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Quantifying and Controlling Fine Particulate Matter in New York City

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1. Executive Summary

Controlling emissions of $PM_{2.5}$ and its precursors is a pressing policy problem for New York City (NYC). Exposure to fine particulate matter ($PM_{2.5}$) is linked to serious health effects, including premature death from heart and lung disease, cardiovascular symptoms, cardiac arrhythmias, heart attacks, respiratory symptoms, asthma attacks, and bronchitis. With one of the highest burdens of asthma problems in the country, NYC residents are highly susceptible to increases in air pollution. The high ambient concentrations of $PM_{2.5}$ in NYC are thus a major threat to public health.

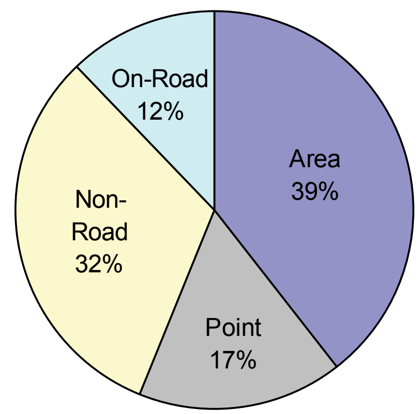
The strong and increasingly thorough scientific record demonstrating the severity and extent of health effects from exposure to $PM_{2.5}$ has prompted policy and regulatory activity on all levels of government. The City of New York has taken actions to improve air quality, the most recent of which include the use of ultra-low-sulfur diesel in Metropolitan Transit Authority vehicles and the retrofit of those vehicles with diesel particulate filters. Despite this and other local and state efforts, air quality problems in the city persist. All five boroughs are in violation of the current federal National Ambient Air Quality Standards (NAAQS), set in 1997.

Even as compliance with the 1997 $PM_{2.5}$ NAAQS has proved elusive for NYC thus far, more stringent regulations are forthcoming. In September 2006, the U.S. EPA lowered the 24-hour $PM_{2.5}$ standard to 35 $\mu g/m^3$, well below peak ambient levels in all five boroughs. Although recent federal regulations, such as the EPA's rulemaking on the sulfur content of diesel fuel, will help to improve air quality, additional actions must be taken in the near term if NYC is to meet the new federal standard. Moreover, attaining the lower 24-hour NAAQS may not sufficiently reduce the threat to public health in NYC, where interactions between weather and the complex three-dimensional urban landscape can result in markedly different $PM_{2.5}$ concentrations in sheltered areas than at sites chosen for federal air quality monitoring.

Formulating an effective policy response to $PM_{2.5}$ pollution requires an understanding of where it originates. Ambient $PM_{2.5}$ concentrations are made up of numerous constituents coming from different sources in different geographic locations, both inside and outside the jurisdiction of the city and the state. This report concentrates on a subset of the emissions that affect the city's air quality—those that are emitted in NYC—because state and local policies may be able to target them more easily than emissions outside of the region. To this end, we present an inventory of NYC $PM_{2.5}$ emissions that has been revised to more accurately portray local economic and demographic conditions there.

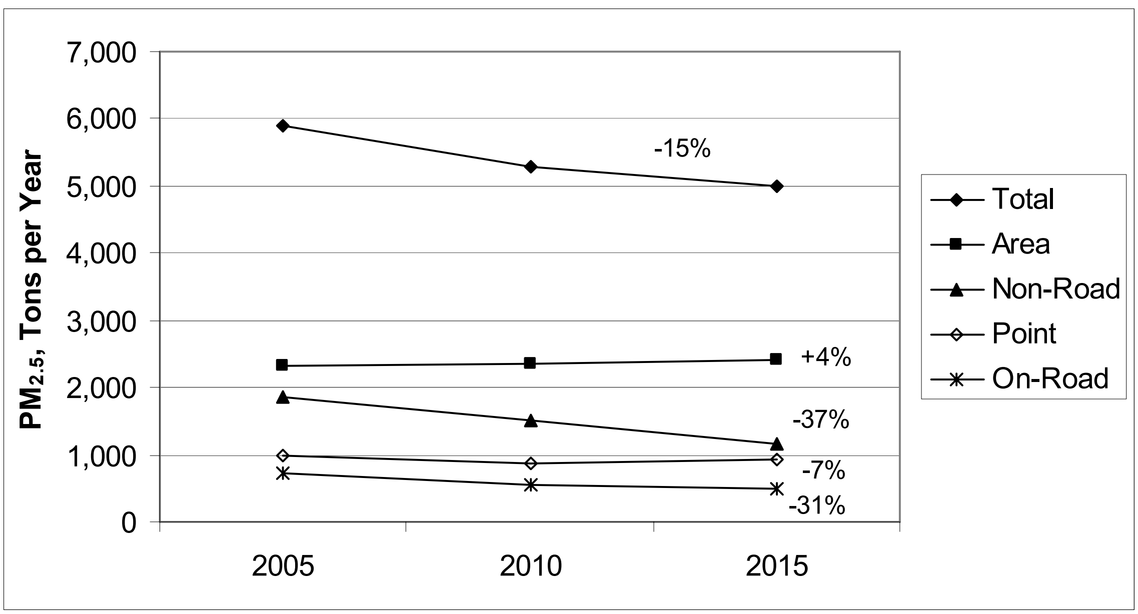
The largest part (39%) of 2005 $PM_{2.5}$ emissions in NYC originate in the area source sector, mainly comprised of residential and commercial furnaces, waste incinerators, small distributed generation, and combined heat and power (CHP) applications. Sources in the non-road sector, such as airplanes, industrial and construction equipment, and marine transportation, are estimated to account for 32% of NYC's $PM_{2.5}$ emissions in 2005. Dominated by a few large electricity generators, point sources comprise 17% of total $PM_{2.5}$ emissions. On-road sources are estimated to contribute the remaining 12%.

Estimated 2005 Direct $PM_{2.5}$ Emissions in NYC



Total: 5,903 tons

The study presents indicative trends for $PM_{2.5}$ emissions from the different source sectors in NYC. Between 2005 and 2015, total annual direct $PM_{2.5}$ emissions are projected to fall by roughly 15 percent, or nearly 900 tons. The largest portion of the overall reduction comes in the non-road sector, driven by federal regulations on diesel equipment. Emissions from on-road sources are also projected to fall significantly as a result of federal regulations on diesel trucks and the installation of retrofit equipment on Transit Authority buses. Emissions from large point sources are projected to fall by a small amount over the study period, as increasing production to meet growth is offset by the addition of new, cleaner plants in the electricity sector. Area source emissions increase by a small amount, as growth in commercial and industrial activity more than offsets reductions in the residential sector projected to result from the continuing shift from oil to gas for residential heating.



Policymakers have a wide range of tools they can use to reduce NYC's contribution to the problem. We evaluated nineteen measures that can be implemented on the state or local level on the basis of cost, benefits for reducing other pollutants and increasing efficiency, political feasibility, and other considerations. This analysis identifies six key policies for achieving much-needed reductions, including:

- Energy efficiency programs – both electric and thermal – are very low-cost ways of reducing fuel combustion, provide substantial reductions in other pollutants, and stimulate local economic development.
- Development of new combined heat and power systems provides zero-cost emission reductions, as energy cost savings typically result in net savings over the life of the system.
- Mandated use of low or ultra low-sulfur fuel oil for heating in the area source sector could reduce emissions significantly at very low costs. Reduced maintenance costs and extended equipment life could offset most or all of the increased fuel cost.
- Controls on "other" buses operating in NYC (i.e., those not already controlled) would provide modest reductions (roughly 40 tons per year) at modest cost, with co-benefits in reduced CO and HC emissions.
- Fuel switching at both point sources and area sources offers significant emission reductions; however, movement from oil to natural gas is happening naturally in these sectors and is putting additional strain on gas infrastructure and upward pressure on natural gas prices.
- Further research is needed into the efficacy and suitability of emissions controls for energy plants firing residual fuel for electricity and steam generation within New York City.

