3]Retrofit, 200 MW (NESCAUM 1998)

In sum, past experience with power plant NO_x and SO_2 control technologies clearly shows that control costs can decline dramatically, both in the initial transition to regulation and thereafter, as firms strive on an ongoing basis to minimize compliance costs.

B. Pre-Regulatory Overestimates: The Roles of Innovation and Conservative Assumptions

Several observers of the regulatory process have noted that projected compliance costs are systematically overestimated, not only by industry, but by regulatory agencies as well. As shown above, one reason for this is that regulatory agencies, like industry, frequently fail to account for the cost impacts of technological innovation over time. (Goodstein and Hodges 1997; Harrington, Morgenstern et al. 1999; Morgenstern, Pizer et al. 1998). In addition, regulatory agencies -- though they may lack industry's incentive to bias cost estimates for political reasons -- must frequently rely on industry sources of data which may be obtained either directly from regulated firms or indirectly from third party consultants or equipment vendors (Harrington, Morgenstern et al. 1999).

In 1991, a study by Resources for the Future (Harrington, Morgenstern et al. 1991) analyzed ten air quality regulatory impact statements and found that EPA overestimated future compliance costs in six cases, was approximately correct in three cases[85], and underestimated once (with respect to vehicle inspection and maintenance programs). The study did not attempt to analyze industry estimates, noting that a finding of "bias in the cost estimates from industry (or environmental) sources is perhaps to be expected."

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[85] Two of these may have been overestimates as well, but the comparison was less inconclusive.

B.1 Regulatory Cost Estimates by the Automobile Industry

Isolating the costs of emissions controls is considerably more difficult for automobiles than it is for power plants. This is true for a number of reasons. First, automobile technologies are complex, closely integrated, and change rapidly, sometimes with successive model years. Second, fuel changes play a large role that is difficult to capture. Third, much of the relevant data are proprietary. Finally, many of industry’s pre-regulatory cost estimates were clearly conservative and used to delay the regulatory process. The wide gap between some industry estimates and contemporaneous government estimates during the early 1970s suggests that political motivations played some part in the dramatic cost claims being made by industry at that time.

A similar gap emerged in more recent debates concerning the costs associated with California's LEV program (Phases 1 and 2). Table V-4 illustrates the difference between estimates provided by two industry sources (ACG and AAMA)[86], contemporaneous estimates by the California Air Resources Board (CARB), and actual costs as of 1998. The table indicates that industry estimates exceeded the estimates of the CARB by as much as five to ten times. In two cases, even CARB's estimates later proved to overstate actual compliance costs; this was not the case for ULEVs which as of 1998 were costing slightly more than CARB anticipated in 1994.

Table V-4: Estimated Costs of California’s LEV Program (per vehicle)

Vehicle	ACG ‘93	CARB ‘94	AAMA ‘94	Actual
TLEV	$273	$66	$298-487	$35
LEV	$788	$120	$911-1343	$83
ULEV	$679-1,326	$227	$1,666-4,005	$251

B.2 SO2 Control Costs: The Ever Decreasing Costs of Title IV Compliance

Pre-regulatory cost estimates were similarly high prior to the implementation of Title IV SO_2 requirements. Table V-5 shows two pre-regulatory cost estimates for Phase I of the program versus actual costs calculated by MIT (Smith, Platt et al. 1998). In this case, EPA’s pre-regulatory cost range proved consistent with actual costs, whereas EPRI overestimated eventual control costs by more than 80 percent.

Table V-5: Estimates vs. Actual Costs of Phase I of the Acid Rain Program in 1995 Dollars

Study	Total Cost ($ Billions)	Average Cost ($/ton)
EPA 1990	0.45-0.86	144-208
EPRI 1993	1.34	307
MIT 1997	0.73	187-210

Source: Smith, Platt et al. 1998

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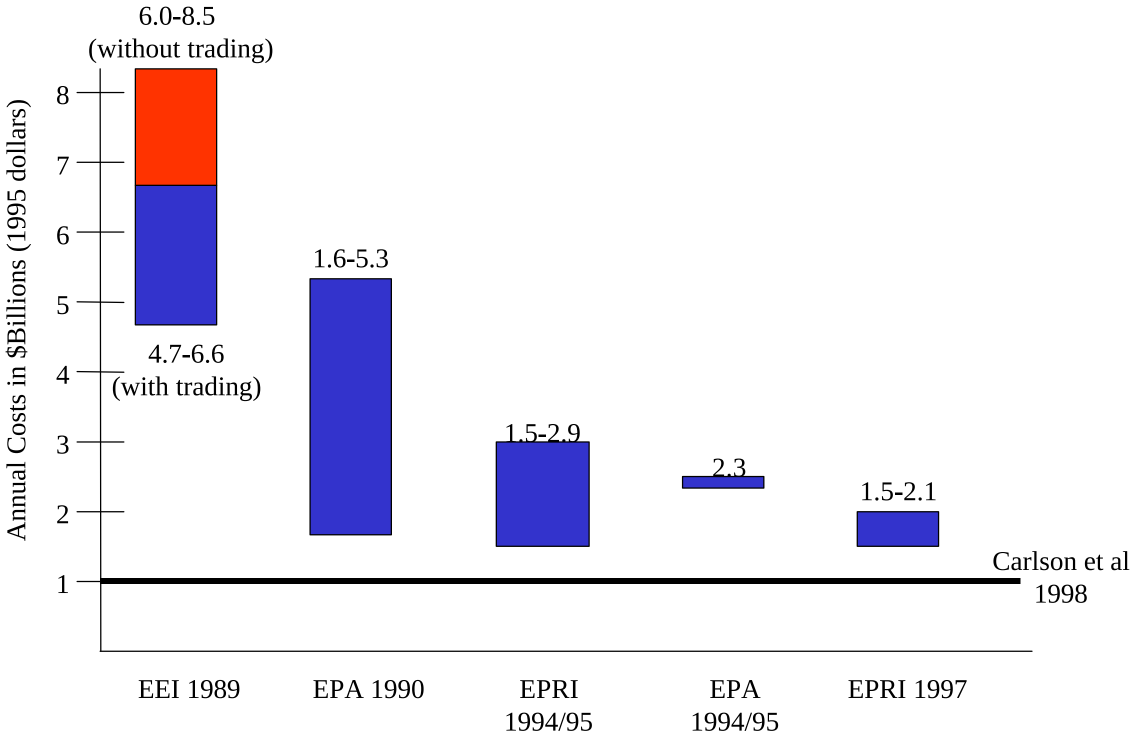
[86] The sources of these estimates are cited in Chapter II; ACG stands for the Automotive Consulting Group and AAMA stands for the American Automobile Manufacturers Association.

A classic trajectory of declining cost estimates is illustrated by Figure V-2 which shows the evolution of anticipated Phase II (which started in January 2000) control costs over time based on EPA and industry estimates. (Note that the final data point shown in this figure is taken from a comparative analysis by independent researchers (Carlson, Burtraw et al. 2000).) Both the orderly downward progression of these estimates and the magnitude of the discrepancy between early industry estimates and more recent estimates are striking. Thus, the Edison Electric Institute's (EEI) estimates at the height of the acid rain debate of the late 1980s predicted annual control costs of $4.7 to 6.5 billion with trading and $6.0 to 8.5 billion without trading. These estimates were widely cited in Congressional testimony, trade magazines, and media reports at the time and were substantially higher than the cost estimates advanced by EPA, which similarly failed to account for the eventual impacts of technological innovation in scrubber technology and the emergence of lower-cost compliance options (such as fuel switching).

The next round of estimates emerged in the mid-1990s when the political atmosphere around SO_2 regulation was significantly less charged. These estimates were significantly lower and reflected a better understanding of the flexibility gains associated with trading and available compliance options (notably, the economic viability of relying on low-sulfur coal from the Powder River Basin). By this time, industry and EPA estimates had converged and were reasonably consistent. Finally, the even lower figures cited in the Carlson study (1998) are testimony to the continued potential for cost-reductions as technologies improve and firms gain experience in selecting compliance options that are optimum for their specific units.

Figure V-2: Cost Estimates for Phase II Compliance in 1995 Dollars



Source: Burtraw 1998; Smith, Platt et al. 1998

B.3 Conservative Estimates and the NOx Case Study

Similar examples of industry's tendency to overstate costs during the pre-regulatory phase of technology development can be found for the NO_x control technologies shown in Tables V-2 and V-3. While comparable EPA estimates were not available for these technologies[87], the early industry estimates shown in these tables were issued in the context of active political debates about the need for and likely extent of future regulations. Hence, they probably reflect the conservative bias typical of industry estimates that are developed with the intent of influencing such debates. A similar tendency to exaggerate likely control costs can be found in numerous industry statements made in public forums in the 1990s (many of these claims are not documented in the scientific literature). These estimates often exceeded even EPRI's 1989 cost projections by several-fold (Amar 1997).

C. Summary and Conclusions

Pre-regulatory estimates of control costs, both by industry and regulatory agencies, tend to overstate eventual implementation costs, often dramatically. One reason for this is that cost models frequently fail to account for technological innovation, and particularly for the large degree of innovation that may be stimulated by recent market-based approaches to pollution control. A desire to influence the regulatory debate toward delaying or relaxing control requirements creates another potent reason for the conservative bias consistently associated with many early industry estimates.

