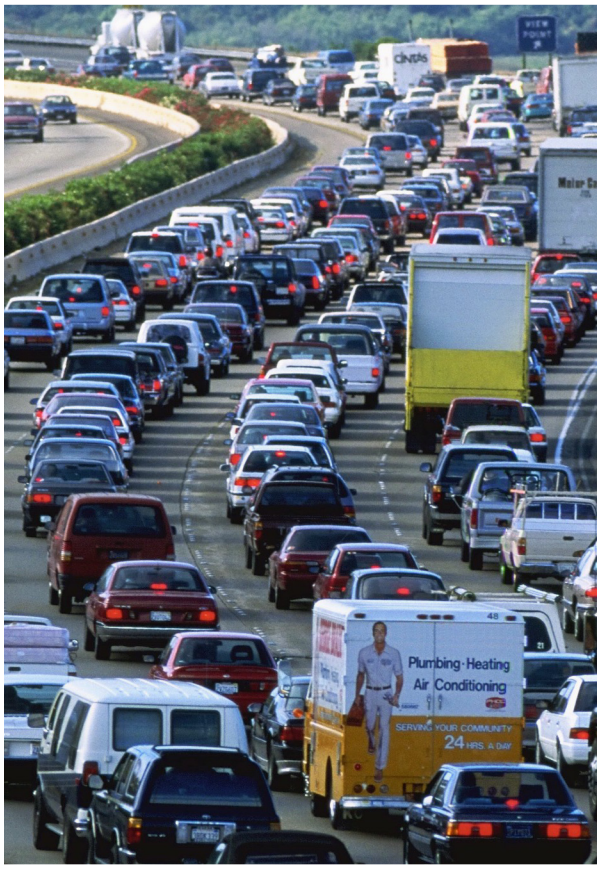
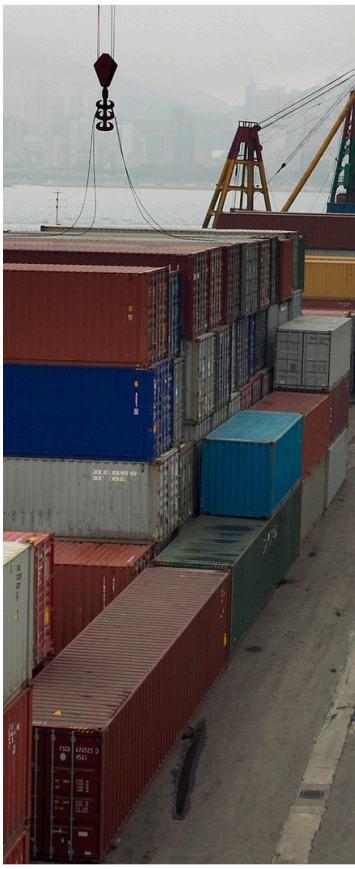
Controlling Fine Particulate Matter Under the Clean Air Act:

A Menu of Options





STAPPA
State and Territorial Air Pollution Program Administrators

ALAPCO
Association of Local Air Pollution Control Officials

March 2006

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Acknowledgements

On behalf of the State and Territorial Air Pollution Program Administrators (STAPPA) and the Association of Local Air Pollution Control Officials (ALAPCO), we are pleased to provide *Controlling Fine Particulate Matter Under the Clean Air Act: A Menu of Options.* Our associations developed this document to assist states and localities in determining the most effective ways to control emissions of fine particles ($PM_{2.5}$) and $PM_{2.5}$ precursors from sources in their areas. We hope that states and localities find this document useful as they prepare their State Implementation Plans (SIPs) for attaining or maintaining the $PM_{2.5}$ standard.

STAPPA and ALAPCO express gratitude to M.J Bradley & Associates, Inc. for its assistance in drafting this document, in particular, Ann Berwick, Michael Bradley, Tom Curry, Will Durbin, Dana Lowell and Chris Van Atten. We thank Brock Nicholson (North Carolina) and Lynne Liddington (Knox County, Tennessee), co-chairs of the associations' Criteria Pollutants Committee, under whose guidance this document was prepared. We also appreciate the efforts of the STAPPA and ALAPCO $PM_{2.5}$ Menu of Options Review Workgroup, who helped shape the options presented in this document. We thank Bill Becker, Executive Director of STAPPA and ALAPCO, and Amy Royden-Bloom, Senior Staff Associate of STAPPA and ALAPCO, who oversaw the project. Finally, we express our gratitude to EPA for providing the funding for this project.

Once again, we believe that *Controlling Fine Particulate Matter Under the Clean Air Act: A Menu of Options* will serve as a useful and important resource for states and localities as they develop approaches to regulate emissions of $PM_{2.5}$ and $PM_{2.5}$ precursors and thank all who contributed to its development.

Eddie Terrill	John Paul
STAPPA President	*ALAPCO President*

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About STAPPA and ALAPCO

The State and Territorial Air Pollution Program Administrators (STAPPA) and the Association of Local Air Pollution Control Officials (ALAPCO) are the two national associations of air quality officials in the states, territories and major metropolitan areas throughout the country. The members of STAPPA and ALAPCO have primary responsibility for implementing our nation's air pollution control laws and regulations. The associations serve to encourage the exchange of information and experience among air pollution control officials; enhance communication and cooperation among federal, state and local regulatory agencies; and facilitate air pollution control activities that will result in clean, healthful air across the country. STAPPA and ALAPCO share joint headquarters in Washington, DC.

For further information, contact STAPPA and ALAPCO at 444 North Capitol Street, NW, Suite 307, Washington, DC 20001 (telephone: 202-624-7864; fax: 202-624-7863; email 4cleanair@4cleanair.org) or visit our associations' web site at www.4cleanair.org.

Introduction

The State and Territorial Air Pollution Program Administrators (STAPPA) and the Association of Local Air Pollution Control Officials (ALAPCO) have prepared *Controlling Fine Particulate Matter Under the Clean Air Act: A Menu of Options ($PM_{2.5}$ Menu of Options)* to assist state and local air pollution control officials in evaluating the options for reducing fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursor emissions.

Areas throughout the eastern U.S. and California (and one area in Montana) currently exceed EPA's National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$, and states must submit State Implementation Plans (SIPs) by April 2008 detailing their plans for achieving the national standards.

Meanwhile, the $PM_{2.5}$ NAAQS are once again undergoing the periodic review that §109(d)(1) of the Clean Air Act requires take place at five-year intervals. Under the terms of a consent decree, EPA is to issue final standards by September 27, 2006. The Agency proposed new standards on January 17, 2006.

EPA estimates that meeting the current $PM_{2.5}$ standards would avoid tens of thousands of premature deaths annually and save hundreds of thousands of people from significant respiratory and cardiovascular disease. The Agency further estimates that the monetized health benefits of improvements in $PM_{2.5}$ air quality exceed the costs by a substantial margin.

$PM_{2.5}$ is a complex pollutant with many sources contributing to the ambient air quality problem. As a result, this *$PM_{2.5}$ Menu of Options* addresses a broad array of emission source categories, ranging from household furnaces to petroleum refineries. The challenge confronting air quality officials is tremendous, as evidenced by the sheer number of options that we identify for improving air quality. But therein lie the opportunities, as well.

Like STAPPA's and ALAPCO's previous document—*Controlling Particulate Matter Under the Clean Air Act: A Menu of Options*—this document compiles and analyzes secondary information. It is intended to serve as a general reference for a national audience, and it will in no way substitute for a thorough analysis by state and local agencies of local emissions sources and conditions, using appropriate guidance from EPA and other available information.

What To Regulate

The national focus of this report should not obscure an absolutely central point: local choices about the sources and pollutants to control will need to be informed by highly local considerations. A particular source category may account for a small share of national $PM_{2.5}$ emissions, but it may nonetheless dominate the local inventory.

The chemistry and physics of $PM_{2.5}$ formation in the atmosphere is incompletely understood. Some $PM_{2.5}$ is

released directly to the atmosphere, and some forms from emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) (which are currently viewed as the most significant precursors and are the only ones addressed in this report). Ammonia and volatile organic compounds (VOCs), which are not included in this report, can also contribute to ambient $PM_{2.5}$. Direct $PM_{2.5}$ emissions may be largely responsible for one area's nonattainment, while SO_2 emissions may cause the problem elsewhere. The choice of whether to focus on reducing direct $PM_{2.5}$, SO_2 or NO_x—or all of them, or ammonia or VOCs—will depend on local source contributions and atmospheric chemistry.

There are further challenges for SIP writers. In a perfect world, control-efficiency and cost-effectiveness data would be at hand; however, it is not consistently available. Of course, even when information of this sort can be found, it may not be applicable to all sources.

And another source of uncertainty complicates the job. As we discuss in Chapter 3, *Fine Particulate Matter and Precursor Emissions*, there are important distinctions between filterable and condensable $PM_{2.5}$. Further, some methods used to measure PM emissions reflect only the filterable components and, to exacerbate the problem, the filterable components vary depending on the test method used. Although we discuss this issue in Chapter 3 in the context of the national $PM_{2.5}$ inventory, the distinction between filterables and condensables also raises regulatory and permitting issues.

The Authority to Regulate

Having decided what sources and pollutants need to be controlled in order to address $PM_{2.5}$ nonattainment, regulators must then ascertain their authority to do so.

The Clean Air Act divides responsibility for various types of air pollution sources and air pollutants between the states and localities on the one hand and the federal government on the other. Generally, state and local regulators share responsibility with EPA for regulating so-called "criteria" pollutants from stationary and area sources (see Chapter 4, *The Clean Air Act*), with states and localities assigned the lead role in addressing emissions from these source categories.

States and localities are free under *federal* law to adopt more stringent standards for stationary and area sources than the Clean Air Act requires. However, some states may be limited by state law or policy in whether they can enact requirements that are more stringent than federal standards. Here, we outline the possible approaches to tightening federal standards that states and localities may consider, and to developing standards where no federal programs exist.

For states that have no latitude or little latitude beyond what the Clean Air Act prescribes, the priority will be to ensure strict compliance with the limits that the Act and federal regulations impose on particulates and precursor pollutants. In these states, the precise language of the statutory limitation will inform the degree of regulatory latitude. For example, regulators in at least some of these states may not be able to set more stringent standards for those sources that federal law or regulations actually address, but in some of these states regulators may see their way clear to setting standards for smaller sources than those covered by federal requirements.

Moreover, there are no actual federal Reasonably Available Control Technology (RACT) standards—EPA issues only guidelines (and although the RACT standards are intended to reflect real-time advancements in technology, many of the guidelines are seriously outdated). Since the guidelines do not set actual limits, even state prohibitions against enacting more stringent state standards may be inapplicable.

States and localities that are not limited to the requirements promulgated under federal law will want to look to the most stringent standards that regulators in other jurisdictions have imposed; we have identified these throughout this *Menu of Options*. State and local authority to impose such limits derives from the federal requirement to attain the NAAQS. The options for imposing more stringent requirements than current federal regulations include the following:

- Under the state or local version of federal regulatory air pollution programs, or through permit determinations, adopt the most stringent standards that appear to be feasible, even if they are more stringent than federal rules impose; or apply the federal or stricter standards to sources that are smaller than those covered by the federal requirements.
- Craft state or local regulatory programs or permits that impose on sources the most stringent standards that appear to be feasible. For example, this might include the imposition of Best Available Control Technology (BACT)-level standards on existing sources, even in the absence of a modification that would trigger New Source Review (NSR).
- Through regulations or permits, set limits on sulfur levels in coal and oil for sources that burn these fuels.
- For sources that are permitted to burn more than one type of fuel, impose permit conditions that strictly limit the extent to which they may burn the more polluting fuel.
- Consider the imposition of regulatory standards that can be met by most, but not necessarily all, sources to which the standard is applicable, with an opportunity

for sources to demonstrate that the standards are technically infeasible in light of particular circumstances.

- Adopt a state-level cap-and-trade program or participate in a regional trading program for a particular source category or group of source categories.

The discussion above applies to stationary and area sources, but not to mobile sources, as to which all states other than California have less leeway to impose their own standards. For *new* vehicles, states are limited to federal standards or to the more stringent standards that California has adopted. For *existing onroad* vehicles, all states can impose their own standards; although for *existing nonroad* vehicles, they once again have only the choice of federal or California standards.

However, by no stretch of the imagination does this mean that states should overlook the possibilities for mobile source strategies as a way of tackling $PM_{2.5}$ nonattainment. As we discuss in the chapters that follow, states have a range of opportunities for addressing these sources.

Energy Efficiency

The rising cost of fossil fuels has focused the nation's attention on the opportunities for reducing fuel consumption, including energy efficiency measures, some of which are addressed in this report. For example, Chapter 18, *Residential Fuel Combustion and Electricity Use*, discusses several demand-side efficiency measures. However, other source categories surely present opportunities for increased efficiency that regulators should not overlook.

On the supply side, energy efficiency measures involve increasing the efficiency of the fuel combustion process or of the way the fuel is utilized. At a conventional power plant, *two-thirds* of the potential energy in the fuel burned to produce electricity is inevitably lost to waste heat. Meanwhile, facilities burn additional fuel to satisfy their thermal needs (for hot water, space heating and the like). Combined heat and power (CHP or cogeneration) facilities located at or near a facility address this problem by recovering the waste heat and putting it to productive use. CHP systems can achieve overall efficiencies of greater than 80 percent (Elliott, 1999; EPA, 2000). In the late 1990s, 9 percent of this country's electricity came from cogeneration plants, although a number of other countries garnered a much higher percentage: Denmark (40 percent), Finland and the Netherlands (30 percent each), the Czech Republic (18 percent), and Germany (15 percent) (Elliott, 1999).

A number of the industry sectors we profile in this report are candidates for cogeneration. The petroleum refining and pulp and paper industries already employ cogeneration to some degree, but the practice has room to grow further in those industries and others, such as cement manufacturing and iron and steel production (Elliott, 1999).

There are unquestionably disincentives to the development of CHP in this country (e.g., high prices for excess power that CHP projects sell to the grid, long tax depreciation periods for CHP equipment), although increasing fuel prices make cogeneration more attractive. Environmental regulators can reverse some of the disincentives; for example, by writing air pollution permits on an electricity (and, where appropriate, thermal) output rather than on a heat input basis, to encourage efficiency in the use of fuel.

This Report

As indicated, this report addresses a broad range of source categories. These sources do not represent the entire inventory of $PM_{2.5}$, SO_2 and NO_x emissions, although they do cover a large share of the national inventory. Each source category chapter provides an overview of the category, background on the technical as opposed to the policy options for reducing emissions, and an overview of existing regulatory authority (with the regulatory authority issues discussed up-front in the mobile source chapters because of the preeminence of preemption considerations). Each chapter concludes with a discussion of state and local policy measures.

Additionally, the report has two separate technology chapters—one on boiler and another on diesel engine technologies. The boiler technology chapter informs the industrial and commercial boiler and electric generating unit chapters, as well as the chapters on other source categories that burn process fuels (e.g., pulp and paper). The chapter on diesel engine technologies is useful for understanding the three mobile source chapters, as well as substantial portions of the airport and marine port chapters.

The report begins with the *The Highlights* of the source category chapters. Although these do not substitute for the detail provided in each chapter, they cull the most significant emissions reductions opportunities. Prior to the sector-specific chapters, Chapter 2 discusses the health effects of $PM_{2.5}$, Chapter 3 discusses the national emissions inventory, and Chapter 4 provides an overview of the Clean Air Act.

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Chapter 1

The Highlights

Introduction

The highlights that follow identify the most significant emissions reduction opportunities for fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursors from each of the industries addressed in the sector-specific chapters of this report. We emphasize, however, that local considerations need to inform local choices about the sources and pollutants to control in order to address $PM_{2.5}$ pollution most effectively.

Additionally, almost all of the items we identify in *The Highlights* fall within the purview of environmental regulators. However, in certain instances we have included strategies that would require action by other agencies or branches of government, such as measures to reduce total vehicle miles traveled. We have done so only when these strategies are particularly effective.

Industrial and Commercial Boilers

Industrial and commercial boilers represent about 40 percent of all energy use in the industrial and commercial sectors. Although most commercial boilers are small (less than 10 million British thermal units per hour (MMBtu/hr)), very large industrial boilers (greater than 250 MMBtu/hr) account for almost half of industrial boiler capacity. However, in many fuel and size categories, standards for PM, sulfur dioxide (SO_2) and nitrogen dioxides (NO_x) emissions from industrial and commercial boilers are less stringent than standards for the same pollutant emissions from electric generating unit (EGU) boilers. Although there may be reasons in individual cases why the most stringent EGU boiler limits are not feasible for industrial and commercial boilers, those limits suggest an appropriate starting point for consideration of limits for industrial and commercial boilers larger than 250 MMBtu/hr, and even for those larger than 100 MMBtu/hr.

Apart from the differences in EGU and industrial/commercial boiler standards, there are enormous disparities in terms of the stringency of various emissions standards for PM, SO_2 and NO_x for industrial and commercial boilers. These disparities suggest that there is significant room for improvement in the emissions profile of this source category. For example:

- In certain industrial and commercial boiler categories (e.g., new residual oil-fired boilers between 10–100 MMBtu/hr, new and existing natural gas-fired boilers larger than 5 MMBtu/hr), state Best Available Control Technology (BACT) determinations set much tighter PM emissions limits than do the federal Maximum Achievable Control Technology (MACT) standards. For example, compare the BACT limit of 0.02 pounds per MMBtu (lb/MMBtu) to the MACT standard of 0.03 lb/MMBtu for new residual oil-fired boilers between 10–100 MMBtu/hr; and the BACT limit of 0.007 lb/MMBtu to the absence of any MACT limit for new natural gas-fired boilers larger

than 5 MMBtu/hr.

- The same kind of disparity appears between the new federal New Source Performance Standards (NSPS) for SO_2 emissions from industrial and commercial boilers built after February 2005 and the NSPS for SO_2 from existing industrial and commercial boilers. For example, the SO_2 standard for new coal-fired boilers between 100–250 MMBtu/hr is 0.20 lb/MMBtu, compared to 1.2 lb/MMBtu for existing units of that size. The SO_2 standard for new residual oil-fired boilers greater than 100 MMBtu/hr is 0.32 lb/MMBtu, compared to 0.8 lb/MMBtu for existing boilers. State and local regulators will want to consider the feasibility of requiring existing sources to meet these more stringent standards.

- Although wood-fired boilers constitute 4 percent of industrial boiler capacity, they account for fully 20 percent of industrial boiler $PM_{2.5}$ emissions. Average uncontrolled $PM_{2.5}$ emissions rates for wood-fired industrial boilers are higher than those of any fossil fuel-fired boilers. A recent BACT limit for PM for an *existing* wood-fired EGU boiler sets the same limit as the MACT standard for PM emissions for *new* wood-fired industrial and commercial boilers (0.025 lb/MMBtu). This limit is approximately three times more stringent than the MACT standard for PM from existing wood-fired boilers industrial and commercial boilers (0.07 lb/MMBtu).

- For industrial and commercial boilers burning natural gas and residual oil, the San Joaquin Valley Unified Air Pollution Control District (UAPCD) has set some of the most stringent NO_x emissions limits in the country. For example, it imposes a limit of 0.007 lb/MMBtu on natural gas-fired boilers greater than 5 MMBtu/hr, as compared to an NSPS of 0.3 lb/MMBtu for natural gas-fired boilers greater than 100 MMBtu/hr. Also, the San Joaquin Valley UAPCD has NO_x standards that apply to units as small as 0.075 MMBtu/hr, while the federal NSPS apply only to units larger than 100 MMBtu/hr.

State and local agencies have other options for limiting emissions from industrial and commercial boilers in addition to setting emissions limits. For example, Connecticut has set limits of 0.3 percent by weight on the sulfur content of fuel oil used by power plants (with the alternative of a 0.33 lb/MMBtu SO_2 emissions rate), and these limits could be applied to boilers in other industry sectors. New York has set limits on the sulfur content of both oil and coal used by power plants and other sources. The limits vary by area within the state, with the lowest limits in New York City: (1) 0.30 percent sulfur by weight for residual oil, (2) 0.20 percent sulfur by weight for distillate oil, and (3) 0.2 lb of sulfur per MMBtu gross heat content for solid fuels.

States should also consider supporting regional multi-pollutant initiatives (aimed at SO_2, NO_x and mercury emissions from EGUs and large industrial boilers), such as the Clean Air Interstate Rule (CAIR)-Plus initiative of the Ozone Transport Commission (OTC) and the regional air quality initiative of the Lake Michigan Air Directors Consortium (LADCO), discussed in the EGU *Highlights* below.

Electric Generating Units

The electric power sector is one of the dominant sources of $PM_{2.5}$, SO_2 and NO_x emissions in the U.S. Within the EGU sector, coal-fired power plants account for the vast majority of emissions. Nationwide, EGUs account for almost 10 percent of the $PM_{2.5}$ emissions, nearly 70 percent of the SO_2 emissions, and more than 20 percent of the NO_x emissions from all source categories. In 2002, coal-fired power plants were responsible for 92, 95 and 87 percent of EGU emissions of $PM_{2.5}$, SO_2 and NO_x, respectively.

The average emissions rates for SO_2 and NO_x across all coal-fired EGUs in the U.S. in 2002 were 0.94 lb/MMBtu and 0.40 lb/MMBtu, respectively. To put these average emissions rates in perspective, a typical baseload coal plant would generate about 33,000 tons of SO_2 and 14,000 tons of NO_x annually at these rates.

There are many opportunities for states and localities to regulate $PM_{2.5}$ emissions and their precursors from EGUs far more stringently than EPA's CAIR. In fact, several states have already passed laws or regulations aimed at reducing EGU emissions beyond federal requirements. Other states and localities may wish to adopt similar programs. For example, New Hampshire law requires EGUs to reduce their SO_2 emissions 75 percent (based on a rate of 3.0 pounds per megawatt-hour (lb/MWh)) by December 2006, and their NO_x emissions 70 percent (based on a rate of 1.5 lb/MWh) by the same date. Massachusetts regulations also limit coal plant SO_2 emissions to roughly 0.3 lb/MMBtu and NO_x emissions to roughly 0.15 lb/MMBtu within the next few years, well in advance of the second-phase CAIR caps. North Carolina law imposes similar limits, although with a later effective date.

STAPPA and ALAPCO have conducted an analysis identifying the emissions reductions that can be achieved from EGUs by applying BACT. The Associations concluded that EGUs could achieve emissions limits of 0.10 lb/MMBtu for SO_2 and 0.07–0.08 lb/MMBtu for NO_x.

States should also consider national and regional approaches to achieving more stringent and expeditious reductions than CAIR. STAPPA and ALAPCO's strategy calls for a national SO_2 cap of 1.26–1.89 million tons per year (as compared to a baseline of 10.6 million tons in 2001) by 2013, and a NO_x cap of 0.88–1.26 million tons

per year by the same date (as compared to a baseline of 4.7 million tons in 2001).

Additionally, regional groups like the OTC and LADCO are considering options that extend beyond CAIR and could include large industrial boilers. The OTC is evaluating a phased cap-and-trade program for SO_2 and NO_x. In Phase 1, which would be implemented on January 1, 2009, the program would be based on an SO_2 emissions rate of 0.24 lb/MMBtu, and a NO_x emissions rate of 0.12 lb/MMBtu. In Phase 2, which would be implemented beginning January 1, 2012, the caps would be ratcheted down based on an SO_2 emissions rate of 0.14 lb/MMBtu and a NO_x emissions rate of 0.08 lb/MMBtu. The Midwest Regional Planning Organization has been evaluating similar reduction targets, including a Phase 2 SO_2 cap between 0.15 lb/MMBtu and 0.10 lb/MMBtu in 2013 and a Phase 2 NO_x cap between 0.10 lb/MMBtu and 0.07 lb/MMBtu in 2013.

State and local agencies have other options for limiting emissions from power plants in addition to setting emissions limits. For example, as detailed in *The Highlights* for industrial and commercial boilers, Connecticut and New York have both set limits on the sulfur content of fuel.

States should also consider options for promoting renewable energy sources and energy-efficient power generation to meet future energy demands. The District of Columbia and 21 states have adopted Renewable Portfolio Standard (RPS) programs, requiring varying amounts of renewables in their electricity supply. For example, California requires 20 percent renewable generation by 2017, New York requires 25 percent by 2013, and Pennsylvania requires 18 percent by 2020. (These percentages are not exactly comparable, because the states vary in the resources they define as renewable.) States have also established funding initiatives to promote renewable energy projects. These programs can be an important complement to the approaches recommended above.

Pulp and Paper

The pulp and paper industry is divided into three segments: pulp making, paper making and converting operations. The pulp making process is the largest source of emissions, accounting for over 75 percent of the sector's $PM_{2.5}$, SO_2 and NO_x emissions. Over 80 percent of the pulp mills in the U.S. use the kraft pulping process. There are four primary sources of emissions from kraft pulping operations: power boilers, recovery furnaces, lime kilns and smelt dissolving tanks (SDTs).

Power boilers dominate the emissions from pulp mills. The approaches discussed in Chapter 6, *Industrial and Commercial Boilers*, and in *The Highlights* for those sources, are equally applicable to power boilers used in the kraft pulping process.

There are MACT standards for PM emissions from recovery furnaces, lime kilns and SDTs. These standards are 40 to 85 percent more stringent for new sources than they are for existing sources. The MACT standards for new sources limit PM emissions to 0.034 grams per dry standard cubic meter (g/dscm) for recovery furnaces, 0.023 g/dscm for lime kilns and 0.06 kilograms per megagram for SDTs. State and local regulators should consider evaluating the feasibility of requiring existing sources to meet these more stringent standards. For example, upgrades to electrostatic precipitators (ESPs) and replacement of wet scrubbers with ESPs can significantly reduce PM emissions. Older model ESPs on recovery furnaces have collection efficiencies close to 90 percent, while newer model ESPs have collection efficiencies greater than 99 percent.

While there are federal standards for SO_2 and NO_x emissions from power boilers at pulp and paper facilities, there are no federal NSPS and MACT standards for SO_2 or NO_x emissions from other pulping emissions sources. Although the options for reducing NO_x emissions from these sources are more limited, significant reductions in SO_2 emissions from recovery furnaces and lime kilns at kraft pulp mills are feasible. Some facilities have successfully lowered SO_2 emissions from recovery furnaces by reducing the sulfur content of the process-based fuels and by regulating temperatures in the furnace to minimize SO_2 formation. Where these techniques are not practical or successful, facilities should consider using a wet scrubber for SO_2 control.

Much like a number of the other industry sectors we have discussed, pulp and paper manufacturers are candidates for facility-wide emissions caps for PM, SO_2 and NO_x, on account of the number of their emissions sources and potential reduction strategies. In fact, the MACT standards for PM emissions from recovery furnaces, SDTs and lime kilns already include the option of a facility-wide emissions limit as an alternative to compliance with unit-specific standards. If regulators pursue the cap approach for all three pollutants, they should consider including power boilers, in light of their contribution to the overall emissions profile of these facilities.

Cement Manufacturing

The largest source of emissions in cement manufacturing—and the centerpiece of the process—is the kiln. Cement kilns generate over 40 percent of the PM emissions and more than 80 percent of both the SO_2 and NO_x emissions associated with cement manufacturing.

More than 80 percent of the burners used to heat cement kilns use coal, and the remainder use other fossil fuels or waste materials combined with fossil fuels. A significant portion of the NO_x emissions and the SO_2 emissions come from this fuel combustion, although raw material composition also influences SO_2 emissions significantly. PM emissions come from fuel combustion and from the handling, grinding and storing of raw materials, clinker and the final product.

States and localities have significant opportunities to reduce SO_2 and NO_x emissions from cement operations, especially in light of the fact that there are currently no federal NSPS for this industry. Recent advancements in selective non-catalytic reduction (SNCR) technology make it suitable for use on cement kilns. Although there is only one SNCR device currently installed at a cement plant in the U.S., there are over 32 SNCR systems installed on kilns in Germany and many more in the rest of Europe.

Recently approved permits in Florida have required the installation of SNCR controls with low-NO_x burners (LNBs) and multi-staged combustion as BACT for NO_x. BACT determinations that include all three technologies include NO_x limits as low as 1.95 pounds per ton (lb/ton) of clinker (30-day average). Recent BACT determinations that do not include SNCR, but do include LNBs and multi-staged combustion have NO_x limits of 2.8–5.52 lb/ton of clinker.

Sulfur levels in the fuel and raw materials heavily influence SO_2 emissions rates from cement kilns. Cement kiln systems have highly alkaline internal environments that can absorb up to 95 percent of potential SO_2 emissions. For this reason, even if they burn fuels that are relatively high in sulfur, preheater/precalciner kilns can virtually eliminate SO_2 emissions. However, without the use of raw materials that are low in sulfur, uncontrolled emissions from preheater/precalciner kilns can be as high as 7.6 lb/ton of clinker. By contrast, recent BACT determinations have set SO_2 limits ranging from 0.20 to 2.16 lb/ton of clinker. In the absence of add-on controls, the use of low-sulfur raw materials is essential for the control of SO_2.

Where the process itself does not achieve satisfactory SO_2 emissions levels, wet flue gas desulfurization (FGD) technology can provide an SO_2 control efficiency of 90–99 percent. Use of wet FGD systems in the cement manufacturing process can be complicated by particle build-up and clogging, but LADCO has concluded that these problems are manageable if the FGD device is installed downstream of an efficient fabric filter. Of more than 100 cement plants in the country, only five currently use wet scrubbers to control SO_2, suggesting substantial opportunities for the industry to improve its emissions profile. Dry FGD technology (not recommended for wet kilns) and lime spray injection are other SO_2 control options, although they are less effective.

Federal NSPS and MACT standards limit particulate emissions from cement manufacturing. Recently promulgated MACT standards set PM limits for cement kilns using hazardous waste as fuel. These standards are substantially more stringent than the NSPS and MACT standards for PM for fossil fuel-fired cement kilns. State and local regulators should require kilns that burn fuels other than hazardous waste to meet the more stringent standards, absent a showing that a particular plant cannot achieve these levels.

Additionally, recent BACT determinations for PM and particulate matter less than 10 micrometers (PM_{10}) for combined kiln and clinker cooler emissions are about a quarter of the federal NSPS and MACT PM limits for combined kiln and clinker cooler emissions for cement facilities burning *non-hazardous* materials.

Almost all stages of the manufacturing process include particle capture devices, most frequently fabric filters or ESPs, each with control efficiencies of 95–99 percent. Control device collection efficiencies can be improved by rebuilding ESPs with a larger number of collection areas and increased treatment times, and using fabric filters in combination with ESPs.

Regulators should consider as a model the rules recently promulgated by the South Coast Air Quality Management District (AQMD) to control fugitive PM emissions from cement manufacturing. Among other things, the rules require the enclosure of many parts of the cement manufacturing operation, and mandate the ventilation of enclosed areas to a control system.

Iron and Steel

Coke making

Coke making involves the heating of coal in coke ovens at high temperatures until all volatile components evaporate. The best way to reduce emissions from coke making is to reduce the amount of coke produced. Pulverized coal or other fossil fuels may substitute for some portion of the coke used in the blast furnace. Further, a number of relatively new coke production processes reduce coking emissions (e.g., using a non-recovery coke battery), and technologies exist to produce iron and steel without using coke at all.

In the production of coke, it is important to avoid large temperature fluctuations (thereby reducing damage to the coke oven battery) and incomplete coking (which results in “green pushes”), in order to minimize PM emissions. Emissions should also be controlled by staged charging, which involves introducing coal into the oven at a

controlled rate.

All quench towers should have baffles that are cleaned periodically, and clean water should be used for quenching. Dry quenching is expensive, but is even more effective in reducing emissions.

SO_2 emissions can also be reduced by desulfurizing coke oven gas before it is burned. Only 11 of the 16 byproduct recovery coke plants do so, and state and local regulators should consider requiring this. The U.S. Steel plant in Allegheny County, Pennsylvania has managed to produce coke oven gas with hydrogen sulfide levels between 15-20 grains per 100 dry standard cubic feet.

Allegheny County stands at the forefront in a number of other respects, and regulators elsewhere may wish to consider its rules. Allegheny County sets instantaneous limits for visible emissions from doors, charging, lids and offtake systems, as well as for PM emissions from pushing and combustion stacks. Because coking emissions can be controlled to some degree by a careful program of maintenance—e.g., door cleaning and rebuilding, application of sealing material on coke oven doors—workers are required to undergo extensive training. Indiana has also set opacity limits for bypass heat exchanger stacks and for pushing controls.

Iron making

The blast furnace converts iron ore into a more pure and uniform iron. Casting, the main source of blast furnace emissions, is the process of periodically removing molten iron and slag from the furnace. About half of U.S. blast furnaces control casthouse emissions with covered runners and by evacuating emissions through capture hoods ducted to a baghouse. The half of U.S. blast furnaces that do not have these controls have opportunities for significant reductions.

Steel making

Most integrated mills use basic oxygen furnaces, or BOFs, for the final step of making iron into steel. The oxygen blow portion of the furnace cycle, which involves introducing oxygen into the furnace to refine the iron, accounts for the largest share of emissions, followed by tapping (pouring the molten steel into a ladle) and charging (the addition of molten iron and metal scrap to the furnace).

Primary emissions during oxygen blow periods are typically controlled with an open hood directed to an ESP or wet scrubber, or by a closed hood ducted to a wet scrubber. According to EPA, fabric filters would provide significantly better PM control, but are not used at any facility in the U.S. Upgrading old scrubbers to scrubbers with a higher pressure drop and upgrading ESPs will also reduce primary emissions.

About half of BOF shops rely on the primary collection system to capture some of the fugitive emissions from BOF operations. Regulators should consider requiring the addition of secondary collection systems, which would significantly enhance the pollution control of these furnaces.

Sinter plants

There are only five sinter plants in the U.S. These plants convert fine-sized raw material into an agglomerated product (sinter) to be charged into a blast furnace. Although all the plants operate sinter coolers to cool the product prior to storage, only one has a control device. The other four vent directly to the atmosphere. Requiring these four to install control devices for their coolers represents the most significant emissions reduction opportunity for sinter plants.

State and local agencies should also consider Indiana's regulations on the oil and grease content of sinter plant feedstock.

Minimills

Minimills bypass the coke and iron making processes by producing steel from metal scrap using electric arc furnace (EAF) technology. All plants should be required to use a baghouse to control primary emissions from scrap melting, as well as hoods and baghouses to control emissions from the ladle metallurgy process and from the argon oxygen decarburization vessel.

All minimills control fugitive emissions from charging, tapping and melting with baghouses, but ten plants are subject to opacity limits for fugitive emissions that are not as stringent as the NSPS. Regulators should consider adopting opacity limits for these plants that are at least as stringent as the NSPS requirements.

Petroleum Refineries

Petroleum refineries are complex facilities with numerous sources of air pollution, including boilers, process heaters, catalytic cracking units, internal combustion engines and flares. Although no single control technology or combination of controls will be applicable to all cases, facilities have a wide range of opportunities for reducing emissions.

Because of the large number of refinery emissions sources and potential reduction strategies, state and local agencies should consider adopting facility-wide emissions standards for refinery combustion units, allowing sources to average

emissions rates across units. California's Bay Area AQMD limits NO_x emissions from boilers, steam generators and process heaters to a refinery-wide NO_x standard of 0.033 lb/MMBtu. The facility-wide approach has allowed refinery operators to customize compliance strategies for their facilities. For example, one San Francisco Bay Area refinery reduced its process heater NO_x emissions to less than 20 parts per million (ppm), and its power boiler NO_x emissions to less than 25 ppm.

In Texas, the Houston-Galveston region established a NO_x cap-and-trade program in 2000 that included the region's refineries. The goal of the program is to reduce industrial point source NO_x emissions by an average of 80 percent from 1997 levels. Like facility-wide emissions standards, the trading approach allows sources flexibility to address a large number and variety of emissions sources. In response to this flexibility, refinery operators implemented a wide range of strategies, including the retrofit of large gas turbines with selective catalytic reduction (SCR) systems; the decommissioning of smaller, lower efficiency steam boilers; and the conversion of generating units to combined heat and power systems.

The federal NSPS for catalytic cracking units and sulfur recovery plants are outdated. In lieu of, or in combination with, a comprehensive facility-wide approach, state and local agencies should consider adopting more stringent PM and SO_2 emissions standards for these units, and should impose stringent NO_x standards. (There are no NSPS for NO_x for these units.) Since 2000, EPA has entered into consent decrees with 83 refineries (17 companies), comprising 77 percent of the nation's refining capacity. State permitting authorities should look to the emissions limits and control options required by the consent decrees in developing updated PM, SO_2 and NO_x emissions standards for catalytic cracking units, sulfur recovery plants and other units. For example, several of the consent decrees require refinery owners to install wet gas scrubbers on their fluidized catalytic cracking units in order to limit both PM and SO_2 emissions.

State and local agencies should also consider adopting rules to better manage PM, SO_2 and NO_x emissions from flaring activities. The Bay Area AQMD and the South Coast AQMD have adopted similar rules addressing flare gas emissions that should inform other state rulemakings. Both rules, which require the preparation of flare gas minimization plans, were preceded by requirements to monitor and report flare gas emissions. These monitoring requirements led to significant emissions reductions. For example, in 2004, refineries in the South Coast area reported an 80 percent reduction in SO_2 emissions associated with flaring since they began monitoring and reporting their flare gas emissions. Subsequent Bay Area AQMD and South Coast AQMD rules require flare gas minimization plans and are designed to reduce emissions further.

Diesel Trucks and Buses

Although fewer in number than cars, diesel-powered trucks and buses have a greater impact on air quality. NO_x emissions from these vehicles account for about 20 percent of *all* NO_x emissions, including stationary, area and mobile sources. Additionally, almost all of the PM emissions from trucks and buses are $PM_{2.5}$.

More stringent federal NO_x standards for *new* onroad heavy-duty diesel engines will be phased in between 2007 and 2010, and more stringent federal PM standards will go into full effect in the 2007 model year. Also, federal regulations will reduce sulfur levels in onroad diesel fuel in 2006. These new standards will have a dramatic effect on emissions from this sector in the future. However, trucks and buses have a long lifetime, meaning that state and local regulators will have significant opportunities for at least a decade to control emissions from older *existing* vehicles (and there is no Clean Air Act preemption affecting state and local regulation of *existing* trucks and buses).

State and local emissions programs imposing emissions standards on existing trucks and buses fall into three categories: (1) voluntary; (2) mandatory for all vehicles of a given type (e.g., all heavy-duty trucks above a certain weight); and (3) mandatory for certain types of vehicles that the government buys or that are covered by government contracts (e.g., school buses, refuse haulers). States can also increase taxes and registration fees for older vehicles to encourage their retirement.

Voluntary replacement and retrofit programs need funding in order to be successful. Most such programs provide grant funding, as do California's Carl Moyer Memorial Air Quality Standards Program, the Texas Emissions Reduction Program, and programs in New York, New Jersey and the Puget Sound area. Some vehicle replacements and retrofit technologies have short payback periods because they result in fuel savings, and are good candidates for revolving loan programs.

There are also numerous examples of both kinds of *mandatory* programs—those that apply to all vehicles of a given type and those that apply to vehicles subject to government contracts. California has required retrofits of various fleets. New York City has mandated the retrofit of several types of heavy-duty vehicles, including school buses, city-licensed sightseeing buses and garbage trucks used for all city contracts.

Regulators should also adopt idling limitations to reduce the fuel use and emissions of trucks and buses, as more than 20 cities and states have done. These regulations

ban *unnecessary* idling (for example, idling a truck while making deliveries) for more than a specified period of time. By contrast, the idling done by long-haul truckers (sometimes for as much as 12 hours per day) is necessary to maintain heating, cooling and other amenities while drivers are resting in their sleeper cabs. For these vehicles, idling limitations require modifications to individual trucks or the addition of infrastructure at truck stops. However, these investments are likely to have short payback periods because of the resulting fuel savings, and may be amenable to funding through revolving loan programs. EPA has issued guidance on State Implementation Plan (SIP) credit for programs that reduce the idling attributable to the use of sleeper cabs.

Programs that encourage the maintenance of proper tire inflation will reduce fuel use and emissions, especially for long-haul truckers. Moreover, lowering speed limits where possible, and enforcing existing speed limits, will also cut emissions by reducing fuel use. Diesel trucks and buses that lower their speed from 65 to 55 miles per hour (mph) use approximately 20 percent less fuel.

Nonroad Equipment

The New York City emissions inventory for 1999 is illustrative of the air pollution problems associated with nonroad equipment: 45 percent of PM emissions and 26 percent of NO_x emissions from all mobile sources in New York City came from construction equipment.

Nationally, nonroad diesel equipment contributes about as much to the inventory of NO_x emissions as do trucks and buses—about 20 percent of the total, including stationary, area and mobile sources. Also like the emissions from trucks and buses, almost all of the PM from the nonroad category is $PM_{2.5}$. Although the Clean Air Act preempts states from regulating some kinds of nonroad equipment (e.g., aircraft, certain small engines), they nonetheless have significant opportunities to reduce emissions from this sector.

Like the diesel standards for trucks and buses, EPA emissions limits for nonroad equipment are becoming more stringent, but at a slower pace; it will take until 2016 for the onroad and nonroad diesel standards to achieve general parity. In light of the lag in regulations and the long lifetime of this equipment (as much as 40 years), *existing* nonroad equipment is an even better target than onroad vehicles for retirement and retrofit programs.

Similar to trucks and buses, state programs imposing emissions standards on existing nonroad equipment fall into three categories: (1) voluntary; (2) mandatory for all vehicles of a given type (e.g., portable engines, as California has done); and (3) mandatory for certain types of vehicles that the government buys or that are covered by government contracts (e.g., construction equipment on public projects).

As in the onroad context, *voluntary* replacement and retrofit programs need a funding source to be successful, and many of the same grant programs apply to onroad and nonroad vehicles. Grant programs in California, Texas and Washington State have funded hundreds of nonroad emissions reduction projects. And once again, technologies with short payback periods from fuel savings are good candidates for revolving loan programs.

Mandatory retrofit and replacement programs that apply to all vehicles of a given type are more difficult (but feasible) to apply in the nonroad context than the onroad context for a number of reasons. For one thing, privately owned nonroad equipment is usually not required to be registered with the state. Another reason is that states cannot adopt their own standards for existing nonroad equipment (which is different from the Clean Air Act provisions for trucks and buses and light-duty vehicles)—the Clean Air Act confines states to California or federal standards. However, California has recently adopted mandatory retrofit requirements for portable diesel engines used in a variety of equipment, and states are free to mandate these standards. The California rules include requirements for agricultural pumps, airport ground support equipment, oil drilling rigs and portable generators. The rules are intended to result in a 95-percent reduction in PM emissions from these engines by 2020.

The opportunity to encourage or mandate the use of reduced-sulfur fuels arises from the lag time in federal regulations for *nonroad* diesel fuels as compared to *onroad* diesel fuels. Federal regulations will reduce sulfur levels in *onroad* diesel fuel in 2006, but sulfur limits for most *nonroad* diesel fuel will be phased in between 2007 and 2010. As a result, the use of reduced-sulfur onroad diesel fuel in nonroad equipment between now and 2010, or the use of other alternative fuels, can reduce direct PM emissions and, more importantly, will make retrofit devices more effective. (California regulations will also reduce sulfur in onroad diesel fuel in 2006. The adoption of California diesel fuel rules involves some complexities, but would allow a state to *mandate* the use of reduced-sulfur onroad diesel fuel in nonroad equipment, regardless of whether the equipment is used for government services.)

Idling restrictions are somewhat less feasible on nonroad than on onroad vehicles for a number of reasons. However, this is not true for switcher yard locomotives, which often idle excessively, and states and local areas should consider adopting these restrictions. Voluntary programs in a number of states and local areas, including California, Chicago, the Seattle-Tacoma area and Texas provide funding for locomotive idle reduction programs. The payback periods on these programs are often short (6–20

months). EPA has issued guidance on taking SIP credit for locomotive idle reduction programs.

Light-Duty Cars and Trucks

Light-duty cars and trucks, the majority of which are gasoline fueled, contribute about 16 percent of the NO_x emissions from *all* sources—stationary, mobile and area combined. But these vehicles contribute much less direct $PM_{2.5}$ than do heavy-duty diesel vehicles, and the SO_2 contribution of this sector will fall dramatically beginning in 2006, when allowable fuel sulfur levels for gasoline are reduced.

Starting with the 2004 model year, EPA implemented stricter emissions standards for cars and light trucks, as did California. Given the relatively fast turnover rate of the light-duty fleet, these standards will have an effect in the short term, as new vehicles replace older ones. EPA estimates that annual NO_x emissions from light-duty vehicles will fall by 66 percent by 2020 due to normal fleet turnover, despite a 20-percent increase in annual vehicle miles traveled.

Largely because of the turnover rate of these vehicles, retrofits will not be the best strategy for reducing emissions from the light-duty (as compared to the heavy-duty diesel) fleet. Instead, strategies that increase the vehicle turnover rate or encourage fleets and individuals to purchase the cleanest vehicles available will accelerate reductions. Many states and local areas (including the Bay Area, California, Colorado, Connecticut, Illinois, Maine, New Jersey and New York) have adopted such programs, and others should consider doing so. Strategies include:

- monetary incentives for individuals and fleet owners to make clean choices, including the choice of alternative fuel vehicles when buying new vehicles (e.g., scrappage programs, tax rebates, tax exemptions, reductions in vehicle registration fees); and
- non-monetary incentives for the purchase of cleaner vehicles (e.g., permission to use high-occupancy vehicle (HOV) lanes, exemption from state emissions tests, free parking at street meters and municipal parking lots).

About ten states have adopted California's low-emissions vehicle (LEV) standards for new cars, which are more stringent than EPA's standards. Other states should consider adopting these LEV II standards instead of EPA's Tier 2 standards.

Burning less fuel means less air pollution. States and localities can adopt legislation and policies to reduce vehicles miles traveled and otherwise reduce fuel use and emissions from the light-duty fleet, such as:

- increasing or improving public transportation;
- encouraging non-emitting modes of transportation by building or improving bicycle paths and pedestrian walkways;
- adopting and publicizing employee commuting benefits;
- establishing HOV lanes;
- enhancing traffic management and reducing congestion; and
- keeping maximum highway speeds to 60 mph (by reducing maximum speed limits where possible, and enforcing existing speed limits).

Even states without a federal inspection and maintenance (I&M) mandate are free under federal law to adopt an I&M program. A committee of the National Research Council of the National Academies of Sciences concluded that well-structured I&M programs are one of the most cost-effective vehicle strategies for reducing vehicle emissions of those it evaluated. In light of the fact that a small proportion of vehicles create a disproportionate share of emissions, regulators should consider adopting I&M programs designed to target these vehicles.

Airports

The numerous urban $PM_{2.5}$ areas that are home to one or more major airports will be interested in options to address the increasing level of PM and NO_x emissions from airports. Although states have no authority to regulate aircraft engines, which dominate airport emissions inventories, they have numerous opportunities to reduce emissions from airport ground service equipment and ground transportation vehicles.

Airport ground service equipment includes baggage tugs, belt loaders and aircraft pushback tractors, many of which are diesel fueled. These pieces of equipment are candidates for the same kinds of emissions reductions strategies that apply to nonroad equipment generally, including the retrofit and replacement of older vehicles and the use of onroad reduced-sulfur diesel fuel or other alternative fuels.

Ground transportation fleets are also candidates for retrofit and replacement; these include the predominantly diesel-fueled shuttle buses that ferry passengers to airport parking and car rental lots and to hotels. For example, the South Coast AQMD has required airport fleet operators to purchase or lease alternative-fueled vehicles when adding or replacing vehicles. In addition, airports should

adopt and enforce anti-idling rules for diesel buses, which generate significant excess emissions while waiting, at idle, for passengers.

As is the case for marine ports, the optimum mix of control strategies will vary from airport to airport, depending on fuel availability, existing infrastructure, existing vehicle technologies and other factors. However, the variety of emissions sources and the range of available reduction strategies also make airports good candidates for programs that cap their overall emissions. Facility-wide emissions caps encourage the comprehensive evaluation of the most cost-effective control options. For example, the Texas Commission on Environmental Quality (TCEQ) negotiated a voluntary agreement with the Dallas/Fort Worth International Airport to reduce NO_x emissions. As part of the agreement, the air carriers agreed to reduce ground service equipment NO_x emissions by 75 percent relative to 1996 levels. TCEQ also negotiated a voluntary agreement with Continental Airlines, Southwest Airlines and the City of Houston to reduce NO_x emissions. Massport, the operator of Boston's Logan Airport, established a cap on airport NO_x emissions; any emissions increases that result from airport activities must be offset by emissions reductions on site or near the airport, or by the purchase of emissions reduction credits.

The Federal Aviation Administration's Voluntary Airport Low Emissions (VALE) Program provides funding for LEVs, refueling and recharging stations, gate electrification and other airport air quality improvement measures at commercial service airports in nonattainment and maintenance areas.

Marine Ports

Over 30 of the largest U.S. ports are in areas that are in nonattainment for $PM_{2.5}$, ozone, or both. While emissions inventories vary from port to port, the Ports of Long Beach and Los Angeles are instructive: their mobile sources account for about 25 percent of the total PM from all mobile sources in the Los Angeles area.

Most of the PM and NO_x emissions from ports come from marine vessels: ocean-going ships (which states cannot regulate), auxiliary engines on these ships, and commercial harbor craft. Cargo-handling equipment is the biggest land-based mobile source contributor. All of these sources are diesel powered, and almost all of their PM emissions are $PM_{2.5}$.

As home to large numbers of heavy-duty diesel vehicles, marine ports are candidates for the same emissions reduction strategies that otherwise apply to trucks, buses and nonroad equipment. These include the retrofit and replacement of older vehicles, the use of onroad reduced-sulfur diesel fuel or other alternative fuels in nonroad equipment, and limits on vehicle idling. Some port vehicles—like Category 1 marine engines larger than 600 horsepower (e.g., tugboats) and some material handling equipment—are particularly good candidates for repowering because of the greater fuel efficiency of replacement engines. Moreover, because of their typical governance structure (public or semi-public), many ports are in a good position to implement some or all of these measures or to require that terminal operators do so. This is because, as previously noted, mandatory replacement and retrofit programs are more feasible if they apply to nonroad equipment that is subject to government purchasing or contracting requirements.

Their localized nature also provides opportunities for marine ports to make or require changes to nonroad vehicles that might otherwise be infeasible. For example, several ports (e.g., the Ports of Los Angeles and of Juneau, Alaska) have made the infrastructure improvements that allow the "hotel loads" on large ships to be supplied by land-side electric power (called "cold-ironing") while they are docked, rather than with on-board auxiliary engines. (Cruise ships typically spend up to a full 24 hours docked, while exchanging passengers; and some cargo vessels take 100 hours or more to unload.) Similarly, ports from New York to Seattle have replaced diesel-powered cranes with electric cranes.

Marine ports have a range of other options for reducing emissions, such as programs that encourage ships to operate at lower speeds near the coast (e.g., programs at the Port of Long Beach and the Port of Los Angeles); and operational changes that reduce truck queuing and idling (e.g., measures at the Georgia Ports Authority and the Port of Virginia). Because of the variety of options available, marine ports, like airports, are excellent candidates for programs that cap their overall emissions, thereby facilitating the identification of the most cost-effective reduction opportunities.

Residential Fuel Combustion and Electricity Use

The residential source category produces $PM_{2.5}$ and $PM_{2.5}$-precursor emissions on-site from the direct consumption of fuels—such as natural gas, liquefied propane gas, kerosene, fuel oil, coal and wood. Additionally, an even larger share of the emissions attributable to the source category occurs off-site, at fossil fuel-fired power plants. In light of emissions considerations and widespread concern regarding the rising costs of fossil fuels, residential energy-efficiency programs should be part of the strategy for delivering air quality improvements. State and local regulators have a number of options in this regard.

Regulators should consider promoting the tax incentives

contained in the Energy Policy Act of 2005 and also adopt complementary state and local programs to further encourage the deployment of energy-efficient technologies. For example, under the Energy Policy Act, households that purchase and install energy-efficient windows, insulation, and heating and cooling equipment can receive a tax credit of up to $500 beginning in January 2006.

If they have not already done so, state and local agencies should consider regulating NO_x emissions from residential furnaces, one of the largest on-site sources of NO_x emissions in the residential category. In California, several air districts, including the South Coast AQMD, the Bay Area AQMD and the San Diego County Air Pollution Control District, have adopted NO_x emissions standards for natural gas-fired central furnaces. These rules suggest a starting point for establishing state standards. However, regulators should also evaluate the feasibility of more stringent standards, in light of the fact that the standards in California were first established in 1978, and burner technology has made significant advances in NO_x control since that time.

States that are reliant on home heating oil should regulate its sulfur content to 0.05 percent sulfur by weight (500 parts per million (ppm)). Currently in the U.S., heating oil for residential use has an average sulfur content of about 0.20–0.25 percent. Switching to low sulfur content fuel could eliminate 75–80 percent of the SO_2 emissions generated by residential oil heating systems, as well as 80 percent of $PM_{2.5}$ emissions. Switching to low-sulfur oil can also reduce maintenance and service requirements. The American Society for Testing and Materials, an international voluntary standards development organization, has approved a Low Sulfur No. 2 Heating Oil specification, and industry trade associations have advocated a switch to low-sulfur heating oil.

Replacing an older wood stove with an EPA-certified model can significantly reduce a home's direct $PM_{2.5}$ emissions. This is particularly true as the costs of heating oil and natural gas rise and households become more reliant on wood stoves for heating. Programs in Libby, Montana and Allegheny County, Pennsylvania, initiated in 2005, provide a model for other communities considering a wood stove changeout initiative. The Energy Policy Act of 2005 provides tax incentives for high-efficiency wood stoves. States can promote this incentive and also supplement the program with funding of their own. Other strategies should be considered as well (e.g., requiring all wood stoves that are not EPA-certified to be removed prior to the sale of a property).

State and local agencies should consider regulating PM emissions from residential outdoor wood-fired boilers, which generate large quantities of smoke. There are an estimated 100,000 of these units in the U.S., providing an alternative source of energy in the face of rising fossil fuel prices. Local news stories and growing numbers of complaints to local health agencies provide evidence of the adverse impact of these boilers on local air quality.

Regulators should also consider banning the burning of household garbage, which generates emissions of toxics and particulates. The Western Lake Superior Sanitary District has developed an extensive toolkit for local officials to assist them in evaluating and implementing alternatives.

Commercial Cooking

Charbroiling generates over 80 percent of total $PM_{2.5}$ from the commercial cooking sector. The sector's $PM_{2.5}$ emissions account for 6 percent of the *total* direct $PM_{2.5}$ emissions generated by all point source categories in 1999 (e.g., power plants and industrial facilities).

Commercial cooking establishments use two types of charbroilers: underfired and chain-driven. Most emissions (74 percent) come from the use of underfired charbroilers, although regulatory efforts have focused on control of the chain-driven charbroilers used predominantly by fast-food restaurants.

State and local air pollution control agencies should consider regulating PM emissions from new and existing chain-driven charbroilers. Some areas have already adopted regulations. In California, the South Coast AQMD requires operators of both new and existing chain-driven charbroilers to install a catalytic oxidizer (but allows alternative control devices if they are equally effective). Catalytic oxidizers appear to reduce PM emissions by over 80 percent, and are highly cost-effective ($1,680–$2,800 per ton of PM and VOCs reduced).

Control options are available for reducing emissions from underfired charbroilers, but are more costly than those available for chain-driven charbroilers. Because the South Coast AQMD has concluded that none of the options available for controlling PM emissions from underfired charbroilers meets its cost-effectiveness criteria, the agency has not regulated this source category.

Commercial cooking establishments consume substantial amounts of energy, some portion of which is wasted. For example, charbroilers generally idle at a rate close to their full heat input to be ready for the next round of cooking. Charbroilers contribute to the cooling loads in a kitchen, as they generate excess heat. Further investigation of strategies for reducing the energy use of charbroilers is warranted.

Fugitive Dust

Fugitive dust refers to particles, most commonly derived from soil, that are lifted into the air by human activities and natural forces, such as agricultural tilling, motor vehicle use and wind. The major sources of fugitive dust are paved and unpaved roads, agricultural operations, construction projects and wind erosion from both agricultural and non-agricultural lands.

There are two basic options for controlling fugitive PM emissions from paved roads: (1) prevention strategies aimed at reducing the amount of dirt and sand deposited to roadways; and (2) mitigation strategies like street sweeping, which remove the material after it has been deposited on the road surface. Unpaved roads can be paved or can be addressed with surface treatments. The primary options for reducing fugitive dust from agricultural operations include limiting tillage activities during windy conditions and reducing tillage in various ways (e.g., by adopting low-till agricultural practices).

In designing a dust control program, state and local agencies should consider focusing on targeted programs in order to minimize the costs of control, such as, paving only the most heavily traveled roadways or prioritizing them for dust control measures. The San Joaquin Valley UAPCD imposes a 25 mph speed limit on unpaved roads with over 25 vehicles per day. The District also requires the paving of unpaved roads and road shoulders, with priority given to roads with the highest traffic volumes. Clark County, Nevada requires control measures such as paving for existing unpaved roads with at least 150 vehicles per day.

Cars and trucks sometimes deposit dirt or debris onto the surface of a paved road when leaving a worksite or unpaved road. This "trackout" can be controlled with the construction of gravel beds or other control devices, which remove the dirt prior to the vehicle's entering the roadway. For example, the South Coast AQMD requires trackout control devices for construction projects exceeding five acres.

For paved roads, street cleaning operations can be targeted to minimize the costs of control by focusing on cleaning anti-skid materials and cleaning dirt deposited on a busy road as a result of wind and rain. Apart from these targeted strategies, the cycle of particle deposition on road surfaces and subsequent resuspension in the air will generally outpace efforts to keep roads swept, thereby limiting their effectiveness as a control option.

With respect to agricultural operations, the South Coast AQMD limits fugitive dust by promoting soil conservation practices such as low-till agriculture. The South Coast AQMD also limits tilling activities during high wind events: tilling and mulching activities must cease when wind speeds are greater than 25 mph.

Chapter 2

Effects of Particulate Matter on Human Health and the Environment

Introduction

Airborne particulate matter (PM) has been associated with adverse effects on human health since early in the 20th century. In fact, episodes of acute PM pollution that took place decades ago in different parts of the world spurred the development of many of the first air pollution guidelines. During such episodes—including at the Meuse Valley in Belgium in 1930; Donora, Pennsylvania in 1948; and London, England in 1952—extremely high PM levels were associated with a dramatic increase in daily mortality. In Donora, 20 residents died and 7,000 people—half the town's population—were hospitalized with difficulty breathing due to a poisonous mix of airborne particulates and gases from the smokestacks of the local zinc smelter and other sources. This tragedy, in particular, shocked the U.S. and marked a turning point in the nation's complacency about air pollution and its effects on human health.

PM is the generic term for a broad class of chemically and physically diverse substances that exist as discrete particles (liquid droplets or solids) over a wide range of sizes. Particles originate from a variety of anthropogenic stationary and mobile sources as well as from natural sources. Particles may be emitted directly or formed in the atmosphere by transformations of gaseous emissions such as sulfur dioxide (SO_2), nitrogen oxides (NO_x) and volatile organic compounds. The chemical and physical properties of PM vary greatly with time, region, meteorology and source category, thus complicating the assessment of health and welfare effects.

PM can be divided into (and is currently regulated under) two size ranges: $PM_{2.5}$ and PM_{10}. $PM_{2.5}$ denotes particles equal to or less than 2.5 micrometers (µm) in diameter.[1] PM_{10} particles are those with diameters equal to or less than 10 µm. $PM_{2.5}$ can be further divided into ultrafine particles (particles less than approximately 0.1 µm in diameter). Throughout this discussion, references to $PM_{2.5}$ include all particles equal to or less than 2.5 µm in diameter, including ultrafine particles.

$PM_{2.5}$ or "fine" particles are of particular concern to human health. One-twentieth the width of a human hair, these fine particles can be inhaled deep into the gas-exchange regions of the lung, where the thin-walled alveoli replenish the blood with oxygen.

"Coarse" particles, covering the range from about 2.5 to 10 µm in diameter, also cause adverse health effects. Some of these coarse particles are generated naturally by sea-salt spray, wind and wave erosion, volcanic dust, windblown soil and pollen. They are also produced by human activities such as construction, demolition, mining, road dust, tire wear and industrial processes involving the grinding and crushing of rocks or metals. Larger coarse particles tend to settle out of the air more rapidly than fine particles and will usually be found relatively close to their source. Fine particles, however, can be transported long

1. In this report, particle size or diameter refers to a normalized measure called "aerodynamic diameter," which accounts for the irregular shape and varying density of most particles.

distances by wind and weather, traveling thousands of miles from where they were formed.

As discussed later in this report, the concentration and composition of particle pollution in the atmosphere vary by time of year and by location and are affected by several aspects of weather, such as temperature, humidity and wind. For example, $PM_{2.5}$ in the eastern half of the U.S. contains more sulfates than those in the West, while $PM_{2.5}$ in southern California contains more nitrates than in other areas of the country. In the East, $PM_{2.5}$ values are highest from July through September, while in most of the West, $PM_{2.5}$ values are highest in the winter. Carbon is a substantial component of $PM_{2.5}$ everywhere. On a local scale, researchers have observed high concentrations of PM in close proximity to major roads and highways.

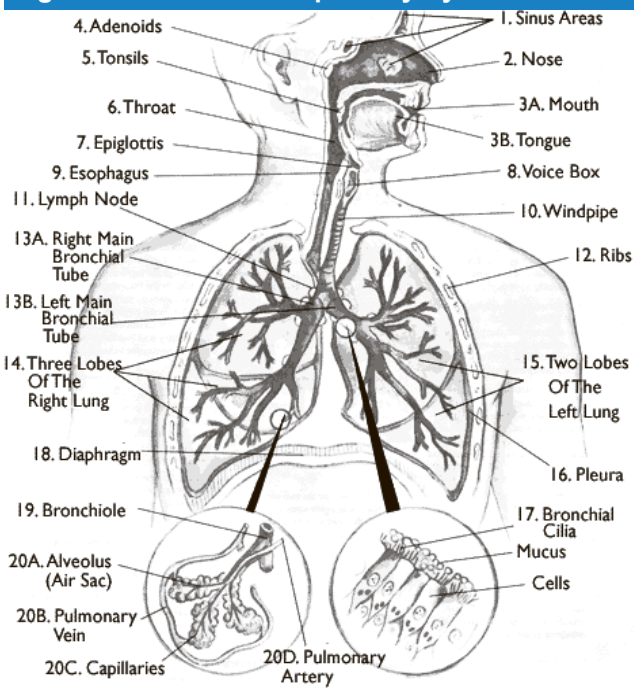
Numerous studies have linked PM (both $PM_{2.5}$ and PM_{10}) air pollution to a broad range of cardiovascular and respiratory health endpoints. Newer studies report associations between short-term exposure to various indicators of PM and cardiopulmonary mortality, hospitalization and emergency department visits and respiratory symptoms. In addition, there is now evidence for associations with cardiovascular health outcomes, such as heart attacks and changes in blood chemistry. Children and the elderly, as well as people with pre-existing cardiovascular or respiratory diseases such as asthma, are particularly susceptible to health effects caused by PM. PM is also an effective delivery mechanism for other toxic air pollutants, which attach themselves to airborne particles. These toxics are then delivered into the lungs, where they can be absorbed into blood and tissue. To the extent that the studies referenced in this chapter refer specifically to $PM_{2.5}$, PM_{10}, or the PM size fraction between 10 and 2.5 µm, we have identified the appropriate size range.

Biological Mechanisms

The health risk from an inhaled dose of PM depends on the size and characteristics of the particles inhaled. Size determines how deeply the inhaled particles will penetrate into the respiratory tract, where they can persist and cause respiratory and other damage. In general, only particles equal to or smaller than 10 µm in diameter (the PM_{10} fraction) can be inhaled deep into the lungs without first being intercepted by the nose or pharynx. Particles deposited in the alveolar region (the air sacs of the lungs) can remain in the lungs for long periods of time because alveoli lack the mucus-lined clearance system of the trachea and bronchi. This presents a particular concern because the alveoli are where oxygen exchange to the blood takes place.

When a foreign material reaches the cells of the lung, macrophages (white blood cells that reside in the tissues

Fig. 2.1 The Human Respiratory System



Source: American Lung Association

and airspaces of the lung) and other protective cells respond to the threat by attempting to engulf, degrade and ultimately expel the invader. The precise biological mechanisms that lead to adverse health effects have not been well defined, although experimental observations have suggested several hypotheses. Particles in the lung will frequently result in an inflammatory response, which can produce cell damage. Particles may also stimulate nerve cells in the underlying tissue, which in turn may affect the nervous system and its control of breathing, heart rate and heart rate variability. Ultrafine particles may themselves enter the blood stream to be transported to the liver, bone marrow and heart, with direct or indirect effects on organ function. Researchers have suggested that several physiological responses might occur in concert to produce health effects (EPA, 2005a).

If it were known which properties of PM were responsible for the preponderance of adverse health effects, emissions and air quality standards could focus on controlling the particles that present the greatest risk. Thus far, however, the laboratory and field evidence do not implicate one specific toxic quality of PM to the exclusion of others (EPA, 2005a). Qualities such as the size of the PM and the presence of certain chemical components (e.g., metals) appear to contribute to its toxicity.

Short-Term Exposure

According to EPA, short-term exposure (hours or days) to $PM_{2.5}$ and PM_{10} can aggravate lung disease, causing

asthma attacks and acute bronchitis, and may also increase susceptibility to respiratory infections. In people with heart disease, short-term exposures have been linked to heart attacks and arrhythmias. Healthy children and adults have not been reported to suffer serious effects from short-term exposures, although they may experience temporary minor irritation when particle levels are elevated.

Particulate air pollution causes greater use of asthma medications and increased rates of school absenteeism, emergency room visits and hospital admissions. Other adverse effects can include coughing and wheezing. Short-term increases in PM levels have been linked to:

- death from respiratory and cardiovascular causes;
- increased numbers of heart attacks, especially among the elderly and people with heart conditions;
- inflammation of lung tissue in young, healthy adults;
- increased hospitalization for cardiovascular disease;
- increased emergency room visits for patients suffering from acute respiratory ailments;
- increased hospitalization for asthma among children;
- increased severity of asthma attacks in children.

In the early 1990s, dozens of short-term community health studies from cities throughout the U.S. and around the world indicated that short-term increases in particle pollution were associated with adverse health effects as outlined above.

The National Morbidity, Mortality and Air Pollution Study (NMMAPS) is the largest multi-city analysis of the short-term effects of PM air pollution on human health. The study included analyses of PM_{10} effects on mortality in 90 U.S. cities (Samet, 2000a, 2000b; Dominici, 2003). Additional, more detailed, analyses were conducted based on a subset of the 20 largest U.S. cities (Samet, 2000b).

The NMMAPS used a uniform methodology to evaluate the relationship between mortality and PM_{10} for the different cities, and synthesized the results to provide a combined estimate of effects across the cities. The authors reported statistically significant associations between both cardiorespiratory mortality and mortality from all causes, and PM_{10} concentrations. The risk estimates for deaths from cardiorespiratory causes were somewhat larger than those for deaths from all causes. The results of the NMMAPS assessment held up using different modeling approaches and adjustments for gaseous co-pollutants.

Another major multi-city study used data from ten of the NMAPS cities where daily PM_{10} monitoring data were available (Schwartz, 2003). Again, the authors reported a statistically significant association between PM_{10} and total mortality, with a reported health risk larger than that reported in the NMMAPS study. The authors of the study suggest that the availability of more frequent monitoring data may partly account for the differences.

Epidemiologic studies have reported associations between short-term exposures to ambient PM (often using PM_{10}) and measures of changes in cardiac function such as arrhythmia, alterations in electrocardiogram patterns, and heart rate or heart rate variability changes (Brook, 2004). These new epidemiologic findings offer some insight into potential biological mechanisms that underlie associations between short-term PM exposure and cardiovascular mortality and hospitalization previously reported in the literature.

The American Heart Association (AHA) has conducted an extensive review of the medical literature on the health effects of particle pollution and issued a statement in 2004 concluding that exposure to $PM_{2.5}$ air pollution contributes to the development of cardiovascular diseases (AHA, 2004). Although the increase in relative risk for heart disease associated with $PM_{2.5}$ for an individual was deemed small in comparison to the impact of such cardiovascular risk factors as high blood pressure and high cholesterol, PM was identified as a serious public health problem due to the very large number of people affected and because exposure occurs over an entire lifetime.

Long-Term Exposure

Long-term exposure, such as that experienced by people living for years in areas with high PM levels, has been associated with problems like reduced lung function and the development of chronic bronchitis—and even premature death. Other symptoms range from premature births to serious respiratory disorders, even when particle levels are very low. Year-round exposure to particulate pollution has also been linked to:

- slowed lung function growth in children and teenagers;
- significant damage to the small airways of the lungs;
- increased risk of death from lung cancer;
- increased risk of death from cardiovascular disease.

Three major studies of the chronic effects of PM exposure have linked increases in mortality and long-term exposure to PM: the Six Cities, American Cancer Society (ACS), and California Seventh Day Adventist (AHSMOG) studies. More recently there has been a comprehensive reanalysis of data from the Six Cities and ACS studies, and new analyses using updated data from the AHSMOG and ACS

studies.

The reanalysis of the Six Cities and ACS studies confirms their original findings, suggesting an association with both total and cardiorespiratory mortality and exposure to $PM_{2.5}$ (Krewski, 2000). Researchers performed an extensive sensitivity analysis using alternative statistical methods and considered the role of 20 potential confounders—such as other pollutants, climate and socioeconomic factors—on study results. The study identified higher educational status as a factor associated with reduced risk to air pollution exposure and reported an association between SO_2 pollution and mortality.

The expanded analysis of the ACS cohort study found significant associations between long-term exposure to *fine particles* (using various averaging periods for air quality concentrations) and premature mortality from all causes, cardiopulmonary diseases and lung cancer (Pope, 2002). In both the reanalyses and extended analyses of the ACS cohort study, long-term exposure to the PM size fraction between 10 and 2.5 µm was not significantly associated with mortality (Krewski, 2000; Pope, 2002). Of all the long-term exposure studies, EPA places greatest weight on the results of the Six Cities and ACS studies because of the data and methodologies used.

Populations at Risk

Individuals with heart or lung disease, older adults and children are considered to be at greater risk from particulate air pollution, especially when they are physically active. Physical activity causes individuals to breathe faster and more deeply, taking more particles into their lungs.

People with heart or lung disease—such as coronary artery disease, congestive heart failure and asthma or chronic obstructive pulmonary disease—are at increased risk because particles can aggravate these diseases (EPA, 2003). Individuals with diabetes may also be at increased risk because they are more likely to have underlying cardiovascular disease.

Older adults are at increased risk, perhaps due to undiagnosed heart or lung disease or diabetes (EPA, 2003). Many studies show that when particle levels are high, older adults are more likely to be hospitalized and to die of aggravated heart or lung disease.

Children are at increased risk from exposure to PM for several reasons: their lungs are still developing; they spend more time at high activity levels; and they are more likely to have asthma or acute respiratory diseases (EPA, 2003).

It appears that the risk associated with PM exposure varies throughout a lifetime and is generally higher in early childhood, lower in healthy adolescence and young adulthood, and higher again in middle age through old age as the incidence of heart and lung disease and diabetes increases. Factors that increase the risk of heart attack, such as high blood pressure and elevated cholesterol levels, also may increase the risk associated with particulate exposure. In addition, scientists are evaluating new studies that suggest that exposure to high particle levels may also be associated with low birth weight in infants, pre-term deliveries, and possibly fetal and infant deaths (EPA, 2003).

Environmental Effects

The particles linked to serious health effects are also a major cause of visibility impairment in many parts of the U.S. Particles in the air reduce the distance at which one can see the color, clarity and contrast of distant objects because these particles scatter and absorb light. In many parts of the U.S., pollution has reduced visual range by 70 percent from natural conditions (EPA, 1997). In the East, the current range is only 14 to 24 miles, compared to a natural visibility range of 90 miles. In the West, the current range is 33 to 90 miles, versus a natural visibility range of 140 miles (EPA, 1997). (Natural visibility in the East is lower than in the West, in part because of higher relative humidity, which causes some particles to become more efficient at scattering light.)

$PM_{2.5}$ can remain suspended in the air and travel long distances. For example, exhaust from a diesel truck in Los Angeles can end up over the Grand Canyon, where one-third of the haze comes from Southern California (EPA, 1997). Emissions from a Los Angeles oil refinery can form particles that in a few days will affect visibility in Colorado's Rocky Mountain National Park. Twenty percent of the haze problem on the dirtiest days in that park is attributed to emissions generated in Los Angeles (EPA, 1997).

In the eastern U.S., reduced visibility is attributable mainly to secondary PM formed in the atmosphere from SO_2 emissions. Although these secondary particles also account for a major portion of particulate loading in the West, primary emissions from sources like wood smoke and NO_x emissions from motor vehicles and other sources contribute a larger percentage of the total particulate loading in the West.

In addition to affecting visibility, airborne particles can also lead to ecosystem damage. The most significant PM-related ecosystem effects result when the long-term, cumulative deposition of nitrates and sulfates exceeds the natural buffering or storage capacity of the ecosystem and affects the nutrient status of the ecosystem, usually by indirectly changing soil chemistry, populations of bacteria

involved in nutrient cycling, and/or populations of fungi involved in plant nutrient uptake. Nitrogen and sulfur in varying amounts are necessary and beneficial nutrients for most organisms. However, excessive amounts of these nutrients from atmospheric deposition can lead to unintended ecosystem changes such as species shifts, loss of species richness and diversity, and impacts on threatened and endangered species.

In addition to altering ecosystem chemistry, particulate pollution also causes damage to vegetation directly. Experiments on seedling and sapling trees have shown significant damage to leaf surface structures after exposure to simulated acid rain or acid mist (resulting primarily from the PM-precursors SO_2 and NO_x) at pH levels of 3.5 (EPA, 2005a).[2] Epicuticular waxes, which function to prevent water loss from plant leaves, can be destroyed by acid rain in a few weeks. The proper functioning of epicuticular wax in conifers is especially crucial because of their longevity and evergreen foliage. For example, red spruce seedlings, which have been extensively studied, appear to be more sensitive to acid precipitation (mist and fog) than other species (EPA, 2005a). Other direct responses of forest trees to acid precipitation include increased permeability of leaf surfaces to toxic materials, water, and disease agents; increased leaching of nutrients from foliage; and altered reproductive processes. All of these effects weaken trees, leaving them more susceptible to other stresses (e.g., extreme weather, pests and pathogens).

Airborne particles can also cause soiling and other damage to materials like concrete and limestone. The effects of PM on materials have been investigated for metals, wood, stone, painted surfaces, electronics and fabrics. Particulate pollution may soil and discolor these materials, reducing their aesthetic appeal. It may also cause other physical and chemical degradation of materials through the action of acidic particles.

PM NAAQS Review

As discussed in Chapter 3, EPA is required by the Clean Air Act to conduct periodic reviews of the National Ambient Air Quality Standards (NAAQS) for criteria pollutants, including PM. EPA must determine whether the latest scientific information suggests a need to revise the standards. In addition, the Act also requires an independent scientific review committee to provide scientific and technical advice to the EPA Administrator on issues related to the NAAQS. The Clean Air Scientific Advisory Committee (CASAC) has performed this function since the 1980s.

On January 17, 2006, EPA published its proposal for addressing the PM NAAQS. Among other things, the Agency proposed lowering the level of the 24-hour $PM_{2.5}$ standard from 65 micrograms per cubic meter ($\mu g/m^3$) to 35 $\mu g/m^3$, retaining the level of the annual $PM_{2.5}$ standard at 15 $\mu g/m^3$, and setting a new 24-hour standard for $PM_{10-2.5}$ at 70$\mu g/m^3$. EPA also proposes to exempt agricultural sources, mining sources and similar sources of crustal material from control in meeting the standards.

Prior to the Agency's proposal on the PM NAAQS, EPA's staff recommendations—which represent an intermediate step between the Agency's detailed scientific review (embodied in the PM criteria document) and the policy decisions that the Administrator must ultimately make—concluded that the current standards are not sufficiently protective and should be tightened (EPA, 2005a). On the whole, CASAC members agreed with most of the EPA staff recommendations (EPA, 2005b). Most notably, the Committee agreed with EPA staff that the primary 24-hour and annual $PM_{2.5}$ standards should be revised to provide greater public health protection.

In justifying its recommendations, the staff paper pointed to several studies demonstrating associations between cardiovascular mortality and morbidity and short-term $PM_{2.5}$ exposure at levels below the current standard. For instance, while the current annual standard is 15 $\mu g/m^3$, three studies have demonstrated increased mortality in areas where $PM_{2.5}$ concentrations ranged between 13 and 14 $\mu g/m^3$. Three other studies demonstrate an increase in emergency department visits in areas with a mean $PM_{2.5}$ concentration at or below 12 $\mu g/m^3$. In addition, risk assessments demonstrate that morbidity would remain high and thousands of premature deaths would occur even if urban areas across the country attained the current $PM_{2.5}$ standard. While the Agency is not required by statute to set standards that will eliminate all risk, the standards must protect public health with an adequate margin of safety.

The current PM NAAQS review process will continue in tandem with states' efforts to achieve the current standards. EPA is under a court order to finalize its decision by September 2006.

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2. The initials pH stand for "potential of Hydrogen" — a measure of how acidic or alkaline a substance is. Acids have pH values under seven and alkalis have pH values over seven.

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Chapter 3

Fine Particulate Matter and Precursor Emissions

Introduction

Particulate matter (PM) is a complex pollutant that occurs throughout the U.S. in both urban and rural areas. Unlike other criteria pollutants, PM comprises a broad class of pollutants that includes a wide range of particle sizes and chemical constituents. Airborne particles can range in size from a few nanometers to upwards of 100 micrometers (µm) in diameter. As noted in Chapter 2, the term $PM_{2.5}$ denotes particles with an aerodynamic diameter equal to or less than 2.5 µm. Some particles are liquid, some are solid and others contain a solid core surrounded by liquid. Common constituents include sulfates, nitrates, ammonium, elemental carbon, a variety of organic compounds, water, and inorganic substances (including metals, dust, sea salt and other trace elements) that are often categorized as "crustal" material.

EPA and the states have established a network to monitor and record $PM_{2.5}$ concentrations throughout the U.S. Data from this network show that national average particle concentrations have been declining. Despite progress to date, however, areas throughout the eastern U.S. and California exceed EPA's National Ambient Air Quality Standards (NAAQS) for $PM_{2.5}$.

In this chapter, we discuss the characteristics of ambient PM, including size and chemical composition. We also discuss the sources that contribute to ambient $PM_{2.5}$ concentrations.

Characteristics of PM

PM air pollution is often discussed in terms of particle size because of the distinct characteristics (origin, chemical species and atmospheric behavior) associated with different particle size classes. In the discussion that follows, we distinguish among three particle sizes: (1) coarse particles in the range of about 2.5 to 10 µm in diameter; (2) fine particles (excluding ultrafine particles) in the range of about 0.1 to 2.5 µm in diameter, sometimes called accumulation mode particles; and (3) ultrafine particles less than approximately 0.1 micrometers in

diameter, sometimes called nuclei mode particles.[1]

Figure 3.1 shows a distribution of particle sizes typical of an urban environment in the U.S. As the figure shows, there is overlap among the different size classes. Formation mechanisms are also considered when distinguishing among categories of PM. Figure 3.1 reveals that particles larger than 0.1 µm in diameter account for most of the *particle mass* in U.S. urban areas. In terms of total *number*, however, most of the particles in urban areas are smaller than 0.1 µm in diameter (ultrafine particles).

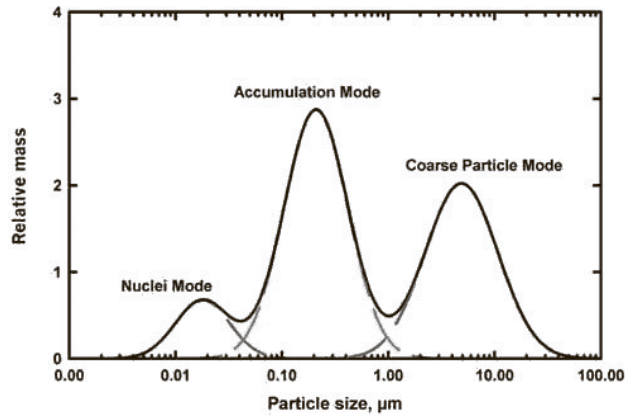
Coarse Particles. In general, coarse PM—in the range of about 2.5 to 10 µm in diameter—is made up of primary particles, emitted directly from their sources as particles (EPA, 2005a). Most coarse particles result from the mechanical disruption of larger particles by crushing or grinding, from the evaporation of sprays or from the re-suspension of dust (crustal material). Specific sources include construction and demolition activities, mining and mineral processing, sea spray, wind-blown dust and the re-suspension of settled biological material from soil surfaces and roads. Some combustion-generated particles, such as fly ash, are also found in the coarse mode.

Fine Particles. In contrast to coarse particles, secondary particles, produced by atmospheric transformation processes from precursor gases, dominate fine or accumulation mode particles (in the range of about 0.1 to 2.5 µm in diameter). Particles in this size range generally are not emitted directly, but rather enter the atmosphere as ultrafine particles or gases that grow by coagulation or condensation and "accumulate" to this size range. Particles in this size range generally do not grow into coarse particles.

Ultrafine Particles. $PM_{2.5}$ can be further divided into ultrafine particles (particles below approximately 0.1 µm in diameter) and nanoparticles (particles characterized

1. The National Ambient Air Quality Standards for PM are defined based on sampling cut points: $PM_{2.5}$ and PM_{10}. $PM_{2.5}$ denotes particles equal to or less than 2.5 µm in diameter. PM_{10} particles are those with diameters equal to or less than 10 µm. The basis for these cut points becomes apparent from the bi-modal distribution of the different particles sizes presented in Figure 3.1.

Fig. 3.1 Particulate Matter Size Distribution in Urban Areas



Source: UCLA Institute of the Environment, 2001

Fig. 3.2 Particulate Matter Size Categories in Persepctive

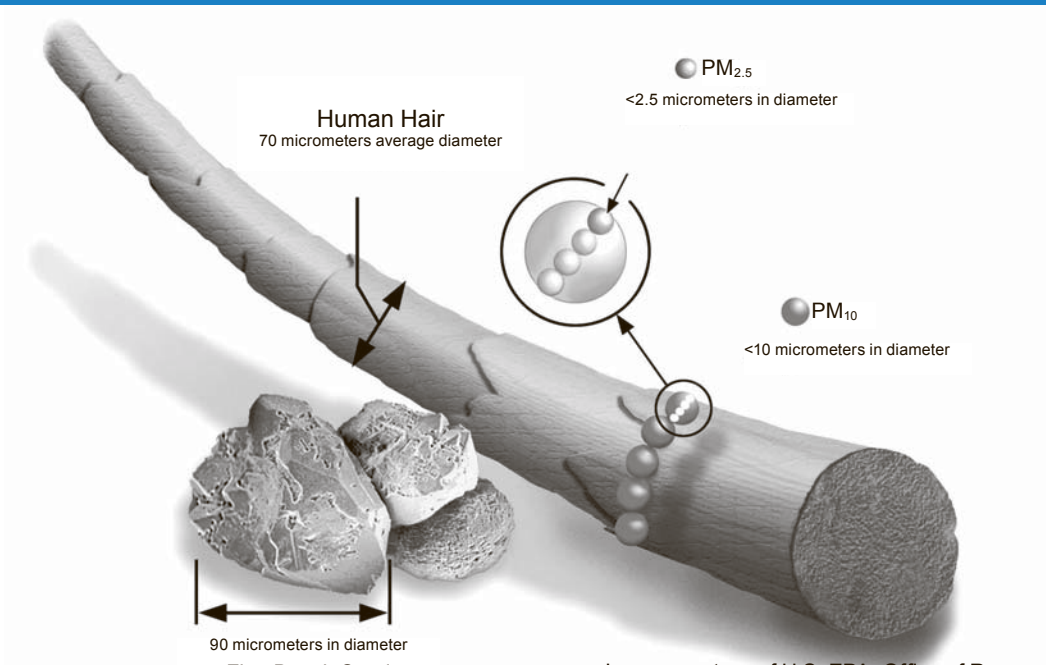


Image courtesy of U.S. EPA, Office of Research and Development

by diameters of less than 50 nanometers or 0.05 µm). Ultrafine particles consist largely of primary combustion products, which undergo reactions in the atmosphere to form larger particles (fine or accumulation mode particles). Some of the formation processes that contribute to the growth of $PM_{2.5}$ in the atmosphere include: (1) nucleation (i.e., gas molecules coming together to form a new particle), (2) condensation of gases onto existing particles, (3) coagulation of particles (i.e., the weak bonding of two or more particles into one larger particle), (4) uptake of water by hygroscopic components and (5) gas phase reactions that form secondary PM. These secondary formation processes can result in new particles or the addition of PM to pre-existing particles. Ultrafine particles have a very short atmospheric life, on the order of minutes to hours, because they readily convert to larger fine or accumulation mode particles.

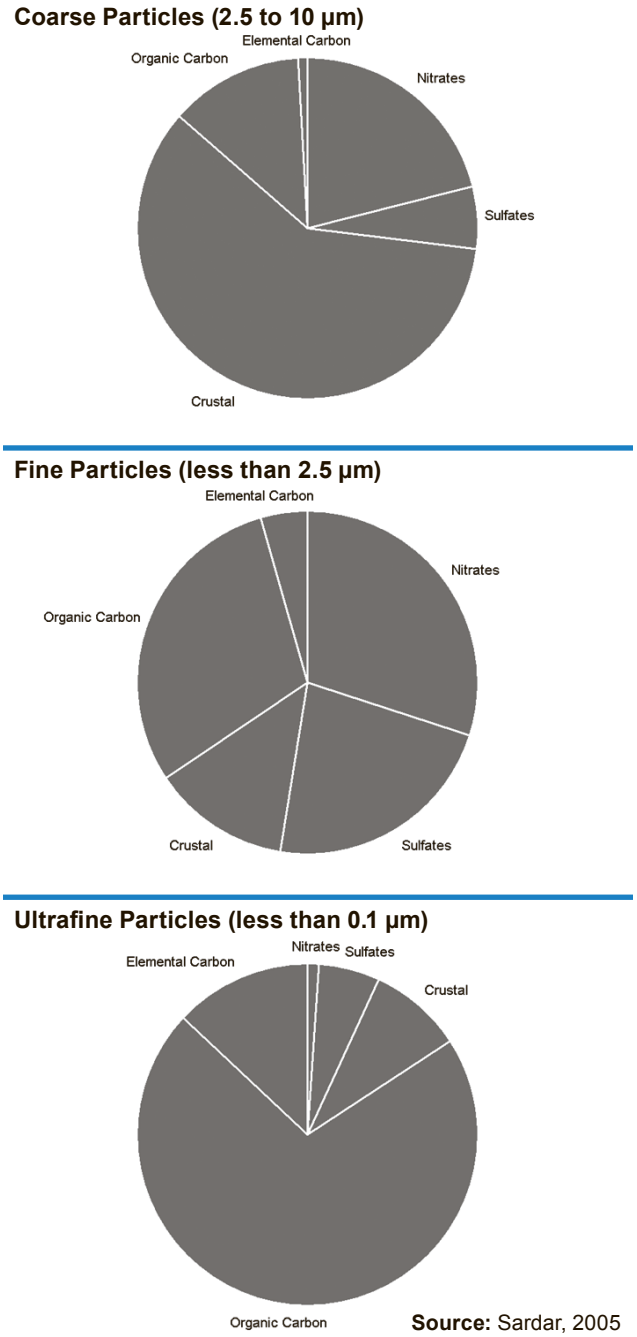
Chemical Composition

The common chemical constituents of PM include sulfates, nitrates, ammonium, elemental carbon, a variety of organic compounds, water and crustal material (including metals, dust, sea salt and other trace elements). Sulfates, ammonium, elemental carbon and secondary organic compounds are found primarily in the $PM_{2.5}$ range. Crustal material—including calcium, aluminum and silicon—is found primarily in the coarse particle range (larger than 2.5 µm). Nitrates are found in both the $PM_{2.5}$ and coarse particle size ranges.

Figure 3.3 describes the chemical composition of PM air pollution measured at a "supersite" location in the Los Angeles metropolitan area, comparing ultrafine, fine and coarse particles (Sardar, 2005). The data show the large crustal contribution to the coarse mode, contrasted with the large contribution of sulfates and nitrates to the fine mode.

Particles, or particle-bound water, can also act as carriers of toxic agents, such as metals and organic compounds (EPA, 2005a).

Fig. 3.3 Supersites Data Los Angeles, CA



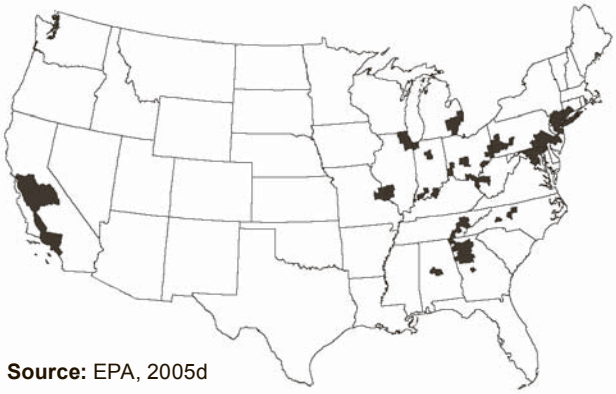
Source: Sardar, 2005

Fate and Transport

The fate and transport of particles in the atmosphere depend in part on their size. Ultrafine particles have a very short ambient residence time—on the order of minutes to hours—because they are likely to undergo gas-to-particle conversions. (These conversions mean they remain in the atmosphere, but in a different form.) Ultrafine particles are also small enough to be removed by diffusion to falling raindrops.

Secondary $PM_{2.5}$, formed from ultrafine particles and gases remains suspended in the atmosphere longer because it is too large to diffuse rapidly to surfaces or to other particles, but it is also too small to settle out or collide with stationary objects. These particles can be transported thousands of miles and remain in the atmosphere for days to weeks. Secondary particles serve as condensation nuclei for cloud droplet formation and are eventually removed from the atmosphere in falling raindrops. Gravity and collisions with surfaces eventually remove secondary particles that are not involved in cloud processes from the atmosphere.

Fig. 3.4 $PM_{2.5}$ Nonattainment Areas



Source: EPA, 2005d

In contrast, coarse particles can settle out from the atmosphere rapidly. As a result, they may be airborne from minutes to days depending on their size, atmospheric conditions and altitude. Larger coarse particles are not readily transported over long distances because they are generally too large to be carried by air streams and because they tend to be easily removed by gravity or impact, or are washed out of the atmosphere by rain. Smaller coarse particles can have longer atmospheric lifetimes and travel longer distances, especially in extreme circumstances. For example, dust storms in desert areas of Africa and Asia lift coarse particles to high elevations; the resulting dust clouds can be transported as far as North America.

Emissions Trends and Regional Characteristics

EPA and the states have established a network of air quality monitors to collect and measure PM_{10} and $PM_{2.5}$ concentrations. The network has been recording PM_{10} concentrations since 1987 and $PM_{2.5}$ concentrations since 1999. The $PM_{2.5}$ monitoring network includes roughly 1,000 monitors, over 90 percent of which are located in urban areas.

EPA has measured a 10 percent decline in national annual average $PM_{2.5}$ concentrations from 1999 to 2003 (EPA, 2004). The Northeast is the only region that did not show a decline between these years: annual concentrations in the region rose slightly (about 1 percent) over the five-year period. Outside of the Northeast, $PM_{2.5}$ concentrations have trended lower during this period, with the largest improvement occurring in those regions that started with the highest concentrations. For example, average annual concentrations declined by 20 percent in the Southeast, 16 percent in southern California and 9 percent in the industrial Midwest. By comparison, the upper Midwest, the Southwest and the Northwest—all of which had lower concentrations to begin with as compared to other regions of the country—posted more modest declines in $PM_{2.5}$ concentrations.

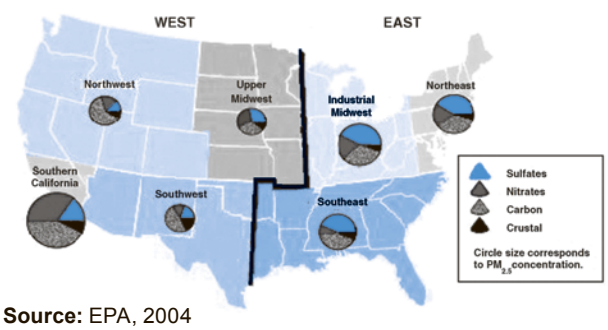
Longer-term trends data are also available, although fewer monitors were in place prior to 1999. According to EPA, $PM_{2.5}$ concentrations have declined on average by about 30 percent over the past 25 years (EPA, 2004).

Recent monitoring data show the highest average $PM_{2.5}$ concentrations in southern California and Pittsburgh, Pennsylvania. Elevated levels have also been recorded in urban areas throughout the Southeast, Northeast and industrial Midwest. Figure 3.4 shows the 39 areas that EPA has designated as exceeding the existing national air quality standard for $PM_{2.5}$ based on air quality monitoring data from 2002 through 2004.

$PM_{2.5}$ concentrations exhibit seasonal variability. For example, $PM_{2.5}$ values in the eastern half of the U.S. are typically higher from July to September, when sulfates are more readily formed from sulfur dioxide (SO_2) emissions. By contrast, $PM_{2.5}$ concentrations tend to be higher in the winter months in many areas of the West—in part because fine particle nitrates are more readily formed in cooler weather, but also because the use of wood stoves and fireplaces produces more carbon.

The composition of $PM_{2.5}$ varies geographically as well as seasonally. $PM_{2.5}$ in the eastern U.S. is dominated by sulfates and carbon. $PM_{2.5}$ in southern California contains more nitrates than in other areas of the country. Carbon is a substantial component of $PM_{2.5}$ concentrations in all regions of the country. Figure 3.5 presents the average composition of urban $PM_{2.5}$ by region.

Fig. 3.5 Urban $PM_{2.5}$ Composition by Region



Source: EPA, 2004

There are no nationwide monitoring networks for ultrafine particles, although some studies have been conducted under the direction of EPA's PM Supersites Program. Measurements taken at a Los Angeles monitoring station show that ultrafine particles make up a small portion of the PM concentration by mass. However, the number of ultrafine particles is significantly larger than the number of coarse or fine particles. Studies have also found that ultrafine PM concentrations can be dramatically elevated

in close proximity to high vehicle traffic areas and during busy traffic periods (EPA, 2005a).

Emissions Sources

The sources of $PM_{2.5}$ and $PM_{2.5}$-precursor emissions are highly diverse, including both natural (biogenic) and human-made (anthropogenic) sources. Sources include motor vehicles, power plants, industrial facilities, wood stoves and fireplaces, forest fires, sea salt, paved and unpaved roads and many others.

Additionally, the contribution of various sources to $PM_{2.5}$ formation varies by geography, time of year and even by time of day. As a result, national and regional inventories of $PM_{2.5}$ and $PM_{2.5}$-precursor emissions—although suggestive of the primary contributors to ambient particulate concentrations—may not provide an accurate characterization of the major sources of emissions at a specific location or during a particular time period.

The relationship between changes in precursor emissions and ambient $PM_{2.5}$ concentrations, moreover, can be nonlinear. Generally, SO_2 emissions reductions lead to reductions in concentrations of sulfate aerosols and nitrogen oxide (NO_x) emissions reductions lead to reductions in nitrate aerosols. However, the direction and extent of changes in ambient $PM_{2.5}$ concentrations as a result of a given level of emissions reduction vary by location and season and depend on fluctuations in ammonia emissions and changes in prevailing meteorology and photochemistry. This complicates the task for state and local officials attempting to prioritize their $PM_{2.5}$ control strategies.

With these cautionary statements in mind, we summarize the major sources of $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from the sources addressed in this report, based on EPA's National Emissions Inventory. A combination of databases (1999 and 2002) have been used to compile this summary, as indicated in Table 3.1.

Table 3.1, at the end of this chapter, presents the total primary $PM_{2.5}$ emissions from all of the source categories addressed in this report. This table reports total primary $PM_{2.5}$ emissions of 5,522,000 tons. Fugitive dust was the largest source of $PM_{2.5}$ emissions, contributing 3,326,000 tons (60 percent).

Table 3.1 also sets out estimates of total SO_2 emissions from the source categories addressed in this report. As shown, emissions total 13,126,000 tons, with electric generating facilities responsible for 10,293,000 tons (78 percent), and industrial point sources contributing another 1,941,000 tons (15 percent).

Finally, Table 3.1 contains estimates of total NO_x emissions from the source categories addressed in this report. Emissions total 19,090,000 tons, with electric utilities contributing 4,700,000 tons (25 percent), highway vehicles contributing 8,167,000 tons (43 percent), and nonroad equipment (including marine vessels and airports) contributing 4,387,000 tons (23 percent).

This report does not present VOC emissions, although a subset of VOCs contribute to ambient $PM_{2.5}$ concentrations. Also, ammonia from sources such as fertilizer and animal feed operations contributes to the formation of sulfates and nitrates that exist in the atmosphere as ammonium sulfate and ammonium nitrate. This report also does not cover ammonia emissions.

Uncertainties in the $PM_{2.5}$ Inventory

The accuracy of the emissions estimates presented in this chapter and throughout this report vary. In some cases, we have a high degree of confidence in the estimates. For example, most power plants continuously monitor and report SO_2 emissions to EPA. In contrast, the estimates of direct $PM_{2.5}$ emissions are less certain and likely to be changing in the coming years with further information. There are several reasons for the uncertainties.

First, there are limited measurements of $PM_{2.5}$ emissions from stationary sources from which to estimate source-specific emissions factors. Many of the emissions stack tests that are the basis of currently available particle size distributions were conducted with methods that have since been shown to introduce errors.

Second, EPA Method 5, the most common test method used for measuring PM emissions from stationary sources, measures total filterable particulates (TFP), not total PM_{10} or total $PM_{2.5}$ specifically (EPA, 2005c). In most cases, sources estimate their $PM_{2.5}$ emissions based on these measurements, rather than specifically testing for the fine fraction.

Third, researchers have suggested that sampling methods may be overstating the $PM_{2.5}$ emissions rates of fugitive dust sources, such as agricultural operations. According to a recent study, the high volume cyclone/impactor system used to develop AP-42 emissions factors for fugitive dust sources has a positive bias (i.e., overestimates emissions) (Comis, 2004; WRAP, 2005).

Fourth, there are important distinctions among filterable $PM_{2.5}$, condensable $PM_{2.5}$ and total $PM_{2.5}$. Filterable $PM_{2.5}$ refers to those particles that are captured on the filter portion of a sampling train. Condensable particles are those that are in gaseous form in the stack but form solid or liquid particles soon after being released to the

atmosphere. Total $PM_{2.5}$ includes both filterable and condensable particles.

There are several different methods (and variations of methods) used to measure PM emissions. In some cases, the test results reflect only the filterable components in the flue gas. Additionally, the filterable components vary depending on the test method used. In particular, filter temperatures influence both the concentration of organic and inorganic vapors that can exist in a gas and also influence chemical reactions that occur in the air sample. At high temperatures, semi-volatile organic and inorganic compounds remain in the vapor phase (passing through the filter). At lower temperatures, these chemicals condense to become filterable PM. Additionally, at high temperatures, reactions between gases like ammonia and SO_2 will be inhibited, so that they remain a gas and pass through the filter. At lower temperatures, these chemicals react, forming particulate-phase chemical compounds such as ammonium sulfate. For example, EPA Test Method 5B uses a high temperature filter (320 ± 25ºF) when collecting a test sample. The sample is then heated in an oven at 320ºF (160ºC) for six hours to volatilize any sulfuric acid that may have condensed on the filter.

There are many variations of EPA Method 5, which are intended to measure filterable PM. Several other EPA test methods are also intended to measure only filterable PM. EPA Method 202, on the other hand, is designed to measure condensable PM, by replicating the cooler temperatures that the flue gas encounters upon exiting a facility stack. Test Method 202 is intended for use in conjunction with a filterable PM test method to capture total PM emissions (EPA, 2005b).

As discussed previously, many emissions inventories do not use the results of site-specific source testing as the basis of their emissions inventory. Much of the emissions inventory is based upon the application of emissions factors. The development of emissions factors is based upon source testing that incorporates many of the underlying uncertainties inherent in the selection of test methods. And these are not the only factors contributing to the uncertainties in both the development and application of emissions factors (for additional information, see the introduction to AP-42 (EPA, 1995)).

The uncertainties related to test method selection are illustrated in a recent advanced announcement of an update of the PM emissions factors for natural gas combustion. EPA has announced that it will revise the estimates of $PM_{2.5}$ emissions from natural gas combustion in the final version of the 2002 National Emissions Inventory (NEI). The revision will decrease the estimated emissions of $PM_{2.5}$ by roughly 95 percent. The Electric Power Research Institute (EPRI) also plans to investigate the available test methods and may suggest some alternative test methods (EPRI, 2005).

It is not yet clear how new test results will affect the overall inventory. Sources for which emissions estimates for condensable PM are not available or are under-estimated may find that the new inventory figures are significantly higher than previous inventory numbers. Other sources, in particular those that have based their estimates on Test Method 202 data (excluding nitrogen purging), may find that their inventory numbers are somewhat lower than under the current inventory (EPA, 2005b).

Lastly, as we have discussed in the *Introduction*, the distinction between filterable and condensable PM is not just an inventory concern; it also raises regulatory and permitting issues.

$PM_{2.5}$ Source Apportionment

As indicated above, emissions of $PM_{2.5}$ and $PM_{2.5}$-precursors come from a variety of sources and vary by location and time. This section discusses several studies that have examined the composition of $PM_{2.5}$ in different geographic areas.

The importance of determining source apportionment for ambient $PM_{2.5}$ in a specific area cannot be overstated; developing a cost-effective approach to controlling $PM_{2.5}$ emissions sources requires an understanding of the relative contribution from local and regional sources. Adequate monitoring data are needed to provide insight into the composition of ambient $PM_{2.5}$ in a given area.

EPA began an effort to deploy over 1,000 $PM_{2.5}$ monitors nationally in 1997. A handful of computer models are available to determine the source apportionment of $PM_{2.5}$ mass in air samples, including the UNMIX and Positive Matrix Factorization (PMF) models. The UNMIX model apportions the PM chemical mass to specific source categories based on an assessment of periods when individual sources do not contribute significantly to the total mass of any species being modeled. The PMF model apportions the PM chemical mass to specific source categories using a least squares modeling technique.

Source apportionment studies have been performed in Phoenix, Denver and various other cities. In Phoenix, the aerosol chemical composition of $PM_{2.5}$ was analyzed using the PMF method (Ramadan, 2000). The results suggest that motor vehicles, vegetation burning and coal-fired power plants are dominant contributors to the city's $PM_{2.5}$ mass. The highest concentrations occurred in the winter and thus corresponded with the peak tourist season and peak fireplace use. A separate study of $PM_{2.5}$ concentrations in Phoenix, using the UNMIX model, arrived at similar results using the same data set (Lewis,

2003). The study found that gasoline engines were the largest overall contributor (33 percent), and that their contribution was higher during the winter months (42 percent). During the summer, the contribution from gasoline engines diminished to 20 percent and was surpassed by the contribution from secondary sulfates (29 percent) and crustal/soil (26 percent).

The different methodologies used in these studies resulted in different conclusions as to the relative contribution of gasoline versus diesel engines: the UNMIX method estimated a higher contribution of diesel in the total vehicle contribution. Other studies have detected similar discrepancies in the contribution assigned to gasoline and diesel use. It will be important to resolve these discrepancies in order to ensure greater confidence in the emissions inventories and source apportionment studies used to tailor control strategies.

In Denver, a tracer method was used to estimate source contributions to fine aerosol concentrations during the winter months, when air quality in Denver is the worst (Lewis, 1986). This study identified motor vehicles (42 percent), electric power generation (23 percent) and wood burning (12 percent) as the most significant contributors to the local $PM_{2.5}$ mix. A more recent study conducted by the Desert Research Institute and Colorado State University evaluated monitoring data and also found that most of the $PM_{2.5}$ in the area came from gasoline vehicles and engines (Lawson, 1998). Specifically, the study found that gasoline vehicles and engines were the largest contributor to $PM_{2.5}$ mass during winter haze episodes (28 percent), followed by dust and debris (16 percent), diesel vehicles and engines (10 percent), wood smoke (5 percent), meat cooking (4 percent), and directly emitted $PM_{2.5}$ from coal-fired power plants (2 percent). Particulate ammonium nitrate and particulate ammonium sulfate, both formed in the atmosphere from a variety of sources, together contributed 35 percent of ambient $PM_{2.5}$.

Numerous other source apportionment studies have been conducted across the U.S. For a discussion of the available literature and various source apportionment methodologies, see EPA's 2003 report entitled *Compilation of Existing Studies on Source Apportionment for $PM_{2.5}$* (EPA, 2003).

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Table 3.1 National Emissions by Source Category (thousand tons)

Source Category		$PM_{2.5}$	SO_2	NO_x	Source
Industrial Boilers					
Pulp and Paper Sector		40	341	210	1999 NEI
Petroleum Refineries		9	103	70	1999 NEI
Iron and Steel Sector		4	32	33	1999 NEI
Other Sector		97	689	478	1999 NEI
Commercial Boilers		16	111	73	1999 NEI
	Total	166	1,276	863	
Electric Generating Units					
Coal		535	9,738	4,097	2002 NEI
Oil		13	343	130	2002 NEI
Gas		11	8	270	2002 NEI
Other		8	191	50	2002 NEI
Internal Combustion		14	13	152	2002 NEI
	Total	582	10,293	4,700	
Industrial Point Source					
Pulp and Paper Sector (except boilers)		59	109	92	1999 NEI
Cement		23	184	234	1999 NEI
Iron and Steel (except boilers)		41	58	65	1999 NEI
Petroleum Refineries (except boilers)		27	314	173	1999 NEI
	Total	150	665	564	
Onroad Vehicles					
Light-Duty Gasoline		45	163	3,797	2002 NEI
Light-Duty Diesel		2	1	21	2002 NEI
Motorcycles[a]		0.3	0.3	19	2002 NEI
Heavy-Duty Gasoline[a]		7	13	497	2002 NEI
Heavy-Duty Diesel		95	81	3,832	2002 NEI
	Total	149	259	8,167	
Nonroad Equipment					
Agricultural Diesel		60	54	568	2002 NEI
Construction and Mining Diesel		60	84	816	2002 NEI
Industrial Diesel		10	15	133	2002 NEI
Commercial Diesel		8	8	77	2002 NEI
Logging Diesel		1	3	24	2002 NEI
Rail Diesel		23	56	1,075	2002 NEI
Other nonroad[b]		37	11	469	2002 NEI
	Total	199	231	3,161	
Airports					
Aircraft		14	8	86	2002 NEI
Ground Service Equipment		0.9	1	20	2002 NEI
	Total	15	9	106	

Continued

Table 3.1 National Emissions by Source Category (thousand tons)

Source Category		$PM_{2.5}$	SO_2	NO_x	Source
Marine Ports					
Marine Ships Diesel		40	201	1,070	2002 NEI
Marine Ships Other		33	2	49	2002 NEI
	Total	74	203	1,120	
Residential					
Natural Gas		24	2	259	1999 NEI
Oil		6	125	53	1999 NEI
Wood		419	6	45	1999 NEI
Coal		5	52	7	1999 NEI
Open Burning		175	5	33	1999 NEI
	Total	630	189	398	
Fugitive Dust					
Paved Roads		688	0	0	1999 NEI
Unpaved Roads		1,304	0	0	1999 NEI
Agricultural Crops		862	0	0	1999 NEI
Agricultural Livestock		88	0	0	1999 NEI
Construction		385	0	0	1999 NEI
	Total	3,326	0	0	
Other					
Agricultural Burning		152	0.012	11	1999 NEI
Commercial Cooking		79	0	0	2002 NEI
	Total	231	0.012	11	
TOTAL		5,522	13,126	19,090	

a. Motorcycles and heavy-duty gasoline trucks are not a focus of this report.

b. Other nonroad equipment, including lawn and garden equipment, recreational vehicles, and non-diesel agricultural, commercial, industrial, construction, logging and mining equipment, are not a focus of this report.

Source: EPA, 2005e (2002 NEI), 2005f (1999 NEI)

Chapter 4

The Clean Air Act

Introduction

Every chapter of this report that addresses specific source categories should be read through the lens of the Clean Air Act. For example, where EPA is required by the Act to establish regulations, state and local agencies are not permitted to set standards less stringent than those imposed by EPA; in addition, the Act places some restrictions on the regulation by states of certain mobile sources. Thus, it will be important for state and local regulators, in assessing what degree of regulatory freedom they have to address emissions from various sources, to understand how the Clean Air Act affects the specific source category or facility under consideration. We provide more detail in individual chapters of this report on applicable federal statutes as they pertain to specific source categories—this chapter is intended as an overview.

The federal Clean Air Act is a long and complicated piece of legislation. It is easier to understand if broken into segments, although there are a number of different ways it can be sliced.

One way to understand the Clean Air Act is to examine the distinctions it makes among broad source categories. The Act treats "stationary" sources (large, non-moving producers of pollution such as refineries and power plants) and "mobile" sources (such as cars, trucks and buses) differently. In addition, it covers "area" sources: facilities that individually are too small to qualify as stationary sources, but that are significant polluters in the aggregate (e.g., fugitive dust, residential wood stoves).