Regardless of the options policymakers elect to pursue, it is critical that additional $PM_{2.5}$ emission reductions be obtained. Uncertainties concerning future federal actions beyond implementation of the existing suite of national strategies focusing on heavy duty trucks, off-road equipment and power plants may leave NYC to act in the absence of national control strategies in the future. Absent a marked change in the direction of national air pollution policy, local, state and regional $PM_{2.5}$ emission reduction strategies will play an increasingly important role in protecting residents for adverse health effects associated with exposure to $PM_{2.5}$ sources.

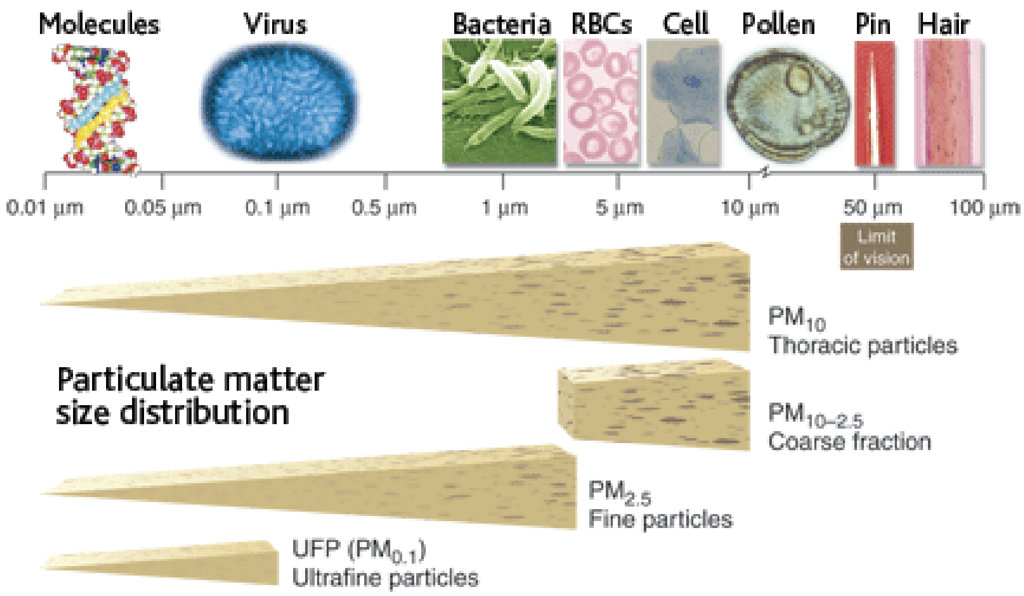
The remainder of this report is outlined as follows. In chapter two, we discuss the health effects associated with exposure to $PM_{2.5}$. Chapter 3 focuses on ambient concentrations in NYC and apportioning those concentrations to sources both inside and outside of the city. In chapters 4 and 5, we draw on state emissions data to construct a baseline emissions inventory and project those emissions to 2010 and 2015. A large range of emission reductions options were identified and initially evaluated in Chapter 6, considering immediate costs and potential co-benefits (including reductions in other emissions). Chapter 7 lays out a portfolio of recommended policy options.

2. Health Effects and Regulation of $PM_{2.5}$

2.1 What is $PM_{2.5}$?

Particulate matter (PM) can be defined as liquid or solid aerosol particles with a broad range of physical characteristics and potential chemical species. PM can include dust, smoke, mist, fumes, or smog, found in the air or emissions.[1] Fine particulate matter, $PM_{2.5}$, includes a wide range of aerosol particles less than 2.5 microns (µg) in diameter (for reference, a micron is just 1/70th of the diameter of a human hair). This entire category of aerosol particles is invisible to the human eye, yet it has a profound impact on public health in several ways.

Figure 2.1: Particulate Matter Size Distribution



Source: P. Huey/Science; Adapted from Brock et al. A1./AHA Scientific Statement No. 71-289, Images EOL Berkeley National Laboratory, CDC, Jupiter Images.

More precisely, $PM_{2.5}$ is a measure of the mass of all airborne particles with aerodynamic diameters less than 2.5 micrometers. $PM_{2.5}$ encompasses a wide range of individual particle sizes, many different chemical species, and contains both primary (directly emitted) and secondary (formed through reactions in the atmosphere) components from a wide variety of sources. A sub-category of $PM_{2.5}$, ultrafine particulates consists of particles less than 0.1 micrometers in diameter.

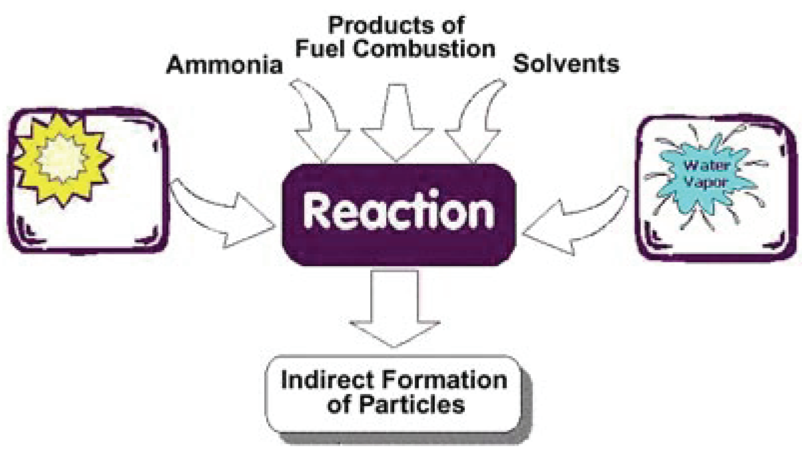
[1] US EPA, http://www.epa.gov/OCEPAterms/pterms.html

Fine particulate matter is emitted from various different processes and types of sources. To distinguish between these processes, the US EPA classifies $PM_{2.5}$ as either primary or secondary. Primary $PM_{2.5}$ does not undergo any chemical transformation in the air after being emitted. These particulates are directly released into the air from a number of sources, including cars, trucks, buses, factories, construction sites, tilled fields, unpaved roads, stone crushing, and burning of wood.[2] The chemical composition of the particles varies depending on location, weather, and time of year[3] and may include varying portions of elemental carbon, organic carbon, nitrate, sulfate, and crustal material.

Primary $PM_{2.5}$ is further categorized as filterable—matter that can be filtered in a solid or liquid state at the source—or condensable—particulates that form during the vapor phase upon cooling and dilution. Fugitive dust, stirred up from unpaved roads, stone crushing, or other processes, is characteristically filterable. Combustion processes, which require temperatures substantially higher than surrounding environmental conditions, generally emit both filterable and condensable $PM_{2.5.}$ Consistent with the definitions used in EPA's National Emissions Inventory, this report uses the terms primary and direct to refer to both filterable and condensable particulate matter.

Secondary $PM_{2.5}$ is formed when certain gases undergo chemical reactions with sunlight, water vapor in the air. Called precursor emissions, these gases include sulfur dioxide (SO_2), ozone (O_3), nitric acid (HNO_3), Volatile Organic Compounds (VOCs), and nitrogen oxides (NOx).

Figure 2.2: Formation of Secondary $PM_{2.5}$



Source: US EPA. http://www.epa.gov/air/urbanair/pm/what1.html

The same source can account for $PM_{2.5}$ emissions through different processes. Fuel combustion is a major source of secondary as well as primary $PM_{2.5}$.[4] For example, $PM_{2.5}$ is emitted directly from automobile exhaust, but it also forms from other tail-pipe emissions like nitrogen oxides (NOx).