The influence of both factors is evident in the case studies presented in this report. First, the impact of technological innovation both in the transition to regulation and in the period following regulation has resulted in substantial cost reductions over time for all the major control technologies involved in automobile emissions and power plant NO_x and SO_2 control. Second, industry estimates tended to diverge most dramatically from government estimates during the pre-regulatory period, when key regulatory decisions had not yet been made. It is often difficult to distinguish between these influences in the early phases of regulation, though for purposes of pending policy decisions it is perhaps only necessary to consider the aggregate effect. Unfortunately, it is probably not possible to derive a universal "adjustment factor" that can be reliably applied to pre-regulatory cost estimates. Nevertheless, the consistent and striking pattern of high cost estimates followed by dramatic cost declines that emerges from the case studies is almost certainly relevant and needs to be considered in the current pre-regulatory debate on the costs of controlling mercury emissions from coal-fired boilers.

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[87] An EPA/TVA study conducted in 1981 had an estimate for SCR. With a retrofit factor of 1.24, capital costs would have been approximately 15-35% lower than a 1982 EPRI estimate (although both were still significantly higher than the actual costs currently seen). However, the two studies didn't have comparable data for a full comparison with the other studies on a cost per ton basis (EPA 1983).

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VI. Status of Mercury Control Technologies for Coal-Fired Boilers

A. Objectives and Approach

In this chapter, we provide an overview of mercury control technology developments, their current status, and outstanding technological issues. The focus is on characterizing mercury emissions and various potential technologies and strategies to reduce them, successes so far, and concerns still remaining. It also discusses what is needed for further development and eventual large-scale commercial implementation of these technologies.

This description of current status and future needs is based on critical review of recent information available in literature (both U.S. and foreign), and is complemented by discussions with individuals actively involved in research and development, including power plant managers involved in field- testing of some of these technologies.

It is important to evaluate and understand the current status of mercury control technologies as well as the projections of their future development and large-scale implementation in the larger context of the three case studies described earlier. For example, the advanced state of technologies for controlling emissions from motor vehicles, and SO_2 and NO_x from power plants was arrived at after, and not before, strong regulatory drivers were put in place. Additionally, the innovation in alternative technologies and continued reduction in costs of control were driven by the need to apply those technologies on a large commercial scale as required by emission standards included in regulations.

The current status of mercury control technologies can best be described as "research and development" stage since there are no established regulatory drivers at the present time to make the technology move forward to the next step of commercial scale application. This chapter therefore describes the progress made mostly in bench and pilot scale studies funded by private and public sectors. The financial investments in research and development by government and industry to date are reflective of early public concern about the public health effects of mercury and the increasing possibility of potential regulatory action, but are much less than what they would be if the regulatory drivers were indeed put in place.

Even with this limited funding, the research done so far on mercury control technologies has answered many basic questions about the applicability of various technical options in the near future (2 to 5 years). One major reason for this substantial understanding of potential mercury technologies is that user and vendor communities by now have developed considerable working knowledge in pollution control as a result of past SO_2, NO_x, and particulate control requirements and many of the technologies to control those pollutants are also expected to play a large role in capturing mercury emissions. Also, other sources such as municipal waste combustors have successfully applied mercury controls. While this working experience with small sources is not directly transferable to large coal-fired boilers because of their different flue gas characteristics, it should still prove to be helpful. Considering these facts, a time frame of two to five years will allow a reasonable time to further develop and apply at full scale various control options in an environment where various technologies and strategies and their combinations compete to find the most optimum and cost effective solutions for specific coal-fired boilers.

Over this time frame of 2 to 5 years, the potential full-scale field demonstrations of technologies such as carbon (or other sorbents) injection, enhanced wet scrubbing (to control SO_2 and mercury), enhanced fine particle controls, fuel switching to natural gas or some innovative combination of these or other emerging technologies and strategies is expected to occur under a regulatory framework that sets overall emission targets and target dates without dictating specific means to meet those targets.

B. Issues with Control of Mercury Emissions

Given the extensive and extremely successful experience with control technologies for other flue gas constituents such as NO_x , SO_2, and particulates, it is fair to ask, "What is so different about mercury?"

To summarize and set a framework for the discussion in this chapter, the following major issues account for the differences and challenges associated with controlling mercury in flue gases from large coal-fired boilers:

- Very small quantities present - typical concentrations of mercury in the flue gas of a coal-fired power plant are about 4 to 5 orders of magnitude (0.01 vs. 100+ ppm) lower than those for NO_x or SO_2. A basic understanding of the physics of mass transfer associated with "trying to contact" the "hard-to-find" mercury molecules, suggests a greater technical challenge than that for NO_x or SO_2.

- Chemical speciation and physical forms present - Unlike NO_x or SO_2 (which are mostly present as gaseous NO or SO_2 in flue gases), mercury is present in both vapor form (as insoluble elemental mercury and as soluble ionic mercury (mercury chloride, mercuric sulfate, and mercuric oxide)) and in particulate form generally adsorbed on flyash. The knowledge of its physical and chemical apportionments is important because current technologies and technologies under development may be effective in capturing some forms but not others.

- Measurement - Again, the very small quantities of various chemical forms of mercury present in the flue gas have made it difficult to develop sufficiently accurate measurement devices. While this has been mostly overcome, it has had a role in slowing the pace of control technology development.

These characteristics of mercury emissions and concentrations pose challenges for future developments in power plant mercury control technologies.

C. Mercury Emissions: Forms and Measurements

C.1Emissions

EPA has estimated that during the period from 1994 to 1995 annual emissions of mercury from human activities in the United States were 159 tons (EPA 1997). Approximately 87% of these emissions were from combustion sources. Coal-fired utilities in the U.S. were estimated to emit 51 tons of mercury per year into the air during this period, accounting for about one third of total emissions nationwide.

The form of mercury emitted from point sources is a critical variable in modeling the patterns and amount of mercury deposition from the atmosphere (EPA 1997; Pai 1997). Both elemental (Hg^0) and oxidized mercury (Hg^{+2}) are emitted to the air from combustion point sources. Elemental mercury has a relatively long residence time in the atmosphere (1-2 years), while oxidized forms of mercury have lifetimes of only a few days (EPA 1997) as a result of the higher solubility of Hg^{+2} in atmospheric moisture. Elemental mercury can thus be transported over long distances, whereas oxidized and particulate forms of mercury deposit within few miles of the point of emission.

C.2Forms of Mercury and Their Effect on Removal

The speciation of mercury in the flue gas of a coal-fired power plant strongly affects the amount of mercury captured by the air pollution control devices because the capture of elemental mercury in flue gas is different from that of oxidized mercury. As discussed in more detail below, elemental mercury does not adsorb on sorbents or unburned carbon as readily as oxidized mercury. Also, oxidized mercury is soluble in flue gas desulfurization systems (FGDs) while elemental mercury is not. In order to understand the technical and economic feasibility of mercury controls in coal-fired power plants, it is therefore necessary to understand the speciation of mercury in flue gas and the potential physical and chemical interactions at various points in the system.

Mercury is present in coal in low concentrations, on the order of 0.1 ppmw (compared to sulfur which varies from less than 0.5% to over 4 % by weight in North American coals). In the combustion zone of a coal-fired power plant, all the mercury in coal is vaporized as elemental mercury, yielding vapor concentrations in the range of 1 to 20 micrograms per cubic meter. At furnace exit temperatures (1700 K), all of the mercury is expected to remain as the thermodynamically favored elemental form in the flue gas. As the flue gas cools after combustion, oxidation reactions can occur, significantly lowering the concentration of elemental vapor phase mercury by the time the post-combustion gases reach the stack. Measurements of the concentration of mercury species taken in the stacks of pilot and full scale coal combustion systems show that the range of observed values is broad; studies have reported anywhere from virtually 100% Hg^0 to 95% Hg^{+2} upstream of the air pollution control device (APCD) (Prestbo 1995; Meij 1994).

Analysis of the results from a number of utility power plants suggests that $HgCl_2$ is the dominant oxidized species in the gas (Senior 2000). The major kinetic pathway to formation of $HgCl_2$ in flue gas is believed to be through the reaction of atomic chlorine (Cl) with elemental mercury (Senior 2000; Kramlich 1997). Although the oxidation of elemental mercury in the convective section of a boiler is assumed to proceed primarily via gas-phase reaction, experimental evidence suggests that some fly ash can catalyze oxidation of elemental mercury. Iron oxide has been shown to promote this oxidation (Ghorishi 1998). Other constituents in the fly ash (carbon, calcium compounds) may also contribute. The presence of acid gases (HCl, SO_2, NO, NO_2) in the flue gas has also been shown to cause oxidation in the presence of fly ash (Carey 1998; Miller 1998). Furthermore, selective catalytic reduction (SCR) technology for NO_x control has been observed to oxidize a portion of elemental mercury (Fahlke 1995).

In summary, coal composition (primarily in terms of chlorine content and ash composition), the operation of the combustion system (primarily in terms of carbon left unburned in the ash), as well as temperature and residence time of the exhaust gas in the particulate control device affect mercury speciation in the flue gas. In turn, this affects the amount of mercury adsorbed on particulate matter, which then can be collected in a baghouse or an electrostatic precipitator, as well as the effectiveness of sorbents and scrubbers in removing mercury.