The Clean Air Act also deals differently with different types of pollutants. So-called "criteria" pollutants—such as ground-level ozone, which is regulated mainly by controlling emissions of nitrogen oxides (NO_x) and volatile organic compounds (VOCs); nitrogen dioxide; carbon monoxide (CO); particulate matter (PM); sulfur dioxide (SO_2); and lead—are regulated separately from "hazardous" air pollutants or HAPs (also known as "air toxics"), such as arsenic, mercury, and a host of others.

Furthermore, the Act divides responsibility for various types of sources and air pollutants between the states and localities on the one hand, and the federal government on the other. Generally, the states and EPA share responsibility for regulating criteria pollutants from stationary and area sources, but states and localities have the lead role in addressing emissions from these source categories. Section 116 of the Act gives states and local agencies the freedom to adopt more stringent standards for criteria pollutant emissions from stationary and area sources than the Clean Air Act requires. As discussed in the Introduction to this report, however, the reality is somewhat different. According to a STAPPA and ALAPCO survey, 26 states are completely or partially precluded by state law or policy from adopting standards that are more stringent than federal law requires and a number of other states report that they face obstacles in

implementing more stringent programs. However, 46 states report that they can adopt requirements if no federal programs or standards exist (STAPPA and ALAPCO, 2002).

The federal government bears most of the burden of regulating air toxics and mobile sources, although the regulation of mobile sources involves a uniquely pragmatic division of responsibility. The Clean Air Act allows California to set new vehicle emissions standards that are stricter than federal standards. This is because California has historically had more severe vehicle-related air pollution problems than the rest of the country and because it already had a history of regulating mobile sources at the time the relevant provisions of the Clean Air Act were enacted. Other states can adopt the California standards (as some have done) for new vehicles, instead of following the federal standards. This ensures that auto manufacturers will have to meet at most two standards for a particular class of vehicles. States are not preempted from imposing their own standards on *existing onroad* vehicles. We discuss these issues at greater length in the section of this chapter on mobile sources.

The National Ambient Air Quality Standards

Under §108 of the Clean Air Act, EPA must set National Ambient Air Quality Standards, or NAAQS, for air pollutants that in the Agency's judgment pose serious problems for public health or welfare. For these pollutants, the Act directs EPA to set "primary" standards to protect public health with an adequate margin of safety (§109(b)(1)), and "secondary" standards to protect public welfare with respect to issues such as visibility and materials damage (§109(b)(2)). Table 4.1 shows the NAAQS for all the criteria pollutants. The Clean Air Act requires that these ambient air quality standards be set without regard to the cost of compliance (American Trucking Associations, Inc. v. Whitman, 2001) or the feasibility of controlling relevant emissions.

Table 4.1

National Ambient Air Quality Standards

Pollutant	Primary Standards	Averaging Times	Secondary Standards
Carbon Monoxide	9 ppm (10 mg/m^3)	8-hour	none
	35 ppm (40 mg/m^3)	1-hour	none
Lead	1.5 µg/m^3	quarterly average	same as primary
Nitrogen Dioxide	0.053 ppm (100 µg/m^3)	annual (arithmetic mean)	same as primary
Particulate Matter (PM_{10})	50 µg/m^3	annual[a] (arithmetic mean)	same as primary
	150 µg/m^3	24-hour[b]	
Particulate Matter ($PM_{2.5}$)	15.0 µg/m^3	annual[c] (arithmetic mean)	same as primary
	65 µg/m^3	24-hour[d]	
Ozone	0.08 ppm	8-hour[e]	same as primary
Sulfur Oxides	0.03 ppm	annual (arithmetic mean)	NA
	0.14 ppm	24-hour[b]	NA
	NA	3-hour[b]	0.5 ppm (1300 µg/m^3)

a. To attain this standard, the 3-year average of the weighted annual mean PM_{10} concentrations at each monitor within an area must not exceed 50 µg/m^3.

b. Not to be exceeded more than once per year.

c. To attain this standard, the 3-year average of the weighted annual mean $PM_{2.5}$ concentrations from single or multiple community-oriented monitors must not exceed 15.0 µg/m^3.

d. To attain this standard, the 3-year average of the 98th percentile of 24-hour concentrations at each population-oriented monitor within an area must not exceed 65 µg/m^3.

e. To attain this standard, the 3-year average of the fourth-highest daily maximum 8-hour average ozone concentrations measured at each monitor within an area over each year must not exceed 0.08 ppm.

Source: EPA, 2005f

Geographic areas within states are classified in terms of where they stand in achieving the NAAQS for each criteria pollutant. Areas are designated as "nonattainment" (they fail to meet the standards), "attainment" (they meet the standards), or "unclassifiable" (not enough information to tell). The classification scheme has additional levels of complexity. For example, under §188 of the Act, areas that are in nonattainment for PM_{10} are classified as either moderate or serious. As the nonattainment classification goes from moderate to serious, more stringent pollution control requirements apply, but state and local authorities are given more time to demonstrate attainment.

EPA promulgated the NAAQS for $PM_{2.5}$ in 1997. In April 2005, the Agency finalized its determination that 208 counties comprising 39 metropolitan areas (now 36, based on reclassifications) were in nonattainment of the standard. Most of these nonattainment areas are east of the Mississippi River; however, three are in the West—two in California and one in Libby, Montana. The deadline for attaining the $PM_{2.5}$ NAAQS is April 2010, but states must submit State Implementation Plans (SIPs) by April 2008. Under §172(a)(2) of the Clean Air Act, EPA can grant a five-year extension for areas with more severe problems.

Meanwhile, the $PM_{2.5}$ NAAQS are again undergoing the periodic Agency review that is required at five-year intervals under §109(d)(1) of the Clean Air Act. Under the terms of a consent decree, EPA issued its proposed $PM_{2.5}$ standards (as well as proposed standards for larger particulates) on January 17, 2006, and must issue final standard by September 27, 2006.

EPA estimates that meeting the current $PM_{2.5}$ standards would avoid tens of thousands of premature deaths annually and save hundreds of thousands more people from significant respiratory or cardiovascular disease. EPA further estimates that—depending on the particular emissions controls required for sources of $PM_{2.5}$-precursor emissions and depending on the locations of those sources—the monetized health benefits of reducing pollutant emissions that lead to $PM_{2.5}$ formation exceed the costs by three to more than 30 times (EPA, 2005d).

State Implementation Plans

Section 110 of the Clean Air Act requires states to write SIPs indicating how they will achieve the NAAQS for each pollutant. States have the lead in controlling air pollution, with EPA stepping in only if a state fails to submit a SIP, submit an adequate SIP, or to implement the SIP. Both the NAAQS and the SIPs are applicable to stationary, area and mobile sources. The stringency of the air pollution controls that states have to impose in their Plans depends on whether the area in question is classified as nonattainment or attainment and—in the case of nonattainment areas—on the degree of nonattainment.

SIPs must also contain provisions for limiting "transport," meaning that states must prohibit the sources within their jurisdiction from sending so much air pollution downwind that areas in *other* states are unable to meet or maintain federal air quality standards (§110(a)(2)(D)).

SIPs are adopted as state regulations and, once approved by EPA, become enforceable as federal regulations. However, §179 of the Act provides that if a state fails to submit a SIP, or submits a SIP that is insufficient, EPA can impose sanctions (such as a loss of federal highway subsidies), or write its own plan for the area in question (known as a Federal Implementation Plan, or FIP).

Stationary and Area Sources

As discussed above, the Clean Air Act addresses criteria pollutants through the NAAQS, which are implemented by the federal government and by state and local agencies through a complicated scheme of shared authority in which the states take the lead. However, several other parts of the Act govern emissions of both criteria pollutants and air toxics, adding layers of requirements to the core NAAQS requirements.

Hazardous Air Pollutants or Air Toxics

Section 112 of the Clean Air Act regulates HAPs, or air toxics, differently from criteria pollutants. The Act specifies 188 HAPs—chemicals that are known to cause, or are suspected of causing, adverse human health or environmental effects. It directs EPA to identify the categories of sources that emit these pollutants and then to develop technology-based national emissions standards, *applicable both to new and existing sources*, for each of the categories.[1] These Maximum Achievable Control Technology (MACT) standards apply to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs.

Section 112(d) requires that MACT standards be based on the maximum reduction of emissions possible through the application of control technologies or work practices. The so-called "MACT floor," or baseline level at which EPA sets the MACT standards for a source category, is determined without regard to the cost of compliance. The MACT floor for existing sources may not be less stringent than "the average emission limitation achieved by the best performing 12 percent of the existing sources." The MACT floor for new sources may not be "less stringent than the emissions control that is achieved in practice by the best controlled similar source." EPA can set standards that are even more stringent than the floor, but if it does

1. For the purposes of §112, "new" sources include those that have been "reconstructed" (i.e., where the cost of reconstruction exceeds 50 percent of the cost of constructing a new source); "existing" sources include those that have been "modified" (EPA, 2004).

so it must take costs into account. Although MACT rules address air toxics, in some instances the rules achieve reductions in PM emissions as a co-benefit. In some MACT rules, for example, EPA regulates PM (but not the fine fraction separately) as a surrogate for the regulation of metals.

Section 112(f) of the Act requires that eight years after promulgating technology-based MACT standards for a source category, EPA must evaluate the "residual" risk, if any, that remains and promulgate regulations to address that risk.

Note that the MACT regulations (as opposed to the later assessment of residual risk that is supposed to follow) reflect an entirely different approach from that embodied in the NAAQS approach to criteria air pollutants. The NAAQS for criteria pollutants are based on a determination about the level of air quality needed to protect public health and the environment, rather than on the technologies or work practices that are available for reducing emissions.

Additional Regulatory Programs

Previous sections of this chapter suggest two ways to understand the Clean Air Act: first, through its classification of emissions sources into stationary, area and mobile source categories and second, with reference to the different treatment of criteria pollutants and HAPs. Another way to understand the Act is by reviewing some of the most significant additional regulatory programs it creates for controlling emissions from stationary and area sources.

RACM/RACT/BACM. Section 172(c)(1) of the Clean Air Act requires nonattainment SIPs to provide for the implementation of all Reasonably Available Control Measures (RACM), which include Reasonably Available Control Technology (RACT). These requirements are *applicable to both new and existing sources*, although—as a practical matter—other, much more stringent requirements dictate the standards for new sources.

Although not specifically defined by the Clean Air Act, EPA has defined RACT as the lowest emissions limitation that a source can meet using control technology that is reasonably available, taking into consideration technological and economic feasibility (EPA, 2005d). For "serious" PM_{10} nonattainment areas, §189(b) of the Clean Air Act requires BACM, or best available control measures.

Federal law does not establish actual RACM, RACT or BACM limits—it only provides guidelines pursuant to which the states adopt regulations. Most of these guidelines are known as Control Technique Guidelines (CTGs, pursuant to §108(b) of the Act) and most address emissions of VOCs. EPA has not issued any new sector-specific CTGs in 25 years. For NO_x, the federal guidelines are referred to as Alternative Control Techniques (ACTs, pursuant to §183(c)). The ACT guidelines describe the control technologies that are available to address NO_x emissions, but do not actually specify the emissions limits that EPA considers to constitute RACT. There are no CTGs for NO_x, and no sector-specific CTGs for SO_2, PM_{10} or $PM_{2.5}$.

RACT limits are significant because they apply not just to new, but also to *existing* sources. However, many ACTs were promulgated a decade or more ago and they have been eclipsed by subsequent technology developments. Also, as was already noted, they are only intended to provide guidance, on which—as EPA acknowledges—significant variation may be appropriate. Like New Source Review (NSR) standards, RACT standards—which are informed by the ACTs—represent moving targets that are intended to reflect advancements in technology.

NSR/PSD. NSR is the Clean Air Act preconstruction permitting program that applies to major stationary sources of air pollution when they are first built or significantly modified. NSR requires—among other things—that sources install updated environmental controls. The term "NSR" is often used to refer to two distinct programs: nonattainment NSR, which operates in areas that are in nonattainment with one or more of the NAAQS (Secs. 171–193), and the Prevention of Significant Deterioration (PSD) program, which operates in attainment and unclassifiable areas (Secs.160–169B).

States also have their own state preconstruction permitting programs that may apply to sources or activities other than those covered by the federal programs. These programs vary widely (Hawkins, 2004).

NSR requires new and modified plants located in attainment areas to implement Best Available Control Technology (BACT); in nonattainment areas, the standard is the Lowest Achievable Emission Rate or LAER (Secs. 165(a)(4) and 173(a)(2), respectively). The key difference between BACT and LAER standards is that BACT takes cost into account while LAER does not (Hawkins, 2004). Unlike New Source Performance Standards (discussed below) and MACT standards for HAPs, BACT and LAER standards are not codified. Instead, they are moving targets, intended to reflect technology advancements as they occur. Current RACT, BACT and LAER guidelines are theoretically listed in EPA's RACT/BACT/LAER Clearinghouse which, unfortunately, is not kept reliably up-to-date.

New plants in nonattainment areas are subject to a number of other important requirements, including—for

ozone—the requirement that new plants provide emissions reductions that would not otherwise be mandated to "offset" any additional air pollution they cause (§173). Offsets are meant to guarantee that the construction of a new source does not increase total pollution emissions in the area where the source is located. The required offset ratio is dependent upon the degree of nonattainment of the area.

NSPS. Unlike NSR, the New Source Performance Standards (NSPS) provided for under §111 of the Clean Air Act apply without regard to an area's attainment status. Like MACT standards, but unlike NSR and RACT standards, the NSPS are codified and do not reflect real-time technology improvements. Although it is relatively unusual, in some circumstances the NSPS apply to *existing* sources. In these instances, §111(d) of the Act requires EPA to establish "emissions guidelines" rather than regulatory limits.

More typically, however, the NSPS apply to new and reconstructed, rather than existing, sources.[2] As with the MACT standards, reconstruction is defined as replacing components such that the cost exceeds 50 percent of the cost of constructing a new source (EPA, 1975).

Under §111, EPA must develop a list of source categories subject to the NSPS. The standards themselves are technology based and are required to reflect "the best system of emissions reduction which…the Administrator determines has been adequately demonstrated." In establishing the NSPS, EPA is required to take cost into consideration.

NSPS requirements are usually less stringent than the standards imposed pursuant to NSR because the NSPS are fixed standards, whereas NSR requirements are decided on a case-by-case basis and are intended to reflect recent technology developments. Also, in nonattainment areas a modification is more likely to trigger NSR because the trigger for NSPS requirements involves an increase in the emissions *rate*, while EPA has historically argued that only an increase in total tons of pollution is required to trigger NSR. Note, however, that EPA has proposed to eliminate this distinction may be eliminated by virtue of rule changes to the NSR program that the Agency proposed in October 2005 (EPA, 2005e).

Regional Haze/BART. Regional haze refers to the visibility impairment caused by emissions of $PM_{2.5}$ and $PM_{2.5}$-precursors from numerous sources located in a wide geographic area. Section 169A(b)(2) of the Clean Air Act requires certain existing stationary sources in attainment and nonattainment areas to install Best Available Retrofit Technology (BART), where BART is defined as "an emission limitation based on the degree of reduction achievable through the application of the best system of continuous emission reduction for each pollutant emitted" (EPA, 2005a). BART applies to large sources in 26 identified source categories that began operation between 1962 and 1997 (§169A(g)(7)).

Section 169A of the Clean Air Act establishes a national visibility goal that calls for "the prevention of any future, and the remedying of any existing, impairment of visibility in Class I areas, which impairment results from manmade air pollution." The term "Class I area" refers to 156 areas: national parks exceeding 6000 acres, wilderness areas and national memorial parks exceeding 5000 acres, and international parks in existence on August 7, 1977.

Pursuant to §169B of the Act, EPA finalized revisions to its original 1980 haze regulations in July 1999 (EPA, 1999). Unlike the 1980 rulemaking that applied to 35 states, the 1999 rule extends coverage to all states, requiring them to establish goals and emissions reduction strategies for improving visibility in the nation's 156 Class I areas.

The regional haze rule requires states to develop implementation plans that:

- ensure *reasonable progress* toward the national goal of eliminating manmade visibility impairment by 2064;
- contain *provisions for BART*; and
- specify *a long-term strategy* for achieving the national goal.

The initial implementation period for the rule extends to the year 2018, with a reassessment and revision of emissions reduction strategies, as appropriate, every ten years thereafter. State strategies must address all types of sources (stationary, mobile and area) and activities that degrade visibility in downwind Class I areas, regardless of whether the affected Class I areas are located within the state's own boundaries.

After protracted litigation that included the D.C. Circuit Court of Appeals' vacatur of substantial portions of the regional haze rule (*American Corn Growers Association v. EPA*, 2002), EPA finalized amendments to the rule on July 6, 2005 (EPA, 2005a). The amendments include so-called "BART guidelines" for states to use in identifying sources subject to BART and applicable controls, as well as general provisions that address the use of trading programs as a substitute for BART. States have not yet developed their

2. NSPS technically also apply to modifications, which are treated as new sources under this program (unlike §112 MACT standards, which treat modified sources as existing sources). However, for complicated reasons having partly to do with EPA's use of certain Internal Revenue Service definitions, most modifications do not trigger the NSPS. Under the NSPS test, a modification occurs if certain kinds of changes increase the hourly emissions rate of a pollutant (EPA, 2000a).

own BART regulations.

EPA also proposed revisions to the regional haze rule on August 1, 2005 (EPA, 2005b). These revisions contain more detailed provisions for the use of trading programs as a substitute for BART; they also propose special provisions for a group of western states and tribes.

States are currently required to develop their regional haze SIPs by December 2007.

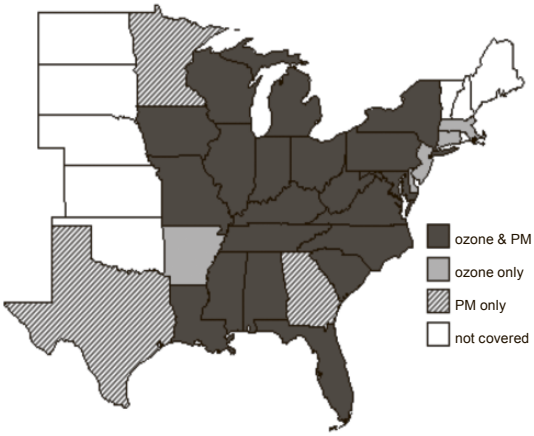
The Acid Rain Program. The Acid Rain Program established by Title IV of the Clean Air Act imposes an additional layer of regulation on electric generating units (EGUs), or power plants, only. The program requires the reduction of NO_x and SO_2 emissions. Title IV imposes a national "cap-and-trade" program—the first of its kind—for EGU emissions of SO_2, but not for NO_x.

A cap-and-trade program sets a cap on total emissions of a particular pollutant from the regulated industry sector and creates authorizations ("allowances") to emit a specified quantity of that pollutant. Allowances are then allocated to sources in the sector, with the total number of allowances equaling the level of the cap. Sources surrender allowances equal to their emissions (e.g., one allowance for each ton of pollutant emitted). This approach gives sources considerable compliance flexibility: they can choose to implement controls or purchase allowances from sources with lower marginal abatement costs. Notwithstanding these provisions, the Clean Air Act prohibits a source from emitting at levels that would violate the applicable SIP, regardless of the number of SO_2 allowances it holds under the Acid Rain Program.

The Clean Air Interstate Rule. On May 12, 2005, EPA finalized the Clean Air Interstate Rule (CAIR) to reduce interstate transport of pollutants under §110(a)(2)(D) of the Clean Air Act (EPA, 2005c). The Agency found that a total of 28 states and the District of Columbia contribute significantly to nonattainment or interfere with maintenance of the NAAQS for $PM_{2.5}$ and/or ozone in downwind states. Like the Acid Rain Program, CAIR requires reductions in emissions of the $PM_{2.5}$-precursors NO_x and SO_2.

CAIR creates three federal cap-and-trade programs: an annual program for SO_2, an annual program for NO_x, and a seasonal program for NO_x. These programs cover EGUs, but states can include in their seasonal NO_x program large industrial boilers (greater than 250 MMBtu/hr) that are included in the NO_x SIP Call's NO_x trading program, so long as they adhere to the cap established under the SIP Call program. These are the only non-EGU sources that can be included in any of the CAIR trading programs without disqualifying the state from participation in the federal trading regimes.

Fig. 4.1 States Affected by CAIR



States have at least four choices for complying with CAIR: (1) participate in the three federal cap-and-trade regimes, which EPA will run; (2) create a cap-and-trade regime within the state for all or some of the pollutants and sources of the state's choice; (3) achieve the required reductions through other methods, such as imposing controls on EGUs or other sources; or (4) create a hybrid of these options.

Figure 4.1 shows the states affected by CAIR.

Although CAIR is currently being litigated, most of the petitioners have raised relatively peripheral issues that do not threaten the core of the rule.

Operating Permits

Under Title V of the Clean Air Act, individual stationary sources implement SIP requirements through operating permits issued by the state. Both the state operating permit programs and individual operating permits are subject to EPA approval. A permit, which lasts for five years, sets specific emissions limits for the source for various pollutants. Permits are intended to collect in one document all of the pollution control, operating, emissions, monitoring, reporting and certification requirements for a facility. Most states have permitting programs for minor sources in addition to their Title V permitting programs for major sources.

Mobile Sources

Under the Clean Air Act, the federal government sets emissions standards for new vehicles. However, as noted previously in this chapter, California can set new vehicle emissions standards that are stricter than federal standards and other states may adopt those standards. States are not preempted from setting their own standards for most types

of existing vehicles.

Although this arrangement means that SIP writers have fewer opportunities to go beyond federal standards in the case of mobile sources (as compared to stationary sources), it would be a mistake to conclude that states have no options for reducing vehicle emissions. In fact, there are countless cases of state and local programs addressed to motor vehicle emissions. For example, in 2005 New Jersey enacted a mandatory diesel retrofit program for onroad vehicles and New York City adopted legislation mandating the retrofit of various types of trucks and buses. More than 20 cities and states have requirements that restrict the idling of diesel vehicles and at least ten states have adopted or moved to adopt California light-duty vehicle standards. EPA reported in 2003 that 33 states and the District of Columbia had an inspection and maintenance program for light-duty vehicles in at least one city or county. According to the Department of Energy, eight states and the federal government provided purchase incentives for environmentally friendly vehicles in 2005. The mobile source chapters of this report (Chapters 12–15) discuss state and local strategies, including these, in greater detail. This section aims only to provide an overview of the Clean Air Act as it applies to mobile sources.

Although EPA regulates emissions from new vehicles and vehicle fuels and fuel additives under the Clean Air Act, the National Highway Traffic Safety Administration (within the U.S. Department of Transportation) regulates vehicle fuel efficiency by setting corporate average fuel economy (CAFE) standards under the Energy Policy and Conservation Act of 1975. However, the line between vehicle emissions standards and CAFE standards has become disputed recently in the context of on-going litigation by automakers that challenges California's efforts to reduce greenhouse gas emissions from cars. In their lawsuit, the automakers argue that the California rules amount to de facto fuel economy standards and are therefore preempted by federal law, which puts vehicle efficiency beyond the reach of state regulatory authority.

Motor Vehicles, Nonroad Engines, Aircraft

The Clean Air Act divides mobile sources into three categories: on-road motor vehicles, (such as cars, trucks and buses); aircraft; and nonroad vehicles and engines (which is a large catch-all category for vehicles and equipment like lawnmowers, tractors, forklifts, motorboats, ships, locomotives, construction equipment, and the like). Generally, it is the federal government rather than the states that bears the responsibility for regulating emissions from new sources in these categories. The principal exception to this is the authority granted to California to adopt its own more stringent emissions standards for new mobile sources. Other states are permitted to adopt these California standards and a number have exercised this important authority. For example, to date at least ten states have adopted or moved to adopt California's low emissions vehicle program; nine states have adopted California's heavy-duty diesel truck standards for model years 2007 and later; and 13 states have adopted California's diesel truck "not-to-exceed" requirements (STAPPA and ALAPCO, 2005).

For *existing* onroad motor vehicles, states and localities have much more leeway; in fact, they are free to adopt their own standards for these vehicles. However, in the case of *existing nonroad equipment*, they are limited to California standards (§209(e)). The Clean Air Act also preempts state regulation of aircraft (§233), new locomotives, and small construction and farm equipment (§209(e)).

Among federal vehicle standards, the so-called "Tier 2" standards govern exhaust emissions from light-duty vehicles, light-duty trucks, and medium-duty passenger vehicles (a category that captures some minivans and SUVs) (§§202 (a) and (b); EPA, 2000b). Although a separate statutory program governs vehicle fuels (§211), the Tier 2 program also requires gasoline refiners to lower the sulfur in gasoline because sulfur in fuel interferes with emissions control equipment.

EPA has promulgated the so-called "Phase 2" rule (as distinct from the Tier 2 rule) to reduce emissions from heavy-duty engines and vehicles. Under the Phase 2 rule, sulfur levels in diesel fuel will decline by 97 percent compared to current levels (EPA, 2000c). Special provisions under §219 of the Clean Air Act impose more stringent PM standards for urban buses than for other heavy-duty engines (EPA, 1993a). Section 219(d) establishes an urban bus rebuild/retrofit program (EPA, 1993b). Additionally, EPA has promulgated a series of rules since 1995 regulating a number of categories of nonroad engines.

Apart from provisions governing emissions standards, the Act provides authority for a host of other specific vehicle programs, such as those governing evaporative emissions, onboard diagnostics (to determine whether a vehicle's emissions control systems are functioning properly), and vehicle fleets in certain ozone and carbon monoxide nonattainment areas.

Fuels and Fuel Additives

Under §211(c) of the Clean Air Act, EPA has the authority to regulate vehicle fuels and fuel additives. As we have seen in the context of the Tier 2 and Phase 2 rules discussed above, the Agency has increasingly tended to treat emissions standards and fuel requirements as a single system rather than as separate issues. This is largely because of the effect of some fuel components (e.g., lead, sulfur) on required pollution control equipment.

Additionally, the Act creates a number of specific programs for vehicle fuels, including programs to control the volatility of gasoline (§211(h)); to address the reformulation of gasoline in certain areas (§§211(k) and 211(c)); and to prohibit the sale of leaded gasoline for use in motor vehicles (§211(n)). The recent Energy Policy Act of 2005 eliminated the requirement that oxygenates be used in reformulated fuel to reduce emissions of CO and ozone precursors, but it also required that ethanol use for vehicle fuel be tripled by 2012.

As discussed in Chapter 4, *The Clean Air Act*, if EPA has regulated a fuel characteristic or component, states other than California may *not* adopt either their own or California's regulation—they must follow the federal regulation. The only exception to this is the situation in which the Administrator finds that state regulation is necessary for achieving the NAAQS. On the other hand, California may adopt its own fuel regulations (§211(c)(4)).

Transportation Measures

In addition to emissions standards for mobile sources and provisions for fuels and fuel additives, the Clean Air Act includes a variety of transportation-related measures. Although many of these provisions are addressed to ozone nonattainment areas, they are worthy of consideration for PM reduction purposes, since NO_x is a precursor not only of ozone, but also of PM. The transportation provisions in §182 of the Act include inspection and maintenance requirements for automobiles as well as transportation control measures aimed at reducing vehicle miles traveled, congestion, and work-related vehicle trips.

The conformity provisions of §176(c) of the Clean Air Act are *not* limited to ozone nonattainment areas. Although these provisions have been applied most frequently to federal transportation projects, §176(c) prohibits federal action—including federal financial assistance—for any transportation project that does not conform to the applicable SIP.

Finally, §209(d) of the Act specifically provides that "[n]othing…shall preclude or deny to any State or political subdivision thereof the right otherwise to control, regulate, or restrict the use, operation, or movement of registered or licensed motor vehicles." Therefore, state and local programs such as those aimed at limiting idling, reducing congestion and encouraging shared transportation are clearly permissible.

The Proposed $PM_{2.5}$ Implementation Rule

On September 8, 2005, EPA issued its proposed $PM_{2.5}$ implementation rule, which addresses requirements that states must meet in developing their SIPs for attaining the $PM_{2.5}$ NAAQS (EPA, 2005d). The proposed rule includes the following provisions:

Attainment dates: Attainment demonstrations and SIPs are due by April 2008. Attainment dates are five years after designation, or April 2010 for most areas. Extensions are possible.

Classifications: EPA prefers not to have a classification system for nonattainment areas but may decide to establish a two-tiered system, with areas classified as moderate or serious.

$PM_{2.5}$-precursors: SO_2 emissions must always be addressed in $PM_{2.5}$ SIPs. VOCs and ammonia need not be addressed unless EPA or the state demonstrates that either is a significant contributor to $PM_{2.5}$ formation in the area. NO_x must be addressed in the absence of a demonstration that it is not a significant contributor.

RACT: EPA suggests a number of possible approaches to RACT. First, it is unlikely that RACT will be required for all stationary sources in all nonattainment areas because of the Agency's view that certain areas will attain the standards by 2010 solely as a result of existing federal emissions reduction programs. With some caveats, previous RACT determinations for other programs may be acceptable for purposes of $PM_{2.5}$ RACT.

NSR: The NSR/PSD proposals are extensive. They include:

- a major source threshold of 100 tons per year;
- a significant emissions increase level of ten tons per year;
- an offset ratio of 1:1, with the possibility of inter-pollutant trading;
- PM_{10} as a surrogate for $PM_{2.5}$ until the final rule is promulgated.

EPA's proposed $PM_{2.5}$ implementation rule also includes a special provision for RACT and EGUs. If a state meets its CAIR SO_2 cap through EGU reductions only, then EGUs that are located in nonattainment areas and that are complying with CAIR would be found to meet RACT. The same would be true with respect to the state's NO_x cap, but only if EGUs that have installed selective catalytic reduction control technology operate this technology year round.

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Chapter 5

Boiler Technologies

Introduction

Boilers account for a large share of the fuel consumed by American industry. In 2003, the electric power sector alone consumed more than 23,000 trillion British thermal units (TBtu) of fuel for steam production (EIA, 2004). Apart from the electric industry, the most steam-intensive industries are forest products (1,381 TBtu), chemical manufacturing (1,055 TBtu), and petroleum refining (680 TBtu) (EERE, 2004). A substantial share of the overall fuel consumed by these industries is used for steam production; steam boilers account for 84 percent of the total energy used in the pulp and paper industry, 47 percent of the energy used in the chemical manufacturing industry, and 51 percent of the energy used in the petroleum refining industry (DOE, 2002).

In short, a boiler is a closed vessel or tank in which heat produced from the combustion of fuels such as natural gas, fuel oil, or coal is used to generate hot water or steam. The steam produced in boilers is used for a range of applications, including process heating, electricity production, and facility heating.

This chapter discusses pollution control techniques for coal-, residual oil-, natural gas- and wood-fired boilers. The technologies and methods described are used to control fine particulate matter ($PM_{2.5}$), sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions from electric-generating boilers (EGUs, discussed in Chapter 7) as well as from industrial and commercial boilers (discussed in Chapter 6 and in several of the sector-specific chapters of this report).

The control options discussed in this chapter are divided into four categories: (1) fuel switching and fuel cleaning, (2) combustion control technologies, (3) post-combustion control technologies, and (4) multi-pollutant control technologies. Switching to a cleaner-burning fuel or treating a fuel prior to combustion can reduce $PM_{2.5}$, SO_2 and NO_x emissions, as well as a host of air toxic emissions. Combustion controls are primarily used to control NO_x emissions. Post-combustion controls generally target a single pollutant or category of pollutants. Multi-pollutant controls, as the name implies, are integrated systems designed to reduce several pollutants.

This chapter focuses exclusively on the technical options for reducing emissions from boilers. Chapter 7, *Electric Generating Units*; Chapter 6, *Industrial and Commercial Boilers*; and the sector-specific chapters (e.g., pulp and paper) include policy options for addressing boiler emissions.

Fuel Switching

Fuel switching encompasses several different control options. One option is blending lower-polluting fuels to reduce overall emissions. For example, many boiler

operators blend lower sulfur (subbituminous) coals with higher sulfur (bituminous) coals to reduce SO_2 emissions. Another option is to substitute a lower emitting fuel entirely. For example, power plants will sometimes switch from using oil to natural gas during the summer ozone season to comply with seasonal NO_x emissions requirements, or switch fuels year round to reduce annual emissions.

The feasibility of fuel switching will depend in part on the characteristics of the plant and the particular type of fuel change being considered. Many plants will be able to switch from high-sulfur to low-sulfur bituminous coal without serious difficulty, but switching from bituminous to subbituminous coal may present greater challenges and costs. In some instances, fuel switching will require significant investment and modifications to an existing plant. Switching to a lower sulfur coal can affect coal handling systems, boiler performance, PM control-effectiveness and ash handling systems (Rastogi, 2001; Rubin, 2001).

In general, for fuel substitution to be practical, there must be a suitable replacement fuel available at an acceptable cost. Boiler owners must evaluate the availability, performance, and cost of the alternatives. While most industrialized areas have access to a variety of fuels, some fuels may not be practical in certain locations. Natural gas is generally supplied by pipeline. Owners of facilities that are not located near existing pipelines may find it expensive to arrange for a natural gas supply. Since coal is typically supplied by railroad and the characteristics of coal from different areas of the country vary widely, some types of coal may not be available at a given location because of shipping costs.

In some cases, fuel switching will have an impact on the performance of the boiler (even if it is a fuel of the same type with different characteristics). For example, many coal-fired boilers burn low-sulfur Powder River Basin (PRB) coal to reduce SO_2 emissions. However, while the PRB coals are inherently low in sulfur, they also have a lower heating value than eastern bituminous coal. Facilities must test-burn the fuel and typically must limit their use of PRB coal (e.g., a 15/85 percent blend of PRB/bituminous) to avoid having to de-rate the unit or face other operating difficulties. Fuel substitution is not feasible for sources where the substitution would require excessive retrofits or would entail substantial performance losses (EPA, 1998).

If alternative fuels are available, retrofit costs are

Table 5.1

Average Fuel Prices

	2004		2005 (Jan.-Sep.)	
Fuel	**Price**	**Price per MMBtu**	**Price**	**Price per MMBtu**
Bituminous coal[a]	$30.56/ton	$1.27	NA	NA
Subbituminous coal[a]	$8.12/ton	$0.41	NA	NA
Residual oil[b] (1% sulfur or more)	$0.835/gal	$5.57	$1.081/gal	$7.21
Residual oil[b] (less than 1% sulfur)	$0.692/gal	$4.61	$0.923/gal	$6.15
Distillate oil[c]	$1.31/gal	$9.36	$1.845/gal	$13.18
Natural gas[d]	$6.09 - $6.43/$10^3$ cf	$5.97 - $6.30	$6.94 - $7.50/103 cf	$6.80 - $7.35

NA = Not Available
MMBtu = million British thermal units

a. Average price from U.S. Energy Information Administration (EIA), Table 31. Average Open Market Sales Price of Coal by State and Coal Rank, 2004, available at: http://www.eia.doe.gov/cneaf/coal/page/acr/table31.html. Price per MMBtu calculated assuming a heat content of 12,000 Btu per pound for bituminous coal and 10,000 Btu per pound for subbituminous coal.
b. Average price from EIA, Table 19. U.S. Refiner Residual Fuel Oil Prices, Average Sales to End Users, available at: http://www.eia.doe.gov/pub/oil_gas/petroleum/data_publications/petroleum_marketing_monthly/current/pdf/pmmtab19.pdf. Price per MMBtu calculated assuming a heat content of 150 MMBtu per thousand gallons.
c. Average price from EIA, U.S. No. 2 Distillate Prices by Sales Type, Industrial Consumers, available at: http://tonto.eia.doe.gov/dnav/pet/pet_pri_dist_dcu_nus_a.htm. Price per MMBtu calculated assuming a heat content of 140 MMBtu per thousand gallons. Note that distillate oil is not normally used in electric generating boilers.
d. Average industrial and electric industry price from EIA, U.S. Natural Gas Prices. Available at: http://tonto.eia.doe.gov/dnav/ng/ng_pri_sum_dcu_nus_a.htm. Price per MMBtu calculated assuming a heat content of 1,020 MMBtu/10^6 standard cubic feet (scf).

reasonable, and performance loss is manageable, fuel prices will tend to be the determining factor in deciding whether fuel switching is viable. Table 5.1 reports recent (2004–2005) average fuel prices for coal, oil and natural gas.

The replacement fuel must also provide a meaningful reduction in emissions to merit the change. The effect that the replacement fuel has on emissions of pollutants other than the pollutant in question should also be considered. For example, switching to low-sulfur coal to reduce SO_2 emissions may increase PM emissions. (See discussion of electrostatic precipitator (ESP) to understand why this might occur.)

Switching from residual oil to natural gas or distillate oil is one option for reducing emissions. Residual oil contains higher concentrations of sulfur and ash compared to distillate oil and natural gas. Fuel oils are classified by number: No. 1 and No. 2 fuel oils are both distillate oils. No. 5 and No. 6 fuel oils are residual oils. No. 4 fuel oil can be distillate or a mixture of residual and distillate (EPA, 1998).

EPA estimates greater than a 90 percent reduction in $PM_{2.5}$ emissions when switching from residual oil to distillate oil or natural gas. Note, however, that distillate oil is not normally used in electric-generating boilers (EPA, 1998).

Table 5.2 provides EPA estimates of the potential $PM_{2.5}$ reductions associated with switching away from bituminous or subbituminous coal to lower emitting fuels.

Table 5.3 shows EPA's estimates of the potential reductions in SO_2 emissions from fuel substitution.

Switching from coal to natural gas reduces NO_x emissions (EPA, 1998). Switching from residual oil to distillate oil or natural gas can result in limited NO_x emissions reductions, but achieving significant NO_x reductions requires the application of combustion and post-combustion control technologies.

Fuel Cleaning

Coal washing is widely practiced in the U.S. to remove impurities (e.g., sulfur) and to increase the coal's heating value.

Conventional (physical) coal washing techniques remove ash and sulfur from coal by crushing the fuel and separating the different components in a liquid bath, such as water.[1] The lighter coal particles float to the top of the bath for recovery, while the heavier impurities sink to the bottom for removal.

Sulfur in coal exists in two forms. The first is as an inorganic form called "pyrite." The second is organic sulfur, which is chemically bound in the molecular structure of coal itself. Conventional coal washing removes only inorganic sulfur and nitrogen—physical cleaning will not remove organic sulfur.

1. Coal washing generates solid and liquid wastes that must be properly handled and disposed of in order to avoid contaminating surface and ground water supplies.

Table 5.2

Estimated Potential Direct $PM_{2.5}$ Emissions Reductions with Fuel Switching away from Coal

Sector	Original Fuel	Switch to Subbituminous (% Reduction)	Switch to Residual Oil[a] (% Reduction)	Switch to Natural Gas (% Reduction)	Switch to Distillate Oil[b] (% Reduction)
Industrial[c]	Bituminous Coal[e]	21.4	7.4	93.1	99.1
	Subbituminous Coal[f]	–	–	91.2	98.8
Utility[c,d]	Bituminous Coal[e]	21.4	14.8	97.5	–
	Subbituminous Coal[f]	–	–	96.8	–

a. Assuming ash content of 0.03% by weight and sulfur content of 2.5% by weight.
b. Assuming ash content <0.01% by weight and sulfur content of 0.22% by weight; typically not used in utility boilers.
c. Based on emissions from dry bottom boilers.
d. Utilities tend to operate more efficiently than industrial units and have lower resulting PM emissions.
e. Assuming ash content of 8.6% by weight.
f. Assuming ash content of 5.2% by weight.

Source: EPA, 1998

Table 5.3

Estimated SO_2 Emissions Reductions with Fuel Switching

Original Fuel	Subbituminous Coal (Percent Reduction)	Distillate Oil[b] (Percent Reduction)	Natural Gas (Percent Reduction)
Bituminous Coal	72.9	91.2	99.9
Subbituminous Coal	–	69.5	99.9
Residual Oil[a]	–	91.5	99.9
Distillate Oil[b]	–	–	99.7

a. Assuming ash content of 0.03% by weight and sulfur content of 2.5% by weight.

b. Assuming ash content <0.01% by weight and sulfur content of 0.22% by weight; typically not used in utility boilers.

Source: EPA, 1998

Coal washing removes varying degrees of ash and sulfur, depending on the type of coal being washed and the cleaning process used. Organic sulfur, which cleaning does not remove, accounts for 35–75 percent of the total sulfur content in Illinois Basin coals, for example (TVA, 2005). Physical coal cleaning typically removes 40–50 percent of the pyritic sulfur in typical llinois coal. The end result is a 20–25 percent reduction in SO_2 emissions (Southern Illinois University, 2005).

In addition to lowering air emissions, coal washing also increases the heating value of the fuel. This lowers the transportation cost of the fuel per unit of energy, offseting the costs associated with the coal washing. Coal washing also reduces the maintenance costs associated with the boiler. According conference paper, unplanned outages—resulting from the use of coal with higher ash and sulfur content—can cost a modest size coal plant millions of dollars in lost revenue (Cole, 2004).

Chemical and biological cleaning methods have been developed in an effort to remove organic sulfur, but these are used only infrequently.

The cost of coal cleaning varies depending on the type of coal and is offset by reduced ash disposal requirements, increased generation per ton of coal, reduced SO_2 and PM emissions, decreased transportation costs, reduced equipment wear and tear and lower operating and maintenance costs.

Combustion Control Technologies

Combustion controls, also known as combustion modifications, attempt to suppress the formation of NO_x within the boiler by controlling peak flame temperatures, oxygen concentrations, and residence times in the active combustion zones. The most common technologies are low-NO_x burners (LNBs) and overfire air (OFA), used alone or in combination.

There are two basic pathways of NO_x formation within a boiler. First, the combustion process releases nitrogen bound in the fuel, which combines with oxygen to form "fuel NO_x." Second, high combustion temperatures break apart stable nitrogen molecules in the air, which then recombine with oxygen to form "thermal NO_x."[2] Fuel NO_x is sensitive to oxygen concentrations in the combustion zone and can be limited by using a lower nitrogen fuel or by limiting oxygen availability during the early stages of combustion. Thermal NO_x is sensitive to temperature and oxygen availability and can be controlled by increasing the size of the combustion zone or by reducing the temperature in the combustion zone. Relatively little thermal NO_x is formed at temperatures below 1,700 degrees Fahrenheit (°F) (Srivastava, 2005).

Bituminous and subbituminous coals usually contain between 0.5 and 2 percent nitrogen by weight. Fuel nitrogen can account for as much as 80 percent of total NO_x emissions from coal combustion (EPA, 1995). Fuel NO_x is also the more important NO_x-forming mechanism in boilers that burn residual oil. Thermal NO_x is the dominant NO_x-forming mechanism in units fired by distillate oils; it is the only NO_x-forming mechanism in natural gas combustion systems (EPA, 1995). Wood-fired boilers normally operate at lower combustion temperatures relative to other fuels (around 1,500°F for wood-fired boilers compared to over 2,000°F for natural gas- or oil-fired boilers). Since wood-fired boilers normally burn at lower temperatures, the units have inherently lower NO_x emissions (Washington DEC, 2003). This is especially true for older and smaller wood-fired boilers. As a result, NO_x combustion control technologies are not applicable to wood-fired boilers. However, post-combustion NO_x controls (such as selective catalytic reduction) can be effective on wood-fired boilers that operate at higher temperatures. (Newer and larger wood-fired boilers use technologies that allow for higher

2. NO_x also forms from the reaction of molecular nitrogen with hydrocarbon radicals (prompt NO_x). However, prompt NO_x accounts for only a small fraction of total NO_x formation and is not the focus of control efforts (Srivastava, 2005; Washington DEC, 2003).

Table 5.4

Estimated Costs (2004$) and Reduction Potential for Low-NO_x Burners, Overfire Air, and Flue Gas Recirculation on Industrial Boilers

Boiler Fuel Type	Control Technology	NO_x Reduction Potential[a]	Boiler Size (MMBtu/hr)	Capital Cost ($/MMBtu/hr)[b]	Cost Effectiveness ($/ton removed)[b]
Coal	LNB	50%	100	5,100	500–3,000
	LNB	50%	250	3,900	400–2,300
	LNB/OFA	50%–65%	100	7,300	600–4,400
	LNB/OFA	50%–65%	250	5,500	500–3,300
Oil	LNB/OFA	30%	100	2,600	600–5,300
	LNB/OFA	30%	250	1,900	500–4,000
	LNB/OFA/FGR	30%–50%	100	4,000	600–11,000
	LNB/OFA/FGR	30%–50%	250	3,000	500–8,600
Natural gas	LNB/OFA	60%	100	2,500	600–11,000
	LNB/OFA	60%	250	1,900	400–8,000
	LNB/OFA/FGR	80%	100	4,000	700–12,300
	LNB/OFA/FGR	80%	250	3,000	500–9,400

a. From Table III-4 in NESCAUM, 2005

b. From Table III-4 in NESCAUM, 2005. Rounded to the nearest 100.

combustion temperatures.)

Some of the combustion techniques described below involve running the boiler slightly inefficiently in order to create a cooler combustion zone. Generally speaking, the lower the flame temperature, the less NO_x is formed. However, operating the boiler to create a cooler combustion zone results in incomplete combustion and can increase PM, carbon monoxide (CO) and volatile organic compound (VOC) emissions.

Combustion controls have been installed on more than 50 percent of coal-fired boilers in the electric generating sector; however, only a small percentage of industrial boilers have combustion controls (NESCAUM, 2005).

Low-NO_x Burners

LNBs reduce the formation of both thermal and fuel NO_x by staging the combustion process to produce fuel-rich and fuel-lean zones within the combustion flame (EPA, 1999). LNBs and OFA (discussed below) are widely used with coal-fired boilers (Srivastava, 2005). According to the U.S. Department of Energy (DOE), LNBs are now installed on approximately 75 percent of the suitable coal-fired electric generating capacity in the U.S. (DOE, 2005). According to EPA, LNBs and flue gas recirculation (discussed below) are the two most common combustion control techniques for natural gas-fired boilers (EPA, 1995).

Table 5.4 lists the NO_x reduction potential and estimated costs for LNBs and LNBs with OFA on coal-fired boilers, as well as the costs and reduction potential of LNBs with OFA and flue gas recirculation on oil- and natural gas-fired boilers. LNBs and OFA have little impact on operating costs; hence, selection is driven primarily by the capital or retrofit costs of the technology (NESCAUM, 2005). Table 5.4 is specific to *industrial* boilers. NESCAUM reports a cost-effectiveness for LNBs of $200–$500 per ton of NO_x removed for boilers in the electric generating sector (NESCAUM, 2005). The Midwest Regional Planning Organization (RPO) reports a cost-effectiveness of $200–$1,000 for LNBs for a coal-fired boiler (Midwest RPO, 2005).

LNBs are capable of reducing coal plant emissions to roughly 0.25–0.65 pounds per million British thermal units (lb/MMBtu), depending on coal type and combustion conditions (DOE, 2005). DOE is currently working to develop a next generation of LNBs that will be capable of achieving an emissions rate comparable to the control levels currently achievable with post-combustion controls. Pilot studies using these LNBs have achieved an emissions rate below 0.15 lb/MMBtu with some ranks of coal. This is equivalent to a 70–75 percent emissions reduction (Richards, 2002).

Overfire Air

OFA, or staged combustion, diverts a portion of the

combustion air (i.e., the air supplied to the burner) from the burners to the region above the burner zone. This two-stage combustion process lowers the combustion temperature in the boiler, reducing NO_x formation. OFA is frequently paired with LNBs when used on coal-, natural gas-, and oil-fired boilers (Davis, 2000).

As a stand-alone modification to a coal-fired boiler, OFA can reduce NO_x emissions up to 30 percent. Combined systems can achieve greater reductions. OFA in combination with LNBs can reduce NO_x formation by greater than 50 percent. Table 5.4 provides estimates of the capital costs of installing LNBs with OFA on *industrial* boilers. A more detailed presentation of these data is available in Chapter 6, *Industrial and Commercial Boilers.*

The capital costs for LNBs with OFA on a coal-fired *utility* boiler are estimated to be \$15,000–\$25,000 per MW or \$1,500–\$2,500 per MMBtu per hour (Davis, 2000).[3] The Midwest RPO reports a cost-effectiveness of \$250–\$600 per ton removal for OFA for a coal-fired boiler (Midwest RPO, 2005).

Flue Gas Recirculation

Flue gas recirculation is a temperature reduction process that involves recirculating a portion of the flue gas with the combustion air. Recirculation lowers the combustion temperature, reducing NO_x formation (EPA, 1999). Flue gas recirculation can be paired with LNBs and OFA.

Flue gas recirculation is frequently used on gas-fired boilers and can be used on oil-fired boilers (EPA, 1995). EPA reports a 55–65 percent reduction in NO_x emissions using flue gas recirculation with natural gas-fired boilers and a 15–30 percent NO_x reduction with residual oil-fired boilers (EPA, 1995).

LNBs and OFA are frequently used in combination with flue gas recirculation to reduce NO_x emissions (EPA, 1999). Table 5.4 provides cost and control-efficiency estimates for this combination with an industrial boiler.

Atomizer Optimization/Replacement

Oil-fired boilers use atomizers to produce oil droplets small enough to burn at a reasonable rate. Mechanical atomizers use the pressure of the oil supply for atomization and thus require relatively high oil pressure for operation. Steam atomizers use steam (or in some cases, compressed air) to form droplets from the fuel; they are often able to form smaller droplets than mechanical atomizers.

Existing atomizers should be in good working order to ensure minimal particulate (unburned carbon) emissions. When mechanical atomizers are used for heavier fuels, some heating of the atomizers will be necessary to lower the oil viscosity enough to allow proper droplet formation.

Replacement of older atomizers, or of mechanical atomizers with steam atomizers where cost allows, can reduce unburned carbon emissions and opacity on the order of 50 percent in some cases. Lower unburned carbon emissions will be reflected in improved boiler thermal efficiencies.

Reburning

Reburning, sometimes classified as a combustion control (NESCAUM, 2005) and sometimes as a post-combustion control (Srivastava, 2005), involves injecting coal, oil, or natural gas above the primary combustion zone. The objective is to produce an area with hydrocarbon fragments that react with NO_x to form other nitrogen-containing materials that can be reduced to nitrogen (Srivastava, 2005). Generally, it is more economical for a facility to use the same fuel for reburning as it does for primary combustion, although there are exceptions.

Reburning can be used on natural gas-, oil- or coal-fired boilers to control NO_x emissions. In order to use coal as a reburning fuel, it must be finely ground, which requires additional pulverizing equipment. A number of coal-fired electric generators with natural gas-fired reburning equipment have stopped using the equipment because of the rising cost of natural gas (Srivastava, 2005).

Worldwide, over 40 boilers have installed reburning equipment with reported or expected NO_x reductions of 25–78 percent. In the U.S., reburning equipment has been installed on 26 coal-fired utility boilers since 1991. Of those, only four are currently operating. The remaining installations are either on decommissioned boilers or are not operational for site-specific reasons such as the cost of natural gas (Srivastava, 2005). EPA estimates that 50–70 percent NO_x reductions can be achieved with reburning on natural gas-fired boilers (EPA, 1999).

Capital costs for reburn range from \$15 to \$30 per kilowatt (kW) for reburn when gas is the primary fuel, and from \$30 to \$60 per kW when coal is the primary fuel. Cost-effectiveness is estimated at \$500–\$2,000 per ton of NO_x removed (NESCAUM, 2005).

Post-Combustion Controls

Post-combustion controls are pollution control devices that are placed downstream of the boiler to remove pollutants from the flue gases. Post-combustion controls are effective at removing PM (including $PM_{2.5}$), SO_2 and NO_x. The sections below are organized by pollutant and control technology, beginning with PM controls.

3. Assuming a heat rate of 10,000 Btu per kilowatt-hour.

PM Post-Combustion Controls

Particulate control is an issue only with coal-, oil-, and wood-fired combustion units. Natural gas-fired boilers are inherently low emitting.

Fabric Filters

A fabric filter, more commonly known as a baghouse, traps particulates in the flue gas before they exit the stack. Fabric filters are made of a woven or felted material in the shape of a cylindrical bag or a flat, supported envelope. The system includes a dust collection hopper and a cleaning mechanism for periodic removal of the particulates. Initially, industrial facilities (e.g., steel manufacturing) were the primary users of baghouses, but over the past two decades the thermal resistivity of filter material has been improved and the use of fabric filters on coal-fired utility boilers has increased (Davis, 2000).

EPA does not recommend using fabric filters for boilers that use oil exclusively. Particles from oil combustion tend to clog the filter. EPA recommends using fabric filters for wood- and coal-fired utility and industrial boilers (EPA, 1998).

According to EPA, a fabric filter on a coal-fired utility boiler can capture up to 99.9 percent of total particulate emissions and 99.0–99.8 percent of $PM_{2.5}$. Fourteen percent of coal-fired utility boilers in the U.S. have installed fabric filters (EPA, 2004).

Table 5.5 summarizes cost estimates for three common types of fabric filters. The first of these filters—a filter with a mechanical shaker—is effective on coal-fired boilers, while the other two types of filters are effective on both coal- and wood-fired boilers. The various types of fabric filters have similar collection methods; however, they differ in terms of the methods used to clean the filters and remove the dust. Costs are reported in terms of the gas flow of the unit. The quantity of flue gas to be handled is one of the key parameters that dictates the sizing and hence the cost of a fabric filter.

Over the past several years, the development of new filter media and special coating and finishing techniques has extended the life and improved the performance of fabric filters. Modern computer controls continuously monitor and adjust the cleaning process, improving emissions control and lowering costs (IEA, 2003).

A baghouse downstream of an ESP (described below) provides high rates of $PM_{2.5}$ removal. Because the precipitator removes the bulk of the particulate burden, the baghouse can be relatively small, reducing its cost. For example, the Compact Hybrid Particulate Collector (COHPAC) is a technology patented by the Electric Power Research Institute (EPRI) in which a baghouse is placed downstream of an existing ESP to improve overall particulate collection efficiency. A hybrid PM collection system can achieve 99.99 percent particulate control for all particles from 0.01 to 50 micrometers (µm) in diameter, and can be retrofit to an existing ESP (IEA, 2003).

Electrostatic Precipitators

An ESP uses an electrical charge to separate the particles in the flue gas stream under the influence of an electric field. In brief, an ESP works by imparting a positive or negative charge to particles in the flue gas stream. The particles are then attracted to an oppositely charged plate or tube and removed from the collection surface to a hopper by vibrating or rapping the collection surface.

There are four common types of ESPs: dry wire-pipe, dry wire-plate, wet wire-pipe, and wet wire-plate. Older ESPs are generally of the wire-pipe design, while newer ones are generally of the wire-plate design. The standard technology for utility and industrial boilers is the dry wire-plate ESP (EPA, 2003e). Wet ESPs are used in situations where the material to be collected is wet, sticky, flammable, or explosive, or has a high resistivity (EPA, 2003g).

Table 5.5

Estimated Costs and Reduction Potential for Fabric Filters[a]

Filter Type	PM Reduction Potential	Capital Cost ($/scfm)	O&M Cost ($/scfm)	Cost-Effectiveness ($/ton removed)
Mechanical shaker cleaned	95%–99.9%	8–72	4–24	37–303
Pulse jet cleaned	95%–99.9%	6–26	5–24	42–266
Reverse-air cleaned	95%–99.9%	9–85	6–27	53–337

SCFM=standard cubic feet per minute

a. Filters are effective on $PM_{2.5}$ but the cost estimates are for removal of all particulates.

Source: EPA, 2003a; EPA, 2003b; EPA, 2003c

Table 5.6

Estimated Costs (2002$) and Reduction Potential for Dry and Wet Wire-Plate ESPs

Filter Type	$PM_{2.5}$ Reduction Potential	Capital Cost ($/scfm)	O&M Cost ($/scfm)	Cost-Effectiveness ($/ton removed)
Dry wire-plate	96%–98%	10–33	3–35	35–236
Wet wire plate	99%	20–40	5–40	48–520

SCFM = standard cubic feet per minute

Source: EPA, 2003e; EPA, 2003g

An ESP can be installed at one of two locations. Most ESPs are installed downstream of the air heater, where the temperature of the flue gas is between 130°C–180°C (270°F–350°F). An ESP installed downstream of the air heater is known as a "cold side" ESP. An ESP installed upstream of the air heater, where flue gas temperatures are significantly higher, is known as a "hot side" ESP.

Seventy-two percent of coal-fired utility boilers in the U.S. have installed cold side ESPs and 11 percent have installed hot side ESPs (EPA, 2001a); 80 percent of all ESPs in the U.S. are used in the electric industry (EPA, 2003e).

The effectiveness of an ESP depends in part on the electrical resistivity of the particles in the flue gas. Coal with a moderate to high amount of sulfur produces particles that are more readily controlled. Low-sulfur coal produces a high-resistivity fly ash that is more difficult to control. The effectiveness of an ESP also varies depending on particle size. A dry ESP can capture greater than 99 percent of total PM from a coal-fired boiler, while capturing 80 to 95 percent of $PM_{2.5}$ (EPA, 1998; EPA, 2003e; EPA, 2004).

Experience with wet ESPs on utility and industrial boilers is limited. However, wet ESPs are generally considered to be more efficient than dry ESPs at removing $PM_{2.5}$ and their usage is increasing. Wet ESPs are reportedly capable of achieving 99 percent $PM_{2.5}$ removal efficiency (Altman, 2003).

Table 5.6 lists cost ranges (in 2002 dollars) for conventionally designed dry and wet wire-plate ESPs operating under typical conditions.

As indicated, a large number of existing coal-fired boilers have ESPs installed. Several options are available for improving the performance of these systems (EPA, 2005c):

- **Improved Monitoring.** Improved monitoring includes the installation of an opacity monitor or a continuous PM monitoring device.
- **Conditioning Agents.** Conditioning agents change the properties of the ash resulting from combustion. Conditioning with ammonia, hydrated sodium carbonate/bicarbonate, or other agents can improve particulate collection efficiency.
- **Upgraded ESPs.** Upgrades include increasing the size of the precipitator, replacing older "weighted wire" electrodes with modern "rigid discharge" electrodes, and adding advanced electronic controls.

Wet Scrubbers

Wet scrubbers are also used to control particulate emissions. The two most frequently utilized wet scrubbers for PM control are condensation scrubbers and venturi scrubbers. Condensation scrubbers are designed to control PM with a diameter of 0.25–1.0 µm (EPA, 2003k). Venturi scrubbers are effective in controlling both PM less than or equal to 10 µm in diameter (PM_{10}) and $PM_{2.5}$ (EPA, 2003j). Wet scrubbers can also be used to control SO_2 emissions; SO_2 scrubbers are used in flue gas desulfurization, a technology described below.

Venturi scrubbers are used to control PM emissions from utility, industrial, and commercial coal-, residual oil- and wood-fired boilers. Table 5.7 presents estimates of particulate collection efficiency and cost for condensation and venturi scrubbers.

SO_2 Post-Combustion Controls

Flue Gas Desulfurization

Flue gas desulfurization (FGD) uses a sorbent—usually lime or limestone—to remove SO_2 from the exhaust gases of a fossil-fuel-fired boiler. FGD scrubbers can be wet or spray dry. FGD scrubbers can be used on coal-fired utility and industrial boilers. Outside of the U.S., FGD scrubbers have also been used on heavy oil-fired boilers. Approximately 85 percent of the systems in the U.S. are wet systems, 12 percent are spray dry, and 3 percent are dry systems (EPA, 2003m). FGD scrubbers have been applied to coal-fired boilers ranging in size from

Table 5.7

Estimated Costs and Reduction Potential for Wet Scrubbers

Scrubber Type	PM Reduction Potential	Capital Cost ($/scfm)	O&M Cost ($/scfm)	Cost-Effectiveness ($/ton removed)
Condensation Scrubbers[a]	>99%	6	2.5	59
Venturi Scrubbers[b]	70%–90%	2.5–21	4.4–120	70–2,400

a. 1993$
b. 2002$

Source: EPA, 2003j, 2003k

50 MMBtu per hour to 15,000 MMBtu per hour (EPA, 2003m).

FGD scrubbers are capable of reduction efficiencies ranging from 50 percent to 98 percent. Historically, wet FGD scrubbers have the highest removal efficiencies (greater than 90 percent) while dry FGD scrubbers have the lowest. However, newer dry scrubbers are capable of efficiencies near 90 percent (EPA, 2003m).

Table 5.8 shows EPA cost estimates for wet and spray dry scrubbers (the two most common FGD scrubber types) for various sizes of electric boilers.

Dry Sorbent Injection

Dry sorbent injection involves the direct release of lime (or other suitable sorbent) into the ductwork or boiler upstream of a PM control device. The SO_2 in the flue gas reacts with the powdered sorbent, which can then be collected by the downstream PM control device. The SO_2 control-efficiency of existing dry injection systems ranges from 40 to 60 percent when using lime or limestone, and up to 90 percent using other sorbents (e.g., sodium bicarbonate) (EPA, 2005b). This option is likely to be most cost-effective for smaller units, less than 300 megawatts (MW), because of the lower capital costs of the technology ($70 per kW) relative to a wet scrubber (Argus Air Daily, 2005).

NO_x Post-Combustion Controls

Selective Catalytic Reduction

Selective catalytic reduction (SCR) is a well developed and widely applied post-combustion NO_x control technology (Davis, 2000). SCR reduces NO_x emissions using a reducing agent and a catalyst. SCR is typically used on electric industry boilers, industrial boilers, process heaters, gas turbines, and reciprocating internal combustion engines that burn fossil fuels. SCR systems have been used on coal- and natural gas-fired electric utility boilers ranging in size from 25 MW to 800 MW. The technology is commercially available for oil-fired boilers but not widely demonstrated (Davis, 2000). SCR retrofits on existing boilers can increase capital costs by over 30 percent compared to installing SCR on a new unit (EPA, 2003h).

SCR can be used as a stand-alone technology or in combination with other NO_x control technologies (e.g., LNBs or flue gas recirculation). Capital costs for SCR are substantially higher than other NO_x control options described in this chapter (EPA, 2003h).

EPA estimates a NO_x control-effectiveness of 80–95

Table 5.8

Estimated Costs (2001$) and Reduction Potential for Wet and Spray Dry FGD Scrubbers

Scrubber Type	SO_2 Reduction Potential	Unit Size (MMBtu/hr)	Capital Cost ($/MMBtu)	O&M Cost ($/MMBtu)	Cost-Effectiveness ($/ton removed)
Wet	>90%	>4,000	1,000–2,500	20–80	200–500
Wet	>90%	<4,000	2,500–15,000	80–200	500–5,000
Spray dry	80%–90%	>2,000	400–1,500	40–100	150–300
Spray dry	80%–90%	<2,000	1,500–15,000	100–3,000	500–4,000

Source: EPA, 2003m

Table 5.9

Estimated Costs (1999$) for SCR Technology

Unit Type	Capital Cost	O&M Cost	Cost-Effectiveness ($/ton removed)
Industrial coal boiler (350 MMBtu)	$10,000–$15,000 per MMBtu/hr	$300 per MMBtu/hr	2,000–3,000
Industrial oil, gas, or wood boiler (350 MMBtu)	$4,000–$6,000 per MMBtu/hr	$450 per MMBtu/hr	1,000–3,000
Coal-fired electric industry boiler (100-850 MW)	$50,000–$110,000 per MW	$0.4–$4.2 million per year	1,000–1,500

Source: EPA, 2001b; EPA 2003h; NESCAUM, 2005

percent for coal-fired electric industry boilers (EPA, 2001b). The main factors in determining the emissions reduction potential of this technology are temperature, the amount of reducing agent, injection grid design and catalyst activity.

EPA includes SCR in its Integrated Planning Model (IPM) as one of the post-combustion NO_x controls available for coal-, oil-, and natural gas-fired boilers. The Agency assumes a 90 percent NO_x removal efficiency for SCR systems installed on coal units and an 80 percent NO_x removal efficiency for SCR systems installed on oil- and natural gas-fired boilers (EPA, 2002).

SCR systems require a large volume of catalyst material; as a result, capital costs are significantly higher than for other types of NO_x controls. Table 5.9 summarizes cost estimates for SCR. The costs in Table 5.9 assume a high capacity factor and annual NO_x control. These assumptions can significantly affect operation and maintenance (O&M) costs and hence overall cost-effectiveness.

The cost of retrofitting an oil- or gas-fired boiler with SCR controls is estimated at about one-half to one-third the cost of retrofitting a coal-fired boiler because gas- and oil-fired boilers require less catalyst than a coal-fired boiler (NESCAUM, 2005).

Selective Non-catalytic Reduction

Selective non-catalytic reduction (SNCR) is a post-combustion NO_x control technology that has been installed on a wide range of boiler configurations, including coal-, natural gas-, biomass-, and waste-fired units. There are hundreds of SNCR systems installed in the U.S. on boilers ranging in size from 50 MMBtu/hr to 6,000 MMBtu/hr (EPA, 2003i).

As shown in Table 5.10, SNCR operating alone has a NO_x reduction potential of 30–70 percent. Combined with other controls, such as LNBs, SNCR can achieve reductions of 65–75 percent (EPA, 2003i; Srivastava, 2005).

EPA includes SNCR as one of the post-combustion NO_x controls available for coal-, oil-, and natural gas-fired boilers in IPM. The Agency assumes a 35 percent NO_x removal efficiency for SNCR systems installed on coal units and a 50 percent NO_x removal efficiency for SNCR systems installed on oil- and natural gas-fired boilers (EPA, 2002).

Table 5.10 also shows EPA cost estimates (in 1999 dollars) for SNCR systems installed on industrial boilers greater than 100 MMBtu per hour in size. The cost of a retrofit is approximately 10–30 percent greater than the cost of SNCR for a new boiler (EPA, 2003i). SNCR systems

Table 5.10

Estimated Costs (1999$) and Reduction Potential of SNCR Technology

Unit Type	NO_x Reduction Potential[a]	Capital Cost	O&M Cost	Cost-Effectiveness ($/ton removed)
Industrial boilers (>100 MMBtu/hr)	30%–70%	$900–$2,500 per MMBtu/hr ($9,000–$25,000 per MW)	$100–$500 per MMBtu/hr ($1,000–$5,000 per MW)	400–2,500[a] 2,000–3,000[b]

a. annual NO_x control

b. seasonal NO_x control

Source: EPA, 2003i; Srivastava, 2005

have a lower capital cost than SCR, but higher operating costs. EPA reports a wide range of cost-effectiveness estimates for SNCR due to site-specific conditions. SNCR is considered an effective control option for seasonal NO_x reductions since capital costs are relatively low (EPA, 2003i).

The Midwest RPO reports a cost-effectiveness of $800–$1,500 for SNCR for a coal-fired boiler (Midwest RPO, 2005).

Multi-Pollutant Control Technologies

Control technology manufacturers are developing a new generation of multi-pollutant controls designed to capture multiple air pollutants more cost-effectively than existing single-pollutant controls. In addition to better performance and reduced cost, multi-pollutant control technologies can offer the following benefits:

- fewer system components, reducing space requirements;
- reduced operational complexity;
- lower auxiliary power requirements, resulting in less impact on heat rate (i.e., efficiency) of a unit;
- lower collateral emissions (e.g., ammonia slip); and
- reduced water consumption, wastewater discharge and solid waste (in some cases with production of a marketable co-product) (Khan, 2005).

Dozens of new multi-pollutant control technologies have been developed and tested in recent years. A few of these technologies have advanced to the point where they are now commercially available. This section provides descriptions of commercially available multi-pollutant technologies—including cost and performance estimates where available—together with descriptions of the most promising of the emerging technologies.

Commercially Available Multi-Pollutant Control Technologies

LoTOx™

Low temperature oxidation (LoTOx™) employs ozone injection to oxidize NO_x and elemental mercury for subsequent removal in a wet scrubber. The technology is licensed by the BOC Group and is being marketed as a control option for NO_x that is less expensive than SCR and also has the ability to remove mercury. LoTOx™ can be used on new or existing units (BOC Group, 2005).

An on-site ozone generator is used to produce the ozone feed for the system from an oxygen source. Once injected into the LoTOx™ reactor, the ozone oxidizes the NO_x and elemental mercury present in the flue gas. The oxidized compounds can then be removed in a downstream wet scrubber (EPA, 2003; MJB&A, 2004).

Full-scale commercial LoTOx™ systems have been installed successfully in a variety of industries, including the steel and recycling sectors. The technology has been demonstrated on a pilot-scale only at coal-fired electric generating facilities.

The LoTOx™ system is reported to control 70–95 percent of the NO_x in flue gas to levels below 0.15 lb/MMBtu (Khan, 2005). BOC normally recommends installing LoTOx™ alone when inlet NO_x levels are below 0.3 lb/MMBtu. Above 0.3 lb/MMBtu inlet NO_x, LoTOx™ is recommended as part of an integrated control approach, meaning that it would operate in series with an alternate control process such as LNBs, OFA, or selective auto-catalytic reduction. By combining two technologies, users may be able to achieve NO_x reductions similar to those achieved by an SCR system alone, but at lower capital and operating costs (EPA, 2005a).

Total mercury removal rates of 90 percent from all coal ranks have been estimated for this process, although additional testing at the pilot- and full-scale level are needed to demonstrate mercury removal capabilities (MJB&A, 2004).

When used with FGD, 95 percent of SO_2 can be removed. According to BOC Group, LoTOx™ enhances the SO_2 removal efficiency of the FGD system by approximately 5 percent (Keeth, 2003). CO_2 emissions increases of 1.0–2.5 percent can be expected due to the auxiliary power requirements associated with the system (EPA, 2005a).

Suppliers estimate capital costs for the LoTOx™ process to be $35–$70 per kW (for LoTOx™ only; not including FGD), depending on the inlet NO_x level and unit size (EPA, 2005a). Annual operating and maintenance costs are estimated to range from 1.7 to 2.4 mills/kWh (for LoTOx™ used in conjunction with selective auto-catalytic reduction, not including FGD) (MJB&A, 2004).[4]

MobotecSystem™

The MobotecSystem™ combines three different control technologies to reduce NO_x, CO, SO_2, mercury and other heavy metal emissions.

The MobotecSystem™ is based on three separate technologies that can be installed together or individually: (1) rotating opposed fired air (ROFA); (2) rotary mixing of

4. One mil is equal to 1/10 of one cent (or $0.001).

chemicals (Rotamix); and (3) an in-duct catalyst (SCR) that may be used in conjunction with the previous two systems for lower cost than a standard SCR system operated alone (Mobotec, 2005).

- ROFA affects combustion and NO_x emissions directly by staging the furnace and turbulently mixing the flue gas with high-velocity secondary air.
- Rotamix is an advanced SNCR system that makes use of the well-mixed flue gas and high-velocity air achieved with ROFA to inject and evenly distribute various chemicals (e.g., ammonia, urea, trona, limestone) or additional fuel (e.g., reburn gas) into the flue gas to remove NO_x, SO_2, and heavy metals (including mercury). Because ROFA and Rotamix are generally designed as one system, the MotobecSystem™ is considered a multi-pollutant control option.
- In-duct SCR is an independent NO_x control option that can be used in conjunction with ROFA and Rotamix to increase the overall NO_x control-effectiveness of the MobotecSystem™ to a level comparable to that achievable with a full size, stand-alone SCR system (approximately 90 percent) (Coombs, 2004). Upstream NO_x reductions by the ROFA and Rotamix technologies decrease the SCR costs by reducing the amount of catalyst required to achieve 90 percent control efficiency.

There are currently 18 ROFA-Rotamix installations in the U.S. and 17 in Sweden (mainly on stoker-fired, fluidized-bed, and dry-bottom boilers burning coal, heavy fuel oil, wood wastes, and municipal waste). The system can reportedly be employed with most fuel types and firing configurations (EPA, 2005a).

In terms of potential O&M impacts, ROFA can be expected to improve combustion efficiency and lower CO emissions. However, the system can lead to reductions in superheat and reheat temperatures. The impact of the system on O&M costs is very site-specific (depending on the boiler design, operating conditions, and fuel characteristics). The system requires auxiliary power to operate, as well as inputs of various chemicals such as ammonia, urea, limestone, and trona (EPA, 2005a).

Air pollutant removal efficiencies for the MobotecSystem™ are estimated at 50–85 percent for NO_x, 60–90 percent for SO_2, and 65–90 percent for mercury (Keeth, 2003; Coombs, 2004; Bierman, 2005). The ROFA system alone reduces NO_x emissions by 40–60 percent. The ROFA-Rotamix systems combined reduce NO_x emissions by a total of 60–80 percent, and the complete system, with the in-duct SCR, reduces NO_x emissions by 90 percent. Several units, operating commercially, have demonstrated the NO_x reduction capability of the system. There is less experience with the system in reducing SO_2 and mercury emissions (Haddad, 2003a). The first commercial application of the system for mercury control is currently planned at two Minnesota power plants (Duluth News Tribune, 2005).

The MobotecSystem™ is marketed as a cost-effective alternative to SCR, based primarily on its lower catalyst use (Haddad, 2003b; Coombs, 2004). The total capital costs for a 400 MW and a 700 MW MobotecSystem™ are estimated at $10 million and $15 million, respectively. The annual operating costs for a 400 MW and 700 MW system are estimated at $100,000 and $175,000, respectively. The system also consumes energy to run the ROFA fans (Haddad, 2003b).

Advanced Dry Scrubbing

Advanced dry scrubbing (ADS) is designed to improve the performance of conventional spray dry FGD units and is primarily designed for SO_2 and mercury control on incinerators and coal-fired boilers burning low- to medium-sulfur coals. The process involves the reaction of hydrated lime slurry with exhaust gases in an absorber vessel to capture SO_2, sulfur trioxide (SO_3), hydrochloric acid (HCl), and mercury. ADS technology utilizes a fluidized bed or a flash dryer for the reactor, with a recycle loop to enhance lime utilization (MJB&A, 2004).

For the most part, ADS designs represent variations on circulating fluid bed (CFB) technology, with design features specific to each vendor. A CFB reactor is a vertical reactor in which a dense material bed of recycled products (ash and sorbent) ensures rapid cooling and a high degree of contact between gases and solids. The flue gas flows upward through the bed, with sorbent typically being sprayed as a slurry into the gas upstream of the bed. The ash and sorbent product is recycled back to the reactor through a dedicated cyclone integral to the system, or directly from the final particulate collection device (ESP or fabric filter).

Specific variants of the technology offered by major vendors include (EPA, 2003):

- circulating dry scrubbers (CDS), offered by LURGI;
- gas suspension absorbers (GSA), offered by FLS Miljo;
- reflux circulating fluidized bed absorbers (RCFB), offered by WULFF; and
- rapid absorption process (RAP), offered by Beaumont Environmental.

While not widely used in the U.S., all four CFB-based technologies are commercially available and have been

installed in the U.S. and abroad.

O&M impacts associated with ADS technologies are similar to those associated with conventional spray dryers. Potential impacts include: increased pressure loss, need for increased auxiliary power, increased loading of solids on the downstream particulate collectors, corrosion, and increased water requirements. The magnitude of these impacts is a function of specific design characteristics and operating conditions; it also depends on whether the ESP or fabric filter is new.

ADS technology is reportedly capable of removing more than 95 percent of the SO_2 and SO_3 in flue gas. Less information is available on mercury removal rates, which vary significantly depending on coal type, operating conditions, and the presence of other pollution control devices in the plant. EPA has reported mercury removal rates as high as 97 percent and as low as 50 percent using the LURGI CDS system. The opportunity to add "dedicated" mercury sorbent is predicted to increase overall mercury removal potential (EPA, 2005a).

Capital cost estimates for ADS technologies range from $50 to $150 per kW, depending on the specific technology (Keeth, 2003; EPA, 2005):

FLS Miljo reports a cost of $150/kW for GSA.

- LURGI estimates capital costs for its CDS system at $140/kW.
- According to WULFF, the capital cost of RCFB at a 275 MW plant in Austria was $90/kW.
- Beaumont Environmental reports a cost of $50/kW for a RAP installation at a 150 MW plant.

ReACT

The Japanese electric generating company J-POWER owns the ReACT multi-pollutant removal system, in which flue gas passes through a two-stage bed of activated coke. In the first stage, sulfur oxides are largely removed through oxidation to sulfuric acid. The second stage involves the injection of ammonia to decompose NO_x into nitrogen and water. The adsorber is a moving bed, which is withdrawn for regeneration in a separate vessel. Mercury adsorbed on the activated coke is released in vapor form during regeneration and is captured on a selenium filter, where it is fixed as a stable compound. The technology can produce saleable byproducts, including raw sulfur and sulfuric acid (MJB&A, 2004; Khan, 2005).

ReACT has not been used in North America, but is being used at 17 coal, sintering, incinerator, and oil- and refuse-derived fuel installations worldwide (EPA, 2005a). J-POWER is marketing this technology as a more cost-effective option for SO_2, NO_x and mercury control than the combination of SCR and wet FGD. According to J-POWER, ReACT offers the following advantages (J-POWER, 2005):

- higher removal rates for elemental and oxidized mercury;
- smaller space requirements;
- much less water consumption;
- lower auxiliary power requirements;
- no SO_3 slip;
- fewer retrofit items (duct, fans, etc.);
- no need to retrofit the air preheater and stack; and
- availability of collected activated coke for reuse.

Due to the high heat capacity of the activated coke system, it takes longer than a conventional plant to reach the operating temperature of the de-NO_x process, which means that NO_x emissions will be higher during start-up or that more time will be required for plant start-up. Other O&M impacts are not known, mainly because of the lack of experience in the U.S. Required auxiliary power is estimated to be 0.70 percent of gross output, compared to 1.0–2.5 percent for a wet FGD system (EPA, 2005a).

ReACT reportedly has demonstrated removal rates of greater than 95 percent for SO_2, 10–50 percent for NO_x, greater than 90 percent for mercury and greater than 95 percent for PM (with an ESP). J-POWER is targeting a NO_x removal rate of 70 percent. NO_x control-effectiveness is higher when SO_2 concentrations at the inlet of the adsorber are lower. The amount of ammonia injected also affects NO_x reduction, as do the oxygen concentration of the flue gas and the inlet gas temperature (J-POWER, 2005; Khan, 2005).

According to J-POWER, capital costs for the ReACT system (including the cost of equipment to make byproducts) are projected to range from $100 to $160 per kW for units 300–900 MW in capacity. Variable operating costs are estimated to range from 1.7 to 2.6 mills per kWh (J-POWER, 2005; Keeth, 2003).

Emerging Multi-Pollutant Control Technologies

SO_x-NO_x-RO_x Box

The SO_x-NO_x-RO_x Box (SNRB) offered by Babcock and Wilcox combines the removal of SO_2, NO_x, and particulates in one unit—a high temperature baghouse. The baghouse, located before the air heater, operates in

the 425°C–455°C (800°F–850°F) temperature range. SO_2 removal involves injecting calcium- or sodium-based sorbent, sodium carbonate and sodium bicarbonate into the flue gas upstream of the baghouse. NO_x removal is accomplished by injecting ammonia to reduce NO_x selectively in the presence of an SCR catalyst placed on the high-temperature ceramic filter bags of the baghouse. The baghouse also removes particulates.

The SNRB system can be applied to both new and existing units and has been demonstrated successfully at Ohio Edison's Burger Unit No. 5 (a 156 MW boiler burning high-sulfur bituminous coal). However, demonstration on a larger scale is needed to better assess the performance and economics of this technology.

Based on pilot-scale testing, the SNRB can remove 90 percent of the NO_x and 80–90 percent of the SO_2 in flue gas, while reducing PM levels to less than 0.02 lb/MMBtu.

The only potential O&M impact of this technology relates to its effects on the life expectancy of fabric filters. The SNRB filters are ceramic, incorporating a catalyst for NO_x control, and their reliability is affected by both mechanical and thermal stress due to changing operating conditions and clean-up of the bags.

The supplier of these systems estimates that capital costs for a 150 MW retrofit would be $253 per kW (in constant 1994 U.S. dollars), assuming 3.5 percent sulfur coal, baseline NO_x emissions of 1.2 lb/MMBtu, a 65 percent capacity factor, and 85 percent SO_2 and 90 percent NO_x removal. No estimates of O&M costs are available (Keeth, 2003; EPA, 2005a).

ECO®

Powerspan's ECO® technology removes $PM_{2.5}$, SO_2, NO_x and mercury from the flue gas of coal-fired power plants. It treats the flue gas in three steps to achieve multi-pollutant removal, using a barrier discharge reactor, an ammonia scrubber, and a wet ESP. An absorber tower contains the ammonia scrubber and precipitator. Mercury collected from the absorber tower is removed from the liquid stream with an activated carbon filter. Since this technology includes a scrubber, its primary market is unscrubbed existing plants and new plants burning medium- to high-sulfur bituminous coal (EPA, 2003; MJB&A, 2004; Khan, 2005).

The ECO® technology is in the demonstration stage. Powerspan has been operating a 50 MW commercial demonstration unit at FirstEnergy Corporation's R.E. Burger Plant since February 2004. The demonstration unit is located downstream of the plant's ESP and the treated flue gas is returned to the existing plant ductwork just before the stack. In September 2005, FirstEnergy announced plans to install an ECO® system at the company's Bay Shore Plant Unit 4 (215 MW) (Boyle, 2005).

The main O&M impact of the ECO® process is its auxiliary power consumption. Powerspan estimates that approximately 3 percent of the plant's output is needed for the ECO® process to reduce inlet NO_x levels from 0.30–0.05 lb/MMBtu. Also, cooling water is required for the wet ESP and heating is required for the acid recovery and ash drying. While these auxiliary power requirements do not present a technical challenge, they will have some impact on a plant's mass and energy balances, as well as on its overall efficiency. At this point, there is not enough information to assess the magnitude of this impact, but it is not expected to be substantial (EPA, 2005a).

Based on testing performed to date, ECO® technology is estimated to reduce SO_2 emissions by 98 percent, NO_x emissions by 90 percent, and mercury emissions by 85 percent. The 50 MW demonstration at the R.E. Burger Plant achieved results of this magnitude (Boyle, 2005).

Powerspan and others performed a detailed cost estimate for an ECO® unit at AmerenUE's Sioux plant. The capital cost of the system for this 510 MW installation was estimated at $120.4 million. This figure includes costs for the fertilizer plant and balance-of-plant modifications (EPA, 2005a). FirstEnergy estimates the cost of the system at its 215 MW Bay Shore Plant, including a fertilizer processing plant, to be approximately $100 million (FirstEnergy, 2005). Although the performance and costs of this technology remain uncertain, the results of the demonstration project suggest that an estimated cost of $200 per kW is reasonable. Projected O&M costs are 1.5 mills/kWh (Keeth, 2003; EPA, 2005a).

Powerspan has entered into an alliance with Wheelabrator Air Pollution Control to commercialize the ECO® technology (Powerspan, 2005). As the process is scaled, the main uncertainties relate to whether it can replicate the performance that it has achieved at smaller scale. Also, the economic benefits of the technology are dependent, in part, on the ability to market the fertilizer co-product produced by the process. Powerspan expects the capital costs of the technology to be approximately 10–20 percent less than a combination of conventional technologies that achieves similar performance levels, and the operating costs are generally expected to be lower depending on a host of factors (Boyle, 2005).

Pahlman™ Process

EnviroScrub's Pahlman™ Process is a closed-loop dry sorbent system comprised of two discrete steps. One step involves capturing the target pollutants—NO_x, SO_x, mercury, and particulates—using Pahlmanite™ dry

mineral sorbent compounds. The other step involves the regeneration of the spent or partially spent sorbent compounds for reuse and separation, and the isolation of useful byproducts for use in fertilizers and industrial chemicals. The Pahlmanite™ sorbents are low-density oxides of manganese in the form of a fine black powder. The sorbent is injected in a reactor, which operates at a temperature between ambient temperature and 320°F. According to the supplier, the technology can be used on different types of reactors and can be applied to new units or retrofitted to existing units (EPA, 2003; MJB&A, 2004; Khan, 2005).

The Pahlman™ technology is in a pilot-scale stage. A trailer-mounted pilot plant has been tested at a number of power plants, using flue gas slipstreams (1,000 scfm) (EPA, 2005a). Results from testing at Minnesota Power's Boswell Station indicate that the technology achieved SO_2 reductions above 99 percent, NO_x reductions of 94–97 percent, and mercury reductions of up to 67 percent (Khan, 2005).

O&M impacts for this technology are unknown. Auxiliary power demand is expected to increase, but no specific estimates are available (EPA, 2005a). According to the supplier, capital costs for a 500 MW plant are projected to be approximately $150 per kW (Keeth, 2003). The supplier projects variable O&M costs at approximately 1.45 mills/kWh (EPA, 2005a).

The technology is still at an early development stage and requires further demonstration and assessment for a more comprehensive picture of its cost-effectiveness. Removal efficiencies for mercury and particulates need to be demonstrated. The pollutant removal and sorbent regeneration steps have been commercially demonstrated individually, but not in an integrated system. An integrated 20 MW demonstration project is currently planned for installation at the Boswell Station (EPA, 2005a).

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Chapter 6

Industrial and Commercial Boilers

Introduction

Industrial and commercial boilers combust large quantities of fuel to produce steam and hot water for a variety of uses. These boilers represent about 40 percent of all energy use in the industrial and commercial sectors (EEA, 2005). Industrial boilers usually drive mechanical equipment or heat process materials (e.g., petroleum). Commercial boilers (which include institutional boilers) generally provide space heating and hot water for hotels, hospitals and other large facilities. Some boilers produce electrical power for on-site industrial consumption, sometimes cogenerating steam and electricity.

In this chapter we discuss fine particulate matter ($PM_{2.5}$), sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions from the industrial and commercial boiler source category and the opportunities for controlling these emissions. The discussion includes boilers fired by coal, residual oil, natural gas and wood. Boilers fired by process byproducts (e.g., black liquor solids in pulp and paper recovery boilers) are discussed in other chapters. We do not address boilers fired by other fuels (e.g., distillate oil, tires or biogas) because they contribute a small share of emissions.

In order to avoid excessive repetition, the more detailed technical discussion of emissions control opportunities for industrial, commercial and electric power sector boilers appears in Chapter 5, *Boiler Technologies*.

Sector Profile

In the U.S., there are an estimated 43,000 industrial boilers with a total capacity of 1,600,000 million British thermal units per hour (1.6 million MMBtu/hr), and 120,000 commercial boilers with a total capacity of 1.1 million MMBtu/hr. Industrial boilers tend to be larger than commercial boilers, with an average size of 36 MMBtu/hr as compared to 9.6 MMBtu/hr. Industrial boilers also tend to have higher utilization rates (EEA, 2005). Boilers in the electric industry are larger still.

The universe of industrial and commercial boilers is

Table 6.1

Industrial and Commercial Boilers in the U.S. by Capacity

Capacity Range (MMBtu/hr)	Industrial Boilers	% of Total Industrial Boilers	Commercial Boilers	% of Total Commercial Boilers
0-10	23,800	56%	480,650	95%
10-100	14,860	35%	24,890	5%
100-250	2,560	6%	1,120	<1%
>250	1,480	3%	130	<1%
TOTAL	**42,700**	**100%**	**506,790**	**100%**

Source: Alvis, 2002; EEA, 2005

difficult to characterize for a number of reasons, the primary one being that some boilers, commercial boilers in particular, are so small that little detailed information is routinely collected on them. Additionally, data on boilers are reported in a variety of categories—by fuel type, boiler type and industry sector. As a result, inventory and emissions data are far from precise.

In support of a recent rulemaking, EPA estimated the number of *industrial* boilers in the U.S. by capacity and fuel type.[1] According to the estimates, 49 percent of *industrial* boilers are fired by natural gas, 29 percent by coal, 10 percent by residual oil, 4 percent by wood and 7 percent by other fuels (Alvis, 2002).

According to a review of industrial and commercial boiler populations completed by Energy and Environmental Analysis (EEA) for Oak Ridge National Laboratory in 2005, most data on boilers at commercial facilities are incomplete. EEA estimates that 85 percent of commercial boilers are fired by natural gas, 11 percent by oil, 1.3 percent by coal and 2 percent by other fuels. Of the 130 large capacity commercial boilers (larger than 250 MMBtu/hr), EEA estimates that 80 are coal-fired and 50 are gas-fired.

Table 6.1 summarizes the number of industrial and commercial boilers by capacity.

Table 6.1 shows that while there are many more commercial than industrial boilers in operation in the U.S., the majority of them are smaller than 10 MMBtu/hr. This difference in capacity is reflected in the emissions data presented below.

At the end of this chapter, Table 6A.1 provides additional information on the industrial boiler population by capacity and predominant fuel type. Units with multiple fuels are a small proportion of the total number of units, and they have been grouped according to their primary fuel type.

Based on the data presented, more than half of industrial boilers in the U.S. have capacities under 10 MMBtu/hr; however, these small boilers account for only a small share of industrial boiler capacity. Coal-fired industrial boilers larger than 250 MMBtu/hr and natural gas-fired industrial boilers between 10 and 100 MMBtu/hr account for the largest share of capacity in the industrial boiler sector, with each accounting for about 19 percent of total capacity. There are 594 coal-fired boilers larger than 250 MMBtu/hr, and 10,738 natural gas-fired boilers between 10 and 100 MMBtu/hr. Natural gas-fired industrial boilers larger than 250 MMBtu are the next largest category, with 572 boilers accounting for 16 percent of total industrial boiler capacity.

Emissions

Table 6.2 provides a breakdown of 1999 annual emissions in the U.S. from industrial and commercial boilers by fuel type, based on data from EPA's National Emission Inventory (NEI). Note that the commercial boiler numbers include institutional boilers.

Although there are almost triple the number of commercial boilers in the U.S. compared to industrial boilers, the vast majority of them are small. The predominance of small boilers combined with the large number (85 percent) that are natural gas-fired, means that commercial boilers contribute far fewer emissions when compared to the industrial boiler category.

According to the current inventory data, despite being a relatively clean burning fuel, natural gas-fired boilers account for about 40 percent of $PM_{2.5}$ emissions from the industrial and commercial boiler sectors. The sheer number of natural gas-fired boilers (about half of industrial boilers and 85 percent of commercial boilers) is partly responsible for this result. However, during a review of the 2002 NEI, EPA concluded that the emissions factors used to estimate $PM_{2.5}$ emissions from natural gas combustion in the 2002 NEI and prior inventories were too high. EPA

1. In support of the Industrial/Commercial/Institutional Boiler and Process Heaters National Emission Standards for Hazardous Air Pollutants (EPA, 2004).

Table 6.2

Industrial and Commercial Boiler Sector Emissions

	$PM_{2.5}$		SO_2		NO_x	
Fuel	**Tons/Year**	**% of Sector Total**	**Tons/Year**	**% of Sector Total**	**Tons/Year**	**% of Sector Total**
Industrial boilers						
Coal	12,242	7%	728,809	57%	288,393	33%
Bituminous & subbituminous	12,091	7%	670,175	53%	266,756	31%
Anthracite & lignite	151	0%	58,634	5%	21,638	3%
Oil	10,417	6%	178,025	14%	80,015	9%
Residual	8,332	5%	160,876	13%	62,565	7%
Distillate	2,085	1%	17,149	1%	17,450	2%
Natural gas	58,630	35%	51,952	4%	253,456	29%
Wood/bark	32,568	20%	17,369	1%	73,636	9%
Other	36,474	22%	188,935	15%	94,843	11%
Industrial boilers subtotal	**150,331**	**90%**	**1,165,090**	**91%**	**790,343**	**92%**
Commercial boilers						
Coal	2,961	2%	84,033	7%	22,772	3%
Bituminous & subbituminous	2,417	1%	79,291	6%	20,737	2%
Anthracite & lignite	545	0%	4,742	0%	2,035	0%
Oil	1,616	1%	19,092	1%	9,074	1%
Residual	449	0%	14,254	1%	6,570	1%
Distillate	1,167	1%	4,838	0%	2,504	0%
Natural gas	7,553	5%	3,022	0%	33,681	4%
Wood/bark	3,230	2%	2,818	0%	4,065	0%
Other	477	0%	2,034	0%	3,024	0%
Commercial boilers subtotal	**15,837**	**10%**	**110,999**	**9%**	**72,617**	**8%**
Industrial and commercial boiler sector total	**166,168**	**100%**	**1,276,089**	**100%**	**862,960**	**100%**

Source: EPA, 2005a, based on facilities listed under Source Classification Codes starting with 102 and 103

intends to revise the 2002 NEI, dramatically reducing (by 95 percent) its estimates of the $PM_{2.5}$ emissions from natural gas combustion (EPA, 2005d). It is important to note that the figures reported in Table 6.2 reflect the *earlier* methodology.

While coal-fired boilers account for 29 percent of industrial boiler capacity and 6 percent of the number of industrial boilers, they account for more than 60 percent of SO_2 emissions and a third of NO_x emissions from the industrial and commercial boiler sector. They were assumed to account for 9 percent of $PM_{2.5}$ emissions from the industrial and commercial boiler sector, although this will change with the revision of the $PM_{2.5}$ estimates for natural gas-fired units. In the commercial sector, coal-fired boilers account for almost all the SO_2 emissions.

Industrial wood-fired boilers account for 20 percent of industrial and commercial boiler $PM_{2.5}$ emissions, despite the fact that they are only 4 percent of industrial boiler capacity and 2 percent of the industrial boiler inventory. This share will also increase with the revision of the $PM_{2.5}$ estimates for natural gas-fired units.

The "Other" categories in Table 6.2 includes about a fifth of all $PM_{2.5}$ emissions from the industrial and commercial boiler sector. A large percentage of these emissions come from boilers firing process materials (discussed in the specific sector chapters). However, many of the control options discussed in this chapter and in Chapter 5 can be used or modified for use on process fuel-fired boilers.

SO_2 compounds form when sulfur in coal or oil is oxidized during combustion. Fuel oil sulfur content depends on grade, but it is generally less than 1 percent. Natural gas has virtually no sulfur and its combustion therefore produces almost no SO_2. About 95 percent of the sulfur in fossil fuels is released as gaseous SO_2 when combusted (EPA, 1995).

NO_x formation results from the thermal fixation of atmospheric nitrogen in the combustion flame ("thermal NO_x"), and from oxidation of nitrogen in the fuel ("fuel NO_x"). Bituminous and subbituminous coals usually contain from 0.5 to 2 percent nitrogen by weight. Fuel nitrogen can account for as much as 80 percent of total NO_x emissions from coal combustion (EPA, 1995). Fuel NO_x is also the more important NO_x-forming mechanism in residual oil-fired boilers, but thermal NO_x dominates in units firing distillate oils.

Table 6.3 shows the emissions rates of uncontrolled industrial and commercial boilers by fuel type. While individual boiler units may have emissions rates different from those in the table (depending on the boiler type), the table is intended to provide a sense of the range of emissions rates for uncontrolled boilers.

As a further point of reference, in developing the control

Table 6.3

Emissions Rates for Uncontrolled Industrial and Commercial Boilers by Fuel Type

	Emissions Rates (lb/MMBtu)		
Fuel Type	**$PM_{2.5}$**	**SO_2**	**NO_x**
Coal (high-sulfur)[a]	0.11–0.37[b]	5.25–5.70	0.36–1.65
Coal (low-sulfur)[c]	0.11–0.37	0.875–0.95	0.36–1.65
Residual Oil[d]	0.026–0.046	0.6–1.04	0.13–0.37
Natural Gas[e]	0.007	0.0005	0.098–0.230
Wood[f]	0.25–0.72	0.008–0.025	0.04–0.49

lb/MMBtu = pounds per million British thermal units

a. Assumes sulfur content = 3% and ash content = 5%.

b. Compared to low-sulfur coal, high-sulfur coal produces only slightly more direct PM, since boilers convert virtually all of the sulfur in coal to SO_2. SO_2 may lead to secondary PM, but the SO_2 column in the chart (as opposed to the PM column) reveals this characteristic.

c. Assumes sulfur content = 0.5% and ash content = 5%; assumes heat value of 10,000 MMbtu/lb.

d. For #6 oil, assumes sulfur content = 1% and PM coefficient (A) = 1.49.
For all oil types, assumes heat value = 0.15 MMbtu/gallon.
For all oil types, assumes normal firing (i.e., not tangential).

e. Assumes heat value of 1,020 Btu/standard cubic feet (scf), 100% conversion of fuel sulfur to SO_2, and sulfur content = 2,000 grains/106 scf.

f. Assumes heat value of 0.0045 MMBtu/lb.

Source: EPA, 1995; Davis, 2000

cost estimates for the Clean Air Interstate Rule (CAIR), EPA assumed uncontrolled emissions for industrial boilers of 0.72 pounds of NO_x per MMBtu (lb/MMBtu) for coal-fired boilers; 0.25 lb/MMBtu for natural gas-fired boilers; and 0.22 lb/MMBtu, 0.36 lb/MMBtu, and 0.5 lb/MMBtu for oil-fired boilers. (The two higher oil-fired NO_x values were added to better reflect the variability in inventory data (Khan, 2003a).) EPA conducted similar analyses for SO_2 control technologies, but the Agency did not specify uncontrolled emissions rates (Khan, 2003b).

Emissions Control Opportunities

Numerous options are available for reducing $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from industrial and commercial boilers. In general, the options fall into three categories: (1) improving combustion efficiency; (2) fuel switching to lower emitting fuels; and (3) using control technologies (including combustion and post-combustion controls). The discussion that follows provides an overview of the control options for industrial and commercial boilers. A more detailed discussion of the controls for coal-, oil-, natural gas- and wood-fired boilers, including fuel switching, is contained in Chapter 5, *Boiler Technologies*.

Improving Combustion Efficiency

A properly tuned and maintained boiler requires less fuel to generate steam efficiently, thereby reducing the amount of fuel burned in the boiler, which in turn reduces the amount of air pollution exiting in the flue gas. Some states require boiler owners to conduct boiler tune-ups to optimize combustion efficiency. For example, New York requires owners of small boilers (e.g., 20–50 MMBtu/hr in some levels of nonattainment area) to perform an annual tune-up (NY DEC, 2004). In general, larger boilers are better equipped to continuously optimize combustion efficiency (e.g., by continuously monitoring oxygen and carbon monoxide levels in the flue gas), although there are opportunities for owners of smaller boilers as well.

Fuel Switching

Fuel switching can be an effective strategy for reducing $PM_{2.5}$, SO_2 and NO_x emissions. For example, switching from a high-sulfur (bituminous) coal to a lower-sulfur (subbituminous) coal can reduce a unit's SO_2 emissions by more than 70 percent. However, switching to a lower-sulfur coal can affect coal handling systems, boiler performance, PM control-effectiveness and ash handling systems (Rastogi, 2001; Rubin, 2001). Therefore, the feasibility of switching to a lower-sulfur coal will require a facility-specific determination, taking into consideration the cost and availability of different types of coal in different areas. Blending lower-sulfur and higher-sulfur coal has also been used to control SO_2 emissions and often requires fewer modifications than a complete substitution. Coal blending has been successfully implemented in the electric generating sector (Russell, 2005).

Although expensive, switching from coal to natural gas produces even more dramatic results, virtually eliminating a plant's SO_2 and direct $PM_{2.5}$ emissions and significantly lowering NO_x emissions. Switching from oil to natural gas is easier than switching from coal to natural gas, and many plants have dual oil- and gas-firing capability. Going from a higher- to a lower-sulfur residual oil or from residual to distillate oil is also possible, although facility-specific issues related to differences in the viscosity of the fuel may need to be addressed.

Fuel switching is not without its complications. The availability of fuel switching as a control strategy for a specific boiler will depend heavily on the location of the facility. For example, a facility may be unable to access lower-sulfur coal supplies or may lack access to a natural gas pipeline. Also, since transportation costs make up a large fraction of the delivered price of coal, the price of low-sulfur coal varies widely across facilities.

Chapter 5, *Boiler Technologies*, includes a more detailed discussion of fuel switching, including emissions reduction and cost estimates.

Boiler Control Technologies

There are two basic categories of technologies that are available to control emissions from industrial and commercial boilers: combustion controls and post-combustion controls.

Combustion controls involve modifications to the combustion process that reduce the formation of NO_x in the boiler. The most common technologies are low-NO_x burners (LNBs) and over fire air (OFA), used either alone or in combination. Flue gas recirculation (FGR) is another common technology, used alone or in combination with LNBs and OFA.

Post-combustion controls involve the application of back-end control systems to capture or convert pollutants before they are released to the air. Post-combustion control technologies are available for all of the pollutants addressed in this report. The most common technologies include selective catalytic reduction (SCR) for NO_x control, flue gas desulfurization (FGD) or scrubbers for SO_2 control, and fabric filters (also known as baghouses) and electrostatic precipitators (ESPs) for PM control. Table 6.4 summarizes the available technologies. Chapter 5, *Boiler Technologies*, also discusses emerging multi-pollutant control technologies. However, these are more likely to be used within the electric generating sector because of state and federal control requirements specific to electric generating units (EGUs).

Table 6.4

Combustion and Post-Control Control Options for Industrial and Commercial Boilers

Control Type	Control Technology	Description	Applications and Commercial Availability
Combustion controls (for NO_x control)	LNBs	Burner configuration that limits NO_x formation by controlling temperature profile in burner zone.	Commercially available for tangential and wall-fired boilers.
	OFA	Diverts some combustion air and reinjects it above burner zone.	Commercially available. Applicable to most boiler types. Must have sufficient furnace height above burners.
	LNBs with OFA	Combination of new burner design and injection of air above main combustion zone.	Available in new boiler designs and can be retrofitted to a variety of boiler types.
	Reburn	Injection of reburn fuel (natural gas, fuel oil or coal) for combustion above the main combustion zone.	Available but not in wide use. Must have sufficient furnace height above burners.
Post-combustion controls	SCR	Injection of ammonia (NH_3) into boiler flue gas; flue gas then passed through a catalyst bed, where the NO_x and NH_3 react to form nitrogen and water vapor.	Available on new boilers and as retrofit.
	SNCR	Injection of NH_3 or urea in the convective pass for NO_x control.	Available on new boilers and as retrofit.
	FGD (Wet scrubber)	Slurry of lime used to absorb SO_2.	Commercially available and in wide use.
	FGD (Spray drying)	Alternative scrubbing technique using calcium hydroxide slurry that vaporizes in spray vessel for SO_2 control.	Commercially available and in wide use. Primarily for low- to medium-sulfur fuels.
	ESP	Use of grounded electrodes to collect particles.	Commercially available and in wide use.
	Fabric filter (or baghouse)	Filtering elements (bags) used to collect particles.	Commercially available and in wide use.

In general, combustion controls involve lower capital investment and operating costs than post-combustion controls but produce more modest emissions reductions. LNBs and OFA can achieve a 40–60 percent reduction in NO_x emissions, while SCR controls can achieve 70–90 percent control. Post-combustion controls often require large capital expenditures with significant on-going operating and maintenance costs. They can also make significant energy demands, requiring a meaningful portion of a facility's electricity output to operate.

To compare the various control options, we have compiled information on the cost-effectiveness of retrofitting controls. In general, cost-effectiveness increases as boiler size and capacity factor (a measure of boiler utilization) increases. For example, EPA estimates a cost-effectiveness of about $1,000 per ton of NO_x removed with the application of SCR controls on a boiler with an 85 percent capacity factor. With a capacity factor of 50 percent, the cost-effectiveness of control rises to about $2,000 per ton of NO_x reduced (Khan, 2003a).

Table 6A.2, at the end of this chapter, summarizes the control-effectiveness and cost-effectiveness of retrofit with NO_x controls for industrial boilers. EPA based its calculations on three different size industrial boilers (100 MMBtu/hr, 250 MMBtu/hr and 1,000 MMBtu/hr), at three different capacity factors. These control options are not appropriate in all situations. Selective non-catalytic reduction (SNCR) can be difficult to apply to industrial boilers that modulate or cycle frequently. SNCR technology injects ammonia (or urea) into the flue gases at specific flue gas temperatures. In a boiler that cycles frequently, the location of the flue gases at the specified temperature will be constantly changing.

Table 6.5 presents similar data for retrofit with SO_2 control technologies on coal- and oil-fired industrial

boilers.[2] In two recent studies of control technologies for industrial boilers, both the Midwest Regional Planning Organization and the Northeast States for Coordinated Air Use Management used these data for their own analyses (MRPO, 2005; NESCAUM, 2005).

Control devices for wood-fired boilers are similar to control devices for fossil fuel-fired boilers. One notable exception is for NO_x controls. As discussed in Chapter 5, *Boiler Technologies*, wood-fired boilers burn at lower combustion temperatures than fossil fuel-fired boilers and have lower thermal NO_x emissions. This is especially true for older and smaller wood-fired boilers. As a result, NO_x combustion control technologies are not applicable to wood-fired boilers. However, post-combustion NO_x controls (such as SCR or SNCR) can be effective on wood-fired boilers that operate at higher temperatures. (Newer and larger wood-fired boilers use technologies that allow for higher combustion temperatures.)

Table 6.6 includes control efficiencies for wood- and combination wood- (hog fuel) fired boilers,[3] as compiled by the Washington State Department of Ecology (DEC) for a Reasonably Available Control Technology (RACT) analysis (Washington DEC, 2005).

Table 6.7 summarizes the Washington DEC's cost-effectiveness data on controls for PM and acid gases (primarily SO_2 and hydrochloric acid), for three sizes of wood-fired boilers: small boilers (<3,500 lb/hr steam), medium size boilers (15,000-45,000 lb/hr steam), and large boilers (>200,000 lb/hr steam). The Department concluded that low-sulfur fuels were RACT for SO_2, and proper boiler

2. More information on the assumptions used to develop these estimates can be found in *Methodology, Assumptions, and References—Preliminary NO_x Controls Cost Estimates for Industrial Boilers*; and *Methodology, Assumptions, and References—Preliminary SO_2 Controls Cost Estimates for Industrial Boilers* (Khan, 2003a; 2003b).

3. Combination wood or hog fuel is a biomass fuel composed of wood products (e.g., bark, wood, sawdust) shredded or ground into half-inch to 6-inch pieces for firing.

Table 6.5

Cost-Effectiveness and Control-Effectiveness of SO_2 Control Options for Retrofit on Industrial Boilers

Fuel	Technology	SO_2 Reduction (%)	Capacity Factor (%)	Cost-Effectiveness ($/Ton of SO_2) 100 MMBtu/hr	250 MMBtu/hr	1,000 MMBtu/hr
High-sulfur coal[a]	FGD (Dry)	40	14	3,543	2,471	1,703
			50	1,292	992	776
			83	943	763	633
Lower-sulfur coal[b]	FGD (Dry)	40	14	4,283	2,952	1,986
			50	1,504	1,131	870
			83	1,075	849	697
Coal	FGD (Spray dry)	90	14	3,920	2,611	1,500
			50	1,209	842	531
			83	790	569	381
High-sulfur coal	FGD (Wet)	90	14	3,513	2,708	1,789
			50	1,046	820	563
			83	664	528	373
Lower-sulfur coal	FGD (Wet)	90	14	4,495	3,460	2,273
			50	1,326	1,036	704
			83	836	661	461
Oil	FGD (Wet)	90	10	10,160	7,801	5,082
			50	2,126	1,654	1,109
			86	1,285	1,011	693

a. Assumes sulfur content = 3.43% and ash content = 12.71%.

b. Assumes sulfur content = 2.0% and ash content = 13.2%.

Source: Khan, 2003b

Table 6.6

Control Efficiencies for Wood-Fired Boilers

	PM Removal Efficiency (%)	SO_2 Removal Efficiency (%)	NO_x Removal Efficiency (%)
Single cyclone	50–90	NA	NA
Multi-cyclones	>90	NA	NA
Dry ESP	95.5–99.9	NA	NA
Wet ESP	95.5–99.9	80–95	NA
Spray dryer w/ESP	98.5–99.9	60–75	NA
Spray dryer w/FF	99.0–99.9	65–80	NA
Dry FGD and ESP	98.5–99.9	60–70	NA
Dry FGD and FF	99.0–99.9	70–80	NA
Spray dryer, dry FGD and FF	99.0–99.9	80–90	NA
ESP and wet FGD (1)	98.5–99.9	50–60	NA
ESP and wet FGD (2)	98.5–99.9	90–95	NA
SCR/SNCR	NA	NA	>90

Source: Washington DEC, 2005

Table 6.7

Cost-Effectiveness for Wood-fired Boilers

Pollutant, Control Technology (Assumed Removal Efficiency)	Boiler Size	Cost-Effectiveness ($/ton removed)
PM, Multi-cyclone (90%)	Small	180
	Medium	64–83
	Large	55–57
PM, Venturi scrubber (90%)	Small	1,229
	Medium	126–306
	Large	40-56
PM, ESP (99.5%)	Small	594
	Medium	203–292
	Large	114–130
PM, Fabric filter (99.5%)	Small	958
	Medium	147–279
	Large	91–107
Acid gas (SO_x and hydrochloric acid), FGD	Small	1,349
	Medium	598–713
	Large	458–479

Source: Washington DEC, 2005

design was RACT for NO_x.

Regulatory Authority

As we have indicated in Chapter 4, the states and federal government share responsibility for regulating criteria pollutants (e.g., PM, SO_2, NO_x) from stationary sources like industrial and commercial boilers. The federal government has more of the burden of regulating toxic air pollutants, but—for both criteria pollutants and air toxics—states and local areas are free under federal law to adopt more stringent standards than the Clean Air Act requires.

This section describes federal Clean Air Act standards for emissions of PM, SO_2 and NO_x from the industrial and commercial boiler sector.

Reasonably Available Control Technology

State Implementation Plans (SIPs) for nonattainment areas must include RACT for new and *existing* sources. RACT standards are non-binding federal guidelines; for volatile organic compounds, the guidelines are known as Control Technique Guidelines (CTGs). Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

There are ACTs applicable to NO_x emissions from industrial and commercial boilers, which can be found at www.epa.gov/ttn/catc/products.html#aptecrpts (EPA, 1994).