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[2] US EPA, http://www.epa.gov/air/urbanair/pm/what1.html

[3] US EPA, http://www.epa.gov/pmdesignations/basicinfo.htm

[4] US EPA, http://www.epa.gov/air/urbanair/pm/what1.html

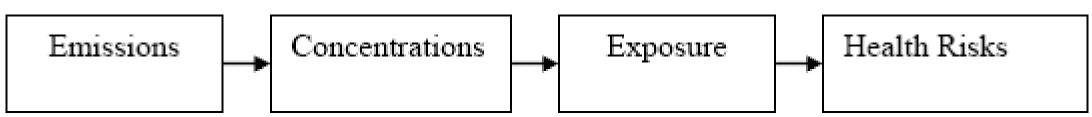
While PM and $PM_{2.5}$ are classified by size for regulatory and public health research purposes, other physical and chemical characteristics may also play a role in the extent to which a particular subcategory of $PM_{2.5}$ can adversely affect public health in New York City. To better understand the influence of particular source classes on $PM_{2.5}$ levels in NYC, it is useful to point out some of these subcategories and their sources.

The **solid fraction** of diesel PM is often referred to as "elemental carbon" ("EC") or "black carbon" and also includes ash deposits. The solid fraction of total PM is the primary source of the black smoke associated with heavy-duty diesel engines. Ash, on the other hand, primarily emanates from lubricating oil and metals due to engine wear. As newer engines are developed to produce less EC, the proportion of ash in the exhaust tends to increase. The **soluble organic fraction** (SOF) is composed of organic material from engine fuel and lubrication oil. SOF consists essentially of hydrocarbon deposited on the surface of elemental carbon (EC) particles and is often referred to as "wet PM." SOF formation is dependent on the duty cycle, with light load causing higher concentrations because of low engine temperature.[5] **Sulfate** (SO_4) is a combustion by-product that combines the sulfur in diesel fuel with water vapor, to form sulfuric acid mist. Reaction with ammonia in the ambient air subsequently forms ammonium sulfate or bisulfate particles.

Many variables come into play to determine the level of ambient $PM_{2.5}$—that is, $PM_{2.5}$ in the open-air—at a given place and time. Particulate matter can be suspended in the air for long periods of time, but it can also be transported many miles from its source by air currents. Therefore, the ambient concentration level here—consisting of directly emitted $PM_{2.5,}$ particles formed from precursors, or some combination of the two—depend on any number of anthropomorphic activities and weather conditions, both here and elsewhere.

The environmental and health effects linked to $PM_{2.5}$, discussed further in the following section, involve additional variables. There are several different steps that lead to the health threats caused by $PM_{2.5}$. Figure 2 illustrates the pathway leading from emissions to health risks as follows:

Figure 2.3: Pathway from Emissions to Health Effects



In other words, primary $PM_{2.5}$ and its precursors are emitted from a source (such as a power plant or a car) and result in $PM_{2.5}$ concentrations outdoors and indoors. Concentrations of $PM_{2.5}$ at any one point in space and time are determined by many factors, including the transport of primary emissions over long distances, and the creation of secondary $PM_{2.5}$ through complex chemical processes. Exposure occurs when people

[5] This is especially significant when considering diesel oxidation catalysts (DOCs) as a candidate retrofit device, since they are effective in reducing only the SOF portion of diesel PM. As emission regulations prompt generally higher operating temperatures in newer engines, SOF emissions decline.

spend time in areas with $PM_{2.5}$ concentrations. Finally, exposure leads to public health risks, if the exposure concentrations are high enough.

The number of variables involved in the creation and dispersion of $PM_{2.5}$, and in its exposure and health effects to humans, creates challenges for policy makers. Reducing public exposure to $PM_{2.5}$ may require controlling $PM_{2.5}$ and other pollutants, from many sources, in any number of places.

2.2 Health Effects of Fine Particulates

The health effects of exposure to airborne particles have been investigated over the last 50 years in New York City and elsewhere. Originally such research focused on the health impacts associated with exposure to coarse particles, which yielded to emphasis on fine particles as researchers developed more sophisticated detection technologies and a better understanding of the physiological responses to exposure to aerosol particles. Additional research identified sensitive populations, young children, the elderly and asthmatics as particularly susceptible to adverse health effects for fine particle exposure. A growing body of literature links exposure to PM to numerous adverse health impacts, including aggravation of chronic respiratory and cardiovascular diseases, decreased lung function, increased hospitalizations, emergency room visits and premature mortality.[6]

Combining research advancements in the fields of toxicology, human exposure assessment and epidemiology, with a better understanding of lung physiology, researchers believe that particles less than 2.5 µm in diameter, upon inhalation, are in the size range to penetrate into the deepest portions of the lung without being removed in the upper airway and are more likely to influence health than inhalation of larger particles.

Research on the health effects of exposure to $PM_{2.5}$ has compelled policy change on the state and federal levels. Based on some of the earlier research, EPA in 1997 established an annual exposure limit for $PM_{2.5}$ of 15 $\mu g/m^3$ with a 24 hour average of 65 $\mu g/m^3$, which it deemed sufficient to provide an adequate margin of safety to protect public health.[7] In 2003, the California Air Resources Board, utilizing more recent and sophisticated research on the health effects of exposure to aerosol particles adopted a fine particulate matter $PM_{2.5}$ annual exposure standard of 12 $\mu g/m^3$.[8] The sizeable and

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[6] Bell et al., Time-Series Studies of Particulate Matter, Annu. Rev. Public Health 2004 25:247, at 250.

[7] Clean Air Act Section 109(b)(1), 42 U.S.C. 7409(b)(1). See US Environmental Protection Agency, National Ambient Air Quality Standards for Particulate Matter, Final Rule 62 Fed.Reg. 38652 (July 18, 1997).

[8] California Air Resources Board (CARB), Final Regulation Order for the Rulemaking To Consider Amendments to Regulations for the State Ambient Air Quality Standards for Suspended Particulate Matter and Sulfates Final Regulation Order (June 4, 2003). Section 70200 Table of Standards. CARB adopted the more protective standard as required by state for the "[p]revention of excess deaths and illness from long-term exposure. Illness outcomes include, but are not limited to, respiratory symptoms, asthma exacerbations, and hospital admissions for cardiac and respiratory diseases. Sensitive subpopulations include children, the elderly, and individuals with preexisting cardiopulmonary disease." See http://www.arb.ca.gov/regact/aaqspm/aaqspm.htm.

growing literature informed the most recent NAAQS review, which adopted a more protective 24-hour standard for ambient $PM_{2.5}$ concentrations.

Particular aspects of the research literature on the health effects of exposure to $PM_{2.5}$ are summarized below. More detailed discussions and reviews of additional research findings of the health effects from exposure to fine particulate matter can be found elsewhere (US EPA, 2004).

We can categorize the evidence for $PM_{2.5}$ into studies of acute effects and of chronic effects. The most substantial body of literature is related to premature death (and premature death tends to dominate benefits assessments), so we focus on those studies here.