C.3 Mercury Measurements in Power Plants

Before discussing specific control technologies, it is important to note that the reliable measurement of mercury (including speciated measurements) in flue gas has posed problems to the development of mercury control technologies. This is because the actual performance of these technologies cannot be determined without sufficiently accurate measurements at the inlet and outlet of control devices. Low mercury concentrations and interferences from other species in the flue gas have contributed to the difficulty in obtaining accurate measurements. It is useful to briefly summarize the history of mercury measurement in combustion flue gas and to note some remaining technical concerns.

In the early 1990's extensive field testing programs were undertaken by EPRI and U.S. DOE to quantify emissions of hazardous air pollutants from coal-fired power plants. As part of these programs, mercury speciation was measured, even though there was no validated method for this measurement. At that time, reliable methods were available only for total mercury. Two methods were used in those studies: EPA Method 29 and the Mercury Speciation Adsorption (MESA) method. Although both of these methods measure total mercury accurately, they have been shown to speciate mercury incorrectly in certain circumstances (Brown 1999). Method 29 over-reports oxidized mercury when the concentration of SO_2 in the flue gas is greater than approximately 500 ppm. The MESA method is affected by interactions between SO_2 and NO_x. When SO_2 was present at concentrations of greater than 500 ppm and NO_x, greater than 250 ppm, the fraction of oxidized mercury was overestimated by as much as 70%. (Values of SO_2 in the flue gas of coal-fired power plants range from 300 to over 2000 ppm, while NO_x concentrations can vary from 100 ppm to over 500 ppm).

The lack of accurate speciation data hampered the development of basic understanding of mercury chemistry and mercury sorbents until about 1997. Both EPA Method 29 and the Ontario Hydro method (which is a derivative of the former method) are based on bubbling a small sample of gas through a series of liquid impingers containing different solutions. Generally, the initial impingers contain a solution that removes oxidized mercury and the final impingers, elemental mercury. Problems with EPA Method 29 centered on the interaction of acid gases with the initial impinger solutions, which allowed these impingers to capture some of the elemental mercury in addition to the oxidized mercury. Thus, the total amount of mercury was measured correctly, but the amount of oxidized mercury was overestimated by Method 29 in some cases.

This problem has largely been resolved, with the aid of concentrated efforts by EPRI and U.S. DOE to validate one of several modifications to EPA Method 29 (Laudal 1997). The Ontario Hydro method is now widely accepted as an accurate method for measuring elemental and ionic mercury in coal-fired power plants. The EPA Information Collection Request (ICR) for speciated measurement is based on the use of the Ontario Hydro method, and is described later in this chapter.

Some continuous emission monitors (CEMs) for mercury have been tested at coal-fired power plants and in hazardous waste combustors (French 1999). Currently there are no commercially available CEMs which can measure speciated mercury, although several are under development. The Ontario Hydro method (or any other impinger-based method) is a batch method, and is slow and time-consuming: several hours are needed to make a measurement (and there is considerable set-up time) and the data are not available for a few days after that. As the development of mercury control technologies broadens from the laboratory to pilot and full-scale applications under field conditions, there should be less need for on-line speciation measurements. There are several instruments currently commercially available for total mercury measurements (French 1999), which range in price from $45,000 to $140,000. As these measurement technologies continue to improve, they will help in reliably estimating the effectiveness of applied control technologies.

D. Review of Control Options

At present, EPA has not defined MACT (Maximum Achievable Control Technology) for controlling mercury emissions from electric generating, coal-fired boilers. If and when EPA establishes an emission limit for mercury under MACT, effective strategies will be needed to meet those limits. Much work has been done on options for controlling mercury emissions ranging from switching (and blending) of fuels to retrofit control technologies. A brief review of various control options follows.

D.1 Pollution Prevention Options

Several pollution prevention options to comply with potential mercury control regulations may be available to coal-fired boilers. These include fuel switching and additional coal cleaning. It is possible that one or combination of these options may prove to be the optimal choice for some boilers. For instance, fuel switching to low-sulfur coal has proven popular in complying with SO_2 controls, and has provided some affected sources with near-zero cost compliance strategies (Ellerman et al. 1997). These options require little in the way of technological innovation, as they involve well-understood technologies and/or operations in wide commercial use. Reductions in mercury emissions may be obtained by switching to or blending with fuels which have lower mercury content (Pavlish 1998). Based on data from the USGS COALQUAL database (Toole-O'Neil 1999), the mean concentration of mercury in uncleaned (or in-ground) coal is approximately 0.2 ppmw. Mean values for various important coal-producing regions range from 0.08 ppmw to 0.22 ppmw. Thus, there is room for reduction in mercury by switching to a different coal. However, this is only feasible if the other properties of a coal (e.g., heating value, ash content, and sulfur) are compatible with the combustion system. Most coal-fired power plants have a limited range of fuel properties that are acceptable.

Co-firing with or switching to natural gas is an obvious alternative whose main challenge may be the cost differential between gas and coal. On the other hand, replacement of coal with natural gas or a conversion to a combined cycle natural gas plant offers the substantial benefits of essentially zero emissions of mercury and SO_2, and major reductions in fine particles, NO_x, and CO_2 emissions.

Coal cleaning has been shown to remove some mercury from coal. Conventional cleaning methods exploit differences in physical properties (density, surface properties) between minerals and macerals (i.e., the carbon-containing portion of the coal). A recent study (Toole-O'Neil 1999) reported mercury reduction in coal obtained from cleaning 24 eastern bituminous coals. In 20 out of 24 cases, mercury was reduced as a result of cleaning with an average reduction (on an equal energy basis) of 37%. However, the mercury reduction was not consistent from coal to coal which probably reflects differences in the forms of mercury (organic versus sulfide) among the coals. Coal cleaning may be an option in some specific cases. As a large percentage of eastern coals are already cleaned, further advanced cleaning may or may not be cost effective (Srivastava 2000).

D.2 Conventional Control Technologies

Most coal-fired power plants already have air pollution control devices (APCDs) such as fabric filters (baghouses) and electrostatic precipitators (ESPs) for particulate control, scrubbers for SO_2 control and low-NO_x burners (LNBs), selective catalytic reduction (SCR) or selective non-catalytic reduction (SNCR) for NO_x control. Most of these APCD's have from minor to substantial impacts on mercury emissions as well as speciation.

Fabric Filters and ESP's

Significant amounts of mercury can be removed in baghouses (Amrhein 1999; Butz 1999) and lesser amounts in electrostatic precipitators (Amrhein 1999; DeVito 1998). Oxidation of elemental mercury has been observed in a pilot-scale fabric filter system during combustion of eastern bituminous coals, whereas a pilot-scale ESP collecting the same ash did not oxidize elemental mercury (Amrhein 1999). If there is mercury in the particulate phase at the inlet to an ESP or fabric filter, these devices should remove it efficiently. Unburned carbon has been suspected of adsorbing mercury for both eastern and western bituminous and sub-bituminous coals. Often as a consequence of using low-NO_x burners, pulverized coal boilers can produce increased levels of unburned carbon when burning bituminous coals (DeVito 1998), or less commonly, sub-bituminous coals (Butz 1999). Mercury has been found to concentrate in the carbon-rich fraction of fly ash (Li 1997; Huggins 2000). However, it is not yet possible to generalize from the limited research conducted to date about the relationship between the carbon content of ash and potential mercury emissions reductions.

In some cases, ash from western sub-bituminous coals has been observed to adsorb large amounts of mercury in the particulate collection device (particularly a baghouse) even with little or no unburned carbon present (Butz 1999). Some component of the ash from these coals appears to adsorb elemental mercury since little oxidized mercury is generally present in the flue gas from western fuels (owing to the low chlorine content of the coals). At this time, the mechanism by which elemental mercury is removed by the ash from western sub-bituminous ash is not well understood.

Adsorption of mercury by fly ash has been observed to be dependent on temperature for both eastern bituminous coals and western sub-bituminous coals (Amrhein 1999; Butz 1999; Srinivasachar 1999). Tests at the ABB-Alstohm Power Plant Laboratories (Srinivasachar 1999) pilot scale combustor showed increased capture of mercury by the ash upstream of the pilot scale ESP at 100º C (212º F) as compared to 150º C (300º F). Furthermore, a special wet ESP operating at low temperatures (75º to 120º F at the outlet) captured approximately 50% of the mercury in combustion gas from a Powder River Basin (PRB) sub-bituminous coal. In recent work reported at the A&WMA Specialty Conference on Mercury (September, 1999), (Butz 1999), tests on a pilot-scale baghouse were conducted at Comanche Station which burns a PRB sub-bituminous coal. At the nominal operating temperature of the full-scale baghouse (275º F or 135º C), the ash removed 10% to 40% of the mercury (which was predominantly elemental). Mercury removal increased to 60% when the flue gas was cooled to 230º F (110º C). PRB coals from the western United States are widely used for power generation because of their low sulfur content.

As noted above, some fly ash has been observed to oxidize elemental mercury in both laboratory scale apparatus (Norton 2000; Ghorishi 2000), and pilot-scale baghouses (Amrhein 1999). Laboratory experiments using well-controlled gas compositions indicate that the composition of the flue gas, particularly the amounts of HCl, NO_x, and SO_2, is critical for mercury oxidation. The composition of the ash is also important. Experiments with model fly ash compounds (Ghorishi 1998) exposed to mercury in a laboratory fixed bed reaction have shown that iron oxide is particularly effective in oxidizing elemental mercury in simulated flue gas. Many eastern bituminous coals and lignites contain substantial amounts of iron oxide in the ash.