SIP writers should bear in mind that although the ACTs may provide a starting point, they are unlikely to represent the actual RACT limits that states will choose to adopt. That is the case because RACT standards are intended to reflect advancements in technology, and most of the ACTs are outdated—some, seriously so. Additionally, the ACTs are guidelines only, not actual federal limits. Even if they were federal limits, many states would have the leeway to enact more stringent standards. And since the ACTs do not set limits, even state prohibitions against enacting more stringent standards may be inapplicable.

New Source Performance Standards

New Source Performance Standards (NSPS) apply to stationary sources in specific source categories, in both attainment and nonattainment areas, when they are constructed, reconstructed or modified.

40 CFR §§60.40b, et seq. (subpart Db) apply to steam generating units that commence construction, reconstruction or modification after June 19, 1984 and that have a heat input capacity greater than 100 MMBtu/hr. The rules include different limits for NOx at facilities that commence construction or reconstruction after July 9, 1997 that have a heat input capacity greater than 100 MMBtu/hr.

40 CFR §§60.40c, et seq. (subpart Dc) apply to steam generating units that commence construction, reconstruction or modification after June 9, 1989 and that have a maximum design heat input capacity of 100 MMBtu/hr or less, but greater than or equal to 10 MMBtu/hr.

On February 27, 2006, EPA published fuel-neutral (except for modified wood) NSPS for emissions from industrial, commercial and institutional steam-generating boilers. The new limits apply to facilities constructed, reconstructed or modified after February 28, 2005 (EPA, 2006).

The NSPS for existing industrial and commercial boilers appear in Table 6A.3, and the NSPS for new boilers appear in Table 6A.3-1, at the end of this chapter.

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, Best Available Retrofit Technology (BART) applies in attainment and nonattainment areas to $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from large sources in 26 identified source categories, that began operation between 1962 and 1997. The source categories include fossil fuel-fired boilers of more than 250 million Btu per hour heat input. However, states have not yet developed their BART rules.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) and Prevention of Significant Deterioration (PSD) apply to new and modified large stationary sources in nonattainment and attainment areas, respectively. The programs subject plants in nonattainment areas to Lowest Achievable Emission Rate (LAER) standards, and in attainment areas to BACT. Neither BACT nor LAER standards are codified, because they are intended to reflect technology advancements as they occur.

Air districts in California have made a number of BACT determinations for PM_{10} and NO_x emissions from natural gas-fired industrial and commercial boilers. These determinations are summarized in Table 6.8.

Examples of recent determinations from EPA's RACT/BACT/LAER Clearinghouse for industrial boilers are

summarized in Table 6A.4, at the end of this chapter.

The BACT determinations in Table 6A.4 show some of the emissions limits that regulators have considered appropriate for industrial boilers during the past ten years. Note that the PM emissions limits for natural gas- and residual oil-fired boilers are based on good combustion practices, while the coal-fired units are required to meet PM_{10} emissions limits lower than 0.05 lb/MMBtu with fabric filters.

Maximum Achievable Control Technology

Maximum Achievable Control Technology (MACT) standards address emissions of hazardous air pollutants (HAPs) from new and existing sources in identified industry categories in both attainment and nonattainment areas. MACT standards apply to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs. In some MACT rules, such as those referenced here, EPA regulates PM as a surrogate for HAPs.

40 CFR §§63.7480, *et seq.* (subpart DDDDD) regulate PM emissions from industrial and commercial boilers. A unit is "existing" unless it commenced construction or reconstruction after January 13, 2003.

The MACT standards for industrial and commercial boilers appear in Table 6.9.

Clean Air Interstate Rule

Under the Clean Air Interstate Rule (CAIR), there are federal trading programs: annual for SO_2, annual for NO_x and seasonal for NO_x. These programs cover electric generating units (EGUs), but a state can include in its seasonal NO_x program large industrial boilers (greater than 250 MMBtu/hr) that are included in the NO_x SIP Call's NO_x trading program, so long as the state adheres to the cap established under the SIP Call program. These are the only non-EGU sources that can be included in any of the federal CAIR trading programs without disqualifying the state from participation in the federal trading regimes.

Table 6.8

BACT Determinations for New Natural Gas-Fired Boilers in California

Capacity (MMBtu/hr)	District	Startup Date	PM_{10} Limit	PM_{10} Control	NO_x Limit[a]	NO_x Control
4.2	South Coast AQMD	6/2000	0.007 lb/MMBtu	ND	2 ppmvd @ 3% O_2 (~0.003 lb/MMBtu)	SCONOx Catalytic Absorption System
5.1	South Coast AQMD	6/1999	ND	ND	20 ppmvd @ 3% O_2 (~0.03 lb/MMBtu)	LNB, FGR
6.2	San Joaquin Valley UAPCD	6/2000	0.007 lb/MMBtu	ND	12 ppmvd @ 3% O_2 (~0.016 lb/MMBtu)	LNB
10.0	South Coast AQMD	4/2003	ND	ND	12 ppmvd @ 3% O_2 (~0.016 lb/MMBtu)	Forced internal recirculation
16.4	South Coast AQMD	2/2000	ND	ND	11 ppmvd @ 3% O_2 (~0.015 lb/MMBtu)	SCR
16.5	South Coast AQMD	ND	ND	ND	7 ppmvd @ 3% O_2 (~0.009 lb/MMBtu)	SCR
20.9	South Coast AQMD	4/2000	ND	ND	9 ppmvd @ 3% O_2 (~0.012 lb/MMBtu)	LNB
21.0	South Coast AQMD	12/2000	ND	ND	7 ppmvd @ 3% O_2 (~0.009 lb/MMBtu)	SCR
21.5	South Coast AQMD	4/2002	ND	ND	9 ppmvd @ 3% O_2 (~0.12Btu)	LNB

ND=no data

ppmvd= parts per million on dry volume basis adjusted to 3% oxygen dry.

a. lb/MMBtu approximated as ppmvd divided by 750.

Source: California Air Resources Board BACT Clearinghouse, 2005

Table 6.9

MACT Standards for Industrial and Commercial Boilers (40 CFR §§63.7480, et seq. (subpart DDDDD))

Source	Category	Size (MMBtu/hr)	PM Limit (lb/MMBtu)
New or reconstructed	Solid Fuel[a]	>10	0.025
		≤10, >10% capacity factor	0.025
	Liquid Fuel[b]	>10	0.03
		≤10, >10% capacity factor	0.03
	Gaseous Fuel[c]	NA	No limit
Existing	Solid Fuel	>10	0.07
	Solid Fuel-Limited Use[d]	>10	0.21
	Liquid Fuel	NA	No limit
	Gaseous Fuel	NA	No limit

NA=Not Applicable

a. Includes coal and wood.

b. Includes residual oil and process liquids.

c. Includes natural gas, process gas and refinery gas.

d. Limited use is defined as having a federally enforceable annual average capacity factor equal to or less than 10%.

State and Local Policy Measures

As discussed in the *Introduction* to this report, state and local agencies are free under federal law to adopt more stringent standards than the Clean Air Act requires, although in some instances state laws limit this authority. SIP writers are presumably familiar with any such constraints under applicable local laws and regulations. The *Introduction* also outlines possible approaches for imposing more stringent state and local standards, such as requiring sources that have not yet been subject to BACT/LAER standards to achieve those limits or imposing air pollution limits on sources that have heretofore been unregulated.

The discussion below summarizes emissions standards for PM, SO_2 and NO_x that state and local officials should consider for industrial and commercial boilers. We also outline other opportunities to reduce emissions from these sources.

Emissions Standards for New and Existing Industrial and Commercial Boilers

In Tables 6A.5—6A.7 , at the end of this chapter, we include summary information on the most stringent PM, SO_2 and NO_x emissions limits for industrial and commercial boilers we were able to identify, in order to highlight the limits that regulators have viewed as feasible under some circumstances. We note federal limits where they are more stringent than any state or local limits we found.

Note that these summary tables include some BACT determinations for *EGUs*, notwithstanding that the table headings refer to *industrial and commercial* boilers. We include these data on account of our recommendation (in the section entitled "*Emissions Standards for New and Existing EGU Boilers,*" below) that where EGU boiler limits are more stringent than any industrial and commercial boiler limits, the EGU limits suggest an appropriate starting point for consideration of limits for industrial and commercial boilers greater than 250 MMBtu/hr and even those greater than 100 MMBtu/hr.

PM. No federal or state rules or permit determinations currently specify $PM_{2.5}$ limits for industrial and commercial boilers, and data on the $PM_{2.5}$ fraction of emissions are limited. Therefore, Table 6A.5, at the end of this chapter, highlights PM limits, on the theory that more stringent PM limits will also result in reduced emissions of $PM_{2.5}$.

Some recent BACT determinations include both filterable and condensable PM,[4] although federal regulatory limits and most BACT determinations have addressed only filterable PM. STAPPA and ALAPCO have commented in rulemakings that PM limits should include both. We discuss this issue further in Chapter 3, *Fine Particulate Matter and Precursor Emissions*.

4. Recent permitting decisions for coal-fired EGUs in Iowa and Wisconsin include both filterable and condensable PM, and set limits of 0.025 lb/MMBtu and 0.018 lb/MMBtu, respectively (STAPPA and ALAPCO, 2005).

SO_2. Note, as discussed below, that state permits for industrial and commercial coal- and oil-fired boilers can limit the sulfur content of fuels as an alternative means of limiting SO_2 emissions. Table 6A.6, at the end of this chapter, highlights SO_2 limits.

NO_x. The San Joaquin Valley Unified Air Pollution Control District (UAPCD) and the New Jersey Department of Environmental Protection (DEP) State of the Art (SOTA) Manual for Boilers and Process Heaters set the highest bar for NO_x emissions from natural gas- and oil-fired industrial and commercial boilers. We provide more detail on their limits in the sections that follow. Also, the emissions limits imposed by the San Joaquin Valley UAPCD and the New Jersey SOTA cover units much smaller than the federal NSPS, which apply to units greater than 100 MMBtu/hr. Table 6A.7, at the end of this chapter, highlights NO_x limits.

Additional Information on California and New Jersey Standards

The San Joaquin Valley UAPCD has set California's most stringent limits for NO_x emissions from industrial and commercial boilers, steam generators and process heaters. The rules apply to units with a heat input greater than 0.075 MMBtu/hr, as described in Table 6A.8 at the end of this chapter.

In 2004, the New Jersey DEP updated its SOTA Manual for Boilers and Process Heaters referenced above. To do so, DEP reviewed recent permits issued in New Jersey and other states, state and federal regulations, BACT data from the Bay Area Air Quality Management District, BACT data from the California Air Resource Board, and data from EPA's RACT/BACT/LAER Clearinghouse.

Table 6.10 shows New Jersey's SOTA performance levels for NO_x emissions from newly constructed, reconstructed and modified natural gas- and distillate oil-fired (note—only distillate, and not residual oil) industrial and utility boilers and process heaters. New Jersey DEP notes that it evaluates boilers fired with other fuels (e.g., coal or wood) on a case-by-case basis.

The values in Table 6.10 are an order of magnitude more stringent than the NSPS for NO_x emissions from natural gas- and distillate oil-fired boilers. However, with the exception of distillate oil-fired boilers greater than 75 MMBtu/hr, they are not as stringent as the San Joaquin Valley UAPCD limits for natural gas-fired boilers.

Emissions Standards for New and Existing EGU Boilers

In comments on proposed changes to the NSPS for EGUs and industrial and commercial boilers, STAPPA and ALAPCO take the position that EPA should apply the same limits to large industrial and commercial boilers (greater than or equal to 250 MMBtu/hr) as it applies to EGUs (STAPPA and ALAPCO, 2005). The Associations further advocated for more stringent SO_2 and NO_x emissions than EPA was proposing, and for the inclusion of both filterable and condensable PM in establishing PM limits. (For a summary of the proposed NSPS for EGUs, see Chapter 7, *Electric Generating Units*.)

Although there may be reasons in individual cases why EGU boiler limits are not feasible for industrial and commercial boilers, the EGU boiler standards suggest an appropriate starting point for consideration of limits for industrial and commercial boilers greater than 250 MMBtu/hr and even those greater than 100 MMBtu/hr.

Regional CAIR "Plus" Approach

Another alternative for states to consider is supporting the Ozone Transport Commission (OTC) Multi-Pollutant Position, or similar programs that other regions may develop, aimed at achieving NO_x, SO_2 and mercury

Table 6.10

New Jersey SOTA NO_x Levels for Natural Gas- and Distillate Oil-Fired Boilers and Process Heaters

Fuel	Capacity (MMBtu/hr)	Controlled NO_x Emissions Level (lb/MMBtu)	Control Technology
Natural gas	10 and <50	0.0350	LNB with FGR or ultra LNB
	≥50 and <75	0.0200	LNB with FGR or ultra LNB
	≥75	0.010	LNB with FGR and/or SCR
Distillate oil	≥10 and <75	0.0600	LNB with FGR
	≥75	0.0300	LNB with FGR and SCR

Source: NJ DEP, 2004

emissions reductions from large industrial boilers, as well as from EGUs. (The EGU reductions would be deeper and come sooner than under CAIR.) The OTC's approach is to use a regional partnership and a model rule to implement a program that goes beyond CAIR. The OTC would use the basic structure of CAIR to implement tighter emissions caps. The OTC aims to develop regional partnerships and a model rule in 2006, followed by a first phase of controls in 2009 and a second phase in 2012 (InsideEPA, 2006). The support of states outside the OTC for such efforts could increase the likelihood of other regional programs.

Fuel Sulfur Limits

In addition to the stack emissions limits described above, state and local agencies have opted in some cases to regulate the concentrations of sulfur in fuel, for example, by regulating the sulfur content of fuel oil.

In 2000, the Connecticut Department of Environmental Protection (DEP) adopted regulations limiting the sulfur content of fuel oil used by power plants (CT DEP, 2000). Effective January 1, 2002, affected units that combust liquid or gaseous fuels are subject to a fuel sulfur limit of 0.5 percent by weight. Effective January 1, 2003, the fuel sulfur limit is reduced to 0.3 percent by weight. Alternatively, an affected source can meet an average SO_2 emissions rate of 0.33 lb/MMBtu. The DEP Commissioner can suspend the fuel sulfur limits in the event of a fuel supply emergency, in which case the Commissioner can require affected sources to offset excess emissions with SO_2 allowances.

The New York State Department of Environmental Conservation regulates the sulfur content of oil and coal used by stationary combustion sources (NY DEC, 2005). The sulfur limits vary by area within the state. The lowest limits apply in New York City: (1) 0.30 percent sulfur by weight for residual oil, (2) 0.20 percent sulfur by weight for distillate oil and (3) 0.2 lb of sulfur per MMBtu gross heat content for solid fuels. As in Connecticut, the Commissioner has the authority to suspend the standards in the event of a supply shortage.

Limits on Back-Up Fuels

Over the past two decades, state and local officials have approved a large number of natural gas-fired boilers. According to an official in the Puget Sound Clean Air Agency, applicants frequently list oil as a back-up fuel. The official noted that as the cost of natural gas increases, operators might switch from natural gas to oil (Van Slyke, 2006). Since oil emissions limits are likely less stringent than natural gas emissions limits, this fuel switching negatively impacts air quality. As a result, it is important for state and local officials to evaluate the emissions impacts of both primary and back-up fuels during the review process.

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Table 6A.1

Industrial Boilers in the U.S. by Fuel Type and Capacity[a,b]

Primary Fuel	Capacity Range (MMBtu/hr)	Average Capacity Factor (MMBtu/hr)	Estimated Total Capacity (MMBtu/hr)	% of Industrial Boiler Sector Capacity[c]	Percent of Capacity by Fuel	Number of Industrial Boilers	% of Total Industrial Boiler Sector Boilers[c]	% of Industrial Boilers by Fuel
Coal	0-10	3	358	<1%	<1%	139	<1%	5%
	10-100	53	63,231	3%	12%	1,188	3%	44%
	100-250	172	133,877	7%	25%	778	2%	29%
	>250	583	346,320	19%	64%	594	1%	22%
Subtotal	**-**	**201**	**543,786**	**29%**	**100%**	**2,699**	**6%**	**100%**
Residual oil	0-10	3	1,908	<1%	1%	636	1%	25%
	10-100	37	53,946	3%	28%	1,458	3%	57%
	100-250	172	56,244	3%	29%	327	1%	13%
	>250	547	80,956	4%	42%	148	<1%	6%
Subtotal	**-**	**75**	**193,054**	**10%**	**100%**	**2,569**	**6%**	**100%**
Natural gas	0-10	3	61,467	3%	7%	20,462	48%	62%
	10-100	33	355,554	19%	39%	10,738	25%	33%
	100-250	164	201,240	11%	22%	1,224	3%	4%
	>250	510	291,644	16%	32%	572	1%	2%
Subtotal	**-**	**28**	**909,905**	**49%**	**100%**	**32,996**	**77%**	**100%**
Wood	0-10	5	1,018	<1%	1%	201	<1%	24%
	10-100	31	13,868	1%	18%	450	1%	53%
	100-250	177	17,851	1%	23%	101	<1%	12%
	>250	500	45,988	2%	58%	92	<1%	11%
Subtotal	**-**	**93**	**78,725**	**4%**	**100%**	**844**	**2%**	**100%**
Other	>0	-	82,088	7%		3,592	9%	
TOTAL	**-**	**-**	**1,807,558**	**100%**	**-**	**42,700**	**100%**	**-**

a. This table is based on data supplied in support of the Industrial/Commercial/Institutional Boiler and Process Heaters National Emission Standards for Hazardous Air Pollutants (EPA, 2004). Eastern Research Group (ERG) used the Industrial Combustion Coordinated Rulemaking (ICCR) Inventory and Survey data to populate a new database of facilities separated by fuel type, capacity and control technology.

b. The database used as a source for this data combined different fuel types to fit into general categories. For example, the "natural gas" fuel in this table includes other gases such as landfill gas, blast furnace gas, propane and butane. Based on an independent industrial boiler population analysis (EEA, 2005), it appears that the vast majority of the boilers listed as "natural gas" in this table are natural gas-fired boilers.

c. Smaller sectors (e.g., distillate oil) are not included in this table, but are included in the total used to calculate percent by sector. As a result, the percent calculations do not add to 100%.

Source: Alvis, 2002

Table 6A.2

Cost-Effectiveness and Control-Effectiveness of Combustion and Post-Combustion NO_x Control Options for Retrofit on Industrial Boilers

Fuel	Technology	NO_x Reduction (%)	Capacity Factor (%)	Cost-Effectiveness ($/Ton NO_x)		
				100 MMBtu/hr	250 MMBtu/hr	1,000 MMBtu/hr
Subbituminous coal	LNB	51	14	3,033	2,304	1,520
			50	849	645	426
			83	512	389	256
	LNB/OFA	65	14	3,428	2,608	1,727
			50	972	743	496
			83	593	454	306
Bituminous coal	LNB/OFA	51	14	4,358	3,317	2,197
			50	1,239	947	634
			83	757	581	392
Coal	SCR	80	14	7,262	5,924	4,481
			50	2,141	1,766	1,359
			83	1,349	1,123	876
	SNCR	40	14	4,970	4,015	2,962
			50	2,073	1,814	1,510
			83	1,625	1,473	1,285
Oil	LNB/OFA (0.5 lb/MMBtu inlet NO_x)	30	10	5,260	3,986	2,630
			50	1,052	797	526
			86	612	464	306
	LNB/OFA/FGR (0.5 lb/MMBtu inlet NO_x)	50	10	4,973	3,790	2,505
			50	1,028	791	533
			86	615	477	326
	LNB/OFA/FGR (0.36 lb/MMBtu inlet NO_x)	30	10	11,303	8,613	5,694
			50	2,337	1,798	1,210
			86	1,399	1,085	741
	SCR (0.36 lb/MMBtu inlet NO_x)	80	5	25,838	20,113	14,601
			50	2,767	2,178	1,622
			86	1,694	1,343	1,017
	SCR (0.5 lb/MMBtu inlet NO_x)	80	5	18,544	14,443	10,458
			50	2,014	1,595	1,191
			86	1,245	997	760
	SNCR (0.5 lb/MMBtu inlet NO_x)	40	10	7,399	5,892	4,271
			50	2,367	2,070	1,749
			86	1,840	1,670	1,485
	SNCR (0.36 lb/MMBtu inlet NO_x)	40	10	9,842	7,753	5,497
			50	2,853	2,444	1,995
			86	2,123	1,889	1,628

Table 6A.2, continued

Cost-Effectiveness and Control-Effectiveness of Combustion and Post-Combustion NO_x Control Options for Retrofit on Industrial Boilers

Fuel	Technology	NO_x Reduction (%)	Capacity Factor (%)	Cost-Effectiveness ($/Ton NO_x)		
				100 MMBtu/hr	250 MMBtu/hr	1,000 MMBtu/hr
Natural gas	LNB/OFA	60	5	10,521	7,973	5,260
			50	1,052	797	526
			94	559	424	280
	LNB/OFA/FGR	80	5	12,374	9,415	6,204
			50	1,278	981	656
			94	700	543	368
	SCR	80	5	26,859	21,095	14,815
			50	2,933	2,330	1,670
			94	1,689	1,354	986
	SNCR	40	5	27,105	20,870	14,165
			50	3,735	3,116	2,452
			94	2,521	2,193	1,842

Source: Khan, 2003a

Table 6A.3

NSPS for Industrial and Commercial Boilers (40 CFR §§60.40b, *et seq.* (subpart Db), and 40 CFR §§60.40c, *et seq*. (subpart Dc))

Fuel Type	Capacity (MMBtu/hr)	PM Limit	SO_2 Limit	NO_x Limit
Coal	10 - 30	NA	90% reduction from potential SO_2 emissions levels and SO_2 emissions rate less than or equal to 1.2 lb/MMBtu (assuming 100% coal)[a]	NA
Coal	30 - 100	0.051 lb/MMBtu (assuming 100% coal); no more than 20% opacity (6-minute average)		
Coal[b]	>100	0.051 lb/MMBtu (assuming 100% coal)	90% reduction from potential SO_2 emissions levels or SO_2 emissions rate less than or equal to 1.2 lb/MMBtu (assuming 100% coal)	0.5-0.8 lb/MMBtu depending on coal and boiler type 0.3 lb/MMBtu for facilities commencing construction or reconstruction after July 9, 1997[c]
Residual oil	10 - 100	No more than 20% opacity (6-minute average)	0.50 lb/MMBtu or combust oil 0.5% sulfur by weight or less	NA
Residual oil	>100	0.1 lb/MMBtu	90% reduction from potential SO_2 emissions levels or SO_2 emissions rate less than or equal to 0.8 lb/MMBtu (assuming 100% oil)	0.30 lb/MMBtu (low heat release), 0.40 lb/MMBtu (high heat release)[d] 0.3 lb/MMBtu for facilities commencing construction or reconstruction after July 9, 1997
Natural gas	10 - 100	NA	NA	NA
Natural gas	>100	NA	NA	0.10 lb/MMBtu (low heat release), 0.40 (high heat release) 0.3 lb/MMBtu for facilities commencing construction or reconstruction after July 9, 1997[e]
Wood	>30	0.1 lb/MMBtu, nothing greater than 20% opacity (6-minute average, capacity factor greater than 30%)	NA	NA
Wood	>100	0.1 lb/MMBtu (capacity factor greater than 30%)	NA	NA

a. Alternate SO_2 limits exist for facilities using fluidized bed combustion (with coal) or an emerging technology for the control of SO_2 emissions.
b. There are exceptions to standards for coal- and oil-fired units greater than 100 MMBtu/hr that commenced construction, reconstruction or modification after June 19, 1984 but on or before June 19, 1986. There are further exceptions to the PM limits for coal-fired units that commenced construction after June 19, 1984 but on or before November 25, 1986.
c. The 1997 limits for coal-, residual oil- and natural gas-fired boilers greater than 100 MMBtu/hr include exceptions for affected facilities with annual capacity factors for coal, residual oil or natural gas of 10% or less.
d. The heat release points are defined by fuel and steam generating unit type in 40 CFR §60.44b.
e. If a natural gas-fired boiler has a low heat release rate and combusts natural gas in excess of 30% of the heat input, the limit is determined by an equation in 40 CFR §60.44b.

Table 6A.3-1

NSPS for Industrial and Commercial Boilers Constructed, Reconstructed or Modified after February 28, 2005

Fuel Type	Capacity (MMBtu/hr)	PM Limit[a]	SO_2 Limit
Coal, oil, natural gas, wood	30–100	0.03 lb/MMBtu	No change (see Table 6A.3)
	>100	0.03 lb/MMBtu	0.20 lb/MMBtu[b]

a. Modified wood-fired units greater than 30 MMBtu/hr and less than or equal to 250 MMBtu/hr have a limit of 0.10 lb/MMBtu. Modified units greater than 250 MMBtu/hr have a limit of 0.085 lb/MMBtu.

b. Units burning only oil that contains no more than 0.3 percent sulfur by weight or any idividual fuel that, when combusted without SO_2 emissions control, have an SO_2 emissions rate equal to or less than 0.32 lb/MMBtu, are exempt from other SO_2 emissions limits.

Source: EPA, 2006

Table 6A.4

Recent BACT Determinations for Industrial and Commercial Boilers Posted in the RACT/BACT/LAER Clearinghouse

Fuel (Basis)	Throughput MMBtu/hr	RBLC ID[a]	Permit Date (Type)	PM		SO_x		NO_x	
				Limit	Control	Limit	Control	Limit	Control
Natural gas (BACT/PSD)	80	WI-0207	1/2004 (New)	0.0075 lb/MMBtu	Good combustion	ND		0.04 lb/MMBtu	LNB
Lignite (BACT/PSD)	250	ND-0020	8/2004 (New)	0.048 lb/MMBtu (PM_{10}, 0.02 lb/MMBtu)	Fabric Filter	0.09 lb/MMBtu	FGD	0.1 lb/MMBtu	SNCR
Wood (BACT/PSD)	230	MN-0058	6/2005 (New)	PM_{10}, 0.025 lb/MMBtu	ESP	ND		0.15 lb/MMBtu	SNCR
Oil (BACT/PSD)	50.2	WA-0313	11/2003 (New)	0.02 lb/MMBtu	Reduced Hours	ND		0.09 lb/MMBtu	LNB, FGR, Atomization
Natural gas (BACT/PSD)	34	IN-0108	11/2003 (Modified)	PM_{10}, 0.0076 lb/MMBtu	ND	0.0006 lb/MMBtu	ND	ND	
Natural gas (BACT/PSD)	22	AR-0077	7/2004 (New)	PM_{10}, 0.0076 lb/MMBtu	ND	0.0006 lb/MMBtu	ND	ND	
Oil (BACT/PSD)	94	NJ-0046	3/1996 (Modified)	ND		0.32 lb/MMBtu	ND	0.28 lb/MMBtu	LNB
Coal (BACT/PSD)	249	NC-0092	5/2001 (Modified)	0.16 lb/MMBtu	Multicyclone and wet scrubber	0.8 lb/MMBtu	Wet scrubber	0.4 lb/MMBtu	Good combustion
Coal (BACT/PSD)	238	OH-0241	5/2004 (Modified)	PM_{10}, 0.031 lb/MMBtu	Fabric Filter	1.6 lb/MMBtu	No Control	0.7 lb/MMBtu	OFA
Coal (BACT/PSD)	146.7	VA-0267	8/2001 (New)	PM_{10}, 0.02 lb/MMBtu	Fabric Filter	1.6 lb/MMBtu	FGD	0.25 lb/MMBtu	Low excess air, staged combustion
Residual oil (BACT/PSD)	150	VA-0278	3/2003 (Modified)	PM_{10}, 0.05 lb/MMBtu	Good combustion practices	0.52 lb/MMBtu	Low-sulfur fuel	0.4 lb/MMBtu	LNB

ND = No Data

a. RBLC ID = RACT/BACT/LAER Clearinghouse identification number.

Source: EPA, 2005b

Table 6A.5

Summary of Most Stringent PM Limits for New and Existing Industrial and Commercial Boilers in the U.S.

Fuel Type	Boiler Size (MMBtu/hr)	Limit (lb/MMBtu)	Source [a]
Coal	<10	0.025 (New)	Federal MACT
	10-250	0.025 (New), 0.07 (Existing)	Federal MACT
	>250	0.015 (New), 0.07 (Existing)	New: BACT determinations for coal-fired EGUs (WY-0057, PR-0007, IA-0051); Existing: Federal MACT
Residual oil	0.075-10	0.03 (New)	Federal MACT
	10-100	0.02 (New)	Washington State BACT determination (WA-0313)
	100-250	0.03 (New)	Federal MACT
	>250	0.015 (New)	BACT determinations for coal-fired EGUs (WY-0057, PR-0007, IA-0051)
Natural gas	0.4-5.0	0.007 (New)	South Coast AQMD BACT Determination
	>5.0	0.007 (New); 0.0076 (Existing)	New: South Coast AQMD BACT determination; Existing: Indiana BACT determination (IN-0108)
Wood	<10	0.025 (New)	Federal MACT
	10-250	0.025 (New)	Federal MACT
	>250	0.015 (New)	BACT determinations for coal-fired EGUs (WY-0057, PR-0007, IA-0051)

a. RBLC ID (RACT/BACT/LAER Clearinghouse identification number) included where applicable.

Table 6A.6

Summary of Most Stringent SO_2 Limits for New and Existing Industrial and Commercial Boilers in the U.S.

Fuel Type	Boiler Size (MMBtu/hr)	Limit (lb/MMBtu)	Source[a]
Coal	10-100	1.2 (New)	Federal NSPS
	100-250	0.16 (New), 0.8 (Existing)	New: Virginia BACT determination (VA-0267)a; Existing: North Carolina BACT determination (NC-0092)
	>250	0.10 (New)	BACT determinations for coal-fired EGUs (IA-0067, WY-0057)
Residual oil	10-100	0.32 (New, Existing)	New Jersey BACT determination (NJ-0046)
	100-250	0.8 (New, Existing)	New: Federal NSPS; Existing: North Carolina BACT determination (NC-0092)
	>250	0.8 (New)	Federal NSPS
Natural gas	>5.0	0.0006 (New, Existing)	New: Arkansas BACT determination (AR-0077); Existing: Indiana BACT determination (IN-0108)
Wood	>250	0.10 (New)	BACT determinations for coal-fired EGUs (IA-0067, WY-0057)

a. RBLC ID (RACT/BACT/LAER Clearinghouse identification number) included where applicable.

Table 6A.7

Summary of Most Stringent NO_x Limits for New and Existing Industrial and Commercial Boilers in the U.S.

Fuel Type	Boiler Size (MMBtu/hr)	Limit (lb/MMBtu)	Source [a]
Coal	100-250	0.3 (New), 0.4 (Existing)	New: Federal NSPS; Existing: North Carolina BACT determination (NC-0092)
	>250	0.07-0.08 (New)	BACT determinations for coal-fired EGUs (IA-0067, KS-0026, WY-0057, MO-0050, IA-0051)
Residual oil	0.075-0.4	0.093 (New, Existing)	San Joaquin Valley UAPCD
	0.4-2.0	0.036 (New, Existing)	San Joaquin Valley UAPCD
	2.0-250	0.052 (New, Existing)	San Joaquin Valley UAPCD
	>250	0.03 (New), 0.052 (Existing)	New: New Jersey SOTA; Existing: San Joaquin Valley UAPCD
Natural gas	0.075-0.4	0.093 (New, Existing)	San Joaquin Valley UAPCD
	0.4-5.0	0.036 (New, Existing)	San Joaquin Valley UAPCD
	>5.0	0.007 (New, Existing)	San Joaquin Valley UAPCD
Wood	100-250	0.15 (New)	Minnesota BACT determination for wood-fired EGU boiler (MN-0058)
	>250	0.07-0.08 (New)	BACT determinations for coal-fired EGUs (IA-0067, KS-0026, WY-0057, MO-0050, IA-0051)

a. RBLC ID (RACT/BACT/LAER Clearinghouse identification number) included where applicable.

Table 6A.8

San Joaquin Valley UAPCD NO_x Levels for Gaseous Fuel- and Liquid-Fired Boilers, Steam Generators and Process Heaters

Fuel[a]	Capacity (MMBtu/hr)	Units Covered	Controlled NO_x Emissions Level (lb/MMBtu)
Gaseous fuel	≥0.075 and ≤0.4	All units supplied, sold, installed or solicited for installation on and after January 1, 2007	0.093 lb/MMBtu
	>0.4 and <2.0	All units supplied, sold, installed or solicited for installation on and after January 1, 2007	0.036 lb/MMBtu
	≥2.0 and ≤5.0	All units; owners must register units by January 1, 2008	0.036 lb/MMBtu (30 ppmv)
	>5 MMBtu/hr	Units with a rated heat input equal to or less than 20.0 MMBtu/hour	Standard option: 0.018 lb/MMBtu (15 ppmv); enhanced option: 0.011 lb/MMBtu (9 ppmv)[b]
	>5 MMBtu/hr	Units with a rated heat input greater than 20.0 MMBtu/hour	Standard option: 0.011 lb/MMBtu (9ppmv); enhanced option: 0.007 lb/MMBtu (6 ppmv)
Liquid fuel	≥0.075 and ≤0.4	All units supplied, sold, installed or solicited for installation on and after January 1, 2007	0.093 lb/MMBtu
	>0.4 and <2.0	All units supplied, sold, installed or solicited for installation on and after January 1, 2007	0.036 lb/MMBtu
	≥2.0 and ≤5.0	All units, owners must register units by January 1, 2008	0.052 lb/MMBtu (40 ppmv)
	>5 MMBtu/hr	All units, must be in full compliance by December 1, 2008	0.052 lb/MMBtu (40 ppmv)

All limits are at 3% stack gas oxygen by volume.

ppmv = parts per million on dry volume basis.

a. The rule specifies different limits for oilfield steam generators; refinery units with a rated heat input greater than 5 MMBtu/hr; load-following units; units limited by a permit to operate to an annual heat input of 9 billion Btu/year to 30 billion Btu/year; and units in which the rated heat input of each burner is less than or equal to 5 MMBtu/hr but the total rated heat input of all the burners in a unit is great than 5 MMBtu/hr.

b. Under the standard option, qualifying units must be in full compliance by June 1, 2007. Under the enhanced option, units must be in compliance by December 1, 2008.

Source: San Joaquin Valley UAPCD, 2005a, 2005b, 2005c

Chapter 7

Electric Generating Units

Introduction

Electric generating units (EGUs) in the U.S. produced 3.953 trillion kilowatt-hours (trillion kWh) of electricity in 2004.[1] Sources that combust coal, oil or natural gas provided 70 percent of that amount (EIA, 2005a). EGUs that use combustion processes include large steam boiler/generators, combustion turbines and stationary internal combustion (IC) engines. Combustion-based EGUs account for 39 percent of total fossil fuel consumption in the U.S. and are the largest industrial point source contributors of fine particulate ($PM_{2.5}$), sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions (EIA, 2005b; EPA, 2005a).

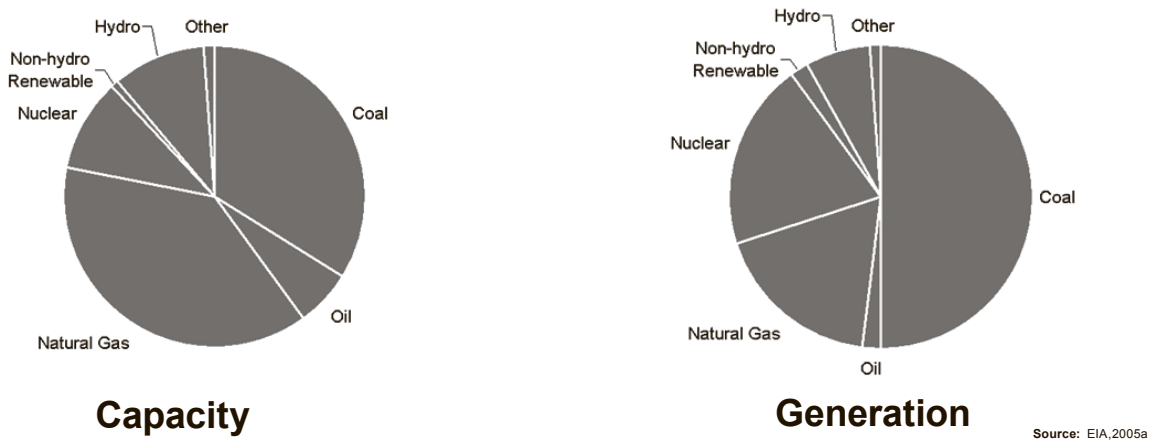
The size, technology, fuel and operating characteristics of EGUs are largely dependent on the purpose they are designed to serve in meeting electricity demand. Large pulverized coal boilers, for example, are designed primarily to operate as base load generating facilities run at high capacity factors. (Capacity factor is a measure of plant utilization; these plants run at capacity factors of 70–90 percent.) Simple cycle combustion turbines and IC engines, on the other hand, can be quickly ramped up and down to meet changing electricity demand and are generally operated as peaking facilities. The average size of a coal-fired boiler in the U.S. is over 200 megawatts (MW) while the average size of a simple cycle natural gas turbine is 50 MW, and that of an IC engine is only about 2 MW (EIA, 2003).

As discussed below, large coal-fired boilers are responsible for the vast majority of emissions from the electric generating sector and are subject to a multitude of existing federal and state programs to reduce emissions. Peaking facilities tend to produce significantly less total air emissions, and in some cases may not be subject to the same regulations or pollution control requirements because of their smaller capacity.

This chapter discusses $PM_{2.5}$, SO_2 and NO_x emissions from EGUs, with an emphasis on technical and policy approaches for reducing emissions. The discussion

1. A kWh is a unit of energy equivalent to one kilowatt of power expended over one hour. A 100-watt bulb burning for ten hours will use one kilowatt hour of energy.

Fig. 7.1 U.S. Generating Capacity and Generation by Fuel Type



Source: EIA,2005a

highlights the importance of generator technology and fuel choice in determining direct $PM_{2.5}$ and $PM_{2.5}$-precursor emissions as well as the control technology options available for different types of EGUs. The chapter includes discussion of the emissions, current regulatory treatment and control options for large pulverized coal and other steam boilers, combustion turbines and IC engines used at electric generating facilities. We conclude by outlining a series of measures that state and local governments could pursue to achieve further emissions reductions.

The discussion provides a general review of control technology options, but it does not duplicate the more detailed discussion of emissions control opportunities for EGU boilers provided in Chapter 5, *Boiler Technologies*.

Sector Profile

The electric power sector in the U.S. is based on a familiar model, one that is replicated throughout the world. Large central power stations, often in remote locations, spin generators to feed a network of high-voltage transmission lines. These carefully synchronized generators are generally powered by coal, natural gas, oil, water or nuclear fission. The electricity pulsing through the transmission grid feeds a system of low-voltage distribution lines, which connect to homes and businesses. Turning on a reading lamp or computer connects the appliance to this network of wires, drawing electricity from the grid. Power plant operators are continually responding to these changes in the system, increasing or decreasing supply to match demand in a carefully orchestrated exercise managed by a central authority or dispatcher.

There are more than 17,000 EGUs in the U.S., with a combined generating capacity of about 950 gigawatts (GW). Of this capacity, 78 percent (over 700 GW) is fueled by the combustion of fossil fuels, including 313 GW of coal, 59 GW of oil and 355 GW of natural gas (EIA, 2003). In 2004, the industry generated almost four trillion kWh of electricity, including 2.78 trillion kWh (70 percent) from fossil fuel combustion (EIA, 2005a). Figure 7.1 illustrates U.S. generating capacity and generation in 2004 by fuel type.

As Figure 7.1 illustrates, the share of generating capacity and generation are different across the different fuel types. For example, coal-fired power plants produced 50 percent of U.S. electricity in 2004, while accounting for only about 34 percent of capacity. In contrast, natural gas-fired power plants, which account for about 38 percent of capacity, generated less than 20 percent of the country's electricity. This dichotomy is reflective of the higher capacity utilization of coal-fired power plants, primarily operated to serve base load, as compared to natural gas facilities, which tend to operate as load following or peaking units. The average capacity factor across all coal-fired power plants in 2003 was 72 percent, *versus* 21 percent for natural gas and 20 percent for oil-fueled sources (EIA, 2003; EIA, 2004a).

Different technologies are used to generate electricity, depending on the energy source. Table 7.1 shows that steam boilers are fueled by a variety of energy sources, but that coal accounts for the largest segment. Natural gas, on the other hand, is the dominant fuel used for turbines operating in a combined cycle mode (i.e., that include a heat recovery steam generator to provide a second generation cycle) and for simple cycle turbines without

Table 7.1

U.S. Generating Capacity by Fuel Type and Prime Mover Technology (2003, Megawatts)

		Prime Mover Technology								
		Steam Boilers	Combined Cycle Turbines	Simple Cycle Turbines	Hydro-electric	IC Engine	Wind/ Solar/ Fuel Cell	Pumped Storage	Other	Total
Fuel Type	Coal	312,453	566							**313,019**
	Natural gas	98,826	145,674	108,770		2,102			119	**355,492**
	Nuclear	99,209								**99,209**
	Oil	34,017	1,396	18,161		4,913			22	**58,509**
	Other	7,908	247	2,430		8			59	**10,651**
	Renewable	5,449	110	152		585	6,004		49	**12,351**
	Water				78,694			20,522		**99,216**
	Total	**557,863**	**147,993**	**129,513**	**78,694**	**7,608**	**6,004**	**20,522**	**249**	**948,446**

Source: EIA, 2003

a steam cycle.[2] Oil (which includes diesel fuel) and to a lesser extent natural gas are used to fuel stationary IC engines (e.g., piston and diesel engines).

Emissions

The electric power sector is the single largest source of $PM_{2.5}$, SO_2 and NO_x emissions among all the major point source categories. Nationwide, EGUs account for nearly 70 percent of SO_2 emissions, more than 20 percent of NO_x emissions and almost 10 percent of $PM_{2.5}$ emissions. Excluding fugitive dust, forest wildfires and agricultural fires (i.e., the miscellaneous categories), EGUs account for more than 20 percent of $PM_{2.5}$ emissions nationwide (EPA, 2005b).

Within the EGU sector, coal-fired power plants account for the vast majority of $PM_{2.5}$, SO_2 and NO_x emissions (EPA, 2005b). Table 7.2 presents 2002 EGU emissions by fuel type. In 2002, coal-fired facilities were responsible for 92, 95 and 87 percent of EGU emissions of $PM_{2.5}$, SO_2 and NO_x, respectively.

Coal combustion dominates EGU emissions because it accounts for 70 percent of fossil fuel-fired electric generation and because coal-fired facilities tend to have relatively high emissions rates, particularly compared to facilities firing natural gas. The higher emissions rates stem from the relatively high concentrations of sulfur, nitrogen and ash in coal, which is converted to PM, SO_2 and NO_x during combustion. The formation of these pollutants as a result of coal, oil and natural gas combustion is described briefly below.

PM. Uncontrolled PM emissions from coal- and oil-fired boilers include the ash from combustion of the fuel and the unburned carbon resulting from incomplete combustion. In pulverized coal boilers, the combustion process is almost complete, so the majority of the PM emitted is composed of inorganic ash residues (EPA, 1998a). The average ash content of bituminous coal consumed by the electric power industry in 2003 was 10.1 percent, while the average ash content of subbituminous coal was 6.4 percent and of lignite 13.3 percent (EIA, 2004b). For fuel oil, PM emissions depend predominantly on the grade of fuel. Combustion of lighter distillate oils results in significantly less PM formation than does combustion of heavier residual oils. Because natural gas is a gaseous fuel, PM emissions resulting from its combustion are typically very low (EPA, 1995).

NO_x. NO_x formation results from the thermal fixation of atmospheric nitrogen in the combustion flame ("thermal NO_x") and from oxidation of nitrogen bound in the fuel ("fuel NO_x"). Bituminous and subbituminous coals usually contain from 0.5 to 2 percent nitrogen by weight. Fuel nitrogen can account for as much as 80 percent of total NO_x emissions from coal combustion (EPA, 1995). Fuel NO_x is also the more important NO_x-forming mechanism in residual oil boilers, but thermal NO_x is the dominant NO_x-forming mechanism in units firing distillate oils and the only mechanism in natural gas combustion systems (EPA, 1995).

2. The combined cycle turbines in Table 7.1 operated on coal are two integrated gasification combined cycle (IGCC) power plants (in Wabash, Indiana and Polk County, Florida) that use gasification technology to convert coal into synthesis gas that can be used to fuel a combined cycle power block.

Table 7.2

EGU Emissions by Fuel Type (2002)

Fuel Type	$PM_{2.5}$		SO_2		NO_x	
	Thousand Tons/Year	Percent of Sector Emissions	Thousand Tons/Year	Percent of Sector Emissions	Thousand Tons/Year	Percent of Sector Emissions
Coal	**535**	**92%**	**9,738**	**95%**	**4,097**	**87%**
bituminous	419	72%	7,317	71%	2,635	56%
subbituminous	95	16%	1,949	19%	1,296	28%
anthracite & lignite	22	4%	465	5%	163	3%
Oil	**13**	**2%**	**343**	**3%**	**130**	**3%**
residual	12	2%	330	3%	121	3%
distillate	1	0%	13	0%	8	0%
Natural Gas	**11**	**2%**	**8**	**0%**	**270**	**6%**
Other	**8**	**1%**	**191**	**2%**	**50**	**1%**
Internal Combustion	**14**	**2%**	**13**	**0%**	**152**	**3%**
Total	**582**	**100**	**10,293**	**100**	**4,700**	**100**

Source: EPA, 2005b

SO_2. SO_2 compounds form when sulfur in coal or oil is oxidized during combustion. The average sulfur content of bituminous coal delivered to EGUs in 2003 was 1.49 percent, while the average sulfur content of subbituminous coal was 0.37 percent and of lignite 0.94 percent (EIA, 2004b). Fuel oil sulfur content depends on grade, but is generally less than one percent. Natural gas has virtually no sulfur and therefore produces almost no SO_2 when combusted. About 95 percent of the sulfur present in fossil fuels is emitted as gaseous SO_2 when combusted (EPA, 1995).

Table 7.3 shows the emissions rates of uncontrolled EGU pulverized coal boilers, along with those of oil- and natural gas-fired combustion turbines and oil- and natural gas-fired IC engines. It should be noted that the uncontrolled fuel *input* emissions rates of natural gas-fired combined cycle power plants are the same as the uncontrolled emissions rates shown for simple cycle turbines because the same combustion occurs in both systems. However, on an output basis (e.g., emissions in pounds per megawatt-hour—lb/MWh), combined cycle systems have lower emissions rates due to the improved efficiency attributable to the heat recovery steam cycle.

The uncontrolled emissions rates shown in Table 7.3 are higher than the emissions rates of many EGUs operating today because most facilities have some form of PM, SO_2 or NO_x emissions control. As a point of reference, the average emissions rates for NO_x and SO_2 across all coal-fired EGUs in the U.S. in 2002 were 0.40 pounds per million British thermal unit (lb/MMBtu) and 0.94 lb/MMBtu, respectively (NRDC, 2004).

Emissions Control Opportunities

Numerous options are available for reducing $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from EGUs. In general, the options fall into three basic categories: (1) fuel switching to lower emitting fuels; (2) control technologies (including combustion and post-combustion controls) for both new and existing facilities; and (3) reliance on lower emitting (including non-emitting) forms of electric power generation. In the discussion that follows, we provide an overview of the control technologies available to the EGU sector, focusing first on boiler control technologies. We discuss combustion turbines and IC engines separately. A more detailed discussion of the control options for coal-, oil-, natural gas- and wood-fired boilers is contained in Chapter 5, *Boiler Technologies*.

A note on cost-effectiveness: Throughout, we report the cost-effectiveness of individual control options applied to a specific type of facility, such as the cost per ton of reducing SO_2 emissions from a coal-fired power plant. However, in many cases emissions of NO_x and SO_2 (though not $PM_{2.5}$) from fossil-fired power plants are regulated under a cap-and-trade system, and the costs of control are

Table 7.3

Typical Uncontrolled Emissions Rates for EGU Facilities

Technolgy & Fuel	$PM_{2.5}$		SO_2		NO_x	
	lb/MMBtu	lb/MWh	lb/MMBtu	lb/MWh	lb/MMBtu	lb/MWh
Pulverized coal bituminous (3% sulfur)[a]	0.3	2.5	4.8	47.5	0.9	9.2
Pulverized coal bituminous (1.5% sulfur)[a]	0.3	2.5	2.4	23.6	0.9	9.2
Pulverized coal subbituminous (0.35% sulfur)[b]	0.2	2.1	0.6	6.5	0.7	6.7
Natural gas combined cycle[c]	Negligible		Negligible		0.32	2.2
Simple cycle turbine natural gas[d]	Negligible		Negligible		0.32	3.1
Simple cycle combustion turbine distillate oil	0.004	0.04	0.03	0.3	0.88	8.6
Internal combustion engine natural gas[e]	Negligible		Negligible		2.7	45.9
Internal combustion engine diesel (high-sulfur diesel—5%)[e]	0.1	1.70	0.51	8.6	3.2	54.4
Internal combustion engine diesel (low-sulfur diesel—0.5%)[e]	0.1	1.70	0.05	0.9	3.2	54.4

a. Assumes coal ash content of 10%, 24 MMBtu/ton coal heating value and 10,000 Btu/kWh heat rate. Rates based on uncontrolled dry-bottom, wall-fired boiler.

b. Assumes coal ash content of 6.4%, 18 MMBtu/ton coal heat content and 10,000 Btu/kWh heat rate. Rates based on uncontrolled dry-bottom, wall-fired boiler.

c. Assumes heat rate of 7,000 Btu/kWh, no steam injection or other controls.

d. Assumes heat rate of 9,800 Btu/kWh, no steam injection or other controls.

e. Assumes heat rate of 17,000 Btu/kWh.

Source: EPA, 1996, 1998a, 2000

determined by many factors, including fuel prices, fuel transportation costs, pollution control system costs and other factors. In theory, the price of allowances under a cap-and-trade program should reflect the marginal cost of compliance, or the cost of reducing one additional ton of SO_2 or NO_x. As a result, the allowance market provides a measure of cost-effectiveness at the prevailing cap level. The cost-effectiveness of a single control option is less informative.

Fuel Switching

Fuel switching can be an effective strategy for reducing $PM_{2.5}$, SO_2 and NO_x emissions. For example, switching from a high-sulfur (bituminous) coal to a low-sulfur (subbituminous) coal can reduce a plant's SO_2 emissions by more than 70 percent. Fuel blending has also been widely used in order to control SO_2 emissions. This has been one of the industry's key strategies for complying with the federal Acid Rain program (Title IV of the Clean Air Act), as evidenced by the sector's growing reliance on low-sulfur coal supplies over the past decade (USGS, 2001). Switching from coal to natural gas produces an even more dramatic result, virtually eliminating a plant's SO_2 and direct $PM_{2.5}$ emissions.

Fuel switching, however, is not without its complications. The feasibility of fuel switching will depend in part on the characteristics of the plant and the particular type of fuel change at issue. Many plants will be able to switch from high-sulfur to low-sulfur bituminous coal without serious difficulty, but switching from bituminous to subbituminous coal may present greater challenges and costs. In some instances, fuel switching will require significant investment and modifications to an existing plant. Switching to a lower-sulfur coal can affect coal handling systems, boiler performance, PM control-effectiveness and ash handling systems (Rastogi, 2001; Rubin, 2001). Rail lines may also limit access to lower-sulfur coals.

Switching from oil to gas is relatively easy and many plants already have dual oil- and gas-firing capacity. Going from a higher- to a lower-sulfur residual oil or from residual to distillate oil is also possible, although issues related to differences in the viscosity of the fuel may have to be addressed.

The availability of fuel switching as a control strategy for a specific power plant will also depend heavily on the location of the facility. For example, a power plant may be unable to access lower-sulfur coal supplies due to its

location, or it may lack access to a natural gas pipeline. Many low-sulfur coals actually cost less than high-sulfur coals, even accounting for the fact that the high-sulfur coals may be higher rank (i.e., the cost of the low-sulfur coal can be less per unit of heat input). But because these fuels frequently have a lower heat content, switching to a low-sulfur coal may reduce boiler output, requiring more fuel consumption per unit of electricity output. Fuel prices may also limit the availability of lower emitting fuels. Since transportation costs make up a large fraction of the delivered price of coal, the prices paid for low-sulfur coal varies widely among power plants. Finally, natural gas is significantly more expensive than coal.

As an example of the cost of fuel switching, in 2002 Manitoba Hydro converted its 132 MW Selkirk plant from coal to natural gas after 40 years of using coal for electricity production. The project required a $30 million capital investment, including the installation of low-NO_x burners (LNBs), fuel control equipment, a burner safety system, the replacement of boiler pressure parts and the construction of 35 miles of high-pressure gas transmission pipeline (University of Manitoba, 2005). In addition to these capital expenses, the company estimates that the plant will cost an additional $10 million per year to operate because of higher fuel costs and lost revenue opportunities stemming from these higher fuel costs. (The actual figure may be higher in light of recent fuel prices.) The emissions benefits of the project have been dramatic, as summarized in Table 7.4.

Fuel switching options for boilers are discussed in more detail in Chapter 5, *Boiler Technologies*.

Fuel Cleaning

There are several options for treating fuels prior to combustion that will reduce emissions at the power plant. In particular, coal washing is widely practiced in the U.S. to remove impurities (sulfur and ash) and to increase the coal's heating value.

Coal washing removes varying degrees of sulfur and ash, depending on the type of coal and the cleaning process. There are two forms of sulfur in coal. The first is an inorganic form called "pyrite." The second is organic sulfur, which is chemically bound in the molecular structure of coal itself. Conventional coal washing removes only inorganic sulfur and nitrogen—physical cleaning will not remove organic sulfur. For example, physical coal cleaning generally removes 40–50 percent of the pyritic sulfur in typical Illinois coal. The end result is a 20–25 percent reduction in SO_2 emissions (Southern Illinois University, 2005).

In addition to lowering air emissions, coal washing also increases the heating value of the fuel. This lowers the transportation cost of the fuel per unit of energy, offseting the costs associated with the coal washing. Coal washing also reduces the maintenance costs associated with the boiler. According to a recent conference paper, unplanned outages—resulting from the use of coal with higher ash and sulfur content—can cost a modest size coal plant millions of dollars in revenue (Cole, 2004).

Table 7.4

Emissions Benefits of Fuel Switching at Selkirk Plant

Pollutant	Coal-Fired (Tons/Year)	Natural Gas-Fired (Tons/Year)	Percent Reduction
PM	1,440	9	99.4%
SO_2	850	3	99.6%
NO_x	623	66	89.4%

Source: University of Manitoba, 2005

Boiler Control Technologies

As indicated in Table 7.1, steam boilers are the dominant technology used in the electric generating sector and coal accounts for the largest segment of the boiler population. There are two basic categories of technologies that are used to control emissions from EGU boilers: combustion controls and post-combustion controls.

Combustion controls involve modifications to the combustion process designed to reduce the formation of NO_x in the boiler. The most common technologies used in the EGU sector are LNBs and over fire air (OFA), either alone or in combination.

Post-combustion controls involve the application of back-end control systems to capture or convert pollutants before they are released to the air. Control technologies are available for all of the major pollutants of concern addressed in this report: $PM_{2.5}$, SO_2 and NO_x. The most common technologies include selective catalytic reduction (SCR) for NO_x control, flue gas desulfurization (FGD) or scrubbers for SO_2 control, and fabric filters (also known as baghouses) and electrostatic precipitators (ESPs) for PM control. A new generation of multi-pollutant control technologies is also under development.

The application of combustion and post-combustion controls varies depending on the type of boiler and the type of fuel combusted. For example, fabric filters are not used on oil-fired boilers because particles from oil combustion tend to clog the filter (EPA, 1998b). In contrast, fabric filters are highly effective in controlling PM emissions from coal- and wood-fired boilers. Table 7.5 summarizes the applications and commercial availability of the various combustion and post-combustion control options for EGU

Table 7.5

Combustion and Post-Combustion Control Options for EGU Boilers

Control Type	Control Technology	Description	Applications and Commercial Availability
Combustion controls (for NO_x control)	Low-NO_x burners	Burner configuration that limits NOx formation by controlling temperature profile in burner zone.	Commercially available for boilers, such as tangential and wall-fired boilers, but not all boiler types.
	Overfire air	Diverts some combustion air and reinjects it above main combustion zone.	Commercially available. Applicable to most boiler types. Must have sufficient furnace height above burners.
	Low-NO_x burners with overfire air	Combination of new burner design and injection of air above main combustion zone.	Available in new boiler designs and can be retrofitted to a variety of boiler types.
	Reburn	Injection of reburn fuel (natural gas, fuel oil or coal) for combustion above the main combustion zone.	Available but not in wide use. Must have sufficient furnace height above burners.
Post-combustion controls	SCR	Injection of ammonia (NH_3) into boiler flue gas; flue gas then passed through a catalyst bed, where the NO_x and NH_3 react to form nitrogen and water vapor.	Available on new boilers and as retrofit.
	SNCR	Injection of NH_3 or urea in the convective pass for NO_x control.	Available on new boilers and as retrofit.
	Wet scrubber	Slurry of lime used to absorb SO_2.	Commercially available and in wide use.
	Spray drying	Alternative scrubbing technique using calcium hydroxide slurry that vaporizes in spray vessel for SO_2 control.	Commercially available and in wide use. Primarily for low to medium sulfur fuels.
	Electrostatic precipitator	Use of grounded electrodes to collect particles.	Commercially available and in wide use.
	Fabric filter (or baghouse)	Filtering elements (bags) used to collect particles.	Commercially available and in wide use.

Table 7.6

Cost-Effectiveness of Combustion and Post-Combustion Control Options for EGU Boilers

Control Type	Control Technology	Control-Effectiveness[a]	Cost-Effectiveness
Combustion controls (for NO_x)	Low-NO_x burners	30%–50%	\$200–\$1,000 per ton of NO_x
	Overfire air	20%–30%	\$250–\$600 per ton of NO_x
	Reburn	30%–60%	\$500–\$2,000 per ton of NO_x
Post-combustion controls	SCR	70%–90%	\$1,000–\$2,000 per ton of NO_x
	SNCR	30%–50%	\$800–\$1,500 per ton of NO_x
	FGD (wet)	>90%	\$200–\$5,000 per ton of SO_2
	FGD (spray dry)	80-90%	\$150–\$4,000 per ton of SO_2
	Electrostatic precipitator	80%–95% ($PM_{2.5}$)	\$48–\$520 per ton of PM[b]
	Fabric filter (or baghouse)	99% ($PM_{2.5}$)	\$37–\$303 per ton of PM[b]

a. Maximum control-effectiveness may exceed the values presented in some cases.

b. Although control effectiveness estimates were available for $PM_{2.5}$, cost-effectiveness estimates for $PM_{2.5}$, specifically, were not available from the sources referenced.

Source: Midwest RPO, 2005; NESCAUM, 2005; EPA, 2003c; EPA, 2003d

boilers.

In general, combustion controls (just for NO_x) involve a lower capital investment and lower operating costs, as compared to post-combustion controls. However, combustion controls generally produce more modest emissions reductions. LNBs and overfire air can achieve a 40–60 percent reduction in NO_x emissions, while SCR controls can achieve greater than 90 percent control. Post-combustion controls (available for a range of pollutants) often require large capital expenditures with significant on-going operating and maintenance costs. Post-combustion controls can also have significant energy requirements, requiring a meaningful portion of a facility's electricity output to operate. Wet scrubber units, in particular, require as much as 1–2 percent of a power plant's total energy output. Table 7.6 summarizes the control-effectiveness and cost-effectiveness of alternative EGU control options.

Renewable Generating Technologies

Renewable forms of energy and nuclear power can also reduce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions by displacing the electricity that would otherwise be produced by a fossil fuel-fired power plant. Currently, renewable sources of energy contribute less than 10 percent of the electricity supplied by the electric power sector; of that 10 percent, hydroelectric power provides 82 percent, biomass provides 9 percent and wind and geothermal each provide 4 percent of renewable energy supply (EIA, 2005c). Nuclear power contributes 20 percent of the sector's electricity output (EIA, 2005a).

The extent to which non-emitting forms of generation reduce $PM_{2.5}$, SO_2 and NO_x emissions will depend on the types of generation displaced, which varies by region, time of day and time of year. Emissions reductions are greatest when renewable energy is used to displace generation at a coal-fired power plant with limited pollution controls.

A number of factors limit the widespread adoption of renewable energy, despite the sector's tremendous potential. Compared with new fossil fuel-fired power plants, renewable technologies reduce long-term operating (primarily fuel) costs, but have high up-front capital costs. (See Table 7.7.) In some cases, the savings in fuel costs eventually pay back the higher capital costs, but in most cases renewable electricity is simply more expensive than fossil fuel-based electricity. (This cost disparity would be reduced and possibly even reversed if electric generating companies were required to internalize the cost of carbon.) Also, renewable energy technologies face technical challenges in terms of scale, dispatch and proximity to transmission systems and, in some cases, strong public opposition. For example, there has been local opposition to several proposed wind power projects in recent years.

Integrated Gasification Combined Cycle

The abundant domestic supply of coal and rising natural gas prices have fueled growing interest in advanced coal-based generating technologies, including integrated gasification combined cycle (IGCC) technology. Rather than burning coal directly, coal gasification reacts coal with steam and carefully controlled amounts of air or oxygen under high temperatures and pressures. A synthesis gas ("syngas") is formed, typically comprised of hydrogen and carbon monoxide (CO). In the IGCC configuration, which combines modern coal gasification technology

Table 7.7

Costs of Conventional and Renewable Energy Technologies

Technology	Size (MW)	Capital ($/kw)	Fixed O&M ($/kw/yr)	Variable O&M ($/MWh)
Natural gas combined cycle	250	503	9.74	1.95
Conventional pulverized coal	600	1,074	23.36	2.92
Combustion turbine	160	383	9.74	3.90
Wind	50	863–1,396	16.12	0
Solar PV	5	3,587	9.49	0
Geothermal	50	1,887–8,985	68–2,274	0
Landfill gas	30	1,369–2,656	93.75	0.0094

O&M = Operation and Maintenance

Note: Region-specific interconnection costs are assumed to increase the capital costs of wind plants.

Source: EPA, 2005d

with both gas turbine and steam turbine power generation, the syngas is used to fuel a combustion turbine. The combustion turbine drives an electric generator, provides air under pressure back to the gasifier, and produces heat to generate steam to drive a steam turbine. This combined use of combustion and steam turbines significantly boosts the efficiency of the system. IGCC can be used for new power plants or for repowering existing facilities.

IGCC has several advantages relative to a conventional coal-based power plant, because gasification facilitates the removal of pollutants that would otherwise be released into the air. The IGCC plant typically achieves 99 percent SO_2 removal and 90 percent NO_x removal, as well as reduced CO_2 emissions (relative to a conventional coal-fired power plant, assuming the same level of output). Table 7.8 presents the permitted emissions rates for a proposed IGCC plant in Wisconsin. Despite the advantages, however, companies have been reluctant to embrace the technology because of the relatively high cost and risk that comes with early adoption of new technology. Two full scale commercial IGCC electric generating units are in operation in the U.S: Tampa Electric Company's 262 MW unit at the Polk plant in Florida and Cinergy's 192 MW unit at the Wabash River plant in Indiana. Outside of the electric generating sector, chemical manufacturers rely on gasification technology for chemical production. For example, the Eastman Chemical plant in Kingsport, Tennessee has utilized coal gasification to produce chemicals for more than 20 years (Ferguson, 2003).

Table 7.8

IGCC Permitted Emissions Rates for the Elm Road IGCC Plant

Pollutant	IGCC Permit Emissions Rates (lb/mmBtu)
NO_x	0.07
PM_{10}	0.011
SO_2	0.03

Source: Clean Air Task Force, 2004

Combustion Turbines and Reciprocating Internal Combustion Engines

The electric generating sector uses combustion turbines and stationary IC engines for peaking and base load power generation, as well as for stand-by or emergency power. Oil, and to a lesser extent natural gas, are used to fuel stationary IC engines in simple cycle or cogeneration (electricity production with heat recovery) mode. Combustion turbines are generally fueled with natural gas and, in some cases, distillate oil. Combined, combustion turbines and IC engines contribute 2 percent of EGU $PM_{2.5}$ and NO_x emissions and less than 1 percent of EGU sector SO_2 emissions (EPA, 2005f).

Internal Combustion Engines. There are two basic types of stationary IC engines: spark ignition and compression ignition. Spark ignition engines, used in stationary applications, typically use a gaseous fuel such as natural gas. Compression ignition engines (or diesel engines) typically use a liquid fuel such as diesel fuel, but they can also operate on a blend of fuels (e.g., natural gas and diesel fuel).

IC engines are used in the electric power sector because of their low initial capital cost, ease of installation and ability to provide intermittent or variable load service. Commercially available stationary engines range from 5 kilowatts (kW) to 7 MW (HARC, 2003). In the electric power sector, several large engines will often be used in tandem in order to meet load requirements.

Most small-scale electrical generators are diesel engines (HARC, 2003). Many diesel engines in the U.S. are used for emergency back-up purposes and operate only a limited number of hours per year. In a study focusing on the Northeast, the Northeast States for Coordinated Air Use Management (NESCAUM) reports that 80 percent of these engines are intended for emergency use, 19 percent are used during periods of peak energy demand and 1 percent are used for baseload power generation (NESCAUM, 2003). Emergency engines are often exempt from emissions standards, with their operation usually limited to a small number of hours or to specific emergency situations.

Compression engines, burning diesel fuel, have relatively high rates of PM and NO_x emissions. NO_x emissions from an uncontrolled diesel engine typically range from 10–14 grams per horsepower hour (g/hp-hr), depending on the horsepower rating (HARC, 2003). On a pound per megawatt hour (MWh) basis, this rate is more than 200 times higher than that of a combined cycle natural gas turbine with SCR controls (NESCAUM, 2003). Diesel exhaust also contains numerous toxic and potentially carcinogenic components. As compared to diesel-fueled compression engines:

- Dual fuel compression engines produce relatively low PM emissions.
- Natural gas-fired spark ignition engines also have relatively low PM emissions.
- Spark ignition engines can produce relatively high NO_x emissions (MECA, 1997).

Table 7.9 reports typical emissions rates for IC engines.

Several options other than fuel switching are available for controlling PM and NO_x emissions from stationary diesel engines. The most effective NO_x control method is SCR. For diesel and dual fuel engines, SCR can reduce NO_x emissions by 90 percent or more (EPA, 2003b). Injection timing adjustments and lean NO_x catalyst technology can provide a more modest 15 to 25 percent NO_x reduction (NESCAUM, 2003). However, timing adjustment modifications are reported to increase PM emissions substantially, and a diesel particulate filter (DPF) is recommended in order to address the increase in PM emissions associated with this option (HARC, 2003).

DPFs and oxidation catalysts can achieve 80–90 percent and 20 percent reductions in PM emissions, respectively (NESCAUM, 2003). Both technologies will also substantially reduce hydrocarbon and CO emissions. Further, low-sulfur diesel fuel can enhance the performance of some of these control options.

Low emissions combustion involves the combustion of a very fuel lean mixture (high oxygen content). A fuel lean mixture lowers cylinder temperatures and lowers NO_x formation. This technology is not effective on diesel engines but does work for dual fuel engines with pilot liquid fuel reduced to 1 percent or less,[3] lean burn engines and rich burn to lean burn engine conversions (NJ DEP, 2003). EPA reports a NO_x control-effectiveness of 80 percent and an average cost-effectiveness for large IC engines using low emissions combustion technology of \$540–\$890 per ton of ozone season NO_x reductions (EPA, 2003b).

The cost-effectiveness of controlling IC engines varies widely, depending on the utilization of the unit. An engine that operates only a limited number of hours will

3. Dual fuel engines are generally started using liquid fuel. Once stable operating conditions are reached, the engine is switched over to natural gas firing. The "pilot liquid fuel" continues to provide a small percentage of the heat input.

Table 7.9

Small Engine PM_{10}, SO_x and NO_x Emissions Factors

Engine Type	PM_{10}[a]		SO_x		NO_x	
	g/hp-hr	lb/MWh	g/hp-hr	lb/MWh	g/hp-hr	lb/MWh
Diesel < 600 hp	1.00	2.95	0.16	0.49	14.06	41.47
Diesel > 600 hp	0.32	0.94	0.16	0.49	10.86	32.04
Natural gas	0.15	0.45	0.002	0.006	10.89	32.12
Gasoline	0.33	0.97	0.16	0.49	5.00	14.72

a. Although this is labeled PM_{10}, according to NESCAUM (2003) $PM_{2.5}$ accounts for most of the overall PM emissions from IC engines.

Source: NESCAUM, 2003

Table 7.10

Cost-Effectiveness of Various Control Options for IC Engines

Technology	Controls	Basis for Cost-Effectiveness Calculation	Cost-Effectiveness Estimate (by Hours of Operation)		
			500 Hours	1,000 Hours	2,000 Hours
Back-up diesel generator (2,848 hp)	Diesel particulate filter (DPF) with ultra-low-sulfur diesel fuel	Combined CO, HC and PM reduction	$19,000 per ton	$10,000 per ton	$5,000 per ton
Shipyard diesel generator (1,030 hp)	DPF and SCR	Combined NO_x, CO, HC and PM reduction	$90,000 per ton	$46,000 per ton	$23,000 per ton
Emergency generator (2,841 hp)	DPF burning regular #2 diesel fuel	Combined CO, HC and PM reduction	$65,000 per ton	$32,000 per ton	$16,000 per ton
Emergency diesel generator (2,220 hp)	Diesel oxidation catalyst burning #1 and #2 diesel fuel	Combined NO_x, CO, HC and PM reduction	$2,000 per ton	$1,000 per ton	$1,000 per ton
Back-up diesel generator (2,220 hp)	DPF burning #2 diesel fuel	Combined CO, HC and PM reduction	$17,000 per ton	$9,000 per ton	$4,000 per ton
Back-up diesel generator for brewery operation (1,109 hp)	DPF burning #2 diesel fuel	Combined CO, HC and PM reduction	$5,000 per ton	$3,000 per ton	$2,000 per ton

HC = hydrocarbons, CO = carbon monoxide

Source: NESCAUM, 2003

face a relatively high cost per ton of pollutant reduction. Other factors that influence the cost-effectiveness of control include the size of the unit, load and fuel type. NESCAUM reports the cost-effectiveness of various control technology combinations applied to stationary IC engines (NESCAUM, 2003). The results of the analysis are presented in Table 7.10. These cost-effectiveness estimates reflect the combined benefits of reducing multiple pollutants, as well as different annual hours of operation. The California Energy Commission reports a cost-effectiveness of $11,000–$38,000 per ton of NO_x reduction for SCR controls on a 1 MW diesel generator (CEC, 2001).

Combustion Turbines. The electric power sector uses combustion turbines, fueled by natural gas and oil, for peaking and base load power generation; these turbines typically range in size from 500 kW to 25 MW, or larger. The primary pollutants of concern from a combustion turbine are NO_x and CO. Modern combustion technology and back-end control systems are capable of achieving extremely low NO_x and CO emissions levels.

Combustion turbines used in the electric power sector are based on aircraft jet engine technology. A combustion turbine power plant consists of a gas compressor, fuel combustors and a gas expansion turbine. Air is compressed in the gas compressor and energy is added to the compressed air by combusting liquid or gaseous fuel in the combustor. The hot compressed air is expanded through the gas turbine, which drives both the compressor and an electric power generator. Gas turbine power plants are available as heavy-duty "frame" machines specifically designed as stationary engines, or as aeroderivative machines—aircraft engines adapted to stationary applications.

Several control options are available for reducing combustion turbine NO_x emissions. The basic options are: (1) water or steam injection into the combustion zone; (2) dry low-NO_x combustion; (3) catalytic combustion; (4) SCR; and (5) other catalyst technologies (e.g., SCONOx). In some cases, these control options, some of which are back-end and some of which are combustion technologies, are used in combination to achieve extremely low NO_x emissions rates.

Water or steam injection systems are designed to lower

the combustion zone temperature, which will reduce NO_x emissions. As the combustion zone temperature decreases, NO_x production decreases exponentially. There is a trade-off, however, between the efficiency of the unit and the amount of water injected. Turbine manufacturers have been increasing the firing temperatures of gas turbines in order to improve their thermodynamic efficiency. Water injection reduces the overall thermal efficiency of the unit and tends to increase CO emissions. Water injection can generally achieve an emissions rate of 25 parts per million (ppm) with units firing natural gas and 42 ppm with units firing oil (Schorr, 1999).

Like the systems used for boilers and IC engines, SCR controls for gas turbines involve the injection of a reagent—ammonia—into the exhaust gas stream. The reagent and NO_x react in the presence of a catalyst to form molecular nitrogen and water.

New gas combustion systems are capable of achieving extremely low levels of NO_x emissions without add-on controls. For example, GE frame turbines are capable of achieving emissions levels of 9 ppm NO_x with dry low-NO_x combustion (Schorr, 1999). The technology uses staged combustion and lean premixed fuel-air mixtures to reduce NO_x emissions.

SCONOx is a catalytic technology that removes both NO_x and CO without the use of supplementary chemical reagents (e.g., ammonia). The system is currently installed only on a 32 MW cogeneration unit in California and a 5 MW unit in Massachusetts (Czarnecki, 2005).

XONON is a catalytic combustion technology, developed by Catalytica and Kawasaki, which operates at much lower gas temperatures, vastly reducing the amount of thermal NO_x generated in the combustion process. The California Air Resources Board (CARB) certified XONON in June 2002 at 2.5 ppm (dry volume basis) NO_x and 6.0 ppm (dry volume basis) CO. The system is currently installed at a 1.5 MW unit in California.

EPA reports a cost of $3,000–$6,000 per ton of NO_x removed with SCR controls on a large (greater than 75 MW) gas turbine and a cost of $2,000–$10,000 per ton of NO_x removed for a small (5 MW) turbine (EPA, 2003a). GE reports the annual cost of reducing NO_x using SCR from 9 ppm to 3.5 ppm for a specified 170 MW gas turbine operating 8,000 hours per year at $8,000–$12,000 per ton of NO_x removed (Schorr, 1999).

Regulatory Authority

EGU emissions contribute significantly to local, regional and national air pollution problems, including $PM_{2.5}$ pollution. Numerous state and federal programs stemming from Titles I and IV of the Clean Air Act regulate these emissions. The mix of regulatory programs results in a range of EGU emissions limits and performance standards, imposed by permits and state and federal regulatory programs, including emissions trading programs such as the Clean Air Interstate Rule (CAIR) and the Acid Rain SO_2 Program.

As we have indicated in other chapters, the states and the federal government share responsibility for regulating criteria pollutants (e.g., PM, SO_2, NO_x) from stationary sources like EGUs. The federal government has more of the burden of regulating toxic air pollutants, but—for both criteria pollutants and air toxics—states and local areas are free under federal law to adopt more stringent standards than the Clean Air Act requires.

This section describes federal standards that apply under the Clean Air Act to pollutant emissions from EGUs.

Reasonably Available Control Technology

State Implementation Plans (SIPs) for areas that are in nonattainment of federal ambient air quality standards must include Reasonably Available Control Technology (RACT) requirements for new and existing sources. RACT standards are non-binding federal guidelines; for volatile organic compounds (VOCs), the guidelines are known as Control Technique Guidelines (CTGs). Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

There are ACTs applicable to NO_x emissions from stationary gas turbines and utility boilers, which can be found at EPA's website (www.epa.gov/ttn/catc/products.html#aptecrpts). SIP writers should bear in mind that although the ACTs may provide a starting point, they are unlikely to represent the actual RACT limits that states will choose to adopt. That is the case because RACT standards are intended to reflect advancements in technology, and most of the ACTs are outdated—some, seriously so. Additionally, the ACTs are guidelines only, not actual federal limits. Even if they were federal limits, many states would have the leeway to enact more stringent standards. And since the ACTs do not set limits, even state prohibitions against enacting standards that are more stringent than federal requirements may be inapplicable.

New Source Performance Standards

New Source Performance Standards (NSPS) typically apply to major stationary sources (for EGUs, greater than 250 MMBtu per hour heat input) in specific source categories, in both attainment and nonattainment areas, when they are first constructed, reconstructed or modified.

Table 7.11

NSPS for Fossil Fuel-Fired Steam Generators Greater than 250 MMBtu per hour (40 CFR §§60.40, *et seq.* (subpart D), 40 CFR §§60.40a, *et seq.* (subpart Da))

Fuel Type	Applicable Dates[a]	PM Limit	Opacity Limit [b]	SO_2 Limit[c]	NO_x Limit[c]
Gases (natural gas)	After August 17, 1971	0.1 lb/MMBtu	20%	NA	0.2 lb/MMBtu
	After September 18, 1978	0.03 lb/MMBtu	20%	0.8 lb/MMBtu and 90% reduction, or 0% reduction when emissions are less than 0.2 lb/MMBtu	0.2 lb/MMBtu[d]
Liquids (oil)	After August 17, 1971	0.1 lb/MMBtu	20%	0.8 lb/MMBtu	0.3 lb/MMBtu
	After September 18, 1978	0.03 lb/MMBtu (70% reduction from potential combustion concentration)[e]	20%	0.8 lb/MMBtu and 90% reduction, or 0% reduction when emissions are less than 0.2 lb/MMBtu	0.3 lb/MMBtu[f]
Solids (coal)	After August 17, 1971	0.1 lb/MMBtu	20%	1.2 lb/MMBtu	0.7 lb/MMBtu[g]
	After September 18, 1978	0.03 lb/MMBtu (99% reduction from potential combustion concentration)	20%	1.2 lb/MMBtu and 90% reduction, or 70% reduction when emissions are less than 0.60 lb/MMBtu[h]	0.6 lb/MMBtu[i]

a. The dates after which units commenced construction, reconstruction or modification.
b. In all cases, except for one six-minute period per hour of not more than 27% opacity.
c. Subpart D contains an equation for facilities that burn multiple fuels and have an applicable date of August 17, 1971. Subpart Da contains an equation for facilities that burn multiple fuels and have an applicable date of September 18, 1978.
d. Coal-derived gaseous fuels are subject to a NO_x limit of 0.5 lb/MMBtu.
e. "Potential combustion concentration" means the theoretical emissions (lb/MMBtu heat input) that would result from combustion of a fuel in an uncleaned state without emissions control systems), and is further defined at 40 CFR § 60.41a.
f. Coal-derived liquid fuels and shale oil are subject to a NO_x limit of 0.5 lb/MMBtu.
g. Except for lignite used in units commencing construction after December 22, 1976 and mined in North Dakota, South Dakota or Montana and burned in a cyclone-fired unit (NO_x limit of 0.8 lb/MMBtu), and other lignite (NO_x limit of 0.6 lb/MMBtu).
h. SO_2 emissions are limited to 1.2 lb/MMBtu for any affected facility that combusts 100% anthracite; and 1.2 lb/MMBtu and 85% reduction for any affected facility that combusts solid solvent refined coal.
i. Except subbituminous and coal-derived fuels (NO_x limit of 0.5 lb/MMBtu) and any fuel containing more than 25% lignite by weight (NO_x limit of 0.8 lb/MMBtu).

There are two sets of NSPS requirements, the applicability of which depends on the date that plant construction, reconstruction or modification commenced. Plants that commenced construction after August 17, 1971 have one set of NSPS limits (40 CFR §§60.40, et seq. (subpart D)); and plants that commenced construction after September 18, 1978, a different set (40 CFR §§60.40a, et seq. (subpart Da)). Thus, plants built or under construction prior to August 17, 1971 are not subject to NSPS limits without a major modification after the NSPS rules took effect.

The NSPS are not dependent on the type of boiler technology, but they do vary according to the type of fossil fuel consumed (although this is not the case for the new NSPS discussed immediately below). Table 7.11 shows the current NSPS limits.

In February 2005, EPA proposed changes to the NSPS limits at 40 CFR §§60.40a, et seq. (subpart Da). The current standards were originally promulgated on June 11, 1979. The final rule was released on February 9, 2006 and amends the emissions limits for PM, SO_2 and NO_x from EGU steam generating units. Only those units that begin construction, reconstruction or modification after February 28, 2005 will be affected by the proposed amendments. The fuel-neutral limits for these new units are summarized in Table 7.12, as are the limits advocated by STAPPA and ALAPCO in response to EPA's proposal.

EPA promulgated the current standards for stationary combustion turbines, shown in Table 7.13, in 1979. They are available at 40 CFR §§60.330, et seq. (subpart GG). In February 2005, EPA also proposed changes to the NSPS for NO_x and SO_2 emissions from stationary combustion

Table 7.12

Fuel-Neutral NSPS for EGU Boilers Constructed, Reconstructed or Modified after February 28, 2005

		Limits Promulgated by EPA		Limits Suggested by STAPPA and ALAPCO
Pollutant	**Facility Type**	**lb/MWh**	**lb/MMBtu**	**lb/MMBtu**
PM[a]	New or reconstructed	-	0.015 or 99.9% reduction	-
	Modified	-	0.015 or 99.8% reduction	
SO_2	New	1.4 or 95% reduction[b]	-	0.10
	Reconstructed	-	0.15 or 95% reduction	
	Modified	-	0.15 or 90% reduction	
NO_x	New	1.0	-	0.07–0.08
	Reconstructed	-	0.11	
	Modified	-	0.15	

a. NSPS limit does not include condensable PM. STAPPA and ALAPCO have recommended that EPA adopt a PM NSPS that includes both filterable and condensable PM.

b. The SO_2 limit for new EGUs that burn over 75% coal refuse (by heat input) is 1.4 lb/MWh or 94% reduction.

Source: EPA, 2006; STAPPA and ALAPCO, 2005

turbines, and these, too, were released on February 9, 2006. The new NSPS apply to new stationary combustion turbines that commence construction, reconstruction or modification after February 18, 2005 (EPA, 2006a). (We have not included a table showing the new NSPS for new stationary combustion turbines because of the complexity of the new standards, which vary based on the fuel input of the unit at peak load, fuel, application, and location of the turbine.)

There are no existing NSPS for IC engines. However, on July 11, 2005, EPA proposed a rule that would reduce emissions of air pollutants from stationary compression ignition IC engines. The proposed NSPS would limit emissions of PM, SO_2 and NO_x from stationary diesel IC engines to the same levels required by EPA's nonroad diesel engine regulations (see Chapter 14, *Nonroad Equipment*). New, modified and reconstructed stationary diesel engines would have to comply with the proposed rule. A new stationary diesel engine is one that is ordered after July 11, 2005 and manufactured after April 2006. Stationary diesel engines that start modification or reconstruction after July 11, 2005 also are subject to the rule. The proposed rule also contains fuel requirements that limit the amount of sulfur in the diesel fuel used to run these engines (EPA, 2005g).

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, Best Available Retrofit Technology (BART) requirements apply to $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from large sources that fall into 26 identified source categories and that began

Table 7.13

NSPS for Stationary Gas Turbines (40 CFR §§60.330, *et seq.* (subpart GG))

Fuel Type	Applicable Dates	Turbine Size	NO_x Limit	SO_2 Limit
Natural gas	October 3, 1977	>100 mmBtu/hour	75 ppm[b]	150 ppm at 15% on a dry basis[a]
	October 3, 1977	10–100 mmBtu/hour	150 ppm[b]	

a. Fuel must have a sulfur content less than or equal to 0.8% by weight (8000 ppm).

b. May be slightly higher or lower based on site-specific calculations.

operation between 1962 and 1997, in both attainment and nonattainment areas. The source categories include fossil fuel-fired steam electric plants of more than 250 MMBtu per hour heat input. However, states have not yet developed their BART rules.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) and Prevention of Significant Deterioration (PSD) requirements apply to new and modified large stationary sources in nonattainment and attainment areas, respectively. The programs subject plants in nonattainment areas to Lowest Achievable Emission Rate (LAER) standards and in attainment areas to Best Available Control Technology (BACT). Neither BACT nor LAER standards are codified because they are intended to reflect technology advancements as they occur.

STAPPA and ALAPCO completed a review of SO_2 and NO_x BACT determinations for coal-fired EGU boilers in March 2004 (STAPPA and ALAPCO, 2004). As part of their review, STAPPA and ALAPCO identified BACT levels for new and existing coal-fired EGU boilers. They compiled the new boiler BACT levels based on data in EPA's RACT/BACT/LAER Clearinghouse, and the existing boiler BACT levels from recent EPA consent decrees in PSD cases.

For SO_2, STAPPA and ALAPCO reported a BACT level of 0.10 lb/MMBtu for new coal-fired EGUs and a BACT level of 0.15 lb/MMBtu for existing coal-fired EGUs. For NO_x, they reported a BACT level of 0.07 lb/MMBtu for new EGUs and a BACT level of 0.10 lb/MMBtu for existing EGUs.

STAPPA and ALAPCO did not review BACT determinations for natural gas- or residual oil-fired EGU boilers, but simply notes that BACT levels more stringent than those for coal-fired boilers are feasible for gas- and residual oil-fired EGU boilers (STAPPA and ALAPCO, 2004).

CARB made BACT determinations for PM, SO_x and NO_x emissions from a modified natural gas-fired EGU boiler in 2002. The limits were 0.01 grains per dry standard cubic foot for PM; 0.2 lb/MMBtu for SO_x; and 5 ppm (at 3 percent oxygen, approximately 0.005 lb/MMBtu) for NO_x. CARB based the PM and SO_x determinations on good combustion practices and the low sulfur content of natural gas. It based the NO_x determination on the combined use of LNBs, FGR and SCR (CARB, 2005).

Officials at CARB have indicated in discussions that new natural gas-fired combined cycle turbines and cogeneration facilities are generally permitted at a NO_x limit of 2.0 ppm over a one hour average, and simple cycle turbines are permitted at a limit of 2.5 ppm over a one hour average (Kato, 2005). Also, different limits sometimes allow higher emissions levels during certain periods, for example when a unit is ramping up or down.

The Minnesota Pollution Control Agency (MPCA) recently issued BACT determinations for two wood-fired utility boilers. The boilers are both new processes at existing facilities. The BACT determinations set limits for PM_{10} and NO_x. The MPCA based the PM_{10} BACT limit on use of an ESP and set the limit at 0.025 lb/MMBtu. It set the NO_x limit at 0.15 lb/MMBtu based on use of selective non-catalytic reduction (SNCR) controls (EPA, 2005e).

Maximum Achievable Control Technology

Maximum Achievable Control Technology (MACT) standards address emissions of hazardous air pollutants (HAPs) from new and existing sources in identified industry categories. They apply in both attainment and nonattainment areas to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs.

We have discussed MACT standards for certain other source categories, where EPA has regulated PM as a surrogate for HAPs—which it does not do for EGU boilers, IC engines or combustion turbines. For that reason, we have not included a discussion of MACT standards in this chapter.

The Acid Rain Program

The 1990 Amendments to the Clean Air Act established the Title IV Acid Rain Program to reduce EGU SO_2 and NO_x emissions. The Title IV SO_2 reduction program uses a cap-and-trade approach: power plants are not subject to specific emissions rate limits but, instead, total SO_2 emissions from affected sources are capped at a specific tonnage. Affected facilities under the program comply by obtaining emissions allowances in an amount equal to their annual emissions. EPA has allocated sources a calculated number of allowances, and sources must purchase allowances in the market or from EPA to meet any shortfall. Sources that install pollution controls may have an excess of allowances, enabling them to sell allowances.

The Acid Rain SO_2 Program has two phases. Phase I began in 1995 and affected 263 EGUs at 110 mostly coal-fired power plants located in 21 eastern and midwestern states. Phase II, which began in 2000, established an annual tonnage cap of 8.95 million tons and brought smaller coal-fired EGUs and oil and natural gas generators into the program. Currently, the SO_2 program covers virtually all EGUs burning coal, oil and natural gas, with an output capacity greater than 25 MW.

The Acid Rain Program also reduces NO_x emissions in two phases, with boiler design-specific emissions limits (not with a cap-and-trade approach). In general, power plants affected by the NO_x limits must install some type of NO_x pollution control device, such as LNB technology. The NO_x requirements apply nationally to existing EGUs and to new power plants beginning operation after the start of the program.

Clean Air Interstate Rule

In 1998, EPA promulgated the NO_x SIP Call, requiring NO_x emissions reductions in 22 states in the eastern U.S. The rule required affected states to amend their SIPs and limit NO_x emissions to meet EPA-specified ozone season NO_x emissions budgets. Although the NO_x SIP Call did not mandate the specific sources that were required to reduce emissions, EGU emissions reductions were generally the most cost-effective option and were at the core of each affected state's strategy to meet its assigned NO_x budget.

EPA followed the NO_x SIP Call in March 2005 with CAIR, aimed at addressing pollution transport that contributes to eight-hour ozone and $PM_{2.5}$ nonattainment in the eastern U.S. CAIR identifies 28 states and the District of Columbia (Figure 4.1 in Chapter 4, *The Clean Air Act*, shows the affected area) as contributing to unhealthy levels of ozone and/or $PM_{2.5}$ in downwind nonattainment areas, and subjects them to additional NO_x and SO_2 emissions reductions. CAIR requires states contributing to $PM_{2.5}$ nonattainment to achieve annual NO_x and SO_2 emissions reductions and states contributing to ozone nonattainment to achieve further seasonal NO_x emissions reductions. States may choose the way in which they achieve these reductions, but it is expected that most, if not all states, will opt (at a minimum) to cap emissions of EGUs according to the budgets established by EPA and to participate in the EPA administered emissions trading programs (EPA, 2005c).

Assuming that all of the affected states opt to participate in the EPA administered trading program, EGU sources in the CAIR states will have to retire two SO_2 allowances for every one ton of emissions beginning in 2010. This effectively requires a 50 percent reduction in allowable emissions. Banked allowances, issued for years prior to 2010, can be used on a one-for-one basis. In the second phase of the program, beginning in 2015, SO_2 allowances (vintage 2015 and beyond) must be retired at a ratio of 2.86 allowances for every one ton of emissions. In terms of NO_x, reductions, the CAIR program establishes two separate NO_x trading programs, an annual NO_x trading program and an ozone season NO_x trading program. The first phase of NO_x reductions will be implemented in 2009, and the second phase in 2015.

Table 7.14

State Programs to Reduce EGU NO_x and SO_2 Emissions beyond Federal Requirements

State	Affected Sources	Pollutant	Reduction Requirement	Effective Date	Comments
MA	Six oldest power plants in state	NO_x	Annual NO_x rate of 1.5 lb/MWh	Oct. 2004	Implemented through regulations. $PM_{2.5}$ and CO set aside for future consideration. CO_2 cap proposed in Dec. 2005.
		SO_2	Phase 1: 6.0 lb/MWh Phase 2: 3.0 lb/MWh	Phase 1: Oct. 2004 Phase 2: Oct. 2006	
NC	14 power plants in state	NO_x	77% reduction (annual NO_x)	2009	Enacted through legislation in 2002. Future consideration to be given to mercury and CO_2.
		SO_2	73% reduction	2013	
NH	Six units at three plants	NO_x	70% reduction based on 1.5 lb/MWh rate	Dec. 2006	Enacted through legislation in 2002. Also includes a CO_2 cap at 1990 levels.
		SO_2	75% reduction based on 3.0 lb/MWh rate	Dec. 2006	
NY	All units 25 MW or greater	NO_x	Annual NO_x cap-and-trade budget based on at 0.15 lb/mmBtu	2004	Implemented through regulations.
		SO_2	50% reduction below Phase 2 Acid Rain levels	2005-2008 phase in	

State and Local Policy Measures

As we have discussed in the *Introduction* to this report, the Clean Air Act allows state and local agencies to adopt more stringent standards than the Act itself requires, although in some instances state law limits this authority. We assume that SIP writers are familiar with applicable local laws and regulations that address this issue. In the *Introduction*, we have also outlined possible approaches to the imposition of more stringent state and local standards, such as requiring sources that have not yet been required to meet BACT/LAER standards to achieve those limits.

State and local authority to tighten Clean Air Act requirements on new and existing sources and to impose air pollution limits on sources that have heretofore been unregulated applies to the EGU sector as well as other industry sectors. However, a number of additional opportunities to reduce emissions from EGUs are worthy of consideration. Depending on the scope of state environmental law, some of the measures discussed below will fall within the purview of environmental regulators; others may require additional legislative authority.

Note that we discuss the opportunities for emissions reductions through the promotion of energy-efficiency programs mainly in Chapter 18, *Residential Fuel Combustion and Electricity Use*, rather than in this chapter. Nonetheless, the demand side management opportunities for EGUs are substantial.

State Laws and Regulations Limiting Emissions from New and Existing EGUs

As described above, a number of federal programs limit emissions from EGUs. However, as a matter of federal law, existence of these programs does not preclude states from passing their own laws or regulations to achieve further reductions. Several states have passed laws (New Hampshire, North Carolina) or regulations (Massachusetts, New York) aimed at reducing EGU emissions beyond federal requirements. Table 7.14 provides a brief overview of these programs.

State programs can be patterned on existing emissions trading programs, which establish an emissions budget and allow for emissions trading flexibility (e.g., the New York program), or they can impose an emissions performance requirement on EGUs, with limited or no flexibility for emissions averaging or trading. (The Massachusetts, New Hampshire and North Carolina programs limit emissions and constrain compliance flexibility.) Either approach can achieve emissions reductions beyond those mandated by federal laws, although trading programs generally reduce the costs of compliance.

Additionally, some smaller EGUs fall below federal regulatory size thresholds, leaving them exempt from federal air pollution control limits. In some cases, state and local regulators have imposed emissions standards on these sources. For example, California (in particular, the San Joaquin Valley Unified Air Pollution Control District), New Jersey and Washington have all regulated units much smaller than the federal standards address. The Ozone Transport Commission (OTC) has also developed a model rule with fuel-specific emissions standards for NO_x for both new and existing non-emergency natural gas and diesel engines. In addition, the Regulatory Assistance Project has developed a model rule for controlling emissions from distributed generation units. The rule—which deals with PM_{10}, not $PM_{2.5}$—has been or is soon expected to be adopted either partially or entirely by Connecticut, Delaware, Maine, Maryland, Massachusetts and Rhode Island (Regulatory Assistance Project, 2006).

Fuel Sulfur Limits

In addition to the stack emissions limits described above, state and local agencies have opted in some cases to regulate the concentrations of sulfur in fuel, for example, by regulating the sulfur content of fuel oil.

In 2000, the Connecticut Department of Environmental Protection (DEP) adopted regulations limiting the sulfur content of fuel oil used by power plants (RCSA §22a-174-19a). Effective January 1, 2002, affected units that combust liquid or gaseous fuels are subject to a fuel sulfur limit of 0.5 percent by weight. Effective January 1, 2003, the fuel sulfur limit is reduced to 0.3 percent by weight. Alternatively, an affected source can meet an average SO_2 emissions rate of 0.33 lb/MMBtu. The DEP Commissioner can suspend the fuel sulfur limits in the event of a fuel supply emergency, in which case the Commissioner can require affected sources to offset excess emissions with SO_2 allowances.

The New York State Department of Environmental Conservation regulates the sulfur content of oil and coal used by stationary combustion sources (6 NYCRR §225). The sulfur limits vary by area within the state. The lowest limits apply in New York City: (1) 0.30 percent sulfur by weight for residual oil, (2) 0.20 percent sulfur by weight for distillate oil and (3) 0.2 lb of sulfur per MMBtu gross heat content for solid fuels. As in Connecticut, the Commissioner has the authority to suspend the standards in the event of a supply shortage.

State Utility Regulatory Programs

An alternative approach for states to consider is to provide EGU owners incentives to install pollution controls by enabling the installation costs to be recovered through utility rates for a limited time only. Rate recovery for pollution control investments exists in states that have not deregulated (or "restructured") their electric industry. However, some states that still regulate retail electricity

supply have enacted measures that go even a step further to facilitate the recovery of pollution control costs, e.g., pre-approval statutes and a variety of tax measures (NARUC, 2004). Indiana, Arkansas, Kentucky and Wisconsin are among the states that facilitate recovery of control costs.

Implementation of this type of utility rate program would be more difficult in states with deregulated electricity markets, although even in these states it would be possible, using a wires or system benefits charge as the mechanism for recovering the control costs. This type of mechanism would enable unregulated power producers to receive compensation for installing controls on power that they sold to regulated retail electricity providers (i.e., the providers of transmission and distribution services) in the state.

Regional CAIR "Plus" Approaches

Another alternative for states to consider is supporting a regional approach for pursuing deeper or quicker reductions in SO_2 and NO_x emissions from EGUs than those provided for in CAIR. STAPPA and ALAPCO published a strategy that calls for a national SO_2 cap of 1.26-1.89 million tons per year (as compared to a baseline of 10.6 million tons in 2001) by 2013, and a NO_x cap of 0.88-1.26 million tons per year by the same date (as compared to a baseline of 4.7 million tons in 2001) (STAPPA and ALAPCO, 2004).

The OTC plans to use a regional partnership and a model rule to implement a program that goes beyond CAIR for EGUs and large industrial boilers. The OTC would use the basic structure of CAIR to implement tighter emissions caps. The OTC aims to develop regional partnerships and a model rule in 2006, followed by a first phase of controls in 2009 and a second phase in 2012 (InsideEPA, 2006). In Phase 1, the program would be based on an SO_2 emissions rate of 0.24 lb/MMBtu and a NO_x emissions rate of 0.12 lb/MMBtu. In Phase 2, the caps would be ratcheted down based on an SO2 emissions rate of 0.14 lb/MMBtu and a NOx emissions rate of 0.08 lb/MMBtu (OTC, 2005). State support for such efforts could increase the likelihood of other regional programs.

States should also consider national and regional approaches to achieving more stringent and expeditious reductions than CAIR.

CAIR NO_x Allowance Allocations

Another option is to pursue an allocation of CAIR NO_x emissions allowances that provides incentives for renewable energy and energy-efficient electricity generation. Under the CAIR program, states are assigned NO_x budgets by EPA, but are given flexibility in allocating the allowances to their sources. (The same flexibility is not available for SO_2 allowance allocations, which are based on the historic permanent allocation methodology embodied in the Acid Rain SO_2 Program.) For NO_x allocations, however, states can choose their allocation methodology, including a distribution that allocates allowances directly to or to a set-aside for non-emitting sources, energy-efficiency projects or other recipients. States are also free to choose an output-based allocation scheme, which favors more efficient (and thus less polluting) generation. State allocations to renewable energy, energy-efficiency or system benefits programs can serve two purposes: (1) tightening the cap on affected sources—which will have fewer allowances allocated to them and, therefore, more of an incentive to install pollution controls; and (2) providing a source of revenue for funding projects or programs that help to reduce emissions. STAPPA and ALAPCO have prepared the document *Alternative NO_x Allowance Allocation Language for the Clean Air Interstate Rule,* which contains alternative regulatory language for NO_x allocation that can provide opportunities to promote clean technologies.

Renewable Energy Programs

Apart from allocation schemes for CAIR NO_x allowances that encourage the development of renewables, most states have programs in place to support renewable energy. To the extent non-emitting renewable energy resources (such as wind and solar) replace new or existing fossil generation, emissions from EGUs are reduced. Of course, not all renewables used to generate electricity will result in zero emissions. For example, combustion of renewable biomass resources will not necessarily be beneficial in addressing $PM_{2.5}$ air quality concerns.

A Renewable Portfolio Standard (RPS) requires that a minimum amount of renewable energy be included in the portfolio of electricity resources of retail electricity providers serving a state. RPS programs already have been adopted by the District of Columbia and 21 states—Arizona, California, Colorado, Connecticut, Hawaii, Illinois, Iowa, Maine, Maryland, Massachusetts, Minnesota, Montana, Nevada, New Jersey, New Mexico, New York, Pennsylvania, Rhode Island, Texas, Vermont and Wisconsin (Pew Center on Global Climate Change, 2006). These programs aim to increase the amount of electricity produced from renewable energy resources by expanding the markets for this product. For example, California requires 20 percent renewable generation by 2017, New York requires 25 percent by 2013 and Pennsylvania requires 18 percent by 2020 (Pew Center on Global Climate Change, 2005). (These percentages are not exactly comparable, because the states vary in the resources they define as renewable.) RPS programs have the disadvantage that they do not specify where renewable energy resources must be developed, so that a state's RPS may stimulate development of renewables elsewhere, limiting the program's impact on local air quality.

Another means of supporting renewables is through public

benefits funding. The funding pool can be created through fees placed on electricity companies or customers. Such fees are sometimes referred to as system benefit charges. A number of states adopted system benefits charges as part of electricity restructuring initiatives. A state seeking to improve EGU emissions performance should consider enacting or expanding renewable energy funding initiatives to improve the overall emissions profile of the its EGU fleet.

Limits on Back-Up Fuels

In the face of rising fuel prices, state and local agencies should evaluate whether increases in the use of back-up fuels are increasing emissions significantly. In some cases, the emissions limits that apply to back-up fuels are less stringent than those applicable to a facility's primary fuel. For example, a natural gas-fired boiler with oil back-up will generally be subject to a more restrictive standard when burning natural gas. One way for operating permits to address this issue is to restrict the operating hours of the unit on the back-up fuel. Cap-and-trade programs that restrict emissions-sector wide can also address this concern.

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Chapter 8

Pulp and Paper

Introduction

The pulp and paper industry converts harvested wood and recycled fibers (e.g., rags or wastepaper) into commodity grades of wood pulp, primary paper products and paper board products. The U.S. is the world leader in the pulp and paper business, producing about 28 percent of the world's pulp and 25 percent of total world output of paper and paperboard. Fueling this large manufacturing sector is the nation's demand for paper products. The U.S. consumed close to 96 million tons of paper products in 2001, about 691 pounds per capita (DOE, 2005). The industry is especially important in many rural economies, where the local paper or lumber mill is often the area's largest employer.

The pulp and paper sector is divided into three distinct segments. Pulp making turns raw wood fiber or recycled fiber into a pulp and bleaches the pulp to varying degrees of brightness as required by the final product. Paper making takes the bleached or unbleached pulp and processes it into sheets of paperboard or paper. Converting operations use these primary materials to manufacture more specialized products, such as paperboard boxes, writing paper and tissue paper (Davis, 2000; EPA, 2002).

Among the various processes involved in the production of paper, the pulping process—which produces particulate matter (PM), sulfur dioxide (SO_2), nitrogen oxides (NO_x) and other air pollutants—is the largest source of emissions. Paper mills and converting facilities have far lower emissions and are discussed only briefly in this chapter. The vast majority of pulp mills (over 80 percent) use the kraft pulping process.

There are four primary sources of emissions in the kraft pulping process: (1) recovery furnaces (or recovery boilers); (2) smelt dissolving tanks (SDTs); (3) lime kilns; and (4) power boilers.[1] The last of these sources, power boilers, are discussed primarily in Chapter 5, *Boiler Technologies*, is Chapter 6, *Industrial and Commercial Boilers*. This chapter focuses on emissions, emissions limits and control technologies for recovery furnaces, SDTs and lime kilns.

1. Recovery boilers are used to recycle chemicals used in the pulping process. Power boilers are used to generate energy for the pulping process and steam for paper making dryers.

Table 8.1

Pulp and Paper Sector Emissions

Process Type	$PM_{2.5}$ Tons/Year	$PM_{2.5}$ Percent of Sector Emissions	SO_2 Tons/Year	SO_2 Percent of Sector Emissions	NO_x Tons/Year	NO_x Percent of Sector Emissions
Pulp & paper mills	78,798	77%	372,704	81%	239,862	78%
Paperboard mills	20,222	20%	77,964	17%	61,625	20%
Converted paper and paperboard products manufacturing	3,079	3%	7,199	2%	6,982	2%
Total	102,099	100%	457,868	100%	308,470	100%

Source: EPA, 2005

Sector Profile

As noted in the *Introduction*, the U.S. produces about 25 percent of the world's paper and paperboard (DOE, 2005). More than 50 percent of the wood cut in the U.S. is eventually used for paper and paperboard (STAPPA and ALAPCO, 1999). In 1998, the U.S. Census Bureau reported 514 pulp and paper mills and over 6,000 converting facilities in the U.S. The majority of converting facilities are small, with approximately 75 percent having fewer than 100 employees. By contrast, two-thirds of pulp and paper mills have more than 100 employees (EPA, 2002).

There are three general types of pulp and paper facilities: pulp mills, paper and paperboard mills, and converting facilities. Pulp mills separate the fibers of wood or other recycled materials to create pulp. In the pulping process, mills may use chemical, semichemical, or mechanical processes. Paper and paperboard mills are primarily engaged in manufacturing paper and paperboard from wood pulp and other fiber pulp. Converting facilities turn paper and paperboard into a variety of products, including paper bags, boxes, envelopes and stationary products. The paper mill sector includes integrated mills, where pulping and paper making occur on-site. As a result, regulations covering emissions from the pulping process refer to the regulated facilities as "pulp and paper mills" (EPA, 2002).

The geographic distribution of pulp and paper mills varies by mill type. Pulp mills are located primarily in regions of the country where trees are harvested from natural stands or tree farms—that is, in the Southeast, Northwest, Northeast and North Central regions. Pulp mills that process recycled fiber are generally located near sources of waste paper. Paper mills are located near pulping operations or near converting markets. The distribution of paperboard mills reflects the location of manufacturing facilities in general, since manufacturing operations are the primary market for paperboard products (EPA, 2002).

Emissions

Table 8.1 summarizes the emissions generated by the pulp and paper sector based on EPA's 1999 National Emissions Inventory. Mills that are primarily engaged in pulp, paper or paperboard manufacturing are the dominant source of $PM_{2.5}$, SO_2 and NO_x emissions within the sector. Converting facilities, which produce a wide range of products like envelopes and paper bags, account for 3 percent or less of emissions. (Note that some integrated mills, classified as paper or paperboard mills, may also manufacture converted products.) In terms of the many processes involved in producing paper, recovery boilers and power boilers are the dominant source of emissions within the sector, as illustrated by Table 8.2, which shows NO_x and SO_2 emissions reported by the Fraser NH LLC integrated pulp and paper mill in Berlin, New Hampshire.

Table 8.2

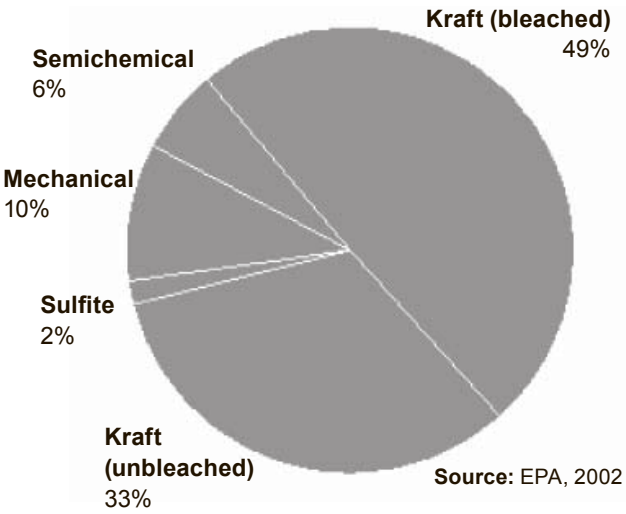
Sample Pulp Mill Emissions Inventory: Fraser NH LLC, Berlin, New Hampshire (2003)[a]

Device	SO_2 tons (%)	NO_x tons (%)
Recovery boiler	43 (2%)	226 (33%)
Hog-fuel boiler[b]	8 (<1%)	124 (18%)
Five oil-fired boilers	1,819 (87%)	312 (46%)
Lime kiln	226 (11%)	17 (3%)
Other	5 (<1%)	2 (<1%)

a. $PM_{2.5}$ emissions were not reported.
b. Hog-fuel is waste wood (e.g., bark, shavings, sawdust) used for energy production.

Source: New Hampshire DES, 2005

Fig. 8.1 Production by Pulping Process, 2000



Source: EPA, 2002

The Paper Making Process

The first step in the paper making process is the preparation of the raw material inputs: (1) debarking and chipping newly harvested wood; or (2) preparing recycled products for pulping. Wood chipping operations, including chip conveyor systems, produce fugitive PM emissions; however, there is limited information on the particle sizes or quantity of emissions from wood chipping (Davis, 2000; Sharifsoltani, 2003).

After chipping wood or preparing recycled products, manufacturers begin the process of pulping (breaking down the wood into fibers). Within the pulp and paper industry, the pulping process is the main source of air emissions. The process is based on chemical wood pulping or mechanical wood pulping. There are three types of chemical pulping: kraft (bleached or unbleached), acid sulfate pulping and semichemical pulping. Figure 8.1 presents the share of pulp making by pulp production process in 2000 (EPA, 2002). The kraft pulping process, discussed below, is the dominant pulping process and is used for more than 80 percent of overall pulp production (on a tonnage basis) in the U.S. (EPA, 2002).

During the actual paper making process, manufacturers spread a wet mixture of pulp onto screens and use vacuums and gravity to remove water. Manufacturers then pass the fibers through rollers and dryers that compress the material into sheets and remove the remaining moisture (EPA, 2002). Emissions are mainly in the form of water vapor, with little or no PM emissions. NO_x or SO_2 emissions may be generated by the dryers, depending on the type of fuel used for heating. These emissions are small because of the low heat levels needed and the large amounts of air used in the process (EPA, 2002; Davis, 2000).

Once the paper has been spread and dried, the converting process—which involves the cutting, gluing or folding of paper products into final form—begins. Emissions from these processes result primarily from the indirect emissions associated with energy consumption. Emissions may result from the burning of fossil fuels on-site to generate hot air for drying or heating, but there appear to be no data on these emissions (Davis, 2000).

Since pulping is the primary source of PM, SO_2 and NO_x emissions, the remainder of this chapter focuses on the pulping process.