2.2.1 Exposure to $PM_{2.5}$ and Premature Mortality

Numerous time-series studies of particulate matter mortality risks have been published in settings across the world. Time series analysis of morbidity and mortality data investigates associations between short-term changes in pollution concentrations with short-term changes in acute health outcomes. In general, these studies suggest that an increase of 10 $\mu m/m^3$ in daily average PM_{10} concentrations results in a 0.5-1 percent increase in total daily deaths (US EPA, 2004). The deaths tend to be due to either cardiovascular or respiratory disease and appear to result in a loss of life expectancy of at least one month.[9] It has also been shown that mortality risks occur at relatively low particulate matter concentrations, well below both air quality standards and concentrations in NYC.[10]

There is also evidence of long-term mortality risks from particulate matter exposure. Two large prospective cohort studies (Dockery et al. 1993; Pope et al. 2002) evaluated the relationship between long-term pollution exposure and mortality risk, controlling for individual risk factors such as smoking, diet, or occupational exposures.[11] In the Six Cities study (Dockery et al. 1993), which focused on over 8,000 white adults living in the eastern half of the US, the risk of death was 26 percent higher in the most-polluted city when compared with the least-polluted city (controlling for other factors). This corresponded to an approximate 1.3-percent increase in mortality for every $\mu g/m^3$ of annual average $PM_{2.5}$, substantially higher than the time-series estimates. The American Cancer Society study followed over 500,000 adults in all 50 US states, finding that a 1 μg/m3 increase in annual average $PM_{2.5}$ concentrations was associated with a 0.6-percent

[9] Schwartz J, Ballester F, Saez M, et al, The Concentration-Response Relation Between Air Pollution and Daily Deaths. Environ Health Perspect. 2001 Oct; 109(10);1001-6.

[10] Daniels MJ, Dominici F, Samet JM, Zeger SL, Estimating Particulate Matter-Mortality Dose-Response Curves and Threshold Levels: An Analysis of Daily Time-Series for the 20 Largest US Cities. Am J Epidemiol. 2000 Sep 1;152(5):397-406; Schwartz J, Laden F, Zanobetti A, The Concentration-Response Relation Between PM(2.5) and Daily Deaths. *Environ Health Perspect.* 2002 Oct;110(10);1025-9.

[11] Dockery DW, Pope CA III, Xu X, et al, An Association Between Air Pollution and Mortality in Six U.S. Cities. *N Engl J Med.* 1993 Dec 9;329(24):1753-9; Pope CA III, Burnett RT, Thun MJ, et al, Lung Cancer, Cardiopulmonary Mortality, and Long-Term Exposure to Fine Particulate Air Pollution. JAMA 2002 Mar 6;287(9):1132-41.

increase in mortality rates.[12] Significant effects were seen on both cardiopulmonary and lung cancer death, and there was no evidence of a threshold.

In summary, there is substantial evidence supporting a relationship between particulate matter exposure and cardiopulmonary health (including premature death). This effect appears to be associated with fine particles ($PM_{2.5}$), although uncertainty remains regarding the precise sizes and chemical constituents most associated with health impacts. Because the cohort mortality effect is larger than the time-series mortality effect and theoretically includes some fraction of the acute deaths, the cohort mortality concentration-response functions are usually used for risk assessment calculations.

2.2.2 Potential Additional $PM_{2.5}$ Risk Factors in New York City

A specific question of interest for this report would be whether the health effects of $PM_{2.5}$ might be expected to differ in NYC as compared with the average national impact. One might expect differences for a variety of reasons: the levels of particulate matter may differ, the composition of particulate matter may differ, time-activity patterns influencing personal exposures from ambient concentration may differ, and the relative proportion of high-risk individuals may differ. Although the evidence base to allow for deviations from nationally-derived concentration-response functions is limited, we briefly consider each of these issues in turn.

The first of these issues has to do with whether there is a threshold for $PM_{2.5}$, or in general, whether the concentration-response function displays non-linearities that would lead to a different slope at levels in NYC versus levels elsewhere. The literature in this area is somewhat limited, but as mentioned above, studies such as the National Morbidity, Mortality, and Air Pollution Study (NMMAPS)[13] and the American Cancer Society cohort study[14] found no evidence of thresholds or non-linearities within the range of concentrations reported.[15]

[12] Pope CA III, Thun MJ, et al, Particulate Air Pollution As A Predictor of Mortality In A Prospective Study of U.S. Adults. *Am J Respir Crit Care Med.* 1995 Mar; 151(3 Pt 1):669-74; Pope CA III, Burnett RT, Thun MJ, et al, Lung Cancer, Cardiopulmonary Mortality, and Long-Term Exposure to Fine Particulates in Air Pollution. *JAMA* 2002 Mar 6;287(9):1132-41.

[13] Samet JM, Zeger SL, Dominici F, Curriero F, et al, The National Morbidity, Mortality, and Air Pollution Study. Part II: Morbidity and Mortality from Air Pollution in the United States. *Res Rep Health Eff Inst.* 2000 Jun; 94(Pt. 2):5-70:discussion 71-9. See also Dominici F, McDermott A, Daniels M, et al, Revised Analyses of the National Morbidity, Mortality, and Air Pollution Study: Mortality Among Residents of 90 Cities. *J Toxicol Environ Health* A. 2005 Jul 9-23;68(13-14:1071-92.

[14] Pope CA III, Burnett RT, Thun MJ, Calle EE, et al, Lung Cancer, Cardiopulmonary Mortality, and Long Term Exposure to Fine Particulate Air Pollution, JAMA. 2002 Mar 6;287(9):1132-41; Daniels MJ, Dominici F, Samet JM, Zeger SL, Estimating Particulate Matter-Mortality Dose-Response Curves and Threshold Levels: An Analysis of Daily Time-Series for the 20 Largest US Cities. Am J Epidemiol. 2000 Sep 1;152(5):397-406.

[15] Likewise, long term mean $PM_{2.5}$ concentration at levels below EPA's standard of 15 $\mu g/m^3$ has been associated with excess hospital visits by children under 18, consistent with the lack of evidence for a threshold below which the adverse incremental effects of $PM_{2.5}$ will not occur. Pre-filed Direct Testimony of George D. Thurston, Sc.D. before the New York State Board on Electric Generation Siting and the Environment. Case 99-F-1627. Mar 13, 2005, p. 11-13.

As the more protective California fine particle matter $PM_{2.5}$ annual exposure standard of 12 $\mu g/m^3$ implies, existing ambient $PM_{2.5}$ concentrations in NYC are exposing residents to greater potential adverse health effects. In its 2002 review of studies of exposure-response relationships between $PM_{2.5}$ and mortality, the Staff of the California Air Resources Board (CARB) reported that linear, nonthreshold models provided a better fit than non-linear models.[16] The most recent review of the $PM_{2.5}$ NAAQS assumes a human health-effects threshold of 10 $\mu g/m^3$ but acknowledges that "effect thresholds can neither be discerned nor determined not to exist" in either the short or long term and called for more research "to determine the existence and level of any thresholds that may exist or the shape of nonlinear concentration-response curves at low levels of exposure that may exist."[17] EPA's AirNOW public advisory program associates health concerns for unusually sensitive groups with $PM_{2.5}$ concentrations of 15 $\mu g/m^3$, well below the recently-adopted federal 24-hour standard of 35 $\mu g/m^3$.[18]

While some have argued that epidemiological studies based on central site monitors lack the resolved personal exposure data necessary to detect thresholds or non-linearities were they to exist, at a minimum, it is difficult to conclude that the concentration-response function in NYC would differ from elsewhere strictly based on ambient concentrations. However, since ambient $PM_{2.5}$ concentrations in NYC are 13-17 $\mu g/m^3$, it would be relatively more likely that health effects would be seen, relative to geographic locations with lower levels. As fine aerosol particle matter demonstrates an ability to penetrate deep within the respiratory systems of humans and other animals, it has the potential to reach the circulatory system, making the composition of such particles another area of concern. The composition question of fine aerosol particle matter is complicated by the limited evidence available on differential toxicity of particle constituents.