Flue Gas Desulfurization (FGD)

In addition to particulate removal devices, other air pollution control technologies have been shown to affect the speciation of mercury in the flue gas as well as remove some of the mercury from the flue gas. Of the 1039 coal-fired boilers in the U.S., 159 employ wet FGD systems for control of SO_2. Using 1996 data (Brown 1999), this represents 23% of the total capacity (70 GW out of about 300GW). Recent sampling campaigns on full-scale utility boilers (DeVito 1998; Laudal 1999) and a large pilot-scale unit (Amrhein 1999) have provided data on the speciation of mercury at the inlets and outlets of the installed FGDs. Many scrubbers have been observed to remove 85-95% of oxidized mercury, but essentially no elemental mercury. Based on a detailed study of the behavior of mercury in a pilot-scale wet scrubber, the adsorption of oxidized mercury appears to be strongly correlated with the mass transfer of gas to liquid in the scrubber, usually expressed as liquid-to-gas (L/G) ratio and weakly dependent on pH of the scrubber solution (Amrhein 1999).

In addition, the type of FGD system (forced vs. natural oxidation, for example, or limestone vs. magnesium-lime) also affects the amount of oxidized mercury removed in the scrubber (Senior 2000).

Oxidized mercury is adsorbed from the flue gas into the aqueous scrubbing solution. Under some conditions, limestone scrubbers have been observed to reduce adsorbed mercury back to Hg^0 (which is released back into the flue gas and exits the scrubber) giving rise to higher concentrations of elemental mercury at the outlet than at the inlet (Amrhein 1999). Notwithstanding this phenomenon, the ability of scrubbers to reduce total mercury emissions is well established.

In the U.S., a small number of coal-fired power plants (~2% or 5 GW out of 300 GW) have spray dryer absorbers (SDAs) for control of sulfur dioxide emissions, usually followed by fabric filters. SDAs employ alkaline sorbents (similar to FGDs); an aqueous slurry of the sorbent is sprayed into a vessel with sufficient time to dry the sorbent particles such that they can be collected in a conventional particulate control device. SDAs were documented to capture mercury in a wide range (6-96%) based on data for seven installations on coal-fired power plants (Gleiser 1994). Although mercury speciation was not measured, the amount of mercury removal increased with coal chlorine content suggesting that SDAs preferentially remove oxidized mercury. The wide range seen in collection efficiency is one of the recurring problems and indicates a lack of basic understanding of some of the physical and chemical processes taking place in the control devices.

The marked contrast between the removal of elemental and oxidized mercury in SO_2 scrubbers means that it is presently difficult to generalize about the effectiveness of scrubbers for removing mercury without an understanding of mercury speciation in the system. For example, FGD systems on plants burning North Dakota lignite were shown to remove less than 5% of the mercury in the flue gas (Laudal 1999), while FGD systems on plants burning eastern bituminous coals had mercury removals in the range of 56-75% (Amrhein 1999; DeVito 1998). In both cases, the scrubbers remove 85-95% of the oxidized mercury, but in the former case (lignite) there is very little oxidized mercury in the flue gas at the FGD inlet, while in the latter case (eastern bituminous), about 80% of the mercury is in the oxidized form at the FGD inlet.

SCR/SNCR

Although SCR and SNCR systems are not yet widely used on U.S. power plants, these technologies are expected to be quite widespread in the near future (2 to 3 years) due to recent federal and state NO_x control regulations. A recent wave of planned and completed SCR retrofits supports this assumption (see Chapter III for the case study on NO_x controls).

SCR systems have been shown to oxidize elemental mercury (Fahlke 1995). Thus, when an SCR system is placed upstream of an FGD system, increased capture of mercury in the scrubber should result. At present, it appears that SCR systems are capable of increasing the amount of oxidized mercury, which should result in increased effectiveness of downstream FGDs for total mercury removal. Current work at EPRI, DOE, and EPA is targeting this research area. Specifically, EPA's ICR effort, described later, includes field studies of coal-fired boilers equipped with SCR-FGD systems, the data from which should be very helpful in answering this question in the near future. The effects of SNCR systems on mercury speciation have not been documented in the open literature to date.

D.3 Mercury-Specific Control Technologies

Various technologies have been proposed as retrofits to existing power plants in order to specifically remove mercury from flue gas. To date, no technology has been tested at full scale (i.e., in commercial utility boilers), although several pilot scale (i.e., in sub-scale combustion facilities or on slipstreams from commercial boilers) demonstrations have been made. The most widely studied systems for capture of mercury from coal-fired power plants employ sorbents injected into the flue gas and then collected in the existing particulate control devices. Oxidation of mercury in the gas phase prior to the FGD has also been tested as a means to increase removal of mercury in the FGD. Much of this work has been reviewed previously (Brown 1999; EPA 1997). In this section, we emphasize newer information, particularly that which has become available since the 1999 DOE Critical Review (Brown 1999), as well as information regarding mechanisms which could help identify and overcome current technical problems.

Sorbents

Carbon-based

The most commonly studied sorbent has been activated carbon. This material has been successfully used as a sorbent in municipal and hazardous waste combustors, although the concentrations and speciation of mercury, as well as the levels of other acid gases are typically much higher in these systems than in coal-fired power plant flue gases. Therefore, the successful application of sorbent-based technologies to large coal-fired boilers will have to consider their much lower mercury concentrations in flue gas and much higher flow rates (this is the “mass transfer” problem noted earlier).

When attempting to evaluate the effectiveness of activated carbon for removal of mercury from flue gas, it is important to understand the underlying mechanisms. The major factors which influence mercury removal include: sorbent type and properties, gas-phase mercury species (Hg^0 or $HgCl_2$), temperature, concentration of acid gases (HCl, SO_2, NO, NO_2) in the flue gas, residence time in the duct and dispersion of the sorbent in the flue gas.

Most of the experimental investigations conducted so far involve passing a simulated flue gas through a fixed bed of sorbent. This has provided much useful information on the effect of flue gas constituents, temperature and mercury speciation. Some of the key findings related to mechanisms for mercury sorption (and oxidation) by activated carbon are summarized below.

1. Capacity of activated carbons for elemental mercury and $HgCl_2$ adsorption increases as temperature decreases (Carey 1998).

2. The equilibrium capacity of some activated carbons has been shown to be higher for Hg^0 than for $HgCl_2$ (Carey 1998). (Activated carbons can be made from a variety of materials, with different surface areas, pore size distributions, etc.). However, the reactivity of a given carbon is higher for $HgCl_2$ than for Hg^0, meaning that a given activated carbon will likely adsorb more $HgCl_2$ than Hg^0 over short contact times, all other things being equal. Thus, in practice, activated carbon often appears to be a more effective sorbent for $HgCl_2$ than for Hg^0.

3. Acid gases affect the sorption of both Hg^0 and $HgCl_2$. Below about 500 ppm SO_2, sorption capacity of activated carbon increases dramatically (Carey 1998). However, most coal-fired power plants have levels at or greater than that level at the inlet to the particulate control device, so the practical effect of SO_2 may be small. HCl increases the sorption of both Hg^0 and $HgCl_2$, although its effect on Hg^0 is more dramatic (Carey 1998). In the absence of HCl or SO_2, NO_x increases the sorption capacity of both Hg^0 and $HgCl_2$ (Miller 1998). However, presence of both NO_x and either HCl or SO_2 in the flue gas seems to decrease the capacity of activated carbon for both Hg^0 (Carey 1998; Miller 1998), and $HgCl_2$. (Carey 1998). The presence of HCl, NO_x, and SO_2 also promotes oxidation of elemental mercury across activated carbon. Under some circumstances, this oxidation is accompanied by increased adsorption. NO_2, in particular, appears to affect the oxidation of elemental mercury strongly (Carey 1998).

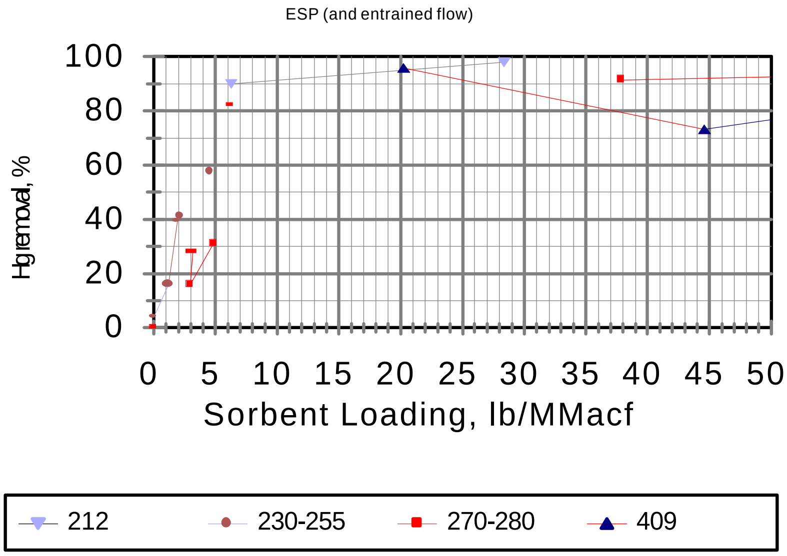
These general observations are based on research conducted on fixed bed or monolith catalysts. There is comparatively little data for activated carbon in an entrained mode (e.g. duct injection), although this would be the more economical method for the commercial implementation of activated carbon in coal-fired power plants.