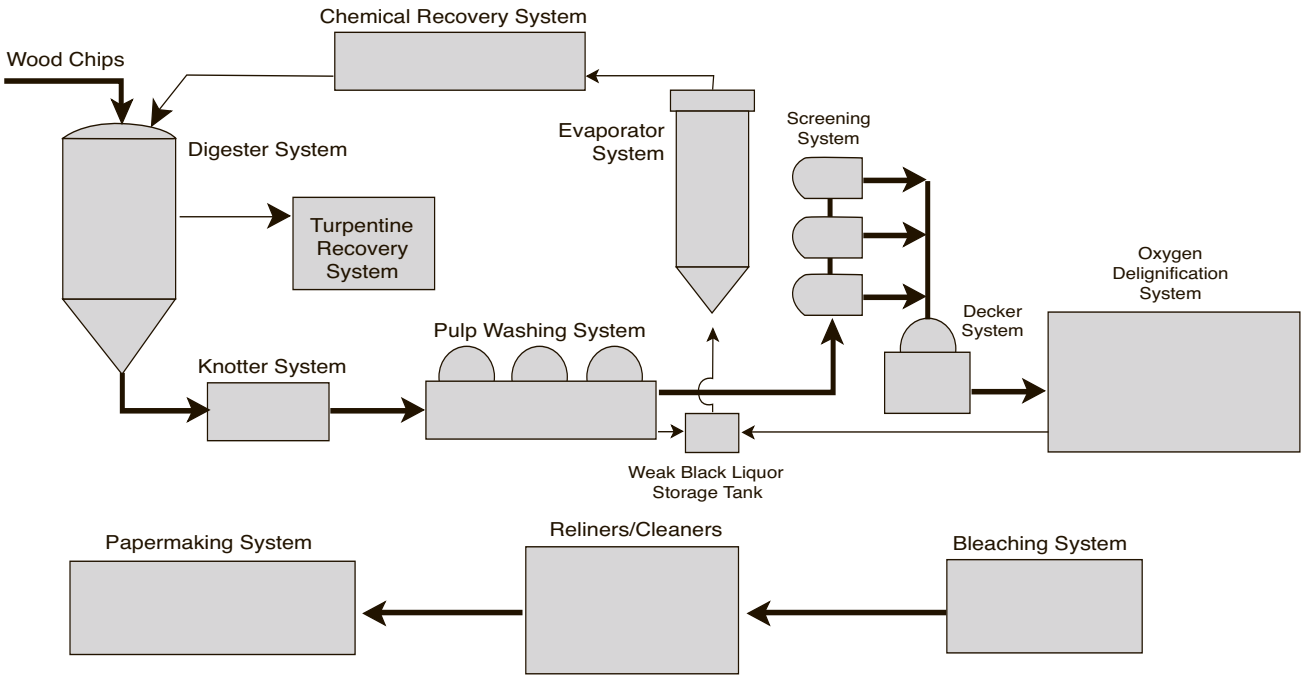
Kraft Pulping

The chemical processes that form the basis for kraft pulping have not changed fundamentally since the patent office issued a patent for this process in 1884 (EPA, 2002). Figure 8.2 provides an overview of a kraft pulping mill with an integrated paper making system. The descriptions of the kraft process in this section focus on sources of air emissions, the majority of which occur in the chemical recovery system. For a more detailed description of the kraft pulping process see Chapter 18 (Wood Processing Industry) in Davis, 2000; or Smook, Handbook for Pulp and Paper Technologists, 2nd Edition, 1992, which EPA recommends (EPA, 2002).

As noted, there are four significant emissions sources within the kraft pulping process: recovery furnaces (or recovery boilers), SDTs, lime kilns and power boilers. As shown in Figure 8.3, all of these sources— except for power boilers—are directly or exclusively involved in the chemical recovery system. (Power boilers supply energy to systems throughout the facility.) The chemical recovery system collects waste liquids and gases from the chemical pulping process and treats them before recycling them back into the chemical pulping process or removing them from the process. Chemical recovery is used in all of the chemical pulping processes (kraft, sulfite and semichemical), reducing discharges to the environment and reducing operating costs (EPA, 2002). Note that this section focuses on the chemical recovery system in the kraft process; semichemical and sulfite chemical recovery processes are different and are treated separately.

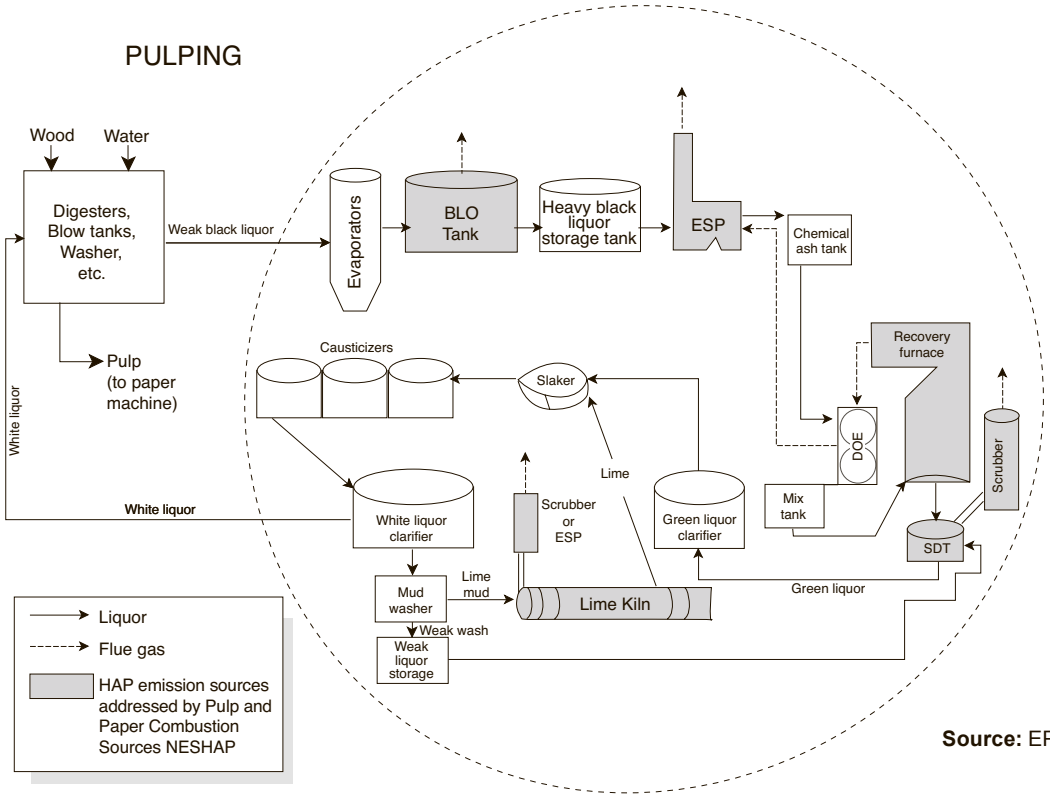
Kraft mills produce pulp from wood chips or recycled products by digesting them in a solution called white liquor (Davis, 2000). One of the byproducts of this digestion is weak black liquor. This liquor contains the pulping chemicals and lignin from the trees. Weak black liquor (about 15 percent solids) is concentrated into heavy black liquor (usually 68–75 percent solids) by evaporation. There are two evaporator types: direct contact evaporators (DCEs) and non-direct contact evaporators (NDCEs). DCEs—used in older furnaces—use direct contact with the hot furnace flue gas to heat the weak black

Fig. 8.2 Overview of Kraft Pulping with Paper Making



Source: EPA, 1998

Fig. 8.3 Chemical Recovery System Details



Source: EPA, 1998

Table 8.3

Emissions from Kraft Recovery Furnaces

Pollutant	DCE Kraft Recovery Furnace		NDCE Kraft Recovery Furnace	
	Emissions Range (lb/ton BLS)	Emissions Mean (lb/ton BLS)	Emissions Range (lb/ton BLS)	Emissions Mean (lb/ton BLS)
PM	0.41–0.98	0.45	0.31–1.66	1.04
SO_2	0.025–10.8	2.06	0.01–54.1	2.63
NO_x	1.06–2.44	1.45	0.63–2.30	1.20

BLS = black liquor solids

Source: Davis, 2000

liquor. In newer furnaces, the liquor is concentrated in closed NDCEs. These have the advantage of preventing emissions of sulfur compounds (total reduced sulfur or TRS) that are stripped from the liquor by direct contact with the flue gas.[2]

Recovery Furnaces and Power Boilers

Pulp facilities burn concentrated black liquor in recovery furnaces to produce energy and recover sodium and sulfur. Recovery furnaces are similar in design to fossil fuel-fired boilers, but are several times larger than a typical industrial boiler. Nevertheless they have comparable output as a result of the relatively low heating value of the black liquor (about 6,000 British thermal units per pound (Btu/lb)). Sodium and sulfur compounds are collected as molten smelt from the bottom of the recovery furnace.

Table 8.4

Estimated Energy Sources for the U.S. Pulp and Paper Industry

Energy Source	Percent (1999)
Purchased steam	1.5
Coal	12.5
Fuel oil	6.3
Natural gas	17.6
Other purchased energy	6.7
Waste wood and wood chips and bark	13.5
Spent liquor solids	40.3
Other self-generated power	1.6
Total	100

Source: EPA, 2002

Table 8.3 reports the estimated emissions of a recovery furnace. The PM emissions rates assume that an electrostatic precipitator (ESP) is installed. The type of evaporator used in the process influences emissions from the recovery furnaces. About 90 percent of PM emissions consist of PM_{10}, and of this subset about 50 percent of particles are smaller than 1 micrometer (µm) in diameter (Davis, 2000). Note that NDCEs do not appear to significantly reduce PM, SO_2 or NO_x emissions. NDCEs are preferred over DCEs because of their impact on TRS, not on PM, SO_2 and NO_x emissions.

SO_2 concentrations in the stack gases from a recovery furnace average between 0 and 500 parts per million (ppm) (Davis, 2000). SO_2 emissions from recovery furnaces are highly variable and depend on liquor properties, combustion patterns and furnace design. NO_x concentrations in the stack gases average between 50 and 100 ppm (Davis, 2000).

The energy generated by the recovery furnace is used to power the facility; however, supplemental energy is generally required to power all processes. Industry-wide, combustion of spent liquor solids in the recovery furnace makes up about 40 percent of overall energy requirements, as shown in Table 8.4. Manufacturers make up for the shortfall with fossil fuel- and wood-fired boilers. Table 8.4 shows that natural gas is the largest fuel source for the non-recovery boilers (power boilers), followed by wood, coal and fuel oil. Emissions factors for industrial boilers burning these types of fuels are discussed in Chapter 6, *Industrial and Commercial Boilers.*

The total boiler capacity of the pulp and paper sector, including recovery and power boilers, is estimated at 376,000 million British thermal units per hour (MMBtu/hr), or 25 percent of total industrial boiler capacity in the U.S. (Energy and Environmental Analysis, 2005). The sector is also the largest consumer of boiler fuel within the

2. Note that TRS has historically been the focus of regulation at kraft pulp mills (along with particulates) because it is the source of the strong "rotten egg" odor frequently associated with the pulp and paper industry. TRS compounds are a byproduct of the reaction between the wood and the white liquor.

industrial sector, consuming more than 2,200 trillion Btu per year (Energy and Environmental Analysis, 2005). This places the pulp and paper sector ahead of the chemical, petroleum refining, and primary metals industries in terms of boiler fuel consumption.

Smelt Dissolving Tank

The molten smelt collected from the bottom of the recovery furnace's combustion chamber contains sodium and sulfur from the combusted liquor. Molten smelt is quenched with water in the SDT to form green liquor, which is a solution of sodium sulfide and sodium carbonate (EPA, 2002). The steam produced in the quenching process is a source of PM emissions. PM is usually removed with a wet scrubber, such as a venturi scrubber (Davis, 2000). (Chapter 5, *Boiler Technologies*, includes more information on wet scrubbers.) Table 8.5 summarizes the controlled emissions rates of an SDT.

Table 8.5

SDT Emissions

Pollutant	Emissions Range (lb/ton BLS[a])	Emissions Mean (lb/ton BLS[a])
PM[a]	0.04–0.18	0.13
SO_2	ND–0.075[c]	0.016
NO_x	ND–0.151[c]	0.033

BLS = black liquor solids
ND = not detected
a. After a venturi scrubber

Source: Davis, 2000

Lime Kilns

Once the smelt has been treated in the SDT, further processing is required to produce white liquor and calcium carbonate (Davis, 2000). The calcium carbonate is sent to a lime kiln, where it is converted into quicklime for reuse in the white liquor recycling process. Like SDTs, lime kilns are usually controlled with venturi scrubbers. Kraft mill lime kilns are large rotary kilns, which typically fire either natural gas or residual oil (Davis, 2000). Table 8.6 provides emissions estimates for lime kilns.

Table 8.6

Emissions from Lime Kilns

Pollutant	Emissions Range (lb/ton CaO[a])	Emissions Mean (lb/ton CaO[a])
PM[b]	0.84–4.0	1.82
SO_2	0.01–1.63	0.23
NO_x	0.08–9.0	2.19

a. CaO, reburned lime product
b. After venturi scrubber

Source: Davis, 2000

Acid Sulfite Pulping

Acid sulfite pulping operations in the U.S. produce less than 2 percent of total pulp. Use of the process has been declining for decades; in 1999 there were only 11 mills in the U.S. using sulfite pulping (Davis, 2000) and fewer than five mills in 2005 (Lockwood-Post Directory, 2005). Sulfite pulping involves digesting wood chips with an acidic solution. Some sulfite operations use chemical recovery with evaporators and recovery furnaces.

SO_2 emissions are a significant concern from acid sulfite pulping; 10–70 pounds of SO_2 can be released per ton of pulp if care is not taken when excess gases are released from the digester (Davis, 2000). Additional SO_2 can be released during the washing process if gases are not collected and redirected to a scrubber. Uncontrolled emissions from washing are normally 1–4 pounds per ton (lb/ton) of pulp, but can be as high as 16 lb/ton (Davis, 2000).

Semichemical Pulping

Semichemical pulping is used to produce approximately 6 percent of total pulp in the U.S. (EPA, 2002); approximately ten plants rely on the process (Lockwood-Post Directory, 2005). This pulping process involves partial chemical digestion of the wood by using either a lower temperature, shorter cooking time or more dilute liquor than in a conventional chemical process. A neutral sulfite digesting liquor is commonly used for partial chemical digestion, followed by grinding of the wood chips in a refiner. Chemical recovery is not practiced at any of the semichemical operations in the U.S.

The major process difference between chemical pulping and semichemical pulping is that semichemical pulping uses lower temperatures, more dilute cooking liquor or shorter cooking times, and mechanical disintegration for fiber separation (EPA, 2002).

Mechanical Pulping

Mechanical pulping operations in the U.S. produce 10 percent of total pulp (EPA, 2002). Despite its name, mechanical pulping can involve some steaming or sulfite softening of the wood prior to the grinding operation. Mechanical pulping typically results in high pulp yields, up to 95 percent, as compared to chemical pulping yields of 45–50 percent, but energy usage is also high. To offset its structural weakness, mechanical pulp is often blended with chemical pulp (EPA, 2002). There are approximately 24 mechanical pulp mills in the U.S. (Lockwood-Post Directory, 2005).

Emissions Control Opportunities

As discussed in the previous section, the primary sources of $PM_{2.5}$, SO_2 and NO_x emissions from pulp and paper mills are recovery furnaces, SDTs, lime kilns and power boilers. Chapter 5, *Boiler Technologies*, discusses emissions control technologies for power boilers that burn coal, natural gas, oil or wood.

Controlling PM Emissions

Recovery furnaces rely on an ESP or a combination of an ESP and wet scrubber to control PM emissions. Most PM emissions from lime kilns and SDTs are controlled with wet scrubbers—typically, venturi scrubbers (Davis, 2000). Opportunities exist to reduce current PM emissions levels by upgrading or replacing older wet scrubbers or ESPs (NESCAUM, 2005). For example, on recovery furnaces, older model ESPs have collection efficiencies close to 90 percent, while newer model ESPs have collection efficiencies greater than 99 percent (Davis, 2000).

The cost of retrofitting a recovery furnace with an ESP is heavily influenced by site-specific factors. Table 8.7 provides one estimate of the costs for a unit equipped with DCE and a unit equipped with NDCE. Capital costs range from $18.4–$29.3 million.

Lime kilns installed in the past ten years are likely to have an ESP control system. Emissions from older units can be reduced by replacing wet scrubbers with ESPs. Table 8.8 shows estimated costs for implementing this type of replacement to meet the 0.01 grains per dry standard cubic foot limit set in the recent MACT standard (described below).

Table 8.7

Estimated Recovery Furnace ESP Retrofit Costs

Furnace Characteristics	Capital Cost	O&M (annual)
NDCE, 3.7 MMlb BLS/day	$29.3 million	$1.9 million
DCE, 1.7 MMlb BLS/day	$18.4 million	$1.2 million

BLS = black liquor solids

Source: NESCAUM, 2005

Controlling SO_2 Emissions

Other than power boilers, recovery furnaces are the main source of SO_2 emissions from pulp and paper mills. In general, recovery furnaces are designed to maximize sulfur recovery while minimizing sulfur air emissions. This is accomplished by operating an overall reducing atmosphere in the combustion zone, producing sodium sulfide from the sulfate in the black liquor rather than SO_2 (North Carolina DENR, 2000).

Table 8.8

Estimated Lime Kiln ESP Retrofit Costs

Lime Kiln Characteristics[a]	Capital Cost	O&M (annual)
270 CaO/day	$3.4 million	$0.2 million

a. CaO, reburned lime product

Source: NESCAUM, 2005

Compared to the more traditional fuels burned in power boilers (i.e., natural gas, coal, wood and oil), the black liquor solids burned in the recovery furnace can have widely varying sulfur content. This results in variable SO_2 emissions, making control difficult.

Despite this difficulty, some facilities have successfully lowered SO_2 emissions by reducing the sulfur content of the black liquor before combustion and by regulating temperatures in the furnace to minimize SO_2 formation (Davis, 2000). Only one example of a pulp mill utilizing a wet scrubber for SO_2 control was identified (North Carolina DENR, 2000). Table 8.9 presents estimated costs for installing scrubbers that reduce emissions to between 10 and 50 ppm; however, costs are reported to be highly site specific.

Another source of SO_2 emissions is the combustion of TRS. As briefly mentioned in the kraft process description, the pulping process results in the formation and release of TRS (a pollutant that is well known for its "rotten egg" odor). One control option for TRS is collection and combustion. If the combustion is uncontrolled, this option can result in SO_2 emissions of about 5.7 pounds per air-dried ton of unbleached pulp. When TRS is burned in lime kilns or

Table 8.9

Estimated Recovery Furnace SO_2 Scrubber Retrofit Costs

Furnace Characteristics	Capital Cost	O&M (annual)
NDCE, 3.7 MMlb BLS/day	$12.8 million	$1.6–$1.8 million
DCE, 1.7 MMlb BLS/day	$8 million	$1.1–$1.3 million

BLS = black liquor solids

Source: NESCAUM, 2005

wood residue-fired boilers, the lime or bark ash alkalinity captures a significant portion of the SO_2. Another option is to remove SO_2 with alkaline scrubbers (Davis, 2000).

Acid sulfite pulping is used at a declining number of pulping operations, but it can emit large amounts of SO_2 per ton of pulp. Raising the pH of the digester before releasing excess gas reduces the amount of SO_2 released. Alternatively, SO_2 can be scrubbed out using an alkaline solution (with recovery of up to 97 percent) (Davis, 2000).

Controlling NO_x Emissions

Apart from the power boilers, lime kilns and recovery furnaces are the primary sources of NO_x emissions from pulp and paper mills; however, options to reduce these emissions are limited. Staged combustion can reduce NO_x emissions, but care must be taken because changes in firing patterns may increase SO_2 emissions (Davis, 2000). Selective non-catalytic reduction (SNCR) and selective catalytic reduction (SCR) technologies for recovery furnaces have been investigated but have been determined not to be technically feasible. Further, low-NO_x burners appear to adversely affect efficiency and energy usage (NESCAUM, 2005).

Several permits were reviewed to determine whether NO_x controls have been applied to lime kilns and recovery furnaces in the pulp and paper industry. A best available control technology (BACT) analysis for a proposed lime kiln evaluated the feasibility of low-NO_x burners, flue gas recirculation, oxidation/reduction scrubbing, SCR, SNCR and non-selective catalytic reduction, concluding that none of these options is technically feasible for lime kiln NO_x emissions control. Rather, the regulatory agency concluded that "good design and operation practices" constitute BACT, and established a NO_x limit of 175 ppm at 10 percent oxygen (Georgia DNR, 2003).

Similarly, a BACT analysis for a kraft recovery furnace concluded that SCR and SNCR were both technically infeasible for recovery furnace NO_x emissions control. The regulatory agency determined that combustion controls (staged combustion) constitute BACT for the recovery boiler, and reported that the RACT/BACT/LAER Clearinghouse includes combustion controls as BACT for recovery furnaces with NO_x emissions ranging from 75–120 ppm (North Carolina DENR, 2000).

Regulatory Authority

As indicated in the more detailed discussion of the Clean Air Act in Chapter 4, states and the federal government share responsibility for regulating criteria pollutants (e.g., $PM_{2.5}$, SO_2, NO_x) from stationary sources like those operating at pulp and paper mills. The federal government has more of the burden of regulating toxic air pollutants, but—for both criteria pollutants and air toxics—state and local agencies are free under federal law to adopt more stringent standards than the Clean Air Act requires.

This section describes federal standards that apply under the Clean Air Act to emissions from pulp and paper mills.

Reasonably Available Control Technology

State Implementation Plans (SIPs) for areas that are in nonattainment of federal ambient air quality standards must include Reasonably Available Control Technology (RACT) requirements for new and existing sources. RACT standards are non-binding federal guidelines; for volatile organic compounds (VOCs), the guidelines are known as Control Technique Guidelines (CTGs).

Table 8.10

NSPS for Kraft Pulp Mills (40 CFR §§60.280, et seq. (subpart BB))[a]

Source	Particulate Matter Limit	Opacity Limit
Recovery furnace	0.10 g/dscm (0.044 gr/dscf)[b,c]	35%
SDT	0.1 g/kg (0.20 lb/ton) BLS	NA
Lime kiln (gaseous fossil fuel such as natural gas)	0.15 g/dscm (0.066 gr/dscf)[d]	NA
Lime kiln (liquid fuel such as oil)	0.30 g/dscm (0.13 gr/dscf)[d]	NA

BLS = black liquor solids

a. The kraft NSPS also include TRS limits for various sources, including recovery furnaces, lime kilns, and SDTs.

b. g/dscm stands for "grams per dry standard cubic meter;" gr/dscf stands for "grains per dry standard cubic feet."

c. Corrected to 8 percent oxygen

d. Corrected to 10 percent oxygen

Table 8.11

MACT Standards for Chemical Recovery Combustion Sources at Kraft Pulp Mills (40 CFR §§63.860, et seq. (subpart MM))[a]

Facility Type	Source	PM Emissions Limit	
		Existing Source[b]	New Source
Kraft pulp mill	Recovery furnace	0.10 g/dscm (0.044 gr/dscf)[c,d]	0.034 g/dscm (0.015 gr/dscf)[e]
	SDT	0.10 kg/Mg (0.20 lb/ton) BLS	0.06 kg/Mg (0.12 lb/ton) BLS
	Lime kiln	0.15 g/dscm (0.064 gr/dscf)[e]	0.023 g/dscm (0.01 gr/dscf)[e]

BLS = black liquor solids

a. These standards also apply to soda, sulfite and stand-alone semichemical pulp mills.

b. As an alternative to the PM emissions limits for existing kraft pulp mills, each owner or operator of a kraft pulp mill may establish PM emissions limits for each existing kraft recovery furnace, SDT, and lime kiln that operates 6,300 hours or more per year using a methodology described in Subpart MM. Under the methodology, the combined emissions from the recovery furnace, SDT, and lime kiln cannot exceed the combined emissions limits. Limits established using this methodology are not allowed to exceed applicable NSPS limits.

c. g/dscm, gram per dry standard cubic meter; gr/dscf, grains per dry standard cubic feet

d. Corrected to 8 percent oxygen

e. Corrected to 10 percent oxygen

Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

EPA has not published ACT guidelines for pulp and paper mills.

New Source Performance Standards

New Source Performance Standards (NSPS) typically apply, in both attainment and nonattainment areas, to new and reconstructed sources within specific source categories.

The requirements of 40 CFR §§60.280, *et seq.* (subpart BB) apply to kraft pulp mills that commence construction, modification or reconstruction after September 24, 1976. The NSPS standards do not apply to semichemical, mechanical or sulfite mills, which account for only a small segment of the industry's emissions. The kraft NSPS limit PM from recovery furnaces, SDTs and lime kilns.

The NSPS for kraft pulp mills appear in Table 8.10.

Power boilers used to generate electricity and steam at pulp and paper facilities are subject to the NSPS for industrial boilers (40 CFR §§60.40b, *et seq.* (subpart Db) and 40 CFR §§60.40c, *et seq.* (subpart Dc)), discussed in Chapter 6, *Industrial and Commercial Boilers.* Newer boilers are subject to limits for PM, SO_2 and NO_x emissions under these rules.

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, Best Available Retrofit Technology (BART) requirements for the control of $PM_{2.5}$ and $PM_{2.5}$-precursor emissions apply to certain large sources (in both attainment and nonattainment areas) that began operation between 1962 and 1997. The 26 source categories subject to BART requirements include kraft pulp mills. However, states have not yet developed BART rules under the regional haze program.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) requirements apply to new and modified large stationary sources in nonattainment areas, while Prevention of Significant Deterioriation (PSD) requirements apply to large stationary sources in attainment areas. Under these programs, plants in nonattainment areas are required to implement Lowest Achievable Emission Rate (LAER) controls and plants in attainment areas are required to implement BACT. Neither BACT nor LAER standards are codified because they are intended to reflect technology advancements as they occur.

Several permits were evaluated to determine what is generally considered BACT for lime kilns and recovery boilers.

As indicated above, "good design and operation practices" were determined to constitute BACT for lime kiln NO_x emissions control (Georgia DNR, 2003). Combustion controls (staged combustion) were determined to constitute

BACT for recovery boiler NO_x emissions control (North Carolina DENR, 2000). The Washington State Department of Ecology determined that good combustion practices (with an emissions limit of 95 ppm) constituted BACT for recovery furnace NO_x emissions control (Washington DEC, 2003).

In terms of SO_2, "no control" was determined to constitute BACT for a recovery boiler because of the high cost per ton removed with the application of a wet scrubber (North Carolina DENR, 2000; Georgia DNR, 2003). Recent BACT determinations are reported to limit SO_2 emissions to between 50 and 300 ppm (NESCAUM, 2005).

In terms of PM, the Georgia Department of Natural Resources concluded that an ESP with a 99.8 percent control efficiency constituted BACT for recovery boiler PM_{10} emissions control (Georgia DNR, 2003).

Maximum Achievable Control Technology

Maximum Achievable Control Technology (MACT) standards address emissions of hazardous air pollutants (HAPs) from new and existing sources in identified industry categories. They apply in both attainment and nonattainment areas to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs.

Under 40 CFR §§63.860, *et seq.* (subpart MM), PM emissions from kraft and sulfite pulp mills are regulated as a surrogate for HAPs. The subpart was developed by EPA as part of a "cluster rule" that covered both air emissions and water discharges from the pulp and paper industry. It includes standards for combustion sources at kraft and sulfite mills.

The owner or operator of an existing affected source or process unit must comply with the requirements in Table 8.11 no later than March 13, 2004. The owner or operator of a new affected source that has an initial start-up date after March 13, 2001 must comply with the requirements immediately upon start-up.

The new source MACT standards in Table 8.11 for recovery furnaces and lime kilns are based on the emissions levels achieved by units equipped with ESPs, while the standards for SDTs are based on emissions levels attained by units with wet scrubbers (EPA, 2001).

Clean Air Interstate Rule

In March 2005, EPA promulgated the Clean Air Interstate Rule (CAIR), which aims to address pollution transport that contributes to the nonattainment of eight-hour ozone and $PM_{2.5}$ air quality standards throughout the eastern U.S. States can choose how to achieve these reductions, but it is expected that most, if not all, states will opt to cap emissions according to budgets established by EPA and participate in EPA-administered emissions trading programs. None of the sources operating at pulp and paper mills that are discussed here can be included in any of the federal CAIR trading programs without disqualifying the state from participation in the federal trading regimes.

State and Local Policy Measures

With the updated MACT standards, EPA anticipates a national reduction in PM emissions and a reduction in SO_2 emissions as a secondary benefit of the PM control measures (Telander, 2002). While these reductions will reduce $PM_{2.5}$ emissions, there are additional measures states might consider.

Update PM Emissions Limits

There are MACT standards for PM emissions from recovery furnaces, lime kilns and SDTs. These standards are 40 to 85 percent more stringent for new sources than they are for existing sources. State and local regulators should evaluate the feasibility of requiring existing sources to meet more stringent standards. For example, at existing facilities, upgrades to ESPs and replacement of wet scrubbers with ESPs can significantly reduce PM emissions. Older model ESPs on recovery furnaces have collection efficiencies close to 90 percent, while newer model ESPs have collection efficiencies greater than 99 percent.

Limit SO_2 Emissions from Kraft Pulp Mills

As described in the section of this chapter that discusses regulatory authority, the kraft NSPS limit PM and TRS emissions, but not SO_2. However, some states and local areas, such as Washington and the Lane Regional Air Pollution Authority in Oregon, have established SO_2 emissions limits for recovery furnaces, lime kilns and other combustion devices at kraft pulp mills (WAC, 1991a; LRAPA, 2005).

States have also used BACT determinations to limit SO_2 emissions from individual kraft mills, with limits ranging from 50 to 300 ppm (NESCAUM, 2005).

Limit SO_2 Emissions from Sulfite Pulp Mills

States with sulfite pulping operations should consider limiting SO_2 emissions from these facilities. (In 2005, fewer than five sulfite pulping facilities remained in the U.S.) For example, Washington State limits the SO_2 emissions from sulfite pulp mills as summarized in Table 8.12.

Table 8.12

Washington State Emissions Guidelines for Sulfite Pulp Mills

Source	SO_2 Limit
Sulfite pulp mill with incineration	10 g/kg (20 lb/ton) air dried, unbleached pulp
Sulfite pulp mill without incineration	2 g/kg (4 lb/ton) air dried, unbleached pulp
Blow system	0.1 g/min (15 min average)/ kg (0.2 lb/min/ton) air dried, unbleached pulp
Recovery system and acid plant	800 ppm (hourly average)
Recovery systems after Jan 24, 1972	300 ppm (hourly average)
Emissions unit (except recovery system), blow system, or acid plant	1000 ppm (hourly average) @ 7% oxygen

Source: WAC, 1991b

Power Boilers

Power boilers are a major source of $PM_{2.5}$, SO_2 and NO_x emissions within the pulp and paper industry, and several options are available for reducing their emissions, as discussed in Chapter 6, *Industrial and Commercial Boilers*. In contrast, sources indicate that many of the technologies that are used to control fossil fuel boilers, such as SCR, are not feasible for the control of NO_x and SO_2 emissions from lime kilns and recovery furnaces (NESCAUM, 2005). At the same time, however, there are wide variations in the emissions generated by these sources. For example, NESCAUM reports SO_2 emissions from kraft recovery furnaces ranging from almost 0 to 300 ppm.

State and local agencies might consider facility-wide $PM_{2.5}$, NO_x and SO_2 limits for pulp and paper facilities, rather than simply limiting power boiler emissions, which appear to be the most feasible to control. This facility-wide approach would increase the flexibility that companies have to meet their compliance obligations, while reducing the overall costs of compliance and the aggregate emissions from these sources.

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Chapter 9

Cement Manufacturing

Introduction

Cement is used mainly as an ingredient in concrete, a ubiquitous building material. Cement acts as an adhesive when mixed with other ingredients, including water.

Cement is made in several steps: quarrying and crushing the raw materials; grinding the materials to a fine powder; heating the materials in a kiln to produce "clinker" (the intermediate products); and grinding the clinker, together with gypsum, to a fine powder.

Cement kilns produce over 80 percent of the sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions associated with cement manufacturing and over 40 percent of the particulate (PM) emissions. The kiln is a long, brick-lined cylindrical steel shell that is rotated with an electric drive. It is set at an angle, such that the material introduced at one end moves to the other end by gravity. At the lower end, burners, most of which fire coal, provide the heat that turns the raw materials into clinker.

In 2003, there were 116 portland cement plants in 37 states and two in Puerto Rico. California, Texas, Pennsylvania, Michigan, Missouri and Alabama were the six largest producers, with about one-half of the country's total production. After falling during the 1980s, cement production in the U.S. has grown over 25 percent since 1990. In 2003, cement plants in the U.S. produced approximately 87 million tons of portland cement and 4.5 million tons of masonry cement (which contains portland cement) (USGS, 2004).

Sector Profile

Most commonly, cement refers to a fine, powdery substance made of heated and pulverized limestone and clay that sets into a solid mass when mixed with water. This type of cement is also called "hydraulic cement" because it requires water to set. Cement is used mainly as an ingredient in concrete.

Buildings, dams, roadways and countless other structures are either made of, or supported by, concrete. Concrete has four ingredients: cement, water, coarse aggregate (e.g., rocks) and fine aggregate (e.g., sand). Cement reacts with the water to form a paste that hardens around the aggregates to make concrete. The proportions of ingredients vary based on the desired properties of the concrete. A typical batch of concrete is about 10–15 percent cement, 60–70 percent aggregate (coarse and fine) and 15–20 percent water (Portland Cement Association,

2005).

The most common type of cement is portland cement, so named by Joseph Aspdin, who patented it in 1824, because the color of the concrete he made with the cement reminded him of limestone quarried on the Isle of Portland in the English Channel (Portland Cement Association, 2005). Portland cement accounts for about 93 percent of the cement produced in the U.S. Of the remainder, about 5 percent is masonry cement and about 2 percent is blended.

The raw materials for portland cement come from four categories of substances: calcium carbonate and oxides of silicon, aluminum and iron. Limestone is most often used to fulfill the calcium carbonate requirement and is the predominant raw material in cement. There are more than 30 raw materials that can be used to fulfill the other mineral requirements. The mix of raw materials used depends on their availability and on the desired properties of the cement (EPA, 2000; Davis, 2000).

Both masonry cement and blended cements are produced in portland cement plants and contain portland cement as an ingredient (Davis, 2000). Masonry cement commonly contains a finely ground mixture of portland cement, limestone and air-entraining agents (EPA, 2000). Blended cements are produced by blending two or more types of fine materials, primarily portland cement, ground granulated blast furnace slag, fly ash, silica fume, calcined clay, hydrated lime and pre-blended combinations of those materials (Portland Cement Association, 2005).

The process of making portland cement involves (EPA, 2000):

- quarrying and crushing raw materials;
- proportioning and grinding the raw materials to a high degree of fineness;
- pyroprocessing (the heating process that takes place in the cement kiln) the crushed raw materials in a rotary kiln to produce clinker (intermediate products of fused raw materials); and
- grinding the clinker to a fine powder, along with an appropriate proportion of gypsum (calcium sulfate, a white or clear sulfate of lime also used in plaster) to produce cement.

The American Society for Testing and Materials (ASTM) specifies eight different types of portland cement. The types have various physical and chemical attributes and are designated by Roman numerals and letters (Types I, IA, II, IIA, III, IIIA, IV and V). Type I is considered a general purpose cement, suitable for all uses, while the other types have specific qualities such as rapid setting, high strength and resistance to high sulfate soil or groundwater that can damage cement (Portland Cement Association, 2005). The different types of portland cement are made by using appropriate combinations of raw materials, blending the intermediate product (clinker) with specified amounts of

Table 9.1

Cement Manufacturing Sector Emissions

	$PM_{2.5}$		SO_2		NO_x	
Process Type	**Tons/year**	**Percent of Sector Emissions**	**Tons/Year**	**Percent of Sector Emissions**	**Tons/Year**	**Percent of Sector Emissions**
Dry kilns	7,209	31%	64,467	35%	90,000	39%
Wet kilns	1,232	5%	80,288	44%	71,638	31%
Preheater/precalciner kilns	866	4%	15,076	8%	23,904	10%
Preheater kilns	165	1%	NA	NA	9,003	4%
In-process fuel	647	3%	12,548	7%	14,622	6%
Material handling, storage and grinding	5,621	24%	NA	NA	NA	NA
Manufacturing (not classified)	2,181	9%	NA	NA	NA	NA
Clinker cooler	2,028	9%	NA	NA	NA	NA
Other	3,123	14%	11,787	6%	24,522	10%
Total	23,072	100%	184,167	100%	233,689	100%

Source: EPA, 2005, based on facilities listed under SIC code 3241

gypsum (calcium sulfate) and grinding the final product to a specific degree of fineness (EPA, 2000).

There are four types of kilns used at cement and clinker facilities: wet kilns, long dry kilns, dry kilns with a suspension preheater and dry kilns with a precalciner system. Wet kilns are the least energy-efficient and are becoming less common; their use has declined by 77 percent since 1974. All kilns built since 1975 (56 percent of existing U.S. capacity) use the dry process. About 81 percent of the cement produced in the U.S. is manufactured using the dry process (Portland Cement Association, 2005).

Emissions

Most SO_2 and NO_x emissions from the cement manufacturing process come from the operation of the kiln. About 41 percent of fine particulate matter ($PM_{2.5}$) emissions come from the kiln.

PM emissions also result from handling, storing and grinding both raw materials and the intermediate product (clinker). Historically, the cement industry has focused on control of PM emissions, both because it is in the financial interest of manufacturers to capture and recycle PM back into the process and because federal standards limit PM emissions. Almost all stages of the manufacturing process include particle capture devices, most frequently fabric filters or electrostatic precipitators (ESPs). Table 9.1 summarizes $PM_{2.5}$, SO_2 and NO_x and emissions from the cement manufacturing industry.

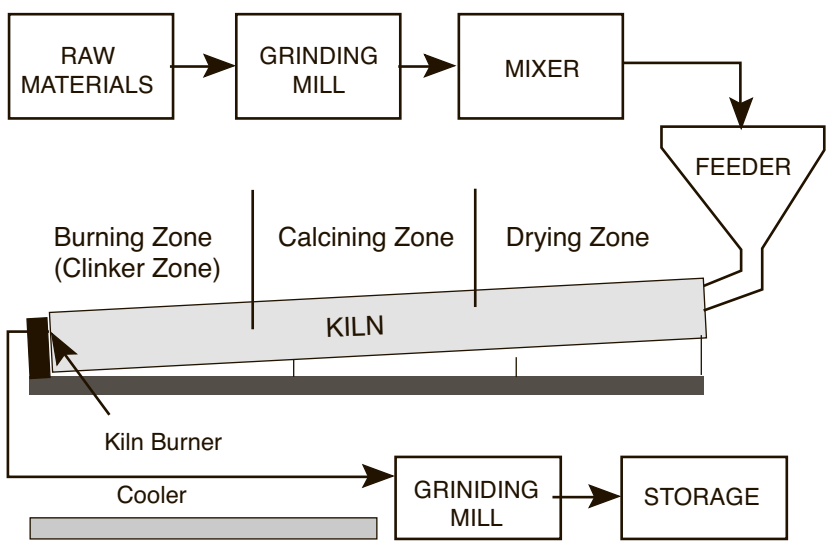
Note that all new kilns are preheater/precalciner kilns; as their numbers increase, their share of the overall emissions attributable to this sector will also increase while the share of emissions attributable to dry and wet kilns will decline.

Cement Manufacturing

Facilities produce portland cement by grinding and then heating a mixture of calcium carbonate with oxides of silicon, aluminum and iron to a high temperature. The raw materials used to create this mixture depend on the mineral content of available sources. The materials can include clay, shale, sand, slag, fly ash, or waste rock from a limestone quarry for silicon or aluminum; and mill wastes, pyrite, or iron ore for iron. Calcium carbonate, usually from quarried limestone, is the most important ingredient and plants are frequently located near limestone sources (EPA, 2000).

Figure 9.1 shows a basic cement manufacturing process. Raw materials are gathered and then crushed in a grinding mill, after which they can be stored separately for later use or immediately combined in proper proportions in the mixer. Raw materials can be introduced in a dry form or in a 30–40 percent liquid slurry. The mixture of raw materials enters the kiln through the feeder, where the heating process begins. A cement kiln is a long rotating tube that is heated at one end (EPA, 1994). As the material moves through the kiln, it is gradually heated and turned into clinker. Clinker pieces are hard, gray spheres with diameters of one-eighth to 2 inches, composed of compounds resulting from the chemical reactions that take place in the kiln (Davis, 2000).

Fig. 9.1 Basic Cement Manufacturing Process



Source: EPA, 1995

Table 9.2

Controlled PM Emission Factors for Different Kiln Types[a]

Cement Kiln Type	Control Device	PM Emission Rate[b] (lb/ton of clinker)
Wet kiln	ESP	0.77
	Fabric filter	0.46
Long dry kiln	ESP	1.0
	Fabric filter	0.20
Preheater kiln	ESP	0.26
	Fabric filter	0.25
Preheater/precalciner kiln	ESP	0.048
	Fabric filter	0.21

a. This table shows controlled as opposed to uncontrolled emissions because PM emissions from kilns are normally controlled. Uncontrolled emissions are approximately 130 pounds per ton (lb/ton) of clinker from wet process kilns and 250 lb/ton of clinker from preheater kilns.

b. Includes filterable PM (collected on or before the filter of an EPA Method 5 sampling train); does not include condensable PM (collected in the impinger portion of a PM sampling train).

Source: EPA, 1995

After leaving the kiln, the clinker is cooled to lock in its mineral properties. Clinker can be stored or immediately ground into the final product. During the final grinding, approximately 5 percent by weight of gypsum is added to the clinker to control the setting time of the cement (Davis, 2000).

Raw material extraction, handling and grinding; and clinker cooling and grinding can result in fugitive PM emissions. If a manufacturer uses the raw materials in a slurry form (as in the wet process), PM emissions are very low for the raw material grinding and handling portions of the manufacturing process. Since the final mix (in either the wet or dry process) is a finely ground powder, it is a source of fugitive PM emissions during handling and transport.

The largest source of emissions in cement manufacturing, and the centerpiece of the process, is the kiln. As shown in Table 9.1, over 40 percent of PM emissions, about 87 percent of SO_2 emissions and about 84 percent of NO_x emissions are associated with cement manufacturing are generated by cement kilns (EPA, 1994). The kiln is a long (150–750 feet long, 10–25 feet in diameter) brick-lined cylindrical steel shell that is rotated with an electric drive at one to three revolutions per minute (EPA, 2000). The kiln is set at an angle that allows the material fed in at one end to move by gravity to the other end. At the lower end, burners provide the heat that turns the raw materials into clinker.

The burners used to heat the kiln use coal or other fossil fuels (EPA, 1994). In 1998, about 82 percent of cement plants were coal-fired, about 3 percent burned natural gas and approximately 1 percent used oil. The remaining 14 percent of the plants used other combinations of fuel, such as waste materials supplemented with coal or another fossil fuel (EPA, 2000).

There are four processing stages, regardless of the type of kiln that is employed: evaporation, dehydration, calcination and reaction. Both the evaporation and dehydration stages take place in the drying zone shown in Figure 9.1. During evaporation, the material is heated to 212 degrees Fahrenheit (°F) to remove water. The material temperature is then increased during dehydration to approximately 800°F. As the material moves through the kiln, temperatures increase during the calcination stage to 1,600°F–1,800°F, and carbon dioxide and calcium oxides are formed. The heating of the material to a temperature that is high, but below the melting or fusing point, results in the loss of moisture and the decomposition of carbonates and other compounds. Finally, during the reaction stage (in the burning zone in Figure 9.1), the calcium oxides are heated to 2,750°F to form clinker (EPA, 2000).

As noted previously, of the four types of cement kilns, wet kilns are the least energy-efficient and their use has declined dramatically. Wet kilns and long dry kilns use older technologies and have only one fuel combustion zone. Newer preheater and precalciner kilns have two combustion zones and use dry raw materials; some older kilns with preheaters have only one combustion zone. Kilns with two combustion zones suppress some thermal

Table 9.3

Controlled PM Emissions Factors for Other Processes

Process	Control Device	PM Emissions Rate (lb/Ton of Clinker)
Clinker cooler	ESP	0.096
	Fabric filter	0.13
	Gravel bed filter	0.21
Raw mill	Fabric filter	0.012
Raw mill feed belt	Fabric filter	0.0031
Raw mill weigh hopper	Fabric filter	0.019
Raw mill air separator	Fabric filter	0.032
Finish grinding mill	Fabric filter	0.0080
Finish grinding mill feed belt	Fabric filter	0.0024
Finish grinding mill weigh hopper	Fabric filter	0.0094
Finish grinding mill air separator	Fabric filter	0.028
Primary limestone crushing	Fabric filter	0.0010
Primary limestone screening	Fabric filter	0.00022
Limestone transfer	Fabric filter	0.000029
Secondary limestone screening	Fabric filter	0.00031

Source: EPA, 1995

NO_x formation.[1]

The main process difference between a wet kiln and a long dry kiln is that a 30–40 percent liquid slurry of raw materials is introduced to the feeder in a wet kiln, while a dry feed of raw materials is introduced in a long dry kiln. Historically, producers using raw materials with higher moisture content (15–20 percent) favored the wet process (EPA, 2000). However, there have been significant efficiency advancements in the dry process, especially with the use of suspension preheater and precalciner systems. Accordingly, new kilns always use the dry process.

Suspension preheater systems preheat the dry feed before it enters the kiln, using a four-stage system of funnels, called cyclones.[2] As the material moves through each of the four cyclones it is heated, so that it is already 40–50 percent calcined by the time it enters the kiln. This process allows the kiln to be shortened and significantly increases the efficiency of the system compared to wet and long dry kilns (EPA, 2000).

In a precalciner system, a second burner is used to carry out calcination in a separate vessel. The raw material is almost 95 percent calcined before entering the kiln. The precalciner system is the most energy-efficient cement-making process and is used by all newer cement plants in the U.S. (EPA, 2000).

Precalciner systems are much larger than the other systems. Precalciner systems in the U.S. have an average capacity of about 869,000 metric tons per year, compared to 307,000 metric tons per year, 265,000 metric tons per year, and 406,000 metric tons per year for long wet, long dry, and suspension preheater kilns, respectively. This size differential has led to a decline in the number of kilns but an increase in total U.S. cement-making capacity (EPA, 2000).

Kiln PM emissions come from the exhaust gases generated during the heating process. These emissions consist of a mix of particles from the various stages of material processing and from the combustion of the heat source. Collectively, the particulates are known as cement kiln dust (CKD) and are collected with ESPs or fabric filter for use or disposal. Handling CKD can result in fugitive dust

1. Four types of NO_x form in cement manufacturing: fuel NO_x, thermal NO_x, feed NO_x and prompt NO_x. Fuel NO_x forms from the oxidation of nitrogen in the fuel; thermal NO_x results from the oxidation of nitrogen in the process air at high temperatures; feed NO_x comes from the heating of the raw materials; and prompt NO_x is generated by the reaction of fuel-derived radicals with elemental nitrogen in a hydrocarbon flame. Of these four, fuel NO_x and thermal NO_x are the most significant. Prompt NO_x is particularly insignificant to total NO_x formation (Greer, 2003).

2. Preheater cyclones should not to be confused with cyclones that can be used for PM removal.

Table 9.4

Uncontrolled SO_2 Emission Factors for Different Kiln Types

Cement Kiln Type	SO_2 Emission Rate (lb/Ton of Clinker)
Wet kiln	8.2
Long dry kiln	10
Preheater kiln	0.55
Preheater/precalciner kiln	1.1

Source: EPA, 1995

emissions (Davis, 2000).

Tables 9.2 and 9.3 present the best available information on PM emissions from this industry sector. Note that the data in both of these tables are from AP-42 and have a rating of "D" or "E" on a rating scale in which "A" denotes excellent data quality and "E" denotes poor data quality. This reflects the difficulty of obtaining emissions rates for cement kilns.

SO_2 and NO_x emissions rates for cement kilns vary widely throughout the industry. Continuous monitoring has shown that emissions from a single source can vary by an order of magnitude over time (Davis, 2000). The amount of sulfur in the raw materials and in the fuel determines the amount of SO_2 emissions, while fuel combustion is a major source of NO_x emissions (Davis, 2000). Partly as a result of these uncertainties, there are as yet no federal New Source Performance Standards (NSPS) for SO_2 or NO_x emissions from cement plants.

Tables 9.4 and 9.5 present the best available information on SO_2 and NO_x emissions from this industry sector. The data in Table 9.4 are also from AP-42 and have a rating of "C" or "D."

Table 9.5 shows uncontrolled NO_x emissions as reported in EPA's 1994 "Alternative Control Techniques (ACT) Document—NO_x Emissions from Cement Manufacturing" and similar data from AP-42. The ACT data were collected from 51 cement kilns (22 wet kilns, ten long dry kilns, ten preheater kilns and nine precalciner kilns). The AP-42 data all have ratings of "D." According to EPA's year 2000 review of NO_x control technologies, the ACT data provide the best available information on cement manufacturing emissions (EPA, 2000).

Emissions Control Opportunities

Most $PM_{2.5}$, SO_2 and NO_x emissions from cement manufacturing are produced by cement kilns. Additional $PM_{2.5}$ emissions come from the handling, grinding and storing of raw materials, clinker and the final product.

Over the last half century, the cement-manufacturing industry has moved away from wet kilns in favor of more efficient preheater and precalciner dry kilns. The movement towards more efficient kilns has had the indirect benefit of reducing uncontrolled emissions of SO_2 and NO_x per ton of clinker (reflected in the average emissions rates shown in Tables 9.4 and 9.5). However, the change from wet to dry kilns has increased the potential for PM emissions, since the raw materials are more likely to produce fugitive dust emissions. Industry-wide use of fabric filters and ESPs has controlled a large percentage of these emissions.

Additional process modifications to increase energy efficiency, such as instruction and training of kiln operators and consistent raw material and coal preparation, can reduce PM, SO_2 and NO_x emissions. However, many of these process modifications are dependent on site-specific characteristics, making it difficult to provide general estimates of cost-effectiveness (EPA, 2000).

Controlling PM Emissions

Federal standards limit PM emissions; moreover, it is also in the financial interest of manufacturers to capture and recycle PM. As a result, almost all stages of the manufacturing process include particle capture devices

Table 9.5

Uncontrolled NO_x Emission Factors for Different Kiln Types

Cement Kiln Type	ACT NO_x Emission Rate (lb/Ton of Clinker)	Range of NO_x Emissions (lb/Ton of Clinker)	AP-42 NO_x Emission Rate (lb/Ton of Clinker)
Wet kiln	9.7	3.6-19.5	7.4
Long dry kiln	8.6	6.1-10.5	6.0
Preheater kiln	5.9	2.5-11.7	4.8
Preheater/precalciner kiln	3.3	0.9-7.0	4.2

Source: EPA, 1994, 1995a

(most frequently, fabric filters or ESPs).

PM emissions come from combustion sources in the kiln and from material handling, grinding and storage operations (with the latter producing fugitive emissions). Control devices available for combustion sources are fabric filters and ESPs. Both fabric filters and ESPs are discussed in detail in Chapter 5, *Boiler Technologies*. As they do for boilers, the control devices remove PM from an exhaust stream. However, given the high pollutant loadings associated with cement operations, control devices must be relatively large compared to boiler control devices. Both fabric filters and ESPs have PM collection efficiencies of 95–99 percent (LADCO, 2005).

Control devices for sources of fugitive PM emissions are also primarily fabric filters and ESPs. In order to capture fugitive PM emissions, material handling, grinding and storage operations must be enclosed and vented to the control device (LADCO, 2005). Dust collectors can also control non-kiln PM sources. Dust collectors are similar to fabric filters but have temperature restrictions (a limit of 200°F or 400°F, depending on the filter media) and are effective only on low-flow applications. However, dust collectors require less room than fabric filters or ESPs and are therefore useful for areas with space restrictions. Dust collectors have a PM control-efficiency of over 99 percent (LADCO, 2005).

If there is adequate space for ductwork and the control device is properly sized, multiple PM sources can be directed to and controlled by one control device (LADCO, 2005). If fugitive emissions sources are not enclosed, they can be controlled using wind screens, water sprays, equipment enclosures and other fugitive dust controls (EC, 2000).

Where existing control device collection-efficiencies are lower than desired, several performance improvements are possible. These include rebuilding ESPs, possibly with a larger number of collection areas and increased treatment times, and using fabric filters in combination with ESPs. In the settlement of a recent lawsuit in Wyoming, one cement manufacturer agreed to replace an existing ESP with a fabric filter to reduce PM emissions that were causing a kiln to exceed opacity limits (Associated Press, 2005).

Controlling SO_2 Emissions

The clinker-making process itself provides substantial SO_2 control and has historically been considered the best available technology for controlling cement kiln SO_2 emissions (Davis, 2000). A number of Best Available Control Technology (BACT) determinations since 2000 have listed "process" as the technology for controlling SO_2 emissions (Sumter Cement Company, 2005). Cement kiln systems have highly alkaline internal environments that can absorb up to 95 percent of potential SO_2 emissions. However, variations in the kiln feed can lower the sulfur absorption rate to 70 percent (EPA, 1995a). This makes raw material selection an important consideration in controlling SO_2 emissions (Linero, 2001). Where raw material sulfur is very low and preheater/precalciner kilns are used, almost all SO_2 emissions are scrubbed during the manufacturing process and emissions are effectively zero (Linero, 2005b).

Fabric filters on cement kilns are also reported to absorb some SO_2, although an absorbing reagent—such as calcium oxide, an intermediary product of clinker formation—must be present in the filter cake for SO_2 capture to occur (EPA, 1995b).

As much as 50 percent of the SO_2 can be removed from the kiln exhaust gases when this gas stream is used in drying raw materials. When this approach is used, moisture and calcium carbonate are simultaneously present for sufficient time for the chemical reaction with SO_2 to occur (EPA, 1995b).

Cement manufacturing plants also use flue gas control technologies such as flue gas desulfurization (FGD). Advanced, wet and dry FGD are all possible SO_2 control technologies. Advanced FGD is estimated to have a control-efficiency of 95–99.5 percent and, as an added benefit, generates gypsum—a mineral needed in the final stages of cement production—as a byproduct. However, no cement manufacturing units in the U.S. currently use advanced FGD (LADCO, 2005). Descriptions of all three types of FGD systems are included in Chapter 5, *Boiler Technologies*. Note that wet and dry FGD systems are sometimes referred to as wet and dry scrubbers.

Wet FGD has an SO_2 control-efficiency of 90–99 percent. Using wet FGD in the cement industry requires more care than using this technology on utility boilers because of particulate build-up and clogging. Most of these problems can be avoided if the FGD is installed downstream of an efficient fabric filter (LADCO, 2005). Five cement plants in the U.S. currently operate with wet scrubbers to control SO_2 emissions (Sumter Cement Company, 2005).

Dry FGD has an SO_2 control-efficiency of 90–95 percent. Most dry FGD systems are installed on preheater or preheater/precalciner kilns. Dry FGD is not recommended for wet kilns because it can cause corrosion problems in that setting (LADCO, 2005).

Lime spray injection is an alternative to traditional wet and dry scrubbers and has been proposed as a less expensive alternative for reducing SO_2. In this process, a fine mist of lime product is sprayed into the exhaust gas before it reaches the PM removal equipment. The lime droplets

Table 9.6

SO_2 Control Technology Cost Estimates for Long Dry Kilns

Control Technology	Size (Tons of Clinker per Year)	Capital Costs ($ Million)	Annual Costs ($ Million)	Cost Effectiveness ($ per Ton)
AFGD	300,000	7.3–22.9	3–6	2,000–4,000
	600,000	14.1–45.9	6.1–11.9	
	1,200,000	28.1–91.6	12.1–23.7	
Wet FGD	300,000	2.43–36.5	3–9	2,000–6,200
	600,000	4.9–73.0	6–18.4	
	1,200,000	9.5–142.5	11.9–36.8	
Dry FGD	300,000	1.45–37.0	3–9	1,900–7,000
	600,000	2.9–84.9	5.5–20.0	
	1,200,000	5.6–165.5	10.7–38.9	

Source: Lake Michigan Air Directors Consortium, 2005

react with the SO_2 and are captured by the fabric filter or ESP (Linero, 2001).

The Lake Michigan Air Directors Consortium (LADCO) conducted an economic analysis of Best Available Retrofit Technologies (BART) for controlling emissions of SO_2 (and other pollutants) from long dry kilns and preheater kilns using model facilities. Tables 9.6 and 9.7 present results from the LADCO analysis.

Note that the cost estimates in Tables 9.6 and 9.7 are based on modeling commissioned by LADCO in support of its "Cement Best Available Retrofit Technology (BART) Engineering Analysis."[3] The LADCO analyses included a detailed review of available technologies and control costs. As a result of the modeling assumptions and the inclusion of a wide spectrum of operating conditions, some of the cost ranges are extremely large. LADCO emphasizes that site-specific analyses are required to assess facility-level costs accurately.

Controlling NO_x Emissions

There are three categories of NO_x emissions controls for cement manufacturing facilities: process modifications, combustion controls and post-combustion controls.

It is important to note that preheater and precalciner kilns

3. The LADCO website includes detailed information on the modeling. Supporting documentation is available at: www.ladco.org/Regional_Air_Quality.html.

Table 9.7

SO_2 Control Technology Cost Estimates for Preheater Kilns

Control Technology	Size (Tons Clinker per Year)	Capital Costs ($ Million)	Annual Costs ($ Million)	Cost Effectiveness ($ per Ton)
AFGD	300,000	4.5–14.5	1.1–3.0	13,600–38,600
	600,000	8.9–29.0	2.3–5.9	
	1,200,000	17.8–58.0	4.5–11.8	
Wet FGD	300,000	1.5–23.1	0.9–4.8	9,700–64,600
	600,000	3.1–46.3	1.6–9.5	
	1,200,000	6.2–92.5	3.2–18.9	
Dry FGD	300,000	0.9–26.3	0.9–5.4	10,000–72,800
	600,000	1.8–52.6	1.7–10.6	
	1,200,000	3.6–105.2	3.3–21.0	

Source: Lake Michigan Air Directors Consortium, 2005

have lower NO_x emissions rates than wet and long dry kilns, because preheater and precalciner kilns have two combustion zones. With two combustion zones, the fuel combustion temperatures for the preheating or precalcining step can be lower than the combustion temperatures for the kiln firing; at lower temperatures, less NO_x is formed.

Because of the different firing configurations, not all controls are applicable to all kiln types. While the process modifications discussed below can be used on any of the four kiln types, mid-kiln firing is effective only on wet or long dry kilns. Low-NO_x burners (LNBs) are effective only on indirect-fired kilns.

Process Modifications. Many process modifications focus on improving energy efficiency, thereby resulting in a reduction of all emissions. Two process controls that specifically target NO_x emissions are continuous emissions monitoring systems (CEMS) and a patented feed modification technique known as the CemStar Process (EPA, 2000).

CEMS allow operators to continuously monitor oxygen and carbon monoxide (CO) emissions in cement kiln exhaust gases. The levels of these gases indicate the amount of excess air in the combustion zone. At a given excess air level, NO_x emissions increase as the temperature increases. Knowing the excess air level allows operators to maintain a lower temperature and thereby minimize NO_x creation. Additionally, some studies indicate that reducing excess air by half can reduce NO_x emissions by about 15 percent (EPA, 2000).

The CemStar Process, developed and patented by TXI Industries, is a raw feed modification process that can reduce NO_x emissions by about 30 percent and increase production by approximately 15 percent. It involves the addition of a small amount of steel slag to the raw kiln feed. Steel slag has a chemical composition similar to clinker and many of the chemical reactions required to convert steel slag to clinker take place in the steel furnace. By substituting steel slag for a portion of the raw materials, facilities can increase thermal efficiency and thereby reduce NO_x emissions. In 2000, 11 facilities in the U.S. were using or planning to incorporate CemStar (EPA, 2000).

Combustion Controls. Combustion modifications to reduce NO_x emissions from cement kilns fall into two categories: staged combustion of air and staged combustion of fuel. The primary staged air combustion controls are LNBs. LNBs reduce the formation of NO_x by staging the combustion process, producing fuel-rich and fuel-lean zones within the combustion flame (EPA, 1999). LNBs are commonly used on electric generating unit and industrial boilers and are described in more detail in Chapter 5, *Boiler Technologies*.

LNBs have been used on cement kilns for over 20 years. However, they are effective only on indirect-fired kilns, which account for a small share of the existing U.S. kiln population. The Portland Cement Association reported to EPA in 2000 that a survey of 81 percent of the kilns in the U.S. indicated that 14 percent of these kilns use some type of LNB (EPA, 2000). New kilns are designed to incorporate indirect firing and LNBs to reduce NO_x emissions. LNBs with indirect firing have been determined to be BACT for several new facilities since 2000 (Sumter Cement Company, 2005).

As compared to direct-fired systems, indirect-fired systems use less air to convey fuel to the combustion zone; they therefore add less air to the combustion zone. This reduces the oxygen available for NO_x formation. Conversion of a direct-fired kiln to an indirect-fired kiln involves adding particle separation equipment, such as a cyclone or a baghouse and a fan, to provide the air used to transport the powdered coal from storage to the kiln. In addition to reducing NO_x formation, indirect firing increases overall energy efficiency by allowing a greater proportion of hot clinker-cooler air to be used as secondary combustion air (EPA, 2000).

LNBs can be paired with flue gas recirculation (FGR) for additional NO_x reductions of 15–38 percent (EPA, 2000). FGR is a temperature reduction process that involves recirculating a portion of the flue gas with the combustion air. Recirculation lowers the combustion temperature, reducing NO_x formation (EPA, 1999). FGR is not effective on all kilns because of the high temperatures required to create clinker. If use of FGR lowers the temperature too much, the process can be compromised (EPA, 2000).

All new cement kilns are of the preheater or precalciner variety. These take advantage of staged fuel combustion by having two firing zones. One zone is located before the place where the material enters the kiln; the other zone is at the end of the kiln. These kilns are more fuel efficient and have lower NO_x formation than wet or long dry kilns. Multi-staged combustion can be used with preheater/precalciner kilns to reduce NO_x formation. Multi-staged combustion reduces NO_x through the use of staged air or staged fuel combustion at the burning locations (Sumter Cement Company, 2005).

Wet and long dry kilns can also make use of staged fuel combustion by injecting solid fuel into the calcining zone of the kiln. This technique is called mid-kiln firing. Mid-kiln firing requires solid and slow burning fuels—most systems use whole tires, although at least one uses baled industrial waste. The fuel is dropped into the calcining zone, where it burns at the calcining temperature (between 1,600°F and 1,800°F) instead of at the clinker-forming temperature (2,750°F). The lower combustion temperature reduces NO_x formation. The average NO_x reduction using

Table 9.8

NO_x Control Technology Performance

Technology	Average Emissions Reduction (%)	Range of Emissions Reductions (%)	Average Annual Cost-Effectiveness ($ per Ton NO_x Reduced)[a]	Range of Annual Cost-Effectiveness ($ per Ton NO_x Reduced)[a]
CemStar	33	25–40	550	ND
Indirect firing with a low-NO_x burner[b]	27	4–47	440	300–620
Long dry kiln with low-NO_x burner[c]	40	ND	ND	166–1,299
Preheater kiln with low-NO_x burner[c]	40	ND	ND	175–1,201
Mid-kiln firing (wet kilns only)	41	28–59	55	(460)–730
Mid-kiln firing (dry kilns only)	33	11–55		
SNCR[d]	40	10–50	1,750	1,000–2,500
SNCR[e]	ND	31–83	ND	605–700
SCR[f]	90	85–95	2,700	ND
SCR[e]	ND	31–83	ND	530–1,135
Long dry kiln with SCR[c]	ND	70–90	ND	586–1,902
Preheater kiln with SCR[c]	ND	70–90	ND	504–1,232

ND = No Data

a. In 1997$, except where noted.

b. Converting a direct-fired kiln to an indirect-fired kiln and retrofitting with an LNB roughly doubles the cost effectiveness.

c. From LADCO, 2005. Cost-effectiveness in 2004$.

d. These estimates are based on the use of an SNCR technology called NO_x-OUT® at three facilities.

e. From Haug, 2002. Emissions reductions range includes all data from the study. Cost-effectiveness in 2002$. Junker, 2000 supports SNCR emissions reductions of 80-85%.

f. From European Commission, 2000; based on one demonstration project in the Netherlands. Cost effectiveness in 2000$.

Source: EPA, 2000; European Commission, 2000; Huag, 2002; LADCO, 2005

this method is 41 percent for wet kilns and 33 percent for long dry kilns (EPA, 2000).

Tire-derived fuel can also be used in precalciners in place of all or some portion of the fossil fuel used for combustion. In one case, NO_x emissions reductions of about 30 percent occurred when tire-derived fuel was substituted for 47 percent of the coal (EPA, 2000).

Post-combustion Controls. The potential flue gas treatments for reducing NO_x emissions are selective catalytic reduction (SCR) and SNCR. Both of these technologies are commonly used on electric generating and industrial boilers and are described in more detail in Chapter 5, *Boiler Technologies*. SCR reduces NO_x emissions by chemical reduction, using a reducing agent (ammonia) and a catalyst. SCR is currently in operation on a cement kiln in Germany (Solnhofer Portland Cement Plant) and has been used on pilot cement plants in other parts of Europe. It is not, however, currently used on kilns in the U.S. (Sumter Cement Company, 2005).

Data from the SCR pilot projects in Europe suggest reduction efficiencies of 85–95 percent. One study from the Netherlands reported a cost-effectiveness of €2,500 per metric ton of NO_x (about $2,700 per ton) in late October 2005 (EC, 2000). The BART analysis conducted by LADCO estimated an SCR removal efficiency of 70–90 percent, and cost-effectiveness of $600–$1,900 per ton of NO_x removed on long dry kilns, and $500–$1,200 per ton of NO_x removed on preheater kilns (LADCO, 2005).

A review conducted by the Solnhofer Portland Cement Company with the German Federal Environmental Office (the organization that funded the project) and Lurgi (designers of the SCR equipment used in the project) compared the costs of SCR and SNCR on a preheater kiln (Haug, 2002). According to the results of that study, the costs of an SNCR system that reduces NO_x emissions from 5.4 pounds per ton (lb/ton) of clinker to 3.6 lb/ton of clinker is €560 per ton of NO_x ($605 per ton of NO_x). The same review estimated the cost to reduce NO_x with an SNCR to 2.3 lb/ton of clinker and 0.9 lb/ton of clinker to be approximately €590–€650 per ton of NO_x ($640–$700 per ton of NO_x), respectively. By contrast, the review found that the cost of control by SCR per ton of pollutant decreased as the reductions increased, from approximately €1,050–€490 per ton of NO_x ($1,135–$530 per ton of NO_x) for reductions to 3.6 lb/ton and 0.9 lb/ton of clinker, respectively. The costs of control with an SNCR and SCR are equal when facilities reduce emissions to 2.3 lb/ton of clinker (about a 56% reduction in NO_x).

SNCR also relies on the reduction of NO_x by exposure to a reducing agent (such as ammonia), but it does not involve the use of a catalyst. The catalyst in an SCR increases the reaction rate; without the catalyst, temperature becomes very important. The temperature range needed for SNCR occurs in the middle of wet and long dry kilns. It is difficult to inject the reducing agents in the middle of the kiln at a continuous rate, which makes it difficult to use SNCR on wet and long dry kilns. On preheater and precalciner kilns, the temperatures at the cooler end of the kiln and in the preheating area are appropriate for SNCR use (EPA, 2000).

As of 2000, at least 18 kilns in Europe had installed SNCR (EC, 2000). By 2004, there were approximately 32 SNCR installations in Germany alone (Verein Deutcher Zementwerke, 2004). Currently, only one full-size kiln in the U.S. has installed SNCR. Average control-efficiencies for these installations are 50–60 percent (Linero, 2005a). The Florida Department of Environmental Protection (DEP) has recently approved at least three cement permit applications that include SNCR with LNBs and multi-stage combustion as BACT (Florida DEP, 2005).

Table 9.8 summarizes the potential NO_x reductions and cost-effectiveness associated with control technologies discussed in this section.

Regulatory Authority

As we have indicated in Chapter 4, which discusses the Clean Air Act in more detail, the states and the federal government share responsibility for regulating criteria pollutants (e.g., PM, SO_2, NO_x) from stationary sources like those operating at portland cement plants. The federal government has more of the burden of regulating toxic air pollutants, but—for both criteria pollutants and air toxics—states and local areas are free under federal law to adopt more stringent standards than the Clean Air Act requires.

This section describes federal standards that apply under the Clean Air Act to pollutant emissions from portland cement plants.

Reasonably Available Control Technology

State Implementation Plans (SIPs) for areas that are in nonattainment of federal ambient air quality standards must include Reasonably Available Control Technology (RACT) requirements for new and existing sources. RACT standards are non-binding federal guidelines; for Volatile Organic Compounds (VOCs), the guidelines are known as Control Technique Guidelines (CTGs). Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

ACT guidelines applicable to NO_x emissions from cement manufacturing can be found at www.epa.gov/ttn/catc/products.html#aptecrpts. SIP writers should bear in mind that although the ACTs may provide a starting point, they are unlikely to represent the actual RACT limits that states will choose to adopt. That is the case because RACT standards are intended to reflect advancements in technology, and most of the ACTs are outdated—some, seriously so. Additionally, the ACTs are guidelines only, not actual federal limits. Even if they were federal limits, many states would have the leeway to enact more stringent standards. And since the ACTs do not set limits, even state prohibitions against enacting more stringent standards may be inapplicable.

New Source Performance Standards

The NSPS typically apply, in both attainment and nonattainment areas, to new, modified and reconstructed sources within specific source categories.

Under the Clean Air Act, 40 CFR §§ 60.60, *et seq.* (subpart F), NSPS requirements apply to portland cement plants that commenced construction, modification or reconstruction after August 17, 1971. The following sources at portland cement plants are covered: kilns, clinker coolers, raw mill systems, finish mill systems, raw mill dryers, raw material storage, clinker storage, finished product storage, conveyor transfer points, and bagging and bulk loading and unloading systems.

The NSPS for portland cement plants appear in Table 9.9.

Table 9.9

NSPS for Portland Cement Plants (40 CFR §§60.60, *et seq.* (subpart F))

Source	Particulate Matter Limit	Opacity Limit
Kiln (wet or dry)	0.15 kg/metric ton of feed (dry basis) (0.30 lb/ton)	20%
Clinker cooler	0.050 kg/metric ton of feed (dry basis) to the kiln (0.10 lb/ton)	10%
Any covered plant source besides the kiln or the clinker cooler[a]		10%

a. Raw mill systems, finish mill systems, raw mill dryers, raw material storage, clinker storage, finished product storage, conveyor transfer points, and bagging and bulk loading and unloading systems.

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, BART requirements for the control of $PM_{2.5}$ and $PM_{2.5}$-precursor emissions apply to certain large sources (in both attainment and nonattainment areas) that began operation between 1962 and 1997. The 26 source categories subject to BART requirements include portland cement plants. However, states have not yet developed BART rules under the regional haze program.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) requirements apply to new and modified large stationary sources in nonattainment areas, while Prevention of Significant Deterioration (PSD) requirements apply to large stationary sources in attainment areas. Under these programs, plants in nonattainment areas are required to implement Lowest Achievable Emission Rate (LAER) controls and plants in attainment areas are required to implement BACT. Neither BACT nor LAER standards are codified because they are intended to reflect technology advancements as they occur.

There have been a number of BACT determinations for cement kilns since 2000. Many of the BACT determinations for NO_x have included LNBs and multi-stage combustion. Recent permit applications in Florida have included SNCR in addition to LNBs and multi-staged combustion. The BACT determinations that include all three technologies include proposed NO_x limits as low as 1.95 lb/ton of clinker (30-day average). Recent BACT determinations that include LNBs and multi-staged combustion but no SNCR have NO_x limits ranging from 2.8 to 5.52 lb/ton of clinker (Florida DEP, 2005; Sumter Cement Company, 2005).

BACT determinations for SO_2 from preheater/precalciner kilns since 2000 have limits (and some pending proposed limits) that range from 0.20 to 2.16 lb/ton of clinker. Some of the limits are based on very low sulfur in the raw materials and SO_2 removed during the process; other limits are based on wet scrubbing and dry scrubbing (Florida DEP, 2005; Sumter Cement Company, 2005).

Recent BACT determinations for PM and PM_{10} are as low as 0.1 lb/ton of process feed. This limit is for the combined kiln and clinker cooler emissions and is about a quarter of the federal Maximum Achievable Control Technology (MACT) and NSPS limits discussed below (Florida DEP, 2005).

The Florida Department of Environmental Protection maintains a list of NSR and PSD cement plant construction permits for plants in Florida on their website. The have additional technical information on BACT for cement plants. The information can be found at www.dep.state.fl.us/Air/permitting/construction.htm.

Maximum Achievable Control Technology

MACT standards address emissions of Hazardous Air Pollutants (HAPs) from new and existing sources in identified industry categories. They apply in both attainment and nonattainment areas to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs.

Under the Clean Air Act, 40 CFR §§63.1340, *et seq.* (subpart LLL), PM emissions from portland cement manufacturing plants are regulated as a surrogate for HAPs.[4]

The MACT standards for portland cement plants appear in Table 9.10.

4. The MACT standards include limits for dioxins and furans and total hydrocarbon, which are not included in this discussion.

Table 9.10

MACT Standards for the Portland Cement Manufacturing Industry (40 CFR §§63.1340, *et seq.* (subpart LLL))

Source	PM Emissions Limit	Opacity Limit
All kilns and in-line kiln/raw mills[a]	0.15 kg/Mg[b] (0.30 lb/ton) of feed (dry basis)	20%
All clinker coolers	0.050 kg/Mg (0.10 lb/ton) of feed (dry basis)	10%
Raw mills and finish mills	NA	10%
All raw material dryers and material handling points	NA	10%

a. In-line kiln/raw mill refers to a part of the portland cement production process in which a dry kiln system is integrated with the raw mill so that all or a portion of the kiln exhaust gases are used to perform the drying operation of the raw mill without use of an auxiliary heat source. In this system, the kiln is capable of operating without the raw mill, but the raw mill cannot operate without the kiln gases; consequently the raw mill does not generate a separate exhaust gas stream.
b. Mg = megagrams, 1 Mg = 1 metric ton.

On October 12, 2005, EPA published final MACT standards (at 40 CFR §§63.1200, *et seq.* (subpart EEE)) for hazardous waste combustors, which include facilities that combust hazardous waste in cement kilns.[5] The standards, which may be the subject of ligitation, cover portland cement plants that use hazardous waste as a fuel source and regulate PM emissions as a surrogate for HAPs.[6] A summary of the standards is included in Table 9.11.

Clean Air Interstate Rule

In March 2005, EPA promulgated the Clean Air Interstate Rule (CAIR), which aims to address pollution transport that contributes to the non-attainment of eight-hour ozone and $PM_{2.5}$ air quality standards throughout the eastern U.S. CAIR identifies 28 states and the District of Columbia as contributing to unhealthy levels of ozone and/or fine particles in downwind nonattainment areas and requires these states to implement additional NO_x and SO_2 emissions reductions. CAIR requires states that contribute to downwind $PM_{2.5}$ nonattainment to achieve annual NO_x and SO_2 emissions reductions; states that contribute to downwind ozone nonattainment are required to achieve further seasonal NO_x emissions reductions. States can choose how to achieve these reductions, but it is expected that most, if not all states, will opt to cap emissions of electric generating units (EGUs) according to budgets established by EPA and to participate in EPA- administered emissions trading programs. None of the sources operating at portland cement plants can be included in any of the federal CAIR trading programs without disqualifying the state from participation in the federal trading regimes.

EPA promulgated the predecessor to CAIR in 1998.

5. A facility continues to be regulated as an affected source until it ceases burning hazardous waste and initiates closure requirements pursuant to the Resource Conservation and Recovery Act (RCRA).

6. The MACT standards establish limits for pollutants—including dioxins and furans, mercury, semivolatile metals, low volatile metals, total chlorine, carbon monoxide and hydrocarbons—that are not included in this discussion.

Table 9.11

MACT Standards for Cement Kilns Using Hazardous Waste as a Fuel Source (40 CFR §§63.1200, *et seq.* (subpart EEE))

Source	PM Emission Limit	Opacity Limit[a]
Existing Kilns	0.028 gr/dscf[b]	20%
New or reconstructed kilns	0.0023 gr/dscf	20%

a. The opacity standards do not apply to sources equipped with a bag leak detection system under 63.1206(c)(8) or a particulate matter detection system under 63.1206(c)(9).
b. gr/dscf, grains per dry standard cubic feet corrected to 7% oxygen. 1 grain = 0.002 ounces.

Table 9.12

Expected Emission Limits for Alternative Control Techniques in Federal Implementation Plan

Kiln Type	Emission Limit (lb/ton, 30-day average)
Long wet kiln	6.0
Long dry kiln	5.1
Preheater kiln	3.8
Precalciner or preheater/precalciner	2.8

Source: EPA, 1998

Known as the NO_x SIP Call, it requires 22 states in the eastern U.S. to implement NO_x reductions. Under the rule, affected states were required to amend their SIPs and limit NO_x emissions to meet EPA-specified ozone season NO_x emissions budgets. Although the NO_x SIP Call did not mandate the source categories that were required to reduce emissions, EPA proposed a Federal Implementation Plan (FIP) as part of the process that surrounded the implementation of the SIP call and the resolution of related litigation. EPA's calculation in the proposed FIP included NO_x emissions reductions of 30 percent from uncontrolled levels for large cement manufacturing sources. The rules outlined in the FIP apply to kilns with process rates of at least:

- 10 tons per hour for long wet kilns;
- 12 tons per hour for long dry kilns;
- 16 tons per hour for preheater kilns; and
- 22 tons per hour for precalciner and preheater/precalciner kilns.

EPA proposed a technology standard to implement required NO_x reductions. The technology standard would have required kiln owners to install and operate LNBs, mid-kiln firing or alternative control techniques (subject to EPA approval) that achieve at least the emissions limits shown in Table 9.12.

State and Local Policy Measures

As discussed in the Introduction to this report, states and localities are free under federal law to adopt more stringent standards than the Clean Air Act requires, although in some instances state laws limit this authority. SIP writers are presumably familiar with any such constraints under applicable local laws and regulations. The Introduction also outlines possible approaches for imposing more stringent state and local standards.

A review of BACT determinations indicates that there are numerous opportunities for reducing emissions from the cement manufacturing process. For NO_x control, recent BACT determinations in Florida require the installation of SNCR controls with LNBs and multi-staged combustion. These determinations include NO_x limits as low as 1.95 lb/ton of clinker (30-day average). Although there is only one SNCR device currently installed at a cement plant in the U.S., there are over 32 SNCR systems installed on kilns in Germany and many more in the rest of Europe. Recent BACT determinations that do not include SNCR but do include LNBs and multi-staged combustion have NO_x limits ranging from 2.8 to 5.52 lb/ton of clinker.

Recent BACT determinations have set SO_2 limits ranging from 0.20 to 2.16 lb/ton of clinker for preheater/precalciner kilns, compared to uncontrolled emissions from preheater/precalciner kilns that can be as high as 7.6 lb/ton of clinker (and up to 10 lb/ton of clinker for existing long dry kilns). Where the process itself does not achieve satisfactory SO_2 emissions levels, wet FGD technology can provide an SO_2 control efficiency of 90–99 percent. Of more than 100 cement plants in the country, only five currently use wet scrubbers to control SO_2, suggesting substantial opportunities for the industry to improve its emissions profile. Dry FGD technology (not recommended for wet kilns) and lime spray injection are other SO_2 control options, although they are less effective.

Apart from imposing stringent permit limits on individual plants, state and local areas should consider the following opportunities for the adoption of regulations to address PM emissions from the cement manufacturing sector.

The first option is for states and localities to adopt more stringent PM standards. The MACT PM limits for cement kilns that burn hazardous waste (Table 9.11) are more stringent than the MACT PM standards for cement kilns that burn fossil fuels (Table 9.10). State and local regulators can consider requiring kilns that do not burn hazardous waste to meet the more stringent standards, absent a showing that particular plants cannot achieve these levels.

A second option is for states and localities to adopt rules promulgated by the South Coast Air Quality Management District (AQMD) to control fugitive dust. The South Coast AQMD Board recently adopted a new rule to reduce PM emissions from cement processing equipment, storage piles and facility roadways (South Coast AQMD, 2005). According to the South Coast AQMD staff report, the rule is designed to achieve PM reductions of 2 tons per day in the South Coast basin and has a cost-effectiveness of $1,000–$4,000 per ton of PM reduced (Pham, 2005).

The proposed rule includes opacity limits of 10 percent from any operation at the facility except for open piles, roadways and unpaved areas, where a higher opacity limit of 20 percent applies. Additionally, the operator of a facility cannot cause or allow the facility to emit any dust plume that is visible more than 100 feet away in any direction. The rule includes additional requirement for enclosing conveying systems and material loading, unloading and storage areas. Operators must use dust suppressants as necessary. They must also enclose crushing, screening, milling, grinding, blending, drying, heating, mixing, sacking, palletizing, packaging and other related operations. The enclosed areas must be vented to a control system. Additionally, all kilns and clinker coolers must be vented to control systems (South Coast AQMD, 2005).

Finally, under the South Coast AQMD rule, all control

systems must have outlet concentrations that do not exceed 0.01 grains of PM per dry standard cubic foot for equipment installed before November 4, 2005, and concentrations that do not exceed 0.005 grains of PM per dry standard cubic foot for equipment installed after November 4, 2005. The rule includes the performance requirements for existing baghouses as specified in the U.S. Industrial Ventilation Handbook (South Coast AQMD, 2005).

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Chapter 10

Iron and Steel

Introduction

The U.S. steel industry, which produced about 106 million tons of raw steel in 2004, is comprised of two primary segments: 17 integrated iron and steel plants and approximately 93 "minimills" that melt steel scrap in electric arc furnaces (EAFs). The integrated plants currently account for about 47 percent of U.S. steel production, while the minimills account for the remaining 53 percent (USGS, 2005). Integrated plants use iron ore in blast furnaces to produce molten iron; the molten iron and ferrous scrap are then loaded (the terminology in the industry is "charged") into basic oxygen furnaces (BOF), where oxygen is blown into the vessel to remove carbon and produce steel. The EAFs at minimills produce steel by melting mostly recycled ferrous scrap. Steel is currently used principally in the automotive and construction industries.

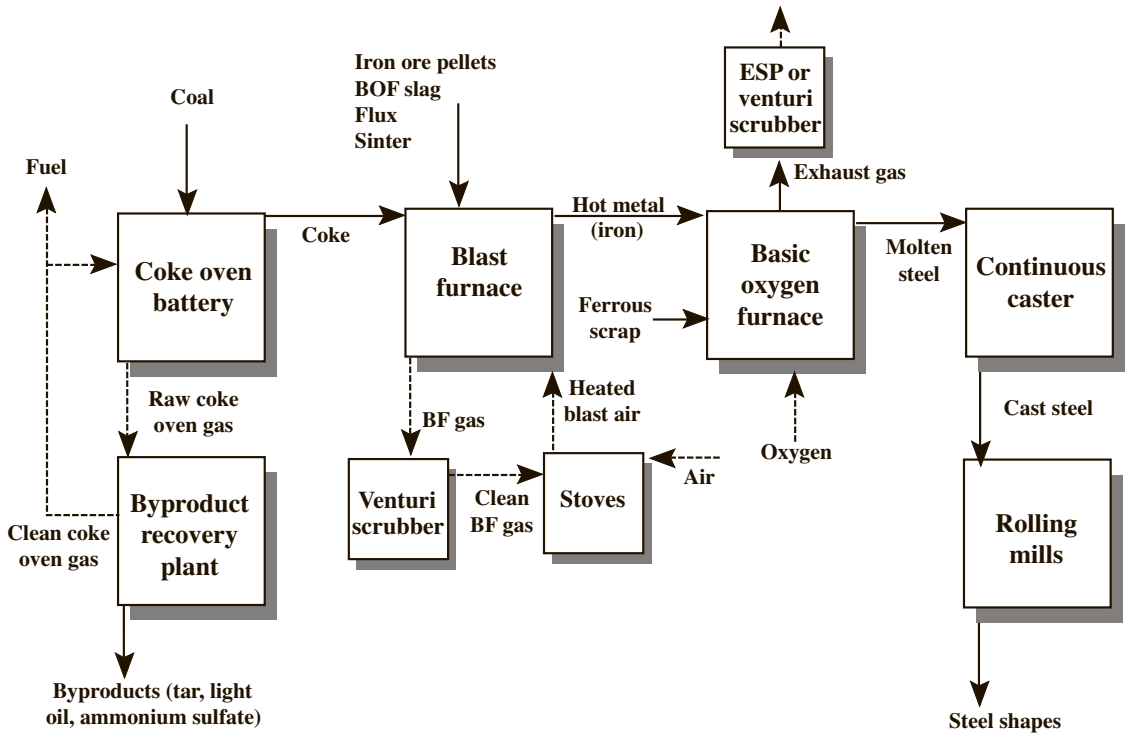
The iron and steel industry grew rapidly with the advent of the railroad in the latter half of the nineteenth century and remained a driving force in the American economy through the first half of the twentieth century, when the U.S. was the world's largest steel producer. Since the end of World War II, however, the U.S. has lost a large share of the world market for steel—particularly since 1969, when U.S. steel production peaked at 141 million tons. In the ensuing years, new, more efficient steel plants have been built overseas, aided by reductions in shipping costs (Gordon, 1991). In addition, other materials, such as plastic, have substituted for steel in various applications. Meanwhile—partly due to the energy crises of the 1970s and the declining cost of scrap metal—smaller minimills have claimed a growing share of steel manufacturing capacity by converting scrap metal to steel, rather than engaging in the traditional steel production process performed by integrated steel mills.

Most integrated steel mills in the U.S. are located in the Great Lakes region near the automotive industry. While minimills operate throughout the country, the majority are located in the traditional steel making states of Pennsylvania, Ohio and Indiana.

The production of steel at integrated steel mills involves numerous processes, all of which have their own extensive set of operations (see Figure 10.1). Each of these many processes is a potential source of emissions; as a result, controlling emissions from iron and steel plants is a complicated matter.

This chapter focuses only on coke production and primary iron and steel production, not on the host of related industries (e.g., ferroalloy plants, ferrous foundries) that either produce inputs for steel plants or cast the finished steel into more refined products. Although these facilities also produce fine particulate matter ($PM_{2.5}$), sulfur dioxide (SO_2) and nitrogen oxide (NO_x) emissions (albeit in lesser quantities than primary steel production), they are beyond the scope of this report. Moreover, even for

Fig. 10.1 Processes at an Integrated Steel Mill



Source: EPA, 2005k

coke and primary iron and steel production, the approach taken in this chapter is somewhat different than that taken for other industry sectors discussed in this report. One of the reasons for this is the very large number of emissions points in the coke, iron and steel production process. Additionally, perhaps partly because many of these emissions are fugitive in nature, information on the emissions reductions that can be achieved through various approaches and on the cost-effectiveness of controls is scant (EPA, 2002b). Although we are unable to present complete information on all emissions control options and their cost-effectiveness, we attempt to provide a robust presentation on the range of available control opportunities for the most significant sources of emissions associated with the iron and steel industry's primary production processes.

Sector Profile

Since the 1970s, American steel production has ranged between 80 and 110 million tons per year, with the U.S. ceding the spot as the world's top steel producer to China during that time (DOE, 2001). The past few years have seen a continuation of the downward trend in U.S. production, as large quantities of relatively low-cost steel from overseas have further eroded the American steel base. Since 1998, over 30 steel companies have declared bankruptcy (EPA, 2004a; USGS, 2005). Nonetheless, the American iron and steel industry is still a formidable economic force: in 2004, 95 facilities produced $51 billion of product (EPA, 2004a).

Steel is manufactured in two different types of facilities: integrated mills and minimills. Each accounts for about half of total steel production and each has a very different emissions profile. The two types of facilities use different input materials, but the output from both is molten steel, which is then formed into a variety of products. Integrated mills are typically used for high tonnage production of carbon steels, while minimills produce carbon steels and specialty steels (EPA, 1995).

An integrated steel mill includes a blast furnace for making iron and a BOF for making steel. A few integrated mills also have coke plants and sinter plants; the latter convert fine-sized raw materials into an agglomerated product (sinter) of suitable size for charging into a blast furnace. All have continuous casters for casting the molten steel and almost all have rolling mills onsite or nearby to make the final steel shapes. The basic raw material inputs include coke, iron ore pellets (taconite), sinter and flux materials for removing impurities from molten metal.

Coke is produced by heating coal in the absence of oxygen at high temperatures in coke ovens. Next comes the production of molten iron, which involves heating the

coke, iron ore and limestone in a blast furnace. Then, molten iron from the blast furnace is combined with flux and scrap steel in the BOF and high-purity oxygen is injected to remove carbon and produce steel. Forming and finishing operations follow (EPA, 1995).

There are 17 integrated steel mills in the U.S., with a total of 34 blast furnaces, 44 BOFs and five sinter plants. Most of these integrated mills are concentrated in the Great Lakes region, particularly Indiana, Michigan and Ohio (EPA, 2001a, updated November 2005).

Minimills produce steel from metal scrap using EAF technology. Minimills use primarily scrap steel as the input material; the scrap steel is melted and refined by passing an electric current through it. Minimills do not involve the coke making and iron making steps that take place at integrated mills due to the nature of their raw materials (EPA, 1995). There are 93 minimills operating throughout the country.

Emissions

Table 10.1 summarizes emissions from the iron and steel industry. EPA provides highly detailed data on emissions from this sector, but these data do not reveal how emissions break down by major categories, i.e., coke making, iron making, steel making in BOFs, and steel making in EAFs.

Table 10.1

Iron and Steel Sector Emissions (1999)

Process Type	$PM_{2.5}$ (tons)	SO_2 (tons)	NO_x (tons)
Iron and steel primary production	44,922	89,305	97,977

Source: EPA, 2005a

Integrated Steel Mills

As noted above, integrated steel mills use a three-step process to produce steel:

- coke making, from coal in a coke oven;
- iron making, using coke in a blast furnace to produce molten iron; and
- steel production, from molten iron and ferrous scrap in a BOF.

Coke making

Most coke is produced in by-product recovery ovens by heating pulverized bituminous coal in an oxygen-free atmosphere until all volatile components evaporate. It is used as a fuel and as a reducing agent to melt and reduce iron ore.

Coal, which is first pulverized to prepare for coking, is fed into the coke oven (in a process known as "charging") through ports in the top of the oven. The ports are then sealed, and the coal is heated at high temperatures for 14 to 36 hours. Coke manufacturing is done in batches, with each coke battery comprising a series of ten to 100 individual ovens, side by side. The walls separating adjacent ovens, as well as the end walls, contain a series of vertical heating flues. Volatile compounds are driven from the coal and the solid carbon remaining in the oven is the coke (EPA, 1995, 2000a).

The volatile matter driven from the coal moves upward through cast iron "goosenecks" into a common horizontal pipe called the collecting main, which connects all the coke ovens in series. The collecting main directs the gases evolved during coking to an adjacent byproduct recovery plant, where combustible gases and other coal byproducts are recovered. The cleaned gas is used to underfire the coke ovens and for fuel elsewhere in the plant. Coke oven gas is the most common fuel for underfiring coke ovens (EPA, 2000a).

When the coking is complete, doors on both sides of the oven are removed and the coke is pushed out into a rail "quench" car. The quench car moves to a quench tower, where the hot coke is sprayed with water to cool it and prevent it from continuing to burn. The car then discharges the coke onto a wharf to drain and cool further, after which grates on the wharf are opened and the coke falls onto a conveyor that carries it to the crushing and screening station. After it is broken into appropriate sizes, the coke goes to the blast furnace or to storage (EPA, 1995, 2000a).

PM is emitted at a number of different points in the coke making process, including during oven charging, from leaking doors, when pushing coke into the quench car, and during coke quenching (EPA, 2000a). EPA's initial Maximum Achievable Control Technology (MACT) rule for coke ovens targeted leaks from doors and lids, as well as offtake system emissions. For purposes of EPA's more recent MACT rule dealing with integrated iron and steel plants, the emissions sources of concern for coke making are the pushing of coke from the ovens, the quenching of incandescent coke, and the battery stacks (the discharge point for the underfiring system). Coke pushing results in fugitive particulate emissions that may include volatile organic compounds (VOCs), while quenching results in PM emissions with traces of organic compounds (EPA, 2002b).

Coke can also be made at a non-recovery coke oven. In this type of oven, the gas produced by the coking process

Table 10.2

Filterable[a] Particulate Matter Emissions Factors from Coke Ovens (lb per ton of charged coal)

Source	$PM_{2.5}$	Total PM
Coke pushing		
Uncontrolled	0.19	1.15
Controlled with venturi scrubber	0.13	0.18
Combustion Stack	0.44	0.47
Quenching		
Uncontrolled (dirty water)	1.01	5.24
Uncontrolled (clean water)	0.13	1.13
With baffles (dirty water)	0.27	1.30
With baffles (clean water)	0.03	0.54
Oven charging	0.006	0.016
Mobile scrubber car	0.022	0.072

a. Emissions factors are for filterable PM only—not condensable PM.

Source: EPA, 2000a

is immediately combusted in and around the coke oven to provide heat for the coking process, thus eliminating the need for a by-product plant and producing far fewer emissions than a by-product coke oven.

Table 10.2 includes PM emissions factors from the predominant sources of coke oven emissions.

Iron making

The blast furnace, a tall steel vessel lined with heat-resistant brick, converts iron ore into a more pure and uniform iron. Conveyor systems of carts and ladles carry inputs and outputs to and from the blast furnace (EPA, 2002b).

Iron ore is the primary source of iron for integrated iron and steel mills, although iron can also be captured by "sintering"—a process of agglomeration whereby fine grains, pollution control dust, sludge, and other materials are mixed, ignited, and fused into cakes for charging to the blast furnace. Together with the iron materials, coke (for fuel) and limestone (the flux) are charged into the top of the furnace; meanwhile, hot air is forced in from the bottom. These react to form molten reduced iron, carbon monoxide and slag. Some of the carbon also remains in the iron, which allows iron and steel to harden when they are cooled rapidly. Molten iron and slag collect in the hearth at the base of the furnace (EPA, 1995, 2002).

A hole, called a taphole, is drilled at the base of the hearth and the molten iron and slag are periodically removed, or cast, from the furnace. Molten iron flows into runners that lead to transport ladles, while slag flows through separate runners to a slag pit. The blast furnace byproduct gas is cleaned of particulates and used as fuel within the steel plant (EPA, 1986, 2001a).

The primary source of blast furnace emissions is the casting operation, which generates PM emissions when the molten iron and slag come into contact with the air. Drilling and plugging the taphole also contribute to casting emissions (EPA, 1986).

Steel making in a BOF

BOFs are the standard steel-making furnaces used at integrated mills for the final refinement of iron into steel, although two facilities (as of 2002) use EAFs, which we discuss in the section on minimills (EPA, 2002b). Molten iron from the blast furnace, flux, alloy materials and scrap are placed in the BOF, where they are melted and refined by the injection of high purity oxygen. Impurities are removed by the combination of the fluxes and oxygen, which results in the production of slag. In addition to slag, waste products from this process include carbon monoxide and oxides of iron emitted as dust. The furnace is tilted in order to remove both steel and slag. In BOFs, unlike in EAFs, about 67 percent of the iron is molten iron and only about 27 percent is scrap (EPA, 2001c, 2002).

The operations in the BOF process are:

- charging—the addition of molten iron and metal scrap to the furnace;
- oxygen blow—introducing oxygen into the furnace to refine the iron;
- turndown—tilting the vessel to obtain a sample and check temperature;
- reblow—introducing additional oxygen if necessary;
- tapping—pouring the molten steel into a ladle; and
- deslagging—pouring residual slag into a slag pot.

Of these, the oxygen blow portion of the furnace cycle accounts for the largest share of emissions, followed by tapping and then charging (EPA, 2001b, 2001c).

Sinter Plants

There are currently five sinter plants in the U.S., with a combined capacity of 14.6 million tons per year. Like many components of the steel industry, the population of sinter plants has been declining: as recently as 2001, the

industry counted seven operational plants (Branscome, 2005a).

The sintering process converts fine-sized raw materials, including iron ore, coke breeze, limestone, mill scale, and flue dust, into an agglomerated product (sinter) of suitable size for charging into the blast furnace. The raw materials are mixed with water to form a cohesive matrix and then placed on a continuous traveling grate called the sinter strand. A burner hood at the beginning of the sinter strand ignites the coke in the mixture, after which the combustion is self-supporting and provides sufficient heat (2,400–2,700 degrees Fahrenheit) to cause surface melting and agglomeration of the mix. On the underside of the sinter strand is a series of windboxes that draw combusted air down through the material bed into a common duct leading to a gas cleaning device (either a venturi scrubber or a baghouse) (EPA, 1986).

The fused sinter is discharged at the end of the sinter strand, where it is crushed and screened. Undersize sinter is recycled to the mixing mill and back to the strand. The remaining sinter product is cooled in open air or in a circular cooler with mechanical fans. The cooled sinter is crushed and screened a final time, and then the fine material is recycled and the product is sent to be charged to the blast furnace. Generally, 2.5 tons of raw materials, including water and fuel, are required to produce one ton of product sinter (EPA, 1986).

Sinter plants generate emissions (in order from largest to smallest in quantity) from the windbox exhaust, the discharge end (and its associated sinter crushers and hot screens), raw material handling, and the cooler. PM emissions from the windbox come mainly in the form of iron oxides, sulfur oxides, carbonaceous compounds and chlorides (EPA, 1986).

Several factors can affect PM emissions. For example, PM emissions from the windbox are (not surprisingly) affected by the amount of fine material being used (e.g., pollution control dust from the steelmaking process) and its particle size distribution; an increase in fine material can result in more PM emissions as well as smaller particle sizes in the emissions. Operating parameters, such as the bed air flow rate, bed depth, proper proportioning and mixing of the feed materials, and condition of the grate and machine seals also affect particulate generation in the windboxes (EPA, 2001a).

Minimills

Minimills do not involve the coke-making and iron-making steps in the steel making process due to the nature of their feed stock; instead they use an EAF to melt and refine scrap metal as the primary raw material. Table 10.3 shows emissions factors for minimills.

Most of the iron introduced into an EAF—about 88 percent—is scrap. In most EAFs, electric arcs are formed between three carbon electrodes. (A few newer furnaces use direct current and only one electrode.) The EAFs require a power source to supply the charge necessary to generate the electric arc and typically use electricity purchased from an outside source. Flux is blown or deposited on top of the metal after it has melted; air in the furnace and injected oxygen oxidize the impurities. Slag containing some of the oxidation products forms on top of the molten metal. When the process is complete, the furnace is tilted so that the molten steel can be drained through one tap and the slag removed from a separate tap. In addition to slag, the EAF process produces metal dusts and gaseous products (EPA, 1995, 2002).

Emissions Control Opportunities

As indicated, the steel-making process involves multiple steps, each of which comprises many emissions points. This section attempts to identify the most significant sources of emissions at each step of the process, together with control opportunities.

Certain work practices can also be instituted to reduce emissions. For example, fugitive emissions from storage piles, raw material transfer, roads, etc., can be controlled by enclosures, suppression techniques, and road sweeping. Every plant should implement a control plan for fugitive dust. Raw material transfer points that are particularly dusty can be equipped with hoods vented to a baghouse. In addition, it is important to monitor capture and control equipment to ensure that it is working properly at all times.

Coke making

There are a number of different emissions points and steps in the coke-making process, including oven charging, leaking doors, coke pushing (into the quench car) and coke quenching (EPA, 2000a). Burning of uncleaned coke oven gas is also a significant source of emissions. The basic types of control opportunities for these sources of emissions include improved work practices, pollution control technology, limited use or even elimination of coke in the production of steel, and relatively new processes for the production of coke. The best way to reduce emissions is to reduce the amount of coke produced and consumed.

Improved Work Practices and Pollution Control Technology. During the coking cycle, emissions can occur through poorly sealed doors, charge lids, offtake caps, the collecting main and cracks in oven brickwork. These emissions can be controlled to some degree by a diligent maintenance program, including door cleaning and

Table 10.3

Emissions Factors for Minimills (units in lb/ton of steel produced unless otherwise specified)

Source	Controls	NO_x	SO_2	Total PM
Electric arc furnace: charging, melting, slagging, tapping (with direct shell evacuation and roof canopy hood exhausted to baghouse)	Without oxygen lancing and oxy-fuel burners	0.33	0.17	0.064
	With oxygen lancing and oxy-fuel burners	0.31	0.10	
Ladle metallurgy	Controlled by baghouse	0.011	0.035	0.0054
Argon oxygen decarburization vessel	Controlled by baghouse	0.1		
Electric arc furnace and argon decarburization vessel	Fugitive emissions, controlled by roof monitor exhausted to baghouse	0.002		
Electric arc furnace and ladle metallurgy: charging, melting, slagging, tapping, etc. (with direct shell evacuation and roof canopy hood)	Without oxygen lancing and oxy-fuel burners	0.39	0.54[a]	1.1[a]
	With oxygen lancing	0.48	0.22[a]	
	With oxy-fuel burners	0.41	0.096[a]	
	With oxygen lancing and oxy-fuel burners	0.40	0.58[a]	
Electric arc furnace, ladle metallurgy, and continuous caster: charging, melting, slagging, tapping, etc. (with direct shell evacuation and roof canopy hood exhausted to baghouse)	With argon oxygen decarburization vessel	0.38	0.046	
	With oxy-fuel burners and oxygen lancing	0.13	0.14	
Direct reduced iron reformer	Controlled by cyclone and scrubber	0.96	0.048	

a. Exhausted to baghouse.

b. Pounds per million British thermal units (lb/MMBtu) of heat input.

Source: EPA, 2004c

rebuilding, and by the application of lute (seal) material. In addition, operating the coke oven so as to avoid large temperature fluctuations can reduce damage to the battery, which causes emissions.

Almost all plants have capture and control systems for coke pushing emissions. Additional pollution control can be obtained by limiting the frequency of "green pushes," which occur from incomplete coking and overwhelm the PM capture system. Regularly observing opacity during the pushing process and investigating ovens that produce a green push will enhance emissions control (EPA, 2001b).

Coke oven emissions also occur from battery stacks when raw coke oven gas leaks through the oven walls, enters the flues of the underfiring system and is discharged through the stack. These emissions can be limited with appropriate diagnostic procedures and work practices. Good, systematic operation and maintenance of the battery is necessary to prevent green pushes and associated stack emissions (EPA, 2002a). Another way to reduce emissions during quenching is to substitute an inert gas, such as nitrogen, for water when cooling the coke. When water is sprayed on the coke, it breaks up the coke and generates PM emissions. Substituting an inert gas avoids this problem (Davis, 2000). Some plants in Europe have switched from water quenching to dry quenching to limit emissions of PM and VOCs. This does require major construction activities and associated costs (EPA, 2002a). The European Commission (EC) estimates that a dry quenching plant may cost between 10 and 15 times more than a wet quenching station (EC, 2001).

Emissions of PM from coke oven charging can be controlled to a large degree by staged charging. Staged charging involves dropping coal into an oven at a controlled rate; this allows the induced vacuum on the oven to handle all the emissions and not be overwhelmed. Charging emissions can also be controlled to some degree by preventing obstructions through regular cleaning of

the goosenecks and collecting main passages (Davis, 2000; EPA, 2000a). Leaks from coke oven doors can be controlled by cleaning, patching and regular maintenance (Davis, 2000; EPA, 2000a). Emissions from coke pushing are generally controlled using mobile scrubber cars with hoods, a closed shed with vents to evacuate the gas to a cleaning device, or traveling hoods with a duct leading to a gas cleaner (EPA, 2000a). It is especially important to desulfurize coke oven gas before it is burned to reduce SO_2 emissions. Only 11 of the 16 operating byproduct recovery coke plants currently desulfurize this coke oven gas (Burns, 1998, updated).

Emissions from quenching the red-hot coke with water after pushing are controlled by a combination of techniques: all quench towers should have baffles that are periodically inspected and cleaned and total dissolved solids in the water used for quenching should be monitored and limited. Dirty river water should not be used (EPA, 2001b).

Limited Use of Coke and Use of Cokeless Technologies. One approach to limiting the emissions associated with coke making is to substitute pulverized coal for as much as 40 percent of the coke in the blast furnace. Other fossil fuels such as natural gas, oil, or tar/pitch may also substitute for coke to varying and limited degrees, but pulverized coal is the most commonly used. Pulverized coal is also cheaper than coke; however, the coal may degrade the final steel product, so operations producing high quality steel generally avoid substitution (North Carolina DENR, 1998). One company in Japan has even been substituting waste plastic for coke for almost a decade (Iino, 2003).

A number of other technologies exist to produce molten iron and steel without the use of coke. When coke batteries need replacement, they are often replaced by a cokeless Direct Reduced Iron (DRI) process. The MIDREX® and HYL processes are the most common form of DRI and account for approximately 85 percent of worldwide production at DRI plants, which have a combined global capacity of 50 million metric tons (55 million short tons) (Environment Canada, 2002). The DRI can substitute for scrap steel in an EAF and, to a lesser extent, for scrap in a BOF. Using the DRI produces lower emissions while at the same time allowing the EAF to produce a higher quality steel with fewer byproducts. The capital cost of a MIDREX® facility with a capacity of 1.5 million short tons per year is approximately $300 million (Environment Canada, 2002).

Another way to avoid using coke is to engage in direct smelting. One direct smelting method is the Corex process, which uses untreated raw coal in place of coke. The first Corex demonstration plant was developed in 1988. While this 300,000 ton-per-year plant has had a number of technical difficulties, Corex plants built more recently (including ones in South Korea and India) have reportedly been much more successful. In June 2005, China announced plans to construct the world's largest Corex plant—designed to produce 1.5 million tons per year of molten iron (Kuck, 1998; Siemens, 2005). The developer of the technology, Deutsche Voest-Alpine Industrieanlagenbau, claims that compared to traditional steel making, the Corex process reduces PM emissions by over 99 percent, SO_2 emissions by 92 percent, and NO_x emissions by 98 percent (STAPPA and ALAPCO, 1999).

The HIsmelt (high intensity smelt) process, developed by an Australian company, is another type of cokeless iron melting process that converts fine ores into molten iron. The process has been tested since 1991 in Australia, the United Kingdom and the U.S. The first commercial size plant began construction in Germany in 2003 and is expected to begin producing 800,000 metric tons (970,000 short tons) of pig iron per year by mid-2006. Other iron-making facilities may follow (Aker Kvaerner, 2005).

New Coke Production Processes. Employing a non-recovery coke battery is another method for reducing coke emissions. Unlike traditional coke plants, non-recovery coke batteries consume the byproducts of coke production that are typically recovered and burn them directly to produce heat. Because the ovens are under negative pressure, there are no leaks of raw coke oven gas. The process was developed by the Jewell Coal and Coke company, which has operated a plant in Virginia since 1989. In 1998, the Indiana Harbour Coke Company, a subsidiary of Sunoco, built a 1.3 million-ton-capacity non-recovery coke plant in Chicago, Illinois. It uses the flue gas to produce steam and electricity in a cogeneration plant. Capital costs for the plant were $300–$350 million (Boselovic, 1998; Environment Canada, 2002). In addition, EPA estimates the annual costs for non-recovery batteries at approximately $28,300 per battery, due to monitoring, recordkeeping and reporting requirements needed to assure continuous compliance with standards for emissions of hazardous air pollutants (HAPs) (EPA, 2004b).

Other technologies are under development. For example, over the past few years, Japan has been testing a "next-generation" coke oven as part of the SCOPE 21 project (Super Coke Oven for Productivity and Environment Enhancement in the Twenty-First Century). The project reduces coking time (and emissions) by preheating the coal; it also reduces gas leakage by covering the coke oven entirely with steel plates to create an airtight cover. In addition, SCOPE 21 is attempting airtight transportation of hot coke (Itagaki, 1997; Fukada, 2005).

With funding from DOE, the Calderon Energy Company of Ohio has been testing a closed system for coke making

while also attempting to identify ways to clean the raw gases before they are released. The product gases are broken down (cracked) and desulfurized (using limestone) to produce hydrogen, carbon dioxide and elemental sulfur. According to DOE, if the process is successful it will produce no gas emissions, water discharges or solid wastes (DOE, 2005). A similar project, conducted between 1995 and 1997, cost $7.4 million, 41 percent of which was paid by DOE and the rest by Calderon. This pilot project produced a yield of approximately 200 tons of coke per day. A commercial-scale facility would produce approximately 1,600 tons of coke per day (Ailor, 2003).

Summary of World Bank and European Commission Recommendations. The World Bank and the EC have published extensive reports within the past few years outlining opportunities for pollution control from coke making and other industry sectors. For air emissions from coke plants, the EC report considers the prevention of fugitive smoke and the treatment of coke oven gas to be the most important issues. The Commission identifies Best Available Techniques (BAT), which—as explained in its report—are not emissions limits but rather levels that new installations could meet or exceed and that many existing installations could also meet or exceed over time (World Bank, 1998b; EC, 2001).