NMMAPS evaluated regional differences in concentration-response functions for time-series studies (Dominici et al., 2003). The concentration-response function was highest in the Northeast as compared with other regions of the country, although the root cause for this difference is unknown. There has been an extensive recent literature attempting to determine the relative toxicity of various particle constituents, with limited definitive conclusions. In the long-term cohort studies mentioned above (Dockery et al., 1993;

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[16] Staff reported that "multiple analyses of the exposure-response relationships between $PM_{2.5}$ and mortality indicate that the data can be fitted most parsimoniously with linear, nonthreshold models. Given the apparent linearity of the exposure-response relationships in the epidemiological data, it is difficult to determine at what concentrations within the $PM_{2.5}$ distributions in each study adverse health effects begin. Intuitively, one would expect greater biological responses and larger numbers of adverse events occurring at higher concentrations, everything else being equal. Nonetheless, in a linear exposure-response relationship, effects may be observed at lower levels as well" (Staff of the California Air Resources Board and the Office of Environmental Health Hazard Assessment, "Staff Report: Public Hearing to Consider Amendments to the Ambient Air Quality Standards for Particulate Matter and Sulfates", May 3, 2002. p. 7-94)

[17] US Environmental Protection Agency, National Ambient Air Quality Standards for Particulate Matter, Final Rule, 40 CFR Part 50. (September 21, 2006)

[18] Levels of health concerns are expressed as an Air Quality Index (AQI). An AQI of 100 for $PM_{2.5}$, classified as moderate to unhealthy, corresponds to a level of 40 $\mu g/m^3$ averaged over 24 hours. See http://airnow.gov/index.cfm?action=aqibroch.aqi#aqipar and http://airnow.gov/index.cfm?action=static.publications.

Pope et al., 2002), there was a relatively consistent association between sulfate particles and premature mortality, but data were not available to consider other particle constituents. In the time-series literature, most relevant analyses considered sulfate particles as well, finding generally positive associations (EPA, 2004).

In the limited literature that has addressed numerous particle constituents, statistical power has often been lacking to disentangle the effects of individual constituents. That is why a few studies have used factor analysis techniques to attribute PM-related health effects to source categories. Laden and colleagues (2000) applied statistical methods to elemental data from the Six Cities Study and found that factors for motor vehicles and coal had statistically significant effects on premature mortality, with the motor vehicle factor approximately a factor of three greater than the coal factor (per unit concentration).[19] A crustal factor was not significant. Although the confidence intervals were wide, there was some evidence that cardiovascular deaths were more closely related to motor vehicle particles and respiratory deaths were more closely related to coal-derived particles. Other factor analysis studies similarly concluded that combustion particles appeared more toxic than non-combustion particles.[20] Combustion of oil and, to a lesser extent, natural gas creates fine particulate emissions containing metals, including cadmium, barium, chromium, molybdenum, and zinc. Compared to emissions from burning oil, a higher percentage of particle emissions from gas combustion are ultra-fine particulates, which may increase the toxicity of the metal components.[21]

In terms of the personal exposure versus ambient concentration question, this depends in large part on where people spend their time and how their homes are built. In a dense urban area, one might anticipate that people would spend substantial time in close proximity to motor vehicles, where PM concentrations have been shown to be significantly higher.[22] However, reduced time in motor vehicles due to public transportation use may be an offsetting factor. Air exchange rates in the home, associated in part with air conditioning in the summer, can modify personal exposures per unit ambient concentration. A study by Janssen and colleagues (2002) found that cities with higher air conditioning prevalence had lower PM concentration-response functions for

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[19] Laden F, Neas LM, Dockery DW, et al, Association of Fine Particulate Matter From Different Sources With Daily Mortality in Six U.S. Cities, *Environ Health Perspect*. 2000 Oct;108(10):941-7. See also Laden F, Schwartz J, Speizer FE, et al, Reduction in Fine Particulate Air Pollution and Mortality: Extended Follow-up of the Harvard Six Cities Study, *Am J Respir Crit Care Med*. 2006 Jan 19; [Epub ahead of print]

[20] Mar TF, Norris GA, Koenig JQ, et al, Associations Between Air Pollution and Mortality In Phoenix, 1995-1997. *Environ Health Perspect.* 2000 Apr;108(4);347-53.

[21] Pre-filed Direct Testimony of George D. Thurston, Sc.D. before the New York State Board on Electric Generation Siting and the Environment. Case 99-F-1627. Mar 13, 2005, p. 36. See also Sioutas C, Delfino RJ, Singh M.Exposure assessment for atmospheric ultrafine particles (UFPs) and implications in epidemiologic research. *Environ Health Perspect.* 2005 Aug;113(8):947-55; Delfino RJ, Sioutas C, Malik S. Potential role of ultrafine particles in associations between airborne particle mass and cardiovascular health. *Environ Health Perspect.* 2005 Aug;113(8):934-46.

[22] Zhu Y, Hinds WC, Kim S, et al, Cocentration and Size Distribution of ultrafine particles near a major highway. *J Air Waste Manag Assoc.* 2002 Sep;52(9);1032-42.

hospital admissions.[23] NYC was not one of the 14 cities considered, but similar cities in the Northeast (i.e., New Haven and Pittsburgh) had lower air conditioning prevalence as compared with cities in the South, and had correspondingly higher concentration-response functions.

Finally, addressing the population composition question, one might think that risks would differ per unit exposure in NYC if there were demographic, social, behavioral, or medical factors that differed substantially in NYC compared to the rest of the nation (and if these factors were linked with PM health effects). Scientific research has demonstrated that the very young, the very old, and those with pre-existing health conditions (such as heart disease and asthma) are particularly affected by air pollution. With one of the highest burdens of asthma problems in the country, NYC residents are highly susceptible to increases in air pollution. Population groups with high incidence of asthma, including Hispanics and the poor, are well represented in NYC neighborhoods near emissions-producing power plants.[24],[25] Evaluating these data is beyond the scope of this report, but is a topic that would need to be addressed for formal risk assessments, especially if focused on subpopulations of interest.

NYSERDA is currently sponsoring a number of studies that will shed additional light on fine particulate sources, concentrations and health effects in NYC and New York State. One such study, at the NY Department of Health, is focusing on fine particulate matter and acute asthma in NYC. The goal of this work is to evaluate temporal associations between a number of air pollutants, including fine particulates, and emergency room visits for asthma. Two areas of NYC are being studied, one in the South Bronx and one in Manhattan.[26]

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[23] Janssen NA, Schwartz J, Zanobetti A, et al, Air Conditioning and Source-specific Particles as Modifiers of the Effect of PM(10) on Hospital Admissions for Heart and Lung Disease. *Environ Health Perspect.* 2002 Jan;110(1):43-9.

[24] Thurston (2005) reports that Hispanics and the poor comprise a far higher percentage of the residential population in NYC than they do in other counties in New York State and in the U.S. Populations in the vicinity of Astoria, Queens, the site of a cluster of power plants, have even higher percentages of Hispanics and persons living below the poverty line. Another at-risk population, children under 18, are more highly represented in these census tracts than in the U.S. as a whole. (Pre-filed Direct Testimony of George D. Thurston, Sc.D. before the New York State Board on Electric Generation Siting and the Environment. Case 99-F-1627. Mar 13, 2005, p. 26-27.)