Recent EPA work on entrained flow (Serre 2000) on the adsorption of elemental mercury by commercial activated carbon has revealed that the initial reactivity of the carbon, not its adsorption capacity, was important in determining the amount of mercury adsorbed on the carbon, as was previously observed. Sorbent properties such as reactivity and capacity are a function of many parameters including surface area, porosity, and functional surface form. Reactivity refers to the initial rate of reaction. Capacity refers to the sorbent ability to adsorb mercury. The rate of mercury adsorption appeared to be high initially but then decreased sharply with time. Adsorption was sensitive to temperature, decreasing as temperature increased. These experiments were conducted with large amounts of both sorbent and mercury in the gas-phase such that mass transfer limitations were minimal. The gas composition did not simulate the flue gas from a coal-fired plant, in particular because of the absence of acid gases. Therefore, the results while important to further our basic understanding, are applicable to the behavior of actual flue gases only with care.

Pilot-scale testing of sorbent injection has also been carried out for several years as better methods for measuring mercury species in flue gas have improved the understanding of the process. Duct injection of a commercial activated carbon upstream of a pilot-scale ESP was carried out by ABB in 1996-97 (Srinivasachar 1999). An eastern low-sulfur bituminous coal was burned and sorbent was injected in the duct upstream of an ESP with about a one second of residence time. The temperature in the duct and ESP was controlled at either 100^{o}C or 140^{o}C. MESA as well as Ontario Hydro measurements at the ESP inlet indicated that the gas-phase mercury was primarily in an oxidized form. At a ratio of sorbent to mercury of 40,000:1 (6.3 lb sorbent/MMacf), 82% of the mercury was removed at 140^{o}C and 90% was removed at 100^{o}C. Other pilot scale data from Public Service Electric & Gas of New Jersey's Hudson Station burning an eastern bituminous low-sulfur coal (Butz 1999) show similar trends for the injection of commercial activated carbon upstream of an ESP, as does laboratory testing of an activated carbon sorbent in flue gas from combustion of a Pittsburgh seam (medium sulfur) bituminous coal. (Morency 2000) These data are shown in Figure VI-1.

Figure VI-1 shows that for sorbent loading higher than about 5 lb per million actual cubic feet (MMacf) (typically sorbent to mercury ratios of about 40,000:1 for a mercury concentration in the flue gas of 10 micrograms per cubic meter) mercury removals of greater than 80% were obtained over a range of temperatures from 212º F to 280º F. For sorbent loadings greater than 20 lb/MMacf (typically sorbent to mercury ratios of about 150,000:1), mercury removals of greater than 90% have been obtained.

Figure VI-1: Mercury Removal versus Activated Carbon Loading for Bituminous Coal

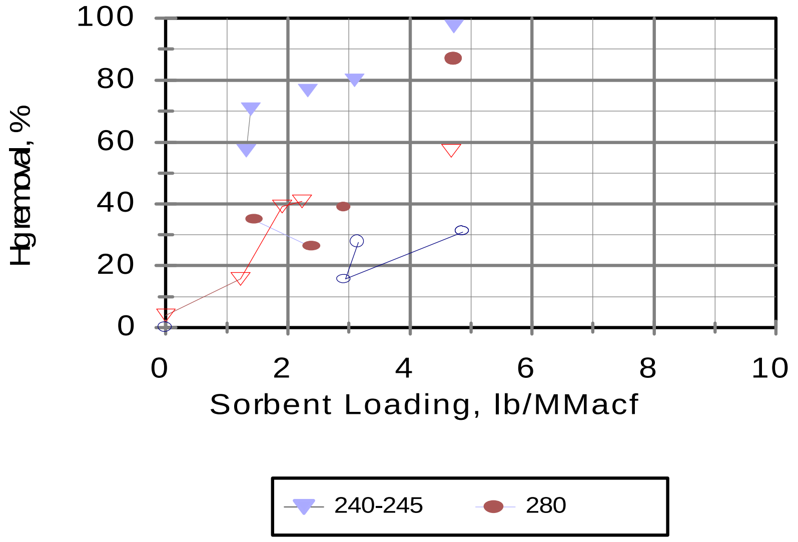


When the same sorbent was injected upstream of a pilot-scale ESP at Public Service of Colorado’s Comanche Station (Brown 1999), very different results were obtained. Greater than 50% capture was achieved at a sorbent injection rate of 3.5 lb/MMacf for temperatures ranging from 270º to 350º F. In this case, however, the native fly ash accounted for about half of the mercury capture.

The different results between the Comanche and the Hudson tests (Butz 1999) were largely due to differences in fuel and resulting flyash. Hudson Station burned a low-sulfur, eastern bituminous coal, while Comanche burned a western sub-bituminous coal from the Powder River Basin. In the former case, most of the gaseous mercury in the flue gas was oxidized at the ESP inlet, while in the latter case, the gaseous mercury was predominantly elemental. Furthermore, the fly ash from Comanche Station, which is high in calcium oxide and low in iron oxide, was observed to adsorb, on average, 50% of the mercury in the flue gas with no sorbent present. As discussed previously, ash from western coals that is high in calcium has been observed to adsorb mercury in baghouses.

Figure VI-2 contrasts pilot-scale ESP data (open symbols) with baghouse data (closed symbols) for sorbent injection, and the same eastern bituminous coal burned at PSE&G's Hudson Station (Butz 1999). As expected, given the better contact between sorbent and gas in a baghouse, the baghouse data show higher mercury capture for the same sorbent loading. This was also observed at Comanche Station burning a western subbituminous coal (Brown 1999).

Figure VI-2: Mercury Removal versus Activated Carbon Loading in a Pilot Scale ESP (open symbols) and Baghouse (closed symbols)



Modified activated carbons (e.g., doped with sulfur or iodine) have also been studied. There are many studies on methods for making and characterizing doped activated carbons, but few studies in which the performance of the sorbent was characterized at realistic gas compositions. In general, sulfur impregnation increases the capacity of activated carbon for both Hg^0 and $HgCl_2$, although not to the same degree (Hsi 1998). Research has been done on processes for preparing sulfur-impregnated activated carbons because it is recognized that the form of sulfur and morphology of the activated carbon are important in determining the capacity of the sorbent for mercury. Pore structure as well as the amount of elemental sulfur on the sorbent which is accessible to the gas have been observed to be critical factors in determining the ability of sulfur-impregnated carbons to adsorb mercury (Hsi 2000). The temperature of sulfur impregnation affects the amount of elemental sulfur deposited on the sorbent which is then available to react with mercury.

The cost of sulfur-impregnated activated carbons may be substantially higher than undoped activated carbons because of the need for elaborate preparation methods for sulfur impregnation. In one experiment designed to investigate a solution for this problem, cloth made from activated carbon fiber was impregnated with sulfur and tested for adsorption of mercury in a simulated flue gas (Hsi 2000). High capacity for elemental mercury was demonstrated at bench scale, indicating that a fabric filter might efficiently adsorb mercury, perhaps at a lower cost than that for injection of powdered activated carbon.

As already discussed, coal fly ash containing carbon also shows activity as a mercury sorbent (Butz 1999). However, little is known about the properties of unburned carbon which contribute to mercury sorption, and more work at the pilot scale will be necessary to evaluate the efficacy of high-carbon fly ash as a mercury sorbent. It is expected that fly ash-derived sorbents may be, in some cases, a cost-effective alternative to activated carbon.

Non-Carbon Based

Non-carbon sorbents have also been investigated for removal of mercury. Although activated carbon currently seems to be the most likely candidate for mercury removal, there are some drawbacks. Chief among these is the potential increase in the carbon content of the fly ash beyond a level which current purchasers may not find acceptable. It is possible that some of the non-carbon based sorbents would be cheaper than activated carbon. Calcium-based sorbents have been characterized most extensively (Ghorishi 1998, 1999). In addition sorbents based on zeolites are under development (Morency 2000), and noble metals have also been tested as mercury sorbents (Brown 1999).

Co-injection of activated carbon and calcium-based sorbents in air pollution control equipment has been known to increase the removal of mercury from flue gas. This was first demonstrated in spray dryer absorber (SDA) systems on full-scale coal-fired power plants (Laudal 1999). Recently, there has been pilot work on injection of calcium-based sorbents upstream of a baghouse (Butz 1999), and into a specialized fluidized bed reactor (Helfritch 1999). Combining activated carbon with hydrated lime can reduce the amount of carbon required (for an equivalent mercury removal) by one-half to one-third. Pilot tests of limestone furnace injection, followed by a cyclone separator, also showed good removal of mercury from flue gas in a pilot scale unit burning eastern bituminous coals (Amrhein 1999).

Recent experience with non-carbon based sorbents is discussed next.