The measures recommended in the World Bank and EC approach with respect to coke making (some of which overlap with the approaches outlined above) are briefly summarized here:

General

- Use cokeless iron- and steel-making processes to eliminate the need to manufacture coke.
- Use and preheat high-grade coal to reduce coking time, increase throughput, and reduce fuel consumption. Beneficiation (crushing and separating ore into more desirable substances) and blending processes can be used to improve the quality of coal feed.
- Maintain a free flow of gas without obstruction in the coke ovens.
- Use enclosed conveyors and sieves for coal and coke handling to reduce dust formation.
- Use sprinklers to suppress dust formation.
- Store materials in bunkers or warehouses and provide windbreaks where feasible.
- Employ a systematic program carried out by specially trained maintenance personnel to extensively maintain and clean various types of equipment, including oven chambers, oven doors and frame seals, ascension pipes (the exhaust pipes used during coking), charging holes, lids, and other equipment.

Charging

- Remove dust particles by evacuation through various pipe systems, followed by fabric filtration. PM emissions rates less than 5 grams (0.011 lb) per metric ton of coke are achievable.

Coking

- Use large ovens to increase batch size and reduce the number of chargings and pushings, thereby reducing associated emissions.
- Operate coke ovens in a manner that avoids large temperature fluctuations (which cause significant emissions).
- Install spring-loaded, flexible-sealing doors or knife-edged doors.
- Regularly clean doors, door frames, and hole lids.
- Seal lids using a slurry.
- Use low leakage door construction.

Firing

- Use desulfurized coke oven gas.
- Incorporate low-NO_x techniques, such as staged combustion, in the construction of new batteries.

Pushing

- Maintain a sufficient coking time, thus avoiding green pushes.
- Use sheds and enclosed cars or traveling hoods. The gases released should be removed and passed through fabric filters. These techniques can achieve a stack emissions rate of less than 5 grams (0.011 lb) of PM per metric ton of coke.

Quenching

- Use dry instead of wet quenching where feasible and filter all gases extracted from the dry quenching unit.
- If wet quenching is used, provide interceptors (baffles) to remove coarse dust.

Coke Oven Gas Desulfurization

- Achieve desulfurization by using absorption systems or oxidative desulfurization, provided that polluting water with toxic compounds can be avoided.

Table 10.4

Installed Capital Cost for a Capture and Control System for Casthouse Fugitive Emissions (2001$)

Item	Cost
Baghouse, capture hoods, ductwork	$1,920,000
Auxiliary equipment (fans, dampers, stacks, etc.)	$577,000
Other (electrical, piping)	$840,000
Site-specific retrofit costs (50 percent)	$1,670,000
Total installed capital cost	$5,007,000

Source: EPA, 2002

Gas-tight Operation of Gas Treatment Plant

- Minimize the number of flanges by welding piping connections wherever possible.
- Use gas-tight pumps (e.g., magnetic pumps).
- Avoid emissions from pressure valves in storage tanks by connecting the valve outlet to the coke oven gas collecting system.

Iron Making

Fugitive PM from the casting operation in the blast furnace is the main source of blast furnace emissions. Casting is the process whereby molten iron and slag are periodically removed (cast) from the furnace through the taphole. PM emissions are generated when the molten iron and slag contact air above their surface. Resulting PM emissions usually comprise iron oxides, magnesium oxide and carbonaceous compounds. Drilling and plugging the taphole also generate casting emissions (EPA, 1986, 2001a).

About half of U.S. blast furnaces control casthouse emissions with covered runners and by evacuating emissions through capture hoods ducted to a baghouse. This suggests that a significant opportunity exists to reduce emissions from the roughly half of U.S. blast furnaces that have not yet employed these types of control measures (EPA, 2001a). Some furnaces also rely on suppression techniques, such as using steam or inert gas to reduce the formation of pollutants by preventing ambient air from contacting molten surfaces (EPA, 1986, 2001a).

Flue gas from the blast furnace is used to preheat the blast air and as fuel for other plant operations. Before blast furnace flue gas can be burned efficiently, PM must be removed. The gases pass first through a settling chamber or dry cyclone to remove about 60 percent of the PM. They then undergo a one or two stage cleaning operation. The primary cleaner is normally a wet scrubber, which removes about 90 percent of remaining PM. The secondary cleaner is a high-energy wet scrubber (usually a venturi), which removes up to 90 percent of the PM that eludes the primary cleaner. Together, these control devices provide a clean fuel with less than 0.02 grains of PM per dry standard cubic foot (gr/dscf) (EPA, 2001c).

Emissions from the blast furnace top are usually controlled effectively with wet-collection systems such as dust catchers, venturi scrubbers and precipitators (Davis, 2000). Other control techniques include changing operational practices, such as increasing the amount of oxygen injected in the blast air and raising the blast air temperature (Environment Canada, 2002).

EPA's "Economic Analysis of the Final Integrated Iron and Steel NESHAP" provides cost estimates for new capture and control systems for casthouses, based on responses to an EPA cost survey of plants that had most recently installed such systems and on experience with a system installed at USS/Kobe Steel (later owned by Republic Technologies International). According to EPA, the capture and control system was designed for 300,000 actual cubic feet per minute (acfm) and an air-to-cloth ratio of 6.3 feet per minute. The total capital cost (increased by 50 percent to reflect the higher cost of retrofitting) is shown in Table 10.4 as approximately $5 million. The annual operating cost was estimated at $730,000 per year, resulting in a total annualized cost of $1.2 million per year.

Summary of World Bank and European Commission Recommendations. As indicated, the World Bank and the EC have published reports outlining opportunities for pollution control from multiple industry sectors. They include BAT, for blast furnace emissions, a summary of which we include below (World Bank, 1998a; EC, 2001). Again, some of the techniques listed may overlap with approaches already discussed.

- Use the direct steel manufacturing process where technically and economically feasible.
- Replace a portion of the coke used in the blast furnace by injecting pulverized coal or by using natural gas or oil. The feasibility of pulverized coal injection at 180 kilograms (397 lb) per metric ton of molten iron has already been demonstrated, but higher injection rates could be possible.
- Recover blast furnace gas to use as an energy source. While this gas contains only 10 percent of the energy content of natural gas, the fact that vast quantities of it are generated means the energy recovery potential is very high.
- Treat blast furnace gas to remove PM and other pollutants. Dry separation techniques (e.g., reusable deflectors) can remove coarse PM, while a scrubber or wet electrostatic precipitator (ESP) can then remove $PM_{2.5}$.

- Use pelletized instead of sintered feed where appropriate.
- During tapping, cover iron runners and use nitrogen blankets to reduce dust emissions.
- Reduce dust by using enclosed conveyor belts and wind barriers.
- Recycle collected dust to a sintering plant where possible.
- Improve blast furnace charge distribution.
- In high top pressure blast furnaces, use an expansion turbine to recover energy from the large volumes of pressurized gas.
- Reduce energy use in the hot stoves by using a computer to aid hot stove operation, by preheating the fuel, and by installing insulation.
- Use low-NO_x burners to reduce NO_x emissions from burning fuel in ancillary operations.
- Use tar-free runner linings.

Steel Making in a BOF

As indicated above, the most significant sources of emissions in the BOF process are charging, tapping, and the oxygen blow, which is the largest emissions source.

Primary[1] Furnace Controls. Emissions during oxygen blow periods are controlled by the primary emissions capture and control system for the BOF, which is either: (1) an open hood directed to an ESP or wet scrubber, or (2) a closed hood ducted to a wet scrubber. There are about twice as many open as closed hood BOF shops in the U.S. EPA has reported that fabric filters have recently been proven effective, but they are not currently used at any facility in the U.S. (EPA, 2001a). The use of fabric filters could provide significantly better PM control.

EPA also reported in 2001 that all BOFs had capture and control systems for primary emissions. Eight of the open hood systems had venturi scrubbers; another eight were controlled by ESPs. All eight of the closed hood shops were controlled by venturi scrubbers. EPA reported that ESPs at the open hood shops performed better than the venturi scrubbers (the MACT standard for open hood systems is therefore based on this technology). For closed hood systems, the MACT standard is based on high-energy venturi scrubbers with a pressure drop of 50 inches of water or more (EPA, 2001c).

Opportunities for improved control of primary emissions will vary from site to site; they typically include the potential for significantly reduced emissions from upgrading old scrubbers to scrubbers with a higher pressure drop and upgrading ESPs.

Secondary Furnace Controls. According to EPA, data on secondary BOF emissions are extremely limited (EPA, 2001c). The BOF produces secondary or fugitive emissions—primarily metal oxides—when the molten iron and scrap metal are charged to the furnace and when the molten steel and slag are tapped from the furnace. Additional sources of secondary emissions include hot metal transfer, deslagging, and ladle metallurgy operations (EPA, 2001b, 2001c). Most plants have capture systems and baghouses for these other sources, and those that do not are logical candidates for installing them. A review of emissions tests should indicate if the baghouse is performing properly—a well designed and properly operated baghouse should achieve a PM emissions rate of less than 0.005 gr/dscf (EPA, 2005h).

According to EPA, control options for secondary furnace

1. In other chapters, "primary" is used as distinct from secondary (precursor) emissions. In this chapter, "primary" is used as distinct from fugitive emissions.

Table 10.5

Pollution Control Devices at U.S. Sinter Plants

Plant	Capacity (million tons per year)	Control Device		
		Windbox	Discharge	Cooler
Mittal Steel (formerly ISG, Bethlehem Steel), Burns Harbor, IN	2.9	Scrubber	Baghouse	None
Mittal Steel (formerly ISG, Bethlehem Steel), Sparrows Point, MD	4.0	Scrubber	Baghouse	Cyclone
Mittal Steel (formerly Ispat Inland), East Chicago, IN	1.4	Baghouse	Baghouse	None
Mittal Steel (formerly ISG, LTV Steel), East Chicago, IN	1.9	Scrubber	Scrubber	None
US Steel, Gary, IN	4.4	Baghouse	Baghouse	None

Source: EPA, 2005b, updated October 2005

emissions include furnace enclosures, local hoods, full or partial building evacuation and—for open hood systems—adapting the primary furnace hood to capture secondary emissions as well (this is an option for shops with relatively large flow capacity in the primary control system). PM removal techniques in use as of EPA's 2001 NESHAP assessment included baghouses and wet scrubbers (EPA, 2001a). EPA reported that about half of the BOF shops had a separate capture and control system for emissions from BOF charging and tapping. Ten of these used baghouses and the other two used scrubbers. The other half relied on the primary collection system to capture some of the emissions (EPA, 2001c). For those shops that rely on the primary collection system, adding secondary collection systems would provide significant additional pollution control (EPA, 2001a).

Summary of World Bank and European Commission Recommendations. The World Bank and EC reports identify opportunities for controlling BOF emissions. These are summarized below (World Bank, 1998a; EC, 2001).

- Use enclosures around the BOF.
- Use efficient evacuation and subsequent purification by a fabric filter or ESP during hot metal pretreatment (including hot metal transfer processes, desulfurization and deslagging).
- Recover BOF gas to use as fuel.
- "De-dust" primary emissions through suppressed combustion and dry electrostatic precipitation (in new or existing situations) or scrubbing (in existing situations).
- Cover the lance hole during oxygen blowing and inject inert gas to dissipate the PM.
- "De-dust" secondary gases (emissions that cannot be captured effectively by the primary ventilation system) during charging, tapping, hot metal handling (reladling operations), deslagging of hot metal, and secondary metallurgy by using efficient evacuation followed by fabric filtration or use of an ESP. These techniques can achieve a capture efficiency of about 90 percent.
- Suppress fumes with inert gases during reladling of hot metal.

Sinter Plants

Of the five sinter plants in the U.S., three use a baghouse to control windbox exhaust, while the other two use a venturi scrubber (Branscome, 2005a). When developing MACT standards in 2001, EPA determined that both baghouses and venturi scrubbers were capable of achieving the MACT floor level and that no other control devices could achieve better emissions control (EPA, 2001c). At the discharge end, emissions are controlled by local hooding and ventilation to one or more baghouses or wet scrubbers. Four sinter plants in the U.S. use baghouses and one uses a wet scrubber (Branscome, 2005a; EPA, 2001a). Finally, while all the plants operate sinter coolers to cool the sinter product prior to storage, only one has a control device. This plant vents to a cyclone, while the others vent directly to the atmosphere (EPA, 2001a).

Requiring the other four plants to install control devices for their coolers thus represents the most significant emissions reduction opportunity at sinter plants in the U.S. The coolers can be a significant source of PM emissions (several hundred tons per year), because of the high volumetric flow rate and PM loading. Two plants that recently closed had baghouses, but currently only one controlled cooler remains (Branscome, 2005a).

Table 10.5 shows the pollution control devices in operation at the five sinter plants in the U.S.

The EC has identified a number of opportunities for further pollution reductions at sinter plants in addition to baghouses and scrubbers (EC, 2001).

Sinter plants can cut SO_2 emissions by:

- Reducing the sulfur content of the sinter feed; in particular, reducing the sulfur content of the coke breeze, which consists of small-grade coke with particle sizes under 5 millimeters (mm).
- Making the sinter feed more basic (as opposed to acidic).
- Using coarser coke breeze (5–6 mm) instead of normal grain sizes (under 3 mm). One plant achieved a 40 percent reduction in SO_2 emissions.
- Using regenerated activated carbon (RAC) for desulfurization. The RAC process allows several components—including SO_2, hydrogen chloride, hydrogen fluoride, mercury, and sometimes, depending on the design, NO_x—to be removed from the waste gas. SO_2 may be reduced by more than 95 percent, while nitrogen control efficiency can be as high as 80–90 percent, depending on operating temperature, ammonia addition and design.
- Focusing controls on the end of the strand, closest to the sinter cooler, since emissions increase along the sinter strand where temperatures are higher.

Plants can also reduce energy use by reusing the waste heat from the main exhaust gas from the sintering machines or by reusing cooling air from the sinter cooler.

Minimills

In an EAF, charging and melting scrap and tapping steel all generate emissions of metal dust and gaseous products. Iron oxides are the primary component of resulting PM emissions, although the composition of the PM varies depending on the composition of the scrap metal and other materials that are used. SO_2 and NO_x are generated in addition to PM (EPA, 2004c).

In general, most minimills in the U.S. are controlled to some degree, because the applicable New Source Performance Standards (NSPS) were developed over three decades ago. Since then, they have been updated only once, in 1984. Minimills have expanded rapidly as a result of modifications and new plants and have therefore become increasingly subject to the NSPS.

During melting, plants control primary emissions by using a shell to carry emissions to a baghouse. All but one plant in the U.S. use a baghouse to control primary emissions from melting, and each of those can meet the NSPS of 0.0052 gr/dscf. The one plant without a baghouse uses a wet scrubber and has an emissions rate approximately ten times higher than the NSPS (Branscome, 2005b).

Fugitive emissions from charging, tapping, and melting are controlled with baghouses at all plants. Ten plants are not subject to opacity limits for fugitive emissions as stringent as the NSPS (6 percent opacity). Most of these plants would have to upgrade to meet the NSPS limit for fugitive emissions. Cost information for improved capture systems is not currently available, but EPA is developing the information and plans to publish it by early 2006 (Branscome, 2005b).

All but one minimill in the U.S. also use hoods and baghouses to control emissions from the ladle metallurgy process and from the argon oxygen decarburization vessel (a processing unit used to produce specialty steels). The

Table 10.6

NSPS for Iron and Steel Plants (40 CFR §§60.140, *et seq.* (subpart N), §§60.140a, *et seq.* (subpart Na), §§60.270, *et seq.* (subpart AA), and §§60.270a, *et seq.* (subpart AAa))

Source	Applicable Date[a]	Unit	PM	Opacity (percent)
Primary emissions from BOFs	After June 11, 1973	Any	50 mg/dscm (0.022gr/dscf)[b]	10[c]
	After January 20, 1983	Open hood	50 mg/dscm (0.022gr/dscf)	10[c]
		Closed hood	68 mg/dscm (0.030 gr/dscf)	10[c]
Secondary emissions from basic oxygen steel making facilities	After January 20, 1983	BOF shop roof monitor or other openings	NA	10[c]
		Control device used only for secondary controls[d]	23 mg/dscm (0.010 gr/dscf)	5
EAFs	Between October 11, 1974 and August 17, 1983	Control device	12 mg/dcsm (0.0052 gr/dscf)	3
		Shop	NA	6[e]
		Dust-handling equipment	NA	10
EAFs and argon-oxygen decarburization vessels	After August 17, 1983	Control device	12 mg/dscm (0.0052 gr/dscf)	3
		Shop	NA	6
		Dust-handling equipment	NA	10

NA = No applicable standards

a. The applicable date is the date after which units commenced construction, modification, or reconstruction.
b. Mg/dscm = milligrams per dry standard cubic meter; gr/dscf = grains per dry standard cubic foot.
c. Opacity between 10 and 20% may occur once per steel production cycle.
d. Fume suppression systems used to control secondary emissions and control devices used to collect both primary and secondary emissions from a BOF are not subject to these limits.
e. There are various exceptions to the 6 percent rule (e.g., during charging and tapping periods).

Source: EPA 2000b, 2000c, 2005g, 2005h

one plant that does not—the same plant mentioned above that lacks a baghouse—evacuates its emissions from the argon oxygen decarburization vessel to the same scrubber it uses to control primary melting emissions (EPA, 2004c).

One option for achieving further emissions reductions is to introduce a Fuchs shaft furnace, which preheats the scrap and uses positioning techniques to reduce air emissions, including PM. These types of furnaces are in operation in Arizona, the United Kingdom, China, Malaysia and Turkey (Environment Canada, 2002).

EAFs may also employ the Consteel process. Developed in the mid-1980s, this process continuously charges scrap to the furnace via an enclosed conveyor system, thereby reducing PM emissions. Moreover, the hot furnace waste gases move in the opposite direction from the scrap and transfer heat energy to the scrap. A number of plants in the U.S. and Japan use the Consteel process (DOE, 1994; Environment Canada, 2002; Worrell, 1999).

In addition, minimills can take a variety of steps to cut energy use, which may not directly reduce $PM_{2.5}$ emissions onsite but will reduce electricity use and its associated emissions. A number of possibilities are outlined in a 1999 report by the Lawrence Berkeley National Laboratory, "Energy Efficiency and Carbon Dioxide Emissions Reduction Opportunities in the U.S. Iron and Steel Sector." For instance, using foamy slag to cover the arc and melt surface (to reduce heat losses), using direct current arc furnaces instead of conventional alternating current, and preheating scrap by using waste heat are measures that all save significant energy (Worrell, 1999).

As discussed in other sections, the World Bank and the EC have also put forth suggestions concerning best available emissions control technologies for minimills (World Bank, 1998a; EC, 2001). Their suggestions include:

- Locate EAFs in enclosed buildings.
- Control dust by using fabric filters, direct off-gas extraction, hood systems, dog-houses (so called because of their appearance), total building evacuation, and bottom tapping. For primary and secondary emissions from EAFs, up to 98 percent collection efficiency is achievable. Moreover, minimizing dust also reduces heavy metal emissions (except for those present in the gas phase, such as mercury).
- Preheat scrap to recover waste heat from off gas.
- Improve feed quality by using selected scrap.
- Reduce NO_x emissions by using natural gas as fuel, low-NO_x burners, and hydrogen peroxide and urea in stainless steel pickling baths.

Regulatory Authority

As indicated in Chapter 4, states and the federal government share responsibility for regulating criteria pollutants (e.g., $PM_{2.5}$, SO_2, NO_x) from stationary sources like those operating at iron and steel plants. The federal government has more of the burden of regulating air toxics but—for both conventional and toxic types of pollution—states and localities are free under federal law to adopt more stringent standards than the Clean Air Act requires.

The sections that follow review federal standards that apply under the Clean Air Act to emissions from iron and steel plants.

Reasonably Available Control Technology

State Implementation Plans (SIPs) for nonattainment areas must include Reasonably Available Control Technology (RACT) for new and existing sources. RACT standards are non-binding federal guidelines; for VOCs, the guidelines are known as Control Technique Guidelines (CTGs). Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

The ACTs applicable to NO_x emissions from iron and steel mills can be found at www.epa.gov/ttn/catc/products.html#aptecrpts (EPA, 2005j).

New Source Performance Standards

The NSPS apply to new, modified and reconstructed sources in specific source categories, in both attainment and nonattainment areas.

The requirements of 40 CFR §§60.140, *et seq.* (subpart N) apply to primary emissions from BOFs that commence construction, modification, or reconstruction after June 11, 1973.

The requirements of 40 CFR §§60.140a, *et seq.* (subpart Na) apply to secondary emissions from BOF steel making facilities that commence construction, modification, or reconstruction after January 20, 1983.

The requirements of 40 CFR §§60.270, *et seq.* (subpart AA) apply to EAFs that commenced construction, modification, or reconstruction after October 21, 1974 and on or before August 17, 1983.

The requirements of 40 CFR §§60.270a, *et seq.* (subpart AAa) apply to EAFs and Argon-Oxygen Decarburization vessels that commence construction, modification, or reconstruction after August 17, 1983.

The NSPS for these sources at iron and steel plants appear in Table 10.6.

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, Best Available Retrofit Technology (BART) requirements apply, in both attainment and nonattainment areas, to $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from large sources in 26 identified source categories that began operation between 1962 and 1997. The source categories include iron and steel mill plants, coke oven batteries, and sintering plants. States have not yet developed their BART rules.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) and Prevention of Significant Deterioration (PSD) requirements apply to new and

Table 10.7

Recent BACT Determinations for Steel Mills[a]

Permit Date	Date of Last Determination	Process	Pollutant	Control	Efficiency (%)
Minimill (Electric Arc Furnace)					
Wheeling Pittsburgh Steel, Jefferson County, Ohio					
6-Jan-05	12-Aug-05	EAF material handling	PM_{10}	Baghouse and vent filters	99
		LMF material handling	PM_{10}	Baghouse and vent filters	99
Rocky Mountain (CF & I Steel), Pueblo, Colorado					
21-Jun-04	20-Jan-05	EAF	PM_{10}	High-efficiency filter baghouse	
			PM	Fabric filter	
			SO_2	Alternative raw materials and process controls	
Quanex Corporation, Jackson, Michigan					
8-Dec-03	13-Oct-05	EAF	PM_{10}	Side draft/canopy hood and baghouse	99.90
North American Stainless, Montgomery, Indiana					
21-Nov-03	6-Feb-04	Ladle metallurgy	PM	Baghouse	
		EAF dust treatment		Scrubber	
		EAF dust handling		Bin vents	
		AOD	NO_x	Natural gas fired oxy fuel burners	
Nucor Steel Decatur, Morgan, Alabama					
7-Jul-02	25-Aug-03	EAF	PM	Baghouse	99
Integrated Steel Mill					
Permit Date	**Date of Last Determination**	**Process**	**Pollutant**	**Control**	**Efficiency (%)**
Wheeling Pittsburgh Steel, Jefferson County, Ohio					
6-Jan-05	12-Aug-05	BOF (material handling)	PM_{10}	Baghouses	99
		BOF	PM	Wet venturi scrubber, combustion hood and flame suppression	99
		BOF (fugitive emissions)		Baghouse and bin vent filters	99

Continued on next page

Table 10.7 (continued)

Recent BACT Determinations for Steel Mills[a]

Coke Plant

Permit Date	Date of Last Determination	Process	Pollutant	Control	Efficiency (%)
Haverhill North Coke Company, Scioto County, Ohio					
27-Feb-01	14-Aug-03	Coking	SO_2	Dry scrubber, lime spray dryer, and low-sulfur coal	92[b]
			NO_x	Staged combustion	85[c]
			PM_{10}	Baghouses	99
		Charging	PM_{10}	Baghouse with traveling hood	93
		Pushing	PM_{10}	Baghouse with a shed extending the length of the battery to capture all emissions	98
		Loading	PM	Load-in coal, coke and coke breeze with stacking tube; load-in coal with water sprays or dust suppressant; load-out coal under pile gravity feed and water spray; load-out coke with under pile gravity feed	70-95[d]
		Handling	PM_{10}	Rail car to use bottom dumping and be enclosed; belt conveyors to be enclosed; wet and/or chemical suppression	
		Coke breeze crushing/screening/ conveying	PM_{10}	Baghouse	

a. All determinations are for PSD (BACT), not NSR (LAER).
b. Also, 0% leakage and negative pressure in oven.
c. Also, no visible emissions from waste gas piping and 0% leakage from coke oven doors.
d. Depending on the emissions point.

Source: EPA, 2005b

modified large stationary sources in nonattainment and attainment areas, respectively. Under these programs, plants in nonattainment areas are subject to Lowest Achievable Emission Rate (LAER) standards, while plants in attainment areas are subject to Best Available Control Technology (BACT) requirements. Neither BACT nor LAER standards are codified because they are intended to reflect technology advancements as they occur.

EPA's RACT/BACT/LAER Clearinghouse has information on BACT determinations since 2000 for a number of minimills, a coke plant, and an integrated iron and steel mill (See Table 10.7). The Clearinghouse does not have information on any LAER determinations for iron and steel plants.

Past BACT determinations for minimills—only some of which are included in Table 10.7—generally include a baghouse to control PM_{10} emissions from identified emissions points; in addition, some also require vent filters and canopy hoods. The coke plant BACT determination requires pollution controls during charging, pushing, loading, handling and conveying. The PM_{10} controls involve a baghouse, while SO_2 and NO_x are controlled by a scrubber and staged combustion, respectively. Finally, the BACT determination for an integrated steel mill requires a baghouse, wet venturi scrubber and flame suppression to control PM_{10} and PM (including particles larger than PM_{10})

Table 10.8a

Existing Byproduct Batteries

	MACT track limits (percent)			LAER extension track limits (percent)		
	12/31/95	1/1/03	7/14/05[a]	11/15/93	1/1/98	1/1/10
Tall doors[b]	6.0	5.5	4.0	7.0	4.3	4.0
Short doors (or otherwise exempted from tall door standards)[b]	5.5	5.0	3.3	7.0	3.8	3.3
Topside port lids[b]	0.6	0.6	0.4	0.83	0.4	0.4
Offtake systems[b]	3.0	3.0	2.5	4.2	2.5	2.5
Visible emissions per charge[c]	12 seconds			12 seconds		

Table 10.8b

New and Rebuilt Byproduct Batteries

	New or greenfield batteries[d] (percent)		Brownfield or padup rebuild[d] batteries (percent)	
Emissions points	Same technology	New technology[e]	MACT track limits	LAER extension track
Tall doors[b]	0.0	4.0	Same as for existing batteries (see Table 10.8a)	4.0
Short doors (or otherwise exempted from tall door standards)[b]	NA	3.3		3.3
Topside port lids[b]	0.0	0.4		0.4
Offtake systems[b]	0.0	2.5		2.5
Visible emissions per charge[c]	34 seconds	12 seconds		12 seconds

NA = No applicable standards

a. Residual risk standard, finalized April 15, 2005 (EPA, 2005e).

b. Based on a 30-run rolling average of percent leak.

c. Based on the logarithmic 30-day rolling average.

d. New and greenfield batteries cannot qualify for the LAER extension track and must meet the residual risk standard (shown in Table 10.8a).

e. Padup rebuild means a coke oven battery that is a complete reconstruction of an existing coke oven battery on the same site and pad, as of November 15, 1990, without an increase in design capacity.

f. EPA determines case-by-case limits, which must be more stringent than the limits shown, or less than the equivalent level of mass emissions.

Source: EPA 2005f

from identified BOF-related operations.

Maximum Achievable Control Technology

MACT standards address emissions of HAPs from new and existing sources in identified industry categories in both attainment and nonattainment areas. MACT standards apply to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs. In some MACT rules, such as those referenced here, EPA regulates PM as a surrogate for HAPs.

MACT Standards for Coke Oven Leaks. Under 40 CFR §§63.300, *et seq.* (subpart L), visible emissions limits (opacity standards) apply to doors, lids, offtake systems and charging operations for new and existing coke oven batteries.

The MACT standard for coke oven leaks offers existing facilities two options: a MACT and a LAER track. The LAER track exempts existing facilities from a residual risk standard until 2020 but requires them to meet standards, some of which are more stringent and apply earlier than the MACT standards. Table 10.8a summarizes emissions limits for existing coke oven batteries under both tracks.

A battery is considered "new" if it commenced

Table 10.9

MACT Standards for Integrated Iron and Steel Plants (40 CFR §§63.7780, *et seq.* (subpart FFFFF) and Coke Ovens (40 CFR §§63.7280, *et seq.* (subpart CCCCC))

	Type of Plant	Unit	New or Existing	PM Limit	Opacity Limit (percent)
Integrated iron and steel manufacturing facilities	Sinter	Windbox exhaust stream	New	0.3 lb/ton product sinter	
			Existing	0.4 lb/ton product sinter	
		Discharge end	New	0.01 gr/dscf[a]	10[b]
			Existing	0.02 gr/dscf	20[b]
		Cooler stack	New	0.01 gr/dscf	
			Existing	0.03 gr/dscf	
	Blast furnace	Casthouse	New	0.003 gr/dscf	15[b]
			Existing	0.01 gr/dscf	20[b]
	Basic oxygen furnace[c]	Primary emissions control system – closed hood	New and Existing	0.03 gr/dscf	
		Primary emissions control system –open hood	New	0.01 gr/dscf	
			Existing	0.02 gr/dscf	
		Secondary emissions control system	New	0.0052 gr/dscf	
			Existing	0.01 gr/dscf	
		Hot metal transfer, skimming, or desulfurization	New	0.003 gr/dscf	
			Existing	0.01 gr/dscf	
		Ladle metallurgy	New	0.004 gr/dscf	
			Existing	0.01 gr/dscf	
		Roof monitor	New		10[d]
			Existing		20[e]
Coke ovens	Cokeside shed		New or Existing	0.01 gr/dscf	
	Moveable hood	Vented to a stationary control device		0.02 lb/ton coke	
	Mobile scrubber cars[f]	Short batteries		0.03 lb/ton coke	
		Tall batteries		0.01 lb/ton coke	
	Mobile scrubber cars[g]			0.04 lb/ton coke	
	Battery stack	Normal coking cycle			15[h]
		Extended coking			10[h]

a. Mg/dscm = milligrams per dry standard cubic meter; gr/dscf = grains per dry standard cubic foot.
b. Six-minute average.
c. Basic oxygen furnace includes both top and bottom blown furnaces, but does not include argon oxygen decarburization furnaces.
d. Three- or six-minute average; opacity level may reach up to 20% for one period per each steel production cycle.
e. Three-minute average.
f. Not capturing emissions during travel.
g. Capturing emissions during travel.
h. Daily average.
Source: EPA, 2003, 2005i

construction or reconstruction as of November 15, 1990. Table 10.8b shows the emissions limits for new and rebuilt batteries.

Finally, fugitive emissions from sources built on or after August 9, 2004 are subject to an opacity limit of 20 percent and a PM limit from charging emissions control devices under 0.0081 pounds per ton (lb/ton) of dry coal charged.

MACT Standards for Integrated Iron and Steel Plants. PM emissions from integrated iron and steel manufacturing facilities are regulated by 40 CFR §§63.7780, *et seq.* (subpart FFFFF). In particular, the subpart covers emissions from sinter plants, blast furnaces and BOFs. A unit is considered an "existing unit" if construction or reconstruction commenced before July 13, 2001; a "new unit" is one for which construction or reconstruction commences on or after July 13, 2001.

PM emissions from coke ovens are regulated under 40 CFR §§63.7280, *et seq.* (subpart CCCCC). In particular, it covers emissions from pushing, quenching and battery stacks. A unit is considered "existing" if it commenced construction or reconstruction before July 3, 2001; a "new unit" is one for which construction or reconstruction commences on or after July 3, 2001.

The MACT standards for these sources at iron and steel plants appear in Table 10.9.

As a result of MACT requirements, a number of plants have installed new or improved emissions control systems. For instance, Indiana's ISG Burns Harbor plant is in the process of installing two new baghouses for its blast furnaces, a new baghouse for the BOF vessels, and broken bag detectors and monitoring equipment. Ispat Island in East Chicago, Illinois is currently upgrading the scrubber at its BOF (Perry, 2005).

In addition, EPA proposed amendments to the MACT standards for integrated iron and steel manufacturing facilities on August 30, 2005, in response to a lawsuit brought by several steel companies. The revised limits would apply to each sinter cooler instead of to each sinter cooler stack and would set a 10 percent opacity limit instead of the current PM emissions limit of 0.03 gr/dscf. The proposal would also revise the operating limit for an ESP that controls emissions from a BOF; specifically, it would require that the hourly average opacity of emissions from the control device be maintained at or below 10 percent. Finally, the proposed amendments would reduce the frequency of performance tests, allow alternative monitoring systems, and change operating and maintenance plans (EPA, 2005c).

Clean Air Interstate Rule

None of the sources operating at iron and steel mills that are discussed here can be included in any of the federal Clean Air Interstate Rule (CAIR) trading programs without disqualifying the state from participation in federal trading regimes.

State and Local Initiatives

The federal MACT standards for coke plants and for integrated iron and steel plants require self-monitoring. It is important for local and state agencies to ensure that facilities adhere to the self-monitoring requirements, including reporting the data to the permitting agency.

In addition, a number of coke and steel plants have received compliance extensions for meeting MACT standards. To the extent possible, areas should limit compliance extensions to the absolute minimum.

In terms of states and localities imposing standards stricter than the federal standards, Allegheny County, Pennsylvania stands at the forefront. Most importantly, the Allegheny regulations set instantaneous limits for visible emissions from doors, charging, lids, and offtake systems, as well as for PM emissions from pushing and combustion stacks. The federal MACT standards are based on an average (usually 30 days or runs) (ACHD, 2005; EPA, 2005f). Instantaneous standards coupled with frequent inspections by Allegheny County staff prevent Allegheny coke plants from delaying action to correct problems (Westman, 2005).

To comply with these stricter regulations, coke plants in Allegheny County employ both technology measures and work practice standards. In terms of technology, the coke plants use either a stationary shed or a moving hood, either of which evacuates pushing emissions to a baghouse. These options are equally effective at emissions reduction. (While stationary sheds have the added benefit of also capturing door emissions, doors may experience more leaks because they are not easily visible and may not be cleaned or repaired as readily with a shed in place.)

It is also extremely important that coke plants use effective operational practices. Workers at the Allegheny plants undergo extensive training on door cleaning, sealing, lid operation, pollution control device management, and various other operations in the coke plant, and there is substantial oversight of workers' performance (Westman, 2005).

The U.S. Steel plant in Allegheny County represents an interesting case study of options for controlling SO_2 emissions from coke plant operations. A few years after the Allegheny County Health Department established

a hydrogen sulfide (H_2S) standard of 40 grains per 100 dry standard cubic feet (40 gr/100dscf) for desulfurized coke oven gas, pressure mounted on U.S. Steel to reduce emissions further because of lingering odor issues and the region's PM problem. In response, U.S. Steel authorized plant operators to do what they thought necessary to achieve emissions under 30 gr/100dscf. The operators installed more sensitive controls and better instruments to improve their understanding of all emissions points and engaged in a number of fine-tuning operations to achieve the 30 gr/100 dscf goal. With the success of the "Under 30" program, U.S. Steel implemented a similar challenge for an "Under 20" program to make even further emissions reductions to under 20 gr/100 dscf. The plant now produces coke oven gas with H_2S levels between 15 and 20 gr/100dscf (Westman, 2005).

Allegheny County is also home to U.S. Steel's Edgar Thomson plant, which uses an entirely enclosed BOF. Designed as a closed-in structure two decades ago to deal with the area's "dustfall" PM problem, the plant has reduced fugitive emissions substantially using primary and secondary emissions controls (baghouses). However, the plant continues to generate some fugitive emissions from leaking side doors and a roof.

Indiana has a number of regulations for both coke plants and integrated iron and steel mills that impose standards stricter than the federal MACT requirements. For instance, in the case of coke plants, Indiana has set opacity limits for bypass heat exchanger stacks and for pushing controls, whereas the MACT standards have no opacity limits (Kuh, 2001a). Likewise, in the case of integrated iron and steel mills, Indiana has established opacity limits for BOFs, whereas the federal MACT standards require only that site-specific opacity limits be developed during initial performance tests on ESPs controlling BOF emissions. Finally, Indiana regulates both the oil and grease content of sinter plant feedstock, while the federal MACT standards regulate only oil content (Kuh, 2001b).

The state of Michigan has a regulatory scrap management program for iron and steel plants. While the primary goal of the program is to reduce mercury emissions by removing switches from automobiles, the program also cuts SO_2 emissions by requiring that scrap be clean and free of oily residues. This approach is also useful for reducing emissions of condensable particulate emissions caused by melting oily scrap.

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Chapter 11

Petroleum Refineries

Introduction

Petroleum is the largest source of energy in the U.S., accounting for approximately 40 percent of the nation's total energy consumption (EIA, 2003). Measured by energy content (i.e., British thermal units, or Btus, consumed), Americans use more petroleum than either coal or natural gas. Before petroleum can be used, however, it must first go through an extensive refining process.

Using physical, thermal and chemical processes, petroleum refineries separate crude oil into its major fractions, which are then processed further into finished petroleum products. The primary products of the industry fall into three major categories: fuels (gasoline, diesel, jet fuel); finished products (solvents, lubricating oils, asphalt); and chemical industry feedstocks (ethylene, propylene, butadiene). Fuels account for almost 90 percent of the petroleum products produced in the U.S., while finished products and petrochemical feedstocks each account for about 5 percent (EPA, 1995a).

The petroleum refining process involves a vast complex of operations, from desalting[1] and distilling[2] to cracking (breaking down large hydrocarbons into smaller molecules) and polymerizing (combining small hydrocarbons to form larger molecules). Different refineries may have slightly different sets of operations, in part because the composition of crude oil—a mixture of various hydrocarbons with differing amounts of impurities—varies substantially depending on its source. Also, refineries serve different markets with different product needs. As a result, no two refineries are exactly alike.

The petroleum refining industry consumes large amounts of energy derived from a variety of fuel sources. The industry obtains about 65 percent of the energy it consumes for heat and power from byproducts of the refining process, including refinery gas, petroleum coke and other oil-based byproducts (DOE, 2000). Refineries also rely on natural gas for a substantial share of their energy needs (DOE,

1. Crude oil often contains water, inorganic salts, suspended solids and water-soluble trace metals. Desalting is the process of removing these contaminants.

2. Distillation is the process of separating different petroleum products.

2000).

The petroleum refining process produces a host of pollutants—including fine particulate matter ($PM_{2.5}$), sulfur dioxide (SO_2) and nitrogen oxides (NO_x)—from the combustion of various fuels in boilers, process heaters, engines and flares. This chapter discusses technology and policy options for addressing these different sources of emissions. It addresses the refining process only and does not cover the exploration, production or transport of petroleum products.

Sector Profile

A small number of large plants dominate the petroleum refining industry and an even smaller number of companies own and operate those plants. There are currently 144 refineries in the U.S., concentrated in a relatively small number of states (EIA, 2005a). Table 11.1 presents a summary of U.S. refining capacity by state. States with large numbers of refineries are those that either possess crude oil sources (onshore petroleum terminals, oil and gas extraction areas) or many consumers (heavily industrialized areas). U.S. petroleum refineries produce approximately $220 billion worth of products annually and employ 64,000 people (EIA, 2005b).

Over the past two decades, the refining industry has undergone substantial restructuring. In 1981, the government terminated the Crude Oil Entitlements Program, which had encouraged smaller refineries. This forced the least efficient plants to close. The 1990s witnessed increasing industry consolidation and concentration, as a growing number of firms merged in order to reduce costs by sharing assets and minimizing operating costs. The number of major U.S. energy companies dropped from 19 in 1990 to nine in 2001; meanwhile the merger of Exxon and Mobil created the world's largest publicly traded energy company. Of the 300 refineries that were in operation in 1982, only 176 remained in 1994—and only 144 continue to operate today (EPA, 1995a; EIA, 2005a). Throughout this period, however, refining capacity did not diminish. In fact, even as 47 refineries shut down during the 1990s, overall refining capacity increased 6 percent (EIA, 2002).

Emissions

A host of operations produce air emissions within the petroleum refining process; however, there are a few dominant sources. Table 11.2 provides an estimate of $PM_{2.5}$, SO_2 and NO_x emissions from the petroleum refining industry by process category. The estimates are based on EPA's 1999 National Emissions Inventory. Boilers, process heaters, and catalytic cracking units are dominant sources of $PM_{2.5}$, SO_2 and NO_x emissions. Internal combustion engines and flares produce a more modest, but still significant, share of $PM_{2.5}$, SO_2 and NO_x emissions. Petroleum refineries are also major sources of volatile organic compounds (VOCs), hazardous air pollutants (both organics and metals) and carbon monoxide (CO). These pollutants are not the focus of this report.

Catalytic cracking units are a major source of air pollution at refineries. Emissions from catalytic cracking processes

Table 11.1

Capacity and Number of Petroleum Refineries by State

Rank	State	Capacity (thousand barrels/day)	% of Total U.S. Capacity	Number of Refineries	% of Total Number of U.S. Refineries
1	Texas	4,628	27%	25	17%
2	Louisiana	2,773	16%	17	12%
3	California	2,005	12%	21	15%
4	Illinois	896	5%	4	3%
5	Pennsylvania	770	5%	5	3%
6	Washington	616	4%	5	3%
7	New Jersey	615	4%	5	3%
8	Ohio	551	3%	4	3%
9	Oklahoma	485	3%	5	3%
10	Indiana	433	3%	2	1%
Subtotal		13,772	81%	93	65%
U.S. Total		17,006	100%	144	100%

Source: EIA, 2005a

Table 11.2

Petroleum Refining Sector Emissions (1999)

Process Type	$PM_{2.5}$		SO_2		NO_x	
	Tons/Year	Percent of Sector Emissions	Tons/Year	Percent of Sector Emissions	Tons/Year	Percent of Sector Emissions
Boilers	9,437	26%	103,026	25%	70,260	29%
Internal combustion engines	1,515	4%	862	<1%	18,449	8%
Catalytic cracking units	9,646	26%	136,780	33%	29,603	12%
Process heaters	9,880	27%	71,789	17%	100,380	41%
Flares	394	1%	27,763	7%	8,503	3%
Cooling towers	1,706	5%	--	--	--	--
Elemental sulfur production	7	<1%	18,872	5%	274	<1%
Other process related	4,145	11%	57,956	14%	16,288	7%
Total	36,729	100%	417,048	100%	243,757	100%

Source: EPA, 2005e (based on facilities listed under SIC code 2911)

come from two sources: process heaters and flue gas from catalyst regeneration. (Catalyst regeneration is the process by which catalysts are cleaned and restored after being contaminated in a reaction.) The catalyst regenerator emits particulates, SO_2, NO_x and other pollutants. Fluid catalytic cracking units are a particular concern because of their high rates of PM emissions (EPA, 1995b).

Sulfur compounds present in crude oil are converted to hydrogen sulfide in the cracking and hydro treating processes. Many refineries convert hydrogen sulfide gas to elemental sulfur using a "Claus" sulfur recovery plant. The conversion process produces elemental sulfur, which can be sold for agricultural chemicals and other applications. The process also produces SO_2 emissions.

Petroleum refineries use flares to combust vapors rather than discharging them directly to the atmosphere. The flares produce $PM_{2.5}$, SO_2 and NO_x emissions. Steam may be injected into the flare tip to reduce visible smoke.

Refineries also produce significant fugitive emissions. Fugitive emissions are generated by thousands of potential emissions sources, including valves, flanges, pump and compressor seals, tanks and process drains. Evaporation of spilled petroleum liquids and gases also causes fugitive emissions. Generally, the control of fugitive emissions involves minimizing leaks and spills through equipment changes, procedure changes, and improved monitoring, housekeeping and maintenance practices (EPA, 1995a).

The Petroleum Refining Process

The petroleum refining process is often described in terms of five basic steps: fractionation, conversion, treatment, formulation and blending, and auxiliary processes. Different refineries perform some or all of these activities.

Petroleum refining begins with the crude, or unprocessed, oil that is pumped from the ground or ocean floor. Crude oil contains various elements or compounds, but the key components are carbon and hydrogen that form hydrocarbon molecules of different lengths and structures, from straight chains to branching chains and rings. Fractionation (distillation) is the separation of crude oil in atmospheric and vacuum distillation towers into groups of hydrocarbon compounds with differing boiling-point ranges, called fractions or cuts. Because of the large volumes of crude oil subjected to this initial processing step, atmospheric and vacuum distillation account for a significant share (35–40 percent) of the energy a refinery consumes (DOE, 2000).

Refineries use cracking, reforming and other conversion processes that change the size and structure of the hydrocarbon molecules to convert most distillation products into more usable products. These processes break down longer chain molecules into smaller ones by heating or using catalysts.

The converted products are then subjected to various treatment and separation processes to remove undesirable constituents and improve product quality. Treatment may include the removal or separation of aromatics as well as impurities and undesirable contaminants, such as sulfur, nitrogen and heavy metals. It may involve chemical or physical separation (such as dissolving, absorption or precipitating) by means of a variety of processes, including

desalting, drying, hydrodesulfurizing, solvent refining, sweetening, extracting solvents and solvent dewaxing.

Formulating and blending is the process of mixing and combining hydrocarbon fractions, additives and other components to produce finished products with specific performance properties. For example, refiners combine different mixtures of hydrocarbons to produce gasoline with different octane ratings.

There are several auxiliary processes that are vital to the functioning of a petroleum refinery, including power generation, solid waste and wastewater treatment, process-water treatment, and cooling and other utility services.

Petroleum refineries use combustion equipment (e.g., boilers and process heaters) throughout the refining process to produce electricity, steam and heat. Process heaters fueled by refinery gas (a byproduct of the refining process), natural gas, residual fuel oils or some combination thereof are used to produce the heat that is necessary in many of the refining processes (EPA, 1995b). Electric motors, steam turbines and internal combustion engines are used to drive pumps, compressors, blowers and other refinery process equipment. Fuel combustion by these types of equipment is responsible for a significant share of the refinery sector's $PM_{2.5}$, SO_2 and NO_x emissions.

Because of their coincident demand for electrical and thermal energy, petroleum refineries are one of the nation's largest users of combined heat and power technology (also known as CHP or cogeneration). CHP systems are integrated technologies that produce useful thermal and power output, thereby increasing the overall fuel efficiency of the unit. CHP technologies also offer significantly lower pollution emissions rates compared to separate heat and power systems.

Emissions Control Opportunities

Modern pollution control systems can reduce air pollution emissions from petroleum refineries dramatically, as evidenced by the agreements that EPA has reached with several U.S. oil companies. Most recently, EPA and the Department of Justice announced an agreement with ExxonMobil to reduce NO_x and SO_2 emissions by 11,000 tons per year and 42,000 tons per year, respectively, at the company's seven U.S. petroleum refineries (EPA, 2005d). According to EPA, companies that together account for nearly 77 percent of domestic refining capacity have agreed to legally binding agreements to reduce their air pollutant emissions (EPA, 2005d). Process changes, equipment upgrades, pollution control equipment, increased monitoring and better equipment maintenance are all options for controlling refinery air emissions (Davis, 2000). Process heaters, boilers, catalytic cracking units and flares provide the primary opportunities for $PM_{2.5}$, SO_2 and NO_x control.

Process Heaters

Virtually every step of the petroleum refining process relies on process heaters, but their main use is as preheaters for distillation, catalytic cracking, hydroprocessing and hydroconversion (EPA, 1993a). Process heaters raise the temperature of process materials for additional processing (heated feed) and drive chemical reactions in the process heater tubes (reaction feed). For example, process heaters preheat crude oil prior to distillation.

A process heater is an enclosed device in which solid, liquid or gaseous fuels are combusted for the purpose of heating a process material (e.g., crude oil). There are two basic types of process heaters: direct and indirect. Direct-fired systems place the combustion gases in *direct* contact with the process material. *Indirect* systems rely on tubing to separate the combustion gases from the process material. Indirect-fired process heaters are similar to boilers, except that boilers heat water to produce steam rather than to heat process materials. The petroleum sector

Table 11.3

Process Heater Emissions Rates

Fuel	Emissions Rate (lb/MMBtu)		
	$PM_{2.5}$	SO_2	NO_x
Distillate Oil	0.014	0.304	0.143
Residual Oil	0.026–0.046	0.628–1.047 (352–587 ppm)	0.367[a] (286 ppm)
Refinery gas	no data	0.084	0.087

a. For process heaters over 100 million British thermal units per hour (MMBtu/hr). Process heaters under 100 MMBtu/hr have a NO_x emissions rate of 0.313 lb/MMBtu.

Source: EC, 2003; EPA, 1995b.

relies primarily on indirect-fired process heaters. Process heaters are fueled by refinery gas, natural gas, residual fuel oils or combinations thereof, depending on cost, operating conditions and emissions requirements.

Several options are available for controlling emissions from process heaters, including fuel processing prior to combustion, fuel switching, combustion modifications and post-combustion controls.

Fuel processing is a common strategy used to control SO_2 emissions from process heaters. A large portion of the fuel consumed by refineries is generated internally by the different refinery processes and collected in a refinery gas system. Depending on the method of fuel processing used, refinery gas contains varying amounts of sulfur. For example, refinery gas produced by the crude distillation process contains higher concentrations of sulfur than refinery gas produced by catalytic reforming. Higher sulfur refinery gas is normally treated by amine scrubbing to remove hydrogen sulfide before being released to the refinery gas system.

Fuel switching is also an option for reducing emissions from process heaters. For example, SO_2 emissions can be reduced by switching from liquid fuel oils to refinery gas or natural gas. Several of the consent decrees stemming from EPA's national petroleum refinery compliance initiative require the affected facilities to discontinue the burning of fuel oil in all heaters and boilers, except in instances of natural gas curtailment, in an effort to reduce SO_2 emissions. In some cases, the agreements further specify that only low-sulfur fuel oil be used during periods of natural gas curtailment (*U.S. v. Valero*, 2005).

Options are also available for controlling NO_x emissions from process heaters. Natural gas and refinery gas contain little or no fuel-bound nitrogen, but combusting these fuels at high temperature results in the formation of thermal NO_x. The reaction of atmospheric nitrogen and oxygen in the combustion air produces thermal NO_x. Firing refinery fuel gas generally results in higher thermal NO_x formation than firing natural gas due to the higher flame temperatures caused by the higher hydrogen content of refinery gas (EPA, 1993a). Combustion of distillate and residual fuel oils results in the formation of both thermal and fuel NO_x. According to EPA, low-NO_x burners (LNBs), ultra-low-NO_x burners (ULNBs), selective noncatalytic reduction (SNCR) and selective catalytic reduction (SCR) can be used to control NO_x emissions from process heaters (EPA, 1993a). Also, LNBs can be used in combination with flue gas recirculation (FGR), SNCR and SCR. Each of these control technologies is described in Chapter 5, *Boiler Technologies*.

Companies have retrofit many existing process heaters with low-NO_x burners, allowing existing units to achieve NO_x emissions rates in the range of 0.015–0.025 pounds per million British thermal units (lb/MMBtu) (Deason, 2005). New process heaters are capable of NO_x emissions rates in the range of 0.010–0.015 lb/MMBtu per hour (Deason, 2005). Few existing process heaters have been retrofit with SCR controls, in part because of severe space constraints at petroleum refineries. New process heaters, which allow greater flexibility in terms of design and layout, are generally better suited to accommodate an SCR control system (Deason, 2005). Other factors may also limit the application of post-combustion controls. Refinery process heaters are diverse in terms of their size, load and temperature duty. These factors will all influence the feasibility as well as the cost-effectiveness of available control options.

Table 11.4 provides estimates of the cost-effectiveness of alternative NO_x control technologies for an average-

Table 11.4

NO_x Control Options for Refinery Process Heaters

Technology	Cost-Effectiveness ($/ton) by Capacity Factor		
	10%	50%	90%
Low-NO_x burners	12,800–21,900	2,570–4,370	1,430–2,430
Ultra-low-NO_x burners	8,830–14,800	1,770–2,950	981–1,640
Selective noncatalytic reduction	25,900–31,200	5,450–6,670	3,170–3,940
Selective catalytic reduction	71,900–80,100	14,800–16,400	8,460–9,370
Low-NO_x burners with FGR	19,100–26,700	3,960–5,480	2,270–3,120
Low-NO_x burners with SNCR	20,200–25,400	4,300–5,340	2,530–3,119
Low-NO_x burners with SCR	69,300–74,100	14,200–15,200	8,110–8,640

Source: EPA, 1993

sized process heater (77 MMBtu per hour). The cost-effectiveness of different control options varies depending on the utilization rate of the unit, as well as the type of process heater retrofit.

Boilers

The petroleum refining industry also makes extensive use of boilers. Roughly 50 percent of the energy consumed by the sector is devoted to steam production (DOE, 2002). Refinery boilers are used primarily for heating and separating hydrocarbon streams and, to a lesser extent, for producing electricity. When used for heating, the steam usually heats the petroleum indirectly in heat exchangers and returns to the boiler. In direct contact operations, the steam serves as a stripping medium or a process fluid. Refinery boilers are fueled by refinery gas, natural gas and residual fuel oils.

There are three basic options for reducing $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from refinery boilers: combustion controls, post-combustion controls and fuel switching. Chapter 6, *Industrial and Commercial Boilers*, provides a detailed discussion of the control options for industrial and commercial boilers.

Catalytic Cracking Units

The catalytic cracking process converts heavier hydrocarbons into higher value, lower boiling point products—like gasoline and liquefied petroleum gas (LPG)—by combining the process feed (i.e., petroleum residues) with a catalyst at high temperatures. The inputs to the catalytic cracking process come from various parts of the refinery, including the atmospheric and vacuum distillation units. The petroleum refining industry uses several types of catalytic cracking units; however, the fluidized-bed reactor is the most common.

Catalytic cracking units produce $PM_{2.5}$, SO_2 and NO_x emissions from several sources. First, process heaters and/or waste heat from the catalyst regenerator unit (discussed below) pre-heat the oil-based residues that are sent to the catalytic cracking unit. The catalytic cracking process also relies on boilers. Steam is used to enhance the reactions of the process feed and the catalyst and to strip oil from the catalyst. The control technology options for process heaters and boilers are discussed above.

The primary source of emissions from the catalytic cracking process is the catalyst regenerator unit (EC, 2003). (The waste heat from the regenerator unit also provides much of the heat required by the catalytic cracking process.) During the cracking process, coke is deposited on the surface of the catalyst, deactivating the material. The catalyst is regenerated by burning off the coke at high temperatures. The flue gas from the regenerator unit contains $PM_{2.5}$, SO_2 and NO_x (including catalyst fines), and other air pollutants. SO_2 emissions from a catalytic cracking unit can range from 150 to 3,000 parts per million on a dry volume basis (ppmv), NO_x emissions range from 50 to 400 ppmv, and particulate emissions typically range between 200 and 650 milligrams per normal cubic meter of gas (Eagleson, 2005). The particles from the regenerator unit consist of silica/alumina and the nickel and vanadium present in the feedstock. Ninety percent of these particles are smaller than 10 micrometers (μm) in diameter (PM_{10}) (EC, 2003).

Cyclones and electrostatic precipitators (ESPs) are often used to control particulate emissions from regenerator units. Cyclones capture 30–90 percent of total particulates. However, cyclones are less effective at controlling PM_{10}. ESPs can achieve control efficiencies in excess of 95 percent and are effective at capturing PM_{10}. Baghouses are generally not used with catalytic cracking units because of the space required and because of the pressure drop they cause in the flue gas stream (EC, 2003).

Refineries make widespread use of wet scrubbing for the simultaneous control of SO_2 and particulates from regenerator units. A wet gas scrubber can achieve greater than 95 percent SO_2 control and moderate NO_x removal (70 percent) with the inclusion of an additional treatment tower. A wet gas scrubber can also reduce particulate emissions, although the technology is reported to be less effective for the control of smaller particles (EC, 2003).

Belco Technologies, a provider of air pollution control systems, reports that its EDV® wet scrubbing system can readily meet New Source Performance Standards (NSPS) for particulate and SO_2 emissions (see Table 11.5). Testing performed at a U.S. refinery found that the system was achieving greater than 99 percent SO_2 removal efficiency and greater than 92 percent particulate control (Eagleson, 2005).

Several of the consent decrees arising from EPA's refinery enforcement initiative require the installation of wet gas scrubbers at fluid catalytic cracking units. For example, Sunoco is required to install wet gas scrubbers at its Marcus Hook refinery in Toledo, Ohio and at its refinery in Philadelphia, Pennsylvania (*U.S. v. Sunoco*, 2005). The agreement further specifies that SO_2 concentrations be limited to 25 ppmv on an annual basis (at 0 percent oxygen).

Refineries also use SO_2 adsorption catalysts to control SO_2 emissions from regenerator units. A catalyst is added to the system, liberating the sulfur in the coke for capture in the refinery's amine scrubbing system. Control-effectiveness depends on the amount of the additive injected into the system.

NO_x emissions can be reduced with SCR or SNCR

controls. A modest but growing number of SCR systems have been installed at fluid catalytic cracking units. The first SCR control system installed in the U.S. on a fluid catalytic cracking unit was at the ExxonMobil Torrance Refinery in 2000 (*U.S. v. ExxonMobil*, 2005). Currently, the only SNCR installation at a catalytic cracking unit is in Japan (EC, 2003). However, several EPA consent decrees in the U.S. require the installation of SCR or SNCR controls at fluid catalytic cracking units.

The BOC Group has developed a NO_x control technology, known as LoTOx™, that can achieve reductions comparable to those of an SCR system. LoTOx™ is a selective, low-temperature oxidation technology that uses ozone to oxidize NO_x to water soluble nitric pentoxide, which reacts with moisture in the flue gas to form nitric acid. A wet scrubber then removes the oxidized NO_x compounds. The LoTOx™ technology is integrated within the EDV® wet scrubbing system discussed previously for simultaneous control of $PM_{2.5}$, SO_2 and NO_x. The system reduces NO_x emissions to 10 ppm or less (Sexton, 2004).

Sulfur Recovery Unit

The sulfur recovery process converts hydrogen sulfide gas to elemental sulfur, which can be sold for other applications. SO_2 emissions from the sulfur recovery process are directly related to recovery efficiency. Higher recovery efficiencies mean less sulfur is released to the air. Several factors influence the efficiency of a sulfur recovery unit. In some cases, two or more catalytic stages are used in series to recover the sulfur. A two-bed catalytic Claus plant can achieve 94–96 percent efficiency. A scrubber at the end of the recovery unit can capture the remaining sulfur compounds (EPA, 1993b).

Flares

Petroleum refineries use flares to combust vapors rather than discharging them directly to the atmosphere. The flares produce $PM_{2.5}$, SO_2 and NO_x emissions. They are used for non-steady-state operations, such as maintenance, startup, shutdowns and upsets. Some of these operations are predictable, while some are not. Flares provide an outlet for gases that might otherwise result in fires or explosions and are therefore a critical component of the petroleum refining process. Flares also convert noxious and odorous gases to less hazardous gases. Frequent flaring in routine, nonemergency situations or to bypass pollution control systems can produce excess emissions and may violate the Clean Air Act.

In particular, the combustion of acid gases—a highly concentrated waste stream of hydrogen sulfide gas and sour water stripper gas—can produce high rates of SO_2 emissions. Typically, a refinery relies on a sulfur recovery plant to process hydrogen sulfide gas and sour water stripper gas to produce elemental sulfur for sale. According to EPA, a single day of acid gas flaring can generate more SO_2 than would be generated by an entire year of permitted sulfur recovery activity (EPA, 2000a). Improved operational control systems, improved training of operators, proper sizing of the sulfur recovery unit to handle all of the acid gases produced by the refinery, and optimal performance of the sulfur recovery unit can all prevent the flaring of acid gases (EPA, 2000a). In one case, a refinery had been sending its acid gases to an adjacent chemical plant, resulting in the need to incinerate the gases when the chemical plant was unable to process them. Under a consent decree, the owner of the refinery agreed to install a sulfur recovery plant at the refinery to eliminate the problem (Sunoco, 2005). A refinery in Oregon, Ohio was able to reduce its annual SO_2 emissions from about 180 tons to 49 tons through equipment and operational changes that eliminated conditions that had previously required higher rates of flaring (EPA, 2000a).

Refineries can also install flare gas recovery systems to collect and treat refinery gases, reducing the need for flaring while at the same time recovering the gas for use in refinery furnaces and boilers. Several such installations already exist in New Jersey, and California's Bay Area Air Quality Management District (AQMD) has just issued a rule encouraging these systems (Bay Area AQMD, 2005a).

Regulatory Authority

As is the case for other stationary sources, state and local agencies share responsibility for regulating petroleum refineries with the federal government. The federal government establishes two sets of emissions standards that apply to refineries: NSPS for PM and SO_2 emissions, and Maximum Achievable Control Technology (MACT) standards for hazardous air pollutants (HAPs).

Reasonably Available Control Technology

State Implementation Plans (SIPs) for areas that are in non-attainment of federal ambient air quality standards must include Reasonably Available Control Technology (RACT) requirements for new and existing sources. RACT standards are non-binding federal guidelines; for VOCs, the guidelines are known as Control Technique Guidelines (CTGs). Guidelines known as Alternative Control Techniques (ACTs) outline the control technologies that are available to address NO_x emissions from specific source sectors, but do not specify actual RACT limits. There are no sector-specific ACTs or CTGs for SO_2, PM_{10} or $PM_{2.5}$.

ACT guidelines applicable to NO_x emissions from equipment used at petroleum refineries (e.g., boilers and process heaters) can be found at www.epa.gov/ttn/catc/products.html#aptecrpts.

Table 11.5

NSPS for Petroleum Refineries (40 CFR §§60.100, *et seq.*, (subpart J))[a]

Source	Applicable Date	Controls	PM	Opacity	SO_2[b]
Fluid catalytic cracking unit catalyst regenerator	After June 11, 1973	With or without controls	2.0 lb/ton of coke burn-off[c]	30%[d]	NA
		With control device	NA	NA	90% reduction or 50 ppm, whichever is less stringent
		Without control device	NA	NA	20 lb/ton of coke burn-off[e]
Claus sulfur recovery plant	After October 4, 1976	Reduction or oxidation control system followed by incineration	NA	NA	250 ppm[f]
		Reduction control system not followed by incineration	NA	NA	300 ppm[g,h]

a. Any fluid catalytic cracking unit in which a contact material reacts with petroleum derivatives to improve feedstock quality and in which the contact material is regenerated by burning off coke and/or other deposits, and that commenced construction, reconstruction or modification on or before January 17, 1984 is exempt from these standards.
b. Any fluid catalytic cracking unit catalyst regenerator that commenced construction, reconstruction or modification on or before January 17, 1984 is exempted from the SO_2 standards.
c. In addition, if the gases discharged by the unit pass through an incinerator or waste heat boiler in which auxiliary or supplemental fossil fuel is burned, PM emissions may not exceed 0.10 lb/MMBtu of heat input attributable to such fossil fuel.
d. Exception: opacity may exceed 30% for one 6-minute average during any 1-hour period.
e. Another option is to use fresh feed (any petroleum derivative feedstock stream charged directly into the riser or reactor of a fluid catalytic cracking unit, except for petroleum derivatives recycled within the unit, fractionator or gas recovery unit) in the fluid catalytic cracking unit that has a total sulfur content no greater than 0.30% by weight.
f. At 0% excess air.
g. Additional limit of 10 ppm hydrogen sulfide.
h. The emissions limit is 300 ppmv of reduced sulfur compounds, calculated as ppmv SO_2 (dry basis) at 0% excess air.

New Source Performance Standards

The NSPS typically apply to new, modified and reconstructed sources in specific source categories in both attainment and nonattainment areas.

The NSPS for petroleum refineries are outlined in 40 CFR §§60.100, *et seq.* (subpart J). See Table 11.5 for refinery NSPS standards for PM and SO_2, as well as for applicable opacity limits. The standards apply to fluid catalytic cracking unit catalyst regenerators and fuel gas combustion devices that commenced construction, reconstruction or modification after June 11, 1973. They also apply to any Claus sulfur recovery plant of at least 20 long tons (22.4 short tons) per day capacity that commenced construction, reconstruction or modification after October 4, 1976.

In addition to the limits for PM and SO_2 set forth in Table 11.5, the NSPS also require refineries to limit the concentration of hydrogen sulfide from any fuel gas combustion device (including during flaring operations) constructed after June 11, 1973. Specifically, hydrogen sulfide concentrations must be below 230 milligrams per dry standard cubic meter (0.10 grains per million dry standard cubic feet). Flaring hydrogen sulfide can produce high ambient concentrations of SO_2. This standard does not apply if there is a process upset or emergency (40 CFR §§60.100, *et seq.* (subpart J)).

On July 22, 2005, EPA, Our Children's Earth Foundation and the Sierra Club filed a proposed consent decree requiring EPA to review the NSPS for SO_2 and PM and to finalize any revisions, if needed, within 30 months (EPA, 2005c). The NSPS for petroleum refineries have not been updated since 1974.

Power boilers at refineries, which are used to generate electricity and steam, are subject to the NSPS for industrial boilers (40 CFR §§60.40b, *et seq.* (subpart Db); and 40 CFR §§60.40c, *et seq.* (subpart Dc)) discussed in Chapter 6, *Industrial and Commercial Boilers*. Newer boilers are subject to limits for PM, SO_2, and NO_x emissions under these rules.

Regional Haze/Best Available Retrofit Technology

Under the regional haze program, Best Available Retrofit Technology (BART) requirements apply to emissions of $PM_{2.5}$ and $PM_{2.5}$-precursors from large sources that fall into 26 identified source categories and that began operation between 1962 and 1997, in both attainment and nonattainment areas. Petroleum refineries are among the identified source categories. However, states have not yet developed their BART rules.

New Source Review/Prevention of Significant Deterioration

New Source Review (NSR) and Prevention of Significant Deterioration (PSD) requirements apply to new and modified large stationary sources in nonattainment and attainment areas, respectively. Under these programs, plants in nonattainment areas are subject to Lowest Achievable Emission Rate (LAER) standards, while plants in attainment areas are subject to Best Available Control Technology (BACT) standards. Neither BACT nor LAER standards are codified, because they are intended to reflect technology advancements as they occur.

The Arizona Department of Environmental Quality has recently issued an air permit for the proposed Arizona Clean Fuels refinery (Arizona DEQ, 2005). The permit provides an example of the controls to be installed at a new refinery:

- The facility's two steam boilers will be fired with natural gas and will be equipped with low-NO_x burners and FGR.
- The facility's 18 process heaters will be equipped with low-NO_x burners; nine will also be equipped with SCR controls.
- Catalyst regenerators at the catalytic reforming unit will be equipped with a wet scrubber.
- The permit prohibits the use of flares as pollution control devices for intermittent or routine, non-emergency hydrocarbon releases.
- The permit prohibits the combustion of fuel oil in the refinery's boilers and heaters. Natural gas and fuel gases generated within the refinery are the only fuels allowed.
- The permit requires the refinery to meet several equipment design standards and work practice requirements in order to minimize SO_2 emissions during upsets and malfunctions of the sulfur recovery process.

Maximum Achievable Control Technology

MACT standards address emissions of HAPs from new and existing sources in identified industry categories, in both attainment and nonattainment areas. MACT standards apply to sources with potential emissions of 10 tons per year of a single HAP or 25 tons per year of combined HAPs.

Petroleum refineries are subject to the MACT standards at 40 CFR §§63.1560, *et seq.* (subpart UUU); in particular, these standards address catalytic cracking units, catalytic reforming units and sulfur recovery units. Units are considered new if they commenced construction, reconstruction or modification after September 11, 1998; all other units are considered existing.

The MACT standards regulate PM and SO_2 as surrogates for HAPs.[3] Specifically, PM is used as a surrogate for metal HAPs in catalytic cracking units and SO_2 is used as a surrogate for sulfur HAP emissions from sulfur recovery units.

The MACT standards mandate the same PM limits as the NSPS for catalytic cracking units and sulfur recovery units subject to the NSPS (see description above). Catalytic cracking units not subject to the NSPS have four compliance options:

1. meet the NSPS standards for PM;
2. meet a PM limit of 2.0 pounds per ton (lb/ton) of coke burn-off;[4]
3. meet a nickel emissions limit of 13,000 milligrams per hour (0.029 pounds per hour (lb/hr)); or
4. meet a nickel emissions limit of 1.0 milligram per kilogram (0.002 lb/ton) of coke burn-off in the catalyst regenerator.

Sulfur recovery units not subject to the NSPS have two options: (1) meet the NSPS standards; or (2) meet a total reduced sulfur emissions limit of 300 ppmv of total reduced sulfur compounds, expressed as an equivalent SO_2 concentration (dry basis) at 0 percent oxygen.

Power boilers and process heaters at refineries are subject to the Industrial/Commercial/Institutional Boilers and Process Heaters National Emission Standards for Hazardous Air Pollutants (40 CFR 63 Subpart DDDDD),

3. Note that 40 CFR §§63.640, *et seq.* (subpart CC) include additional HAP limits for petroleum refineries that do not use PM or SO_2 as surrogates.

4. While the limit of 2.0 lb/ton is the same as the NSPS, these units are *not* subject to the NSPS requirement if the gases discharged by the unit pass through an incinerator or waste heat boiler in which auxiliary or supplemental fossil fuel is burned; in that case, PM emissions may not exceed 0.10 lb/MMBtu of heat input attributable to such fossil fuel.

discussed in Chapter 6, *Industrial and Commercial Boilers*. Larger liquid-fueled boilers and process heaters are subject to PM, hydrogen chloride and CO limits. Larger gaseous-fueled units are subject to a CO limit. Different limits generally apply to new and existing units. A new or reconstructed unit is one that commenced construction or reconstruction on or later than January 13, 2003.

Petroleum Refinery Enforcement Initiative

In addition to the regulatory requirements described above, EPA in 1998 initiated an enforcement program known as the Petroleum Refinery Initiative to address alleged Clean Air Act violations at petroleum refineries (EPA, 2005b). The program addresses four areas:

- PSD/NSR;
- NSPS;
- Leak Detection and Repair (LDAR) requirements; and
- Benzene National Emissions Standards for Hazardous Air Pollutants.

EPA has entered into consent decrees with 83 refineries (17 companies), comprising 77 percent of the nation's refinery capacity.[5] The consent decrees already lodged will reduce annual refinery SO_2 emissions by 237,000 tons per year and NO_x emissions by 81,000 tons per year, as well as an undetermined amount of PM and other pollutants. Refineries have agreed to spend over $4 billion on new control technologies, pay over $60 million in civil penalties and spend over $65 million in environmental projects in 25 states (EPA, 2005b, 2005d).

We provide details on the ConocoPhillips consent decree as an example of the control measures that the government is requiring in its petroleum refinery cases. In its January 27, 2005 consent decree with EPA, ConocoPhillips agreed to make substantial reductions of PM, SO_2 and NO_x emissions. Specifically, the company must install an SCR system or a Scrubber-Based NO_x Emission Reduction Technology (SNERT) on various units in order to achieve a NO_x concentration of between 20 and 40 ppmv (the actual limit depends on the type of unit) on a 365-day rolling average basis; and 40 ppmv on a seven-day rolling average basis, at 0 percent oxygen. Other units are required to use NO_x-reducing catalyst additives and low-NO_x combustion promoters. Furthermore, combustion units other than internal engines must install SCR, SNCR, new ultra-low-NO_x burners or other technologies that the company demonstrates will reduce NO_x emissions to 0.040 lb/MMBtu or lower. Internal combustion engines must install combustion controls to reduce NO_x emissions by 80 percent or more compared to an uncontrolled internal combustion engine (*U.S. v. ConocoPhillips*, 2005).

For SO_2 control, the agreement requires six fluidized catalytic cracking units (FCCUs) to install wet gas scrubbers and meet an SO_2 concentration limit of 25 ppmv or less on a 365-day average basis; and 50 ppmv or lower on a seven-day rolling average basis, at 0 percent oxygen. For PM control, five FCCUs will install wet gas scrubbers to achieve an emissions limit of 1 lb/ton of coke burned on a three-hour average basis (*U.S. v. ConocoPhillips*, 2005).

Finally, the company must install continuous emissions monitoring systems to demonstrate compliance, and some facilities must also install continuous opacity monitoring systems (*U.S. v. ConocoPhillips*, 2005).

State and Local Policy Measures

As discussed in Chapter 4, *The Clean Air Act*, state and local authority varies with regard to the imposition of more stringent regulations than the Clean Air Act and federal regulations require. Some states and local agencies have no limitations in this regard, while the limitations in other states vary. For the purposes of this discussion, we will assume that SIP writers are familiar with applicable state and local laws and regulations.

For states that can impose more stringent requirements than those promulgated under federal law, several options are available.

For example, the South Coast AQMD Rule 1105.1 limits PM_{10}[6] and ammonia emissions from fluid catalytic cracking units at oil refineries. Refinery operators can comply with one of three PM_{10} emissions limits: 3.6 lb/hr, 0.005 grain per dry standard cubic foot of flue gas corrected to 3 percent oxygen dry, or 2.8 lb per thousand barrels of fresh feed. Oil refineries are expected to implement several control technologies to meet the emissions limits, including ESPs, SO_x reducing agents and wet gas scrubbers. The South Coast AQMD is also developing a rule to limit SO_2 emissions from flaring operations. The rule builds on a pre-existing requirement (Rule 1118) that required eight refineries and two other facilities to collect emissions data from their vent gas flaring operations (South Coast AQMD, 2004a).

The Bay Area AQMD adopted a rule on July 20, 2005 limiting emissions from flaring operations. The rule builds on a June 2003 rule requiring refineries to monitor and report flare emissions data. It requires that each refinery prepare a Flare Minimization Plan that determines how best to minimize flaring. The Bay Area AQMD must

5. The consent decrees are available at www.epa.gov/compliance/resources/cases/civil/caa/oil/index.html.

6. The South Coast rule explicitly limits filterable PM_{10} emissions. Filterable PM consists of directly emitted particles captured on the filter of a stack test train and can include $PM_{2.5}$.

approve the plans, which must be updated annually to incorporate the latest technologies and practices (Bay Area AQMD, 2005a).

The Bay Area AQMD also adopted a rule (Rule 9-10) on July 1, 1997 requiring refineries to limit NO_x emissions from boilers, steam generators and process heaters to a refinery-wide NO_x standard of 0.033 lb/MMBtu. Smaller units are exempt from the rule (Bay Area AQMD, 2002b). Seebold provides a case study of how one San Francisco Bay refinery achieved the standard (2001). With the application of new burner technology, the facility was able to reduce NO_x emissions from its process heater to less than 20 ppm. No back-end controls were required. Also, the facility was able to reduce NO_x emissions from its power boilers to less than 25 ppm without the use of back-end controls.

Although VOCs have not been the focus of this discussion, the Bay Area AQMD has also adopted a stringent rule (Rule 8-18) to control VOC emissions from refinery valves, flanges, pumps, compressors, pressure relief valves, pipes, connections and other refinery components. The rule was initially developed in 1980, strengthened significantly in 1992, and amended at various points thereafter, most recently in 2004. The 2004 version of the rule requires that refineries develop an LDAR program to locate, minimize and repair leaks from this equipment. The rule has the most stringent leak requirements in California (and the nation), and has significantly reduced fugitive emissions from the area's refinery facilities (Bay Area AQMD, 2004).

State permitting authorities may also find it useful in the context of permitting determinations to consider the emissions limits and control options required by the many petroleum sector settlement agreements. For example, several of the agreements require SCR controls on fluid catalytic cracking units.

In general, flexible compliance mechanisms for facilities like refineries can significantly reduce compliance costs, allowing better use of limited resources. Refineries are complex operations with many different kinds of emissions sources. Boilers and process heaters, for example, will vary in terms of their age, size, capacity, utilization and space limitations. No single control technology or combination of controls will be appropriate in all situations. Recognizing this, the Houston-Galveston region established a NO_x cap-and-trade program in 2000 that included the region's refineries (TCEQ, 2000, 2002). In response to the flexibility afforded by the cap-and-trade approach, refinery operators implemented a wide range of strategies aimed at reducing NO_x emissions. This included the retrofit of large gas turbines with SCR systems, the decommissioning of smaller, lower efficiency steam boilers, and the conversion of generating units to CHP systems (Deason, 2005). This diverse group of control strategies also appears in the agreements that the Texas Commission on Environmental Quality negotiated with several refineries in the Beaumont/Port Arthur ozone nonattainment area. The agreements require the installation of scrubbers, the replacement of burners and the shut down of certain boilers (TCEQ 2004a, 2004b, 2004c).

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Chapter 12

Diesel Engine Technologies

Introduction

Diesel engine technology has changed dramatically in the last 20 years. EPA regulations have reduced the nitrogen oxides (NO_x) and fine particulate matter ($PM_{2.5}$) emissions from new onroad diesel truck and bus engines by 80 percent[1] and 90 percent, respectively, since 1988.

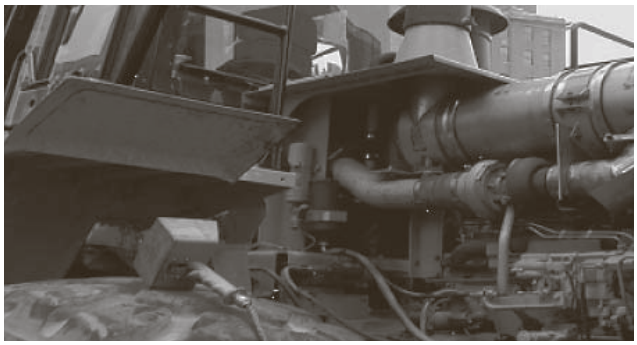
To meet increasingly stringent emissions standards, engine manufacturers have made fundamental changes to engines and added exhaust "after-treatment" devices. Stricter standards that take effect between 2007 and 2010 will reduce allowable NO_x and $PM_{2.5}$ emissions by a further 90 percent and are driving additional changes—primarily the use of even more effective after-treatment technologies.

Regulations for the diesel engines used in nonroad equipment such as ships, trains and agricultural and construction tractors have lagged—emissions standards for these engines will not achieve regulatory parity with those for onroad engines until 2015 or later. However, because many of these engines are very similar to onroad engines, the same technologies being developed for the onroad market can be used in many cases to reduce their emissions in advance of regulatory mandates.

While great progress has been achieved in reducing emissions from new diesel engines, there is a significant legacy fleet of older diesel engines still in use. Many of these engines were made before diesel emissions standards went into effect. Diesel trucks and buses can stay in service for 20 years, while some nonroad equipment can last for more than 40 years.

There are five ways that exhaust emissions from existing diesel vehicle fleets can be reduced—most of which take advantage of the progress that has been made in the development of new diesel engines. These approaches include:

- **Replace or Repower**: Retire vehicles or engines "early" and replace with new, cleaner engines.
- **Retrofit**: Add an "after-treatment" device to the tail pipe and/or modify an existing engine to make it



1. Due to the effect of engine "defeat devices," actual NO_x reductions for long-haul Class 8 onroad trucks have been less than what would be implied by the change in standards for engines built prior to 2002 (see Chapter 13, *Diesel Trucks and Buses*).

cleaner.

- **Repair/Rebuild**: Perform proper maintenance to ensure that all engines are as clean as possible given their technology level.
- **Reduce Idling**: Reduce unproductive engine idling, thereby reducing emissions and saving fuel.
- **Refuel**: Use a "cleaner" alternative diesel fuel.

This chapter will discuss each of these approaches. For retrofits, alternative diesel fuels and idle reduction, we have included a brief description of the technologies that can be used, their costs, implementation issues and emissions benefits.

The information presented here merits its own chapter because it is relevant to the emissions reduction strategies discussed in several other chapters, all of which rely on the application of technology to reduce emissions from in-use diesel engines.

Engine/Vehicle Retirement

For many fleets of older diesel vehicles, particularly those built prior to 1988, dramatic reductions in both $PM_{2.5}$ and NO_x emissions can be achieved by retiring older vehicles and replacing them with new vehicles equipped with new engines that meet much stricter emissions standards. In many cases, these new engines will also achieve better fuel economy, providing some economic benefit to the owner.

Another approach is to keep the vehicle but retire the engine—by repowering with a new engine. While some fleet operators have taken this approach with onroad vehicles,[2] it is much more likely to be cost-effective for nonroad fleets, including some marine vessels, locomotives, and large specialty construction equipment. This type of equipment typically has a much longer lifespan than onroad trucks and buses; in addition, the engine generally represents a smaller percentage of the value of the entire vehicle.

For some nonroad equipment, it may be possible to repower with an onroad engine certified to lower emissions standards than new nonroad engines and thereby produce even greater emissions reductions (see Chapter 14, *Nonroad Equipment*).

Engine Modifications

Some of the changes that have been made to onroad diesel engines over the last 20 years include the change from a two-stroke to four-stroke cycle, improved cylinder designs, improved fuel injectors and higher injection pressures, electronic fuel control, the addition of turbochargers, and the addition of exhaust gas recirculation. These changes have contributed to both improved fuel economy and lower emissions. To a lesser extent, these technology advances have also been applied to nonroad engines.

Diesel engines, particularly the very large nonroad engines used in marine vessels, locomotives and some construction equipment, typically go through one or two major overhauls during their lifetime. Some of the changes that have been applied to new engine designs can be applied as an upgrade to existing engines during an overhaul, resulting in lower emissions. As with repowering, this approach is generally more practical and cost-effective for the largest and longest-lived nonroad engines.

In general, diesel engine design involves a fundamental trade-off between NO_x and PM formation in the cylinder. Relatively simple changes to the fuel injection timing of existing diesel engines can reduce NO_x emissions by 20–40 percent, but at the expense of an unacceptable increase in PM emissions. At least one manufacturer has taken the approach of combining engine timing changes with retrofit of a diesel oxidation catalyst or diesel particulate filter to achieve simultaneous $PM_{2.5}$ and NO_x reductions.

Repair/Rebuild

Engines that are properly maintained and tuned perform better and typically emit less pollution than poorly maintained engines, regardless of how old they are. Proper maintenance includes the periodic overhaul or rebuilding of the entire engine to evaluate and replace as necessary major components that can wear out, such as cylinder liners. Proper maintenance and rebuilding of severely worn engines may also increase fuel economy and extend engine life.

Any maintenance problem that restricts air flow into the engine or results in over-fueling is likely to increase PM emissions from a diesel engine. Increased emissions can result from something as simple as a dirty air filter. Other maintenance problems that can increase emissions include clogged or worn fuel injectors, worn fuel injection pumps, a malfunctioning turbocharger, worn cylinder liners, improper engine timing and leaks in the air intake manifold.

Retrofit Technologies

"After-treatment" refers to a device or technology installed in a vehicle's exhaust system to reduce emissions. Unlike in-engine and fuel technologies, these devices do not reduce the emissions produced by the engine; rather, they act to cleanse the exhaust after it has left the engine, but

2. Between 2000 and 2001, MTA New York City Transit repowered 600 model year 1990 and 1993 urban transit buses with new engines.

Table 12.1

Emissions Reduction Potential (% reduction)

Technology	NO_x	HC & CO	$PM_{2.5}$	Notes
Diesel oxidation catalyst (DOC)	no reduction	20%–50%	10%–30%	$PM_{2.5}$ reductions are primarily the "wet" liquid HC portion of $PM_{2.5}$.
Passive diesel particulate filter (DPF)	no reduction	60%–90%	80%–90%+	$PM_{2.5}$ reductions of both HC and solid carbon particles. While DPFs do not affect the total mass of NO_x emissions, they usually increase the ratio of NO_2 to NO directly emitted, from less than 10%-30% or more.
Active diesel particulate filter (ADPF)	no reduction	60%–93%	80%–90%+	$PM_{2.5}$ reductions of both HC and solid carbon particles. While DPFs do not affect the total mass of NO_x emissions, they usually increase the ratio of NO_2 to NO directly emitted, from less than 10%-30% or more.
Flow through filter (FTF)	no reduction	50%–89%	50%–76%	Eliminates more $PM_{2.5}$ carbon particles than a typical DOC.
Closed crankcase filter system (CCFS)	no reduction	no reduction	5%–10%	Additional PM reduction when combined with other PM after-treatment devices.
Exhaust gas recirculation (EGR)	30%–40%	no reduction	no reduction	NO_x reduction for both low- and high-pressure systems. EGR can achieve an 80% reduction in HC, CO and $PM_{2.5}$ when combined with a DPF.
Selective catalytic reduction (SCR)	60%	no reduction	no reduction	SCR can achieve a 25% reduction in HC, CO and $PM_{2.5}$ when combined with a DOC; higher reductions are possible when combined with a DPF.
NO_x reducing catalyst (NRC)	20%–30%	no reduction	no reduction	NRC can achieve a 25% reduction in HC, CO and $PM_{2.5}$ when combined with a DOC; higher reductions are possible when combined with a DPF.
NO_x adsorber	90%+	10%–90%	10%–90%	
Emulsified diesel (ED) fuel	5%–30%	increase (up to 35%)	20%–50%	
Biodiesel	increase (up to 10%)	0%–50%	0%–50%	
Oxygenated diesel (OD)	increase (0%–10%)	0%–50%	0%–50%	Ethanol is a renewable fuel that results in net fuel-cycle CO_2 reductions.
Fuel-borne catalysts	0%–10%	0%–50%	0%–50%	

Notes: For all technologies, the reported range of emissions reductions is from emissions test results under the EPA and California Air Resources Board (CARB) technology verification programs.

before it enters the atmosphere. With some exceptions and limitations, as noted below, after-treatment devices can be retrofit onto existing diesel vehicles to reduce their emissions.

There are a number of different approaches to diesel exhaust after-treatment, but most include a catalyst that promotes chemical reactions in the exhaust, oxidizing the hydrocarbons (HC) and $PM_{2.5}$ to carbon dioxide (CO_2) and water, or reducing NO_x to elemental nitrogen. In general, any after-treatment technology can either oxidize $PM_{2.5}$ or reduce NO_x, but not both at the same time. Different technologies can be combined in a single commercially available product, however, resulting in both lower $PM_{2.5}$ and lower NO_x emissions simultaneously. The nine most common diesel retrofit technologies are described in detail below; they include both PM oxidation technologies and NO_x reduction technologies. These technologies are all potentially applicable to a wide variety of diesel engines. Some are commercially available now for a significant number of applications, others for a smaller number of specific engines.

$PM_{2.5}$ Technologies

Diesel Oxidation Catalyst. A diesel oxidation catalyst (DOC) is a flow-through metal or ceramic substrate coated with a precious metal catalyst (e.g., platinum) and packaged into a metal container similar to an exhaust muffler/resonator. The DOC sits in the exhaust stream of a vehicle and promotes the oxidation of unburned PM, HC and carbon monoxide (CO) in the exhaust passing through it, producing CO_2 and water.

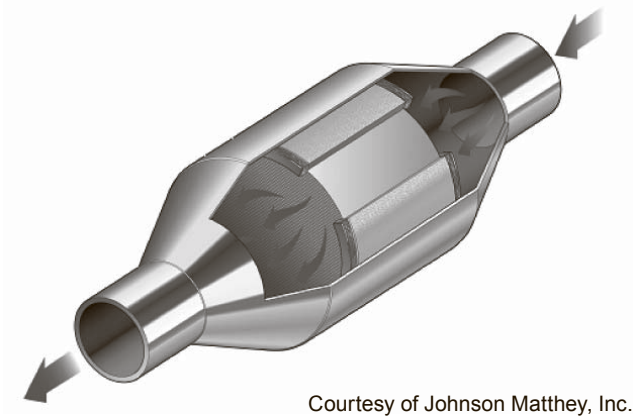
The following implementation issues[3] and costs[4] apply when installing and operating DOCs:

- If properly sized, DOCs can be used on virtually any diesel engine, with an appropriate catalyst formulation.
- DOCs may not be appropriate for very old two-stroke engines with high oil consumption, as excess PM can plug the flow channels in the DOC.
- Retrofit installation is generally straightforward and takes only a few hours.
- Fuel should not have more than 500 parts per million (ppm) sulfur. With higher sulfur fuels, excessive sulfate $PM_{2.5}$ is created across the catalyst.

3. For all technologies, information on implementation issues is taken from technology summaries on the EPA/OTAQ website, discussions with manufacturers and the author's project experience.

4. Typical costs for all technologies are from technology summaries available on the EPA/OTAQ website and from the Manufacturers of Emissions Controls Association (MECA) independent cost survey noted in the reference section.

Typical Diesel Oxidation Catalyst



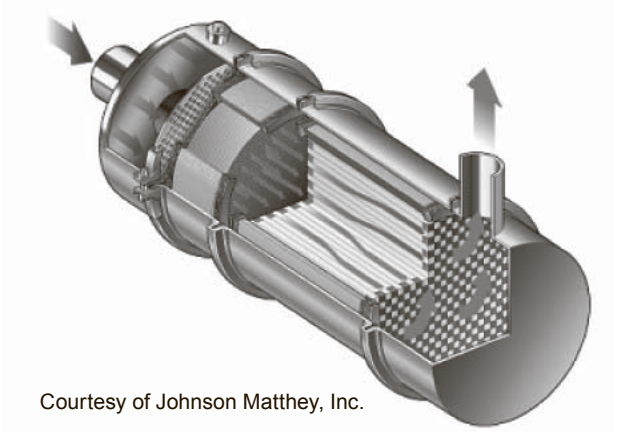
Courtesy of Johnson Matthey, Inc.

- Purchase cost is $1,000–$2,000 for a typical truck or bus engine.
- Larger engines generally require larger, more expensive devices.
- DOCs have minimal effect on fuel economy.

Passive Diesel Particulate Filter. A passive diesel particulate filter (DPF) combines a DOC with a porous ceramic, metal mesh or silicon carbide filter in a metal container similar to an exhaust muffler/resonator. There are several variations on the design: some DPFs have a separate flow-through catalyst section in series with an un-catalyzed filter, while others use a filter with the catalyst applied directly to it. The DPF sits in the exhaust stream of the vehicle like a typical muffler/resonator.

Inside the device, the gaseous components of the exhaust pass through the porous walls of the filter, while the solid $PM_{2.5}$ particles are physically trapped on and in the filter walls. Carbon will typically oxidize only at temperatures greater than 600 degrees Centigrade (°C); the catalyst promotes oxidation of the trapped $PM_{2.5}$ (carbon and

CRT™ Diesel Particulate Filter



Courtesy of Johnson Matthey, Inc.

hydrocarbons) at the temperatures more typical of diesel exhaust (200–400°C), which then exits the filter as gaseous CO_2 and H_2O. The catalyst also oxidizes gaseous HC and CO in the exhaust like a typical DOC.

The following implementation issues and costs apply when installing and operating DPFs:

- DPFs put a restriction in the exhaust and increase "back pressure" on the engine pistons; they must be carefully designed to minimize this impact, since high back pressure can lead to progressive engine damage.
- Non-combustible components of lube oil collect as ash in the filter over time. Approximately once per year, DPFs must be removed from the vehicle and cleaned to remove this collected ash.
- As ash or excess carbon builds up in the filter, engine back pressure rises. DPFs should always be used with a back-pressure monitoring system that triggers a maintenance light once the back pressure rises above a set threshold.
- Fuel cannot have more than 50 ppm sulfur. Higher levels of sulfur reduce the oxidation efficiency of collected $PM_{2.5}$ and can result in filter plugging. As discussed in Chapters 13 and 14, onroad diesel fuel will be required to have less than 15 ppm sulfur beginning in late 2006. This ultra-low-sulfur fuel is available in some parts of the country today and can be used with DPFs. Nonroad diesel fuel with sulfur levels of 500 ppm or more will continue to be available through 2010. DPF retrofits will not be possible on nonroad equipment until after 2010 unless onroad fuel is used.
- DPFs will not work on all engines. The more PM the engine produces, the larger the filter and catalyst must be to work continuously without plugging. For some very old, very dirty engines it may not be practical to design a DPF that will work consistently, due to cost issues and space constraints.
- The DPF also requires a minimum exhaust temperature of 240–300°C for 35–40 percent of the time to oxidize the collected $PM_{2.5}$ in the filter. This is easily achievable for many vehicles, but certain engines and certain duty cycles may not have sufficient exhaust temperature to use a DPF effectively. In general, DPFs can be used for duty cycles in which the diesel engine operates for a majority of the time under high loads. Lightly loaded duty cycles may not be appropriate for DPFs. Evaluation of the temperature profile for the engine/duty cycle is highly recommended before retrofitting DPFs on a vehicle type for the first time.
- Installation is relatively straightforward and usually takes less than five hours per vehicle.
- Purchase cost is $5,000–$10,000 for a typical truck or bus engine.
- Larger engines generally require larger, more expensive devices.
- DPFs have minimal effect on fuel economy.

Active Diesel Particulate Filter. An active diesel particulate filter (ADPF) system uses the same porous filter as a passive DPF to remove $PM_{2.5}$ from diesel exhaust. Like a passive DPF, these systems may also employ a catalyst to lower the temperature at which the collected $PM_{2.5}$ will oxidize out of the filter. However, in order to accommodate a wider range of duty cycles, they also incorporate some kind of active system to raise the temperature inside the filter.

The most common method used to raise the temperature in the filter is to inject additional diesel fuel into the exhaust stream across a small catalyst, downstream of the engine but in front of the filter. Oxidation of this fuel raises the temperature.

It is expected that virtually all new onroad diesel trucks and buses will use ADPF systems beginning in 2007 to comply with the stricter EPA PM standards that will take effect at that time. While there are currently very few ADPF systems commercially available for retrofit, it is expected that more will become available after 2007.

The following implementation issues and costs apply when installing and operating ADPFs:

- ADPFs increase back pressure on the engine pistons and must be carefully designed to minimize this impact, since high back pressure can lead to progressive engine damage.
- Non-combustible components of lube oil collect as ash in the filter over time. Approximately once per year, ADPFs must be removed from the vehicle and cleaned to remove this collected ash.
- ADPFs will work on a wider range of engines and duty cycles than passive DPFs because they are not limited by the inherent exhaust temperature profile.
- ADPF systems are significantly more complicated than passive DPFs. In addition to the filter element, an active DPF system will typically contain a fuel pump, a fuel injector into the exhaust, a separate catalyst, back-pressure and temperature monitors and an Electronic Control Module (ECM).
- Depending on whether or not a catalyst is used to help oxidize collected PM out of the filter, ADPF systems may be usable with fuel that has up to

500 ppm sulfur. Fully active systems that raise the exhaust temperature above 600°C to oxidize collected carbon can operate on the higher sulfur fuels. However, systems that incorporate a catalyst will be limited to operation on fuel with less than 50 ppm sulfur in the same way as DPFs. Most systems designed for use with onroad vehicles are expected to include a catalyst and will require the lower sulfur fuels that are due to become readily available for onroad vehicles beginning in late 2006.

- Purchase cost for early retrofit devices is expected to be comparable to current DPF prices: $5,000–$10,000 for a typical truck or bus engine. As manufacturing volume increases to accommodate the installation of ADPFs on all new trucks after 2007, this price is expected to fall significantly.
- Larger engines generally require larger, more expensive devices.
- ADPFs incur a fuel economy penalty of 4–7 percent for operation of the active system.

Flow Through Filter. A flow through filter (FTF) is very similar to a DOC, but it uses a different type of substrate to hold the catalyst material. Different manufacturers use wire mesh, wire fleece or sintered metal, all coated with a precious metal catalyst and packaged into metal containers similar to those used to package a DOC.

As in a DOC, the catalyst promotes the oxidation of unburned carbon, HC and CO in the exhaust, producing CO_2 and water. Because of the substrate formation, individual $PM_{2.5}$ particles typically have greater opportunity to contact a catalyst site than in a typical DOC so that an FTF eliminates a greater percentage of $PM_{2.5}$ than a typical DOC.

The following implementation issues and costs apply when installing and operating FTFs:

- FTFs, if properly sized and made with an appropriate catalyst formulation, can be used on a greater range of diesel engines than passive DPFs.
- Like passive DPFs, FTFs require a minimum exhaust temperature to work consistently and therefore cannot be used in all duty cycles.
- Retrofit installation is generally straightforward and takes only a few hours.
- Fuel should not have more than 500 ppm sulfur. With higher sulfur fuels, excessive sulfate $PM_{2.5}$ is created across the catalyst.
- Purchase cost is $3,000–$4,000 for a typical truck or bus engine.
- Larger engines generally require larger, more expensive devices.
- FTFs have minimal effect on fuel economy.

Closed Crankcase Filter System. Most current diesel engines have "open" crankcases, which are vented to the atmosphere to relieve any pressure that builds up as combustion gases leak from the cylinders into the crank case. While the flow from these vents is generally much smaller than exhaust flow from the tail pipe, crank case vent emissions include combustion gases, unburned fuel, various HC vapors, diesel PM and engine oil droplets.

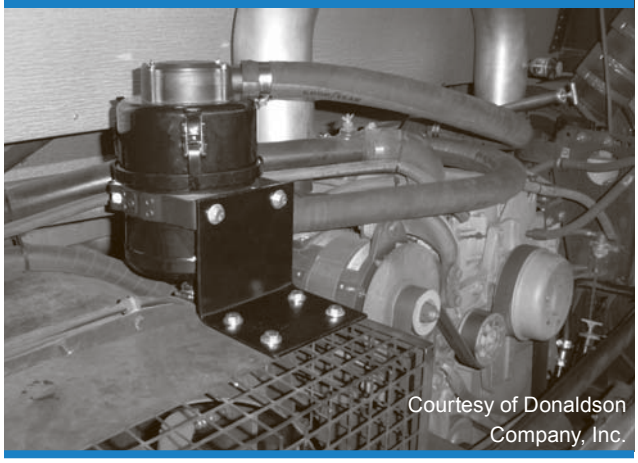
Under current EPA standards and test procedures, these crank case vent gases are not included in exhaust emissions limits. Nonetheless, they can be significant. Testing conducted under the EPA Verification Program (discussed below) on a 1998 onroad diesel engine showed that PM emissions from the crank case vent totaled 16–23 percent of the PM emissions coming from the tail pipe (RTI, 2003).

A closed crankcase system routes gases from the crankcase

Particulate Reactor™ Flow Through Filter

Courtesy of Fleetguard Emission Solutions

Closed Crankcase Filter Installed on a Bus



Courtesy of Donaldson Company, Inc.

vent to the engine's air intake manifold, effectively closing the crank case. These systems generally include a pressure-regulating valve and an oil filter to keep oil in the vent gases from fouling the engine's turbocharger.

The following implementation issues and costs apply when installing and operating a closed crankcase filter system:

- Closed crankcase systems can be installed on virtually any diesel engine.
- Retrofit installation is generally straightforward and takes only a few hours.
- Removing crankcase vent emissions generally results in a cleaner vehicle engine compartment.
- Purchase cost is $250–$500 for a typical truck or bus engine.
- Closed crankcase systems have no effect on fuel economy.

NO_x Technologies

Exhaust Gas Recirculation. In an exhaust gas recirculation (EGR) system, some of the exhaust gas from the diesel engine is rerouted back to the engine's air intake manifold to be mixed with the combustion air before it enters the cylinder. This introduces inert gas (CO_2) into the cylinder that dilutes the air charge, absorbs heat and reduces peak combustion temperature in the engine, thus reducing the production of NO_x during combustion. Technically, EGR is not an after-treatment technology because it fundamentally affects the engine's combustion cycle and reduces engine emissions.

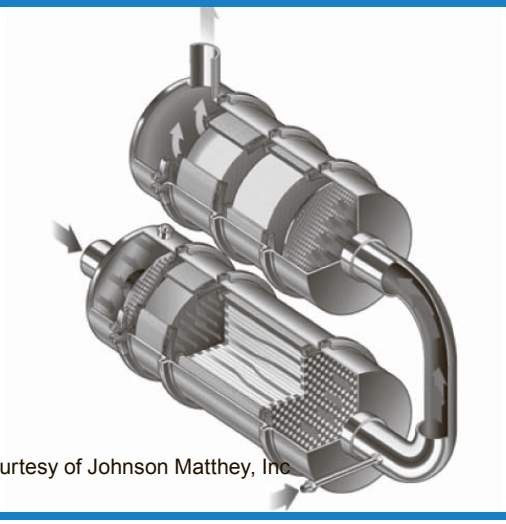
Since 2003, most new onroad diesel engines have included high-pressure cooled EGR systems that redirect exhaust flow to the air intake using an electronically controlled valve mounted just downstream of the engine turbocharger. Control of this exhaust valve in response to differing engine conditions is accomplished using the engine's ECM. These types of systems could theoretically be retrofit on electronically controlled engines.

Low-pressure EGR systems are also available that can be retrofit onto virtually any diesel engine. These systems typically redirect exhaust flow to the air intake using a valve mounted much further back in the exhaust stream, where the valve is controlled with an ECM separate from the engine ECM. These systems must be used in conjunction with a passive or active DPF, because the redirected air entering the air intake must have low levels of particulate.

The following implementation issues and costs apply when installing and operating an EGR system:

- Both low- and high-pressure EGR systems add equipment to the engine that may be difficult to fit in existing engine compartments and that may increase maintenance costs.
- Retrofit of a high-pressure system may also require an upgrade to the existing engine cooling system, since the EGR cooler can add up to 40 percent additional heating load to the engine cooling loop.
- High-pressure EGR systems force more PM into the engine oil, reducing effective oil life.
- Low-pressure EGR systems require engine mapping to develop effective control algorithms.
- EGR applied to a new engine adds a modest amount to the cost of the engine.
- Retrofits can cost $10,000–$15,000, including the cost of the required DPF.

SCRT™ SCR System



Courtesy of Johnson Matthey, Inc.

Longview® Lean NO_x Catalyst System

Courtesy of Cleaire and Fleetguard Emission Solutions

- EGR systems incur a fuel economy penalty of 2–4 percent.

Selective Catalytic Reduction. Selective Catalytic Reduction (SCR) technology uses a reducing agent in conjunction with a special SCR catalyst to reduce the NO_x in diesel exhaust to elemental nitrogen. The SCR catalyst sits in the exhaust stream, much like a DOC or DPF, and the reductant is injected into the exhaust ahead of the catalyst, much like the diesel fuel in an active DPF system.

The most common reductant used with SCR is a solution of urea in water, which is kept in a separate tank on the vehicle. Once injected, the urea becomes ammonia and a chemical reduction reaction between the ammonia and NO_x takes place across the SCR catalyst. The urea injection is managed by an ECM that carefully matches the volume of injected urea to the engine's exhaust flow.

Since some ammonia can slip through the SCR catalyst, most SCR systems also include a DOC or DPF downstream from the SCR catalyst to clean up the ammonia. The DOC or DPF will also reduce PM emissions.

The following implementation issues and costs apply when installing and operating an SCR system:

- SCR systems can be used on virtually any diesel engine.
- SCR systems add significant equipment to the engine system and increase ongoing maintenance costs. In addition to the SCR catalyst, an SCR system will typically require a urea tank, a urea pump and injector, back-pressure and temperature monitors and an ECM.
- SCR requires the use of urea reductant. While urea is produced in plentiful quantities as an industrial chemical and is relatively inexpensive ($0.80/gallon), the retail urea distribution infrastructure in place in the U.S. is currently limited. Fleet operators can contract for urea delivery to their own centralized fueling locations, but vehicle operators currently have very limited ability to purchase urea at existing retail fuel outlets.
- SCR systems typically use approximately one gallon of urea for every 20 gallons of diesel fuel.
- SCR systems require engine mapping and exhaust temperature measurement to develop effective control algorithms.
- Fuel should have no more than 500 ppm sulfur for operation of the SCR system itself. If used in conjunction with a DPF, fuel sulfur cannot be more than 50 ppm, as noted above.
- An SCR system typically costs $10,000–$20,000 for a truck or bus engine.
- As with the other retrofit technologies described above, the cost of SCR systems generally increases with engine horsepower.
- SCR systems have minimal effect on fuel economy

NO_x Reducing Catalyst. A NO_x reducing catalyst (NRC) system works in the same way as an SCR system, but it uses diesel fuel as a reductant instead of a urea solution. The NRC catalyst sits in the exhaust stream much like a DOC or DPF and the diesel fuel reductant is injected into the exhaust ahead of the catalyst. HC from the injected fuel contribute to a chemical reduction reaction with the NO_x in the exhaust, across the NRC catalyst.

Since diesel fuel is not as effective a reductant as urea, NRC systems reduce less than half as much NO_x as SCR systems.

Because of the potential for increased HC and CO emissions from the injected fuel, most NRC systems also include a DOC or DPF downstream from the NRC catalyst.

The following implementation issues and costs apply when installing and operating an NRC:

- NRC systems can be used on virtually any diesel engine.
- NRC systems add significant equipment to the engine system and increase ongoing maintenance costs. In addition to the NRC catalyst, an NRC system will typically contain a fuel pump, a fuel injector, back-pressure and temperature monitors (because exhaust temperature is critical in these applications) and an ECM.
- Fuel should have no more than 500 ppm sulfur, though ultra-low-sulfur diesel with less than 15 ppm sulfur is preferable, if available.
- Purchase cost of $6,500–$10,000 for a typical truck or bus engine.
- As with the other retrofit technologies described above, the cost of NRC systems generally increases with engine horsepower.
- NRC systems incur a fuel economy penalty of 4–7 percent because of the fuel used as a reductant.

NO_x Adsorber. Unlike SCR and NRC systems, which continually reduce NO_x to elemental nitrogen by combining the oxygen in NO_x with HC (diesel fuel or ammonia), the materials in a NO_x adsorber chemically

combine with NO_x under lean (high oxygen content) conditions typical of diesel engine exhaust. In this way, the NO_x adsorber chemically collects nitrogen over time. Eventually, the nitrogen must be released in order to make room to collect more. This is done by creating rich (low oxygen, high HC) conditions in the exhaust. Since rich operation is not typical of diesel engine operation, this is done by injecting fuel into the exhaust (similar to an NRC system).

NO_x adsorbers are typically set up as a dual bed system with a series of dampers to minimize the fuel penalty.

NO_x adsorber technology is expected to take a leading role in complying with the more stringent EPA onroad NO_x standards that take effect between 2007 and 2010. EPA believes that NO_x adsorbers are applicable to nonroad engines and that they will also be used to comply with the Tier 4 nonroad NO_x standards that will take effect between 2008 and 2015.

Due to the high level of engine integration necessary to make adsorber systems work, it is unclear at this time whether NO_x adsorber technology will be available as a retrofit option and if so, when.

The following implementation issues and costs apply when installing and operating a NO_x adsorber system:

- NO_x adsorber systems can theoretically be used on virtually any diesel engine.
- NO_x adsorber systems add significant equipment to the engine system, increasing on-going maintenance costs. In addition to the NO_x adsorber beds themselves, a NO_x adsorber system typically contains a fuel pump, a fuel injector, back-pressure and temperature monitors and an ECM. It may also contain dampers to control flow to multiple adsorber beds.
- The materials used in NO_x adsorbers capture and store sulfur compounds that can be released only with a very high temperature cycle. For this reason, fuel with no more than 15 ppm sulfur must be used.
- Costs are unknown at this time. In its regulatory impact analysis for the 2007 onroad diesel engine standards, EPA projected that NO_x adsorbers would ultimately cost about 25 percent more than DPFs. In the short term, early retrofit systems are expected to cost as much or more than current SCR systems.
- NO_x adsorbers incur a fuel economy penalty of 4–7 percent because of the fuel used to release collected nitrogen from the adsorber catalyst.

Alternative Diesel Fuels

Various fuel parameters can affect both NO_x and $PM_{2.5}$ emissions, regardless of both engine and after-treatment technology. One approach to reducing emissions is to use a non-standard diesel fuel formulation.

Most alternative "clean" diesel fuels are based on blending a small amount (up to 20 percent by volume) of another substance with commercially available petroleum diesel. Various approaches are available, including blending with water (emulsified diesel), alcohol (oxy-diesel), converted vegetable oil (biodiesel) or metal-based powdered catalysts in a liquid suspension (fuel-borne catalysts). In the future, synthetic diesel fuel made from natural gas may also be available.

In addition, reducing sulfur levels in diesel fuel can itself reduce PM emissions, while also improving the effectiveness of some retrofit PM control devices, as discussed above. EPA rules will require significant diesel sulfur reductions beginning in late 2006. In some parts of the country, ultra-low-sulfur diesel fuel with less than 30 ppm sulfur has been available for several years, ahead of EPA mandates. Some fleets have already begun to use this fuel in order to reduce emissions.

The implementation dates for mandatory sulfur reductions in nonroad and marine fuels lag those for onroad fuel by several years. Between now and 2012, the use of reduced-sulfur onroad diesel fuel in nonroad construction, locomotive and marine engines could complement other emissions reduction strategies. See Chapters 13 (Trucks and Buses) and 14 (Nonroad Equipment) for a full discussion of this issue.

Emulsified Diesel Fuel. Emulsified diesel fuel (ED) is commercial diesel fuel blended with up to 20 percent water and a small amount of a proprietary hydrocarbon additive. The additive creates a stable emulsion that will not separate, in which the water molecules are completely enclosed by fuel molecules. This prevents the water from coming into contact with the engine and fuel system components, thereby preventing corrosion and maintaining lubricity.

During combustion, evaporation of the water contained in the fuel decreases peak combustion temperatures; this lowers NO_x formation and also results in greater atomization of the fuel, which reduces $PM_{2.5}$ emissions.

The following implementation issues and costs apply when using ED:

- Emulsified fuel can be used in virtually any diesel engine without modification.
- Creation of the emulsion requires specialized

blending equipment. Blending can be done at the fuel supplier's terminal or at a customer site.

- Emulsified fuel is stable, but it can stratify if left in a tank for an extended period without use.
- Because of its water content, emulsified fuel has 10–30 percent lower effective energy content per gallon than standard diesel, resulting in lower fuel economy (miles per gallon).
- Since fuel is typically metered into an engine's cylinder by volume, the use of emulsified fuel will result in a 10–30 percent reduction in power output at the same throttle setting.
- ED typically costs $0.01–$0.10 per gallon more than the base diesel fuel. In addition, typical vehicles will use 10–20 percent more fuel to do the same amount of work.