[25] Compared to national and state averages, the counties with the highest ambient $PM_{2.5}$ levels have relatively large minority and low-income populations. Of the five NYC boroughs, average annual ambient levels of $PM_{2.5}$ from 2000-2004 are highest in Manhattan, followed by Brooklyn and the Bronx. Manhattan has a large Hispanic population (27%) relative to the citywide (17%) and statewide (15%) Hispanic populations. The percent of families that are below the poverty level is much higher in Brooklyn (22%) than in the state as a whole (12%). Likewise, African Americans comprise a much larger share of the population in Brooklyn (36%) than in New York State (16%). The Bronx is home to large populations of African Americans (36%), Hispanics (48%) and families below the poverty level (28%). (U.S. Census Bureau, Census 2000 Redistricting Data (Public Law 94-171) Summary File, Matrices PL1 and PL2.)

[26] For more information on this and other NYSERDA-sponsored projects, see: www.nyserda.org/programs/environment/emep.asp.

In conclusion, the available $PM_{2.5}$ research appears to suggest that existing ambient concentrations are sufficient to influence cardiorespiratory mortality and morbidity in NYC. Although the evidence for location-specific concentration-response functions is limited, one might anticipate slightly higher toxicity from particulate matter in NYC as compared with the national average, given the higher ambient concentration, the preponderance of combustion particles, and the proximity to motor vehicle sources.

2.3 Regulation of Fine Particulates

The Clean Air Act requires the EPA to promulgate National Ambient Air Quality Standards (NAAQS) for pollutants considered "harmful to public health and the environment." Two types of standards are established. Primary standards are set to protect public health, including "the health of 'sensitive' populations such as asthmatics, children and the elderly." Secondary standards protect "public welfare, including protection against decreased visibility, damage to animals, crops, vegetation and buildings."

In 1971, the EPA promulgated the first national particulate matter standard establishing a maximum ambient concentration not to be exceeded in a 24-hour period of 260 $\mu g/m^3$ and an annual average concentration not to be exceeded of 75 $\mu g/m^3$.[27] The standard measured "total suspended particulate" (TSP) matter, e.g., particulates up to 45 µg in diameter.

By 1987, spurred by a deeper understanding of the adverse health affects posed by exposure to smaller aerosol particles, the EPA eliminated the TSP standard in favor of a PM_{10} standard, targeting particles 10 µg and smaller. Under this standard, an area cannot exceed the 24-hour maximum ambient concentration (150 $\mu g/m^3$) more than once per year, and additionally must have a 3-year weighted-average of 50 $\mu g/m^3$ or less. The borough of Manhattan (New York County) was designated as a non-attainment area for PM_{10}.[28]

In 1997, EPA again revised the standard and created a $PM_{2.5}$ standard. The $PM_{2.5}$ standard consists of two parts: an annual standard[29] of 15.0 $\mu g/m^3$ and a 24-hour standard[30] of 65 $\mu g/m^3$. This standard was challenged by business groups led by the American Trucking Association and the U.S. Chamber of Commerce. In February 2001, the U.S. Supreme Court ruled that EPA did have the authority under the Clean Air Act to promulgate standards for pollutants that harm public health.[31] The following month, the

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[27] U.S. Environmental Protection Agency, National Primary and Secondary Ambient Air Quality Standards 36 Fed.Reg. 8186 (1971).

[28] U.S. Environmental Protection Agency, http://www.epa.gov/oar/oaqps/greenbk/pncs.html#NEW%20YORK.

[29] To attain this standard, the 3-year average of annual arithmetic mean $PM_{2.5}$ concentrations from single or multiple community-oriented monitors must not exceed 15.0 ug/m3.

[30] 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 ug/m3.

[31] "Will Christie Whitman be Carol Browner?," *The Electricity Daily*, Vol. 16, No.41, 1 March 2001.

D.C. Circuit Court rejected all remaining challenges, allowing EPA to move forward with enforcement of the $PM_{2.5}$ standards. The agency finalized the $PM_{2.5}$ designations in December of 2004, and all five boroughs of New York City are non-attainment areas, as well as Nassau, Orange, Rockland, Suffolk and Westchester Counties.[32] States will be required to submit state implementation plans (SIP) by April 2008.[33] New York State's Division of Air Resources (DAR) has yet to begin modeling potential control strategies for inclusion in its SIP.[34]

Since EPA set the first $PM_{2.5}$ NAAQS in 1997, numerous studies have demonstrated an increased risk of premature death from short and long term exposures at ambient concentrations below current standards.[35] As a part of the most recent review cycle for $PM_{2.5}$ NAAQS, the January 2005 draft Staff Paper assessed the risks of ambient $PM_{2.5}$ and recommended a range of more stringent $PM_{2.5}$ standards. This Draft recommended two options for the new standard: 15 $\mu g/m^3$ annual average combined with 35 to 25 $\mu g/m^3$ daily; or 14 to 12 $\mu g/m^3$ annual average combined with 35 to 40 $\mu g/m^3$ daily. Both recommended standards use the 98th percentile form.[36,37]

On June 6, 2005, EPA's panel of independent science reviewers, the Clean Air Scientific Advisory Committee (CASAC), endorsed adoption of a more stringent set of standards than contemplated in the draft Staff Paper. In its letter to the Administrator, the Committee stated that most members favored setting the 24-hour standard at concentrations between 35 to 30 $\mu g/m^3$ with the 98th percentile form, along with an annual NAAQS in the range of 14 to 13 $\mu g/m^3$.[38] Stakeholders have recommended even stricter standards than the CASAC. For example, the American Lung Association advocates changing the annual standard to 12 $\mu g/m^3$, and the 24-hour standard to 99th-percentile, not-to-exceed 25 $\mu g/m^3$. Both examples would substantially increase public health protection by expanding coverage to more of the populations affected by $PM_{2.5}$ and requiring stronger policy measures in current non-attainment areas.

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[32] 40 CFR Part 81; Air Quality Designations and Classifications for the Fine Particles

(PM2.5) National Ambient Air Quality Standards; Final Rule. See also U.S. Environmental Protection Agency, http://www.epa.gov/pmdesignations/finaltable.htm.

[33] U.S. Environmental Protection Agency, http://www.epa.gov/pmdesignations/documents/120/timeline.htm.

[34] Sheehan, Mike (Bureau of Air Quality Planning, a division of DAR). Personal communication, January 17, 2007.

[35] For a summary of these studies, see "Adverse Health Effects of Particulate Matter: New Science Shows Effects Below Current Standards." American Lung Association. http://www.cleanairstandards.org/filemanager/download/129/Science%20Summary%20FINAL.doc.

[36] U.S. Environmental Protection Agency OAQPS Staff. January 2005. "Review of the National Ambient Air Quality Standards for Particulate Matter: Policy Assessment of Scientific and Technical Information" Second Draft. Available at http://www.epa.gov/ttn/naaqs/standards/pm/s_pm_cr_sp.html.

[37] US EPA, National Ambient Air Quality Standards for Particulate Matter, Proposed rule. 40 CFR Part 50 [OAR-2001-0017; FRL-8015-8] RIN 2060-AI44.

[38] Particulate Matter Review Panel of the EPA Clean Air Scientific Advisory Committee, US EPA, "EPA's Review of the National Ambient Air Quality Standards for Particulate Matter (Second Draft PM Staff Paper, January 2005)". Available at http://www.epa.gov/sab/pdf/casac-05-007.pdf.