Calcium-Based Sorbents

Recent laboratory investigations of calcium-based sorbents for mercury control (Ghorishi 1998, 1999) have shed light on the mechanisms involved which offer the potential for more efficient use of such sorbents across a range of applications. Fly ash, hydrated lime, and Advacate™ (a trademarked, pressurized fly ash-lime mixture) were tested for mercury sorption in a fixed bed reactor (Ghorishi 1998). All calcium-based sorbents captured $HgCl_2$ from simulated flue gas at 100°C (although less than a commercial activated carbon). Addition of SO_2 to the gas mixture decreased the sorption of $HgCl_2$ suggesting that there is competition for the same alkaline sites between the two species. Recent work has suggested the existence of formation of a Hg-O bond between $HgCl_2$ and CaO (Gullett 2000). In contrast, the calcium-based sorbents showed little or no removal of Hg^0 in the absence of SO_2. Addition of SO_2 to the gas greatly enhanced the uptake of elemental mercury, suggesting the possibility of some chemical reaction on the surface. Increased sorbent surface area and internal pore volume also increased the capture of elemental mercury by calcium-based sorbents.

- *Zeolites*

Zeolites are another class of sorbents being tested. Zeolites are aluminosilicate sorbents with crystalline structures that contain large, monosized pores and have high surface areas. Zeolites reversibly absorb many molecules and the structure can be made selective to specific molecules. Naturally occurring zeolites are mined and available at costs comparable to activated carbon. Recent fixed bed and entrained flow laboratory work has shown that zeolites doped with proprietary agents can adsorb mercury from coal combustion flue gas as efficiently as commercial activated carbon. Pilot scale testing at a coal-fired power plant is in progress for this process (Morency 2000).

- *Noble Metals*

Mercury is known to amalgamate (or alloy) with noble metals such as gold and silver. Mercury can be collected in this way and removed by a simple thermal desorption process which forces the mercury out of the amalgam. A process for removing mercury from flue gas using monoliths containing a noble metal sorbent has been demonstrated at the pilot scale. Work is ongoing to demonstrate this process at a commercial scale (Brown 1999).

Enhanced FGD Performance

Considerable effort is being devoted to improving the effectiveness of FGD systems for capturing mercury in flue gas by increasing the fraction of oxidized mercury in the gas at the inlet to the scrubber. If a suitable catalyst material could be found, it might be placed upstream of the FGD. Many different catalysts, from traditional industrial catalysts to various types of fly ash, have been investigated (Richardson 1999). Pilot-scale testing of promising catalysts is underway. If this technology proves to be successful in field applications, it offers a great potential as an alternative to sorbent-based technologies since many power plants have already been retrofitted with FGDs. As noted earlier, the SCR-FGD combination appears to allow the oxidation of elemental mercury by the catalyst used in SCR with subsequent capture of total mercury by the downstream FGD. The EPA’s ICR data, when available, should be able to answer this question.

Emerging Technologies

A number of other technologies are at various stages of research and development, including bench scale and slipstream testing. The interest in developing these technologies as alternatives to more developed sorbent-based technologies or enhanced wet scrubbing reflects a strong desire on the part of various vendors to provide innovative and potentially cost-effective strategies to control mercury. Approaches such as plasma corona discharge are being tested at Alabama Power Plant Miller Unit 3; EPRI's COHPAC/TOXECON is being tested at PSE&G Hudson Unit 2, EEC's Circulating Fluid Bed is being tested at PSE&G Mercer Station; and Powerspan's Electro-Catalytic Oxidation (ECO) is being tested at First Energy Eastlake Unit 5.

These and other technologies along with more established technologies described earlier will continue to develop towards commercialization. However, as the three historical case studies clearly indicate, this movement towards large scale filed application will be accelerated only after regulatory drivers establishing performance standards (emission rate limits) are put in place which will allow all technologies to compete at a level playing field. Brief descriptions of these technologies follow:

- <u>Mercury Control by Corona Discharge - Environmental Elements Corporation (EPA, website www.utility.rti.org)</u> The technology involves the generation of an intense corona discharge typically in front of an ESP and wet scrubber. This creates oxygen-carrying reactive species which, in turn, oxidize mercury in the flue gas. Oxidized mercury is subsequently captured in the wet scrubber. In addition some SO_2 is oxidized to SO_3 in the process which may augment the performance of the ESP and improve particulate collection. A slip stream plant was installed at Alabama Power Miller plant Unit 3. Initial tests indicated 80% mercury removal and complete oxidation of elemental mercury at 10 and 20 watts/cfm, respectively.

- EPRI's COHPAC-TOXECON - PSE&G Hudson Station (Butz 1999) This technology is based on the combination of an ESP-Pulse-Jet Baghouse with sorbent injection technology. This approach focuses on improving the efficiency of sorbent injection by providing high efficiency particulate collection as well as a good "contact" scheme for the sorbent and mercury (i.e. the baghouse). COHPAC is a commercial, highly effective technology. Some results with carbon injection at Hudson station were presented earlier in this chapter.

- Circulating Fluid Bed for Mercury and Fine Particulate Control - PSE&G Mercer Station (Helfritch 1999, EPA website www.utility.rti.org) This technology uses a circulating fluid bed to establish a zone of high particle density. This bed is enhanced with activated carbon for mercury adsorption. Carbon utilization is enhanced due to the high residence times in the bed, while fine particles tend to agglomerate through "collisions" in the bed, facilitating their subsequent capture in a conventional ESP. The technology can be used with lime injection for control of acid gases. Tests indicated mercury capture of up to about 80% using iodine -impregnated activated carbon.

- Powerspan's Electro-Catalytic Oxidation Technology - First Energy Eastlake Unit 5 This technology is capable of capturing a number of pollutants (SO_2, NO_x, mercury, and fine particulate matter). This technology uses a dielectric barrier discharge to convert NO_x and SO_2 to acids and oxidize elemental mercury. A condensing, wet ESP is used to collect acid mists, fine particulate and mercury. The planned 50 MW demonstration project follows a 2 MW pilot project at First Energy's Burger Station in Ohio, where test results showed mercury emission reduction of 68% (Clean Air Compliance Review, May 31, 2000). The technology has caught the attention of the industry as exemplified by the recent investment in Powerspan by AEP (www.energyonline.com/Restructuring/news/0725env.html 2000).

Summary of Technologies

This section provided a brief review of a number of pollution prevention (e.g., fuel switching and coal cleaning) and retrofit technology-based options for controlling mercury emissions from existing coal-fired power plants.

Retrofit technology approaches can be divided into two general categories: (1) those that use/optimize existing APCD controls for simultaneous and optimum capture of mercury and other pollutants; and (2) new technologies dedicated specifically to mercury capture.

In the first group, developments are underway to characterize and optimize mercury capture in existing particulate (FF/ESP) and SO_2 control (FGD) equipment. In addition, sorbent injection developments targeting different sorbents, as well as injection technology configurations, seem to be the most active areas of development. Carbon-based sorbents are the most tested and best understood to date. As sorbent properties, such as surface area and pore size have a significant impact on performance of the sorbent, development is not restricted to carbon products only. Calcium, zeolites, and noble metals are among those that may prove to be attractive alternatives in the future.

Better understanding of oxidation mechanisms and subsequent development of practical methods to oxidize elemental mercury in the flue gas will likely yield significant benefits since capture of oxidized mercury in FGD systems and by carbon-based sorbents is greater than that of elemental mercury. Early indications that SCR technology can promote oxidation of elemental mercury may also have significant impact, as the SCR technology is poised for widespread use in the U.S. in response to recent federal and state regulations to control emissions of oxides of nitrogen.

New and emerging technologies may eventually provide more cost-effective options for mercury controls either by providing multi-pollutant control capability and "spreading" the cost over several benefits (e.g., Powerspan's ECO), or by increasing the control efficiency for mercury through dedicated equipment (e.g., EPRI's COHPAC/TOXECON, EEC's Circulating Fluid Bed).

E. Cost Estimates of Controlling Mercury from Coal-Fired Boilers

Perhaps an important question after "Does the technology work?" is "How much does it cost?" It is important to note that the current and past estimates made by EPA and DOE are rather preliminary in nature. These estimates were made for “model plants” by EPA and DOE, since none of the technologies have been implemented at full scale under real-world conditions.

Most of these estimates have been done for activated carbon injection and assume a mercury reduction efficiency ranging from 70 to 90%. A number of assumptions are made about size of the boilers, type of coal burned (bituminous or sub-bituminous), gas flow rates, chlorine level in the coal, inlet uncontrolled mercury level, and the amount of activated carbon needed to remove a given amount of mercury in the flue stream (C/Hg ratio, usually expressed as grams of carbon /grams of Hg). The results of these cost studies differ significantly mostly due to different assumptions, especially the assumption about the C/Hg ratio, since the cost of activated carbon is a large part of the overall cost of controlling mercury.

These studies (Brown 1999; EPA-452/R-97-003 1997; Srivastava 2000; EPA 1999) have included detailed technology-specific estimates as well as estimates of overall national costs predicated on projected technology performance and costs. It is not the purpose of this section to provide detailed cost estimates included in the cited references. However, an important observation about these cost estimates is their sharp downward trend. For example, the March 1999 EPA report states that a 70% reduction in current mercury emissions from coal-fired boilers in the U.S. is expected to cost in the range of $1.7 to 1.9 billion per year. These estimates are more than 60% lower than the $5 billion cost projected only a year earlier in the EPA’s Mercury Study (EPA-452/R-97-003 1997). More importantly, when these costs are expressed in terms of the cost to the ratepayers in terms of mills per kWh of electricity produced, they are quite comparable to the costs currently being incurred for pollutants such as NO_x (see Table VI-1).