Biodiesel. Biodiesel (sometimes called BD) is a renewable fuel with high oxygen content and low sulfur content. It is composed of mono-alkyl esters and long-chain fatty acids derived from vegetable oils or animal fat. Biodiesel has very similar properties to petroleum diesel.

Oxygenates in biodiesel lower $PM_{2.5}$ emissions by supplying additional combustion oxygen. This can also increase peak combustion temperature, resulting in additional NO_x formation.

The following implementation issues and costs apply when using biodiesel fuel:

- Biodiesel can be used in virtually any diesel engine with no or minimal modifications.
- Biodiesel is a robust solvent that can loosen accumulated deposits in a vehicle fuel system. When switching to biodiesel, additional fuel filter changes are usually required.
- While biodiesel can be used undiluted (B100), it is generally recommended that biodiesel not constitute more than 20 percent of the total fuel mix. This is because a B20 blend achieves much of the potential $PM_{2.5}$ reduction benefit while minimizing potential NO_x emissions increases.
- Recently, some engine manufacturers have indicated that the use of biodiesel blends with greater than 5 percent biodiesel will affect their engine warranties, due to concerns over the effect of biodiesel on fuel injectors.
- In parts of the country with significant biodiesel use, 100 percent biodiesel has in the past typically been 1.5–2.0 times more expensive per gallon than petroleum diesel. At these prices, a B20 biodiesel blend would cost $0.05–$0.10 per gallon more than the base diesel fuel. Given recent increases in petroleum diesel fuel prices, biodiesel is likely to be more cost competitive, at least while petroleum diesel prices remain high. Some states also give biodiesel preferential tax treatment, lowering the differential cost compared to petroleum diesel, which is taxed at a higher rate.

Oxygenated Diesel Fuel. Oxygenated diesel (sometimes called OD) is a blend of standard petroleum diesel fuel with a small amount of an alcohol (up to 10 percent) and a proprietary hydrocarbon co-solvent additive that keeps the alcohol from separating out of the diesel. In a diesel engine, the alcohol provides increased combustion oxygen—similar to biodiesel and with similar results (lowering $PM_{2.5}$ emissions and potentially increasing NO_x emissions). Ethanol is low in reactivity and high in oxygen content, making it the preferred oxygenate, but methanol can also be used.

The following implementation issues and costs apply when using oxygenated diesel fuel:

- Oxygenated diesel can be used in virtually any diesel engine with no modifications
- The alcohol increases the vapor pressure inside a closed fuel tank above that of typical diesel fuel, to levels similar to gasoline.
- Oxygenated diesel typically costs $0.05–$0.10 per gallon more than the base diesel fuel.

Fuel-Borne Catalysts. When metal-based powdered catalysts in a liquid suspension are added to diesel fuel in very low concentrations, they can promote more complete combustion in a diesel engine, thus potentially reducing emissions of both NO_x and $PM_{2.5}$. Various companies sell proprietary catalyst packages, which may include small amounts of platinum, cerium, other precious metals or iron compounds.

At present, no fuel-borne catalyst has been verified by either EPA or the California Air Resources Board (CARB) as a stand-alone product (see below), though several are included as part of verified systems when used in conjunction with a DOC or FTF. Based on the verification results, the use of a fuel-borne catalyst appears to provide additional benefits beyond those achieved by the retrofit devices alone.

The following implementation issues and costs apply when using fuel-borne catalysts:

- Fuel-borne catalysts can be used in virtually any diesel engine with no modifications.

- Some catalyst metals such as cerium pose inhalation hazards; therefore, since metals are increasingly implicated in PM health effects, fuel-borne catalysts that contain these metals should only be used in conjunction with a high-efficiency DPF.
- While catalyst formulations can theoretically be added by the operator directly to the fuel tank on individual pieces of equipment, this is not recommended. In order to better monitor and control dosage, on-board or at-the-pump dosing systems should be used, or the fuel supplier can add catalyst to bulk fuel.
- Fuel-borne catalysts typically add $0.05–$0.06 per gallon to fuel costs.
- Some manufacturers claim that fuel-borne catalysts can increase fuel economy by up to 5 percent, but these claims have not been confirmed through EPA verification testing (see below).

Synthetic Diesel Fuel. The Fischer Tropsch process can be used to make a high-quality liquid fuel from natural gas that has fuel properties very similar to those of standard diesel. With virtually zero sulfur and naturally high cetane, synthetic diesel can be used in any diesel engine without modification. Preliminary emissions testing has shown that synthetic diesel fuel produces fewer PM and NO_x emissions than standard petroleum diesel in at least some diesel engines. Several energy companies are currently developing manufacturing facilities to produce this fuel on a pilot basis, but it is not commercially available anywhere in the U.S. today and is not expected to be for a number of years.

Technology Verification Programs

In order to provide end-users with standardized information on the effectiveness of various retrofit and alternative diesel fuel technologies, both EPA and CARB operate technology verification programs (EPA, 2005a; CARB, 2005). Manufacturers who want to receive verification must submit test results from standard emissions tests, along with information on the limitations and special requirements of the technology (e.g., minimum fuel sulfur level). CARB also requires manufacturers to provide minimum warranties for their products.

The EPA and CARB programs have slightly different requirements, but each organization has agreed to provide reciprocity to products verified under the other's program. The EPA program requires manufacturers to verify a product or technology separately for each of a number of different engine families and also reports the actual emissions reductions achieved for NO_x, PM,[5] HC and CO.

5. While these programs do not distinguish between $PM_{2.5}$ and total PM, virtually all particulate produced by most diesel engines is less than 2.5 micrometers in diameter.

The CARB program also provides verification for specific engines and engine families. However, it separates technologies into three levels, based on emissions reduction effectiveness for PM, and it only reports the level that a product falls into, not the actual PM reductions achieved. The CARB categories include Level 1 (greater than 25 percent PM reduction), Level 2 (greater than 50 percent PM reduction) and Level 3 (greater than 85 percent PM reduction). In the case of technologies that also reduce NO_x, the actual NO_x control-effectiveness is reported.

Either EPA or CARB, or both, has verified at least one commercial product for each of the retrofit technologies and alternative diesel fuels discussed above, except NO_x adsorbers and oxygenated diesel fuel. In addition, a number of commercial products have been verified that combine several technologies together (e.g., SCR plus DPF, NRC plus DPF, DOC plus fuel-borne catalyst). In addition, several commercial products have been verified that combine engine modifications to reduce NO_x with after-treatment to reduce PM.

The latest information on verified technologies appears on the EPA and CARB websites. (See references at the end of this chapter.)

In addition to the EPA and CARB verification programs, the Swiss Agency for the Environment, Forests and Landscape has operated a verification program specifically for DPFs since the early 1990s. This program focuses on evaluating filtration efficiency and durability specifically for particles between 20 and 300 micrometers in size.

The Swiss Agency publishes a list of filters that meet its stringent criteria for filtration efficiency and durability, including greater than 95 percent reduction of total particle count (Suva, 2005). This list is called the VERT list. VERT-approved filters are recognized as "best available control technology" to reduce diesel PM emissions by numerous European environmental agencies, as well as by the U.S. Mine Safety and Health Administration. Under its verification program, CARB accepts VERT test results but requires supplemental proof of compatibility with the particular diesel engines proposed for retrofit. Some VERT-approved technologies are commercially available in the U.S. and could be considered by states as acceptable retrofit technologies regardless of their EPA or CARB verification status.

Idle Reduction

The primary role of a diesel engine in most vehicles is to provide power to move the vehicle down the road or around the work site. However, most onroad and nonroad diesel equipment has auxiliary loads (pumps, air compressors, heating/air conditioning, etc.) that are also supplied by the engine, both when the equipment is moving and when it is

stationary.

In certain situations, the fact that a vehicle operator must idle the vehicle's very large diesel engine in order to supply a fairly small load results in excess emissions. Common examples include long-haul truck drivers who leave their trucks running all night in order to provide lights and heating/cooling in the truck cabin while they are sleeping, or charter bus operators who idle their buses while waiting for their passengers to return from an outing—again to maintain comfort in the bus cabin.

Opportunities exist to reduce total vehicle emissions by finding an alternate means to supply these auxiliary loads, thus allowing the operator to shut off the main engine.

Approaches include adding equipment to the vehicle (e.g., a battery system or a small auxiliary engine) or installing infrastructure where specific vehicles congregate (e.g., electrical connections at truck stops or waiting facilities for bus drivers at major tourist attractions).

Besides onroad trucks and buses, certain types of nonroad diesel equipment may also be a good target for idle reduction efforts: in particular, switchyard locomotives and some passenger and container ships that have high on-board electrical loads while in port. These loads, which are usually supplied by on-board diesel engines, could be supplied by the land-side electrical grid, resulting in significant net emissions reductions.

Depending on how much idling is eliminated, idle reduction technologies can reduce total fuel consumption by up to 15 percent on trucks and 25 percent on locomotives (Argonne, 2005), with coincident reductions in NO_x and PM emissions. In some cases, the required idle reduction equipment will pay for itself in a relatively short time through fuel savings.

The following is a general description of the types of idle reduction technologies available for both onroad and nonroad equipment.

Engine Idle Management. The control module in an electronically controlled engine can perform a number of functions, including automatically shutting down the engine after it has been idling for a certain period of time. While this does not address idling required to supply auxiliary loads, automatic shutdown can complement efforts to educate vehicle operators to eliminate truly unnecessary idling. This capability is standard on virtually all modern, electronically controlled engines, but must be turned on by a maintenance technician using software provided by the engine manufacturer. Some engine manufacturers sell a slightly more sophisticated system that can also restart the vehicle as required to maintain engine coolant and/or cab temperatures.

Some systems are available for retrofit on locomotive engines without electronic control. These systems have their own ECM, as well as various sensors to monitor the engine and other vehicle systems. The systems are designed to shut down the engine automatically and to restart it as required to maintain appropriate engine coolant temperature, brake air pressure and battery voltage, etc.

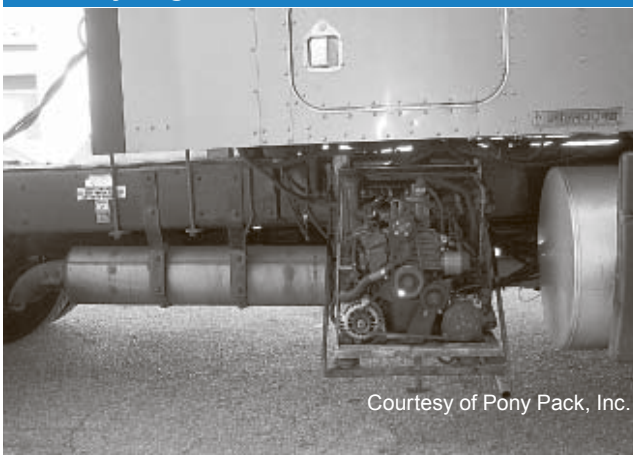
The following implementation issues and costs apply when using engine idle management systems:

- Automatic shutdown systems will not cause vehicles to stall in traffic since they are programmed not to shut down the engine with the vehicle in gear.
- Automatic shutdown on electronically controlled engines does not require any new equipment to be added to the vehicle.
- Shutdown/restart systems for non-electronically controlled engines require installation of an ECM and sensors.
- Basic automatic idle shutdown is available on electronically controlled engines at no extra cost.
- Shutdown/restart systems cost $1,000–$1,500 for onroad Class 8 trucks and $5,000–$10,000 for locomotives.[6]

Electrically-Driven Auxiliary Systems. Some vehicle auxiliary loads are driven directly by the engine (e.g., cab heating), while others are supplied electrically but are indirectly powered by the engine through the engine-driven alternator (e.g., air conditioning). Various companies sell equipment that will allow engine-driven loads to be driven electrically and/or allow electrical loads to be supplied even with the engine off. Some of these technologies rely on the engine's existing starter battery to

6. Information on all idling reduction technology costs is taken from the EPA website: www.epa.gov/otaq/smartway/idling technologies.htm.

Auxiliary Engine Installed on a Truck



Courtesy of Pony Pack, Inc.

supply the required electrical power, while others include additional supplementary batteries for longer service time.

The following implementation issues and costs apply when using electrically-driven auxiliary systems:

- In general, equipment must be added to the vehicle. Space requirements and installation time vary significantly by system.
- Costs are $500–$4,000 for onroad Class 8 trucks, depending on whether heating or air conditioning loads are covered and whether a supplementary battery is provided.

Auxiliary Power Units. The most versatile way to power various vehicle auxiliary loads with the main engine off is to use an auxiliary power unit. These units combine a small diesel engine and generator to provide electrical power for heating, air conditioning, pumps, lighting, etc. The engine is sized to match the required load, so that using these systems is much more efficient than idling the vehicle's much larger main engine.

In some cases these are stand-alone units that provide heated/cooled air directly to the vehicle cab, as well as a place to plug in standard electrical equipment. In other cases, there may be a high degree of integration with existing vehicle heating, air conditioning, lighting and other systems.

The following implementation issues and costs apply when using auxiliary power units:

- Equipment must be added to the vehicle. Space requirements, installation time and degree of integration with existing vehicle systems vary significantly.

Shore Power Idle Reduction System

Courtesy of IdleAire Technologies, Inc.

- Costs are $6,000–$9,000 for onroad Class 8 trucks and $30,000–$40,000 for locomotives.

Supplementary Diesel Heating Systems. Cab heat and/or heat to maintain engine coolant temperature for easy starting can be supplied by a diesel fuel-fired heater (similar in design to those used for residential heating, but much smaller). Power for the heater's fuel pump is supplied by the vehicle battery. Heated air can be supplied from the heater directly to the cab, or the heater can be used to heat the engine coolant and cab heat can be supplied through the vehicle's normal heating system. Some systems include timers for automatic starting as well as thermostats to maintain specific cab temperatures.

The following implementation issues and costs apply when using supplementary diesel heating systems:

- Installation is generally straightforward and required fuel is supplied from the vehicle's existing fuel tank.
- Costs are $900–$3,000 for onroad Class 8 trucks, and $15,000–$20,000 for locomotives.

Grid Power Systems. The electrical power required for on-vehicle auxiliary loads can be supplied from the electrical grid rather than being generated on-board. However, this always requires that infrastructure be installed at locations where vehicles congregate (e.g., trucks stops/rest stops). Some systems require only this fixed infrastructure to provide heated/cooled air to the truck cab along with lighting and a place to plug in electrical equipment. These services are usually bundled with other amenities, such as telephone and cable TV service.

Another approach is to supply electrical power to the vehicle from fixed infrastructure as a means of providing heating/cooling using the truck's existing on-board systems. In this case, modifications must also be made to the truck, so that its existing electrical systems can accept grid-supplied power.

Power plug-in units can also be designed for locomotives; these generally require both fixed infrastructure and modifications to the vehicle itself. This is an approach that may be well suited to commuter trains and yard switchers, but it is unlikely to be effective for many long-haul locomotives.

The following implementation issues and costs apply when using grid power systems:

- In all cases, fixed infrastructure must be installed at likely locations where truckers will stop to rest.
- Depending on the system, modifications may be necessary to enable the truck to accept and use power

from the grid.

- Truck modifications (if required) cost $100–$2,000 per vehicle.
- The cost of installing fixed infrastructure can be $5,000 or more per parking space. Generally, this cost is recovered through the charges levied for use of the system by truckers. Depending on what services are provided, these charges can range from $0.50 to $1.50 per hour.

Hybrid Electric Locomotive. Typical locomotives are propelled by electric motors. Electricity to power these propulsion motors is produced on board using a very large diesel engine coupled to a generator. The diesel engine is sized to provide the peak power needed to start the locomotive from a stop and to pull large loads up an incline.

Unlike long-haul locomotives, whose diesel engines work at or close to peak load for much of the time, switcher yard locomotives operate in short bursts and their diesel engines spend a lot of time at idle or working at very low loads. This type of duty cycle is appropriate for a "hybrid electric" drive system, in which the diesel engine is sized for the average load (and can therefore be much smaller), while peak loads are provided by a battery pack.

At least one company has developed such a hybrid switcher locomotive. It uses a 125 horsepower diesel (smaller than the engines used in Class 8 trucks) and 60,000 pounds of lead-acid batteries. The propulsion motors are powered by the batteries and the diesel engine continuously charges the batteries as they become depleted. This configuration is more efficient than using a large diesel engine and completely eliminates unproductive engine idling.

The following implementation issues and costs apply when using hybrid electric locomotives:

- This is a new technology that is currently undergoing prototype testing.
- Batteries are expected to last over ten years, but this has not been proven.
- Existing locomotives could be converted in lieu of an engine rebuild.
- Conversion of an old locomotive costs approximately $200,000, which is generally competitive with engine rebuilds, assuming that acceptable battery life can be achieved (Argonne, 2005).

Dock Electrification. Some ships, while docked in port, have large electrical loads that are supplied by onboard diesel-driven auxiliary generators. If this power were instead provided from the land-side electrical grid, net PM and NO_x emissions would be reduced significantly in most cases.

Dock electrification requires both land-side infrastructure and expensive modifications to the ships themselves. This approach is not practical for all ships, but it can be cost-effective for individual vessels with a large number of long-duration port calls in the same location and high dock-side electrical loads. The ship types most likely to be good candidates for dock electrification include cruise ships, refrigerated container vessels and some large tankers. See Chapter 17, *Marine Ports*, for a full discussion of dock electrification.

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Chapter 13

Diesel Trucks and Buses

Introduction

The onroad heavy-duty vehicle sector is composed of a wide variety of vehicles, from 18-wheel tractor-trailer combinations, to school and transit buses, to dump trucks and refuse haulers. These vehicles can be found in large numbers on major highways as well as on urban streets. The vast majority are powered by diesel engines.

Heavy-duty vehicles are categorized by weight class. In terms of numbers of vehicles, and especially fuel used annually, the heaviest Class 8 vehicles dominate. In 2002 there were over 2,082,600 Class 8 trucks on the road in the U.S. (U.S. Census Bureau, 2004) and they used over 17 billion gallons of diesel fuel (Davis, 2004).

The majority of these Class 8 trucks are long-haul tractor-trailers used to move goods over the nation's highways. Other examples of Class 8 vehicles include transit buses and refuse haulers.

Table 13.1

Heavy-Duty Truck Classifications

Class	Gross Vehicle Weight Rating (lb)
2B	8,501–10,000
3	10,001–14,000
4	14,001–16,000
5	16,001–19,500
6	19,501–26,000
7	26,001–33,000
8	> 33,000

Source: U.S. Census Bureau, 2004

Truck traffic in the U.S. is growing. The number of miles traveled annually by Class 8 trucks is expected to increase by approximately 40 percent through 2020 (EPA, 2000a). While future emissions regulations will mitigate the air quality impacts of increased miles traveled, heavy-duty onroad vehicles are, and will remain, a significant contributor to direct fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursor emissions.

While the number of heavy-duty trucks on the road is significantly smaller than the number of light-duty gasoline-fueled cars, their air quality impact is greater

Table 13.2

Heavy-Duty Truck Statistics (2002)

Class	No. Trucks (x000)	VMT (million miles)	Fuel Use (million gallons)
2B	396.7	5,031.2	318.2
3	621.1	8,428.6	1,075.1
4	287.3	4,184.2	533.7
5	291.1	3,949.2	503.7
6	855.8	11,361.3	1,449.1
7	419.1	5,726.7	995.9
8	2,082.6	100,167.0	17,420.3

VMT = vehicle miles traveled

Source: U.S. Census Bureau, 2004

because current diesel vehicles emit significantly more $PM_{2.5}$ and nitrogen oxides (NO_x) for each gallon of fuel burned than do gasoline vehicles. This is partly due to the nature of diesel combustion and partly due to a lag in EPA emissions regulations for diesel engines. In addition, heavy-duty diesel vehicles and engines tend to last much longer than gasoline cars and engines, so it takes longer for more stringent regulations to have a significant impact.

This chapter discusses the current and projected inventory of direct $PM_{2.5}$ and PM-precursor emissions from onroad heavy-duty vehicles, available regulatory authority to address these emissions from new and existing vehicles, and the control strategies that state and local authorities could consider to reduce emissions from existing heavy-duty vehicle fleets as they develop their State Implementation Plans (SIPs). Some of the strategies discussed involve the use of various diesel retrofit technologies. These technologies are described in Chapter 12, *Diesel Engine Technologies*.

Sector Profile

The vast majority of direct $PM_{2.5}$ emissions and $PM_{2.5}$-precursor emissions from heavy-duty onroad vehicles come from the combustion of diesel fuel in diesel engines. Almost all of the PM emissions in the exhaust of these vehicles is $PM_{2.5}$. Gasoline powered heavy-duty vehicles do exist, but they account for less than 3 percent of $PM_{2.5}$ and 6 percent of NO_x emissions from onroad sources. In addition, tire wear and brake wear together account for less than 2 percent of $PM_{2.5}$ emissions from heavy-duty vehicles. About a quarter of the PM emissions from tire wear is $PM_{2.5}$, and close to half of the PM emissions from brake wear is $PM_{2.5}$ (EPA, 2005a).

Exhaust from heavy-duty diesel vehicles accounts for about 65 percent of direct $PM_{2.5}$ emissions from onroad vehicles (EPA, 2005a). This is less than 2 percent of the total inventory of direct $PM_{2.5}$ emissions, including stationary sources and nonroad vehicles. However, the hazardous nature of diesel exhaust and the proximity of diesel exhaust sources to sensitive populations, particularly in urban areas, magnifies the health impact. Diesel exhaust is known to contain over 40 substances listed by EPA as hazardous air pollutants, 15 of which have been listed by the International Agency for Research on Cancer as known, probable or possible human carcinogens. Many of these substances are adsorbed onto emitted diesel exhaust particles. For this reason, the California Air Resources Board (CARB) has formally designated diesel PM as a toxic air contaminant (CARB, 1998).

Exhaust from heavy-duty diesel engines also accounts for about 50 percent of NO_x emissions from onroad vehicles and about 20 percent of all NO_x emissions, including those from nonroad diesel and stationary sources (EPA, 2005a).

All mobile sources, including light- and heavy-duty onroad vehicles and nonroad equipment, produce about 2–4 percent of total sulfur dioxide (SO_2) emissions nationally (EPA, 2005a). Mobile source SO_2 emissions will be reduced further beginning in 2006, when allowable fuel

Table 13.3

Diesel Truck and Bus Emissions (2002)

Category	$PM_{2.5}$ (tons/year)	SO_2 (tons/year)	NO_x (tons/year)
Heavy-duty diesel vehicles (highway)	97,000	105,000	3,378,000
% of highway vehicle emissions	65%	38%	46%
% of total mobile source emissions	22%	15%	29%

Source: EPA, 2005f

sulfur levels for both gasoline and diesel fuel will decline by over 90 percent (see below).

Efforts to reduce direct $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from the onroad diesel sector should focus on reducing exhaust emissions of both direct $PM_{2.5}$ and NO_x.

Regulatory Authority

As discussed in Chapter 4, the Clean Air Act gives EPA primary authority to regulate emissions from *new* onroad vehicles, including heavy-duty onroad vehicles. California has authority to adopt its own emissions standards for these vehicles, as long as they are at least as stringent as federal standards. Other states can adopt either the California or the federal standards. States and localities are free to set their own standards for *existing* onroad vehicles. As we discuss below, California standards for new heavy-duty onroad vehicles largely track federal standards.

New Vehicle Standards

EPA considers any vehicle over 8,500 pounds gross vehicle weight to be heavy-duty and therefore subject to different regulations than light-duty cars and trucks. The exception to this is that under EPA Tier 2 regulations for light-duty vehicles, certain very large sport utility vehicles (SUVs) and passenger vans used for personal transportation (8,500–10,000 lb gross vehicle weight) are re-classified as medium-duty passenger vehicles and are subject to the light-duty vehicle rules (DieselNet, 2005).

Unlike light-duty vehicles that are tested and certified at the vehicle level using a chassis test, only the engines themselves are certified for heavy-duty vehicles, using an engine test. Consequently, the numerical emissions limits are not expressed as grams per mile, but rather as grams per brake-horsepower hour (g/bhp-hr), equivalent to grams of emissions per unit of work done by the engine.

EPA first set exhaust smoke opacity standards for new heavy-duty onroad diesel engines beginning in model year 1970. Starting in model year 1974, new engines were required to meet numeric emissions limits for carbon monoxide (CO) and NO_x + hydrocarbons (HC), but PM was not regulated until 1988. Between 1988 and 1998, the EPA limits for HC and CO remained the same, but allowable levels of both NO_x and PM were reduced in several steps. At 4.0 g/bhp-hr NO_x and 0.10 g/bhp-hr PM, the emissions limits for the 1998 model year were 63 percent and 83 percent lower, respectively, than those for the 1988 model year.

In 1997, EPA adopted an even lower standard for heavy-duty diesel engine NO_x emissions to take effect in the 2004 model year. The next year EPA signed a consent decree with the six major heavy-duty engine manufacturers to settle EPA claims. According to EPA, manufacturers had for a number of years been using an "emissions defeat device" that modified engine control software to improve fuel economy, but that increased NO_x emissions during certain high-speed steady-state (highway) driving modes that were not fully captured during the certification test cycle. Among other remedies, the consent decree mandated

Table 13.4

EPA Emissions Standards for New Heavy-Duty Onroad Diesel Engines (g/bhp-hr)

Model Year Beginning	HC	CO	NO_x	PM
1988	1.3	15.5	10.7	0.60
1990	1.3	15.5	6.0	0.60
1991	1.3	15.5	5.0	0.25
1993	1.3	15.5	5.0	0.25 (0.10[a])
1994	1.3	15.5	5.0	0.10 (0.07[a])
1996	1.3	15.5	5.0	0.10 (0.05[a])
1998	1.3	15.5	4.0	0.10 (0.05[a])
2004	0.5	15.5	2.5[b]	0.10 (0.05[a])
2007[c]	0.14	15.5	0.2[d]	0.01

g/bhp-hr = grams per brake-horsepower hour

a. This lower PM limit applies to urban transit buses only.

b. This limit is for NO_x + non-methane hydrocarbons (NMHC).

c. After 2007, any crankcase emissions must be added to tailpipe emissions subject to these limits.

d. The new NO_x limits are phased in between the 2007 and 2010 model years on a percent-of-sales basis. The 2007 rules also add new steady-state tests and not-to-exceed limits for NO_x.

Source: DieselNet, 2005

the 2004 NO_x standard of 2.5 g/bhp-hr for engines built after October 2002. Under the consent decree, engine manufacturers were also required to develop modified software for model year 1993–1998 engines that would reduce off-cycle highway NO_x emissions (this software is often referred to as an ECM or chip "reflash") and to make this software available to vehicle owners free of charge. The consent decree required that all engines be upgraded with this new software at the time of normal engine overhaul or rebuild, which was assumed to occur after 200,000 to 300,000 miles in service (*U.S. v Caterpillar*, 1999).

In December 2000, EPA adopted a NO_x standard of 0.20 g/bhp-hr and a PM standard of 0.01 g/bhp-hr for new onroad heavy-duty diesel engines. This PM standard will go into full effect in the 2007 model year, while the NO_x standard will be phased in between 2007 and 2010 on a percent-of-sales basis.

The 2007 emissions standards introduce additional steady-state tests and not-to-exceed limits in the certification process, to ensure that defeat devices like those that led to the 1998 consent decree will no longer be possible. The 2007 regulations also require that crankcase vent emissions from all diesel engines be controlled (DieselNet, 2005). Previously, engines with turbo-chargers were allowed to vent their crankcase emissions to the atmosphere; these emissions were thus not included in the exhaust limits.

Because the onroad heavy-duty fleet turns over much more slowly than the light-duty fleet, there are still significant numbers of vehicles with unregulated or marginally regulated pre-1990 engines on the road. Even so, over the next 15 years current and pending EPA regulations will begin to have a significant effect. Based on normal fleet turn-over, EPA estimates that annual NO_x and PM emissions from heavy-duty onroad vehicles will fall by 22 percent and 37 percent respectively through 2010, and by 77 percent and 84 percent respectively through 2020, as new vehicles enter the fleet and older vehicles are retired (EPA, 2000b). This decline in annual emissions will occur despite a projected 40 percent increase in annual vehicle miles traveled through 2020.

Urban Bus Retrofit/Rebuild Rule

In 1993, EPA finalized a retrofit/rebuild rule that applies to urban transit bus engines built before 1994. The rule requires that a certified retrofit/rebuild kit be used every time one of these engines is rebuilt or overhauled. These kits will either reduce PM emissions by 25 percent compared to the original certification level of the engine, or will achieve PM emissions of 0.1 g/bhp-hr (EPA, 1998).

To date, EPA has not proposed retrofit/rebuild requirements for any other onroad diesel engines.

Fuel Quality

Congress has also given EPA authority to regulate the quality of fuel and fuel additives used in vehicles, based on direct health effects from the fuel itself and the effect of fuel on the ability of vehicles to meet emissions standards. EPA began to regulate the sulfur content of onroad diesel fuel in 1994, setting a limit of 500 parts per million (ppm). In order to allow the use of advanced catalytic emissions controls that will be required to meet the 2007 PM and NO_x standards, the most recent EPA regulations further reduce the allowable level of sulfur in onroad diesel fuel to 15 ppm beginning in late 2006 (EPA, 2001).

As discussed in Chapter 4, *The Clean Air Act*, if EPA has regulated a fuel characteristic or component, states other than California may *not* adopt either their own or California's regulation—they must follow the federal regulation. The only exception to this is the situation in which the Administrator finds that state regulation is necessary for achieving the National Ambient Air Quality Standards (NAAQS). On the other hand, California may adopt its own fuel regulations.

Other Regulations: California

As has already been noted, California has authority under the Clean Air Act to set its own emissions standards for new vehicles and to establish requirements for fuels and fuel additives.

California's regulation of onroad heavy-duty diesel engines began in 1973, but PM emissions were not regulated until 1987, one year before EPA began regulating PM. In addition, California's 1987 NO_x standard of 6.0 g/bhp-hr was not matched by federal standards until 1990. However, with the exception of engines for urban buses, the numerical emissions limits set by California have been virtually identical to EPA limits since then. California adopted more stringent standards for urban buses between model years 2004 and 2006; these standards included fleet averaging and mandatory fleet-wide emissions reduction requirements. For the 2005 model year, California also adopted supplemental steady state tests and not-to-exceed limits that will not be mandated by EPA until model year 2007 (DieselNet, 2005).

After 2007, California's numerical emissions limits for onroad heavy-duty diesel engines are virtually identical to EPA standards. However, in July 2005 California adopted rules that will require the addition of on-board diagnostic systems (OBD) for onroad diesel engines beginning in the 2010 model year (CARB, 2005d).

Similar to the OBD systems required for light-duty vehicles under current EPA and California rules, these diesel OBD systems will be required to monitor engine systems that affect emissions and to warn the driver of failures that are

Table 13.5

California Air Resources Board (CARB)
Emission Standards for New Heavy-Duty Onroad Diesel Engines (g/bhp-hr)

Model Year	HC	CO	NO_x	PM
Onroad Truck				
1987	1.3	15.5	6.0	0.60
1991	1.3	15.5	5.0	0.25
1994	1.3	15.5	5.0	0.10
1998	1.3	15.5	4.0	0.10
2004	0.5	15.5	2.5[a]	0.10
2005	0.5	15.5	2.5[b]	0.10
2007	0.14	15.5	0.2	0.01
Urban Bus				
1991	1.3	15.5	5.0	0.10
1994	1.3	15.5	5.0	0.07
1996	1.3	15.5	4.0	0.05
2004[c]	0.05	5.0	0.5	0.01
2007	0.14	15.5	0.2	0.01

g/bhp-hr = grams per brake-horsepower hour

a. This limit is for NO_x + non-methane hydrocarbons (NMHC).

b. Adds a Supplemental Engine Test for certification, and not-to-exceed limits of 1.25x these numerical values anywhere on the engine map.

c. This is for diesel engines; there are different limits for alternative fuel engines. CARB also imposed fleet average and mandatory fleet reduction requirements for diesel transit bus fleets.

Source: DieselNet, 2005

likely to increase emissions, so that they can be repaired. Like inspection and maintenance (I&M) programs, OBD systems are designed to ensure that vehicles continue to meet new vehicle emission standards throughout their life.

All of the California requirements noted in this section apply to *new* vehicles. The California programs that are discussed in later sections of this chapter cover *existing* vehicles.

State Authority

Despite limiting the ability of state and local areas to regulate new vehicles, the Clean Air Act allows them to set standards for existing onroad vehicles. In fact, this responsibility falls primarily to the states, with EPA's role in regulating existing vehicles limited to ensuring compliance with new vehicle standards throughout the defined useful life of the vehicle, and potentially regulating emissions from re-built engines used in buses and other heavy-duty onroad vehicles (see above).

Emissions Control Opportunities

Many of the emissions reduction technologies that are and will be applied to new onroad diesel engines to comply with current and future EPA regulations can also be applied to existing engines and vehicles to reduce their emissions. There are three main options for reducing emissions from a diesel vehicle or fleet: use a cleaner engine, use a cleaner fuel or add a retrofit device.

In addition, changes to heavy-duty vehicles, to their operations or to the transportation system that reduce total fuel usage can reduce total emissions and save vehicle operators money.

Use a Cleaner Engine

New onroad diesel engines produce less than 50 percent of the NO_x and less than 20 percent of the PM produced by engines built 15 years ago. Most of these reductions have come from fundamental changes in diesel engine design. Removing older engines from service and replacing them with new engines can therefore have a large emissions benefit.

For most older vehicles it is more cost-effective to retire the vehicle itself. However, for some specialty vehicles—for example, a large construction crane—where the engine represents a relatively small percentage of the vehicle's total value, "repowering" an old vehicle with a new engine may be the preferred alternative.

In limited cases it is also possible to cost-effectively modify existing engines to incorporate more modern design elements and equipment—for example, improved injectors—that reduce emissions. This approach would also include reflashing of 1993–1998 model year engines with upgraded software to reduce off-cycle highway NO_x emissions. (See above discussion of 1998 consent decree.)

Use a Cleaner Fuel

Some alternative formulations of diesel fuel produce lower emissions of NO_x and/or PM in existing engines than standard onroad diesel fuel. The most common of these cleaner fuels are described in Chapter 12, *Diesel Engine Technologies*.

Heavy-duty engines are also available for trucks and buses that operate on natural gas and produce significantly lower PM emissions than current diesel engines. Natural gas engines have PM emissions essentially equivalent to the 2007 EPA PM standards for new diesel engines. These natural gas engines are also certified to produce 30–50 percent lower NO_x emissions than the current EPA NO_x standards for new diesel engines (DOE, 2005).

While it is technically possible to modify existing diesel engines to operate on natural gas, using a purpose-built natural gas engine is generally preferred (ICCT, 2005). Converting an existing diesel vehicle to natural gas also requires that the fuel system be extensively modified to use high-pressure gaseous fuel in lieu of liquid diesel fuel and that high-pressure natural gas fueling stations are available. Given the cost and current limited availability of natural gas fueling stations, conversion to natural gas is most appropriate for fleet vehicles that are centrally fueled and that operate within a limited geographic area.

Add a Retrofit Device

As described in Chapter 12, there are a number of retrofit technologies that can be installed on existing diesel vehicles to reduce emissions. Most of these are after-treatment devices that install in the exhaust system; they do not reduce the emissions produced by the engine rather they clean up the exhaust after it has left the engine, but before it is emitted to the atmosphere. Most of these technologies rely on some kind of catalytic process and can reduce either PM or NO_x emissions, but not both simultaneously. Discrete technologies can, however, be combined into a single commercial product to achieve simultaneous PM and NO_x reductions.

Retrofit devices include closed crankcase and crankcase filter systems that control crankcase vent emissions, which are not currently included in EPA engine emissions limits (but will be included beginning with the 2007 model year). Testing conducted on a 1998 onroad diesel engine under the EPA Verification Program showed that PM emissions from the crankcase vent amounted to 16–23 percent of the PM emissions coming from the tail pipe (RTI, 2003). Controlling crankcase emissions is especially important in terms of reducing PM exposure for certain populations, such as students riding school buses (Hill, 2005).

The three main options described above for reducing emissions from diesel vehicles are not mutually exclusive. In fact, some commercially available retrofit devices work better in conjunction with, or even require the use of, an alternative diesel fuel. Commercial products are also available that combine engine modifications with retrofit devices, while engine repowering can always be combined with retrofits to reduce emissions even further.

Fuel Economy Improvements

Other aspects of the way vehicles are designed and operated—besides the engine itself—can affect total fuel usage and emissions. While much less important for vehicles that operate at relatively low speeds in urban areas, aerodynamic characteristics play a big role in the fuel economy of trucks that operate at high speeds on highways, as does the rolling resistance of tires. Vehicle speed and even the way the operator drives can have a noticeable effect on fuel usage. In addition, the overall efficiency of the nation's transportation system for goods affects the total amount of fuel required to move products from the manufacturer to the consumer.

Technology-based approaches to reducing overall fuel usage for individual trucks and for the transportation system as a whole include the use of more aerodynamic trucks and trailers, the use of lower rolling resistance tires, the use of low-viscosity lubricants and the maintenance of proper tire inflation. Operational approaches that can reduce fuel usage include eliminating unnecessary engine idling, providing driver training, reducing highway speeds and improving freight logistics and intermodal shipping.

While primarily focused on reducing fuel usage, these approaches can also reduce total emissions of both NO_x and PM, especially from the Class 8 tractor-trailer trucks that are widely used to transport goods and that dominate the onroad heavy-duty diesel vehicle sector.

As discussed in Chapter 15, *Light-Duty Cars and Trucks*, keeping maximum highway speeds below about 55 or 60 miles per hour (mph) also limits fuel use, as vehicles operate less efficiently at higher speeds. For cars, fuel economy generally deteriorates rapidly above 60 mph;

in fact, the U.S. Department of Energy (DOE) estimates that fuel economy declines 5–7 percent for every five mile per hour increase in vehicle speed above 60 mph. Diesel trucks and buses that lower their speed from 65 to 55 mph use approximately 20 percent less fuel (EPA, 2005f).

State and Local Policy Measures

Programs that encourage or mandate the retirement or replacement of older diesel vehicles, the use of cleaner fuels or diesel vehicle retrofits can have a large impact on air quality. These programs provide the most significant opportunities for state and local areas to control and reduce PM and NO_x emissions from onroad diesel fleets. As discussed below, whether these policies are voluntary or mandatory, their common focus is on the application of new technology to existing diesel vehicles. Programs that result in reduced fuel usage by onroad diesel vehicles will also reduce PM and NO_x emissions.

Voluntary Retrofit Programs

Replacing vehicles and engines, introducing alternative fuels and installing retrofit devices can be expensive. The Manufacturers of Emission Controls Association estimates that retrofit PM control devices for Class 8 trucks cost between $1,000 and $10,000 per vehicle, depending on the type of retrofit device (MECA, 2000). NO_x control devices can cost as much as $20,000 per vehicle and a new engine can cost $30,000 or more installed. Given this expense, voluntary emissions reduction programs are seldom effective without a funding source to offset some or all of the cost to the vehicle owner.

Programs to offset the cost of emissions controls for vehicle owners can take many forms, such as providing tax reduction or other tax incentives, or a direct rebate for some or all of the cost of implementing specific measures.

However, the most common programs for heavy-duty diesel vehicles are grant programs. These programs are usually competitive, beginning with a Request for Proposals that specifies the types of projects that will be considered. Often, proposals are evaluated for cost-effectiveness (dollars per ton of reduction), with only the most cost-effective projects funded, or with all projects above a specified minimum level of cost-effectiveness funded.

These programs generally provide a grant equal to the incremental cost of the action taken to reduce emissions. A grant program could fund any or all of the following types of emissions reduction projects: early vehicle retirement/scrappage, engine repowering, fuel switching, engine modification or vehicle retrofit.

The two largest grant programs currently operating are California's Carl Moyer Memorial Air Quality Standards Program (Carl Moyer) and the Texas Emissions Reduction Program (TERP). According to the CARB, Carl Moyer has provided $184 million in grant funding for various emissions reduction projects throughout California since its inception in 1998 (CARB, 2003). Approximately 45 percent of these funds have gone to projects to reduce emissions from onroad diesel vehicles. Maximum available funding for the current year totals $140 million. The 2003–2004 budget included $4.6 million for the purchase of new low-emitting school buses, and the 2005–2006 budget includes $25 million to retrofit or replace additional school buses (CARB, 2005b).

Since 2001, TERP has provided $39 million in grants for 41 onroad diesel emissions reduction projects in the 41 Texas counties designated as nonattainment areas. Funded projects included vehicle replacements, repowering, retrofits, purchase of cleaner fuels and infrastructure development (TCEQ, 2004). As discussed in Chapter 14, *Nonroad Equipment*, TERP has also funded nonroad diesel emissions reduction projects.

Both Carl Moyer and TERP make project awards based on a minimum level of cost-effectiveness and both are primarily designed to reduce NO_x emissions, with a secondary focus on PM. Each of these programs funds a range of onroad diesel projects in all of the categories discussed above.

In recent years, EPA has also made demonstration grants under its Voluntary Diesel Retrofit Program and has launched a Clean School Bus Initiative. The most recent grants were announced in February 2005, with a total of $1.6 million going to 18 projects across the country. These projects included retrofits of onroad trucks and buses as well as construction and agricultural equipment. The EPA grant program requires local matching funds (EPA, 2005b).

Over the last four years, the Puget Sound Clean Air Agency has also sponsored and funded approximately 3,000 retrofits of transit and school buses, refuse haulers, public works vehicles, construction equipment and port material-handling equipment, as well as fuel substitution for the world's largest ferry fleet. Its Diesel Solutions program and the Washington State Clean School Bus programs are both voluntary and incentive-based. The Puget Sound Clean Air Agency currently has appropriations of $5 million annually to retrofit school buses statewide (Puget Sound CAA, 2005).

Funding for clean air grant programs can come from a number of sources. The current TERP program is funded by surcharges on the sale or lease of new and used onroad and nonroad diesel vehicles, as well as by fees charged for commercial motor vehicle inspections and by a portion

of fees for certificates of vehicle title. Funding for this program is expected to total approximately $150 million per year through 2010 (TCEQ, 2004). In its early years, Carl Moyer was funded through annual appropriations in the state budget. Beginning in fiscal year 2005, the program has been funded by dedicating a portion of fees from the state's Smog Check program for light-duty vehicles, as well as by fees added to the sale of tires. Local air districts are also allowed to raise additional funds by adding $2 to the fees charged by the Department of Motor Vehicles (CARB, 2005b). In 2005, expected funding for the Carl Moyer program totals $61 million from Smog Check fees, $25 million from tire fees and $55 million raised by local districts (Lemieux, 2005).

In New Jersey, a diesel retrofit program adopted by the state legislature in June 2005 and approved by a substantial margin in the November 2005 general election will be funded by reapportioning 17 percent of the 4 percent portion of the state's corporation business tax that is currently dedicated to the remediation of leaking underground fuel storage tanks. This is expected to generate $14 million per year for retrofits (Iavorone, 2005).

In 1996, the citizens of the State of New York approved the Clean Water/Clean Air Bond Act. Under this program, New York issued $1.75 billion in general obligation bonds. The state used the money to fund a range of environmental programs. Approximately $230 million was spent on air quality projects, including retrofits and fuel conversions of onroad diesel vehicles (NY DEC, 2005).

A number of states have also used funds collected in environmental enforcement actions to fund diesel retrofit projects.

Mandatory Retrofit Programs

In addition to the voluntary programs discussed above, a number of states and localities have begun to mandate the retrofit or replacement of onroad diesel engines in specific types of fleets. California was the first to take this approach, adopting a fleet rule in 2000 for transit buses that mandated the use of cleaner engines, cleaner fuels and the retrofitting of older vehicles. This rule is focused on both NO_x and PM emissions. It mandates incremental progress over a ten-year period through more stringent standards for new engines as well as retrofit requirements (CARB, 2005a).

In 2003, CARB extended the same concept to other fleets by adopting a mandatory retrofit program, to be implemented between 2004 and 2010, for all diesel-fueled solid waste collection vehicles over 14,000 pounds gross vehicle weight. This program provides implementation flexibility because it does not require the use of a specific retrofit device. Instead, it requires the use of the Best Available Control Technology (BACT) that can reduce PM emissions to the greatest extent possible. Under the rules, repowering with a natural gas or cleaner diesel engine can be considered BACT, as can retrofit with a retrofit technology verified by the CARB (CARB, 2004b).

In April 2005, the New York City Council enacted legislation mandating the retrofit of private school buses, city-licensed sight-seeing buses, garbage trucks used for all city contracts and all city-owned or -operated diesel vehicles. Similar to California's regulations, these rules require vehicle owners to install BACT to reduce PM emissions (NY Mayor's Office, 2005).

The 2005 New Jersey program referenced above requires BACT retrofits for all diesel school buses, commercial buses, garbage trucks that are publicly owned or used on a publicly funded contract and other publicly owned onroad diesel equipment. Unlike the California and New York City legislation, the New Jersey program includes provisions to reimburse fleet owners for the cost of the retrofits.

Both the New York City and the New Jersey legislation combine direct and indirect mandates. Direct mandates apply to all vehicles of a given type regardless of who owns or uses them, while indirect mandates are imposed primarily through government purchasing and/or contracting requirements and are therefore more limited in scope. While government-owned and -controlled fleets generally make up only a small percentage of the onroad diesel vehicles in a locality, the appeal of indirect mandates imposed through government contracting policies is that they are usually easier to implement. By helping to build an experience base and develop a commercial market for retrofit products and services that can later be extended to private fleets, the impact of these mandates can also extend beyond the specific vehicles they target. However, experience has shown that the spillover of voluntary actions to the private sector is usually limited absent some direct economic incentive.

Other mandatory actions that state and local governments can take to reduce emissions from existing diesel vehicles include maximum vehicle age requirements for contracted government services like school buses and increased taxes and/or registration fees for older vehicles. These types of policies are designed to encourage retirement of the oldest, most polluting diesel vehicles.

States can also follow California's lead in mandating accelerated efforts to reflash the engine control software in model year 1993–1998 diesel engines to reduce off-cycle NO_x emissions during highway driving. Under the 1998 consent decree between EPA and diesel engine manufacturers, these software upgrades were supposed to be completed during normal engine rebuilds. However,

because the upgrades will also increase fuel usage during highway driving, vehicle owners have little incentive to comply. In early 2004, CARB estimated that fewer than 10 percent of eligible engines had been upgraded (CARB, 2004c).

In December 2004, CARB adopted a rule that requires owners of heavy-duty trucks, school buses and diesel-powered motor homes built between 1993 and 1999 to upgrade their engines by December 31, 2006 pursuant to a phased schedule and starting with the oldest engines. For the most part, the required software is already available and under the terms of the consent decree it must be provided by engine manufacturers free of charge. Compliance will be enforced through fines assessed as part of California's diesel I&M program. CARB estimates that its rule will affect as many as 450,000 vehicles and, when fully implemented, will reduce NO_x emissions by more than 35 tons per day (CARB, 2005c).

Inspection and Maintenance

Numerous studies have shown that a minority of vehicles create a disproportionate share of vehicle emissions (Niemeir, 2004). These high emitters are vehicles that are poorly maintained, misadjusted or have malfunctioning emissions control equipment.

Under the Clean Air Act, certain ozone and CO nonattainment areas are required to put I&M programs in place for light-duty vehicles. However, these requirements give the states a good deal of flexibility as to the type of I&M program they adopt. Moreover, even states without a federal I&M mandate can implement a state I&M program if there is no bar to their doing so under state law.

I&M programs are designed to identify high emitters and require vehicle owners to repair them. Inspections can be performed on a regular basis, usually in conjunction with renewal of vehicle registration or a state-mandated safety inspection, or randomly at the roadside after a pullover. Mandatory repair requirements can be enforced through denial of vehicle registration and/or imposition of a fine that is waived upon proof of repair.

I&M programs for light-duty gasoline vehicles are well developed, and tools for measuring emissions to identify high emitters have evolved over time. They now include devices that directly measure tailpipe CO, HC and NO_x emissions, as well as devices that use information from OBD systems.

Currently 16 states have some kind of I&M program for onroad diesel vehicles, either statewide or in certain counties (Diesel Technology Forum, 2004). The tools available to identify diesel high emitters are more limited than those available to gasoline vehicle I&M programs. Virtually all of the current diesel I&M programs use a green light opacity meter and the SAE J1667 "snap acceleration" test to measure smoke emissions as a proxy for PM emissions; NO_x, CO and HC emissions are not measured.

A review of available data from current U.S. programs concluded that "approximately 5–7 percent of pre-1998 vehicles record smoke emissions on the snap test in excess of standards" and that "smoke inspection programs are effective in reducing both the number of smoky vehicles and the average smoke opacity from any given model year's vehicles" (Diesel Technology Forum, 2004).

Measured smoke opacity from diesel vehicles is highly dependent on test procedures and conditions, and very little data exist to directly correlate measured opacity to PM mass emissions over a typical vehicle driving cycle. However, a study of 20 vehicles that had failed a smoke opacity test using the standards set by most I&M programs showed that after repair, both smoke opacity and PM emissions as measured over the Urban Dynamometer Drive Schedule (UDDS) declined on most vehicles. For pre-1991 vehicles, smoke opacity in a snap acceleration test (i.e., quickly ramping up the engine from idle to maximum governed RPM) declined by 33 percent on average after repairs, while post-repair PM emissions over the UDDS were on average 41 percent lower. The results were similar for 1991 and newer vehicles. The most common repairs performed were replacement of injectors and injector pumps. Repair costs ranged from $85 to $2,053 per vehicle, with an average cost of $1,088 (McCormick, 2003).

While this study is illustrative of the benefits that might accrue from a diesel I&M program, there are a number of practical concerns with these programs. The most important is that EPA's current MOBILE6.2 emissions inventory modeling software does not include any assumptions about high-emitting diesel vehicles, and it does not have any ability to quantify the benefits of a diesel I&M program. In addition, EPA has not issued guidance as to how a state might otherwise quantify the emissions reductions resulting from a diesel I&M program for purposes of developing a SIP for achieving federal air quality standards. Under these circumstances, it is unlikely that any state would be successful in arguing for SIP credit absent significant new testing information.

As noted above, California has mandated OBD systems on new diesel vehicles beginning in the 2010 model year. Adoption of these regulations by other states would eventually allow the use of OBD to identify high emitters in a diesel I&M program, as in many current light-duty I&M programs. However, such a program would be unlikely to have a significant impact prior to 2020.

In addition to diesel OBD, there are other testing technologies under development that could potentially supplement opacity testing in a diesel I&M program, including the use of remote sensing devices (RSD) to measure smoke opacity and gaseous emissions from onroad diesel vehicles as they drive by a fixed sensor. RSD technology is well-developed for light-duty gasoline vehicles and EPA has issued guidance concerning its use in light-duty vehicle I&M programs. However, RSD has only recently been demonstrated as a way to measure emissions from diesels—by the Massachusetts Bay Transportation Authority in Boston, Massachusetts, and in an EPA-funded project at the U.S.-Mexico border in Nogales, Arizona.

It is more difficult to apply RSD testing to heavy-duty than to light-duty vehicles, due to the ubiquity of high-mounted exhaust stacks and a lack of uniformity in stack heights on these vehicles. In addition, as with opacity testing, the lack of EPA guidance on quantifying benefits from an RSD-based diesel I&M program would make it difficult for states to claim SIP credit for such a program at this time.

Idle Reduction

All diesel vehicles, of necessity, idle at some point during their typical daily duty cycle—for example, while sitting at a traffic light. Unfortunately, however, some vehicle operators continue to idle their vehicles when it is not really necessary—examples include delivery trucks that idle while making deliveries or school buses that idle while picking up or dropping off students. Unnecessary idling wastes fuel and increases total daily emissions of NO_x and PM; often it also increases the exposure of sensitive populations to these pollutants. EPA estimates that idling Class 8 trucks use, on average, 0.82 gallons of diesel fuel per hour and produce 144 grams per hour of NO_x emissions (EPA, 2002).

In order to reduce or eliminate unnecessary idling, approximately 26 states and cities have enacted laws that restrict the idling of diesel vehicles. Typically, these laws set a maximum allowable idling time of 2–15 minutes. Operators of vehicles found parked and idling longer than the limit can be fined (ATRI, 2004).

These laws usually provide an exemption for all idling in traffic, as well as idling necessary to ensure safe operation of the vehicle—for example, to maintain the air pressure in the air brake system. Recognizing that starting diesel engines in extremely cold conditions can be difficult, these laws also often provide an exemption to allow additional idling below a specific ambient temperature. Finally, some idling restrictions also provide an exemption to maintain power for heating, cooling or auxiliary equipment. The Maricopa County, Arizona law allows additional idling to maintain bus passenger comfort when the temperature is above 75 degrees Fahrenheit (ATRI, 2004).

One contentious issue associated with many current laws that restrict idling is their effect on operators of coach buses and the degree to which idling to maintain operator and passenger comfort should or should not be considered necessary idling. Because both the heating and air conditioning systems on buses are operated by the diesel engine, many operators keep their buses running while parked to maintain comfortable cabin temperatures for themselves and their passengers.

Many vehicle operators have also complained about the lack of uniformity among idling laws in various locations. In early 2005, EPA announced an initiative to convene workgroups with affected parties to try to reach consensus on “model” idling restrictions.

As with buses, cab climate control and electrical power on tractor-trailer trucks are usually supplied by the truck’s diesel engine. For this reason, long-haul truckers often keep their trucks idling all night as they rest in the sleeper cab. This idling is generally exempt from current idle restriction laws. For example, California law currently prohibits diesel vehicles from idling for more than five minutes, but it exempts idling of trucks equipped with sleeper cabs. However, in October 2005 CARB considered a staff recommendation to extend the idle limit of five minutes to all diesel vehicles, including sleeper-cab trucks, beginning in January 2008. To support this requirement, the proposed regulation would mandate non-programmable idle shutdown features (which could not be modified by the user) on all new diesel trucks beginning in model year 2008 (CARB, 2005c). CARB staff anticipates that truck manufacturers would respond by also installing auxiliary power units, or other methods to supply power to auxiliary loads, as standard equipment on trucks equipped with sleeper cabs. Methods of providing necessary electrical power for climate control, lighting and other amenities while reducing or eliminating the need for extended overnight idling are described as part of the discussion of diesel technology in Chapter 12.

EPA testing has shown that use of these technologies can reduce total NO_x emissions by 90–99 percent compared to idling the truck’s main engine (EPA, 2002). However, these methods require additional equipment to be installed either on the truck, at truck stops, or both.

EPA’s review of currently available devices indicates that on-board idle reduction technologies can cost between $500 and $8,500 per truck, with the auxiliary engine systems (which provide the greatest flexibility) generally costing $5,000 or more. Services that provide cabin heating/cooling, electricity and other amenities at truck stops to preclude the need for on-board equipment generally charge between $0.50 and $1.50 per hour,

depending on the services provided and the contracting arrangements that apply (EPA, 2005d).

States could encourage or mandate the use of these idle reduction technologies in the same way that they encourage or mandate vehicle retirement, engine repowering, retrofit controls and fuel switching, as discussed above. EPA has issued guidance as to how a state can take credit for the benefits of an idle reduction program (EPA, 2004a).

EPA has funded several truck idle reduction demonstration projects, including electrification projects at the Hunts Point produce market in New York City and at a truck stop in Gary, Indiana, as well as a program launched with the Electric Power Research Institute to subsidize the purchase and installation of on-board, battery-powered electric air conditioning and heating systems for trucks (EPA, 2005d). EPA, the states of Oregon and Washington, and the non-profit Climate Trust are funding the installation of electricity connections at 275 parking spaces at truck stops in those states; 18 similar connections have been installed in Wilton, New York on Interstate 87 with funding from the New York State Energy Research & Development Authority (Argus Air Daily, 2005).

In addition to reducing NO_x emissions, EPA testing has shown that the use of technologies to avoid idling can reduce total fuel use by 0.8 gallons per hour or more (EPA, 2002). By eliminating only four hours of idling a day, a trucker could reduce his or her fuel costs by $2,000 per year or more,[1] and pay for the installation of many idle reduction technologies in three years or less. Despite relatively short payback periods, adoption of these technologies could be impeded by a vehicle owner's shortage of capital funds. In such cases, the development of a revolving loan fund could spur technology adoption (see Chapter 14, *Nonroad Equipment*, regarding the West Coast Collaborative revolving fund to finance idle reduction technologies on locomotives).

Reducing Fuel Usage

As discussed above, there are a number of changes besides idle reduction that can be made to individual vehicles, and to the goods distribution system generally, that will reduce fuel usage from onroad heavy-duty diesel vehicles, particularly Class 8 tractor-trailer trucks. These include the use of lower rolling resistance tires, the maintenance of proper tire inflation and the reduction of highway speeds. Additionally, EPA has developed a voluntary partnership program with the transportation industry, called the SmartWay Transport Partnership, to focus attention on these possibilities and their benefits (EPA, 2005e).

A similar program could be adopted or expanded at a local level to improve the transportation system and reduce fuel usage and emissions. In contrast to the emissions retrofit strategies discussed above, the approaches highlighted in the SmartWay program produce a financial payback for vehicle operators in the form of reduced fuel purchases. They are therefore more likely to be adopted without public subsidies, particularly in times of high and rising fuel prices.

In addition, linkages could be made between some of these fuel-saving strategies and the emissions-focused retrofits discussed above ("Green Branding"). In this way, fuel savings could be used to offset some of the cost of adopting emissions reduction technologies, thereby enhancing voluntary buy-in and reducing the total public subsidy required to elicit participation.

1. Assuming $2.50 per gallon for diesel fuel, a fuel savings of 0.8 gallon per hour and 250 days per year of vehicle operation.

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Chapter 14

Nonroad Equipment

Introduction

Nonroad motorized equipment comes in a wide variety of sizes and configurations. This category of emissions sources encompasses everything from lawn mowers and snowmobiles to construction equipment, ocean-going ships and locomotives.

Smaller types of nonroad equipment—which generally fall into the category of either lawn and garden equipment (e.g., lawn mowers, handheld leaf blowers and edgers, etc.) or recreational equipment (e.g., snowmobiles, all-terrain vehicles, small marine outboards, etc.)—are primarily powered by gasoline engines. Traditionally, exhaust emissions from these small engines have not been regulated as strictly as emissions from the larger gasoline engines used in cars and light trucks. For that reason, these pieces of equipment individually have very high emissions relative to the amount of fuel they consume. In addition, there are significantly more of them than there are of the larger, heavy-duty diesel-powered nonroad equipment. As a group, small nonroad gasoline equipment contributes a significant percentage of total volatile organic compound (VOC) emissions nationally, but is not a significant source of the direct fine particulate matter ($PM_{2.5}$) or $PM_{2.5}$-precursor emissions that are the focus of this report (EPA, 2005b). Therefore, small nonroad gasoline equipment is not discussed in this chapter.

With respect to direct $PM_{2.5}$ and major $PM_{2.5}$-precursor emissions, larger diesel-powered nonroad equipment is much more important. This category of sources, which encompasses primarily commercial farm and construction equipment, includes engines in a wide variety of sizes and configurations from as small as 20 horsepower to as large as 5,000 horsepower or more.

The major subcategories of nonroad diesel equipment include agricultural tractors and combines, airport ground-support equipment, construction and mining equipment (e.g., loaders, tractors, pavers, generators, compressors, etc.), commercial and industrial equipment (e.g., generators, compressors, refrigeration units, etc.) and logging equipment. In addition, rail locomotives and marine ships are generally included with these other types of nonroad equipment in emissions inventories. The larger marine ships generally have both very large engines that are used for propulsion and smaller auxiliary engines that

Table 14.1

Major Nonroad Equipment Populations

Equipment Type	Population (x1000)
Diesel	
Agricultural	3,208
Airport ground-support equipment	22
Commercial/industrial	10,456
Construction/mining	3,017
Logging equipment	330
Gasoline	
Lawn and garden	108,460
Recreational	25,488

Source: EPA, 2004b

are used to generate electricity and/or operate pumps on-board.

Technically, aircraft are also considered nonroad equipment, but they are addressed in Chapter 16, *Airports*.

Regardless of size, the diesel engines used in agricultural, construction, industrial and some marine equipment are generally similar; they are also similar to the engines used in onroad diesel vehicles. The engines used in locomotives and some ships are significantly different and are regulated differently by EPA. The propulsion engines in the largest ocean-going ships are unique and generally burn heavier residual fuels. EPA has separate emissions standards for locomotives and ships.

While agricultural equipment is concentrated in rural areas, other nonroad diesel engines contribute significantly to urban air quality concerns. In 1999, 45 percent of PM emissions and 26 percent of nitrogen oxides (NO_x) emissions from mobile sources in New York City came from construction equipment (EPA, 2005c). Likewise, marine emissions can be significant in coastal areas. In 2003, approximately 7.5 percent of all NO_x emissions in Los Angeles came from ships (CARB, 2005a).

This chapter discusses the current and projected inventory of direct $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from nonroad diesel equipment, the regulatory authority to address these emissions from new and existing engines, and the control strategies that state and local authorities can consider to reduce emissions from nonroad diesel equipment fleets as they develop their State Implementation Plans (SIPs). Some of the strategies discussed involve the use of various diesel retrofit technologies, which are described in Chapter 12, *Diesel Engine Technologies*.

Sector Profile

Most direct $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from nonroad equipment come from the combustion of diesel fuel in diesel engines. Approximately 90 percent of the PM directly emitted by these vehicles is $PM_{2.5}$. Some nonroad equipment is gasoline-powered, but it accounts for an insignificant portion of the direct $PM_{2.5}$ and $PM_{2.5}$-precursor inventory (EPA, 2005b). $PM_{2.5}$ from tire and brake wear in nonroad equipment is also insignificant.

The annual direct $PM_{2.5}$ emissions from nonroad equipment are twice those from *all* onroad vehicles, including both heavy-duty trucks and light-duty cars (EPA, 2005b). This is 4 percent of the total direct $PM_{2.5}$ inventory, including stationary sources. However, the hazardous nature of diesel exhaust and the proximity of diesel exhaust sources to sensitive populations magnifies the health impact. Diesel exhaust contains over 40 substances listed by EPA as hazardous air pollutants, 15 of which have been listed by the International Agency for Research on Cancer as known, probable, or possible human carcinogens (CARB, 1998). Many of these substances are adsorbed onto the particles that are emitted in diesel exhaust. For this reason, the California Air Resources Board (CARB) has formally designated diesel PM as a toxic air contaminant.

Exhaust from nonroad diesels also accounts for about 20 percent of *all* NO_x emissions, including those from stationary and other mobile sources—in fact, NO_x emitted by nonroad diesels amounts to about half that produced by onroad vehicles (EPA, 2005b). Therefore, nonroad diesel engines are an important source of $PM_{2.5}$-precursor emissions.

Nonroad equipment accounts for 2.5 percent of total annual sulfur dioxide (SO_2) emissions nationally (EPA, 2005b). SO_2 emissions from nonroad engines are higher than those from onroad vehicles because nonroad diesel fuel currently has significantly more sulfur than gasoline or onroad diesel fuel. However, these nonroad SO_2 emissions will be reduced dramatically beginning in 2007, when allowable fuel sulfur levels for nonroad diesel fuel will decline by over 80 percent (see below). Thus, efforts to reduce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from the nonroad diesel sector should focus on reducing exhaust emissions of both direct $PM_{2.5}$ and NO_x.

Regulatory Authority

As discussed in Chapter 4, the Clean Air Act gives EPA primary authority to regulate emissions from *new* nonroad equipment. California has authority to adopt its own emissions standards for some, but not all, *new* nonroad equipment, as long as its standards are at least as stringent as the federal standards. Other states can adopt either the California or the federal standards. For some *existing*

nonroad equipment, states can adopt California or federal standards, but not their own standards. Note that this stands in contrast to the Act's approach to *existing onroad* vehicles, for which both states and local areas are free to set their own standards. At present, California standards for new heavy-duty nonroad engines largely track federal standards. This is also true for California's onroad diesel rules, though not for California's rules with regard to nonroad gasoline equipment (e.g., leaf blowers and marine pleasure craft outboard engines), which are stricter than federal requirements. California has begun to adopt mandatory retrofit requirements for some categories of nonroad equipment.

That said, certain categories of nonroad equipment are fully preempted under §209(e) of the Clean Air Act, meaning that no state—including California—may set standards for this equipment. Section 209 preemption applies to new engines used in construction equipment or vehicles and new engines used in farm equipment or vehicles, provided the engines are smaller than 175 horsepower; it also applies to new locomotives or new engines used in locomotives. In addition, as discussed below, EPA regulates the emissions of NO_x (but not of other pollutants) from large ocean-going vessels with an eye to international treaty negotiations.

Generally speaking, EPA has three separate regulatory schemes for nonroad diesel engines: one for locomotive engines, one for marine engines and one for all other engines (agricultural, industrial, construction, etc.). Each of these categories of engines will be discussed separately. For regulatory purposes, nonroad engines are those used in *mobile* equipment that is not intended for use on public roadways. Therefore, the definition of "nonroad" is primarily based on mobility or portability. Despite the fact that it may be virtually identical to an engine used in a piece of mobile equipment, any diesel engine that will stay in the same place for 12 months or more—for example, to power a stationary generator or pump—is regulated as a stationary emissions source and is not discussed in this chapter (DieselNet, 2005).

Construction, Agricultural and Industrial Engines

The first EPA emissions standards for new nonroad engines other than marine engines and locomotives were adopted in 1994. These Tier 1 standards were phased in for engines of different sizes starting as early as model year 1996. In 1998, EPA finalized a rule that requires more stringent Tier 2 and Tier 3 standards to be implemented for engines built between 2001 and 2008, and in 2004 the Agency finalized Tier 4 standards that will take effect between 2008 and 2015 (EPA, 2004a, 2004f).

As with emissions standards for onroad truck and bus engines, emissions limits for nonroad engines are expressed in terms of mass per unit of work done by the engine—that is, grams per brake-horsepower hour (g/bhp-hr), or grams per kilowatt hour (g/kWh). Unlike the onroad standards, however, the nonroad standards provide different numerical limits depending on engine size, with smaller engines allowed to have relatively higher emissions. For example, the Tier 1 PM limits for a 175–300 horsepower engine were 0.40 g/bhp-hr, while they were 0.75 g/bhp-hr for an engine smaller than 11 horsepower.

In addition, the Tier 1–3 nonroad limits were significantly less strict than those in place for onroad engines at the same time. For example, a 250 horsepower engine installed in an onroad truck in 2000 would have had to meet a NO_x standard of 4.0 g/bhp-hr and a PM standard of 0.10 g/bhp-hr; virtually the same engine installed in an agricultural tractor in the same year would have been allowed to emit 6.9 g/bhp-hr NO_x and 0.40 g/bhp-hr PM.

When fully implemented in 2016, the Tier 4 nonroad limits will introduce general parity between onroad and nonroad standards for all diesel engines other than the smallest and largest engines. Nonroad engines smaller than 75 horsepower and some engines greater than 750 horsepower will still be allowed to emit significantly more NO_x and PM than onroad and other nonroad engines.

EPA has separate limits for heavy-duty spark-ignition engines (primarily fueled by gasoline). Given the relatively small stock of heavy-duty nonroad equipment powered by such engines, these limits are not discussed here.

Locomotives

EPA first regulated emissions from locomotive engines in 2000. In that year, a set of three standards (Tiers 0–2) became effective, with the operative standard depending on the year the engine was produced. Tier 0 standards apply retroactively to any engine manufactured between 1973 and 2001; Tier 1 applies to new locomotive engines produced from 2002 to 2004; and Tier 2 applies to new locomotive engines produced beginning in 2005 (EPA, 2004f).

Unlike other diesel engine standards, the locomotive standards provide two different numerical emissions limits based on two different test cycles that replicate the major modes of locomotive operation: line haul and switchyard. Line haul operation is characterized by significant time at the highest engine power settings, while in switchyard operations the engine spends a significantly greater amount of time at low power settings and at idle. The EPA Tier 0–2 regulations require that each engine meet both sets of numerical standards.

In addition, unlike other diesel emissions standards, the Tier 0–2 locomotive standards apply not only when the engine is first produced, but also every time the engine is re-manufactured. In effect, this means that every locomotive engine built since 1973 must be upgraded to meet Tier 0 standards when it next has a major overhaul. It is estimated that upgrading an unregulated locomotive engine to Tier 0 standards reduces NO_x emissions by approximately 30 percent, but produces no appreciable reduction in PM emissions. New engines built to comply with Tier 2 standards will have approximately 60 percent lower NO_x and 40 percent lower PM emissions than unregulated engines (DieselNet, 2005).

In June 2004, EPA issued an Advanced Notice of Proposed Rulemaking indicating that it was considering Tier 3 locomotive engine standards. These standards would be implemented for new locomotive engines built as early as model year 2011 and would seek to reduce both PM and NO_x emissions by 90 percent or more compared to the Tier 2 standards. This would bring locomotive engine emissions roughly in line with emissions from onroad trucks under the 2007 rules and with emissions from other nonroad engines under the Tier 4 rules (EPA, 2004a).

Marine Engines

For regulatory purposes, marine engines are divided into three categories based on size (cylinder displacement). The smallest Category 1 engines are very similar to the diesel engines used in onroad trucks and nonroad land-based vehicles. These marine engines are used for propulsion power in personal and some small commercial vessels such as tugboats and other harbor craft. They may also be used as auxiliary engines in larger commercial vessels. Category 2 engines, which are usually larger than 1,000 horsepower, are virtually identical to locomotive engines and are used for propulsion and auxiliary power in large commercial vessels.

The largest, Category 3 marine engines, are unique. Ranging in size from 3,000 to 100,000 horsepower, they are used for propulsion power in very large ocean-going vessels such as cruise ships, freighters and tankers.

In 1999, EPA adopted Tier 2 emissions standards for new Category 1 and Category 2 marine engines. The Tier 2 standards take effect between model years 2004 and 2007. These standards, which vary by engine size, set numerical limits that, for Category 1 engines, are similar to the Tier 2 nonroad standards and, for Category 2 engines, are similar to the Tier 1 locomotive standards. This regulation also designates voluntary "blue sky" emissions standards to which engine manufacturers may choose to certify. A blue sky certified marine engine has 40–50 percent lower NO_x and PM emissions than a Tier 2 engine (DieselNet, 2005).

In 2003, EPA adopted Tier 1 NO_x standards for new Category 3 marine engines. These standards take effect beginning in model year 2004. They are equivalent to the limits in Annex VI to the International Convention for the Prevention of Pollution from Ships (also known as the MARPOL Convention), which was negotiated internationally under the auspices of the International Maritime Organization (IMO). The Tier 1 NO_x standards for Category 3 engines are 9.8–17 g/kWh, depending on engine speed; slower engines are allowed to emit more (DieselNet, 2005). PM and other pollutants are not regulated under either MARPOL Annex VI or the EPA Tier 1 standards.

As with locomotive engines, EPA issued an Advanced Notice of Proposed Rulemaking in June 2004 indicating that it was considering Tier 3 standards for Category 1 and 2 marine engines, to be implemented as early as model year 2011. These proposed standards would seek to reduce both PM and NO_x emissions by 90 percent or more compared to the Tier 2 standards and would bring Category 1 and 2 marine engine emissions roughly in line with emissions from other nonroad engines under the nonroad Tier 4 rules (EPA, 2004a).

To date, EPA has not proposed any tightening of the current Tier 1 emissions standards for Category 3 marine engines. However, the current administration, through the State Department, has supported MARPOL Annex VI and has indicated a desire to further tighten NO_x standards and apply PM limits in future IMO negotiations (Argus, 2005a). The international nature of ocean-going vessels complicates efforts to effectively regulate their emissions without international agreements.

As discussed above, regulation to limit emissions from nonroad diesel engines is a relatively new phenomenon: standards for these types of engines were first introduced

Table 14.2

Standards for Low Sulfur Diesel Fuel

Category	Maximum Sulfur Limit (ppm)			
	2006	2007	2010	2012
Highway	15	15	15	15
Nonroad	3,000[a]	500	15	15
Marine Distillate/ Locomotive	3,000[a]	500	500	15
Marine Residual	20,000	20,000	20,000	20,000

a. 3,000 ppm is an approximate value representing uncontrolled emissions levels.

Source: Dieselnet, 2005

in 1994. The earliest regulation of nonroad diesel engines lagged behind the first standards for PM emissions from onroad diesel engines by six years; standards for onroad diesels, meanwhile, had lagged behind the regulation of gasoline cars by over 15 years. Since nonroad equipment can have a useful life of 40 years or more, this means that the vast majority of nonroad equipment in use today was not subject to any emissions limits when it was manufactured.

In addition, the turnover of nonroad diesel equipment tends to be very slow, so that even the more stringent current and future rules discussed above will not begin to have a major impact for many years. Based on normal fleet turnover, EPA estimates that annual NO_x and PM emissions from nonroad diesel engines will fall by only 10 percent and 20 percent, respectively, through 2010. By comparison, expected NO_x and PM reductions from the onroad heavy-duty diesel fleet over the same time period are expected to total 22 percent and 37 percent, respectively. Through 2020, NO_x and PM reductions for the nonroad fleet are expected to reach 37 and 56 percent, respectively; this compares to anticipated emissions reductions of 77 and 84 percent from the onroad fleet over the same time period. (EPA, 2000; EPA, 2004g).

Fuel Quality

EPA has authority to regulate the quality of fuel and fuel additives used in vehicles based on health effects associated both with direct exposure to the fuel and with the effect of the fuel on the ability of vehicles to meet emissions standards. EPA began to regulate the sulfur content of onroad diesel fuel in 1994 when it limited onroad fuel sulfur content to a maximum of 500 parts per million (ppm). The sulfur content of fuel used in nonroad diesels was unregulated at that time: distillate fuels currently used in construction equipment, locomotives and many marine vessels typically have sulfur levels as high as 3,000 ppm. The sulfur content of the heavier residual oils used to power ocean-going ships can be even higher, as much as 50,000 ppm or more.

To enable the use of advanced catalytic emissions controls, which will be required to meet the nonroad Tier 4 standards and proposed Tier 3 locomotive and marine standards discussed above, EPA has imposed limits on the allowable level of sulfur in nonroad diesel fuel for the first time. All nonroad diesel fuel will be limited to 500 ppm sulfur beginning in June 2007. Fuel sulfur limits will be reduced to 15 ppm for nonroad vehicles in 2010, and for locomotives and marine distillate fuels in 2012 (EPA, 2004f).

These sulfur limits do not apply to the residual fuels used in ocean-going ships, which remain unregulated in the U.S. IMO MARPOL Annex VI establishes a global cap of 45,000 ppm on the sulfur content of marine residual fuels. This treaty also includes a mechanism to establish a 15,000 ppm cap in specific Sulfur Emission Control Areas (SECA)—typically near coasts—where additional emissions reductions are considered critical. While the U.S. has not formally ratified Annex VI, the administration has supported ratification together with the designation of U.S. coastal waters as a SECA zone (Argus, 2005a).

As discussed in Chapter 4, *The Clean Air Act*, if EPA has regulated a fuel characteristic or component, states other than California may *not* adopt either their own or California's regulation—they must follow the federal regulation. The only exception to this is the situation in which the Administrator finds that state regulation is necessary for achieving the National Ambient Air Quality Standards (NAAQS). On the other hand, California may adopt its own fuel regulations.

Other Regulations: California

As noted previously, the Clean Air Act—subject to certain exceptions (e.g., locomotives)—allows California to set its own emissions standards for both new and existing nonroad equipment. However, California has, for the most part, adopted EPA regulations for *new* nonroad diesel equipment with only slight modifications (the same is true for California's *onroad diesel* rules, although its *nonroad gasoline* rules—for equipment like leaf blowers and marine pleasure craft outboard engines—are stricter than federal requirements). California has begun to adopt mandatory retrofit requirements for some categories of nonroad equipment. All of the California programs that are discussed in later sections of this chapter cover *existing* vehicles.

California also has its own regulations on the quality of diesel fuel, both onroad and nonroad. Under the California rules, nonroad diesel fuel sold in California for other than locomotive or marine applications will be limited to 15 ppm sulfur beginning in 2006, four years earlier than the nonroad diesel sold in the rest of the country (CARB, 2004b).

State Authority

States are confined to California standards or to federal standards for new nonroad engines, and they are limited to California standards for existing nonroad engines. In fact, even the authority to adopt California regulations for these sources is largely theoretical at this point, since California has just begun to adopt mandatory retrofit requirements for specific types of nonroad equipment, beginning with portable engines. However, none of the limitations on the imposition of state and local standards prohibits the adoption of voluntary programs, or the inclusion of retrofit requirements in government contracts for construction or other services.

Emissions Control Opportunities

Many of the emissions reduction technologies that are and will be applied to new nonroad diesel engines to comply with current and future EPA regulations can be applied to existing engines and vehicles as well. There are three main approaches to reducing emissions from these vehicles: use a cleaner engine, use a cleaner fuel or add a retrofit device. In many cases, changes to equipment, vehicles and/or operational practices that reduce total fuel usage can reduce total emissions and also save money.

Use a Cleaner Engine

New nonroad diesel engines certified to Tier 2 or Tier 3 EPA standards produce 30–60 percent less NO_x and PM than the unregulated engines built as recently as a few years ago. These reductions have come from changes in diesel engine design, most notably better control of lubrication oil, higher fuel injection pressures, and the addition of electronic fuel control. In some instances, particularly in the case of Tier 3 engines with electronic fuel control, emissions reductions have been accompanied by improved fuel efficiency. Removing older engines from service and replacing them with new engines can therefore have a large positive emissions benefit.

For most older vehicles, it is more cost-effective to retire the vehicle itself. However, in some cases involving specialty equipment, where the engine represents a relatively small percentage of the vehicle's total value, repowering an old vehicle with a new engine may be the better alternative. In particular, repowering has proved to be cost effective for Category 1 marine engines larger than 600 horsepower because resulting improvements in fuel economy can partially offset the cost of a new engine.

For some nonroad equipment, it may be possible to produce even larger emissions reductions by repowering with an *onroad* engine that is certified to even lower standards than new nonroad engines. In limited cases it may also be possible to cost-effectively modify existing engines to incorporate more modern design elements and equipment—such as improved fuel injectors—that result in lower emissions.

Use a Cleaner Fuel

Some alternative formulations of diesel fuel produce lower emissions of NO_x and/or PM in existing engines than standard nonroad diesel fuel. The most common of these cleaner diesel fuels are described in Chapter 12, *Diesel Engine Technologies*.

Heavy-duty engines that operate on natural gas and therefore produce significantly lower PM emissions than current diesel engines are also available for onroad trucks and buses. Natural gas engines have PM emissions essentially equivalent to the 2007 EPA PM standards for new onroad diesel engines. These natural gas engines are also certified to produce 30–50 percent lower NO_x emissions than the current EPA NO_x standards for new onroad diesels.

While it is technically possible to modify existing diesel engines to operate on natural gas, it is generally preferable to use a purpose-built natural gas engine (ICCT, 2005).

Whether using a purpose-built or retrofit engine, the vehicle fuel system must also be modified extensively to store high-pressure gaseous or low-temperature liquefied natural gas fuel in lieu of liquid diesel fuel; in addition, this approach requires that high-pressure natural gas fueling stations be available. This means that conversion to natural gas is most appropriate for fleet vehicles that are centrally fueled at a fixed site and that operate within a limited geographic area.

To date, there have been very few applications of natural gas engines to nonroad equipment. This is at least partially due to difficulty in developing an appropriate fueling infrastructure. Unlike onroad fleet vehicles that often travel to a centralized site for fueling, most construction equipment is regularly fueled at the construction site using mobile fueling equipment.

Nonetheless, there may be targeted opportunities to utilize natural gas fuel in specific types of equipment. The West Coast Collaborative and South Coast Air Quality Management District in California have provided $2.2 million to repower an existing locomotive with a natural gas engine and a liquefied natural gas fuel system (West Coast Collaborative, 2005c). This locomotive will be used to offset existing diesel truck traffic. The Port of Los Angeles has also recently announced a goal to shift from diesel power to the use of natural gas engines for nonroad cargo-handling equipment at the port (Port of Los Angeles, 2005).

In the category of cleaner fuels, electricity could also potentially play a role for some types of nonroad equipment. Depending on the fuel used for electricity generation, net NO_x reductions of 40–80 percent can be achieved by converting a nonroad vehicle from diesel to electric power (EPA 2004c, DieselNet, 2005). While the dirtiest coal- and oil-burning power plants could actually produce greater PM emissions than nonroad diesel equipment, the actual grid mix in many parts of the country would produce net PM reductions of 25–95 percent (EPA 1998, DieselNet, 2005). In either case, the localized air quality benefits are even greater, since all electric motors produce zero PM and NO_x emissions at the point of use.

Unlike onroad vehicles, which typically travel long

distances in a day, many pieces of nonroad equipment are essentially stationary or operate within a very limited geographic area for a relatively long period of time. The range limitations associated with using batteries to power mobile electric vehicles may therefore not be an issue in the case of nonroad equipment. Where such equipment is essentially stationary, the necessary electric power can be pulled directly from the electric grid without the need for onboard batteries and periodic charging.

As discussed further in subsequent sections of this chapter and in Chapters 16 and 17, some specific potential applications of electric power to replace diesel fuel in nonroad engines include: hybrid-electric locomotive yard switchers; electrification of forklifts, cranes and other material-handling equipment in ports and cargo terminals; electrification of construction cranes; gate electrification at airports; and the use of land-side electric power for auxiliary loads when ships are docked in port (cold ironing).

The use of reduced-sulfur onroad diesel fuel in nonroad equipment between now and 2010 can also directly reduce PM emissions from these vehicles, while also making the retrofit of after-treatment PM control technologies more effective. After 2010, nonroad diesel fuel will be required to have the same low-sulfur level as onroad fuel (low-sulfur locomotive and marine distillate fuel will be available somewhat later, in 2012).

Add a Retrofit Device

As described in Chapter 12, *Diesel Engine Technologies*, some retrofit technologies can be installed on existing diesel vehicles to reduce emissions. Most of these are "after-treatment" devices that install in the exhaust system; they clean up the exhaust after it has left the engine. Most of these technologies rely on some kind of catalytic process and can reduce either PM or NO_x, but not both at the same time. However, different technologies can be combined into a single commercial product to produce simultaneous NO_x and PM reductions.

The category of retrofit devices includes closed crankcase and crankcase filter systems that control crankcase vent emissions. These technologies are not currently included in EPA engine emissions limits (but will be included beginning with the 2011 model year). Testing on a 1998 *onroad* diesel engine showed that PM emissions from the crankcase vent totaled 16–23 percent of the PM emissions coming from the tail pipe (RTI, 2003).

The three main options for reducing emissions from diesel vehicles are not mutually exclusive. Some retrofit devices work better in conjunction with, or even require the use of, an alternative diesel fuel. Some commercial products combine engine modifications with retrofit devices, while engine repowering can always be combined with retrofits to reduce emissions even further.

Fuel Economy Improvements

As discussed in Chapter 12, the fuel economy of highway trucks is dramatically affected by the aerodynamics of the vehicle and the rolling resistance of its tires. This is not the case for most nonroad equipment, which operates at relatively low speeds or is essentially stationary. Opportunities to reduce fuel use through vehicle design changes that are unrelated to the engine are therefore much more limited for nonroad equipment.

Some new nonroad engines are both cleaner and more fuel efficient than older engines, so that repowering to reduce emissions can provide additional benefits from fuel savings. In addition, some nonroad sectors may provide opportunities for significant fuel savings based on operational changes. In particular, as discussed below, reduction of locomotive idling and cold ironing of marine vessels in port can result in significant emissions reductions based on reduced diesel fuel usage. The agricultural sector may also provide opportunities for fuel savings and emission reductions based on changes in farming practices.

State and Local Policy Measures

Programs that encourage or mandate the retirement or replacement of older diesel equipment, the use of cleaner fuels or electrification, or the implementation of diesel retrofits provide the most significant opportunity for states to control and reduce PM and NO_x emissions from nonroad diesel fleets. In fact, given the lag in nonroad emissions regulation and the slower turnover of these fleets, nonroad equipment is an even better target for these types of programs than onroad diesel vehicles. Whether voluntary or mandatory, the common focus of these programs is on the application of new technology to existing diesel equipment. Programs that result in reduced fuel usage will also reduce PM and NO_x emissions.

Voluntary Retrofit Programs

Replacement of vehicles and engines, use of alternative fuels, and installation of retrofit devices can be expensive. In general, the larger the engine, the larger and more costly the device. Replacement engines for construction equipment can cost $30,000 or more installed and Category 1 marine engines can cost $100,000 or more (TCEQ, 2004). Given this expense, voluntary emissions reduction programs are seldom effective without a funding source.

Programs to promote emissions reductions from existing engines can take many forms, including the use of tax policy or rebates to offset the cost of implementing specific

measures. The most common programs for heavy-duty diesel vehicles are grant programs. These programs are usually competitive and are often evaluated for cost-effectiveness (as measured in terms of dollar cost per ton reduction), such that funding is limited to only the most cost-effective projects or to those projects that achieve a specified minimum level of cost-effectiveness.

These programs generally provide a grant equal to the incremental cost of the action taken to reduce emissions. A grant program can fund early vehicle retirement/scrappage, engine repowering, fuel switching/electrification, engine modification and vehicle retrofits.

The two largest grant programs of this type currently operating are California's Carl Moyer Memorial Air Quality Standards Program (Carl Moyer) and the Texas Emissions Reduction Program (TERP). Since its inception in 1998, Carl Moyer has provided $184 million in grant funding for various emissions reduction projects throughout California (CARB, 2003). Approximately 30 percent of these funds went to projects to reduce emissions from nonroad diesel equipment, including marine vessels, construction equipment, forklifts and locomotives (CARB, 2005b).

Since 2001, TERP has provided $82 million in grants to replace 98 locomotives and retrofit six yard switchers; $11 million to repower 89 and retrofit six marine engines; and $14 million to replace or repower over 250 pieces of construction equipment, including forklifts, cranes, tractors, loaders, excavators, bulldozers and air compressors in the 41 Texas counties designated as nonattainment areas (TCEQ, 2004).

Both Carl Moyer and TERP make project awards based on a minimum level of cost-effectiveness and both are designed primarily to reduce NO_x emissions, with a secondary focus on PM.

EPA has also been making demonstration grants under its National Clean Diesel Campaign. The most recent grants were announced in February 2005, with a total of $1.6 million going to 18 projects across the country. The nonroad projects included retrofits of commuter rail locomotives, construction vehicles, agricultural equipment, rubber-tired gantry cranes and port straddle carriers. The EPA grant program requires local matching funds (EPA, 2005d).

Over the last four years the Puget Sound Clean Air Agency has also sponsored and funded approximately 3,000 retrofits of both onroad vehicles and nonroad equipment, including construction and port material-handling equipment. Its Diesel Solutions program is voluntary and incentive-based (Puget Sound CAA, 2005). The agency also helped to fund emissions testing of onroad low-sulfur fuel for the Washington State Ferry System, the largest ferry fleet in the U.S. This testing showed that the use of current onroad fuel in the ferries reduced PM emissions by approximately 50 percent. Puget Sound subsequently switched all of its ferries to onroad fuel (McLerran, 2005).

Funding for clean air grant programs can come from a number of sources. See Chapter 13, *Diesel Trucks and Buses*, for a more detailed description of funding mechanisms.

Although some strategies have a relatively short payback period because of fuel savings, a vehicle owner's shortage of capital funds can impede their adoption. In such cases, a revolving loan fund can spur technology adoption. Such a fund typically provides low-interest loans specifically for the purchase and installation of qualifying emissions reduction technologies. As the original recipients repay their loans, the funds are used to make additional loans. In 2005, the West Coast Collaborative developed such a fund to finance idle reduction technologies on locomotives operating on the west coast. This fund was established with $150,000 from EPA and $450,000 from the Climate Trust (West Coast Collaborative, 2005a).

Mandatory Retrofit Programs

States other than California are limited to California standards for existing nonroad equipment.

In 2004, CARB adopted mandatory retrofit requirements for portable diesel engines used in California in agricultural pumps, airport ground-support equipment, oil drilling rigs, portable generators and other nonroad equipment. The rule requires stepped-up reductions in PM emissions from these engines, with a 95 percent reduction to be achieved by 2020 (CARB, 2005). These are the first mandatory retrofit requirements California has adopted for any type of nonroad diesel engines, although CARB is evaluating a similar program for nonroad construction equipment.

Note that broad mandatory retrofit requirements are harder to impose on privately-owned nonroad equipment than on privately-owned onroad vehicles, in part for practical reasons, because—unlike onroad vehicles that use public roadways—nonroad equipment is usually not required to be registered with the state. One of the reasons that California was able to impose mandatory retrofit requirements for portable engines is that owners of these engines have been required for a number of years to register them with the state.

Government Contracting

While no state other than California has directly mandated the retrofit of nonroad diesel equipment, a number of

states and localities have begun—through construction contracting—to *indirectly* mandate the retrofit or replacement of nonroad diesel engines in specific types of fleets, primarily construction equipment. While the nonroad equipment used in government contracting generally makes up only a small percentage of all nonroad equipment used in a locality, the appeal of indirect mandates imposed through government contracting policies is that they are allowed by the Clean Air Act for all states and local areas, they are usually relatively easy to implement, and they may help to build an experience base and a commercial market for retrofit products and services that can later be extended to non-government controlled fleets.

In 2003, for example, the New York City Council adopted Local Law 77, which mandates the use of ultra-low-sulfur diesel fuel and requires retrofits for all diesel-powered construction equipment greater than 50 horsepower used on all *publicly funded* construction projects in New York City. The requirements were phased in between June 2004 and December 2005, depending on the location and size of the project (New York City Department of Design & Construction, 2004).

Local Law 77 requires the use of "best available technology" to reduce PM emissions to the greatest extent possible. Under the implementing regulations developed by the New York City Department of Environmental Protection, contractors must install an EPA- or CARB-verified diesel particulate filter on each affected piece of equipment unless they can demonstrate that these devices are not technically feasible. If so, they can install a diesel oxidation catalyst or use emulsified diesel fuel. (See Chapter 12 for details on these technologies.)

The Illinois Department of Transportation took a similar approach for the rebuilding of the Dan Ryan Expressway through Chicago. Under the Illinois Clean Air Construction Initiative, the Department is mandating the use of cleaner fuels and retrofits for all diesel construction equipment over 50 horsepower used on the project. All equipment will be required to use, at a minimum, highway diesel fuel, which is lower in sulfur than nonroad diesel fuel. Some equipment will be mandated to use ultra-low-sulfur diesel fuel and 25–30 pieces of equipment are expected to be retrofitted with emissions reduction devices (Burke, 2005). The Department will also impose limits on construction vehicle idling and parking and will implement dust control measures.

Requirements adopted by the Port of Seattle for the construction of a new runway at Seattle's Sea-Tac airport provide another example of this approach. In its contract specifications, the Port required that all construction equipment and haul trucks used for the project be late-model equipment certified to the most recent EPA emissions standards. The Port is also requiring that all equipment burn ultra-low-sulfur diesel fuel. This multi-year project is expected to consume 3–5 million gallons of diesel fuel per year (McLerran, 2005).

While experience has shown that the extension of voluntary government actions to the private sector is usually limited absent some direct economic incentive, meaningful results may still be achieved if the affected government sector is large enough. For example, given the large volume of publicly-funded construction in New York City—particularly in the context of the current downtown rebuilding effort—Local Law 77 is clearly intended to spur changes in the city's entire construction sector and to mitigate the direct effect of rebuilding activities on nearby sensitive populations. This is a plausible objective, since contractors presumably use the same equipment on both public and private projects.

State and local agencies also often own and operate a significant amount of nonroad equipment directly. Administrative policies that require retrofits, retirement/repowering or fuel switching for these fleets can complement similar policies for contracted equipment.

Inspection and Maintenance

Chapters 13 and 15 of this report discuss state-mandated inspection and maintenance programs (I&M) for onroad vehicles. These programs are generally designed to identify individual, high-emitting vehicles and require vehicle owners to repair them. As discussed in Chapters 13 and 15, I&M is much more common in the U.S. for cars and light trucks than for heavy-duty diesel trucks and buses. One reason for this is that there are fewer well-developed tools for identifying high-emitting diesel vehicles than there are for identifying high-emitting gasoline vehicles. The development of heavy-duty I&M programs is further hampered by the fact that current EPA inventory models have no facility to give states credit for any emissions reductions that might result.

Developing effective I&M programs for nonroad diesel equipment is even more challenging than developing such programs for onroad diesel vehicles for three additional reasons: (1) there are usually no state registration requirements for nonroad equipment, (2) such equipment operates on private property rather than on public road ways and (3) since most nonroad equipment currently has an unregulated engine, it is less clear what would constitute a high emitter.

No I&M programs for nonroad diesel equipment exist in the U.S. at this time. In 2005, however, CARB entered into a voluntary agreement with the BNSF Railway Company and the Union Pacific Railroad Company to reduce PM emissions in California rail yards. This

agreement requires the railroads to conduct annual opacity tests on every locomotive and to commit to repairing those locomotives that fail the test within 96 hours. These opacity tests are similar to the tests used in current I&M programs for onroad diesel vehicles. The agreement also requires the railroads to implement a pilot program to use remote sensing technology to measure emissions from in-use locomotives. While voluntary, this agreement includes monetary penalties for failure to implement its various requirements (CARB, 2005c).

The California program could be considered a first step toward developing I&M programs for nonroad equipment. However, EPA's current *nonroad* emissions inventory modeling software does not include any assumptions about high-emitting nonroad equipment and does not have any facility to quantify the benefits of a nonroad diesel I&M program that uses either opacity testing or remote sensing. In addition, EPA has not issued guidance as to how a state might otherwise quantify the emissions reductions that result from such a program under its SIP. Thus, it is unlikely that any state would be successful in arguing for SIP credit, absent significant new information based on testing.

Locomotive Idle Reduction

Chapter 13, *Diesel Trucks and Buses*, discusses idling restriction policies in the context of onroad diesel vehicles. These policies generally do not extend to nonroad diesel equipment for two reasons: (1) since this equipment operates on private property rather than public roadways, identifying idling vehicles and enforcing idling restrictions is more difficult and (2) the nature of the duty cycle can make the distinction between "necessary" and "unnecessary" idling more difficult to determine on many pieces of nonroad equipment.

Nonetheless, there are some distinct types of nonroad equipment that are known to idle excessively, particularly switcher yard locomotives. In some cases these locomotives are allowed to idle for long periods of time to maintain engine oil and coolant temperatures; in other cases, they are left to idle to provide a relatively small amount of electrical power for auxiliary equipment. They are often left to idle primarily because of the operator's perception that it is easier to idle than to shut down and restart. Argonne National Laboratory estimates that a typical switcher yard locomotive idles up to 75 percent of the time and that idling consumes 27 percent of the total fuel used (Argonne, 2005).

As discussed in Chapter 12, *Diesel Engine Technologies*, a number of alternative systems can be installed to provide the necessary electrical power for auxiliary loads on locomotives; in addition, systems exist that monitor various engine parameters and automatically shut down the engine when idling is unnecessary, restarting the engine as required. EPA's review of currently available devices indicates that these idle reduction systems cost between $4,000 and $35,000 to install on a locomotive, depending on the type of device used. Given the amount of fuel that can be saved by reducing or eliminating idling, many of these systems have payback periods of six to 20 months (Gaines, 2005).

States and local areas can encourage or mandate the use of these technologies in the same way that they encourage or mandate vehicle retirement, engine repowering, retrofits and fuel switching, as discussed above. EPA has issued guidance concerning how a state can take credit for the benefits of an idle reduction program (EPA, 2004e).

In 2003, EPA awarded the City of Chicago a grant to evaluate emissions, noise reductions and fuel savings available to a Class 1 railroad through the use of idle reduction technology (EPA, 2004d). The West Coast Collaborative has recently provided grant funding to several projects designed to reduce or mitigate the effects of locomotive idling. In addition to the revolving loan fund approach discussed above, it has funded a project to use stationary source emissions control equipment to capture and treat the exhaust from idling locomotives undergoing repairs in a maintenance facility (West Coast Collaborative, 2005b).

The Puget Sound Clean Air Agency has also provided a grant of $50,000 to Tacoma Rail to install idle reduction technology on two locomotives. In addition, Tacoma Rail has installed improved fuel injectors on these units to further reduce both emissions and fuel use (McLerran, 2005).

The TERP program in Texas has provided funding for the purchase of nine "Green Goat" hybrid-electric locomotives to be used by two railroads and a chemical company in Texas. As discussed in Chapter 12, these units can significantly reduce diesel fuel use and emissions from typical switcher yard operations compared to the use of a typical locomotive. In addition, Union Pacific has announced that it will add a total of 111 hybrid locomotives to its Texas operations over the next two years (Argus, 2005b).

The voluntary agreement between CARB and several California railroads to reduce PM emissions in rail yards requires that all locomotives operated by the railroads be equipped with "automatic idling reduction devices" by June 2008 (CARB, 2005c).

Nonroad Diesel Fuel

At approximately 3,000 ppm, the current sulfur content of nonroad diesel fuel impedes the use of effective retrofit

emissions controls on nonroad engines. In general, sulfur negatively affects the catalytic processes used to reduce PM emissions; diesel oxidation catalysts do not work well if fuel sulfur is greater than 500 ppm, and the even more effective diesel particulate filters require fuel with no more than 50 ppm sulfur (see Chapter 12).

While EPA regulations will eventually require nonroad diesel sulfur reductions, the implementation dates for nonroad diesel fuel lag behind those for onroad fuel. Beginning in June 2007, when nonroad diesel will be required to have no more than 500 ppm sulfur, diesel oxidation catalyst retrofits will become practical on a wide range of nonroad equipment. Retrofits with the more effective diesel particulate filters will not be possible until nonroad diesel fuel sulfur is reduced below 50 ppm, which will occur in 2010 for most nonroad equipment under current EPA rules and in 2012 for locomotives and marine distillate fuels. Retrofit programs for nonroad equipment will not be effective prior to these dates without either voluntary or mandatory use of onroad diesel fuel in the retrofit nonroad equipment. For this reason, both New York City's Local Law 77 and the Illinois Department of Transportation's Clean Construction initiative include mandates for the early adoption of reduced sulfur fuels in the affected construction equipment.

California is mandating that all nonroad diesel fuel sold in the state for other than locomotive and marine applications have no more than 15 ppm sulfur beginning in 2006—four years ahead of EPA mandates. While other states can implement similar requirements, such an action would require a finding by EPA that this would be necessary in order for the state to achieve federal air quality standards.

Given the very high sulfur content of the residual fuel oil used to power ocean-going ships, very few retrofit technologies will be effective on the Category 3 marine engines used for propulsion in these vessels. A switch to distillate fuels for these vessels would reduce PM and SO_2 emissions and open up greater opportunities for retrofits. However, this would require modifications to the fuel systems on these engines and would dramatically increase fuel costs for vessel owners.

A more practical approach would be a requirement to use reduced sulfur onroad or nonroad diesel fuel in the auxiliary engines that provide power while vessels are docked in port. For example, the Port of San Francisco is providing monetary incentives to offset the cost to cruise ship operators of using lower sulfur fuels while ships are docked at the port. EPA provided $100,000 toward the program through the West Coast Collaborative; other project partners are providing $141,000 in matching funds (West Coast Collaborative, 2005d).

The Washington State Ferry System operates all of its vessels with onroad diesel fuel to reduce PM emissions and is piloting the use of ultra-low-sulfur diesel and biodiesel in selected ferries (McLerran, 2005).

Marine Programs

Many marine vessels operate auxiliary engines while in port to power their "hotel loads"—that is, onboard electrical equipment and pumps. For certain types of ships, especially cruise ships, refrigerated cargo vessels and some tankers, the loads being supplied can be significant: anywhere from 600 to 7,000 kilowatts. In addition, some types of vessels spend significant amounts of time in port; cruise ships typically spend 12–24 hours at the pier at the beginning and end of each trip while exchanging passengers. Some cargo vessels take can take 30–100 hours or more to unload (Port of Long Beach, 2004).

The combination of high onboard power demand and long port calls results in significant in-port exhaust emissions from the auxiliary engines on these ships. For example, an analysis of port calls at the Port of Long Beach in southern California determined that a single ship that regularly calls at the port produces over 85 tons of NO_x, 9 tons of PM and 79 tons of SO_x annually while docked (Port of Long Beach, 2004). Given that most major ports are in or near urban areas, these emissions can significantly impact sensitive populations.

One approach to reducing these emissions is to provide electrical power from the land-side electrical grid to supply in-port hotel loads instead of using onboard auxiliary diesel engines. The Navy pioneered this approach, which is called cold ironing.

Cold ironing requires significant modifications to both the affected ships and the land-side infrastructure. The first cold ironing project in the U.S. was completed in 2001 in Juneau, Alaska. It involved the installation of necessary shore-side equipment and the retrofit of four cruise ships owned by Princess Cruises to accept shore power during port calls (Port of Long Beach, 2004). Installing the shore-side equipment cost $4.7 million and retrofitting the ships cost an average of $500,000 per vessel (Port of Long Beach, 2004). Princess Cruises is also participating in a shore power project at the Port of Seattle. This project is expected to reduce annual PM and NO_x emissions within the port by 2 tons and 14 tons, respectively (McLerran, 2005).

Unfortunately, given the low cost of marine distillate fuels, the typical cost of electricity supplied from the land-side grid is slightly higher than the cost of fuel for onboard electricity generation.

Given the costs involved, cold ironing is not appropriate for every ship. Nonetheless, the air quality benefits of this

approach can be significant, and it can be cost-effective for ships with a significant hotel load and a large number of annual port calls, each with a long duration. For example, the Long Beach study showed that the use of shore power to supply hotel loads to only three ships could reduce annual in-port emissions by almost 500 tons, at a cost of $11,000 per ton or less (Port of Long Beach, 2004).

The Port of Los Angeles has developed specifications for both the ship-side and shore-side equipment necessary for cold ironing, which it calls Alternative Marine Power (AMP). In June 2004, the port opened the first U.S. container berth equipped to provide shore power (Berth 100 at the China Shipping Terminal); two months later it hosted the first ship capable of accepting the power (the NYK Atlas). The port has plans to install AMP equipment at additional terminals (Port of Los Angeles, 2005).

Other actions that can be taken to reduce ship emissions in and around ports include the use of lower sulfur diesel fuel in auxiliary engines while in port, as discussed above, and efforts to reduce ship speeds near the coast.

Because ships that operate at lower speeds have lower emissions, both the Port of Long Beach and the Port of Los Angeles fund a Voluntary Commercial Ship Speed Reduction Program, which urges vessels to travel at or below 12 knots within 20 miles of the coast. The ports estimate that nearly 70 percent of shipping lines currently calling at the ports participate in the program and that NO_x emissions from ships have been reduced by over 1 ton per day (Port of Los Angeles, 2005a).

The Washington State Ferry system has also reduced operating speeds to reduce fuel usage, in addition to installing improved fuel injectors on its vessels (McLerran, 2005).

Operational Changes

Besides changes to equipment and fuels, some nonroad sectors may present good opportunities for emissions reductions through changes to infrastructure or operational procedures. These types of changes provide indirect PM and NO_x reductions by reducing total fuel usage in the nonroad sector.

For example, the Eastern Washington Farmers Diesel Emissions Reduction Program will offer cash incentives for farmers in six Washington counties to convert over 16,000 acres of crop land from the use of traditional tillage practices to no till/direct seeding. By eliminating several traditional operations prior to seeding, the switch will allow farmers to use their tractors and other nonroad diesel agricultural equipment less. No till/direct seeding is currently practiced by approximately 50 percent of farmers in the Midwest, but by only 10 percent of farmers in Washington. Project sponsors expect that the conversion will reduce total diesel fuel use on the affected farms by over 50,000 gallons per year and result in annual NO_x and PM reductions of 100 tons and 7 tons, respectively (West Coast Collaborative, 2005e).

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Chapter 15

Light-Duty Cars and Trucks

Introduction

More than 97 percent of all vehicles on U.S. roads are personal cars and small trucks (U.S. Census Bureau, 2004; Davis, 2004). In 2002, more than 220 million of these vehicles were registered in the U.S. and they traveled over 2.6 trillion miles, consuming nearly 130 billion gallons of fuel (Davis, 2004). Personal vehicle use continues to grow: annual vehicle mileage for cars and light trucks is projected to increase by more than 20 percent through 2020 (EPA, 2000b).

Traditionally, most Americans have driven cars; in 1990, for example, light trucks (pick-ups, vans, sport utility vehicles (SUVs)) made up only about 25 percent of the U.S. vehicle fleet (Davis, 2004). However, since 2000 nearly 50 percent of all new vehicles sold in the U.S. have been light trucks—primarily SUVs and mini-vans—and light trucks now make up nearly 40 percent of the overall fleet (Davis, 2004). Until recently the trend toward larger vehicles was almost universally expected to continue; recent volatility in gasoline prices has begun to affect light truck sales, but it is impossible to say whether this will develop into a long-term trend.

The increasing use of heavier, less fuel-efficient light trucks has increased both fuel use and exhaust emissions, eroding some of the efficiency gains made through the mid-1980s. As shown in Table 15.1, after a 69 percent increase in fuel efficiency between 1975 and 1987 (the year average fuel efficiency peaked), the average fuel efficiency of the U.S. light-duty fleet declined slightly through 1994 and has remained flat since then, at about 21 miles per gallon (EPA, 2005b). Stagnant fuel economy, coupled with a 60 percent increase in vehicle miles traveled, resulted in a 30 percent increase in fuel use between 1987 and 2002 (Davis, 2004).

This chapter discusses the current and projected inventory of direct fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursor emissions from onroad light-duty vehicles, the regulatory authority to address these emissions from new and existing vehicles, and the control strategies that state and local areas can consider to reduce emissions as they develop their State Implementation Plans (SIPs).

Table 15.1

U.S. Light-Duty Vehicle Fleet

Average Characteristics	1975	1987	2005
Fuel economy (mpg)	13.1	22.1	21.0
Weight (lb)	4,060	3,220	4,089
Horsepower	137	118	212
0-to-60 time (secs)	14.1	13.1	9.9
Truck sales (%)	19%	28%	50%

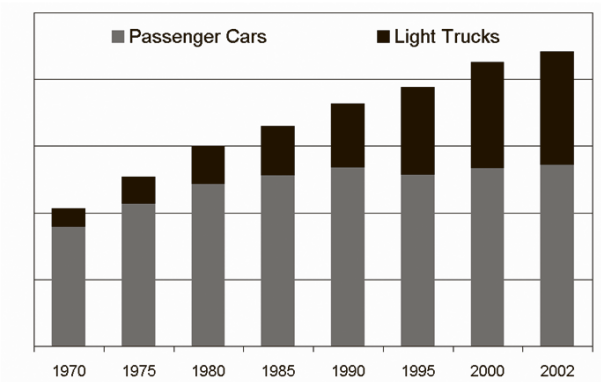
Source: EPA, 2005b

Sector Profile

The vast majority of light-duty vehicles in the U.S. fleet have gasoline engines; in 1998 only 2.2 percent of light-duty vehicles on U.S. roads used diesel fuel (J.D. Power, 2004). While some expect diesel vehicles to account for as much as 15 percent of the overall light-duty fleet by 2014 (J.D. Power, 2004), future regulations that will require emissions parity between light-duty gasoline and diesel engines will mitigate the potential negative air quality impacts of this trend.

Unlike current diesel engines, gasoline engines do not produce significant direct $PM_{2.5}$ emissions. A modern diesel engine without after-treatment will produce over eight times as much $PM_{2.5}$ per gallon of fuel burned as a modern gasoline car engine. Therefore, despite the fact that there are over 40 times as many cars in the U.S. fleet as heavy-duty diesel trucks, cars produce less than one-third as much direct $PM_{2.5}$ as heavy-duty diesel trucks. Moreover, only about half of direct $PM_{2.5}$ emissions from cars comes from engine exhaust; the rest comes from brake and tire wear (EPA, 2005f).

Fig 15.1 Light Duty Vehicle Registrations



Source: ORNL, 2004

The light-duty gasoline vehicle fleet is a much more important contributor to emissions of nitrogen dioxides (NO_x), a $PM_{2.5}$-precursor. In fact, light-duty gasoline vehicles produce about 46 percent of total NO_x emissions from onroad vehicles and account for about 16 percent of *all* NO_x emissions, including those from nonroad diesel and stationary sources (EPA, 2005f).

While sulfur dioxide (SO_2) is also a $PM_{2.5}$-precursor, all mobile sources together—including both light- and heavy-duty onroad vehicles and all nonroad equipment—produce about 2 percent of total annual SO_2 emissions nationwide. Moreover, this contribution will be reduced dramatically beginning in 2006 when allowable fuel sulfur levels for both gasoline and diesel fuel fall by over 80 percent (EPA, 2004).

Efforts to address the contribution of the light-duty gasoline vehicle fleet to ambient $PM_{2.5}$ levels should focus on reducing precursor NO_x emissions. To the extent that efforts are made to reduce direct $PM_{2.5}$ emissions from light-duty vehicles, reducing emissions from brake and tire wear are every bit as important as reducing exhaust emissions (EPA, 2005f).

Regulatory Authority

As discussed in Chapter 4, the Clean Air Act gives EPA primary authority to regulate emissions from *new* onroad vehicles, including light-duty cars and trucks. California has authority to adopt its own emissions standards for new vehicles, as long as they are at least as stringent as federal standards. Other states can adopt either the California or the federal standards.

As discussed below, California standards for cars and light trucks have historically been more stringent than federal standards. This trend has continued with California's recent effort—which is currently being litigated—to begin regulating greenhouse gas emissions from the light-duty fleet. States and localities are free to set their own standards for *existing* onroad vehicles.