On December 20, 2005, EPA announced its recommendation to lower the level of the 24-hour fine particle standard from the current level of 65 µg/m^3 to 35 µg/m^3, and to retain the level of the annual fine standard at 15 µg/m^3.[39] The CASAC requested reconsideration of the proposed ruling in March 2006, citing the significance of the most recent research and stating that "epidemiologic evidence, supported by emerging mechanistic understanding, indicates adverse effects of $PM_{2.5}$ at current annual average levels below 15 µg/m^3."[40] Nevertheless, EPA adopted the December 2005 proposed standard (35 µg/m^3 24-hour, 15 µg/m^3) on September 21, 2006, while deferring consideration of toxicology and epidemiologic studies that were not published in time to be included in the 2004 Criteria Document to the next $PM_{2.5}$ NAAQS review. The Administrator of the EPA set the next review of the PM NAAQS to begin immediately following the September rulemaking, so that the newly published research findings could be considered in a timely fashion.[41]

Concurrent to the $PM_{2.5}$ review process, EPA finalized the Clean Air Interstate Rule (CAIR) in March, 2005.[42] This rule places additional restrictions on the SO_2 and NOx emissions of 28 eastern states and the District of Columbia, to reduce their contributions to $PM_{2.5}$ and 8-hour ozone nonattainment in downwind areas. Under CAIR, states can achieve the required emissions reductions by either requiring power plants to participate in an interstate cap and trade system, or meeting individual state air emission limits through state-defined measures.[43] The final rule requires states to amend and submit their SIPs by September of 2006. Limits on emissions go into effect in two stages. Phase I of CAIR NOx programs begins in 2009; Phase I for SO_2 starts in 2010. Phase II for both SO_2 and NOx commences in 2015. While CAIR is projected to reduce premature mortality due to $PM_{2.5}$ emissions from U.S. power plants, it falls short of meeting current technical potential to reduce emissions. Based on EPA's IPM modeling, the Clean Air Task Force estimated that CAIR will allow premature deaths—9,000 in 2015—that could be prevented with current technology.[44] While CAIR will help reduce ambient $PM_{2.5}$

[39] US EPA, National Ambient Air Quality Standards for Particulate Matter, Proposed rule. 40 CFR Part 50 [OAR-2001-0017; FRL-8015-8] RIN 2060-AI44.

[40] Dr. Rogene Henderson, Chair of the Clean Air Scientific Advisory Committee of the US Environmental Protection Agency, letter to Administrator Stephen L. Johnson. March 21, 2006. Available at http://www.epa.gov/sab/panels/casacpmpanel.html.

[41] US Environmental Protection Agency, National Ambient Air Quality Standards for Particulate Matter, Proposed Rule, 71 Fed.Reg. 2620 (January 17, 2006); US Environmental Protection Agency, National Ambient Air Quality Standards for Particulate Matter, Final Rule, 40 CFR Part 50. (September 21, 2006)

[42] U.S. Environmental Protection Agency, Rule To Reduce Interstate Transport of Fine Particulate Matter and Ozone (Clean Air Interstate Rule); Revisions to Acid Rain Program; Revisions to the NOx SIP Call; Final Rule 70 Fed.Reg. 25161 (May 12, 2005).

[43] http://yosemite.epa.gov/opa/admpress.nsf/d9bf8d9315e942578525701c005e573c/5af79c40e7ba7f2685256ffe00642ad3!OpenDocument

[44] Clean Air Task Force, Comments on Supplemental Proposal for the Rule to Reduce Interstate Transport of Fine Particulate Matter and Ozone (Clean Air Interstate Rule), 69 Fed. Reg. 32684 (June 10, 2004) Docket ID No. OAR-2003-0053, p. 29. See http://www.catf.us/advocacy/legal/IAQR/CATF_CAIR_Comments.pdf

concentrations in NYC, achieving attainment with the 2006 $PM_{2.5}$ NAAQS will nonetheless pose a challenge for the city and state, given that all five boroughs are designated as in non-attainment with the 1997 $PM_{2.5}$ NAAQS and have yet to address how they will comply with the stricter 2006 federal standards.

2.3.1 Efforts to control PM sources in New York City

Adverse public health impacts from exposure to PM are not a new phenomenon in NYC. The NYC Department of Air Pollution Control was created in 1944, at time when coal combustion for heating purposes had been supplanted and in the City by the use of residual fuel oil with high sulfur content (2-3 percent), used extensively in large commercial and residential buildings through the 1970s. Air pollution monitoring began in the 1950s, first on 121st Street and expanding until the 1970s when an extensive air pollution monitoring network of about 40 stations across all five boroughs of NYC.[45] Elevated aerosol particle concentrations were observed throughout all five boroughs and were attributed to vehicle emissions, use of residual fuel for heating, and the on-site incineration of municipal refuse, all of which added to the particulate burden.[46] NYC grappled with the problem, making relatively little progress in protecting public health for several decades. Mortality associated with elevated pollution episodes aggravated by temperature inversions in 1962, 1963 and 1966 spurred both a series of municipal actions and enactment of federal measures that eventually curbed many sources of PM emissions.

In response to the public health problem, recent municipal ordinances have targeted some sources of $PM_{2.5}$. The NYC Metropolitan Transit Authority (MTA) has converted its diesel buses to burn ultra-low-sulfur diesel fuel (ULSD) and retrofitted them with diesel particulate filters (DPFs), and both the MTA and private bus companies subsidized by New York City DOT have refueled buses to run on compressed natural gas. In addition to fuel retrofits, in October 2005 NYC adopted a bill mandating that municipal building projects receiving more than $10 million in city funds be built to “high performance” standards.[47]

As part of a larger initiative to advance sustainability in NYC, in late 2006 Mayor Bloomberg established an environmental goal for the City: achieving the cleanest air quality of any big city in America.[48]

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[45] Lipfert, F.W. Air Pollution and Community Health: A Critical Review and Data Sourcebook, Van Nostrand Rheinhold (New York: 1994) at 166.

[46] Id.

[47] Clean Air-Cool Planet, “NYC sets green standards for public buildings” The Local Climate, Third Edition, November 17, 2005. Available at http://www.cleanair-coolplanet.org/for_communities/localclimate/E3.pdf.

[48] The Mayor set other environmental goals for the city, including providing cleaner power and reducing the City’s global warming emissions by more than 30% by 2030. Other stated goals would affect air quality by allowing higher population density. These include cleaning up all contaminated land for redevelopment, as well as restoring and improving roads, subways, rails, water, and energy infrastructure. (The City of New York, Office of the Mayor. Dec 12, 2006. “Mayor Bloomberg Delivers Sustainability Challenges and Goals for New York City through 2030.” Press release. http://www.nyc.gov/)

2.3.2 Efforts to Control PM in New York State

New York State has actively worked towards improving its air quality for over three decades. Established in 1975 by the state legislature, the New York State Energy Research and Development Authority (NYSERDA) is charged with improving the state's environment, particularly as it is affected by emissions resulting from energy generation and consumption. NYSERDA's research into $PM_{2.5}$ speciation, sources, and health effects through the Environmental Monitoring, Evaluation, and Protection (EMEP) program provides valuable guidance for cost-effective emissions control policies. In addition, NYSERDA has taken a prominent role in advancing non-polluting renewable sources of energy and in reducing energy consumption by homes, businesses and industry in New York through consumer education and assistance programs.

More recently, the state has planned and implemented electricity system policies with air quality improvement as one of several goals. In 2004, New York established a Renewable Portfolio Standard to reduce the negative environmental impacts associated with fossil fuel consumption, improve energy security, and help diversify the state's electricity generation mix. The current standard requires that renewable energy resources account for 24% of energy consumption by 2013.

As another example, New York Governor Pataki initiated the Regional Greenhouse Gas Initiative (RGGI), a cooperative process to establish a cap-and-trade program for CO_2 emissions from power plants in nine Northeast and Mid-Atlantic states. Implementation of RGGI would produce concurrent reductions in greenhouse gases and $PM_{2.5}$, as well as in other federally-regulated air pollutants.