Table VI-1: Comparison of mercury control costs with NO_x control costs

Control	Capital Costs ($/kW)	Total Annual Cost (mills/kWh)
Mercury Controls	0.43 – 52.21	0.17 – 1.76
Low-NO_x Burners	7.31 – 35.89	0.15 – 0.54
Selective Catalytic Reduction	40.88 – 91.51	1.30 – 2.41

Source: Srivastava 2000

The eventual costs of regulating mercury emissions from power plants are, of course, difficult to predict with certainty and depend strongly on the timing and level of control specified. However, the history of costs of control for the three case studies (recognizing fully well that none of the three case studies provides an exact analog to the case of mercury control from coal-fired boilers; see Chapter V) strongly suggests that costs of mercury control should continue to decline as a number of technologies evolve and compete, various technology vendors enter the field to provide alternative technologies best suited to specific coal-fired boilers to meet established emission standards, and as user industry increasingly adopts integrated control strategies capable of reducing multiple pollutants all at once.

F. Current Efforts to Establish Baseline Emissions and Recommendations for Future Work

The previous sections summarized the current state of development of mercury control technologies and how the capture efficiency is strongly influenced by a number of variables including fuel properties, mercury forms, equipment type, etc. In this section, the current data gathering efforts to establish the baseline mercury emissions from coal-fired boilers are described first. Then recommendations for further technology development are presented.

F.1 Assessment of Baseline Emissions of Mercury from Full Scale Power Plants

The Information Collection Request (ICR) (EPA website www.utility.rti.org) initiated by EPA in 1999 was designed to provide information which is expected to be useful for making a regulatory determination about controlling mercury emissions from coal-fired power plants. This effort included a general information gathering (Phase I), a coal quality documentation phase (Phase II), and an actual plant testing program for mercury emissions including mercury speciation from coal-fired power plants (Phase III). Over 75 plants were statistically selected for this testing based on several factors, which included boiler type, configuration of air pollution control equipment, and fuel type. For each plant, the input value of mercury in the coal was measured (along with other coal composition data). Mercury measurements were made at the stack and at the inlet to the last air pollution control device (APCD) using the Ontario Hydro method providing data for elemental, oxidized, and particulate-bound mercury. This will allow gathering of information on mercury emissions for various boiler-APCD configurations, representing the demographics of the overall U.S. boiler population.

At the time this report was being prepared, the data from the ICR were still being analyzed by EPA, DOE, and EPRI. Once the ICR results are properly analyzed, it is expected that we will know substantially more about some of the outstanding issues discussed here. Ideally, this would lead to a better understanding of baseline mercury emissions and reductions currently being achieved for each type of coal, plant type, and APCD for the current population of coal-fired boilers in the U.S.

F.2 Demonstrations of Control Technologies at Full Scale

Technology demonstrations at full-scale are an integral and important part of the technology commercialization and acceptance. It is therefore important that mercury technologies "ready" for full-scale be demonstrated in a timely manner. An example of this is the recent $13 million solicitation from the U.S. DOE titled "Testing and Evaluation of Promising Mercury Control Technologies for Coal Based Power Systems" which envisions several full-scale technology demonstrations in the next few years. As a part of this solicitation, DOE recently announced in August 2000, a $6.8 million contract with a number of organizations (ADA Environmental Solutions, EPRI, PG&E National Energy Group, and Wisconsin Electric Power Company) to test sorbent injection on four power plants.

Sorbent Development

For retrofit systems, activated carbon represents currently the most evaluated material for removal of gaseous mercury from coal-fired power plants, capable of achieving up to 90% removal. Other sorbents have been investigated, although not in as much detail. Coal fly ash containing carbon also shows promise for adsorption of oxidized mercury, while high calcium ash from western coals, has been shown to adsorb elemental mercury. More characterization would be helpful to generalize these findings. Non-carbon sorbents have also been investigated for removal of mercury. Sorbents based on zeolites are under development. Noble metals have also been tested as mercury sorbents. Additional development work is necessary to evaluate the feasibility of some of these sorbents especially those that might offer more cost-effective alternatives to carbon-based sorbents.

Multi-pollutant sorbents, those that remove both acid gases and mercury, show promise for reducing the overall cost of controlling mercury, even if the sorbents themselves are not significantly cheaper than activated carbon. New contacting schemes may be required, such as the fluidized bed reactor (Helfritch 1999) or the TOXECON™ (Butz 1999) baghouse which have already been demonstrated at pilot scale. Again, more extensive pilot-scale as well as full-scale field-testing will provide added confidence in these approaches.

Enhancement of Mercury Removal in Existing Equipment

As discussed, oxidation of elemental mercury in the flue stream upstream of the FGD systems should increase overall mercury removal in wet and dry FGD control systems. As noted before, 17% of the boilers in the U.S. have one of these types of SO_2 control equipment. Also, increasing the amount of oxidized mercury may improve the capture of mercury in fabric filters, at least for plants burning bituminous coals with high carbon in ash. However, only 7% of U.S. power plants employ fabric filters and many of those plants burn western, sub-bituminous coals. There is no relevant data about the effect of increasing the proportion of oxidized mercury on removal rates for fabric filters on eastern coals. Further research is needed to understand what effects that may have.

Preliminary data suggest that SCR systems are capable of oxidizing elemental mercury and that the combination of SCR and FGD (or SDA) would enhance mercury control. Dedicated catalysts have also been proposed for this purpose. In either case, a thorough understanding of the effects of various operating parameters on the rate of oxidation will be needed. Further research is required to characterize the behavior of mercury in SCR (and SNCR) systems so that levels of oxidation can be reliably predicted. This area of research is very timely since a wide scale application of SCR technology to control NO_x emissions is underway in the United States.

Stability of Hg in Coal Combustion Byproducts

If mercury controls are required on coal-fired power plants, mercury will be transferred from the flue gas to other streams in the plant, notably to the ash (if it contains sorbent) or to the scrubber sludge (for wet FGD systems). A preliminary study of mercury in FGD sludge, shows that mercury is not volatile in scrubber sludge at temperatures up to $140^{o}C$ (DeVito 1999). Other recent work has been conducted on the stability of mercury in activated carbon and the leachability of mercury from spent sorbent (Blythe 1999). Commercial activated carbon exposed to either elemental mercury or $HgCl_2$ in simulated combustion gases in a laboratory apparatus was stable in air at room temperature over a six-month period, showing negligible loss of mercury. Heating spent activated carbon to $275^{o}F$ did cause desorption of mercury compounds. The authors of this study concluded that regeneration of activated carbon might be possible as a result. Mercury in activated carbon and other sorbents has shown negligible leachability to date in various experiments.

In summary, spent mercury sorbents appear to be stable in air, and scrubber sludge containing mercury is stable at temperatures up to $140^{o}C$. Spent sorbents do not appear to be leachable using standard leaching protocol, but do emit mercury upon heating. Further research, however, is needed on ash and sorbent residues to evaluate mercury retention and the potential for release back into the environment.

G. Conclusions

Coal-fired power plants presently account for about one-third of total anthropogenic mercury emissions in the U.S., and unlike other major sources of mercury such as municipal waste combustors and hospital waste incinerators, are currently unregulated. Even in the absence of regulatory drivers, limited private and public sector funded research and development efforts have been successful in answering many basic questions about the applicability of various mercury control technology options. The best estimate of the time frame over which a number of existing and emerging technologies are expected to become commercially available is from 2 to 5 years.

Over this time frame, potential full-scale demonstrations of technologies such as carbon (or other sorbents) injection, enhanced wet scrubbing, enhanced particulate controls, fuel switching to natural gas, or some innovative combination of these or other emerging technologies should occur, once firm regulatory requirements are put in place. It is important that these requirements be performance-based and allow for a fair competition between alternative technologies.

In addition to the 2-5 year time frame for controls, it seems plausible that mercury emissions could be controlled at 90% level and beyond, for many coal-fired boilers. Since the technologies at their current “infancy” stage have shown potential to control at such levels and the technology development is expected to take place at an accelerated pace, there is cause for optimism that high levels of control (90% and higher) will be possible in the 2-5 year time frame.

The current national estimates of cost of controlling mercury indicate that they are quite comparable to the costs of controlling traditional pollutants such as NO_x. However, the history of control costs for the three case studies strongly suggests that mercury control costs should continue to decline as a number of technologies evolve and compete, various vendors enter the marketplace to provide alternative options best suited to specific boilers, and as user industry increasingly adopts integrated control strategies capable of reducing multiple pollutants. These alternative options will become available after strong regulatory drivers are put in place, not unlike the three case studies.

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VII. Conclusions

The three case studies summarized in this report involve different pollutants, distinct regulatory processes, varying timeframes -- even, in the automobile example, a different industry. Despite these differences, a strikingly consistent pattern emerges. In every case, effective control technologies became commercially available only after regulatory drivers were introduced. In every case, industry initially resisted the imposition of control requirements, citing lack of need, technological barriers, and expense. In every case, industry nonetheless succeeded in implementing control technologies capable of achieving emissions reductions on the order of 90 percent or more in response to regulations. Finally, in all three cases, the high costs and implementation difficulties originally projected by industry, and, in many instances, even by government officials proved too pessimistic. In fact, early cost projections frequently overstated actual costs by two to ten times.

Of course, none of the case studies presented in this report provides a precise analog to the control of mercury emissions from coal-fired electricity generators. But the elements common to each of them strongly suggest an underlying trajectory of technology development that very likely *does* apply to the mercury situation. If so, policymakers can be reasonably certain of two things: first, that mercury emissions from power plants *can be* substantially and cost-effectively controlled. Second, that the control technologies capable of accomplishing this will not become commercially available until regulatory or other incentives compel their introduction.