New Cars and Light Trucks

EPA began regulating emissions from new light-duty vehicles in the 1970s and continued tightening emissions standards over the years that followed. Most vehicles on the road today were manufactured subject to the Tier 1 standards that were adopted in 1991 and phased in between the 1994 and 1997 model years. These standards allowed higher NO_x emissions from light-duty trucks than from cars and also allowed higher NO_x and PM emissions from diesel than from gasoline vehicles (EPA, 2000a).

In 2000, EPA adopted Tier 2 standards for light-duty vehicles, which will be phased in between the 2004 and

2009 model years. The Tier 2 standards apply the same absolute emissions limit (grams per mile) to virtually all passenger vehicles regardless of size or weight, including cars, pick-up trucks, vans and SUVs. They also apply the same limit regardless of the fuel used, so that new light-duty diesel vehicles will have to be as clean as vehicles with gasoline engines (EPA, 1999, 2004).

The Tier 2 standards allow vehicle manufacturers to certify to any one of eight "bins" with different emissions levels; Bin 1 is the cleanest, with the most stringent standards, while Bin 8 has more lenient standards. The average NO_x emissions from all vehicles that a manufacturer sells must be no more than 0.07 grams/mile, equivalent to Tier 2, Bin 5. This is almost 90 percent lower than the Tier 1 NO_x standard for cars. The Tier 2 PM standard is also 90 percent lower than the corresponding Tier 1 standard (EPA, 2000a; 2004).

Based on normal fleet turnover, EPA estimates that annual NO_x and PM emissions from light-duty vehicles will fall by 66 percent and 5 percent, respectively, through 2020 as new vehicles that comply with Tier 2 regulations enter the fleet and older vehicles are retired (EPA, 1999). Emissions are expected to decline despite a projected 20 percent increase in annual vehicle miles traveled during that time period.

EPA does not regulate carbon dioxide (CO_2) or other greenhouse gas emissions from vehicles. The position taken by the current administration is that the Clean Air Act does not authorize EPA to regulate greenhouse gas emissions from any source. This issue is currently being litigated.

The Department of Transportation (DOT) imposes Corporate Average Fuel Economy (CAFE) requirements on auto manufacturers. Current CAFE standards require manufacturers to maintain a car-fleet-average fuel economy of 27.5 miles per gallon and an average fuel economy of 21.0 miles per gallon for light trucks (STAPPA and ALAPCO, 2005). Higher fuel economy standards decrease the amount of pollution emitted per mile, since a vehicle is using less fuel per mile.

DOT recently proposed revised CAFE standards that will begin to close the "SUV loophole" in the current regulations. The proposed standards divide light trucks into six categories based on vehicle footprint. The smallest truck category would be required to achieve an average fuel economy of 28.4 miles per gallon in 2011, while the largest would be required to achieve only 21.2 miles per gallon. The proposed revisions do not increase CAFE requirements for cars, nor do they extend fuel economy requirements to vehicles over 8,500 pounds (lb)—a category that includes such vehicles as the Hummer H2, Ford Excursion and Chevy Suburban (STAPPA and ALAPCO, 2005).

Fuel Quality

EPA has authority to regulate fuels and fuel additives used in light-duty vehicles, based on direct health effects from the fuel and the effect of the fuel on the ability of vehicles to meet emissions standards. In the past, this has resulted in rules banning lead in gasoline, as well as limits on the volatility of gasoline sold in the summer to reduce evaporative emissions.

In addition, the Clean Air Act, beginning in 1995, required that certain cities with severe ozone problems use reformulated gasoline (RFG) in the summer, and that areas in nonattainment for carbon monoxide (CO) use oxygenated gasoline in the winter. The Clean Air Act requires the use of RFG in cities in 14 states, but it allows limited opt-in for other areas. Currently, RFG is used in part or all of 19 states and the District of Columbia;[1] approximately 30 percent of all gasoline sold in the U.S. is reformulated (EPA, 2005c).

RFG is formulated to meet a number of different specifications aimed at reducing emissions of ozone precursors and toxics such as benzene. One of these requirements is that it contains 2 percent oxygen by weight. The most commonly used oxygenate additive is ethanol, which is manufactured primarily from corn in midwestern states. Another additive that can be used as an oxygenate is methyl tertiary butyl ether (MTBE). MTBE is controversial because it can contaminate ground water. In addition, many people believe that RFG does not need to contain oxygen to improve air quality. California, New York and Connecticut petitioned EPA to have the oxygen requirement removed from their RFG standards, but EPA denied their requests (EPA, 2005c).

The Energy Policy Act of 2005 removed the requirement for RFG to contain oxygen, effective immediately in California and beginning in May 2006 everywhere else. While early versions of the legislation also contained an outright ban on MTBE in gasoline, this provision did not make it into the enacted law. California, New York and Connecticut have all taken individual action that partially or completely bans the use of MTBE in gasoline.

The Energy Policy Act of 2005 also contains a requirement that renewable fuels begin to make up a specified portion of the gasoline sold in the U.S. The law requires that 4 billion gallons of renewable fuel be used in 2006, increasing to 7.5 billion gallons per year in 2012. The most likely fuel to be used to meet this requirement is ethanol (more below).

1. California, Connecticut, Delaware, the District of Columbia, Georgia, Illinois, Indiana, Kentucky, Louisiana, Maryland, Massachusetts, Missouri, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Texas, Virginia and Wisconsin.

The Energy Policy Act of 2005 also allows opt-in to the RFG program at the request of a state, for the 11 northeastern states in the Ozone Transport Region.

Based on the detrimental effect of sulfur on modern emissions after-treatment systems, EPA has also recently focused on reducing sulfur levels in both gasoline and diesel fuel. The Tier 2 light-duty regulations require that most gasoline refiners and importers meet a corporate average sulfur standard of 120 parts per million (ppm), with an absolute cap of 300 ppm, beginning in 2004. This will be reduced to an average of 30 ppm with a cap of 80 ppm by the end of 2006 (EPA, 2004).

As discussed in Chapter 4, *The Clean Air Act*, if EPA has regulated a fuel characteristic or component, states other than California may *not* adopt either their own or California's regulation—they must follow the federal regulation. The only exception to this is the situation in which the Administrator finds that state regulation is necessary for achieving the National Ambient Air Quality Standards (NAAQS). On the other hand, California may adopt its own fuel regulations.

Other Regulations: California

Under the Clean Air Act, California can set its own emissions standards, even for new vehicles. California can also establish its own requirements for fuels and fuel additives.

The Tier 1/Low Emission Vehicle (LEV) California standards in effect through 2003 were very similar to Tier 1 EPA standards, though they included additional, more stringent "low emission vehicle" regulatory categories (Transitional Low Emission Vehicle (TLEV), Low Emission Vehicle (LEV), Ultra-Low Emission Vehicle (ULEV) and Super Ultra-Low Emission Vehicle (SULEV)) for vehicles 6,000–14,000 lb gross vehicle weight). The Tier 1/LEV program established an increasingly stringent fleet-average requirement for emissions of non-methane organic gases (NMOG), which required auto manufacturers to certify an increasing share of their total vehicle sales in the lower emissions categories. The Tier 1/LEV program also required auto manufacturers to begin selling limited quantities of Zero Emission Vehicles (ZEVs), which at the time were generally battery-electric vehicles. In addition, unlike the EPA Tier 1 standards, the California Tier 1/LEV standards applied the same numerical emissions limits to both gasoline and diesel cars and light trucks (EPA, 2000a).

Beginning with the 2004 model year, California began to phase in LEVII standards. LEVII eliminates the Tier 1 and TLEV regulatory categories, significantly reduces allowable PM and NO_x emissions in the LEV and ULEV categories and establishes a SULEV category for vehicles less than 6,000 lb gross vehicle weight. LEVII also establishes Partial Zero-Emission Vehicle (PZEV) and Advanced Technology Partial Zero-Emission Vehicle (AT-PZEV) categories. To certify as a PZEV or AT-PZEV, a vehicle must meet SULEV emissions standards, have zero fuel evaporative emissions and carry a 15-year/150,000 mile full emissions warranty. PZEV and AT-PZEV vehicles can be used by auto manufacturers to meet California's ZEV requirements (CARB, 1999).

California's LEVII standards apply to all cars and light trucks weighing less than 8,500 lb—meaning that most pick-ups and SUVs in California will be required to meet the same standards as cars. Under LEVII, the LEV and ULEV NO_x and PM limits are the same as EPA Tier 2, Bin 5; and the SULEV NO_x and PM limits are the same as EPA Tier 2, Bin 2 (DieselNet, 2005).

The LEVII standards extend the requirements for automakers to certify increasing percentages of vehicles in the ULEV and SULEV categories. As a result, fleet-average emissions levels for new vehicles sold in California will continue to decline through 2010 (CARB, 1999).

In September 2004 the California Air Resources Board (CARB) adopted new rules to regulate greenhouse gas emissions from cars and light trucks, including CO_2, methane, nitrous oxide, and hydrofluorocarbons. (Unlike the other substances, hydrofluorocarbons do not typically occur in engine exhaust but are used in vehicle air conditioning systems.) These new regulations are currently scheduled to phase in between the 2009 and 2016 model years. CARB estimates that the new regulations will reduce greenhouse gas emissions from California's light-duty vehicle fleet by 18 percent in 2020 compared to today, and by 27 percent in 2030 (CARB, 2004b).

Automakers have filed a legal challenge to the CARB greenhouse gas regulations, contending that they are, in effect, fuel economy standards and that only Congress has authority to set such standards.

State Authority

While the Clean Air Act precludes states other than California from implementing their own emissions regulations for new vehicles, states are allowed to adopt California regulations in lieu of EPA regulations. A number of states have exercised this important authority. To date, at least ten states have adopted or moved to adopt California's low-emissions vehicle program; nine have adopted California's heavy-duty diesel truck standards for model years 2007 and later; and 13 have adopted California's diesel truck "not-to-exceed" requirements (STAPPA and ALAPCO, 2005b).

As of January 2006, eight states have adopted California's

light-duty vehicle greenhouse gas regulations (Washington Post, 2006). Many of these states have also supported a legal challenge to EPA's determination that CO_2 cannot be regulated under the Clean Air Act, which is being led by Massachusetts. (The D.C. Circuit Court of Appeals ruled against Massachusetts and the other states in the first round of litigation, but it appears unlikely that the case will end there (*Commonwealth of Massachusetts v. EPA*, 2005).)

Despite the restrictions on state regulation of new vehicles, the Clean Air Act does not limit the authority of state and local areas to set standards for existing onroad vehicles. In fact, this responsibility falls primarily to the various states given that EPA's role in regulating existing light-duty vehicles is limited to ensuring compliance with new vehicle standards throughout the defined useful life of the vehicle.

Emissions Control Opportunities

Virtually all cars and light trucks on the road today already rely on catalytic converters ("after-treatment" devices) in the tailpipe to comply with current EPA Tier 1 emissions standards. Improvements in the efficiency of these devices, along with changes to engine design, were required to meet the EPA Tier 2 and California LEVII standards.

However, it is unlikely that the availability of these improved after-treatment technologies and engines will drive a retrofit market for cars and light trucks. Given the relatively high turnover rate of the light-duty fleet, as well as typical ownership patterns, retrofitting or repowering existing light-duty vehicles is not a practical way to achieve significant emissions reductions. This is different from the situation for the heavy-duty diesel fleet, where future emissions regulations have already spurred the development of a retrofit market and where retrofitting existing vehicles is considered a key pollution control strategy (see Chapters 12, 13, and 14).

Most emissions reductions from the light-duty fleet will have to come from fleet turnover, as new vehicles that meet more stringent standards replace older vehicles, and from net reductions in total fuel usage by the sector. As discussed above, natural turnover is expected to produce significant NO_x reductions. However, given that both the EPA and California emissions standards include a range of certification bins, not all new vehicles will have the same emissions levels. This means that there is an opportunity to accelerate emissions reductions from fleet turnover by encouraging early retirement of the oldest, most polluting vehicles and their replacement by low emitting new cars and light trucks. The entry of new vehicles that use alternative, cleaner fuels can also play a role.

In addition, changes in personal behavior and in the transportation system that reduce vehicle miles traveled and/or relieve congestion will also reduce emissions from the light-duty fleet. Strategies can include smart growth development to promote dense, mixed-use communities that allow cycling, walking and easier use of public transit for commuting and other trips, as well as improvements to and expansion of existing public transit infrastructure.

Reducing and enforcing highway speed limits can also reduce fuel usage and emissions from the light-duty fleet, since cars operated above about 60 miles per hour are less fuel efficient.

Cleaner Vehicles

As discussed above, EPA Tier 2 emissions regulations require that the "average" new car produce no more than 0.07 grams per mile of NO_x. However, the cleanest vehicles will be certified to produce only 0.02 grams/mile—over 70 percent less—while some light trucks will be allowed to produce 0.60 grams/mile—850 percent *more*—through model year 2006 (EPA, 2004). Clearly, not every new vehicle will have the same environmental performance.

EPA maintains a list of the most environmentally friendly vehicles based on certification data (Green Vehicle Guide). For the 2006 model year, the Guide includes 28 different gasoline models that attain EPA's highest rating of 9.5 because their NO_x emissions were certified to the lowest level of 0.02 grams per mile in at least one available configuration (EPA, 2005a). These models include small and mid-size sedans, several small wagons, hybrid-electric sport utility vehicles and performance cars.

Not surprisingly, all of these models also achieved an EPA "greenhouse gas score" of at least 7.0—equivalent to a combined city-highway EPA fuel economy rating of 26–29 miles per gallon (EPA, 2005a). Because exhaust emissions are created by the burning of fuel, in general the higher the average fuel economy a vehicle has, the lower the gram per mile emissions will be.

A review of California certification data shows that, as of November 2005, there were at least 71 gasoline-powered 2006 model cars and light trucks that achieve LEVII ULEV emissions standards, and an additional 13 models that achieve LEVII SULEV standards. These include small and mid-size sedans, wagons, small and medium-size sport utility vehicles and performance cars. In addition, 24 models meet the PZEV emission requirements (CARB, 2005b).

Some of these models are hybrid electric vehicles, which achieve better fuel economy by combining a gasoline engine with a small battery pack and one or more electric motors to better manage energy usage throughout the driving cycle.

While ULEV and SULEV certified gasoline vehicles typically cost the same or only marginally more than equivalent higher emitting vehicles, the most popular hybrid car, the Toyota Prius, costs approximately $3,000 more than a comparable gasoline car. Other hybrid electric cars and light trucks can cost as much $6,000 more than the same vehicle with a gasoline engine (DOE, 2005). At least some of this price difference can be recovered from fuel savings over the life of the vehicle and tax breaks.

Cleaner Fuels

While there are a number of alternative formulations of diesel fuel that produce lower emissions than standard diesel (see Chapter 12), the practical alternatives to gasoline are more limited. The only commercially available liquid fuel that can be used as an alternative to gasoline for cars and light trucks is ethanol, an alcohol produced from corn and other agricultural commodities.

Some of the RFG currently sold in a number of cities during summer months, as well as some of the oxygenated gasoline sold in numerous cities in the winter, contains up to 10 percent ethanol added as an oxygenate. While the Energy Policy Act of 2005 removes the oxygenate requirement for RFG, future "standard" gasoline sold in various areas is likely to contain ethanol to meet the mandate for renewable fuel content that is also included in that law.

In some parts of the country, an "E85" blend of 85 percent ethanol and 15 percent gasoline is sold commercially. Standard gasoline engines and fuel systems must be modified slightly to operate on fuel with a high percentage of ethanol and some auto manufacturers produce models with "flex fuel" engines that can operate on standard gasoline, ethanol, or any mixture of the two, including E85.

According to the U.S. Department of Energy (DOE), there are at least ten 2006 models of mid-size sedans, pick-up trucks, mini-vans and SUVs available from the manufacturer with flex fuel engines (DOE, 2005). These vehicles are certified to either LEV or ULEV emissions standards and, according to EPA, can have up to 20 percent lower PM emissions, up to 10 percent lower NO_x emissions and up to 15 percent lower VOC emissions when operated on E85 instead of standard gasoline (EPA, 2002). These vehicles are generally no more expensive than their gasoline equivalents.

Ethanol has 30 percent lower energy content per gallon than gasoline, so the fuel economy (miles per gallon) of a vehicle operated on E85 will be less than the fuel economy of the same vehicle operated on gasoline (DOE, 2005c). According to DOE, in September 2005 the average price of E85 was $2.41 per gallon, about 13 percent less than the average price of gasoline, at $2.77 per gallon (DOE, 2005b). Given this price differential, it would be marginally more expensive (dollars per mile) to operate a flex-fuel vehicle on E85 instead of gasoline.

Engines are also available for cars and light trucks that operate on natural gas. DOE lists several 2005 and 2006 models available from the manufacturer with dedicated natural gas engines and/or bi-fuel engines that can operate on either natural gas or gasoline (DOE, 2005). These dedicated natural gas and bi-fuel engines are certified to either ULEV or SULEV emissions standards and have no evaporative hydrocarbon emissions due to the use of a sealed fuel system.

It is also relatively easy to convert a standard gasoline vehicle to operate on natural gas, although emissions can actually increase if the conversion is not done properly (ICCT, 2005). There are a number of reputable companies that do conversions of new vehicles, primarily for fleet operators. To ensure that actual emissions reductions are achieved, all natural gas vehicle conversions should be done using an EPA- or CARB-certified conversion kit.

The most significant difference between a gasoline and a natural gas vehicle is the fuel system: while gasoline is a liquid, natural gas is a gas, which is stored on the vehicle under high-pressure. Therefore, use of a natural gas vehicle requires access to a high-pressure natural gas fueling station.

Given the cost and limited availability of public natural gas fueling stations, conversion to natural gas is most appropriate for light-duty fleet vehicles that operate within a limited geographic area, such as taxis and utility service vehicles. In some cities it may already be practical to use a natural gas vehicle for private commuting; a more extensive natural gas fueling infrastructure would allow individuals even greater use of natural gas vehicles.

The incremental cost of natural gas cars and light trucks can be $4,500 or more compared to an equivalent gasoline vehicle (DOE, 2005). Natural gas and gasoline engines have similar efficiency, and natural gas is generally less expensive than gasoline for the same amount of energy. According to DOE, in September 2005 the average price of a "gasoline equivalent gallon" of natural gas delivered for use in a vehicle was $2.12, about 23 percent less than the average price of a gallon of gasoline, at $2.77 (DOE, 2005b). Given this price differential, at least some of the incremental purchase cost of a light-duty natural gas vehicle can be offset over the life of the vehicle based on fuel cost savings.

In response to California Tier 1/LEV requirements that mandated the introduction of ZEVs, many auto manufacturers have produced battery-electric vehicles in

limited quantities in the past. Many of these vehicles were technically successful and gained a loyal following among a small number of customers. However, the limitations of current battery technology—these vehicles typically had a range of 100 miles or less per charge—made electric vehicles impractical for many uses. Their additional cost compared to gasoline vehicles also limited their appeal to the general public. All of the major auto manufacturers have since abandoned their electric vehicle programs in favor of developing hydrogen fuel cell technology and producing vehicles that meet California's LEVII PZEV and AT-PZEV emissions standards. New electric vehicles are still available from several small California companies that convert gasoline cars to electric drive, but their utility for private or even fleet customers is limited.

Hydrogen fuel cells produce electricity directly from gaseous hydrogen and oxygen in a low temperature chemical reaction for which virtually the only by-products are water and heat. A fuel cell vehicle is therefore a zero emissions vehicle, producing no PM, NO_x or other harmful pollutants. Every major auto manufacturer is actively working on fuel cell technology, and a number of demonstration vehicles are on the road today. This work is partially funded by the federal and several state governments.

The current generation of fuel cell vehicles is significantly more expensive than gasoline vehicles and also has limitations in terms of durability and range. In addition to improvements in the vehicle technology itself, fuel cell vehicles will require the development of hydrogen production, distribution and retail fueling infrastructure before they can be used widely. Current expectations are that fuel cell vehicles will not replace a significant portion of the light-duty fleet until 2030 or later.

Using Less Fuel

Some cars and trucks use significantly less fuel than others to go the same number of miles. For vehicles with the same level of engine and after-treatment technology, burning less fuel creates fewer exhaust emissions. In general, the smaller and lighter the vehicle, the more fuel efficient it will be.

Engine technology can also affect fuel use: some gasoline engines are more efficient than others, and hybrid electric vehicles are more efficient still. Other vehicle characteristics can also play a role: vehicles with manual transmissions still get 5–10 percent better fuel economy than most automatics, despite recent improvements in automatic transmission design. Cars with all-wheel drive are also less efficient than cars with two-wheel drive (DOE, 2005d).

Public policies related to land use and transportation infrastructure and management also play a role in limiting light-duty vehicle emissions. Shared transportation is more efficient than the use of single-occupancy vehicles, and public transportation is very efficient if well-utilized. Non-motorized transportation modes (walking, bicycling) are non-polluting. Additionally, measures that reduce vehicle miles traveled not only reduce total exhaust emissions from the light-duty fleet, but also reduce direct $PM_{2.5}$ emissions from brake and tire wear. Brake and tire wear accounts for almost half of the direct $PM_{2.5}$ contribution from light-duty vehicles, but it is unaffected by the engine and fuel technologies previously discussed.

Higher traffic speeds (up to about 60 miles per hour) from reduced congestion reduce total emissions. Keeping maximum highway traffic speeds below about 60 miles per hour also limits fuel use, as higher speeds are less efficient. While every car is slightly different, fuel economy generally deteriorates rapidly above about 60 miles per hour. DOE estimates that for every five miles per hour over 60, fuel economy is reduced by 5–7 percent (DOE, 2005d). While average highway speeds have been increasing across the country since the 1990s, it may be that the current high and rising price of gasoline and the resulting increased focus on personal vehicle fuel economy will create a hospitable environment for rolling back highway speed limits.

State and Local Policy Measures

Programs that encourage or mandate the early retirement or replacement of older cars and trucks, sales of the lowest emitting replacement vehicles available and the use of cleaner fuels can have an important impact on air quality.

These types of programs will probably provide the most significant opportunities for states to control and reduce PM and NO_x emissions from the light-duty fleet. Whether these policies are voluntary or mandatory, their common focus is on accelerating the emissions reductions available from fleet turnover. In addition, inspection and maintenance (I&M) programs for light-duty vehicles will continue to be an important tool for the attainment *and maintenance* of improved ambient air quality.

Approaches that improve the overall transportation system and thereby reduce single-occupancy vehicle trips as well as total light-duty vehicle miles traveled will also produce PM and NO_x benefits based on reduced fuel use by the light-duty vehicle sector as a whole. These include programs that promote the use of public transportation, shared transportation and non-motorized modes; that reduce highway speeds; and that enhance traffic management and reduce congestion.

Accelerating Fleet Turnover

While lower emitting vehicles are sometimes no more expensive than similar vehicles with higher emissions, that is not always the case. In addition, there are many factors besides cost that most people weigh when deciding which vehicle to buy, including size, engine power, safety and amenities. Environmental considerations do not always win out over these other factors.

One way to tip the balance is to provide monetary incentives for individual consumers and fleet customers to make the "clean" choice when buying a new vehicle. Incentives can be in the form of a rebate, a tax credit or reduction, or a reduction in vehicle-related charges such as registration fees. Eight states[2] and the federal government currently provide some kind of purchase incentive for environmentally friendly vehicles (DOE, 2005).

In most cases, these purchase incentives consist of a state income tax rebate or a state sales tax exemption for some percentage of the incremental cost of purchasing a new environmentally friendly vehicle or, in some cases, of converting to a cleaner fuel or technology. The rebate percentages range from 30–100 percent of the incremental cost. Connecticut exempts eligible vehicles from all state sales taxes, while New Mexico exempts vehicles from the state excise tax (DOE, 2005).

Depending on the state, eligible vehicles include hybrid electric vehicles, electric vehicles and vehicles operated on natural gas, liquid petroleum gas, or E85. Colorado bases eligibility on EPA emissions standards, with those vehicles certified to LEV standards eligible for a 50 percent rebate, those certified to ULEV standards eligible for a 75 percent rebate and those certified to SULEV standards eligible for an 85 percent rebate. Colorado also doubles the rebate, up to 100 percent of the incremental cost, for new vehicles that permanently replace vehicles that are ten years old or older (meaning that the old vehicles are scrapped, not resold). New Mexico provides its incentive only for hybrid electric vehicles that get at least 27.5 miles per gallon (DOE, 2005).

Most of these programs also provide incentives for the purchase/installation of alternative fueling infrastructure, such as natural gas fueling stations. Most programs are available to individuals and businesses, but some are available only to government-controlled fleets within the state. Several states operate separate incentive programs for private entities and government fleets.

Between 1992 and 2005 the federal government also provided a tax deduction of $2,000 for the purchase of a hybrid electric vehicle. The Energy Policy Act of 2005 provides tax credits for the purchase of fuel cell, hybrid and alternative fuel vehicles, beginning in 2006 and running through 2010. For hybrids, the credit ranges from $250 to over $3,000 per vehicle, depending on how much of the vehicle's power is provided by the electric motor/battery pack instead of the gasoline engine, as well as its rated fuel economy. For alternative fuel vehicles, the credit is based on a percentage of the incremental cost compared to a standard gasoline vehicle.

Some states and cities are also offering individuals nonmonetary incentives to purchase cleaner vehicles. These range from permission to drive eligible vehicles with a single occupant in high-occupancy vehicle (HOV) lanes (Arizona, California, Georgia and Virginia), to exemption from state emissions tests (Maryland), to free parking at street meters and city-owned parking facilities (cities of Los Angeles, San Jose and Santa Monica in California).

Another way to accelerate fleet turnover is to provide incentives for individuals to retire the oldest, most polluting vehicles earlier than they otherwise would. For example, the Bay Area Air Quality Management District (AQMD) in northern California will buy back pre-1985 vehicles for $650 each. Eligible vehicles must be registered and operational—they are taken to a vehicle dismantler registered with the air district, where they are destroyed so that they cannot be resold and put back on the road (Bay Area AQMD, 2005).

California's Bureau of Automotive Repair operates a similar program. It will pay motorists $1,000 for any vehicle that fails a Smog Check emissions inspection. Eligible vehicles must be taken to a registered dismantler to be scrapped (California Department of Consumer Affairs, 2005).

As a complement to these voluntary vehicle retirement programs, CARB is evaluating the feasibility of a voluntary vehicle repair program. Using remote sensing, such a program would identify eligible high emitting vehicles between biennial smog checks. Vehicle owners would be offered a financial incentive to complete necessary repairs (Panson, 2005).

All of the above are voluntary programs. Though no examples of mandatory programs could be identified for purposes of this report, approaches that might accomplish the same goals could include denying registration or imposing higher registration fees or more frequent emissions inspections on older vehicles. This would provide a greater monetary incentive for scrapping and replacing such vehicles. CARB is considering a mandatory requirement to replace or upgrade emissions control equipment (catalyst mufflers, oxygen sensors, evaporative filter canisters) on older vehicles. Because the efficiency

2. These states are California, Colorado, Connecticut, Illinois, Maine, New Jersey, New Mexico and New York. In addition, Vermont provides state income tax credits for manufacturers of electric, hybrid electric and alternative fuel vehicles with operations in the state.

of these components may degrade with age, mandatory scheduled replacements may provide significant emissions benefits. CARB is currently operating a pilot program to determine the actual benefits of such an approach (Panson, 2005)

As discussed in previous sections, California has adopted more stringent new vehicle emissions standards than the Tier 2 EPA standards currently in effect. Under the Clean Air Act, other states may adopt the California standards and several have already done so. Doing so will accelerate emissions reductions from fleet turnover, as virtually all new vehicles that enter the fleet will be cleaner than they would be under EPA standards. States could complement this approach by requiring operators of publicly owned fleets to purchase only ULEV and SULEV certified vehicles.

Inspection & Maintenance

Studies have shown that a relatively small fraction of vehicles can create a disproportionate share of vehicle emissions (Niemeir, 2004). These vehicles may be poorly maintained or misadjusted or have malfunctioning emissions control equipment. Under the Clean Air Act, certain ozone and CO nonattainment areas are required to put I&M programs in place for light-duty vehicles. However, these requirements give the states a good deal of flexibility as to the type of I&M program they adopt. Moreover, even states without a federal I&M mandate may implement a state I&M program where it is beneficial as part of their SIP strategy.

I&M programs are designed to identify high emitters and require vehicle owners to repair them. Inspections can be performed on a regular basis, usually in conjunction with the renewal of vehicle registration and/or a state-mandated safety inspection. Inspections can also be performed randomly using roadside pull overs. Mandatory repair requirements can be enforced by denying vehicle registration and/or by imposing a fine that is waived upon proof of repair.

Thirty-three states plus the District of Columbia currently have a light-duty I&M program in at least one city or county (EPA, 2003). The earliest programs are those in California and Idaho, which were implemented in 1984; the most recent are those in Louisiana and Missouri, which were implemented in 2000.

Some of these programs require only a visual inspection of the vehicle's emissions control system components, but most require a tailpipe emissions test on most older vehicles to identify high emitters. There are a number of different methods of tailpipe testing, including idle-only tests and tests that require the use of a dynamometer to load up the engine for testing. There are also a number of different test cycles that can be used with a dynamometer; currently the most popular are the ASM2, ASM2525 and IM240 test cycles (EPA, 2003).

Starting with the 1996 model year, EPA began requiring that all new vehicles be equipped with an on-board diagnostic system (OBD). These systems use various sensors to identify failures of engine components or systems that are known to increase exhaust emissions. When a component fails, the OBD system lights an indicator on the vehicle dash to show that the engine requires service. Using software provided by the engine manufacturer, a trained mechanic can retrieve detailed information about the failure that can help to diagnose and repair the problem.

These OBD systems can also be used in an I&M program to identify vehicles with emissions-related problems that are potential high emitters, without the need for a tailpipe test. Currently, 16 states use an OBD scan to identify high emitters in their I&M programs for cars in the 1996 model year and newer; these vehicles do not undergo a tailpipe emissions test.

For many years, light duty I&M programs have been considered a key tool for reducing emissions of smog-forming pollutants (NO_x and VOCs), as well as CO emissions, from cars and light trucks. In its most recent program evaluation, California concluded that in 2002 its I&M program resulted in NO_x reductions of 158 tons per day, along with 360 tons per day of CO and 211 tons per day of hydrocarbons (CARB, 2004a). The reduction in NO_x emissions was estimated to be approximately 18 percent of total NO_x emissions from the gasoline fleet in California.

In 2002, a committee of the National Research Council of the National Academies of Sciences undertook an evaluation, funded by the federal government under the Congestion Mitigation and Air Quality grant program, of the effectiveness of pollution reduction projects. It concluded that "well structured inspection and maintenance programs" were the most cost-effective transportation-related strategies among those evaluated for reducing vehicle emissions (National Research Council, 2002).

Transportation System Improvements

Transportation system improvements that have the potential to reduce total vehicle emissions generally fall into two broad categories: (1) changes that result in fewer trips and fewer miles traveled, especially by single- and low-occupancy private vehicles; and (2) changes that reduce roadway congestion, thus speeding up traffic in and around urban areas and reducing idling emissions. In both categories, strategies can include physical

infrastructure changes as well as operational changes and public education campaigns. Reduction in highway speed limits to 55–60 miles per hour and/or better enforcement of existing 55–65 mile per hour limits can also reduce emissions and fuel consumption from the light-duty fleet.

Trip reduction efforts can focus on improving or supplementing existing public transportation options (bus, subway, light rail) in order to provide travelers with better alternatives to using their own vehicles. In addition to adding buses or subway lines, other approaches to improving public transportation include public education/trip planning services, construction of park and ride facilities and provision of a guaranteed ride home (for example, by taxi) for regular riders who have to work late or who, for some other reason, miss the last scheduled bus or train.

Ride-sharing by private individuals also reduces vehicles miles traveled. In addition to the installing of HOV-only lanes on major commuting routes, strategies available to states and localities include providing equipment and services for van pools, as well as operating a ride-share information clearinghouse to connect individuals who are looking for commuting partners.

Strategies that encourage the use of non-motorized transportation modes (walking, bicycling) also reduce vehicle trips. Measures include the installation of sidewalks and bicycling/walking paths, as well as zoning and land use policies that promote denser development and integration of housing and commercial activities.

EPA recently recognized five communities from California to Florida for their innovative approaches to such "Smart Growth" policies. New housing development in these communities includes features that facilitate walking and bicycling, proximity to existing transit hubs, a range of housing types, and integration with work, shopping, and leisure opportunities to foster a greater sense of community and reduce the use of private cars (EPA, 2005d).

Finally, trip reduction efforts can focus on reducing the total number of commuting and other business trips altogether—for example, by encouraging telecommuting and greater use of teleconferencing for business meetings.

Many current trip reduction programs incentivize employers to provide better options for their employees' commutes. EPA and DOT have developed a voluntary program called Best Workplace for Commuters℠. In addition to providing information for employers and employees, this program recognizes companies that meet a "national standard of excellence" by providing a minimum level of commuting benefits for their employees. The program's benefits include a minimum $30 per month in transit or vanpool passes, a minimum payment of $30 per month in lieu of a subsidized parking space and telecommuting options that reduce total monthly commuting trips by 6 percent or more (DOT, 2005).

Federal tax law allows employers to provide up to $105 per month per employee in transit or vanpool benefits without paying any payroll taxes on that amount. Alternatively, employers can allow employees to reserve up to $105 per month for commuting expenses on a pre-tax basis. Either way, both employees and employers save money on taxes, reducing the cost of commuting (DOT, 2005).

The State of Maryland provides an additional state tax credit of up to 50 percent of the cost of employer-provided commuting benefits up to a maximum value of $50 per month per employee. These tax credits can be claimed for the cost of employer-provided transit passes or vouchers, vanpool expenses, parking cash-out benefits or the costs of a guaranteed ride home program (Maryland DOT, 2005).

While these commuting programs are voluntary, California has implemented a mandatory "Parking Cash-Out Law." The law requires certain employers who provide free parking for their employees to allow employees to "cash out" their parking space—that is, give up their parking space in return for a cash payment—as a means of providing additional monetary incentives for employees to walk, bicycle, carpool or take public transportation to work (CARB, 2005a).

While increased use of public, shared and non-motorized modes of transportation can reduce congestion on crowded roadways, there are other, more direct ways to speed traffic. Probably the simplest way is to improve identified intersections to remove bottlenecks. In many places, computerization and better coordination of traffic signals can also dramatically improve traffic flow in specific corridors. Finally, more sophisticated traffic management tools have become available over the last ten years. These tools are usually described under the general term "Intelligent Transportation Systems" or ITS.

All of these strategies can help to reduce emissions of $PM_{2.5}$ and NO_x from cars and light trucks by reducing vehicle travel and overall fuel use.

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Chapter 16

Airports

Introduction

In many ways a microcosm of the cities they serve, airports comprise a wide variety of particulate matter (PM) and PM-precursor emissions sources, including power plants, construction equipment, shuttle buses, passenger vehicles and aircraft. In aggregate, the emissions from a large airport are comparable to those of a power plant or petroleum refinery, although the characteristics of the source are obviously much different.

The dominant source of airport emissions, aircraft are a growing source of nitrogen oxides (NO_x) emissions in the U.S. as a result of steady growth in air travel demand. Projections by the Federal Aviation Administration (FAA) suggest that flights of commercial air carriers will increase by 18 percent from 2002 to 2010 and 45 percent from 2002 to 2020 in the U.S. (EPA, 2003). A significant increase in air pollution emissions has accompanied this steady growth in air travel. Between 1970 and 1990, aircraft NO_x emissions are estimated to have increased by 133 percent (CCAP/NESCAUM, 2003); moreover, despite the adoption of new aircraft engine emissions standards, the general trend of increasing emissions is expected to continue (Schmidt, 2005).

State and local agencies have limited authority to regulate aircraft-related air pollution because the Clean Air Act preempts states from regulating aircraft engine emissions. However, aircraft are not the only source of emissions at airports. Though they account for a smaller share of airport-related emissions, ground transportation and service equipment and airport power facilities have provided opportunities for meaningful pollution reductions and continue to be a focus of emissions control efforts.[1] For example, some airport operators have converted airport ground vehicles from diesel or gasoline to cleaner alternative fuels. In some cases—such as at the Los Angeles and Dallas/Fort Worth airports—federal or state agencies have imposed measures to bring severely polluted areas into attainment with the Clean Air Act's air quality standards or to mitigate expected emissions increases from airport expansion projects.

This chapter discusses sources of emissions at airports,

1. As some of these basic sources of emissions (e.g., boilers and buses) are discussed in other chapters of this report, the reader is referred to these sections for additional information.

the regulatory authority to address these sources, and the control strategies that state and local authorities might consider to reduce PM and PM-precursor emissions as they develop State Implementation Plans (SIPs).

Sector Profile

As indicated above, airports comprise a wide variety of emissions sources, including mobile and stationary sources.

Fossil fuel combustion is the dominant source of airport $PM_{2.5}$, sulfur dioxide (SO_2) and NO_x emissions. Various studies have estimated the emissions contributions of these facilities, focusing on individual components of the airport source category or, more broadly, on the entire scope of airport-related activities.

In a 2003 study, the Center for Clean Air Policy and the Northeast States for Coordinated Air Use Management (CCAP/NESCAUM) developed emissions inventories for three airports located in the Northeast: (1) Logan International Airport in Boston, Massachusetts, which serves roughly 27 million passengers per year; (2) Bradley International Airport in Windsor Locks, Connecticut, which serves roughly 6.3 million passengers per year; and (3) Manchester Airport in Manchester, New Hampshire, which serves roughly 2.8 million passengers per year. The study focused on three sources of airport-related emissions: aircraft main engines, auxiliary power units and ground service equipment.[2] The study excluded from the inventory ground access vehicles and stationary sources.[3]

Table 16.1 summarizes the NO_x emissions estimates developed in the CCAP/NESCAUM study.[4] In all cases, aircraft main engines were found to be by far the dominant source of emissions among the source categories evaluated.

Table 16.1

Airport Emissions Inventories

Source	Annual NO_x Emissions (Tons, 1999)	% NO_x Emissions (1999)
Logan International Airport		
Aircraft	2,644	86%
APUs	145	5%
GSEs	293	9%
Bradley International Airport		
Aircraft	677	84%
APUs	30	4%
GSEs	96	12%
Manchester Airport		
Aircraft	188	82%
APUs	8	3%
GSEs	33	14%

APU = auxiliary power units
GSE = ground service equipment

Source: CCAP/NESCAUM, 2003

Examples of Airport Emissions Sources

Mobile Sources	Stationary Sources
aircraft main engines	utility plants (e.g., power boilers and combustion turbines)
aircraft auxiliary power units	maintenance facilities
ground service equipment (e.g., baggage tugs, belt loaders, pushback tractors)	restaurants/catering facilities
ground access vehicles (e.g., passenger buses, vans, cars)	emergency services
construction equipment	

EPA reports a similar distribution of NO_x emissions between aircraft and ground service vehicles in its 2002 National Emissions Trends inventory (see Table 16.2). Nationwide, aircraft are estimated to have generated 85,616 tons of NO_x emissions in 2002, or less than 1 percent of all transportation-related NO_x emissions (EPA, 2005a). By comparison, airport service vehicles are estimated to have generated 9,000 tons of NO_x emissions in 2002 (EPA, 2005b).

Nationwide, aircraft are estimated to have generated 7,535 tons of SO_2 emissions in 2002 (EPA, 2005a).

There is wide variation in EPA's estimates of direct $PM_{2.5}$ emissions from aircraft. EPA's 2002 draft emissions inventory estimates annual emissions of 14,000 tons (EPA, 2005a). Research projects are currently underway to better characterize aircraft particle emissions and improved data are expected in the coming years (Chou Wey, 2005).

Diesel-powered airport service vehicles are reported to have generated less than 1,000 tons of direct $PM_{2.5}$ emissions in 2002 (EPA, 2005b).

The national pollution inventory does not tell the full

2. Auxiliary power units are small jet engines onboard aircraft that provide electricity to a jet aircraft parked at a terminal gate.

3. Also, the CCAP/NESCAUM study estimates only aircraft emissions that occur below the "mixing height"—that is, only those emissions that are assumed to contribute to ground-level air quality problems.

4. SO_2 and $PM_{2.5}$ emissions were not estimated in the study.

Table 16.2

National Airport Emissions (2002)

Source Category	$PM_{2.5}$		SO_2		NO_x	
	Tons/Year	Percent of Sector Emissions	Tons/Year	Percent of Sector Emissions	Tons/Year	Percent of Sector Emissions
Aircraft	13,964	94%	7,535	84%	85,616	81%
Ground service vehicles	871	6%	1,457	16%	19,960	19%
Total	14,835	100%	8,992	100%	105,573	100%

NR = not reported

a. Rounded to thousands.

Source: Billings, 2004; EPA, 2005a

story. Several urban areas that are in nonattainment for $PM_{2.5}$ have one or more major airports within their airshed. Within these areas, airport-related activities are likely to contribute a higher share of regional emissions.

In a study of ten urban areas, EPA reports a projected increase in the contribution of aircraft to regional mobile source NO_x inventories; specifically, the average aircraft contribution in these areas is expected to increase from 2 percent in 1990 to about 5 percent in 2010 (EPA, 1999). In at least two of the cities studied, commercial aircraft are projected to contribute as much as 10 percent of total regional mobile source NO_x emissions by 2010. This estimate reflects the expectation that aircraft emissions are likely to increase, while emissions from other mobile source categories are projected to decline.

Some studies have developed airport inventories focused on a broader spectrum of sources, such as the *LAX Master Plan EIS/EIR*, which inventories emissions at the Los Angeles International Airport (LAX). The LAX inventory includes aircraft, auxiliary power units, motor vehicles (on-airport) and stationary sources. In 1996, the airport generated total NO_x emissions of 5,175 tons and total SO_2 emissions of 183 tons. According to the inventory, aircraft accounted for 74 percent of NO_x emissions, ground service equipment accounted for 7 percent, auxiliary power units accounted for 1 percent, traffic on airport roadways accounted for 8 percent and stationary sources accounted for 9 percent. In terms of SO_2 emissions, aircraft accounted for 91 percent of the inventory, ground service equipment accounted for 4 percent, traffic on airport roadways accounted for 1 percent and stationary sources accounted 4 percent (Camp Dresser & McKee, 2001).

In sum, while airports may represent a small share of mobile source emissions overall, the anticipated increase in their emissions and their close proximity to large urban centers make them a candidate for pollution control efforts (see Table 16.3).

Emission Control Opportunities

The technical control options for reducing airport emissions are discussed below. Programs and policies are discussed in the sections that follow.

Aircraft and Auxiliary Power Units

Improving aircraft engine technologies and overall aircraft design can reduce fuel consumption and emissions. There are several promising technologies for reducing aircraft emissions, as described by CCAP/NESCAUM (2003). The National Aeronautics and Space Administration (NASA) conducts most of the original research and

Table 16.3

Airport $PM_{2.5}$ Attainment Status

Airport	National Rank[a]	$PM_{2.5}$ Status
Hartsfield Atlanta Intl	1	Nonattainment
Chicago O'Hare	2	Nonattainment
Los Angeles Intl	3	Nonattainment
Dallas/Fort Worth Intl	4	Attainment
Phoenix Sky Harbor Intl	5	Attainment
Denver Intl	6	Attainment
Las Vegas McCarran Intl	7	Attainment
George Bush Intl	8	Attainment
Minneapolis - St Paul Intl	9	Attainment
Detroit Metropolitan	10	Nonattainment
John F Kennedy Intl	11	Nonattainment
Newark Liberty Intl	12	Nonattainment

a. National rank in terms of enplanements (the number of passengers leaving from an airport).

Source: FAA, 2003

development work on new turbine engine technology and has a multi-year program to develop lower-emitting jet engines. One target is to introduce an engine that can emit 70 percent less NO_x than the limits set by current international standards.[5] Aircraft engine manufacturers have also been working to develop engines with lower NO_x emissions while improving fuel efficiency. However, these approaches are not the focus of this report because the Clean Air Act preempts states from setting their own aircraft emissions standards.

Changing aircraft operations represents a second category of options for reducing emissions. The options include minimizing the use of reverse thrust on landing, using lower engine power at takeoff, using a single engine when taxiing and shutting off main engines while idling. These options are also not emphasized in this report because, in some cases, safety concerns may limit their application. Also, single engine taxi is already widely practiced by U.S. airlines. Delta, for example, reports that single engine taxi has saved the company about 6.3 million gallons of fuel during the first eight months of 2005 (Delta Airlines, 2005).

A third option for reducing aircraft-related emissions is to retrofit airport gates with power and preconditioned air to reduce the use and emissions associated with aircraft auxiliary power units and ground power units. Gate-based power and air conditioning hook-ups have become increasingly common at airports. These hook-ups eliminate or reduce the use of higher-emitting on-board auxiliary power units powered by jet fuel and diesel-powered mobile ground power units. Gate electrification requires an up-front capital investment but, once installed, the system produces fuel and labor savings that typically result in a relatively short payback time of less than two years (CCAP/NESCAUM, 2003). Because of the economic benefits of gate electrification, many larger airports have already adopted the technology. Increased use of these systems could further reduce emissions.

Ground Service Equipment

Ground service vehicles—including baggage tugs, belt loaders and aircraft pushback tractors—are used in a variety of applications, including aircraft fueling, transporting cargo to and from aircraft, loading cargo, transporting passengers to and from aircraft, baggage handling, lavatory service and food service.

As for other mobile sources, there are three basic approaches for reducing emissions from diesel- or gasoline-powered ground service equipment: use a cleaner engine, add a retrofit device or use a cleaner fuel.

5. The 70 percent NASA target is in relation to the standards adopted at the second meeting of the International Civil Aviation Organization's (ICAO) Committee on Aviation Environmental Protection (CAEP), which became effective in 1996.

Cleaner Engines. New nonroad diesel engines certified to the latest EPA standards emit 30–60 percent less NO_x and PM than the unregulated engines built a few years ago. (See Chapter 14, *Nonroad Equipment*, for a discussion of diesel engine standards.) These reductions have come from changes in diesel engine design, most notably better control of lubrication oil, higher fuel injection pressures and the addition of electronic fuel control. In some cases, particularly Tier 3 engines with electronic fuel control, lower emissions have been accompanied by improved fuel efficiency. Removing older engines from service and replacing them with new engines can therefore have a large emissions benefit.

Retrofit Devices. The second basic option for reducing emissions from ground service equipment is to retrofit existing diesel equipment with oxidation catalysts and particulate filters to reduce PM emissions. The more stringent standards that EPA has adopted for nonroad diesel equipment will not take effect for many years and will not apply to the fleet of existing vehicles. Diesel retrofit technologies may be an option for ground service equipment depending on the duty cycle of the vehicle and the availability of controls. Diesel retrofit technologies are discussed in Chapter 12, *Diesel Engine Technologies*.

Cleaner Fuels. Another option for reducing emissions from diesel-powered ground service equipment is to rely on ultra-low-sulfur diesel fuel—that is, fuel with sulfur levels below 15 parts per million (ppm) by volume—prior to the applicable regulatory compliance deadline for nonroad diesel engines in 2010. As discussed in the section of this chapter on regulatory authority, EPA's nonroad clean diesel regulations require the use of ultra-low-sulfur diesel fuel beginning in 2010. Switching a conventional vehicle to lower sulfur fuel will reduce PM and SO_2 emissions to some extent; however, the primary advantage of switching to a lower sulfur fuel is that after-treatment devices, such as diesel particulate filters, can be used with the fuel. Lower sulfur fuel improves the performance and durability of advanced control technologies. In California, state regulations will require ultra-low-sulfur diesel fuel for onroad and nonroad applications starting in 2006, several years before the federal requirements. Chapter 12, *Diesel Engine Technologies*, provides more information on the use of low-sulfur diesel fuel and after treatment control options.

Another option for reducing emissions is to replace diesel- and gasoline-powered ground service equipment with equipment powered by alternative fuels such as electricity, compressed natural gas (CNG) and liquid propane gas (LPG). The CCAP/NESCAUM study discussed previously reports a cost of about $1,700 for converting belt loaders and baggage tractors to LPG; the cost of LPG conversion for aircraft push back tractors is estimated at approximately $2,700 (CCAP/NESCAUM, 2003).

According to the same study, converting gasoline-powered ground service equipment to CNG costs approximately $5,000 (CCAP/NESCAUM, 2003).

The primary barrier to converting airport service equipment to alternative power sources, like CNG and electricity, is the initial capital investment required to install the necessary infrastructure. CNG and LPG both require refueling and storage infrastructure in addition to the purchase of new alternative-fueled vehicles or the conversion of existing gasoline and diesel vehicles. CCAP/NESCAUM (2003) report a cost of $750,000 for a 50-vehicle, fast-fill CNG refueling station. Electric vehicles require battery-charging systems. Government grants can help mitigate these costs. Once made, the investment yields significant pollution reductions in addition to other benefits such as lower fuel costs, lower maintenance costs and longer engine life.

Electric-powered ground service equipment has been a popular option for reducing emissions, particularly in warmer regions, where cold weather does not affect battery performance. For example, the Burbank Airport in California has invested $1.3 million in a system of 42 battery chargers for electric ground service equipment at all 14 of its gates (BGP Airport Authority, 2005). By the end of 2005, the airport authority aims to have 70 percent of its ground service fleet powered by electricity, based on a commitment by Southwest Airlines to convert most of its fleet by the end of 2005. According to CCAP/NESCAUM (2003), electric ground service equipment is initially more expensive than diesel equipment—for example, an electric baggage tractor costs 27 percent more than a diesel version. However, these additional costs are largely offset by reduced fuel and maintenance costs over the life of the machine.

Based on figures reported by Southern California Edison, it costs $3.97 per day to charge an electric-powered belt loader, compared to $8.18 to fuel a gasoline-powered belt loader (Reinhardt, 2005). Sources also report reduced maintenance costs for electric vehicles, which eliminate the need for tune-ups, engine overhauls, exhaust system replacements, transmission maintenance, and oil and other engine fluid changes (Reinhardt, 2005).

Further information on the options for reducing emissions from ground service equipment is available from the FAA Voluntary Airport Low Emissions (VALE) Program guidance document (FAA, 2004). VALE provides funding for low-emission vehicles, refueling and recharging stations, gate electrification and other airport air quality improvements at commercial service airports in nonattainment and maintenance areas.[6]

Ground Access Vehicles

Passenger transport vehicles, including shuttle buses and terminal transport buses, provide an additional opportunity for reducing NO_x and $PM_{2.5}$ emissions. In the case of airport buses, the options are the same as for ground service vehicles: use a cleaner engine, use a cleaner fuel or add a retrofit device. Anti-idling limits are another important strategy for reducing airport shuttle bus emissions.

Removing older engines from service and replacing them with new engines can reduce both NO_x and PM emissions. EPA regulations have reduced NO_x and $PM_{2.5}$ emissions

6. The Vision 100-Century of Aviation Reauthorization Act of December 12, 2003 (P.L. 108-176) directed the FAA to establish a national program to reduce airport ground emissions at commercial service airports located in nonattainment and maintenance areas. The VALE program has its origins in this law.

Table 16.4

Conversion Options

Measure	Emission Reduction			Cost-Effectiveness
	NO_x	CO	VOC	
CNG/LPG replacement of diesel	65%	30%	30%	$1,000–$3,000 per ton of VOC/CO/NOx combined
CNG/LPG conversion from gasoline	25%	45%	50-70%	Cost of conversion is more than covered by fuel cost savings over several years

CNG = compressed natural gas
LPG = liquid propane gas

Source: CCAP/NESCAUM, 2003

Table 16.5

Cost-Effectiveness of Converting Gasoline and Diesel Ground Service Equipment to Electricity

Equipment	Fuel Type	Cost-Effectiveness ($/Ton of NO_x)
Baggage tractor	Gasoline	1,900
	Diesel	5,800
Belt loader	Gasoline	Cost savings
	Diesel	Cost savings
Aircraft tug	Gasoline	Cost savings
	Diesel	2,800

Source: CCAP/NESCAUM, 2003

from new onroad diesel truck and bus engines by 80 percent and 90 percent, respectively, since 1988.

In some cases, airport authorities have converted or replaced shuttle buses with lower-emitting, alternative-fueled vehicles. The fact that airport shuttles follow a fixed route and return to a central location can facilitate the conversion to an alternative fuel. For example, at LAX airport in Los Angeles, all of the courtesy shuttle fleet is powered by alternative fuels (FAA, 2005). At Logan Airport in Boston, the Massachusetts Port Authority (Massport) began converting its shuttle buses in 1995. Now all of Massport's fleet of 33 airport shuttle buses use CNG (Massport, 2005a).

Idling diesel buses can generate excess emissions while waiting for passengers. In Massachusetts, anti-idling regulations prohibit buses from idling for a foreseeable period of time in excess of five minutes. In 2002, EPA sought penalties against two shuttle bus operators at Logan Airport for excessive idling of diesel buses. In one case, a bus was observed idling for more than an hour (EPA, 2002b).

The VALE program described above also provides funding for projects aimed at reducing emissions from airport ground access vehicles.

Chapter 13, *Diesel Trucks and Buses*, discusses pollution control strategies for buses.

Stationary Sources

A combination of on-site power generation and electricity from the grid supply the substantial energy needs of airports. Several options are available for reducing emissions from on-site generating facilities, including control technology installations, equipment upgrades or replacements and fuel switching. The Dallas-Fort Worth International Airport, for example, eliminated a substantial share of its stationary source NO_x emissions by replacing two aging boilers with newer technology (Bergman, 2005). Specifically, the new boilers, fired by natural gas, use TODD Rapid Mix Burners in conjunction with flue gas recirculation to achieve NO_x emission rates of less than 10 ppm (Hodapp, 2005). Options for reducing emissions from power generating technologies are discussed in Chapter 5, *Boiler Technologies*, and Chapter 6, *Industrial and Commercial Boilers*.

Regulatory Authority

Aircraft Engines

Section 233 of the Clean Air Act preempts states from regulating aircraft engine emissions. States are limited to petitioning EPA and the U.S. Department of Transportation to adopt stricter emissions standards for aircraft engines (CCAP/NESCAUM, 2003).

Section 231 of the Clean Air Act directs EPA to issue emissions standards for aircraft engines to address any pollutant that EPA reasonably believes to endanger public health or welfare. Based on this authority, EPA in 1997 promulgated NO_x and CO emissions standards for newly manufactured gas turbine engines consistent with those established by the Committee on Aviation and Environmental Protection (CAEP) of the International Civil Aviation Organization (ICAO).[7] The ICAO establishes minimum standards that member nations are expected to meet, but this does not preclude a nation from establishing more stringent standards (CCAP/NESCAUM, 2003). That said, national standards generally follow the standards adopted by the ICAO. Also, most engine models already meet or exceed the ICAO standards at the time they are adopted.

On November 9, 2005, EPA signed the Final Rulemaking for Aircraft Engine Emission Standards amending the existing NO_x standards for *newly certified* commercial aircraft engines (i.e., new engine models). These new standards are equivalent to the NO_x emission standards of the ICAO adopted in 1999 for implementation beginning in 2004.[8] In March 2005, the ICAO approved new NO_x standards that are 12 percent more stringent than the previous levels agreed to in 1999. The new standards would apply to newly certified engines beginning in 2008. EPA has yet to propose standards that would align U.S. standards with these new international standards.

Nonroad Vehicles

As discussed in Chapter 4, the Clean Air Act gives EPA primary authority to regulate emissions from *new* nonroad equipment. California has authority to adopt its own emissions standards for some, but not all, *new* nonroad equipment, as long as the standards are at least as stringent as federal standards. States can adopt California or federal standards; at present, however, California standards for heavy-duty nonroad vehicles largely track federal standards. States are limited to California standards for existing nonroad vehicles. Also, the regulation of certain nonroad equipment that may be used at an airport is fully preempted under §209(e) of the Clean Air Act. Specifically, no state—including California—may set standards for new engines used in construction and farming equipment that are less than 175 horsepower in size.

7. ICAO is a United Nations intergovernmental body responsible for worldwide planning, implementation and coordination of civil aviation. CAEP coordinates the development of international standards for aircraft engines.

8. These standards are known as the CAEP/4 standards.

The federal emissions standards that apply to ground service equipment vary depending on vehicle characteristics (e.g., engine power rating and model year). In general, ground service equipment is categorized as nonroad equipment fueled by either gasoline or diesel fuel.

For regulatory purposes, nonroad engines are those used in mobile equipment that is not intended for use on public roadways. Therefore, the definition of "nonroad" is based primarily on mobility or portability. Despite the fact that it may be virtually identical to an engine used in a piece of mobile equipment, any diesel engine that will stay in the same place for 12 months or more—for example, to power a stationary generator or pump—is regulated as a stationary emissions source (DieselNet, 2005).

Nonroad Diesel. EPA has regulated emissions from new nonroad diesel engines since 1996, with progressively stricter standards required of later model year engines (the standards are known as Tier 1, Tier 2, Tier 3 and Tier 4). The most stringent Tier 4 emissions requirements are being phased in beginning in 2008 on the basis of vehicle horsepower rating. Standards for larger nonroad diesels, like those used at airports, generally apply in 2011 or later. See Chapter 14, *Nonroad Vehicles*, for a complete discussion of EPA nonroad diesel emissions standards.

Table 16.6

Standards for Low-Sulfur Diesel Fuel

Category	Maximum Sulfur Limit (ppm)			
	2006	2007	2010	2012
Highway	15	15	15	15
Nonroad	3000[a]	500	15	15
Marine/Rail	3000[a]	500	500	15

a. 3000 ppm is an approximate value representing uncontrolled emission levels.

Source: Dieselnet, 2005

At the same time that tighter engine emissions standards are being phased in, EPA regulations will be limiting the sulfur content of diesel fuel. As summarized in Table 16.6, ultra-low-sulfur diesel fuel (i.e., less than 15 ppm sulfur) will not be required for use in nonroad applications until 2010. It will, however, be commercially available prior to this date for use in highway vehicles. The availability and use of low-sulfur diesel will allow engine manufacturers to use increasingly advanced clean engine technologies, facilitating compliance with the Tier 4 emissions requirements.

Table 16.7

Standards for Large Spark Ignition Engines

Tier (year)	HC + NO_x	CO
Tier 1 (2004)	4.0 g/kW-hr	50 g/kW-hr
Tier 2 (2007)	2.7 g/kW-hr	4.4 g/kW-hr

g/KW-hr = grams per kilowatthour

Source: EPA, 2002

Nonroad Gasoline. In 2002, EPA adopted emissions standards for several types of previously unregulated nonroad engines, including spark-ignition nonroad engines powered by gasoline, LPG or CNG, such as airport baggage tractors (EPA, 2002a). The standards apply to newly manufactured engines and therefore have an effect only as turnover occurs and older engines are replaced with new ones. Tier 1 standards for large spark ignition vehicles are effective beginning in 2004. Tier 2 standards are effective beginning in 2007.

Onroad Vehicles

Ground access vehicles, like ground service equipment, are regulated primarily by federal motor vehicle emissions standards, which vary according to vehicle type.

Light-Duty Vehicles. Ground access vehicles include passenger vehicles, such as cars and light trucks, that are subject to EPA or California emissions standards.

As with new nonroad vehicles, the Clean Air Act gives EPA primary authority to regulate emissions from *new* onroad vehicles. California has authority to adopt its own emissions standards (as long as they are at least as stringent as federal standards), and other states can adopt either the California or the federal standards. Unlike the situation with nonroad vehicles, states and local areas do have the authority to set their own standards for *existing* onroad vehicles. See Chapter 15, *Light-Duty Cars and Trucks*, for a more complete discussion of the standards applicable to light-duty vehicles.

Buses and Shuttles. As with light-duty vehicles, the Clean Air Act gives EPA primary authority to regulate emissions from *new* heavy-duty onroad vehicles (e.g., trucks and buses). California has authority to adopt its own emissions standards for these vehicles (as long as they are at least as stringent as federal standards), and other states can adopt either the California or the federal standards. California standards for new heavy-duty onroad vehicles largely track federal standards. States and localities are free to set their own standards for *existing* onroad vehicles. The federal emissions standards for buses are addressed in Chapter 13, *Diesel Trucks and Buses*.

General Conformity

The general conformity requirements of the Clean Air Act require airports in some areas to limit or offset their emissions under certain circumstances. Section 176(c) of the Clean Air Act prohibits federal entities from taking actions in nonattainment or maintenance areas that do not conform to the applicable SIP for attaining and maintaining National Ambient Air Quality Standards.[9]

If a project is forecast to generate more than a *de minimis* increase in emissions and this increase is not reflected in the state's air quality plan, then the project must undergo a conformity assessment. As part of this process, the project authorities will evaluate options for mitigating the impacts of the project. This might include the implementation of mitigation measures, or the purchase of emissions offsets. For example, airport operators have used emissions reductions from ground service equipment to offset the emissions increases expected from airport expansions.

State and Local Policy Measures

As outlined above, a number of control options are available for reducing emissions from airports. The discussion that follows outlines policy approaches that state and local governments can use to mandate or encourage these strategies.

Aircraft Engines

Section 233 of the Clean Air Act preempts states from regulating aircraft emissions; however, there are some limited options that a state can pursue. First, states can petition EPA and the Department of Transportation to adopt stricter emissions standards for aircraft engines. Second, some have suggested that states may have the authority to regulate aircraft ground-level activities (e.g., limiting aircraft idling times), provided they do not impinge on the safety or mobility of the aircraft (CCAP/NESCAUM, 2003). Opponents of such a policy are likely to challenge this authority based on a preemption argument.

Ground Service Equipment

Several examples exist of state environmental agencies negotiating voluntary agreements with airport authorities and air carriers to reduce emissions from ground service equipment. For example, the Texas Commision on Environmental Quality (TCEQ) negotiated a voluntary agreement to reduce NO_x emissions in the Dallas/Fort Worth nonattainment area. As part of the agreement, the air carriers agreed to reduce ground service equipment NO_x emissions by 75 percent relative to 1996 levels (CCAP/NESCAUM, 2003). TCEQ also negotiated a voluntary agreement with Continental Airlines, Southwest Airlines and the City of Houston to reduce ground service equipment NO_x emissions in the Houston-Galveston-Brazoria nonattainment area (CCAP/NESCAUM, 2003).

As part of the modernization plan for LAX, community-based organizations negotiated an agreement to convert all diesel-powered ground service equipment to cleaner fuel alternatives. The agreement also requires all diesel construction equipment to be retrofit with advanced pollution control systems and to use low-sulfur diesel fuel (LAX Coalition, 2004).

Ground Access Vehicles

In general, state and local governments have no special authority to regulate cars and buses at airports, beyond their authority (or limited authority) to regulate any car or bus. However, as the proprietor of an airport, a state does have the authority to control vehicle use at an airport, including idling and parking. For example, a state can control traffic flows and the design of the airport to increase reliance on public transportation, thereby reducing personal vehicle use. A state can also require the use of alternative fuel or low-emitting vehicles in the vehicle fleets that operate at a publicly owned airport. For example, the South Coast Air Quality Management District (AQMD) requires airport fleet operators providing shuttle bus services to purchase or lease alternative-fueled vehicles when adding or replacing transit vehicles (South Coast AQMD, 2000).

Multi-Source Strategies

The airport bubble concept is a possible strategy for reducing overall airport emissions (Schmidt, 2005). As with a cap-and-trade program, the airport bubble would cap multiple sources of emissions. For example, the cap could include all of the aircraft at the airport, or aircraft, ground service equipment and stationary sources. The airport operator and other responsible entities would then implement a range of strategies to ensure that overall emissions do not exceed the cap. The cap could be established as an absolute limit on emissions (e.g., tons per year) or on a rate basis (e.g., tons per passenger).

For example, the operator of Logan Airport, Massport, has established a cap on airport NO_x emissions: any emissions increases that result from airport activities must be offset by on-airport emissions reductions, by emissions reductions near the airport or by the purchase of emission credits. Under the auspices of the Logan Air Quality Initiative, Massport has committed to limiting airport NO_x emissions at or below 1999 modeled levels beginning in 2001. In 2003, airport emissions had declined by more

9. Transportation conformity refers to actions approved or funded by the Federal Highway Administration or the Federal Transit Administration. General conformity refers to projects approved or funded by other federal agencies. Airports are generally subject to the general conformity process.

than 30 percent from the baseline levels. However, most of this decline is attributable to a drop in air travel, and the associated reduction in aircraft emissions, following the terrorist attacks of September 11, 2001 (Massport, 2005b).

Congress introduced legislation in 2005 to direct EPA to study the feasibility of applying airport bubbles as a means of reducing emissions (*Right to Know About Airport Pollution Act of 2005*). However, the bill was not passed before the 2005 session ended.

Another option for policymakers to consider is a revenue-neutral fee-based system. Aircraft are subject to landing fees at most airports to cover the costs of airport services. A fee-based system, designed to reduce emissions, would tie the level of the fee to the emissions performance of the aircraft. This charge would create an economic incentive for air carriers to rely on lower-emitting aircraft. In order to encourage reductions from ground service equipment, the fee could be extended to cover their emissions, or the program could allow air carriers to offset their aircraft emissions with reductions from their service equipment fleets. This approach has been used in Switzerland and Sweden (CCAP/NESCAUM, 2003).

The 2003 CCAP/NESCAUM study provides an overview of the legal issues that would need to be considered if a state were to consider imposing a bubble or landing fee program to reduce airport emissions.

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Chapter 17

Marine Ports

Introduction

Marine ports include a wide variety of fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursor emissions sources, including material-handling equipment, tractor-trailer trucks, locomotives, harbor vessels and ocean-going ships. The many sources of air pollution at a large port can generate significant quantities of air pollution.

There are more than 300 commercial sea and river ports in the U.S., which serve as transfer points for moving a wide variety of goods between ships and truck or rail transport (AAPA, 2005).

Most ports are really a collection of multiple marine terminals, each with piers and docks, as well as associated land-side warehouses and facilities. Most ports are controlled by a public sector agency—often a quasi-public port authority with some independence—but ultimately the real control may rest with elected officials from surrounding jurisdictions. The port authority usually owns the land on which the terminal facilities sit and may also own and operate the terminals, thereby controlling all port operations. More often, the port authority simply leases space to private terminal operator companies, which own and operate the dockside equipment and storage facilities and are directly responsible for loading and unloading ships. In these circumstances, the port authority has only indirect control of port operations through its contractual relationships with the terminal operators.

The transportation of goods dominates port activities; however, cruise ship operations are also significant at some ports, such as at the ports of San Francisco, California and New York Harbor (which serves New York City and parts of New Jersey). Passenger ferries can also account for a significant share of harbor traffic in the waters around some ports, such as those in Seattle, Washington and New York Harbor.

Marine terminals are host to a significant, concentrated volume of truck and locomotive traffic with its attendant diesel exhaust emissions. The transfer of goods from ships to trucks and locomotives involves a wide variety of mobile equipment, such as off-road cranes, fork trucks and port straddle carriers. Much of this equipment is also powered by diesel engines. Although marine ports

Table 17.1

Largest U.S. Marine Ports

Marine Port	Rank	Cargo Volume[a]	$PM_{2.5}$ Status
South Louisiana, LA	1	216.4	Attainment
Houston, TX	2	177.6	Attainment
New York / New Jersey	3	134.5	Nonattainment
Beaumont, TX	4	85.9	Attainment
New Orleans, LA	5	85.0	Attainment
Huntington, WV	6	81.1	Nonattainment
Corpus Christi, TX	7	72.0	Attainment
Long Beach, CA	8	67.9	Nonattainment
Baton Rouge, LA	9	60.6	Attainment
Plaquemines, LA	10	59.1	Attainment
Texas City, TX	11	55.2	Attainment
Los Angeles, CA	12	52.2	Nonattainment

a. Ranked by cargo volume, measured in millions of tons per year

Source: American Association of Port Authorities, 2005a; EPA, 2005a

are home to numerous land-side emissions sources, the dominant source of nitrogen oxides (NO_x) and PM emissions are marine vessels—both the ocean-going ships that transport goods and the harbor craft (tug boats, work boats) that service them—all of which are powered by diesel engines. Approximately 90 percent of the PM emissions from these nonroad diesel engines are $PM_{2.5}$ (EPA, 2005d). In addition, most marine ports also have stationary emissions sources (for example, from vessel painting or repair).

The exhaust emissions from ocean-going ships do not stop when the ships reach the dock. Virtually all of these large vessels have significant "hotel" loads (for lighting, pumps, material-handling equipment, etc.) that must be supplied while the ship is docked. Typically, these loads are served by on-board auxiliary diesel engines coupled to generators that produce electricity. Some ships (particularly cruise ships) also use a significant amount of steam for climate control and other services while at the dock. On-board diesel-fired boilers usually produce this steam.

Since 1970, the value of trade goods shipped through U.S. ports has increased on average more than 10 percent annually (AAPA, 2005). Since 1980, the number of cruise passengers has grown by more than 8 percent per year (Cruise Lines International Association, 2005). Steady growth in shipping and cruise traffic has caused a significant increase in air emissions in and around the ports, and these trends are likely to continue. The American Association of Port Authorities projects that the volume of international trade through U.S. ports will nearly double between 2004 and 2020 (AAPA, 2005).

This chapter discusses sources of emissions at marine ports, relevant regulatory authority for addressing these sources, and control strategies available to state and local authorities to reduce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions as they develop State Implementation Plans (SIPs).

Sector Profile

Marine ports comprise a wide variety of emissions sources, including both mobile and stationary sources.

- **Mobile Sources** include marine vessel main and auxiliary engines, dockside nonroad support equipment (cranes, fork trucks, etc.), heavy-duty onroad trucks and locomotives.
- **Stationary Sources** include maintenance facilities, warehouses, internal combustion engines and combustion turbines.

The California Air Resources Board (CARB) developed a PM_{10} emissions inventory for mobile sources at the Ports of Los Angeles and Long Beach as part of a draft diesel PM exposure assessment (CARB, 2005e). As shown in Table 17.2, the inventory apportions total mobile source PM

emissions into three categories of marine vessel emissions, as well as three categories of land-based emissions.

As the table indicates, marine vessels account for more than 86 percent of all mobile source PM emissions at these ports. The main propulsion engines of ocean-going vessels transiting in and out of the port are the dominant source of emissions—accounting for over 50 percent of the total. Emissions from auxiliary engines, which operate while ships are docked, and from commercial harbor craft are also significant. In terms of land-based mobile sources, cargo-handling equipment is the biggest contributor at these ports—producing almost three times as much PM as onroad trucks and locomotives combined. This information is presented for illustrative purposes—the relative land-side emissions inventory at other ports may be different, depending on how much of the existing material-handling equipment has been electrified, on the split between rail and truck traffic to move goods in and out of the port, and other issues. However, marine emissions almost certainly dominate the PM emissions inventory at most ports.

Nationwide, marine vessels operating in ports and riverways are estimated to have generated 1.1 million tons of NO_x emissions in 2002, approximately 8.9 percent of total national transportation-related NO_x emissions. EPA also estimates that marine vessels produced 73,541 tons of direct $PM_{2.5}$ emissions in 2002, approximately 16.8 percent of national transportation-related direct $PM_{2.5}$ emissions (EPA, 2005d).

Perhaps more important, over 30 of the largest U.S. ports are located in areas that are in nonattainment for $PM_{2.5}$ or ozone, or both. In these locations, marine port activities may contribute a higher share of regional emissions than would be indicated by the national inventory.

For example, the total PM produced by mobile sources at the ports of Long Beach and Los Angeles in 2002 (see Table 17.2) equals approximately 25 percent of the total PM produced by all mobile sources in the Los Angeles metropolitan area (CARB, 2005c).

Efforts to reduce emissions of $PM_{2.5}$ and $PM_{2.5}$-precursors from ports should focus on reducing diesel exhaust emissions from marine vessels. Although states have limited ability to reduce emissions from ocean-going ships, they do have numerous opportunities to reduce emissions from commercial harbor craft and marine auxiliary engines. Efforts to reduce land-based emissions should focus on port-based nonroad material-handling equipment and on the onroad trucks and locomotives that haul goods into and out of the port.

Table 17.2

PM Emissions from Ports of Long Beach and Los Angeles in 2002

Source	PM Emissions	
	Tons	% of Total
Marine Vessels		
Ocean-going vessels	942	53.5%
Auxiliary engine-hoteling	343	19.5%
Commercial harbor craft	244	13.9%
Subtotal – ships	1,529	86.9%
Land-based Equipment		
Cargo handling equipment	172	9.8%
Heavy-duty trucks	41	2.3%
Locomotives	18	1.0%
Subtotal – land based	231	13.1%
Total	**1,760**	**100%**

Source: CARB, 2005e

Regulatory Authority

As discussed in Chapter 4, the Clean Air Act gives EPA primary authority to regulate emissions from all *new* vehicles, both onroad and nonroad. Generally speaking, California has authority to adopt its own emissions standards for these sources, as long as they are at least as stringent as federal standards. Other states can adopt either the California or the federal standards for new vehicles. Current California standards for new onroad trucks and most new heavy-duty nonroad equipment largely track federal standards.

The rules are different for existing vehicles. All states and localities are free to set their own standards for *existing onroad* vehicles. The same is *not* true of *existing nonroad* equipment—for these vehicles, the Clean Air Act allows California to set its own standards and other states can adopt these standards.

That said, the Clean Air Act fully preempts all state and local agencies from setting emissions standards for certain categories of nonroad equipment. Specifically, state and local agencies are fully preempted from setting standards for new engines smaller than 175 horsepower that are used in either (a) construction equipment or vehicles or (b) farm equipment or vehicles. The same preemption applies to new locomotives or new engines used in locomotives.

For the emissions sources of greatest concern at marine ports—ocean-going ships—the reach of federal preemption is somewhat less clear. This is the case for

a number of reasons: (1) the majority of emissions from ocean-going ships calling on U.S. ports are foreign-flagged; (2) EPA regulates NO_x emissions (only) from ships under international treaty; and (3) together with the Clean Air Act, the Ports and Waterways Safety Act regulates these vessels. That said, it can be argued that state authority to regulate emissions from existing ocean-going ships extends to the right to adopt any emissions regulations that California adopts. However, states are preempted from setting their own emissions standards for these ships.

Generally speaking, EPA has one regulatory scheme for onroad trucks (discussed in more detail in Chapter 13, *Diesel Trucks and Buses*) and three separate regulatory schemes for nonroad diesel engines (see Chapter 14, *Nonroad Equipment*): one for marine engines, one for locomotive engines and one for all other engines (agricultural, industrial, construction, material-handling, etc.). We discuss each of these separately.

For regulatory purposes, nonroad engines are those used in mobile equipment that is not intended for use on public roadways. Therefore, the definition of "nonroad" is based primarily on mobility or portability. Despite the fact that it may be virtually identical to an engine used in a piece of mobile equipment, any diesel engine that will stay in the same place for 12 months or more—for example, to power a stationary generator or pump—is regulated as a stationary emissions source (DieselNet, 2005).

Marine Engines

For regulatory purposes, marine engines are divided into three categories based on size (cylinder displacement). The smallest, Category 1 engines, are very similar to the diesel engines used in onroad trucks and nonroad land-based vehicles. Personal vessels and some small commercial vessels, such as tugboats and other harbor craft, use these engines for propulsion power. Larger commercial vessels also use them as auxiliary engines. Category 2 engines, which are usually larger than 1,000 horsepower, are virtually identical to locomotive engines and are used for propulsion and auxiliary power in large commercial vessels.

The largest, Category 3 marine engines, are unique. Large ocean vessels, such as cruise ships, freighters and tankers, use these engines—which range in size from 3,000 to 100,000 horsepower—for propulsion power.

EPA first regulated new marine engines beginning with model year 2004. The standards for Category 1 and 2 engines are currently significantly stricter than the standards for Category 3 engines (DieselNet, 2005). The international nature of ocean-going vessels complicates efforts to regulate their emissions absent international agreements. Current EPA NO_x standards for Category 3 marine engines are equivalent to the limits in Annex VI to the International Convention for the Prevention of Pollution from Ships (also known as the MARPOL Convention), which was negotiated under the auspices of the International Maritime Organization (IMO). Neither EPA nor the MARPOL Convention sets limits on other pollutants from these engines (DieselNet, 2005).

To date, EPA has not proposed tightening these emissions standards, as it has for other nonroad engines. However, the current administration, through the State Department, has supported MARPOL Annex VI and has indicated a desire to further tighten NO_x standards and apply PM limits in future IMO negotiations (Argus, 2005).

See Chapter 14, *Nonroad Equipment*, for a more complete discussion of emissions standards for diesel marine engines.

Material-handling Equipment

EPA emissions standards for nonroad diesel engines have historically lagged behind standards for onroad engines. While the Agency began to regulate onroad diesel engine carbon monoxide (CO), NO_x and hydrocarbon (HC) emissions in model year 1974 (PM was added in 1988), it has regulated emissions from nonroad diesel engines only since model year 1996.

EPA has progressively tightened emissions control requirements for later model year nonroad engines through a succession of standards known as Tier 1, Tier 2, Tier 3 and Tier 4. The Tier 4 standards take full effect in 2016 and will introduce general parity between onroad and nonroad engines for the first time. However, most of the nonroad equipment currently in use, including that used at most ports, is still powered by unregulated or only marginally regulated engines.

Chapter 14 provides a more complete discussion of emissions standards for nonroad diesel engines.

Locomotives

EPA first regulated emissions from locomotive engines in 2000, when a set of three standards (Tiers 0–2) became effective, depending on the year the engine was produced. Tier 0 standards apply retroactively to any engine manufactured between 1973 and 2001; Tier 1 applies to new locomotive engines produced from 2002 to 2004; and Tier 2 applies to new locomotive engines produced beginning in 2005 (EPA, 2004a).

Unlike other diesel engine emissions standards, the Tier 0–2 locomotive standards apply not only when the engine is first produced, but every time the engine is re-

manufactured. This means that every locomotive engine built since 1973 must be upgraded to meet Tier 0 standards when it next has a major overhaul.

In June 2004, EPA issued an Advanced Notice of Proposed Rulemaking for consideration of Tier 3 standards to be implemented for new model year 2011 locomotive engines. These proposed standards would seek to reduce both PM and NO_x emissions by 90 percent or more compared to the Tier 2 standards and would bring locomotive engine emissions roughly in line with emissions from other nonroad engines under the Tier 4 rules (EPA, 2004b).

Chapter 14, *Nonroad Equipment*, provides a more complete discussion of emissions standards for locomotives.

Tractor Trailer Trucks

EPA first set exhaust smoke opacity standards for new heavy-duty onroad diesel engines beginning in model year 1970. Starting in model year 1974, new onroad engines were required to meet numeric emissions limits for CO and NO_x, plus HC. PM emissions were not regulated until 1988. Between 1988 and 1998, the EPA limits for HC and CO emissions remained the same, but the allowable levels of both NO_x and PM were each reduced in several steps. At 4.0 grams per brake-horsepower-hour (g/bhp-hr), the NO_x limit in effect for model year 1998 was 63 percent lower than the NO_x limit in effect for model year 1988, while the 1998 PM limit, at 0.10 g/bhp-hr, was 83 percent lower than the PM limit in 1988 (DieselNet, 2005).

Allowable NO_x and PM standards were lowered even further in two subsequent rounds of EPA rulemaking. Current emissions limits of 0.01 g/bhp-hr for PM and 0.20 g/bhp-hr for NO_x will go into effect for new onroad heavy-duty diesel engines starting with the 2007 model year and will be fully phased in between 2007 and 2010 on a percent-of-sales basis (EPA, 2001).

Table 17.3

Standards for Low-Sulfur Diesel Fuel

Category	Maximum Sulfur Limit (ppm)			
	2006	2007	2010	2012
Highway	15	15	15	15
Nonroad	3000[a]	500	15	15
Marine/rail	3000[a]	500	500	15
Ocean ships	45,000			

a. 3000 ppm is an approximate value representing uncontrolled emission levels.

Source: EPA, 2001, 2004b

Chapter 13, *Diesel Trucks and Buses*, provides a more complete discussion of onroad diesel engine standards.

Stationary Sources

The federal regulations that apply to stationary combustion sources used at marine ports are not specific to the use of these sources in a marine setting. In addition to maintenance facilities and warehouses, such sources include internal combustion engines and combustion turbines (discussed in Chapter 7, *Electric Generating Units*).

Fuel Quality

Congress has given EPA authority to regulate the quality of vehicle fuel and fuel additives, based on the direct health effects of the fuel and the effect of the fuel on the ability of vehicles to meet emissions standards. EPA first regulated the sulfur content of onroad diesel fuel in 1994 when it established a sulfur limit of 500 parts per million (ppm). The sulfur content of fuel used in nonroad diesels was unregulated. The distillate fuels currently used in construction and cargo handling equipment, locomotives and many marine vessels typically contain 3,000 ppm sulfur. The heavier residual oils used to power ocean-going ships can contain 50,000 ppm sulfur or more.

To allow the use of the advanced catalytic emissions controls that will be required to meet nonroad Tier 4 standards and proposed Tier 3 locomotive and marine standards, EPA has—for the first time—imposed limits on the allowable level of sulfur in nonroad diesel fuel. All nonroad diesel fuel will be limited to 500 ppm sulfur beginning in June 2007. A more stringent fuel sulfur limit of 15 ppm will be introduced for nonroad vehicles in 2010 and for locomotives and marine distillate fuels in 2012 (EPA, 2004a).

These sulfur limits do not apply to the residual fuels used in ocean-going ships, which are unregulated in the U.S. IMO MARPOL Annex VI establishes an international cap of 45,000 ppm on the sulfur content of marine residual fuels. This treaty also includes a mechanism to establish a 15,000 ppm cap in designated Sulfur Emission Control Areas (SECA), such as areas near coasts where additional emissions reductions are considered critical. While the U.S. has not formally ratified Annex VI, the current administration has supported ratification as well as the designation of coastal U.S. waters as a SECA zone (Argus, 2005).

As discussed in Chapter 4, *The Clean Air Act*, if EPA has regulated a fuel characteristic or component, states other than California may *not* adopt either their own or California's regulation—they must follow the federal regulation. The only exception to this is the situation in which the Administrator finds that state regulation is

necessary for achieving the National Ambient Air Quality Standards (NAAQS). On the other hand, California may adopt its own fuel regulations.

Other Regulations: California

Under the Clean Air Act, California can set its own emissions standards for both new and existing onroad vehicles, as well as for new and existing nonroad vehicles (with the exception of certain small vehicles and locomotives). However, California has, for the most part, adopted EPA regulations for new onroad and nonroad diesel equipment with only slight modifications.

California also has its own regulations for onroad and nonroad diesel fuel. Under the California rules, nonroad diesel fuel sold in California for other than locomotive or marine applications will be limited to 15 ppm sulfur beginning in 2006, four years earlier than the nonroad diesel sold in the rest of the country (CARB, 2004).

Chapters 13 and 14 provide a more complete discussion of California diesel engine emissions standards.

State Authority

States are restricted to implementing California or federal standards for new onroad and nonroad engines. However, states and local areas have authority to adopt their own standards for existing onroad engines. For existing nonroad engines, states are limited to federal or California standards. In fact, even the authority to adopt California regulations for these sources is largely theoretical at this point, since California has just begun to adopt mandatory retrofit requirements for certain types of nonroad equipment, beginning with portable engines. However, none of the limitations on state and local authority to impose standards prohibit the adoption of voluntary programs or the incorporation of retrofit requirements into government contracts for construction or other services.

Finally, as noted above, all states—including California—are barred from imposing their own limits on certain small nonroad equipment and on locomotives.

Emissions Control Opportunities

Many of the control technologies that are and will be applied to new onroad and nonroad diesel engines to comply with current and future EPA regulations can also be applied to existing marine and land-side engines and vehicles used in and around ports. There are three main approaches for reducing emissions from a diesel vehicle or fleet: use a cleaner engine, use a cleaner fuel or add a retrofit device.

Changes to equipment or operational practices that reduce total fuel usage from port vehicles and equipment can also reduce total emissions and save money in the bargain.

Use a Cleaner Engine

New diesel engines certified to stricter EPA standards produce 30–60 percent less NO_x and PM than the unregulated engines built as recently as a few years ago. These reductions have come from changes in diesel engine design, most notably better control of lubrication oil, higher fuel injection pressures and the addition of electronic fuel control. In some cases—particularly with respect to Tier 3 nonroad engines with electronic fuel control—the emissions reductions have been accompanied by improved fuel efficiency. Removing older engines from service and replacing them with new engines can therefore have a large positive emissions benefit.

For most older vehicles, it is more cost-effective to retire the vehicle itself. However, for some specialty port equipment where the engine represents a relatively small percentage of the vehicle's total value, "repowering" old equipment with a new engine may a better option. This approach has proved to be particularly cost-effective for Category 1 marine engines larger than 600 horsepower where improvements in fuel economy from the new engine can partially offset the cost. These engines provide propulsion power in some tugs and work boats used in marine ports.

For some nonroad port material-handling equipment, it may be possible to achieve even greater reductions by repowering with an onroad engine certified to lower emissions standards than new nonroad engines.

In limited cases it is also possible to cost-effectively rebuild and modify existing engines to incorporate more modern design elements and equipment, thus reducing their emissions. In addition to improved fuel injector designs, some of the cleaner engine and after-treatment technologies that are applicable specifically to marine engines include exhaust gas recirculation, water injection and sea water scrubbers.

Use a Cleaner Fuel

Some alternative formulations of diesel fuel produce lower emissions of NO_x and/or PM in existing engines compared to standard onroad or nonroad diesel fuel. The most common of these cleaner diesel fuels are described in Chapter 12, *Diesel Engine Technologies*.

Cleaner onroad ultra-low-sulfur diesel fuel could also be introduced in ports for use by marine and nonroad material-handling equipment before it becomes mandatory in 2012. This onroad fuel will have significantly less sulfur than the nonroad fuel available between now and

2012 and will produce slightly lower PM emissions when used in typical nonroad engines. Use of lower sulfur fuel will also allow port equipment to be retrofitted with more aggressive after-treatment technologies, resulting in even greater PM reductions (see Chapter 12). After 2012, all nonroad fuel will be required to have the same sulfur level as onroad fuel.

Heavy-duty engines that operate on natural gas are also available for onroad trucks and buses and produce significantly lower PM emissions than current diesel engines. Natural gas engines have PM emissions equivalent to the 2007 EPA PM standards for new onroad diesel engines. These natural gas engines are also certified to produce 30–50 percent lower NO_x emissions than the current EPA NO_x standards for new onroad diesels. While it is technically possible to modify existing diesel engines to operate on natural gas, it is generally preferred to use a purpose-built natural gas engine (ICCT, 2005).

Whether using a purpose-built or retrofit engine, natural gas conversions require that the vehicle fuel system be modified extensively to store either high-pressure gaseous or low-temperature liquefied natural gas (LNG) fuel in lieu of liquid diesel fuel; in addition, LNG or high-pressure natural gas fueling stations must be installed. Given the current limited availability of natural gas fueling stations, conversion to natural gas is most appropriate for fleet vehicles that are centrally fueled at a fixed site and that operate within a limited geographic area. These criteria are often met in the case of land-side material-handling equipment used in ports. For example, the Port of Los Angeles has recently announced a goal to shift from diesel power to the use of natural gas engines for cargo-handling equipment (Port of Los Angeles, 2005d).

Liquid petroleum gas (LPG), like natural gas, can also be used in specially designed engines to power equipment that might otherwise use diesel fuel; it too produces lower PM and NO_x emissions. LPG is not typically used in the U.S. for onroad trucks or large nonroad equipment, but small and medium-sized forklifts are available with LPG engines.

In the category of cleaner fuels, electricity can also play a role in ports. Depending on the fuel used for electricity generation, net NO_x reductions associated with converting nonroad equipment from diesel to electric power can be 40–80 percent (EPA, 2004c, DieselNet, 2005). While the dirtiest coal- and oil-burning power plants could actually produce greater PM emissions than nonroad diesel equipment, the actual grid mix in many parts of the country would produce a net PM reduction of 25–95 percent (EPA, 1998; DieselNet, 2005). In either case, the localized benefits are even greater, since no electric motors emit PM or NO_x at the point of use.

Unlike onroad vehicles, which typically travel long distances in a day, many of the cranes used to unload ships in port are essentially stationary. For this stationary equipment, the necessary electric power can be pulled directly from the electric grid, without the need for batteries on the vehicle and periodic charging. Ports from New York to Seattle have been replacing diesel-powered cranes with electric cranes.

Other land-side material-handling equipment in the port operates within a very limited geographic area for its entire life. Thus, the range limitations associated with battery-powered mobile electric vehicles are not a barrier. In particular, small and medium-sized forklifts are commercially available with electric drives.

Emissions generated by the auxiliary engines of ocean-going ships while they are docked are a significant part of the inventory of most ports. For some of these ships, land-side electric power can be used to supply the necessary ship-side hotel loads in lieu of on-board auxiliary engines ("cold ironing"), resulting in significant emissions reductions.

Add a Retrofit Device

As described in Chapter 12, *Diesel Engine Technologies*, there are a number of EPA- and CARB-verified retrofit technologies that can be installed on existing diesel vehicles to reduce emissions. Most of these are "after-treatment" devices that are installed in the exhaust system to clean up the exhaust after it has left the engine, rather than reducing the emissions produced by the engine. Most of these technologies rely on some kind of catalytic process and can reduce either PM or NO_x emissions, but not both simultaneously. However, discrete technologies can be combined into a single commercial product to yield simultaneous NO_x and PM reductions.

The three main approaches to reducing emissions from diesel vehicles are not mutually exclusive. In fact, some commercially available retrofit devices work better in conjunction with, or even require the use of, an alternative diesel fuel. Products are also available that combine engine modifications with retrofit devices, while engine repowering can always be combined with after-treatment control retrofits to reduce emissions even further.

Fuel Economy Improvements

As discussed above, some new nonroad engines are both cleaner and more fuel efficient than older engines, so that repowering to reduce emissions can provide additional benefits based on fuel savings. This is especially true for engines available to repower marine harbor craft such as tugboats, as well as some cargo handling equipment.

In addition, operational changes at ports may reduce total fuel consumption by vehicles and equipment within the port, thus reducing total emissions. Trucks entering the port to drop off or pick up cargo often end up queuing for long periods of time waiting for an open gate. While waiting, drivers often leave their vehicles idling, which wastes fuel and increases total emissions. Methods that can reduce queuing and attendant idling include better layout of terminal gates and staging areas, the use of gate appointment scheduling systems, expansion of port hours with incentives for off-peak delivery and pick-up, and the use of common pools of dreyage trucks by multiple terminal operators. Voluntary or mandatory idling restrictions for waiting trucks can also help (see Chapter 13, *Diesel Trucks and Buses*).

Many goods today are moved as "containerized" cargo in standardized metal containers that go from truck or locomotive to ship, and vice-versa, without being unloaded. In many ports, the movement and storage of these containers dominate operations. Optimal terminal layout and better tracking and management of individual containers using information technology can minimize the use of nonroad material-handling equipment and resultant emissions.

In some cases greater use of rail (instead of trucks) to move goods in and out of ports can also reduce total fuel use and emissions within the port, while also reducing truck traffic and congestion in surrounding communities.

State and Local Policy Measures

Programs that encourage or mandate the retirement or replacement of older diesel equipment, the use of cleaner fuels or electrification, or the retrofit of older equipment with emissions control devices can have a large impact on air quality. These programs provide the most significant opportunity for states to control and reduce PM and NO_x emissions from equipment used in ports.

Whether these programs are voluntary or mandatory, their common focus is on the application of new technology to existing onroad and nonroad diesel equipment. Programs that result in reduced fuel usage will also reduce PM and NO_x emissions.

Voluntary Retrofit Programs

Replacing vehicles and engines, using alternative fuels and installing retrofit devices can be expensive. The Manufacturers of Emission Controls Association estimates that retrofit PM control devices for Class 8 trucks cost between $1,000 and $10,000 per vehicle, depending on the type of retrofit device being installed (MECA, 2000); NO_x control devices can cost as much as $20,000 or more for the same vehicles. In general, the larger the engine, the larger and more costly the control device.

Replacement engines for material-handling equipment can cost $30,000 or more installed, and Category 1 marine engines can cost more than $100,000 (TCEQ, 2004). Given this expense, voluntary emissions reduction programs are more effective when accompanied by a funding source to offset the cost to the equipment owner.

Programs to offset the cost of emissions controls can take many forms: for example, they can provide a tax reduction or incentive, or rebate some or all of the cost of taking a specific action. The most common emissions reduction incentive programs for the type of diesel equipment typically found in ports are grant programs.

The two largest grant programs of this type currently operating are California's Carl Moyer Memorial Air Quality Standards Program (Carl Moyer) and the Texas Emissions Reduction Program (TERP). Since its inception in 1998, Carl Moyer has provided $184 million in grant funding for various emissions reduction projects throughout California (CARB, 2003). Approximately 30 percent of these funds went to projects to reduce emissions from nonroad diesel equipment that might be found in ports, including marine vessels, forklifts and locomotives (CARB, 2005a).

Since 2001, TERP has provided $82 million in grants to replace 98 locomotives and retrofit six yard switchers and $11 million to repower 89 and retrofit six marine engines in the 41 Texas counties designated as nonattainment areas (TCEQ, 2004).

In recent years EPA has awarded demonstration grants under its National Clean Diesel Campaign. The most recent grants were announced in February and October 2005, with a total of $2.6 million going to 26 projects across the country (EPA, 2005d). The projects include retrofits of rubber-tired gantry cranes and port straddle carriers. The program requires matching funds.

Over the last four years the Puget Sound Clean Air Agency in Seattle, Washington has sponsored and funded approximately 3,000 retrofits of both onroad vehicles and nonroad equipment, including port material-handling equipment. It has also (a) helped fund a pilot program to use ultra-low-sulfur onroad diesel fuel in a Washington State ferry running between the U.S. and Canada and (b) established—in cooperation with a number of private and public partners—a shore power facility (see below) for Princess Cruise Line vessels calling at the Port of Seattle. This facility has been operating since the summer of 2005. The Puget Sound Clean Air Agency Diesel Solutions program is voluntary and incentive-based (Puget Sound CAA, 2005).

A number of states have also used funds from environmental enforcement actions to fund diesel retrofit projects. For example, as part of a 2003 consent decree, the Port of Los Angeles created a $20 million Air Quality Mitigation Program that provides financial incentives to port area businesses to clean up their operations (Port of Los Angeles, 2005a). Funds from this program have been used to retrofit onroad trucks used in the port, as well as to retrofit and repower marine engines.

Chapter 13 provides a more detailed discussion of funding mechanisms for clean air grant programs.

One potential impediment to the adoption of voluntary programs by ports is concern about the effect on future port expansion activities that may be subject to the conformity requirements of the Clean Air Act (see Chapter 4). Given rising cargo volumes across the country, many ports face pressure to expand their operations. In order to expand, ports may need to identify emissions reduction credits for use in meeting conformity requirements. Some ports may be unwilling to take actions to reduce existing emissions unless the reductions can be used for conformity purposes in the context of a later expansion project. A method for ports to "bank" site-specific credits for later use might provide an incentive for ports to take early action.

Despite this problem, conformity requirements can contribute substantially to overall emissions reductions. For example, in order to offset increased NO_x emissions from a 10-year harbor dredging project in New York Harbor, the Port Authority of New York and New Jersey implemented a Harbor Air Mitigation Plan (HAMP) that included retrofitting Staten Island Ferries with selective catalytic reduction (SCR) technology, as well as repowering tug boats. While the NO_x from dredging activities will peak at approximately 450 tons per year (ton/year), the HAMP strategies will produce 800–1,000 ton/year of NO_x offsets beginning in 2006. After the dredging project is over, the implemented projects will reduce total NO_x emissions in the harbor by up to 1,000 ton/year (EPA, 2005b).

Mandatory Retrofit Programs

As stated previously, states other than California are limited to adopting California standards for existing nonroad equipment. In 2004, CARB adopted mandatory retrofit requirements for the portable diesel engines used in agricultural pumps, airport ground-support equipment, oil drilling rigs, portable generators and other nonroad equipment in California. The rule requires stepped reductions in PM emissions from these engines to reach a 95 percent reduction level by 2020 and is likely applicable to some diesel sources in ports (CARB, 2005b). In addition, CARB in December 2005 released a Draft Emission Reduction Plan for Ports and International Goods Movement in California. This plan anticipates future action by CARB to mandate the use of cleaner fuels and retrofits of marine harbor craft and port-based material-handling equipment (CARB, 2005d). Other states can adopt the current CARB requirements for portable engines as well as any other rules applicable to port-based nonroad engines that California adopts in the future.

Indirect mandates imposed through contractual requirements may be more practical in some situations. This approach can be implemented by the port authority through contract arrangements with the companies that operate the terminals. These arrangements will have to be negotiated as modifications to existing contracts or will require the expiration of current contracts.

For example, the Port of Long Beach Authority is drafting a tariff proposal that will require all tenants in the port to develop and implement a plan to significantly reduce PM and NO_x emissions from their operations by 2008. The Authority envisions that the contractual requirements will allow tenants flexibility in achieving the reductions.

As part of a ten-year contract extension for railroad switching services within the ports, the Ports of Long Beach and Los Angeles will require Pacific Harbor Line to replace its existing fleet of 16 locomotives—some of which are more than 50 years old—with cleaner equipment. The new locomotives will include engines that burn emulsified diesel fuel and exceed EPA Tier 2 locomotive emissions standards. Under the agreement, Pacific Harbor will also acquire one locomotive fueled by natural gas and one with a hybrid diesel/battery-electric drive system. Pacific Harbor will also test diesel oxidation catalysts; if successful, they will be retrofit on all its new locomotives. A Carl Moyer grant will partially fund the program, with Pacific Harbor and the ports sharing remaining costs (Port of Long Beach, 2005).

For port authorities that directly own and operate terminal equipment, administrative policies that require retrofits, retirement/repowering or fuel switching could complement policies for tenant equipment.

Idle Reduction

Trucks often queue at marine terminals and idle while waiting to pick up or deliver goods. Unnecessary idling wastes fuel and increases emissions of NO_x and PM. EPA estimates that an idling Class 8 truck uses, on average, 0.82 gallons of diesel fuel per hour and produces 144 grams per hour of NO_x emissions (EPA, 2002).

In order to reduce or eliminate unnecessary truck idling, approximately 26 states and governmental subdivisions have enacted laws that restrict the idling of diesel vehicles (ATRI, 2004). Typically, these laws set a maximum

allowable idling period, anywhere from 2–15 minutes, with fines for violations. Port authorities could implement similar idling restrictions.

Idling laws usually provide an exemption for idling in traffic, as well as for idling necessary to ensure safe operation of the vehicle—for example, to maintain the air pressure in the air brake system. Recognizing that diesel engines can be difficult to start in cold conditions, these laws also often provide an exemption to allow additional idling below a specific ambient temperature, such as 32 or 20 degrees Fahrenheit. Some also provide an exemption to maintain power for heating, cooling or auxiliary equipment.

Switcher yard locomotives also generally idle for long periods of time—an issue in ports that load cargo directly from ships to rail. These locomotives are allowed to idle to maintain engine oil and coolant temperatures, or to provide a relatively small amount of electrical power for auxiliary equipment. They often idle primarily because of the operator's perception that it is easier than shutting down and restarting. Argonne National Laboratory estimates that a typical switcher yard locomotive idles up to 75 percent of the time and that idling accounts for 27 percent of its total fuel use (Argonne, 2005).

As discussed in Chapter 12, *Diesel Engine Technologies*, a number of systems can be installed to (a) provide the necessary electrical power for auxiliary loads on locomotives or (b) monitor various engine parameters and automatically shut down the engine when idling is unnecessary and restart it as required. EPA's review of available systems of this type indicates that they cost between $4,000 and $50,000 to install on a locomotive, depending on the type of device used. Given the amount of fuel that can be saved by reducing or eliminating idling, many of these systems nevertheless have payback periods as short as 6–20 months (Gaines, 2005).

Port authorities could encourage or mandate the use of technologies to reduce or eliminate unnecessary idling in the same way that they encourage or mandate equipment retirement, engine repowering, control device retrofits and fuel switching. EPA has issued guidance to states for taking credit for the benefits of an idle reduction program (EPA, 2004d).

See Chapter 14, *Nonroad Equipment*, for examples of projects to address locomotive idling.

Nonroad Diesel Fuel

At approximately 3,000 ppm, the current sulfur content of nonroad diesel fuel impedes the use of effective retrofit emissions controls on marine and other nonroad engines. Sulfur interferes with the catalytic processes used to reduce PM emissions; diesel oxidation catalysts do not work well if fuel sulfur is greater than 500 ppm, while the most effective diesel particulate filters require fuel with no more than 50 ppm sulfur (see Chapter 12, *Diesel Engine Technologies*).

Although EPA regulations will require reduced sulfur levels in nonroad diesel fuel, implementation dates lag those for onroad diesel fuel. Beginning in June 2007, when EPA will require nonroad diesel and marine distillate fuels to have no more than 500 ppm sulfur, diesel oxidation catalyst retrofits will become practical on a wide range of nonroad equipment used in ports, including marine harbor craft and material-handling equipment. Retrofits with the more effective diesel particulate filters will not be possible until nonroad diesel fuel sulfur is reduced to below 50 ppm, which, under current EPA rules, will not happen until 2010 for most nonroad equipment and until 2012 for locomotives and marine distillate fuels. Retrofit programs for nonroad equipment will not be effective prior to these dates without the use of onroad diesel fuel in the retrofit nonroad equipment. This is why the Puget Sound Clean Air Agency has helped fund a pilot program to use ultra-low-sulfur onroad diesel fuel in a Washington State ferry.

California has mandated that all nonroad diesel fuel other than fuel for locomotives and marine applications have no more than 15 ppm sulfur beginning in 2006—four years earlier than the EPA mandates. Other states can adopt these California fuel regulations—but only if EPA finds that such action is necessary for achieving the NAAQS.

Given the very high sulfur content of the residual fuel oil used to power ocean-going ships, very few retrofit technologies will be effective on the Category 3 marine engines used for propulsion in these vessels. A switch to distillate fuels would itself reduce PM and sulfur dioxide (SO_2) emissions, while opening up greater opportunities for pollution control retrofits. However, this would require modifications to the fuel systems on these engines, and would dramatically increase fuel costs for vessel owners.

A more practical approach would be to require the use of reduced-sulfur onroad or nonroad diesel fuel in the auxiliary engines that provide power while vessels are docked in port. For example, the Port of San Francisco is providing monetary incentives to offset the cost to cruise ship operators of using lower sulfur fuels while their ships are in port. EPA provided $100,000 toward this program through the West Coast Collaborative and other project partners are providing $141,000 in matching funds (West Coast Collaborative, 2005a)

Ocean-Going Ships

As noted previously, many marine vessels operate auxiliary engines while in port to power their "hotel"

loads (i.e., onboard electrical equipment and pumps). For certain types of ships—especially cruise ships, refrigerated cargo vessels and some tankers—the electrical loads being supplied can be significant: anywhere from 600 to 7,000 kilowatts. In addition, some types of vessels spend significant time in port; cruise ships typically spend 12–24 hours docked while exchanging passengers at the beginning and end of each trip. Some cargo vessels, meanwhile, take 30–100 hours or more to unload (Port of Long Beach, 2004).

The combination of high onboard power demand and long port calls results in significant in-port exhaust emissions from ship auxiliary engines. For example, an analysis of port calls at the Port of Long Beach in southern California determined that a single ship that regularly calls at the port produces, on an annual basis, over 85 tons of NO_x, 9 tons of PM and 79 tons of SO_2 while at the pier (Port of Long Beach, 2004).

One approach to reducing these emissions is to provide electrical power from the land-side electrical grid to supply in-port hotel loads, instead of using the onboard auxiliary diesel engines. This approach was pioneered by the Navy and is called "cold ironing."

Cold ironing requires significant modifications to both the affected ships and the land-side infrastructure. The first cold ironing project in the U.S. was completed in 2001 in Juneau, Alaska. It involved the installation of necessary shore-side equipment and retrofit of four cruise ships owned by Princess Cruises. Installing the shore-side equipment cost $4.7 million; ship retrofit costs averaged $500,000 per vessel (Port of Long Beach, 2004). Princess Cruises is also participating in a shore power project at Terminal 30 in the Port of Seattle. This project is expected to reduce annual PM and NO_x emissions within the port by 2 tons and 14 tons, respectively (McLerran, 2005).

Given the low cost of marine distillate fuels, the typical cost of electricity supplied from the land-side grid is slightly higher than the cost of fuel for onboard electricity generation.

Because of the expense involved, cold ironing is not a cost-effective strategy for every ship. Nonetheless, the air quality benefits of this approach can be significant and can also be cost-effective if applied to ships with a significant hotel load and many long port calls. For example, a study by the Port of Long Beach showed that the use of shore power to supply hotel loads to only three ships could reduce annual in-port emissions by almost 500 ton/year, at a cost of $11,000 per ton or less (Port of Long Beach, 2004).

The Port of Los Angeles has developed specifications for both the ship-side and shore-side equipment necessary for cold ironing, which it calls Alternative Marine Power (AMP). In June 2004, the port opened the first U.S. container berth equipped to provide shore power (Berth 100 at the China Shipping Terminal); two months later, the Port of Los Angeles hosted the first ship capable of accepting this power (the NYK Atlas). The port has plans to install AMP equipment at additional terminals (Port of Los Angeles, 2005c).

Other actions that can be taken to reduce ship emissions in and around ports include the use of lower sulfur diesel fuel in auxiliary engines while in port, as discussed above, and reducing ship speeds near the coast. Because ships that operate at lower speeds have lower emissions, both the Port of Long Beach and the Port of Los Angeles implemented a Voluntary Commercial Ship Speed Reduction Program in 2001, which encourages vessels to travel at or below 12 knots within 20 miles of the coast. The ports estimate that nearly 70 percent of shipping lines currently calling at the port participate in the program and that NO_x emissions from these ships have been reduced by more than 1 ton per day (Port of Los Angeles, 2005b).

Operational Changes

Marine ports are complicated commercial enterprises, typically with many different players trying to move many different types of cargo among a number of different transportation modes on a daily basis. Given this complexity, port operations are likely to contain numerous inefficiencies that lead to long wait times for individual trucks as well as wasteful use of dreyage trucks and material-handling equipment. Operational changes that reduce truck queuing and idling, such as extending port hours and providing incentives for off-peak delivery, and better use of information technology to track containers and schedule pick-ups and deliveries, may contribute to increased efficiency and reduced fuel usage.

For example, the Georgia Ports Authority gave all members of the port/terminal community real-time, round-the-clock access to transaction information via a web portal. This system has reduced gate processing times by reducing the number of "trouble transactions" precipitated by outdated or inaccurate data. As a result it has all but eliminated truck queuing outside terminal gates and has reduced truck turn-around time by 30 percent; the Authority estimates that this has reduced NO_x emissions by half a ton on peak days (EPA, 2005b, 2005c).

A common practice at many ports is for each terminal operator to own its own fleet of dreyage trucks (or chassis), which can only be used to move containers that belong to that company. Chassis switching by drivers moving containers from multiple operators can add up to an hour per trip and contributes to congestion on port roadways. In October 2004, the Port of Virginia became the first U.S.

port to require that all of its terminal operators participate in a common chassis pool, meaning that any vehicle can be used to move any container. This has reduced the number of individual trucks stored and used within the port by 20 percent—a reduction of 5,000 chassis. Local drivers who had previously only been able to move two to three containers per shift are now completing up to ten moves a day, increasing their income and improving service to customers while reducing total fuel use and emissions within the port (EPA, 2005c).

Increased use of rail transport to move goods in and out of ports may also contribute to net emissions reductions, while also reducing the use of mobile diesel-powered material-handling equipment in favor of electric-powered stationary cranes. Increased use of rail transport could also provide ancillary benefits to nearby communities by reducing truck traffic and congestion in local neighborhoods. However, this approach is likely to require significant investments in new rail infrastructure and may be constrained by the layout of existing port facilities and the availability of appropriate rights of way.

Many of the operational changes that will help to reduce emissions will also save time and money for terminal operators as well as their customers. Nonetheless, the changes may also entail significant up-front expense and there is likely to be significant competition for the capital funding that might pay for them.

To facilitate the implementation of some of these strategies, it might be appropriate to include them for eligibility in a clean air grant program—in the same way that vehicle retrofits, engine repowering and fuel switching are eligible. In light of the fact that these changes might ultimately save the terminal operators money, a low-interest revolving loan program (discussed in Chapter 14, *Nonroad Equipment*) might be even more appropriate.

Additionally, in the same way that the "bubble" concept is applicable to airports, it is appropriate for consideration in the marine port context. The bubble would cap multiple sources of emissions, such as commercial harbor craft, cargo handling equipment and stationary sources. A range of strategies would ensure that overall emissions did not exceed the cap, with the cap established as an absolute limit on emissions (e.g., tons per year) or on a rate basis (e.g., tons per passenger).

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Chapter 18

Residential Fuel Combustion and Electricity Use

Introduction

In aggregate, households in the U.S. consume large amounts of fossil fuel-based energy—both on-site and indirectly, through the use of electricity. According to the Energy Information Administration (EIA), the residential sector was responsible for 21 quadrillion British thermal units (Btus) or "quads" of energy consumption in 2004—21 percent of total U.S. energy consumption (EIA, 2005). Common uses of energy associated with this sector include space heating, water heating, air conditioning, lighting, refrigeration and cooking.