The technology trajectory clearly indicated by the case studies can be characterized in terms of three distinct phases: a pre-regulatory phase, a regulatory phase, and an implementation phase. During the pre-regulatory phase, the initial identification of an environmental or public health harm is typically followed by several years of scientific and political debate about its causes and potential consequences, the relative contribution of different pollution sources, and the availability of cost-effective means to reduce the harm. Opponents of regulation typically stress scientific uncertainty and the need for further study to resolve the precise relationship between emissions and environmental consequences as a precondition for specifying reductions from particular sources. During this period, early research and development efforts to explore potential control technologies often begin. These may be initiated by public institutions, by third-party interests in anticipation of a future market, or by the affected industry itself, both in preparation for eventual requirements and/or to gain credibility in the larger policy debate. At times, industry has even advanced particular control options on a voluntary basis as a means of influencing the scope or direction of future regulation, as was the case when car manufacturers put forward a modest set of engine adjustments as a substitute for more effective technology -- catalytic converters -- in the late 1960s.

These early efforts, though important, are generally weak and do not lead to widespread demonstration or commercialization of control technologies. This is not surprising since simple economic theory dictates that significant investments in control technology will occur only when such technologies provide economic value. Because private markets often fail to internalize the societal value of environmental protection, definitive government intervention is usually necessary to create the conditions under which control technologies will be fully developed and deployed. Of course, government itself can sponsor technology development, either independently or in partnership with private industry and there are numerous past and current examples of this approach. Publicly funded research helped advance both NO_x and SO_2 controls for power plants in the pre-regulatory phase and is currently responsible for some ongoing work on mercury. Historically, it has played a lesser role in the evolution of automobile controls, although a joint industry-government effort was created a few years ago to address the lack of further progress in automotive efficiency improvement.[88] However, experience suggests that government-sponsored research and development efforts rarely compensate for the absence of real market incentives or regulatory drivers. Though such efforts may result in prototypes, benchmark studies, and pilot projects, more forceful regulatory drivers are usually necessary to bring about the widespread commercialization of effective control technologies.

At present, the development of mercury controls for power plants is in the pre-regulatory phase. Early research and development efforts have identified at least two potentially promising control options at this time: activated carbon injection and enhanced wet scrubbing. Both have been tested in bench and pilot scale studies and offer the potential to achieve substantial (i.e. up to 90 percent and greater) reductions. Experience with full-scale applications of these technologies remains fairly limited but for the reasons described above there would be no reason to expect otherwise at this point in time.[89] Nor should it be surprising that both technologies still face technical and other challenges. On the contrary, current experimental controls for mercury emissions from power plants are at least as advanced – indeed they are probably more advanced – than were NO_x and SO_2 controls or automobile emissions controls prior to the introduction of the first tier of regulatory mandates.

Besides the fact that at least two potentially viable control options have been identified, several other aspects of the current mercury situation bode well for the ability of power plant operators to achieve substantial emissions reductions. First, considerable progress has already been made in understanding the physics and chemistry of mercury interactions in combustion exhaust streams and in developing methods for monitoring and speciating emissions. Other issues, such as waste disposal and byproduct creation, require further work but are generally well understood. Second, power plant operators have by now developed considerable expertise and sophistication in pollution control as a result of past SO_2, NO_x and particulate control requirements. Moreover, many of the well-established technologies alone and in combination (such as FGDs, SCR, baghouses and ESPs) now used to control those pollutants will definitely play an important role in reducing mercury emissions as well. For example, there are early indications that the

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[88] Interestingly, the “Partnership for a New Generation of Vehicles,” as this joint effort is called, was itself a response to the failure of fuel economy regulations – which have remained unchanged for a decade – to elicit further improvement in vehicle efficiency.

[89] Activated carbon injection is currently used to limit mercury and other hazardous emissions from municipal waste combustors. However, there are some important differences between this application and power plants.

SCR-FGD combination, besides substantially reducing SO_2 and NO_x emissions, may also be able to reduce mercury emissions by more than 95 percent at essentially zero cost. At the same time, other sources such as municipal waste combustors have acquired specific experience with mercury controls; this experience, while not directly transferable to the different exhaust characteristics of power plant emissions, should prove helpful. Third, regulatory agencies have themselves become more sophisticated and have developed a wider array of policy instruments for effectively leveraging market forces, encouraging advanced technology development, and promoting integrated pollution control strategies. Finally, add-on devices are not the only options for reducing power plant mercury emissions. Fuel switching, efficiency improvements, and other pollution prevention approaches can be expected to play an important role in reducing overall mercury emissions and will likely provide companies with a wide variety of possible strategies for reducing their regulatory exposure.

The eventual economic and technological consequences of regulating mercury emissions from power plants are, of course, difficult to predict with certainty and depend strongly on the timing and level of control specified. Research and development results to date suggest that the costs of add-on mercury controls for coal-fired power plants are likely to be comparable, on a per kWh basis, to those currently associated with controlling NO_x emissions.[90] Moreover, costs should continue to decline as technology evolves and as industry increasingly adopts integrated control strategies capable of simultaneously reducing multiple pollutant emissions.[91]

A number of recent developments suggest that the pre-regulatory phase for power plant mercury emissions is drawing to a close. Public awareness of methylmercury contamination in fish is high and health experts, environmental advocates, and public officials are calling with mounting urgency for reductions. In perhaps the most telling sign of all, several states have signaled the intent to take regulatory action on their own. The New England states, in conjunction with the eastern Canadian provinces, for example, have committed to the virtual elimination of anthropogenic sources of mercury, with plans to reduce emissions from coal-fired boilers by up to 90 percent by 2010.

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[90] Industry representatives frequently point out that control costs for mercury are likely to be far higher than for NOx or SO2 on a per ton or per pound basis. These figures must be considered in light of the far greater health and environmental hazards presented by a given quantity of mercury when compared to the same quantity of conventional pollutants and hence do not constitute a relevant basis for comparison.

[91] Some observers of the current mercury debate have suggested that the simultaneous deregulation of the power industry, which has led, among other things, to diminished industry funding for cooperative research efforts such as those conducted by the Electric Power Research Institute (EPRI), could pose an additional hurdle to the near term development of mercury control technologies. As the automobile example shows, however, substantial advances in control technology can be achieved in competitive as well as monopolistic industries. The presence or absence of regulatory requirements and the stringency of those requirements, rather than the structure of the industry being regulated, appears to be the more important factor. Ironically, in the automobile case, industry cooperation appears at times to have had the effect of suppressing technology rather than advancing it. Meanwhile, monopolistic industries like regulated electric utilities have in the past proved no more innovative or receptive to the need for regulation – despite the fact that they were arguably in a better position to pass additional costs on to the ratepayers– than their competitive counterparts.

Several individual states, including New Jersey, Massachusetts, Minnesota, and Wisconsin have initiatives in place aimed at limiting future power plant emissions. All of these states, recognizing that mercury is subject to long-range atmospheric transport and cannot be addressed at the state level alone, have also been urging the federal government to take action. Meanwhile, EPA – which already has the legislative authority to impose limits on power plant mercury emissions – is in a position to take a first step toward such action as early as December 15, 2000, when the Agency must reach an initial determination regarding the need for regulation.

As policymakers look ahead to the regulatory and implementation phases of the mercury debate, past experience with technology development and environmental regulation suggests additional considerations. One is that the ultimate response to regulation will depend on a complex mix of factors including macro-economic conditions, fuel prices, energy policies, consumer demand, and other areas of technological development. Hence the most successful regulations have avoided picking technology "winners" but have rather established clear performance standards and allowed market forces to determine how those standards can most efficiently be met.

A second lesson is that industry, while adept at meeting standards, is unlikely to exceed them without further incentives. As a result, weak regulatory mandates can end up functioning as a ceiling, as well as a floor, for future technology development. In the case of SO_2, for instance, substantial advances in flue gas desulfurization (FGD) technology were achieved in the U.S. and overseas in recent decades. This technology is now highly developed and capable of achieving emissions reductions of over 95 percent. The current federal Acid Rain Program, however, requires overall national SO_2 reductions of only 50 percent from electricity generating boilers. Because there is little incentive to cut emissions further, individual facilities with FGD systems are optimizing scrubber performance (since excess emissions allowances have market value under the current cap and trade program), but industry as a whole is not deploying these systems as widely as it could be. Compared to the automobile case, the experience with SO_2 suggests that regulators should be careful not to underestimate technology potential. Weak standards may prove inefficient both in the sense that they lead to sunk investments in control strategies that are ultimately inadequate and in the sense that they necessitate politically costly and unnecessarily protracted reiterations of the regulatory process.

This report summarizes and presents the current state of understanding of mercury emissions and control options for coal-fired boilers in the context of past regulatory experience. In this context, the report makes a compelling case that concerns raised about technology availability should not stand in the way of mandating substantial near-term mercury emissions reductions from existing coal-fired power plants. On the contrary, further development of mercury control technologies might well stall in the absence of regulatory drivers. All the available evidence from more than four decades of environmental regulation in the U.S. suggests that once deadlines and emissions limits are established, the successful commercialization of cost-effective mercury control technologies will soon follow.