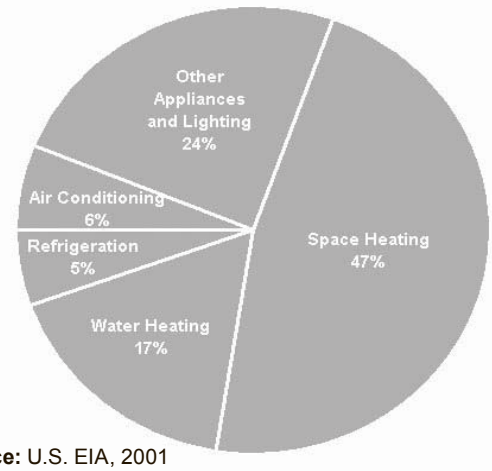
The residential sector produces fine particulate matter ($PM_{2.5}$) and $PM_{2.5}$-precursor emissions on-site from the direct consumption of fuels—such as natural gas, liquefied propane gas, kerosene, fuel oil, coal and wood—primarily for space heating and water heating. Furnaces, boilers, hot water tanks, wood stoves and other equipment produce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions to varying degrees, depending on the fuel consumed and the technology used. This chapter discusses strategies for reducing the direct emissions generated by these sources.

The residential sector is also heavily reliant on electricity to meet its energy needs. According to EIA, the residential sector was responsible for 36 percent of U.S. electricity consumption in 2004 (EIA, 2005). As a result, a substantial share of emissions associated with residential sector energy consumption occur off-site, at fossil-fired power plants. Chapter 7, *Electric Generating Units*, discusses strategies for reducing power plant emissions directly. This chapter reviews a number of strategies for reducing household electricity consumption to further reduce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from power plants.

Sector Profile

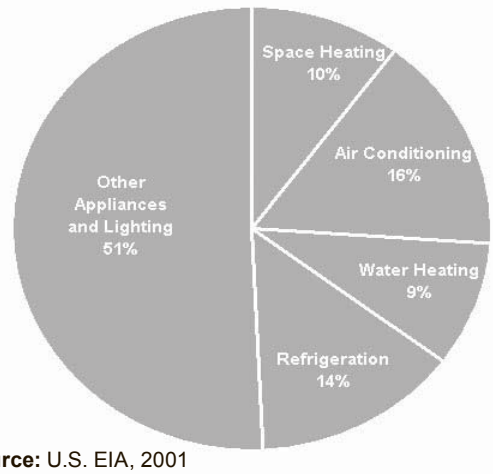
Every four years, EIA conducts a national survey of household energy use known as the Residential Energy Consumption Survey. According to this survey, in 2001, nearly half (47 percent) of all household energy

Fig. 18.1 U.S. Residential Energy Use, 2001



Source: U.S. EIA, 2001

Fig. 18.2 U.S. Residential Electricity End Uses



Source: U.S. EIA, 2001

consumption was devoted to home heating (EIA, 2001).[1] Another 17 percent was used for hot water heating, 6 percent was used for electric air conditioning and 5 percent was used for refrigeration. The remaining 24 percent was used for lighting and to power other appliances. Some regions of the country, such as the Northeast, devote a greater share of energy use to home heating (58 percent) because of their colder climate.[2] This distribution of energy use does not, however, reflect the full energy impact of household electricity use. A comprehensive accounting that includes generation losses at the power plants supplying electricity and the line losses involved in delivering power across the electricity grid would nearly double the energy use attributed to residential sector electric use. This means that energy efficiency improvements in appliances, lighting and air conditioning will have almost twice as much impact on overall energy consumption, compared to non-electric energy use, as Figure 18.1 suggests.

According to EIA, the three most common energy sources for home heating in the U.S. are natural gas, serving 50 percent of households; electricity, serving 36 percent of households; and fuel oil, serving 7 percent of households (EIA, 2001). The U.S. Census Bureau's more recent American Community Survey reports a similar distribution; specifically, it found that nearly 62 million homes (57 percent) used natural gas or liquefied petroleum gas as their heating fuel in 2003. The second most popular home heating source was electricity, used by 33.9 million homes (31.3 percent). According to this survey, only 8.6 percent of households nationwide used fuel oil or kerosene as their primary heating fuel (U.S. Census Bureau, 2004).[3]

Wood is generally not the primary source of energy for home heating in U.S. households. However, widespread use of wood stoves and fireplaces in some areas is an important contributor to ambient levels of particulate matter. In certain winter pollution episodes, more than half of the atmospheric $PM_{2.5}$ concentration in a local area can be attributed to wood smoke (Schauer, 2000). According to EPA, there are 40–45 million wood burning appliances in the U.S., 15 million of which are wood stoves, either free standing or fireplace inserts (Brockman, 2005a). Many of the wood stoves that are in regular use in the U.S. were installed prior to the federal emissions performance standards adopted in 1988.

There are more than 27 million fireplaces in U.S. homes that are used for supplemental heat, aesthetics or as a primary heat source (Houck, 2000). In general, there are two types of fireplaces: manufactured metal and site masonry fireplaces. In 1997, approximately 400,000 factory-built metal fireplaces were sold in the U.S. These fireplaces are designed to last 40 years or more. Masonry fireplaces can last indefinitely.

According to the Hearth, Patio and Barbecue Association, as many as 100,000 homes across North America heat with an outdoor wood-burning furnace, a vast increase from the 2004 figure of 78,000 (Russell, 2005).

1. This includes natural gas, fuel oil, kerosene, liquefied petroleum gas and electricity use. It does not include wood use on-site or fuel consumption at the power plants supplying the electricity.

2. The Residential Energy Consumption Survey is a national survey that collects information on residential energy use. In the most recent survey, conducted in 2001, data were collected from 4,822 households statistically selected to represent the 107 million housing units in the U.S. Data are available for four census regions and nine census divisions. State-level data are available for the four most populated states (California, Texas, New York and Florida). www.eia.doe.gov/emeu/recs/contents.html.

3. The results of the American Community Survey are available at the state and county level. www.census.gov/Press-Release/www/releases/archives/american_community_survey_acs/003053.html.

Table 18.1

Residential Fuel Combustion Emissions, 1999

Fuel Type	$PM_{2.5}$		SO_2		NO_x	
Source Category	**Tons/Year**	**Percent of Sector Emissions**	**Tons/Year**	**Percent of Sector Emissions**	**Tons/Year**	**Percent of Sector Emissions**
Natural gas (including liquefied petroleum gas)	23,981	4%	1,703	1%	258,968	65%
Oil (primarily distillate)	6,229	1%	124,706	66%	52,693	13%
Wood	419,247	67%	5,696	3%	45,402	11%
Coal	5,147	1%	51,555	27%	7,290	2%
Open burning (household and yard waste)	174,936	28%	5,107	3%	33,461	8%
Total direct emissions	**629,540**	**100%**	**188,767**	**100%**	**397,814**	**100%**
Indirect emissions (electricity use)	223,492	NA	4,530,064	NA	2,058,692	NA

Source: EPA, 2005a

Natural gas is also the dominant energy source used for water heating, serving 54 percent of households. Electricity is second, serving 39 percent of households, and fuel oil is third, serving 4 percent of households (EIA, 2001).

Electricity dominates as the energy source for other major household functions, including refrigeration and lighting. Major uses of electricity in U.S. homes are air conditioning, space heating, refrigeration, lighting, water heating and a broad category of uses that we designate as "other" (e.g., televisions, computers, small appliances). Figure 18.2 provides a national break-out of household electricity use by end-use category.

Residential fuel combustion produces $PM_{2.5}$ and $PM_{2.5}$-precursor emissions to varying degrees, depending on the fuel consumed and the technology used. Table 18.1 presents national emissions estimates for primary household energy consumption by fuel type. Wood combustion is the dominant source of direct $PM_{2.5}$ emissions within the residential sector. (Wood stoves and fireplaces also produce a host of hazardous air pollutants including benzene, toluene and polycyclic organic matter.) Oil is the leading source of direct (on-site) sulfur dioxide (SO_2) emissions within the sector, reflecting the higher sulfur content of home heating oil relative to natural gas and wood. Residential coal consumption accounts for an estimated 26 percent of the sector's direct SO_2 emissions. The majority of homes that use coal for home heating are located in just a few states: Pennsylvania (37 percent), Kentucky (7 percent), Indiana (7 percent) and New York (7 percent) (U.S. Census Bureau, 2004). Natural gas produces the largest share of residential nitrogen oxides (NO_x) emissions, in part because of its widespread use.

In addition to direct emissions, the residential sector is also responsible for a substantial share of U.S. electricity use. According to EIA, the residential sector was responsible for 36 percent of U.S. electricity consumption in 2004 (EIA, 2005). Table 18.1 provides an estimate of the indirect emissions attributable to residential electricity use.[4]

Finally, many households also engage in open burning for the disposal of leaves, tree branches and yard trimmings. Open burning produces $PM_{2.5}$ emissions, as well as a host of other pollutants, depending on the materials burned. Burning household trash presents a special concern because of the air toxics generated (EPA, 2003). Research by EPA suggests that a family of four burning its household

4. These figures were calculated by attributing 36 percent of the electric power sector's $PM_{2.5}$, SO_2 and NO_x emissions to the residential sector. The actual electricity-related emissions attributable to the residential sector would vary based on the load profile of household electricity use (i.e., when homes are consuming electricity and which power plants are serving load at these times); however, we consider our more simplified approach to be a reasonable proxy.

wastes on-site will generate as much or more dioxin than a full-scale, 200 ton-per-day municipal waste incinerator (Western Lake Superior Sanitary District, 2005).

Emissions Control Opportunities

A multitude of technologies and conservation measures are available for reducing the air pollution emissions associated with residential energy use. The discussion that follows addresses many of these options and identifies resources for obtaining further information. The discussion focuses on the major uses of energy in the home: space heating (including wood stoves and fireplaces), water heating and powering various household appliances. To touch on all of the options available is beyond the scope of this report. For example, we do not discuss options for enhancing building efficiency through improved insulation and windows. However, we do address many of the options that we believe offer significant potential for reducing residential energy use and emissions.

Space Heating

In many parts of the U.S., space heating is the largest user of energy in the residential sector, accounting for nearly half (47 percent) of all household energy consumption in 2001 (EIA, 2001). Space heating is also one of the most costly items in the household energy budget, particularly as natural gas and fuel oil prices have continued to rise.

As indicated in the sector profile above, natural gas is the most widely used home heating fuel. In 2003, nearly 62 million homes (57 percent) relied on natural gas or liquefied petroleum gas as their heating fuel, 33.9 million homes (31.3 percent) relied on electricity and 8.6 percent of households nationwide relied on fuel oil and kerosene (U.S. Census Bureau, 2004).

Furnaces, boilers and electric resistance heaters supply most of the heating in homes today. In homes heated by natural gas, furnaces are the most common heating system—67 percent of gas-heated homes use central furnaces, with the remainder of homes split among hot water (hydronic) systems, direct-heating furnaces and room heaters. In homes heated by electricity, the primary heating equipment is split nearly evenly among three categories: electric furnaces, heat pumps and baseboard heaters. In oil-heated homes, 41 percent use furnaces and 55 percent use boilers (E Source, 1997). Options for reducing the energy use and emissions associated with these systems include fuel switching, installing add-on controls to make existing systems more energy efficient, replacing older systems with newer, lower-emitting and higher-efficiency models, and replacing conventional technologies with alternatives, such as solar energy space heating technologies or geothermal heat pumps.

Gas Furnaces. NO_x is the major pollutants of concern produced by a natural gas-fired residential furnace. Emissions rates depend primarily on the peak temperature within the combustion chamber, oxygen concentrations and the amount of time that the combustion air is exposed to peak temperatures. As these three factors increase, NO_x emissions levels increase. Natural gas combustion can also produce particulate emissions; however—because natural gas is a gaseous fuel—PM emissions are typically low and emitted particles are generally less than one micrometer (μm) in size. Increased PM emissions may result from poor air/fuel mixing or maintenance problems. Natural gas combustion can also produce SO_2 emissions

Table 18.2

Emissions from Natural Gas- and Oil-Fired Residential Furnaces

Source Category	PM (lb/MMBtu)	SO_2 (lb/MMBtu)	NO_x (lb/MMBtu)
Natural gas furnace[a] (<0.3 106 Btu/hr heat input)	0.007-0.011[b,c]	0.0006	0.092
#2 fuel oil furnace[d] (2,500 ppm sulfur)	0.012	0.25	0.13
#2 fuel oil furnace[d] (500 ppm sulfur)	0.012	0.051	0.13
#2 fuel oil furnace[d] (15 ppm sulfur)	0.012	0.0015	0.13

a. Emissions factors were converted to an energy basis (lb/MMBtu) by using a heating value of 1,020 MMBtu/10^6 standard cubic feet (scf).
b. Assumed to be less than 1.0 μm in diameter.
c. Lower PM value from AP-42 Chapter 1: External Combustion Sources, Section 1.4 Natural Gas Combustion, Supplement D, July 1998. Higher value from Questar Gas, Environmental Information, http://www.questargas.com/brochures/environmental.pdf.
d. All values from AP-42 Chapter 1: External Combustion Sources, Section 1.3 Fuel Oil Combustion. Tables 1.3-1 and 1.3-2, September 1998. Emissions factors were converted from lb/10^3 gallon using a conversion factor of 140 MMBtu/10^3 gallon.

because a sulfur-containing mercaptan is added to the fuel to permit leak detection. Emissions rates for a residential gas furnace are presented in Table 18.2.

Home Oil Use. Heating oil, also known as No. 2 fuel oil, is much like diesel fuel, with at least one important distinction. Heating oil contains between 2,000 and 2,500 parts per million (ppm) sulfur (Bookhart, 2003). By comparison, sulfur levels in onroad diesel fuel are limited by regulation to less than 500 ppm, as discussed in Chapter 13.

Oil heat is used in more than 8 million homes, 6.3 million (78 percent) of which are in the Northeast (Bookhart, 2003). Residential oil use produces $PM_{2.5}$, SO_2 and NO_x emissions; however, SO_2 is the primary pollutant of concern. (The emissions rates of a residential oil furnace are presented in Table 18.2.) Virtually all (99 percent) of the sulfur in fuel oil is oxidized to SO_2 when combusted, and thus there is a direct linear relationship between oil sulfur content and SO_2 emissions from residential boilers (Batey, 2002).

The most effective strategy for reducing SO_2 emissions from residential oil use is lowering the sulfur content of heating oil. Currently in the U.S., heating oil for residential use has an average sulfur content of about 0.20–0.25 percent. Switching to low sulfur content fuel could eliminate 75–80 percent of the SO2 emissions generated by residential oil heating systems, as well as 80 percent of $PM_{2.5}$ emissions (McDonald, 2003).

The American Society for Testing and Materials (ASTM), an international voluntary standards development organization, has approved a Low-Sulfur No. 2 Heating Oil specification. Also, the Oilheat Manufacturers Association has been promoting low-sulfur heating oil, both to improve air quality and to reduce equipment maintenance costs (NORA, 2005). Low-sulfur heating oil reduces the level of residue build-up on the surfaces of boilers and furnaces, improving equipment performance and reducing maintenance costs. According to the Center for Clean Air Policy (CCAP), low-sulfur fuel oil costs about $0.01/gallon more than conventional fuel oil and would add roughly $8.65 a year to a typical customer's annual fuel bill, based on average annual oil consumption of 865 gallons per year (CCAP, 2005).

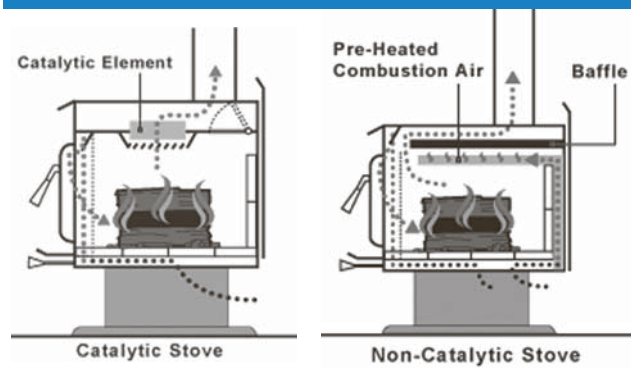
Technology options are also available that can improve the efficiency of oil-fired boilers and furnaces, such as high static flame retention burners. Since 1992, new oil furnaces have been required to meet a minimum energy efficiency standard of 78 percent, while new oil boilers have been required to meet an efficiency standard of 80 percent (DOE, 1989). Earlier model boilers and furnaces are still in service, however, and can have efficiencies as low as 60 percent or lower. A recent analysis estimates that approximately 5 percent of furnaces and 22 percent of boilers in the 15 states that consume the most home heating oil are operating at or below 60 percent efficiency (CCAP, 2005).[5]

The National Oilheat Research Alliance (NORA) has produced a guide for oil heat technicians that outlines options for low- and no-cost techniques to conserve heating fuel: *Efficient Oilheat: An Energy Conservation Guide*.

Wood Stoves and Fireplaces. There are three basic strategies for reducing $PM_{2.5}$ and $PM_{2.5}$-precursor emissions from wood stoves and fireplaces: (1) reducing or eliminating the burning of wood where less polluting alternatives (e.g., natural gas-fired units) are available; (2) relying on higher-efficiency, lower-polluting wood stove technology; and (3) adopting good burning techniques (e.g., using properly sized and properly seasoned wood).

As discussed in more detail below, EPA established smoke emissions limits for new wood stoves in the late 1980s. As a result, all new wood stoves and fireplace inserts must be certified to a maximum limit of 7.5 grams of smoke per hour. (Fireplaces and outdoor wood boilers are exempt from regulation.) The EPA standards have not only improved the emissions performance of new wood stoves, but have also improved their fuel efficiency.

Fig. 18.3 Wood Stove Technologies

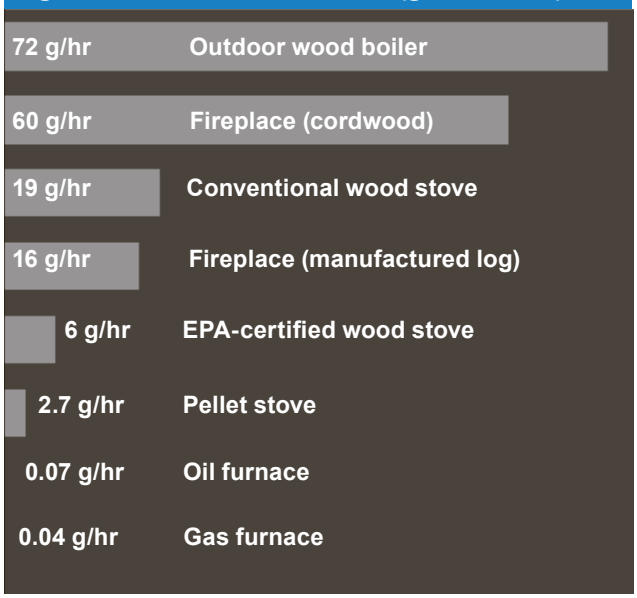


Source: EPA

Wood stove manufacturers use two basic strategies to meet the EPA standards: catalytic and non-catalytic combustion (Wood Heat Organization, 2005). With catalytic combustion, the exhaust from the stove passes through a specially coated ceramic honeycomb inside the stove, where the smoke gases and particles ignite and burn. The on-going performance of the stove depends on the maintenance and replacement of the catalytic honeycomb,

5. These 15 states include Connecticut, Maine, Maryland, Massachusetts, Minnesota, New Hampshire, New Jersey, New York, North Carolina, Pennsylvania, Rhode Island, Vermont, Virginia, Ohio and Wisconsin.

Fig. 18.4 PM Emissions Rates (grams/hour)



Source: Spitzer, 2005b, Houck, 2000, and Houck, 1998.

which can degrade in as little as two years without regular cleaning and maintenance (Wood Heat Organization, 2005).

Non-catalytic stoves rely on the design characteristics of the unit to allow complete combustion of the wood. These design features include firebox insulation, a large baffle to produce a longer, hotter gas flow path, and pre-heated combustion air introduced through small holes above the fuel in the firebox. Non-catalytic stoves have no catalyst to replace; however, some of the internal components of a non-catalytic stove need replacement as they degrade from high temperatures. Most available stoves are non-catalytic. Wood pellet stoves, which burn sawdust and other wood products pressed into small pellets, have grown in popularity and produce lower emissions than a conventional stove. Some pellet stoves can also burn dried corn kernels.

Some of alternative fuel options for wood stoves include manufactured firelogs and natural gas fireplaces, as well as the alternative heating devices described above (e.g., gas furnaces). (See Figure 18.4 for a comparison of average PM gram per hour emissions rates for various home heating devices.) Gas fireplaces are becoming increasingly popular, with newer models that mimic the look of a real wood fire. These self-contained units can be installed into an existing masonry fireplace. Manufactured firelogs, made from wax and sawdust for open-hearth fireplaces, can reduce $PM_{2.5}$ emissions. Studies suggest that manufactured firelogs can reduce $PM_{2.5}$ emissions by an average of 69 percent, relative to burning cordwood in a fireplace (Houck, 2000).

As indicated in Table 18.1, wood stoves and fireplaces are the leading sources of $PM_{2.5}$ emissions within the residential buildings sector. Table 18.3 presents $PM_{2.5}$ emissions factors for a conventional wood stove, catalytic EPA-certified stove non-catalytic EPA-certified stove, and pellet stove. Data come from two sources: (1) EPA's AP-42 emissions factor handbook and (2) a recent study prepared for the Hearth, Patio and Barbecue Association (Houck, 2005a).[6]

The benefits of replacing a conventional stove with an EPA-certified unit can be estimated based on the emissions factors presented in Table 18.3, but should also reflect the higher efficiency of the newer technology. A modern stove provides higher heating value for the same quantity of wood burned. An EPA-certified catalytic or non-catalytic stove is about 10 percent more efficient than a conventional wood stove, and a pellet stove is 20 percent more efficient

Table 18.3

Wood Stove $PM_{2.5}$ Emissions Factors

Stove Type	AP-42 $PM_{2.5}$ Emissions Factors (lb/ton)	Houck $PM_{2.5}$ Emissions Factors (lb/ton)
Conventional wood stove	30.6	66.8
Catalytic EPA-certified wood stove	16.2	15.1
Non-catalytic EPA-certified wood stove	14.6	11.7
Pellet stove	8.8	2.5

Source: EPA, 1996; Houck, 2005a

(Houck, 2005a). Higher-efficiency, cleaner-burning stoves also reduce creosote build-up and may reduce the number of chimney cleanings required.

The average cost of a new stove, including the cost associated with a chimney upgrade, is estimated at $2,500

6. In the Hearth, Patio and Barbecue Association study, the authors adjust the average emissions factors for conventional stoves because they assume lower burn rates than EPA assumed in its AP-42 Emission Factor Handbook. This results in higher emissions. Also, the study suggests lower average emissions factors for EPA-certified stoves, based on research demonstrating that the emissions performance of wood stoves has improved since the earliest models were produced in the early 1990s. Earlier models were used as the basis for the AP-42 emissions factors for EPA-certified stoves. EPA is currently in the midst of re-evaluating the AP-42 emissions factors for wood stoves, which may shed some light on the differences in estimated emissions rates between these two sources (Wood, 2005).

(Houck, 2005b). Assuming that a conventional wood stove with a $PM_{2.5}$ emissions factor of 66.8 pounds per ton (lb/ton) is replaced by a non-catalytic wood stove with a $PM_{2.5}$ emissions factor of 11.7 lb/ton, $PM_{2.5}$ emissions would be reduced by roughly 140 lb per heating season.[7]

According to EPA, preliminary estimates suggest that changing out wood stoves can be a cost-effective option for reducing $PM_{2.5}$ emissions. EPA estimates that this approach can cost less than $2,000 per ton of $PM_{2.5}$ reduced (Houck, 2005b). The *feasibility* of purchasing and installing a new wood stove is a separate issue, which depends on the income of the household. As discussed later in this chapter, grants have been used to fund the replacement of older stoves for low-income families.

The Mid-Atlantic Regional Air Management Association has published a document, *Emissions Inventory Improvement Program (EIIP): Residential Wood Combustion Coordination Project*, as an information resource for state, local and regional organizations that are planning or conducting residential wood combustion surveys. EPA recommends the use of surveys for the development of residential wood combustion emissions inventories.

Water Heating

Water heating is one of the largest uses of energy in the residential sector; it accounts for 17 percent of direct (primary) household energy use according to the Residential Energy Consumption Survey (2001).

In the residential sector, there are generally four destination points or end uses for hot water: faucets, showers, dishwashers and washing machines. Strategies are available that address these end uses, as well as the water heating systems themselves. Just some of the measures that are available for reducing the energy use and emissions associated with water heating are discussed here.

Reducing Hot Water Use. Installing low-flow showerheads and faucet aerators can save significant amounts of hot water. Low-flow showerheads can reduce hot water consumption for bathing by 30 percent. A top quality, low-flow showerhead will cost $10–$20 and can pay for itself in energy saved within four months. Repairing leaks in faucets and showers will also save hot water.

The biggest cost of operating a dishwasher stems from the energy required to heat the water before it ever makes it to the machine. An ENERGY STAR-qualified dishwasher will use less hot water than a conventional dishwasher (generally manufactured before 1994).

As with dishwashers, much of the cost—up to 90 percent—of operating clothes washing machines is associated with the energy needed to heat the water. Unlike dishwashers, clothes washers do not require a minimum temperature for optimum cleaning. Either cold or warm water can be used for washing most laundry loads; cold water is always sufficient for rinsing. Laundry detergents are currently available that are specially formulated for washing in cold water (Procter & Gamble, 2005). Most full-sized ENERGY STAR-qualified washers use 18–25 gallons of water per load, compared to the 40 gallons used by a standard machine.

Reducing Heating Load. Lowering the thermostat setting on a water heater can reduce energy use without any loss of convenience. Although some manufacturers set water heaters at 140 degrees Fahrenheit (°F) (or 60 degrees Celsius (°C)), 120°F (48.9°C) is satisfactory for most household needs. For every 10°F (5.6°C) reduction in water temperature, water-heating energy consumption can be reduced 3–5 percent.

Hot water tanks operate constantly to keep hot water available on demand. Timers are available that can automatically turn hot water heaters on and off, reducing unnecessary energy use, especially in older units. A timer costs approximately $30 and is easily installed. Timers for gas water heaters are less cost-effective than those designed for electric water heaters. Also, a timer will produce only limited energy savings when used with modern water heaters that have relatively low standby losses.

Insulating an older water heater storage tank is a fairly simple and inexpensive improvement that can help maintain the water temperature at the thermostat setting. Newer models of water heaters are well insulated and do not need an added layer. Hot water pipes can also be insulated to reduce heat loss, particularly in unheated areas.

Installing High Efficiency Water Heaters. Replacing an older water heater with a new, high-efficiency model can reduce energy use between 10 and 50 percent. Some of the technologies that are available for reducing energy use include higher-efficiency, conventional gas and electric storage tank water heaters, as well as alternative technologies that tend to be less widely used, such as tankless water heaters, heat pump water heaters and solar water heaters.

More stringent federal efficiency standards for conventional water heaters were recently introduced,

7. This example calculation also assumes an annual average cord usage of 1.75 cords per stove (adjusting the amount of wood used by the new stove to reflect its increased efficiency), and a cord-to-mass conversion factor of 1.4 tons per cord, which is reported to be typical of wood used in the eastern United States (Houck, 2005b).

effective January 20, 2004 (EPA, 2001a). The previous standards had been in place since 1991. Water heaters are not currently assigned ENERGY STAR designations. In January 2004, the Department of Energy (DOE) concluded that it would not be worthwhile to develop ENERGY STAR designations for conventional water heaters, since the differences between the best-performing products and those just meeting the minimum federal standard were deemed too small to justify an ENERGY STAR rating (Karney, 2004). For example, a 50-gallon electric tank water heater is required to achieve a minimum energy factor of 0.904 (Karney, 2004). The best performing model has a factor of 0.95.

According to DOE, heat pump, solar and tankless gas water heaters produce significant energy savings relative to conventional products. However, the Department again concluded that ENERGY STAR designations are not appropriate at this time because the payback periods for these technologies are too long (3.6–19 years) relative to other ENERGY STAR products (Karney, 2004). Also, DOE suggests that the infrastructure to sell and service some of these products is not fully developed.

Regulatory Authority

DOE is responsible for establishing minimum energy efficiency standards for several categories of home appliances, including refrigerators, freezers, air conditioners, fluorescent lamp ballasts, clothes dryers, clothes washers, dishwashers, kitchen ranges and ovens, and water heaters. These standards apply to newly manufactured products only and prohibit the production or import of appliances and other products less efficient than the minimum requirements. Current standards are available from DOE's Appliances and Commercial

Table 18.4

Products Subject to Existing Federal Efficiency Standards

National Appliance Energy Conservation Act of 1987 and 1988	Energy Policy Act of 1992	Energy Policy Act of 2005
Refrigerator-freezers	Fluorescent lamps	Traffic lights and pedestrian signals
Freezers	Incandescent reflector lamps	Building transformers
Room air conditioners	Electric motors (1-200 hp)	Torchiere lamps
Central air conditioners and heat pumps	Commercial packaged air conditioners and heat pumps	Compact fluorescent lamps and fluorescent lamp ballasts
Furnaces, boilers and water heaters	Commercial furnaces and boilers	Residential ceiling fans and ceiling-fan light kits
Direct-fired space heaters	Commercial water heaters	Residential dehumidifiers
Clothes washers and dryers	Showerheads	Commercial refrigerators and freezers
Dishwashers	Faucets and aerators	Commercial unit heaters and large commercial air-conditioners
Ranges and ovens	Toilets	Commercial ice makers and commercial clothes washers
Pool heaters	Distribution transformers[a]	Vending machines[a]
Fluorescent lamp ballasts	Small electric motors (<1 hp)[a]	Pre-rinse spray valves (typically found in dishwashers)
Televisions[a]	High-intensity discharge lamps[a]	Exit signs

a. Specific standards were not set in the legislation, rather DOE was instructed to investigate whether standards are technically feasible and economically justified and to adopt standards in response to their findings.

Source: Nadel, 2005; Edison Electric Institute, 2005

Equipment Standards Program website.[8]

Energy efficiency standards were first adopted at the state level when California passed its 1974 appliance efficiency laws for refrigerators, freezers and air conditioners. Other states, including Florida, Massachusetts and New York, followed California's lead by adopting standards for various products in the early and mid-1980s. In response to the resulting patchwork of state energy efficiency standards, Congress adopted national standards that preempted individual state requirements. Congress adopted national standards in 1987 (under the National Appliance Energy Conservation Act (NAECA) of 1987), in 1988 (when NAECA was expanded to include fluorescent ballasts), in 1992 (under the Energy Policy Act of 1992), and in 2005 (under the Energy Policy Act of 2005). The Energy Policy Act of 2005 establishes energy efficiency standards for several product categories, including dehumidifiers, ceiling fans and various other products. At the time the Act was adopted, Arizona, California, Connecticut, Maryland and New Jersey had already adopted similar efficiency standards. The products covered by federal efficiency standards are summarized in Table 18.4.

Other policy measures that have been used to promote household energy efficiency—apart from efficiency standards—include education programs, rebate programs and building code requirements. For example, the Energy Policy Act of 2005 provides limited tax credits for energy-saving home improvements implemented in 2006 and 2007. Eligible projects include installing insulation, new exterior windows (capped at $200), solar hot water systems and other home improvements. The bill also includes tax credits for contractors who build energy-efficient homes and for manufacturers who make energy-efficient appliances.

In 1988, EPA established New Source Performance Standards (NSPS) for wood stoves, requiring all new wood stoves sold in the U.S. to be certified to meet certain particulate emissions limits by 1992 (EPA, 1988).[9] EPA's mandatory smoke emissions limit for wood stoves is 7.5 grams of smoke per hour (g/hr) for non-catalytic stoves and 4.1 g/hr for catalytic stoves.[10] Some newer stoves have certified emissions in the range of 1–4 g/hr. This compares to an average emissions rate of approximately 42 g/hr for earlier models. Fireplaces and outdoor wood boilers are exempt from regulation under the NSPS. According to EPA, there are approximately ten million wood stoves in active use in the U.S., 75 percent of which are pre-NSPS stoves (Brockman, 2005b). A permanent label on a wood heater indicates that it meets the EPA emissions standards. A consumer information label, which specifies the emissions rate, the heating range of the wood heater and overall efficiency, is also required.

The Energy Policy Act of 2005 provides a rebate to consumers who purchase a renewable energy appliance with a thermal efficiency rating of 75 percent, including wood and wood pellet stoves. The bill provides a 25 percent rebate, capped at $3,000. DOE will be responsible for implementing the program.

Congress has also considered legislation that would provide a $500 tax credit for households in PM_{10} or $PM_{2.5}$ nonattainment areas that replace an old wood stove with an EPA-certified model (Wood Stove Replacement Act of 2005).

State and Local Policy Measures

A multitude of options are available for states seeking to reduce $PM_{2.5}$ and $PM_{2.5}$-precursor emissions associated with household energy use. Because of the number of measures available, we address only some of the options that are believed to offer significant potential for reducing emissions.

NO_x Emissions Standards for Gas Furnaces

In California, several air districts, including the South Coast Air Quality Management District (AQMD), the Bay Area AQMD and the San Diego County Air Pollution Control District, have adopted NO_x emissions standards regulating natural gas-fired central furnaces. The South Coast AQMD was the first to adopt a NO_x emissions standard for natural gas-fired central furnaces. The South Coast AQMD adopted its standard in 1978. The standard applies to central furnaces rated at less than 175,000 Btu per hour. Because the standards apply only to new furnaces, emissions will decline gradually as older, higher-emitting units are replaced and new homes are equipped with certified models. In 2005, the San Joaquin Valley Unified Air Pollution Control District (UAPCD) in California adopted a NO_x emissions standard for new residential furnaces identical to the South Coast AQMD standard: 0.093 lb/MMBtu of heat output or 55 ppm NO_x at 3 percent oxygen stack gas by volume (dry). According to District staff, the difference between the cost of manufacturing a low-NO_x emitting unit and a conventional model is minimal (San Joaquin Valley UAPCD, 2005a).

The South Coast AQMD has not updated its NO_x standard for natural gas-fired central furnaces since 1978. States should consider whether a more stringent standard is achievable based on new or existing technology. For example, researchers at the Lawrence Berkeley National Laboratory have developed a technology, known as

8. www.eere.energy.gov/buildings/appliance_standards/.

9. EPA's website provides a listing of EPA-certified wood stoves: www.epa.gov/compliance/resources/publications/monitoring/.

10. Wood stoves offered for sale in the state of Washington must meet a particulate emissions limit of 4.5 g/hr for non-catalytic wood stoves and 2.1 g/hr for catalytic wood stoves.

ultra-clean low-swirl combustion, that can reduce NO_x emissions to less than 9 ppm (at 3 percent oxygen) in certain applications (Chen, 2003). This compares to a current standard in California of 0.093 lb/MMBtu, which is approximately equal to 55 ppm (at 3 percent oxygen). To date, this technology has been licensed by Lawrence Berkeley National Laboratory for industrial process heaters and has been evaluated in a 15-kilowatt spa heater (where it reduced NO_x emissions from 150 ppm to less than 12 ppm). Thus far, the technology has not been used in a residential furnace application, although the potential does exist to adapt the technology for that application.

Low-Sulfur Diesel Fuel

Significant potential exists for reducing SO_2 emissions from residential boilers and furnaces by substituting low- (500 ppm) and ultra-low- (15 ppm) sulfur oil for current home heating oil, which averages 2,500 ppm sulfur content (Bookhart, 2003).

The New York State Energy Research and Development Authority (NYSERDA) and Brookhaven National Laboratories recently conducted a three-year demonstration project using low-sulfur fuel for residential heating. One thousand homes participated in the study, which used diesel fuel with a sulfur content of 500 ppm. Over the course of the project, use of low-sulfur oil caused maintenance and service requirements to decline substantially, while at the same time dramatically reducing SO_2 emissions (EPSA, 2005).

According to the Empire State Petroleum Association, heating oil companies are willing to market low-sulfur fuel oil to their customers; however, they have been discouraged from doing so because state tax laws treat the low-sulfur fuel as a motor vehicle fuel, subjecting it to the same taxes that are imposed on highway diesel fuel (EPSA, 2005).

Legislators in several states have proposed low-sulfur requirements for home heating oil. Rhode Island Senate Bill 962 and House Bill 6247 would limit the sulfur content of home heating oil to 500 ppm by June 1, 2006. However, both bills failed to come up for a formal vote (Moura, 2005; Sosnowksi, 2005). In New York State, legislation is pending that would direct the Department of Environmental Conservation (DEC) to enact regulations limiting allowable sulfur levels in home heating fuel (Bill No. A6453).

Appliance Efficiency Standards for Boilers and Furnaces

As described above, residential furnaces have been subject, under NAECA of 1987, to a minimum Annual Fuel Utilization Efficiency requirement of 78 percent since 1992. However, furnaces are currently available with efficiency ratings of 90 percent or greater. According to the Boston-based Consortium for Energy Efficiency, Inc., 22 percent of homes built in 2002 were equipped with furnaces that achieved 90 percent Annual Fuel Utilization Efficiency; moreover, as natural gas prices increase, the cost-effectiveness of these higher cost units should continue to improve (Home Builders Association of Metro Denver, 2004). DOE is currently developing updated standards for residential furnaces and boilers; however, the rulemaking has been delayed and is not expected to be completed until fall 2007 (ACEEE, 2005). As a result, the American Council for an Energy-Efficient Economy recommends that individual states adopt minimum efficiency standards for residential furnaces and boilers. Because federal law already covers residential furnaces, states will need to apply for a waiver of federal preemption.

Wood Stove Burn Bans

Local agencies throughout the U.S. have instituted restrictions on wood stove use during periods of poor air quality. For example, since 1988, the Puget Sound Clean Air Agency has instituted temporary restrictions on indoor and outdoor wood burning whenever PM levels exceed a predetermined level. The bans are instituted in two stages. During a first stage ban, residential and commercial buildings are prohibited from burning wood in any stove or fireplace that is not EPA-certified. During a second stage ban, all buildings are prohibited from burning wood in either a certified or uncertified wood stove. Buildings that do not have an adequate source of heat without burning wood are exempted from the bans. In 2005, the state legislature adopted a bill revising the thresholds for a burn ban (HB 1032: Modifying Burn Ban Triggers, 2005). Over the past five years, the number of burn bans in a given year has ranged from zero to 12 (Puget Sound CAA, 2005). Many other localities have instituted similar

Resources available from EPA's wood stove and fireplace website:

- How-To Guide for Implementing a Wood Stove Changeout Campaign
- Fact Sheet: Burn Clean-Hot Tips for a Better Fire
- Guidance for Supplemental Environmental Projects
- Public education materials on clean burning wood stoves and fireplaces
- Additional resources for air quality program officials

http://www.epa.gov/woodstoves

measures.

Wood Stove Changeout Programs

A growing number of communities have been exploring wood stove changeout programs as an option for addressing wintertime particulate air pollution. In 2005, EPA provided funding and technical support for the development of two model programs. A panel convened by the National Academies of Sciences (NAS) has encouraged the Agency to pursue this type of initiative (NAS, 2004). EPA and its project partners initiated one such program in June 2005 in Libby, Montana, a small town in western Montana. Outside of California, Libby is the only area in the West that currently exceeds the $PM_{2.5}$ standard, and wood combustion contributes more than 80 percent of the area's $PM_{2.5}$ load during the winter season (Houck, 2005b). The hearth industry has provided over $1 million in wood stoves and other products to assist in this campaign (Wood, 2005). The Libby experience is expected to be a test case for eastern states considering similar programs.

On September 29, 2005, the Southwest Pennsylvania Air Quality Partnership announced the start of its wood stove changeout program with a $100,000 grant from EPA and others, and an $80,000 grant from the Allegheny County Health Department. The program offers free changeouts for low-income families and replacements at a discount for other residents (Allegheny County, 2005).

EPA's website, www.epa.gov/woodstoves/changeout.html, provides additional information on EPA's wood stove changeout program, including technical resources for launching a local initiative. Other communities, such as Delta County, Colorado, have offered rebates or low interest loans for stove changeouts. In the Delta County program, the amount of the rebate varied depending on the efficiency and type of the new model. For instance, higher rebates were awarded to households that installed pellet or gas stoves.

EPA is developing a guidance document on quantifying the emissions reductions associated with wood stove changeouts for state agencies seeking to use emissions reductions from such programs for State Implementation Plan (SIP) credit. More information should be available from EPA's website when the guidance is completed.

Finally, EPA includes wood stove changeouts in its list of ideas for supplemental environmental projects, or SEPs (EPA, 2005b). A SEP is a project that the defendant in an enforcement action agrees to perform or fund in partial settlement of the case.

Wood Stove Education Campaigns

Between 1999 and 2003, Natural Resources Canada led a multi-stakeholder committee in designing and implementing a national wood heat education campaign, now called *Burn it Smart*. *Burn it Smart* aims to reduce emissions from residential wood combustion by educating consumers on the health and safety aspects of wood combustion, proper methods of wood burning, and the technologies that are available for reducing emissions. State and local agencies seeking to educate the public on these topics will benefit from the educational brochures, fact sheets and poster displays developed by the program. Also, the program conducts community workshops for the general public and industry professionals in an effort to reduce emissions. More information is available at www.burnitsmart.org.

Regulation of Residential Wood Boilers

Over the past year, outdoor residential wood boilers have become a focus of heightened concern and state regulatory attention because of their high air pollution emissions rates and growing popularity. Vermont has proposed a rule to regulate new wood boilers, and the New York Attorney General has petitioned EPA to promulgate NSPS standards for the boilers or revise the existing standards for residential wood stoves to include outdoor wood boilers (Spitzer, 2005a). EPA is considering its response to this petition and has been meeting with boiler manufacturers to discuss options for reducing emissions. Several manufacturers have indicated that they could significantly reduce emissions from new models in less than one year (Wood, 2005). Additional states are encouraging EPA's efforts as the Agency weighs its options, and some states are pursuing their own regulations. A report by the New York Attorney General's Office provides an example of a local ordinance regulating the use of wood boilers (Spitzer, 2005b). The report also provides background information on the cost and performance of wood boiler technology. A federal response could avoid a patchwork of state regulations, both in terms of the performance standard and the test method used to determine emissions performance.

EPA and others are participating in a new ASTM committee that is working to develop a consensus test method and potentially a consensus emissions standard for outdoor wood boilers. This will be a multi-year process (Wood, 2005).

Wood Stove Mandatory Removal Programs

Several communities in California and Nevada, including the San Joaquin Valley, Mammoth Lakes and Reno, require all non-EPA-certified wood stoves to be removed or retired prior to the sale or transfer of a property. This strategy can be a powerful method for accelerating the removal of older stoves and is generally feasible because the cost of removing a stove usually constitutes an insignificant share

of the property sale price.

Outdoor Burning Restrictions

The primary strategy for addressing emissions from open burning is to ban the practice, while providing waste disposal alternatives. Public education and consistent enforcement are also key to program success. Increasing household waste pickups, establishing and promoting free or low-cost drop-off centers, and educating residents about disposal alternatives will all serve to discourage open burning (EPA, 2001b; EPA, 2003).

Connecticut, Maine, Massachusetts, New Jersey, New Hampshire and Vermont have statewide bans on backyard burning of household waste. New York distributes brochures on the hazards of backyard burning. Forsyth County strictly enforces North Carolina's statewide ban against burning garbage by penalizing first-time offenders with fines ranging from $100 to $10,000 (EPA, 2003).

In response to high rates of residential garbage burning in northeastern Minnesota and northwestern Wisconsin, the Western Lake Superior Sanitary District has developed a toolkit of information for local officials to assist them in addressing the problem. The toolkit includes a model ordinance to regulate outdoor burning, public education materials to alert people to the hazards of burning household waste, case studies of successful programs, as well as other resources. The toolkit is available at www.wlssd.com/Open_Burning/Clearing_the_Air_downloadvs.pdf.

Building Envelope and Appliance Efficiency Incentives

The Energy Policy Act of 2005 provides more than $2 billion in tax incentives for advanced energy-saving technologies and practices. Most of these incentives are available in the period 2006–2007 (Nadel, 2005). State policymakers should familiarize themselves with the various provisions of the bill and consider complementary programs to take maximum advantage of the opportunities available. For example, the legislation provides a $2,000 tax credit for new homes that use 50 percent less energy for space heating and cooling relative to homes built pursuant to the 2003 International Energy Conservation Code. The bill also provides tax credits for high-efficiency appliances such as central air conditioners, heat pumps, furnaces and water heaters. State and local energy programs can help in promoting awareness of the tax credits and other provisions of the bill. A complete discussion of these opportunities is available in a report prepared by the American Council for an Energy-Efficient Economy and entitled *The Federal Energy Policy Act of 2005 and its Implications for Energy Efficiency Program Efforts* (Nadel, 2005).

Other Measures

On December 15, 2005, the San Joaquin Valley UAPCD adopted a rule (Rule 9510) requiring developers of larger residential, commercial and industrial projects to adopt measures to reduce NO_x and PM emissions from projected baseline levels. The rule requires developers to reduce NO_x emissions by 33 percent and particulate emissions by 50 percent of a project's baseline. If projected emissions are forecast to exceed the minimum baseline reductions, the developer can offset the difference by paying a fee to the Air District, which is then used to fund clean air projects. Developers have several options for offsetting their baseline emissions, including: (1) not installing wood stoves, (2) increasing energy efficiency, (3) locating homes close to retail outlets or (4) creating higher density developments (San Joaquin Valley UAPCD, 2005b).

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Chapter 19

Commercial Cooking

Introduction

Commercial cooking establishments—including restaurants, industrial-scale food processing facilities and street vendors—are responsible for preparing a growing share of the food consumed in the U.S. One indicator of this trend is the increase in meals consumed outside the home. In 1970, Americans devoted 34 percent of their total food budget to meals consumed away from home. Today, these expenses account for more than 40 percent of total U.S. food expenditures (Dumagan, 1995; USDA, 2005). Demand for ready-made meals prepared in the home has also increased in response to economic and cultural trends (Woon, 2005).

Commercial cooking operations produce a mixture of solid, liquid and gaseous particles, including water and grease, non-condensable gases and solid organic matter. Some portion of the water and grease vapor condenses on the ventilation equipment, while remaining particles and grease droplets are released to the atmosphere. How much of the vapor is condensed rather than released depends on a host of factors.

According to EPA, the majority of PM emissions from the commercial cooking sector come from the cooking of meat products (Roe, 2004). In particular, charbroiling generates over 80 percent of total $PM_{2.5}$ emissions from the commercial cooking sector (Roe, 2004).

Commercial cooking operations use two types of charbroilers: underfired and chain-driven. The vast majority of emissions come from the use of underfired charbroilers, although regulatory efforts have focused on the control of chain-driven charbroilers used by fast-food restaurants (Perryman, 1997).

Emissions rates from commercial cooking vary by type of appliance, type of meat, the fat content of the meat and cooking conditions (McDonald, 2003). High-fat meats cooked on an underfired charbroiler produce the highest fine particulate ($PM_{2.5}$) emissions. PM emissions from meat cooking are primarily in the $PM_{2.5}$ size range (McDonald, 2003).

This chapter discusses the technology and policy options available for reducing $PM_{2.5}$ emissions from commercial cooking establishments.

Sector Profile

There are over 900,000 restaurants in the U.S. and they employ more than 12 million people, making the commercial cooking sector the nation's second largest employer. (Only the government sector employs more people in the U.S.) Industry sales in 2004 were $476 billion, an 11-fold increase from 1970 sales of $43 billion (National Restaurant Association, 2005).

Commercial cooking establishments are spread throughout the U.S. and are expected to continue growing in number as the population increases. Between 2000 and 2020, the U.S. population is expected to grow by between 50 and 80 million people. In part because of the larger population and in part because of greater affluence, *total* U.S. food expenditures are projected to rise a further 26 percent between 2000 and 2020, from $800 billion in 2000 to over $1 trillion in 2020 (Blisard, 2002). While not all of this food will be prepared at commercial cooking facilities, the amount of food purchased outside the home is also expected to increase. For example, the proportion of meat consumed outside the home is expected to rise between 2000 and 2020, as shown in Table 19.1. (As mentioned above, the cooking of meat is the largest source of emissions from commercial cooking operations.) As more Americans go out for meals, emissions from the commercial cooking sector may well become a more substantial air pollution problem.

EPA's National Emissions Inventory divides the commercial cooking sector into five categories: chain-driven (or conveyorized) charbroilers, underfired charbroilers, deep fat fryers, flat griddle fryers, and clamshell griddle fryers.

Chain-driven Broilers. Chain-driven charbroilers, commonly used at fast food restaurants, use a conveyer belt to carry the food product through the heat source, often cooking both sides of the food in a single pass. Most chain-driven charbroilers are fueled by natural gas, although electric models are also available (Perryman, 1997).

Underfired Charbroilers. Underfired charbroilers resemble a barbecue and have a heat source below a heavy metal grid, which marks the food with a distinctive striping. Some restaurants make their charbroiler, which generates flames and smoke, a showpiece within their establishment (Fisher, 2002). Examples include broilers, grill charbroilers, flame broilers and direct-fired barbecues. Charbroilers generate particulate emissions when grease falls from the meat onto the radiant surface. Unburned drippings collect in a grease tray. Most charbroilers burn natural gas, but some use charcoal or wood with ceramic stones to distribute the heat (Perryman, 1997). Because of the smoke they generate, charbroilers require significant ventilation (Fisher, 2002).

Deep Fat Fryers. Deep fat fryers use an exposed hot metal surface to heat cooking oil, which is then used to cook the food. Most raw food products contain a high percentage of water by weight (10–75 percent), and most of the water at the surface vaporizes during the cooking process, creating an oily mist (Perryman, 1997). Fast-food establishments use deep fat fryers to prepare food in batches.

Griddles and Griddle Hybrids. Flat griddles are appliances that consist of an exposed metal plate used to fry food. Most griddles are gas-fired, although some are electric. The Clamshell® griddle is a newer type of griddle that employs a two-sided cooking configuration to reduce cooking time, thereby also reducing emissions (Perryman, 1997).

Table 19.1

Proportion of meat consumed outside the home, 2000–2020

Type of Meat	Commodity Market Share (%) 2000		Commodity Market Share (%) 2020	
	At home	Outside	At home	Outside
Beef	67	33	65	35
Pork	79	21	77	23
Poultry	68	32	69	31
Fish	66	34	64	36
Other meat	77	23	76	24

Source: Blisard, 2002

Table 19.2

National Commercial Cooking Sector Emissions, 2002

Equipment Type	PM_{10}		$PM_{2.5}$	
	Tons/year	Percent of sector emissions	Tons/year	Percent of sector emissions
Underfired charbroiling	60,304	71%	58,295	74%
Chain-driven charbroiling	8,460	10%	8,201	10%
Deep fat frying	ND	ND	ND	ND
Flat griddle frying	15,679	18%	11,916	15%
Clamshell griddle frying	1,073	1%	909	1%
Total	**85,515**	**100%**	**79,321**	**100%**

ND = no data

Source: Roe, 2004

Sector Emissions

A revised inventory methodology developed by EPA and Pechan Associates in 2004 estimates that commercial cooking establishments generate nearly 80,000 tons of $PM_{2.5}$ emissions per year (Roe, 2004).[1] (The updated methodology resulted in a five-fold increase in estimated $PM_{2.5}$ emissions from the sector relative to the 1999 inventory.)

To put this estimate in perspective, 80,000 tons is equivalent to 6 percent of the total *direct* $PM_{2.5}$ emissions that EPA estimates were generated by all point source categories in 1999 (e.g., power plants and industrial facilities). (This is 1.4 percent of the 1999 total direct $PM_{2.5}$ area source emissions inventory.)

1. EPA's revised estimate does not include all cooking processes. For example, it does not include residential or special-event cooking, or cooking processes at institutional facilities. The estimate also does not include criteria pollutant emissions associated with fuel combustion by cooking equipment. Emissions from fuel combustion are instead included in the commercial fuel combustion emissions sector of the National Emissions Inventory (Roe, 2004).

Table 19.2 reports estimated $PM_{2.5}$ emissions from the commercial cooking sector (meat and potatoes only) by equipment type. Underfired charbroilers produced 74 percent of $PM_{2.5}$ emissions, flat griddle frying produced 15 percent, and chain-driven charbroilers produced 10 percent. Particulate emissions from deep fat fryers are below detectable levels (Whynot, 1999).

Particle emissions from meat cooking are primarily in the $PM_{2.5}$ size range (McDonald, 2003). For example, according to EPA's revised inventory data, 97 percent of the PM_{10} emissions attributable to underfired charbroiling are in the fine particle size range (see Table 19.2).

Sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions from the commercial cooking sector are generally not addressed in the literature. Studies of commercial cooking focus on PM, volatile organic compounds (VOCs), and their specific components. NO_x and SO_2 emissions are attributable primarily to the energy use associated with commercial cooking equipment, as opposed to the actual cooking process. The emissions associated with electricity use are addressed in Chapter 7, *Electric Generating Units*.

Table 19.3

$PM_{2.5}$ Emissions Rates from Cooking Meat (lb per unit of meat)

Meat	Fat Content	Appliance	$PM_{2.5}$/1000 lb
Hamburger	21%	Chaindriven charbroiler	4.5–7.4
	25%	Underfired Charbroiler	15.0–40.0
	21%	Underfired Charbroiler	7.1
Steak	3%	Underfired Charbroiler	7.8
Chicken with skin	10-30%	Underfired Charbroiler	7.2–10.4

Source: McDonald, 2003

The $PM_{2.5}$ emissions rates for charbroiling hamburger, steak and chicken range from 4.5 to 40.0 pounds (lb) per 1,000 lb of uncooked meat. High-fat hamburgers (25 percent fat) cooked with an underfired charbroiler produce the highest amount of $PM_{2.5}$ (McDonald, 2003). Table 19.3 summarizes the $PM_{2.5}$ emissions rates of several meat/appliance combinations.

Emissions Control Opportunities

Most $PM_{2.5}$ emissions from commercial cooking establishments are generated when grease from the meat falls into the heating element or flame and combusts. Controlling emissions from the largest sources of emissions, underfired charbroilers, can be very expensive because of the relatively low emissions generated by any single restaurant and the lack of low-cost control options. Emissions from chain-driven charbroilers are more cost-effective to control and are being regulated in southern California by the South Coast Air Quality Management District (AQMD).

Chain-Driven Charbroilers

Chain-driven charbroilers generate PM emissions when grease from the cooking meat falls onto the high temperature radiant surface. The most common technology used to control these emissions is the catalytic oxidizer (Perryman, 1997).

A flameless catalytic oxidizer contains a bed of inert ceramic material coated with a metal catalyst that oxidizes smoke and gases from the cooking process, converting them to carbon dioxide and water. The system is mounted into the charbroiler ventilation duct. Engelhard manufactures a food-service catalytic oxidizer (Engelhard, 2004).

In experiments using a catalytic oxidizer, controlled emissions from chain-driven charbroilers averaged 1.29 lb of PM emissions per 1,000 lb of hamburger, an 83 percent reduction from high-end estimates of uncontrolled emissions (Whynot, 1999). Catalytic oxidizers also reduce VOC emissions.

A catalytic oxidizer adds an estimated $3,700 to the price of a new conveyorized charbroiler (Sacramento Metropolitan AQMD, 2005). For one restaurant in California, the installation of a catalyst also cut duct cleaning from about four to about two times per year, saving approximately $320 per year. In addition—because the catalyst reflects heat back into the charbroiler—natural gas use at the restaurant was reduced by about 7 percent (Ventura County APCD, 2004a).

A large number of restaurants in southern California have installed catalytic oxidizers. The cost-effectiveness of installing catalytic oxidizers on commercial charbroilers is estimated by South Coast AQMD staff at $1,680–$2,800 per ton of PM and VOCs reduced. Restaurant operators, commenting on South Coast AQMD's draft rule, have reported costs of more than $7,000 per ton of emissions reduced (Perryman, 1997). However, according to the South Coast AQMD, this higher cost estimate represented a unique outlier situation where more extensive structural changes to the cooking facility were required (South Coast AQMD, 1997b).

Underfired Charbroilers

Underfired charbroilers produce four times the emissions generated by chain-driven charbroilers per unit of product cooked, and are estimated to generate 74 percent of the sector's $PM_{2.5}$ emissions (Whynot, 1999; Roe, 2004). However, controlling emissions from underfired charbroilers can be very expensive per ton of PM reduced. According to the South Coast AQMD, catalytic oxidizers are not effective with underfired charbroilers because the exhaust temperatures are not sufficient to promote an efficient reaction (Perryman, 1997).

The South Coast AQMD, which has evaluated options for controlling PM_{10} emissions from underfired charbroilers since 1991, concluded in December 2004 that cost-effective control options are not available. In the South Coast AQMD, underfired charbroilers contribute 83 percent of the restaurant sector PM_{10} inventory (South Coast AQMD, 2004).

The South Coast AQMD has focused on two control options: (1) the replacement of underfired charbroilers with Smokless™ broilers, and (2) an add-on scrubber manufactured by CAST, Inc. (South Coast AQMD, 2004).

In 1999, about 70 restaurants in the U.S. used the Smokless™ broiler. This broiler is not a control device, but a complete alternative to the traditional underfired charbroiler. Unlike the underfired broiler, the Smokless™ broiler is not a direct-flame cooker and may result in a product that differs in appearance and taste. This contributed to the decision by South Coast AQMD staff to recommend *against* a rule that would require underfired charbroilers to be replaced with this technology or with some other alternative cooking device. The Smokless™ broiler's capital cost is approximately $5,550, compared to about $3,100 for a similarly sized underfired charbroiler. Replacing an underfired charbroiler with a Smokless™ broiler is reported to reduce PM_{10} emissions at a cost of $3,550 per ton (South Coast AQMD, 2000).

A second emissions control option for underfired charbroilers is the CAST, Inc. add-on scrubber. For a very large restaurant (cooking 2,800 lb of meat per week), the South Coast AQMD estimates a cost-effectiveness of

$1,100–$7,300 per ton of PM_{10} reduced, depending on the type of meat cooked and the volume of make-up air saved by the device (South Coast AQMD, 2004).[2] According to the manufacturer, there are roughly 12 CAST scrubbers installed throughout the U.S. and the system has been improved over the past several years to increase energy savings associated with heating, ventilation and air conditioning (Hopkins, 2005).

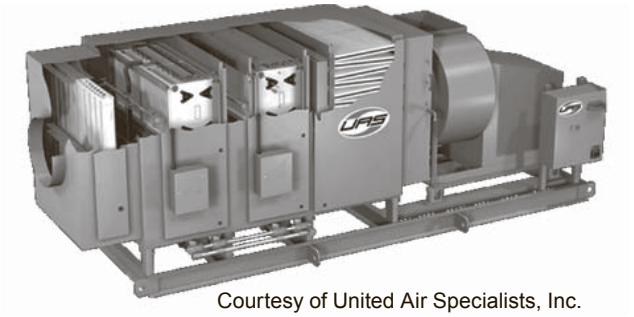
An electrostatic precipitator (ESP) can also be used to control PM emissions. The ESP used for controlling charbroiler particulate emissions (called the "Penney-type" ESP) involves a two-stage process. The exhaust stream enters the first stage from the hood, where it is charged; the charged particles are then collected in the second stage. The South Coast AQMD warns that improper cleaning can interfere with particulate removal efficiencies (Perryman, 1997). There are a few dozen ESPs operating at restaurants located in the South Coast AQMD area (Whynot, 1999). The South Coast AQMD estimates a cost-effectiveness of $10,000 per ton of PM reduced for an ESP (Perryman, 1997).

United Air Specialists, Inc. manufactures an ESP system, known as the Smog-Hog® Air Pollution Control System, that is designed specifically for kitchen emissions control. The system is used by hotels, high-rise buildings, airports, casinos, food courts and sports and entertainment complexes throughout the U.S., including McDonald's, Pizza Hut, Burger King (San Francisco Airport), Planet Hollywood, Four Seasons Hotel Boston and Mohegan Sun Casino (United Air Specialists, undated).

Additionally, wet scrubbers are capable of PM removal efficiencies of 90 percent or higher, depending on particle size, load, flow rate and design pressure drop. Wet scrubbers rely on a finely atomized stream of liquid to capture particulate and gaseous pollutants from an exhaust stream, like that of a restaurant charbroiler. There are several wet scrubbers currently permitted in restaurants in the South Coast AQMD area (Perryman, 1997). The South Coast AQMD estimates a cost-effectiveness of $13,000 per ton of PM reduced for a wet scrubber (Perryman, 1997).

2. Make-up air refers to unconditioned air brought in from the outdoors.

United Air Specialists, Inc. Smog-Hog®



Courtesy of United Air Specialists, Inc.

Apart from these add-on controls, the design of a charbroiler's metal grid can influence the amount of smoke generated by the device. Some grids are designed with a series of gutters that channel grease away from the flames of the broiler and toward the grease pan (Fisher, 2002). Some models have a slanted grid that also serves to divert grease away from the flames. No estimates are currently available regarding the effectiveness of these designs in reducing $PM_{2.5}$ emissions. Also, griddles—which do not have an exposed flame—are available that can create the characteristic stripes of an underfired charbroiler with far less smoke and more efficient energy use (Fisher, 2002).

Flat Griddle and Clamshell® Frying

Regulators have not focused on particulate emissions from flat griddle frying. As shown in Table 19.2, flat griddle fryers account for 15 percent of $PM_{2.5}$ emissions.

The Clamshell® broiler features an infrared broiler-hood mounted over a griddle. This two-sided cooking surface reduces cooking time, thereby decreasing emissions (Fisher, 2002).

Energy Conservation Measures

Charbroilers can be large consumers of energy, as they generally idle at a rate close to their full heat input to be ready for the next round of cooking. Testing in Pacific Gas and Electric Company's test kitchen revealed that a gas underfired broiler has a cooking efficiency of only 5 percent when cooking 100 lb of food over an eight hour period. The heat generated by a charbroiler can also contribute to cooling loads in a kitchen, further contributing to a restaurant's energy needs. Although control systems have been evaluated that would automatically reduce gas flow when a charbroiler is on but not cooking, this technology is not yet commercially available (Fisher, 2002). Further investigation may be warranted in this area in light of the potential energy savings.

Regulatory Authority

There are no federal regulatory requirements for controlling $PM_{2.5}$ emissions from restaurants, and state and local air agencies have generally not been active in regulating commercial cooking facilities. The South Coast AQMD regulates chain-driven charbroilers, as discussed below. The South Coast AQMD also lists catalytic oxidizers in its best available control technology (BACT) guidelines for controlling PM_{10} emissions from chain-driven charbroilers (Perryman, 1997).

Local public health departments may regulate opacity (visible smoke) and odors from commercial cooking establishments in residential or commercial areas. The strategies used to comply with these regulations may also

reduce $PM_{2.5}$ emissions (e.g., installation of a catalytic oxidizer).

State and Local Policy Measures

South Coast AQMD Rule 1138 regulates chain-driven charbroilers (Roe, 2004). It requires operators of both new and existing chain-driven charbroilers to install a catalytic oxidizer. Alternative control devices can be used if they are at least as effective as a catalytic oxidizer in reducing PM and VOC emissions. Chain-driven charbroilers cooking less than 875 lb of meat per week are exempt (South Coast AQMD, 1997a).

Other areas have adopted or have considered adopting the South Coast AQMD rule. The Ventura County Air Pollution Control District and the San Joaquin Valley Unified Air Pollution District, both in California, have adopted the South Coast AQMD rule (Ventura County APCD, 2004b; San Joaquin Valley UAPD, 2002). New Jersey is considering adopting the South Coast AQMD standards for chain-driven charbroilers, and also recommends investigating the use of the Smog-Hog® ESP and catalytic controls on some operations (New Jersey Homes and Restaurants Workgroup, 2005).

The South Coast AQMD conducted extensive research on emissions controls for underfired charbroilers and concluded that none of the options now available met its cost-effectiveness criteria; accordingly, the agency chose not to regulate these charbroilers. Staff have recommended that the Board of Governors look to other industry sectors for more cost-effective particulate reductions (South Coast AQMD, 2004).

In nonattainment areas around Aspen, Colorado, Chapter 6.12 of the Pitkin County Code requires commercial cooking operations to maintain a control device on charbroilers installed on or after January 1, 1993. The control device must reduce uncontrolled PM_{10} emissions by at least 90 percent (Pitkin County Code, 2004). The Pitkin County Code also prohibits the cooking of high fat-content meat (greater than 15 percent precooked fat content by weight) on charbroilers installed after April 25, 1983 but before January 1, 1993, unless an emissions control device has been installed that reduces uncontrolled PM_{10} emissions by at least 90 percent (Pitkin County Code, 2004). Finally, any commercial cooking facility within specified zones in Pitkin County that proposes to extensively remodel its kitchen is required to install a grooved griddle or an equivalent low-emissions cooking device if high fat-content meats are regular menu items. Pitkin County required such controls to be installed on existing restaurants by December 1, 1984 (Pitkin County Code, 2004).

The State of Colorado has published a "Guide to Environmental Regulations for Restaurants and Food Service Businesses." This document instructs facilities to check for smoke from outside hood stacks and to use good cooking practices; it also suggests that restaurants with chain-driven charbroilers consider installing a catalytic converter and that other restaurants with underfired charbroilers consider using a scrubber or ESP (Colorado Department of Public Health and the Environment, 1999).

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Chapter 20

Fugitive Dust

Introduction

Fugitive dust refers to particles, most commonly derived from soil, that are lifted and suspended in the ambient air by human activities and natural forces, such as agricultural tilling, motor vehicle use and wind. Fugitive dust does not include particulate matter (PM) emitted directly from stationary or mobile sources (EPA, 2005a).

The major sources of fugitive dust are paved and unpaved roads, agricultural operations, construction projects and wind erosion from both agricultural and non-agricultural lands (Pace, 2005). Fugitive dust is comprised primarily of coarse PM (between 10 and 2.5 micrometers (µm) in diameter). $PM_{2.5}$ (particles 2.5 µm or less in diameter) is a smaller share of the mix. For example, a study that measured the PM emissions from the unpaved roads of two sand and gravel plants in California found that $PM_{2.5}$ contributed 15 percent of total PM_{10} emissions (Hayden, 2005). A separate study by EPA that reviewed several studies suggests that $PM_{2.5}$ accounts, on average, for 10 percent of fugitive dust PM_{10} emissions (Pace, 2005).

According to EPA's 1999 National Emissions Inventory, fugitive dust is a major component of the direct $PM_{2.5}$ inventory, contributing roughly 50 percent of total direct emissions. EPA estimates that vehicle travel on unpaved roads was responsible for 1.3 million tons of $PM_{2.5}$ emissions in 1999. This is approximately equal to the total direct $PM_{2.5}$ emissions that EPA estimates all point source categories (e.g., power plants and industrial facilities) generated in 1999. EPA estimates that vehicle travel on paved roads produced roughly 700,000 tons of $PM_{2.5}$ in 1999, or roughly half the quantity generated from unpaved roads. Agricultural sources (crops and livestock) are also reported to be a major source of fugitive dust, generating just under one million tons of $PM_{2.5}$ in 1999. See Table 20.1 for EPA's estimates of 1999 fugitive dust emissions.

However, experts suggest that the current inventory has overstated these emissions, in part because of the multipliers that are used to infer the emissions of $PM_{2.5}$ from PM_{10} monitoring data (Pace, 2005). (For example, a multiplier of 0.20 would imply that $PM_{2.5}$ accounts for 20 percent of PM_{10} emissions.) Currently, EPA's AP-42 emissions factors are based on fine to coarse ratios ranging from 0.15 to 0.40 for most fugitive dust sources. However, a recent study by the Western Regional Air Partnership (WRAP) suggests a fine to coarse ratio of 0.10 for fugitive dust. According to WRAP, this bias helps explain why researchers often see a discrepancy in the proportion of fugitive dust found in $PM_{2.5}$ emissions inventories as compared to the proportion on ambient filter samples (WRAP, 2005). Sources also suggest that sampling methods may be overstating the PM_{10} emissions rates of agricultural sources and other categories, from which the $PM_{2.5}$ estimates are derived (WRAP, 2005; Comis, 2004). Efforts to estimate the fine to coarse ratio and to address other fugitive emissions factor issues continue and will likely continue to evolve subsequent to publication of this report. Nevertheless, given that fugitive dust is a

Table 20.1

$PM_{2.5}$ Emissions: Fugitive Dust and Agricultural Burning Sources (1999)

Source Category	$PM_{2.5}$ Emissions (Tons)	Percent of Source Category Emissions (%)
Unpaved roads	1,303,858	37
Paved roads	687,856	20
Agriculture (crop production)	861,172	25
Agriculture (livestock)	88,324	3
Agricultural burning	151,816	4
Construction	384,649	11
Total	**3,477,675**	**100**

Source: EPA, 2005c

major component of the direct $PM_{2.5}$ inventory, controlling emissions from paved and unpaved roads can be an effective way of reducing an area's direct $PM_{2.5}$ emissions.

In the discussion that follows, we focus on the options for reducing PM emissions from paved and unpaved roads (including construction sites) and from agricultural sources. Additional options beyond the strategies discussed in this chapter are available for reducing emissions from construction and demolition activities.

Profile: Paved and Unpaved Roads

According to the Federal Highway Administration, there are nearly four million miles of roads in the U.S. (DOT, 2002a). Roads cover more than 1 percent of the country, an area the size of South Carolina (DOT, 2002a; Gewertz, 2003). Sixty-three percent of this total is paved (2.50 million miles), with the other 37 percent unpaved (1.45 million miles) (DOT, 2002b). In 1999, American automobiles traveled nearly 1.6 trillion miles, or an average of 5,704 miles per capita.

Paved Roads

Paved roads generate PM emissions when motor vehicles and wind cause loose material from the road (including brake and tire wear) to be suspended above the road's surface. The loose material, known as "surface loading," is replenished by various sources, such as snow and ice removal compounds, erosion from adjacent areas and other debris (EPA, 2003a). Trucks carrying dirt and other materials can also contribute to surface loading as a result of spillage, and cars and trucks contribute "trackout" from unpaved roads and work areas. Trackout is the deposition of mud, dirt or similar debris onto the surface of a paved road from the tires and undercarriage of a motor vehicle.

The quantity of emissions that a road generates depends on the speed of the vehicles traveling the road, the traffic levels, the fraction of heavy vehicles (buses and trucks), and the presence or absence of parking lanes and curbs (EPA, 2003a). Emissions from paved roads also vary with the silt loading of the road. Silt loading refers to the mass of silt-size material (under 75 µm in physical diameter) per area of travel surface (EPA, 2003a). In the colder regions of the country, road surface loadings are heaviest during the late winter and early spring months, when the residual loading from snow and ice controls is greatest. In warmer weather, when snow and ice controls are no longer used, the road surface loadings decline significantly (EPA, 2003a).

EPA's PM emissions estimation methodology for fugitive dust from paved roads is in Section 13.2.1 of AP-42 and was last updated in December 2003. Section 7.9 of the California Air Resources Board (CARB) Emission Inventory Methodology provides an alternative methodology for estimating PM emissions from paved roads. CARB's methodology was last updated in July 1997.

Unpaved Roads

Vehicles traveling on unpaved roads pulverize the road, generating particles that are lifted and dropped from the rolling wheels. At the same time, strong air currents from the vehicle act on the disturbed road surface, and the wake continues to generate particles even after the vehicle has passed.

As with paved roads, PM emissions from unpaved roads vary directly with the fraction of silt particles present on the road surface. However, while paved roads have silt material from external sources, unpaved roads possess silt inherently: the road itself is the source of emissions in addition to any surface loading (EPA, 1992). The range

of silt content for different types of unpaved roads can vary over two orders of magnitude, depending on the soil characteristics and meteorology of the area. Moisture, in particular, plays a significant role in this regard, as water acts as a dust suppressant (WRAP, 2004). The types of vehicles traveling on the road also influence emissions. Emissions are highly correlated with vehicle weight, so that roads near industrial sites have higher emissions. In areas with fewer heavy vehicles, the moisture content of the road is likely to be the dominant factor in determining emissions (EPA, 2003b).

Profile: Agricultural Sources

Roughly half of all land in the lower 48 states is farmland, including cropland, land in the Conservation Reserve Program (discussed below), pastureland and rangeland. Both crop and animal production generate pollutants that enter the air and the surface and ground waters (Hopkins, 2004). Agricultural operations generate fugitive dust from four primary sources: wind erosion, tillage (soil preparation), livestock operations and agricultural burning.

Wind generates agricultural dust as it blows across dry, exposed soil. The levels of dust generated depend on wind speed, moisture content of the soil, vegetative cover and the composition of the soil. Tillage operations can increase the concentrations of dust in the air as farm machinery disturbs the soil. As one would expect, loose, dry soil is much more apt to be swept up during mechanical disturbances or windy conditions (WRAP, 2004; USDA, 2000).

Livestock operations generate PM emissions from several sources. For instance, in outdoor operations, animals kick up dust simply by walking on dry soil and manure. Manure piles are prone to dust formation during windy conditions. Bedding materials, low-moisture feedstuffs (such as hay), animal dander and poultry feathers are also sources of emissions. The emissions rates from these sources are highly variable and depend on a variety of factors: the moisture content of the soil and manure (which depends, in turn, on precipitation and watering practices), the degree of exposure of manure piles to wind, the type of ventilation in indoor facilities and other factors (USDA, 2000).

Farms use agricultural burning to reduce or dispose of vegetative debris. Some farm operators burn large areas of land following the harvest to reduce excess plant material; to control crop diseases, weeds and pests; and to maintain crop yields. Farm operators also burn piles of agricultural debris, such as orchard trees, limbs and haystacks, and use fire to clear vegetation out of irrigation ditches and canals (USDA, 2000; EPA, 2005b). Agricultural burning, like any biomass combustion, produces direct emissions of $PM_{2.5}$ as well as smaller amounts of sulfur dioxide (SO_2) and nitrogen oxides (NO_x).

Emission Control Opportunities: Paved and Unpaved Roads

There are two basic options for controlling fugitive PM emissions from paved roads: (1) prevention strategies aimed at reducing the transport and deposition of material on roadways; and (2) mitigation strategies that seek to remove the material after it has been deposited on the road surface. Prevention is generally the preferred option; however, some situations can only be addressed with mitigation measures (EPA, 1992). Unpaved roads can be addressed with surface improvements (i.e., paving) or surface treatments. According to WRAP, the control efficiencies for various options are highly variable (WRAP, 2004).

It is important to recognize that not all control options are equally cost-effective or appropriate in all areas. Cost-effectiveness will vary depending upon the number, size, and configuration of sources within a jurisdiction, and the contribution of that source to local PM concentrations. Cost-effectiveness is also dependent upon the existing degree of control for a given source type. Therefore, cost-effectiveness will vary depending upon the baseline or starting point in each area. The cost-effectiveness numbers offered in this chapter are given as examples only.

Table 20.2 summarizes the cost-effectiveness of a variety of control options that were evaluated on behalf of the San Joaquin Valley Unified Air Pollution Control District (UAPCD). Cost-effectiveness estimates vary by several fold. The estimates were reported only for PM_{10}, and the cost-effectiveness of controlling $PM_{2.5}$ could be many times higher than the estimates in Table 20.2 because the proportion of $PM_{2.5}$ in fugitive dust is small compared to the total concentration of PM_{10}, as noted previously. For example, the WRAP *Fugitive Dust Handbook* provides a sample calculation of the cost-effectiveness of street sweeping using a PM_{10}-efficient sweeper. WRAP estimates that it would cost $960 per ton of PM_{10} reduced to sweep an arterial road once per month (making various site-specific assumptions about the number of vehicles per day, average vehicle speeds, road length, etc.). In contrast, the cost of controlling $PM_{2.5}$ using the same control measure is more than four times greater ($3,930 per ton of $PM_{2.5}$ reduced) (WRAP, 2004). In general, control strategies aimed at reducing fugitive dust seek to reduce PM_{10} emissions, and in some cases data are not available regarding the effectiveness of these control options for reducing $PM_{2.5}$.

Prevention Measures for Paved Roads

As indicated, prevention measures seek to reduce the amount of material deposited on roadway surfaces, including anti-skid materials used during the winter months, soil material from unpaved road shoulders, and mud and dirt carryout from construction sites and unpaved roads. Strategies are available for addressing each of these sources.

Silt loading from anti-skid materials can be a significant source of fugitive dust, at times causing a multi-fold increase in total PM_{10} emissions, although not necessarily in $PM_{2.5}$ (EPA, 1992, 2003c). EPA has found that anti-skid materials are frequently applied well above recommended levels (EPA, 1992). Reducing the application of sanding materials is one option for reducing fugitive dust. A second option is to rely on harder or coarser materials that have a lower initial silt content or greater resistance to forming silt-size particles. For example, the Steamboat Springs, Colorado area requires the use of street sanding materials with less than 2 percent fines (EPA, 2004). This option is limited by price and availability (EPA, 1992). A recent study conducted at Lake Tahoe, Nevada suggests that switching to liquid de-icers can reduce road dust emissions (Gertler, 2005). Finally, improved planning and application techniques can also reduce emissions. For example, officials can limit the application of anti-skid material to intersections, hills, and curves as safety permits (EPA, 1992).

Unpaved road shoulders (i.e., the edges of the road) can be a major source of road dust, particularly in arid regions of the country (EPA, 1992). Passing vehicles, wind erosion and water runoff transport dust from the exposed shoulder to the roadway. The dust loadings from uncurbed

Table 20.2

Control Efficiencies for Roadway Control Measures (2002$)

Control Measure	Cost-effectiveness (dollar per ton of PM_{10})
Paved Roads	
Require 4 foot paved shoulders on all new or modified paved roads	$13,800 - $554,000
Require 4 foot paved shoulders on 50% of existing paved roads with highest traffic	$7,290 - $11,300
Require streets to be swept by PM_{10}-efficient street sweepers once per month	$1,070
Require wind- and water-borne deposition to be removed within 24 hours of discovery	$2,850
Limit the purchase of new street sweepers to PM_{10}-efficient units	$33
Trackout	
Install gravel bed trackout apron (3 inches deep, 50 feet long and full road width)	$13,700 - $322,000
Install a pipe grid trackout control device.	$44,100 - $387,000
Require paved interior roads to be 100 feet long and full road width	$7,930 - $186,000
Unpaved Roads	
Limit maximum speed on unpaved roads to 25 miles per hour	$1,080
Pave unpaved roads and unpaved parking areas	$2,160 - $5,920
Implement watering twice a day for industrial unpaved roads	unknown
Apply dust suppressant annually to unpaved parking areas	unknown
Require paving, 4 inches of gravel or dust suppressants to maintain stabilization surfaces at special event parking	$5,980 - $63,200
Implement measures aimed at limiting visible dust emissions to 20% opacity on unpaved parking areas receiving up to 100 trips per day	$5,230 - $30,500

Source: San Joaquin Valley UAPCD, 2003

shoulders are estimated to be many times greater than those from curbed streets (EPA, 1992).

Paving road shoulders is one option for reducing particle loadings. According to published reports, the cost of constructing a 4-foot wide paved shoulder is approximately $66,000 per mile (both directions), plus maintenance. According to an assessment prepared for the San Joaquin Valley UAPCD, paving costs range from $13,800–$554,000 per ton of PM_{10} reduced (2002$), depending on the amount of traffic. A lightly traveled road reduces the cost-effectiveness of this approach (San Joaquin Valley UAPCD, 2003). (These estimates do not reflect the life-cycle impact of manufacturing and applying paving materials.) The effectiveness of curbing is greater when the adjacent soil is stabilized or covered to prevent wind erosion or tracking onto the street. Wind fences can also be effective in reducing road dust (EPA, 1992).

Mud and dirt carryout from unpaved areas, such as parking lots and construction sites, can generate a significant share of paved road silt loadings (EPA, 1992). Control options include securely covering the tops of trucks carrying dust-prone materials; spraying the tires of vehicles as they leave unpaved areas; installing a gravel bed trackout to remove dirt and mud from vehicle tires; and installing pipe or grate grid trackout devices, which can also be used to remove dirt from tires.

A gravel bed trackout device consists of a layer of washed gravel, rock or crushed rock at the intersection of a paved roadway and a work site entrance, to dislodge mud, dirt and debris from the tires of motor vehicles before they leave the work site. The cost of a gravel bed trackout device is estimated at $500 to construct and $860 per year to maintain (2002$). Maintenance includes the periodic removal, screening and replacement of the gravel to remove accumulated soil (San Joaquin Valley UAPCD, 2003). Based on a control efficiency of 46 percent, the costs are estimated to range from $13,700–$322,000 per ton of PM_{10} reduced. This wide range stems from the assumptions that are made with regard to the number of vehicles exiting the worksite, which influences the level of emissions attributable to the trackout, as well as the assumptions regarding the amount of soil trackout generated by an individual vehicle (San Joaquin Valley UAPCD, 2003).

The WRAP *Fugitive Dust Handbook* provides a sample calculation of the cost-effectiveness of installing a 25 foot gravel apron at a construction site. The cost-effectiveness for the control of PM_{10} emissions is estimated at $45,000 per ton. The cost-effectiveness for the control of $PM_{2.5}$ emissions is estimated at $225,000 per ton (WRAP, 2004).

Another alternative is a pipe grid system—sometimes called a grizzly—that shakes the accumulated dirt and mud from trucks leaving a construction site. The device consists of a series of steel pipes welded into a ladder-like formation. A pipe grid system is estimated to reduce trackout by 80 percent. Cost estimates range from $44,100–$387,000 per ton of PM_{10} reduced (2002$), depending on the assumptions that are made regarding the quantity of emissions that are generated by trackout (San Joaquin Valley UAPCD, 2003).

Mitigation Measures for Paved Roads

Several cities use road cleaning, either by sweeping or water flushing, as a strategy for reducing PM_{10} emissions. Originally, street sweeping was performed exclusively for cosmetic reasons; however, air pollution and stormwater regulations have expanded the practice and have led to the development of new technologies.

The effectiveness of street cleaning as an air pollution control strategy, however, is limited in several respects. Street sweeping equipment varies in terms of its control-effectiveness and may, in some cases, actually increase PM_{10} emissions (DOT, 2005). By sweeping gutters and curb areas, a street sweeper may actually transfer the loose particles to the traveled portion of the road, resulting in a short-term increase in emissions (WRAP, 2004; Kuhns, 2003; Gertler, 2005). Several factors can limit the effectiveness of street sweeping: the presence of parked cars and traffic congestion during sweeping, poor road surface and curb conditions and the presence of construction projects nearby. Also, the cycle of particle deposition on road surfaces and subsequent resuspension in the air will generally outpace efforts to keep the roads swept, thereby limiting its effectiveness as a control option.

Targeted efforts to eliminate anti-skid materials or the cleanup of mud and dirt from road surfaces following a wind or rain event are likely to offer the best opportunities for application of the technology.

There are three basic options in street sweeping technology: traditional mechanical sweepers, vacuum assisted sweepers and regenerative air sweepers.

- Mechanical sweepers employ a rotating gutter broom to remove particles from the street gutter area, with a water spray used to control dust. The particles are then placed in the path of a cylindrical broom that rotates to carry the material onto a conveyor belt and into a storage hopper. This is the most widely used equipment for street cleaning in the U.S.
- Vacuum assisted sweepers also use gutter brooms to remove particles from the street. However, the refuse is then placed in the path of a vacuum intake that sucks the dirt into the hopper. The transported dirt is usually saturated with water. The overall efficiency of vacuum assisted cleaners is generally higher than

that of mechanical cleaners, especially for particles larger than the dust and dirt range (larger than about 3 millimeters).

- Regenerative air sweepers blow air onto the pavement and immediately vacuum it back, to entrain and capture accumulated sediments. Air is regenerated for blowing through a dust separation system.

Street sweepers are most effective when operated at optimal speeds (six to eight miles per hour (mph)), brushes are properly adjusted and appropriate rotation rates and sweeping patterns are used.

The South Coast Air Quality Management District (AQMD) in California was the first agency in the country to enact regulations that included an emissions standard for street sweepers (Rule 1186). South Coast AQMD certifies street sweepers that satisfy minimum pickup and entrainment standards. South Coast AQMD-certified street sweepers are listed at: www.aqmd.gov/rules/doc/r1186/r1186_equip.pdf. South Coast AQMD has certified mechanical sweepers, vacuum assisted sweepers and regenerative air sweepers. The purchase price of a certified PM_{10}-efficient street sweeper is reported to be similar to that of a non-certified unit, and the operating costs are approximately the same (San Joaquin Valley UAPCD, 2003).

The city of Toronto, Ontario has also developed a PM test protocol for street sweepers, concluding that regenerative air street sweepers are most effective in terms of PM_{10} and $PM_{2.5}$ capture efficiency. City officials tested eight street sweeper models, which included several mechanical, vacuum assist and regenerative air models (www.toronto.ca/legdocs/2005/agendas/committees/wks/wks051108/it014.pdf).

Unpaved Roads

The primary control options for fugitive dust from unpaved roads are surface improvements and surface treatments. Limiting the amount and speed of traffic on unpaved roads can also be effective strategies for controlling emissions (EPA, 1992).

Surface improvements are generally one-time or infrequent affairs, unlike surface treatments, which must be reapplied periodically. The most obvious surface improvement is paving; however, because of its high cost, paving is only cost-effective on roads with relatively high traffic volumes. Sources estimate the cost of paving an unpaved road at approximately $400,000 per mile, inclusive of roadway excavation, aggregate base, striping and traffic control (2002$). Over an assumed 25-year useful life, the annualized cost of paving is $44,100 per year per mile (San Joaquin Valley UAPCD, 2003). Moreover, while paving may be a viable option for certain roads, paving operations require large amounts of energy and themselves generate various emissions, including $PM_{2.5}$. Some roads are impractical to pave either because they have poor bases on which to build or because they experience substantial spillage from trucks. Because of the spillage and ensuing dust loading, such roads—even if paved—would still generate significant emissions.

Another option for controlling PM emissions from unpaved roads is to cover the road with a material of lower silt content. For instance, covering a dirt road with gravel reduces dust emissions. This type of measure is initially much less expensive than paving, but has ongoing maintenance costs (grading, reapplication of gravel material). Finally, for roads with very low traffic volume, vegetative cover is an option (EPA, 1992).

Surface treatments come in two types: wet suppression (i.e., watering) and chemical stabilization (EPA, 1992). Reapplication frequencies range from a few minutes for water treatments during a hot summer day to several weeks or months for chemical treatments.

The control efficiency of watering unpaved roads depends on four factors: the amount of water applied per unit area of road surface, the time between applications, traffic volume during that period and prevailing meteorological conditions. EPA field tests suggest low control efficiency (the actual numbers are not explicitly stated) and the need for frequent (almost daily) reapplications (EPA, 1992).

The WRAP *Fugitive Dust Handbook* provides a sample calculation of the cost-effectiveness of watering (twice daily) the unpaved roads of an industrial facility. WRAP estimates a cost of $530 per ton of PM_{10} reduced and a cost of $3,450 per ton of $PM_{2.5}$ reduced (WRAP, 2004).

Chemical treatments, which can be more effective, simulate wet suppression by attracting and retaining moisture on the road surface. Watering often supplements chemical treatments. Most chemical suppressants form a hard cemented surface. The most common type is a petroleum resin, but asphalt emulsions, acrylics and adhesives are also options (EPA, 1992). Potential environmental impacts include surface and groundwater quality deterioration, soil contamination, toxicity to soil and water biota, toxicity to humans during and after application, air pollution from volatile dust suppressant components, accumulation in soils, changes in hydrologic characteristics of the soils, and impacts on native flora and fauna populations.

Emission Control Opportunities: Agricultural Operations

Because of the diversity of agricultural activities and conditions that generate fugitive dust (i.e., tilling, wind

Table 20.3

Efficiency for Various Control Options for Tillage Operations

Control Measure	PM_{10} Control Efficiency	References/Comments
Limited activity during high winds	1%–5%	SCAQMD, 1997. Control efficiency assumes no tilling when wind speed exceeds 25 miles per hour.
Reduced tillage system (conservation tilling)	35%–50%	Coates, 1994. This study identified total PM_{10} emissions generated by five different cotton tillage systems, including conventional tillage systems. Four of the systems combine several tillage operations (e.g., shredding, disking, mulching).
	60%	MRI, 1981. Control efficiency is for a minimum tillage technique that confines farm equipment and vehicle traffic to specific areas (for cotton and tomatoes).
	30%	MRI, 1981; EPA, 1992. Control efficiency is for laser-directed land plane that reduces the amount of land planing.
	50%	MRI, 1981; EPA, 1992. Control efficiency is for using punch planter instead of harrowing (for cotton, corn and lettuce).
	50%	MRI, 1981. Control efficiency is for using plug planting that places plants more exactly and eliminates the need for thinning (for tomatoes only).
	50%	MRI, 1981; EPA, 1992. Control efficiency is for aerial seeding, which produces less dust than ground planting (for alfalfa and wheat).
	91%–99%	Grantz, 1998. Control efficiency is for revegetation of fallow agricultural lands by direct seeding.

Source: WRAP, 2004

erosion, animal operations and agricultural burning), the control opportunities are also varied.

Tillage

The primary options to reduce fugitive dust from tillage involve limiting activity during windy conditions and reducing tillage in various ways. The tillage methods described in Table 20.3 differ in the amount of soil disturbed and the amount of crop residue allowed to remain before the next planting (EPA, 2000).

Conservation tillage methods reduce soil erosion by allowing soil and crop residue to remain undisturbed. Some common methods include no-till, strip-till, ridge-till and mulch-till (EPA, 2000).

- No-till has minimal soil disturbance because the seed is planted with essentially no tillage of the soil and no disturbance of the crop residue.
- Strip-till involves tillage of a narrow strip of soil and planting of the seed or seedling in the tilled area.
- Ridge-till methods disturb a narrow strip of soil that was created during previous cultivation. The crop is planted on the ridge and the crop residue remains between each ridge.
- Mulch-till involves distributing loose plant residue across a field to reduce wind erosion and to conserve soil moisture (EPA, 2000).

Other techniques can also reduce tillage. For instance, the California Air Pollution Control Officers Association (CAPCOA) recommends combining operations to perform several operations during one pass, altering the bed or row size and spacing to reduce the number of passes and soil disturbances by increasing plant density, operating at night when moisture levels are higher and winds are generally lighter, and applying or leaving plant residue on the soil surface to reduce wind erosion of soil (CAPCOA, 2004).

EPA has also suggested using a punch planter to reduce emissions from planting cotton, corn and lettuce (EPA, 1992). A punch planter punches a hole and places the seed into it, as opposed to conventional planters that make a trough and drop the seeds in at a specified spacing. Punch planters leave much of the surface soil and surface crop residues intact. However, large-scale use would require substantial investment.

Planing (flattening the land) and its associated emissions can be reduced in two ways. The first is to use sprinkler

irrigation in place of surface watering, since the latter requires completely flat land. The second method is to use a laser-directed land plane, which has been used in the construction industry for decades. With a more precise leveling blade, the amount of land planing can be reduced. However, both of these options may require substantial up-front capital expenditures (EPA, 1992).

Finally, aerial seeding, or dropping seeds from an airplane, can be an effective way to reduce dust formation. Because of the expense, this technique is rare, although specific conditions may warrant its consideration. For example, a consortium including the University of California Cooperative Extension, the U.S. Department of Agriculture (USDA) Natural Resources Conservation Service and local air pollution districts instituted an aerial re-seeding program in the early 1990s, after a long drought during the previous decade reduced vegetation cover in the Mojave Desert. During the drought, dust storms had destroyed crops and forced farmers to re-seed frequently. Moreover, suspended soil had blown into surrounding areas and caused unhealthy PM_{10} levels. The resulting collaboration involved fire department helicopters planting 2,500 acres of severely degraded land. The maximum daily PM_{10} concentration in one town was reduced from 780 micrograms per cubic meter of air in 1991 to an average of 75.6 during the next ten years, and when drought returned in the late 1990s, the revegetation held the soil in place (Grantz, 2005).

Wind Erosion

Several options are available to reduce wind erosion. The oldest and perhaps easiest is the use of cover crops to cover soil surfaces that would otherwise be exposed and at risk of erosion. Cover plants include traditional crops, such as grasses, wheat, sorghum, corn, legumes and cotton (EPA, 1992). Cover crops such as rye are especially useful because they grow in late fall and provide soil cover during winter.

Cover crops can be effective in reducing erosion, achieving over 90 percent control efficiencies (Grantz, 1998; WRAP, 2004). However, revegetation can be difficult to achieve in any given year, as varying environmental conditions and soil factors make direct seeding unreliable, and employing nursery-grown shrubs does not guarantee success, even with supplemental irrigation (Grantz, 1998).

Mulching can also prevent wind erosion. Mulching and spreading crop residues can be used in conjunction with cover crops, covering soil until the crops have a chance to take root. However, mulching does not last as long as cover crops and can be more vulnerable to wind if the mulch layer is shallow.

Windbreaks or barriers are another option for reducing wind erosion. Shrubs, trees and even manmade fences are effective wind barriers. Grass barriers also work well, but they do not slow the wind as much as shrubs or trees. The principle behind a wind fence or barrier is straightforward: it blocks soil from eroding and provides an area of reduced wind velocity to allow particles in the air to settle. Windbreaks generally consist of trees or shrubs, usually in multiple rows. The effectiveness of the barrier depends on the wind velocity and direction, shape, width, height and porosity of the barrier (EPA, 1992).

Artificial barriers, such as fences constructed of board, bamboo, willow, rock or earth, are effective in reducing wind erosion. Their drawback is the high cost of material

Table 20.4

Efficiency of Control Options for Wind Erosion on Agricultural Lands

Control Measure	PM_{10} Control Efficiency	References/Comments
Artificial wind barrier	64%–88%	EPA, 1992. Assumes a 50% porosity fence.
	54%–71%	Grantz, 1998. Control efficiency is for a wind fence.
	4%–32%	Bilbro and Stout, 1999. Control efficiency based upon reduction in wind velocity by a wind fence made from plastic pipe.
Cover crop	90%	Washington State University, 1998.
Cross-wind ridges	24%–93%	Grantz, 1998. Control efficiency is for furrows.
	40%–80%	Washington State University, 1998.
Mulching	20%–40%	Washington State University, 1998.
Trees or shrubs planted as a windbreak	25%	Sierra Research, 1997. Control efficiency is for trees.

Source: WRAP, 2004

and labor required for construction. As a result, their use has been limited in the U.S. and confined to areas with high-value crops or where overpopulation requires intensive agriculture (EPA, 1992).

Another technique is cross-wind ridges or strip-cropping. Strip-cropping entails establishing crops in strips perpendicular to the prevailing wind direction and arranging crops so that strips susceptible to wind erosion are alternated with strips resistant to wind erosion (CAPCOA, 2004).

Finally, roughening the surface by using tilling methods that leave stable clods of soil is a simple control technique that can substantially reduce fugitive dust (CAPCOA, 2004).

Animal Operations

A host of control options are available to reduce fugitive dust from animal operations, including planting trees and shrubs; maintaining feedlots by removing manure; covering manure piles; and reducing animal activity during the late afternoon, when dust is most likely to form (ISU, 2004). Despite the various control options available, few have been tested extensively, and data on cost and effectiveness, especially for $PM_{2.5}$, are scarce.

The application of water can reduce fugitive dust from outdoor facilities. Facilities employing this method generally rely on a tanker truck to dispense water. EPA notes that this technique may be practical only for small operations, since a large amount of water is needed—one study estimates the amount of water to be equal to cattle drinking-water requirements during the dry season (EPA, 2001). Another potential drawback is that high moisture content can lead to odor problems and may also induce greater microbial activity (although no data have yet shown this to be the case). Indoor facilities have not conducted any studies using sprinklers to reduce $PM_{2.5}$ emissions, but misting systems are already in place in some poultry and swine operations, indicating that water sprays do have potential in an indoor context (EPA, 2001).

Covering manure stockpiles is another method to reduce $PM_{2.5}$ emissions. Impermeable covers, such as polyethylene, can provide substantial reductions. These covers, which range in cost from $1.00–$1.40 per square foot (installed), last about ten years, but they eventually degrade due to snow and wind damage (ISU, 2004). At that point the small bits of plastic shed by the cover can become an environmental and aesthetic problem, not to mention a nuisance. Permeable covers, such as straw, cornstalks, peat moss, foam, rocks and fabric, can also be effective. Costs vary widely, from $0.10 per square foot (applied) for straw and cornstalk, to $0.26 for peat moss. Iowa State University recommends a depth of at least 8, and preferably 10–12, inches (ISU, 2004).

Altering the feed can also reduce emissions. For instance, because some dry, fine-grain feed generates dust emissions during handling and delivery, using higher-moisture feed, agglomerating the feed into pellets, or mixing the feed with vegetable oil or animal fats can reduce emissions. In addition, using enclosed feed delivery systems and covered feeders can cut emissions (EPA, 2001). Evidence suggests that altering the feed may also change fecal characteristics, thereby preventing dust emissions. For example, studies with pigs and cattle show that by adding fat or oil to the diets, feces become stickier and generate less dust (ISU, 2004).

For indoor facilities, a number of options exist to reduce particulate emissions. Perhaps the easiest is to spray vegetable oil on floors and other surfaces (ISU, 2004; EPA, 2001). The oil agglomerates the particles and prevents them from becoming airborne. However, while vegetable oil is generally inexpensive, one drawback to this approach is the increased safety (i.e., slipping) concern for animals and workers.

Another approach is to use filtration. Mechanical filters can be applied to building exhaust from totally or partially enclosed housing. Some filtration systems employ natural materials, such as straw or other crop residues, as the filter medium. However, mechanical filters eventually become clogged and need cleaning or replacement (EPA, 2001). Another type of filtration is the use of trees, shrubs and walls as windbreaks or biofilters (ISU, 2004). These techniques are similar to those discussed above for agricultural erosion control.

Ionization, used in electrostatic precipitators (ESPs), is another possibility for indoor animal operations. With ionization, gas molecules (e.g., oxygen) acquire a charge from high-energy electrons created by an electric field. The ionized gases adhere to particulates, which then move to the nearest grounded surface (e.g., building surface, grounded collection plate) (EPA, 2001). ESPs used in other industries have achieved control efficiencies of upwards of 99 percent, but three studies of agricultural operations achieved only 40-60 percent reductions. The researchers were unable to give definitive explanations for this discrepancy, but the high moisture content of the air stream may have been a factor (EPA, 2001).

Finally, a wet scrubber, described elsewhere in this document, can be an effective way to reduce emissions. In the context of animal feeding operations, a wet scrubber can be installed on the ventilation unit. While wet scrubbers have not been used extensively in the industry, one study showed a 90 percent reduction of total PM (EPA, 2001).

Agricultural Burning

Agricultural burning can produce large quantities of smoke in a short period of time. To reduce impacts, permits are usually required before burns can be conducted. These permits can restrict the type and amount of agricultural material burned and limit burning to those times when air quality and meteorological conditions are most suitable for reducing impacts (EPA, 2005b). Alternatives to burning should also be investigated as the situation permits.

Regulatory Authority

Federal law does not regulate fugitive dust directly. However, state and local authorities have adopted programs to control fugitive dust in an effort to comply with the national PM_{10} air quality standards.

The Clean Air Act requires states to develop plans that include all Reasonably Available Control Measures (RACM) or all Best Available Control Measures (BACM), including fugitive dust control measures, in PM_{10} nonattainment areas. For example, the South Coast AQMD requires all fugitive dust sources to implement specific control measures to prevent visible dust emissions from crossing any property line (Rule 403). Studies indicate that approximately one-third of the South Coast basin's ambient PM_{10} concentrations and over 90 percent of Coachella Valley's ambient PM_{10} levels are the result of fugitive dust (South Coast AQMD, 2005).

In support of the states' effort to implement BACM, EPA has developed guidance on the control measures available for reducing fugitive dust (EPA, 1992). More recently, the San Joaquin Valley UAPCD published an evaluation of BACM, focusing on fugitive dust emissions (San Joaquin Valley Unified APCD, 2003).

Many of the federal programs that regulate agricultural operations are not directed at air emissions (e.g, the Food Quality Protection Act of 1996). However, there are voluntary programs that impinge on agriculture-related air quality issues. The Conservation Reserve Program within the USDA pays farmers to apply soil and water conservation measures and to keep highly erodible and other types of land out of production. The contracts generally last for 10–15 years. At the end of 2004, there were nearly 400,000 farms participating in the program, covering 35 million acres (USDA, 2005a, 2005b). Soil conservation practices are often consistent with efforts to control fugitive dust emissions.

Also, the USDA has evaluated options for reducing fugitive dust emissions from agricultural operations under the auspices of its air quality program.[1]

1. More information is available at www.ars.usda.gov/research/programs.htm.

State and Local Opportunities

The San Joaquin Valley UAPCD in California; Clark County, Nevada; and Maricopa County, Arizona have implemented programs to control fugitive dust emissions from paved and unpaved roads. Toronto, Ontario has evaluated the $PM_{2.5}$ control effectiveness of street sweepers. Few states have regulated agricultural activities as a means of complying with air quality standards.

Paved Roads

San Joaquin Valley, California. Rule 8061 of the San Joaquin Valley UAPCD (amended August 19, 2004) requires new paved roads or modifications to existing paved roads with projected annual average daily vehicle trips of 500 vehicles or more to be constructed with paved shoulders at least 4 feet wide, or the limit of the right-of-way, whichever is smaller. Roads with over 3,000 vehicles per day must have 8-foot shoulders, or shoulders built to the limit of the right-of-way, whichever is smaller (San Joaquin Valley Unified APCD, 2004).

As an alternative to meeting the shoulder width standard, areas may also install curbing next to the travel lane or construct intersections, auxiliary entry lanes and auxiliary exit lanes adjacent to the roadway (San Joaquin Valley Unified APCD, 2004).

If a road is constructed (or modified) with a median, the median must have a shoulder width of 4 feet adjacent to the travel lanes. This standard does not apply to roads with speed limits under 45 mph that have curbing. In addition, areas can avoid this requirement if they landscape and maintain their medians with grass or other vegetative ground cover or dust suppressants.

Rule 8061 also has requirements regarding PM_{10}-efficient street sweepers; it provides that as of July 1, 2005, all purchases of street sweeper equipment by an agency or its contractor must be only PM_{10}-efficient street sweepers, and that each agency or contractor must purchase and use at least one PM_{10}-efficient street sweeper by July 1, 2008. The rule also requires that the routine street sweeper routes with paved curbs and the greatest dirt and silt loadings be cleaned at least once per month. If these provisions cannot be met due to budgetary constraints, the agency may submit a statement of financial hardship and request a waiver.

Finally, the rule specifies actions that agencies must follow after an "event" that results in at least 1-inch thickness of mud or dirt over an area of at least 50 square feet as a result of runoff or erosion from wind, water or a storm. Within 24 hours after discovering this condition, the agency must remove the mud or dirt from the travel lanes or restrict vehicle travel until the material can be removed. The agency must follow dust minimizing practices when

removing the dirt and must remove it from paved shoulders as soon as practicable. Time extensions are available for safety concerns.

Clark County, Nevada. Clark County, Nevada has promulgated regulations for paved roads. Section 93 of the county's air quality regulations deals with fugitive dust from paved roads. The rule was initially adopted in June 2000 and has been amended six times since, most recently on July 1, 2004 (Clark County, 2004a).

Like the San Joaquin Valley rule, §93 requires new construction and modification of paved roads to have 4 feet of paved or stabilized (with gravel) shoulder for roads with average vehicular traffic under 3,000 vehicles per day, and 8 feet for roads with over 3,000 vehicles per day. If the right-of-way is not available for the full length (4 or 8 feet), curbing must be installed instead (Clark County, 2004a).

The rule also specifies median requirements on newly constructed roads or roads modified with shoulders or medians. Such roads must be constructed with one of the following control measures: curbing; solid paving across the median; dust palliatives; 2 inches of gravel; or other materials that prevent trackout, such as landscaping or decorative rock. The measures must achieve control of fugitive dust emissions under 20 percent opacity and silt loading under 0.33 ounces per square foot (Clark County, 2004a).

Toronto, Ontario. The city of Toronto has developed a test protocol for street sweepers that evaluates both PM_{10} and $PM_{2.5}$ control-effectiveness. The testing has examined the amount of material picked up and removed from the test track, redeposited on the sidewalk or road surface, and released into the air. The testing established that the regenerative-air street sweepers achieved the best performance for both PM_{10} and $PM_{2.5}$ in all three test categories. The city has issued a request for proposals limited to regenerative air street sweeper technology, with specific performance thresholds developed based on the testing performed (Welsh, 2005).

Unpaved roads

San Joaquin Valley, California. Rule 8061 of the San Joaquin Valley UAPCD specifies various measures for unpaved roads. First, the rule prohibits building new unpaved roads except for temporary purposes. Second, unpaved roads with over 25 vehicles per day must have a maximum speed limit of 25 mph and post speed limit signs once every mile in urban areas and once every 2 miles in rural areas.

In addition, the rule aims to limit visible dust emissions to 20 percent opacity by requiring unpaved roads with over 25 vehicles per day to implement one of a number of control measures. The measures include watering, installing a uniform layer of washed gravel, using chemical or organic dust stabilizers or suppressants, paving, or any other method that can achieve 20 percent opacity.

The rule also requires agencies with primary responsibility for a road to provide their air pollution control districts with a list of all unpaved roads and a list of paved roads with unpaved shoulders, together with their length and average daily vehicle traffic. By January 1, 2010, each agency is required to pave an average of 20 percent annually of all unpaved roads (to a maximum of 5 cumulative miles within any one urban area), with priority given to roads with the highest vehicle traffic. Also by January 1, 2010, urban areas must pave or stabilize 4-foot shoulders on 50 percent of existing paved public roads with unpaved shoulders, and rural areas must do the same for 25 percent of roads with unpaved shoulders. Each year, the agency responsible for the road must submit a report detailing the number of unpaved road miles paved during the previous year. Areas with budget problems can request a waiver from these paving requirements.

San Joaquin Valley UAPCD Rule 8021 limits fugitive dust emissions from construction, demolition, excavation, extraction and other earthmoving activities. The rule requires the application of water or chemical stabilizers/dust suppressants, in conjunction with optional wind barriers, to limit visible dust emissions to 20 percent

Table 20.5

Clark County, Nevada Compliance Schedule for Unpaved Roads

Nonattainment Area	Fraction of Total Miles of Unpaved Roads Controlled		
	⅓	⅔	All
Basin 212[a]	June 1, 2001	June 1, 2002	June 1, 2003
Basins 216 & 217[b]	April 1, 2003	April 1, 2004	April 1, 2005

a. Applies to roads constructed prior to June 22, 2000.
b. Applies to roads constructed prior to April 1, 2002.

Source: Clark County, 2004b

opacity.

Clark County, Nevada. Clark County has stringent standards for unpaved roads. Section 91 of the county's air quality regulations were initially adopted in June 2000 and have been amended four times since, most recently on July 1, 2004. The rule prohibits the construction of new unpaved roads in specified areas. It requires owners or operators of existing roads with at least 150 vehicles per day to pave the roads, apply dust palliatives, or apply and maintain an alternative control measure approved by the dates specified in Table 20.5. In addition, new unpaved roads must apply dust control measures upon determining that they have more than 150 vehicles per day traveling the road (Clark County, 2004b).

Maricopa County, Arizona. Maricopa County, Arizona has promulgated regulations to control fugitive dust from unpaved roads in the county's PM_{10} nonattainment area. Rule 310.01 regulates unpaved roads as well as open areas, vacant lots and unpaved parking lots. The rule was initially adopted in June 1999 and has been revised twice, most recently on February 17, 2005.

The rule requires the owner of unpaved roads (including alleys) with at least 150 vehicles per day to pave the roads, apply dust suppressants, or apply and maintain gravel. The control measures must achieve 20 percent opacity and keep silt loading under 0.33 ounces per square foot or keep silt content less than or equal to 6 percent.

Agricultural Operations

California regulators have adopted measures to reduce fugitive dust from agricultural activities. Senate Bill 656, passed in 2003, aims to reduce public exposure to $PM_{2.5}$ and PM_{10}. The bill requires CARB, in consultation with local air pollution control and air quality management districts, to adopt a list of the most readily available, feasible and cost-effective control measures that can be implemented by local air districts to reduce $PM_{2.5}$ and PM_{10} (CARB, 2005a). The list is based on rules, regulations, and programs existing in California as of January 1, 2004. Using these measures, air districts were to develop implementation schedules by July 31, 2005. The list of options covers various sectors and sources, including agriculture. Among the options for reducing particulates from agricultural operations are the following:

- Cease tilling/mulching activities when wind speeds are greater than 25 mph.
- Limit fugitive dust from paved and unpaved roads and livestock operations by requiring: (1) the cessation of all hay grinding activities between 2 and 5 p.m. if visible emissions extend more than 50 feet from a hay grinding source, and (2) the treatment of all unpaved access connections to livestock operations and unpaved feed lane access areas with either pavement, gravel (maintained to a depth of 4 inches) or asphaltic road-base.
- Reduce fugitive dust from livestock feed yards by requiring: (1) a dust plan that contains procedures assuring moisture factors between 20 percent and 40 percent for manure in the top 3 inches of occupied pens; and (2) outlines manure management practices, including removal.
- Establish guidelines for smoke management, emphasizing effective planning, coordinating among burners, using the most technically advanced air quality and meteorology burn management tools, and considering alternatives to open burning. Specifically, the guidelines require local air districts to develop smoke management programs that contain three basic components: (1) requirements for a burn authorization system; (2) requirements for smoke management plans by farms conducting prescribed burns; and (3) requirements for burn, no burn and marginal burn days (CARB, 2004).

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State and Territorial Air Pollution Program Administrators (STAPPA)
Association of Local Air Pollution Control Officials (ALAPCO)

444 North Capitol Street, N.W. Suite 307 Washington, D.C. 20001
Phone: (202) 624-7864 Fax: (202) 624-7863
Website: www.4cleanair.org