2.3.3 California Goes Beyond the Federal Standard

In 1999, the Children's Environmental Health Protection Act (California Senate Bill 25. Stats/1999, Ch. 731) required the California Air Resources Board (CARB) along with the Office of Environmental and Human Health Assessment (OEHHA) to "review all existing health-based ambient air quality standards to determine whether, based on public health, scientific literature, and exposure pattern data, these standards adequately protect the health of the public, including infants and children, with an adequate margin of safety." The particulate matter standards were identified by OEHHA as being the first to merit review. Following a 2002 report detailing the OEHHA/ARB staff's review of scientific and health studies regarding particulate matter, it was recommended that California's AAQS for particulate matter be revised. In its report, CARB states that:

> …the potential health impacts from exposure to particulate matter (PM) air pollution are significant. Health effects associated with PM exposure include: premature mortality, increased hospital admissions for cardiopulmonary causes, acute and chronic bronchitis, asthma attacks and emergency room visits, respiratory symptoms, and days with some restriction in activity. These adverse health effects have been reported primarily in infants, children, the elderly, and those with pre-existing cardiopulmonary disease."[49]

[49] "Staff Report: Public Hearing to Consider Amendments to the Ambient Air Quality Standards for Particulate Matter and Sulfates." Prepared by the Staff of the California Air Resources Board and the

CARB advised lowering both the 24-hour and annual standards for particulate matter. **Table** below, compares the federal $PM_{2.5}$ standard to the proposed California standards.

Table 2.1: Federal and California Standards for Ambient $PM_{2.5}$

	Annual	24-Hour
California Standard	12.0 μg/m^3 (1)	25 μg/m^3 (2)
1997 Federal Standard	15.0 μg/m^3 (3)	65 μg/m^3 (4)
2006 Federal Standard	15.0 μg/m^3 (3)	35 μg/m^3 (4)

(1) 24-hour samples, arithmetic mean
(2) Proposed
(3) 3-year average of annual arithmetic mean
(4) 3-year average of the 98th percentile of 24-hour concentrations

CARB eventually adopted all the Staff's recommendations with the exception of the 24-hour standard for $PM_{2.5}$. The Staff's recommendations for that standard were primarily based on a review of health studies performed by outside consultants. During the hearing process, CARB learned that a statistical software package used in several of the studies contained an error which could call certain results into question. CARB decided to wait until the studies had been modified before acting on the Staff's recommendation for that standard.[50] CARB is still reviewing the basis for establishing the 24-hour standard as of May, 2006.

Office of Environmental Health Hazard Assessment, May 3, 2002, ftp://ftp.arb.ca.gov/carbis/research/aaqs/std-rs/pm-final/exesum.pdf, p. 2.

[50] Regarding the deferral see http://www.arb.ca.gov/research/aaqs/std-rs/2_5defer.htm.

3. Ambient Levels and Sources of $PM_{2.5}$

3.1 Ambient Levels in NYC

Data on ambient air quality can be very useful in understanding overall trends in air quality over space and time, for judging the impacts of sources, and for determining compliance with the National Ambient Air Quality Standards (NAAQS). These data also permit calculating relative health risks for given exposure levels, since the epidemiological evidence is generally based on ambient aerosol particle data drawn from central-site monitors.

Ambient PM concentrations do not necessarily represent the levels to which individuals are exposed. There are several reasons for this. First, Americans spend an average of 90 percent of their time indoors, and aerosol particle concentrations, including $PM_{2.5}$ can often be greater indoors compared to outdoor concentrations. Although studies tend to indicate reasonably high temporal correlations between central site ambient $PM_{2.5}$ concentrations and average personal exposures, the relationships for individual subjects vary.[1] In addition, the indoor environment has important sources of many pollutants, including $PM_{2.5}$. These sources include cooking, cigarette smoking, vacuuming, and other activities. As a result, higher concentrations of $PM_{2.5}$ usually are seen indoors than outdoors, especially when buildings are tightly sealed for energy efficiency.

Another factor that may limit the usefulness of ambient monitoring data for characterizing individual population exposures is the fact that monitors are few in number, typically located on roofs two or more stories up, and often do not capture the impact of local sources, e.g., the impact of heavy traffic roadways on nearby residents. Indeed, the siting criteria for ambient aerosol particle monitors emphasize selecting locations that are likely to be representative of average concentrations across the entire airshed. For these reasons, supplementing the ambient monitoring data with personal monitoring provides a more robust characterization of population exposures.

However, these noted limitations of ambient air quality data do not mean that health estimates based on these data are incorrect or meaningless. As noted below, epidemiological studies tell us that on days when air pollution levels are high, more people get sick or die. Although we have a limited ability to know who is at highest risk (because of our lack of personal exposure data and knowledge about population susceptibility), we can still draw population-wide inferences as long as average personal exposures increase when outdoor levels increase. For pollutants that penetrate indoors such as $PM_{2.5}$, this is indeed the case.

$PM_{2.5}$ monitoring is conducted at a number of locations throughout NYC and elsewhere on a regular basis to help evaluate whether there are NAAQS violations. Data from this monitoring system are contained in the AirData System, a database maintained by the

[1] Ozkaynak, H. et. al., *Personal exposure to airborne particles and metals: results from the Particle TEAM study in Riverside, California*, Journal of Exposure Analysis and Environmental Epidemiology, 1996.

U.S. Environmental Protection Agency (http://www.epa.gov/air/data). $PM_{2.5}$ is measured using a variety of monitoring instruments. Manual, filter-based instruments typically collect data from midnight to midnight on a once in every three or six day schedule. The result is a regular, but intermittent, series of 24-hour average data. Continuous, real-time instruments provide hourly records of $PM_{2.5}$ with no gaps. The former provides additional information on the chemical composition of collected aerosol particles, while the latter can provide hourly data, revealing diurnal trends within 24 hour periods.[2]

A map of continuous $PM_{2.5}$ monitoring stations throughout the five boroughs of NYC is shown below in Figure 3.1. Annual average $PM_{2.5}$ concentration data for 2000-2004 from continuous monitors is shown in Table 3.1 on the following page. All boroughs except Queens and Staten Island experienced average $PM_{2.5}$ levels that exceed the annual NAAQS of 15 $\mu g/m^3$. This may reflect the influence of local combustion source emissions in the more highly populated boroughs. Key local sources are likely to include vehicular traffic (throughout the year) and home heating oil combustion (during the heating season). It is worth noting that the absence of an exceedance[3] does not mean that there are no air quality concerns. With only two continuous $PM_{2.5}$ monitors in Queens, there may be air quality concerns closer to neighboring boroughs of Manhattan and the Bronx, where monitors show higher ambient $PM_{2.5}$ levels, or in areas that do not have sufficient air flow. To put these numbers into perspective, Figure 3.2 shows a map of annual average $PM_{2.5}$ concentrations across the US for 2003. $PM_{2.5}$ levels are high in Southern California and in the Eastern US, and the levels in NYC are similar to those seen throughout the northeast.

[2] Roughly half of the monitoring locations in NYC are required by US EPA under Clean Air Act and follow Federal Reference Method (FRM) procedures in siting location, sampling technologies, and laboratory analytical procedures. However, while the FRM monitoring location network is adequate, the instrumentation cannot provide either real-time or near real-time data on current conditions, and NY DEC maintains non-FRM instrumentation, many co-located alongside the FRM instruments to provide real-time data. NY DEC also maintains other monitoring locations due to the non-homogeneous nature of land uses. Although the monitoring program faces budget cuts from the US EPA, maintaining the existing network and enhancing speciation capabilities is critical for evaluating the performance of control measures in the varied terrain and land uses in NYC.

[3] An exceedance is a violation of the pollutant levels permitted by environmental protection standards. (EPA, Terminology Reference System. http://www.epa.gov/trs/, accessed Jan. 18, 2007)

Figure 3.1. Locations of Ambient $PM_{2.5}$ Monitors in New York City

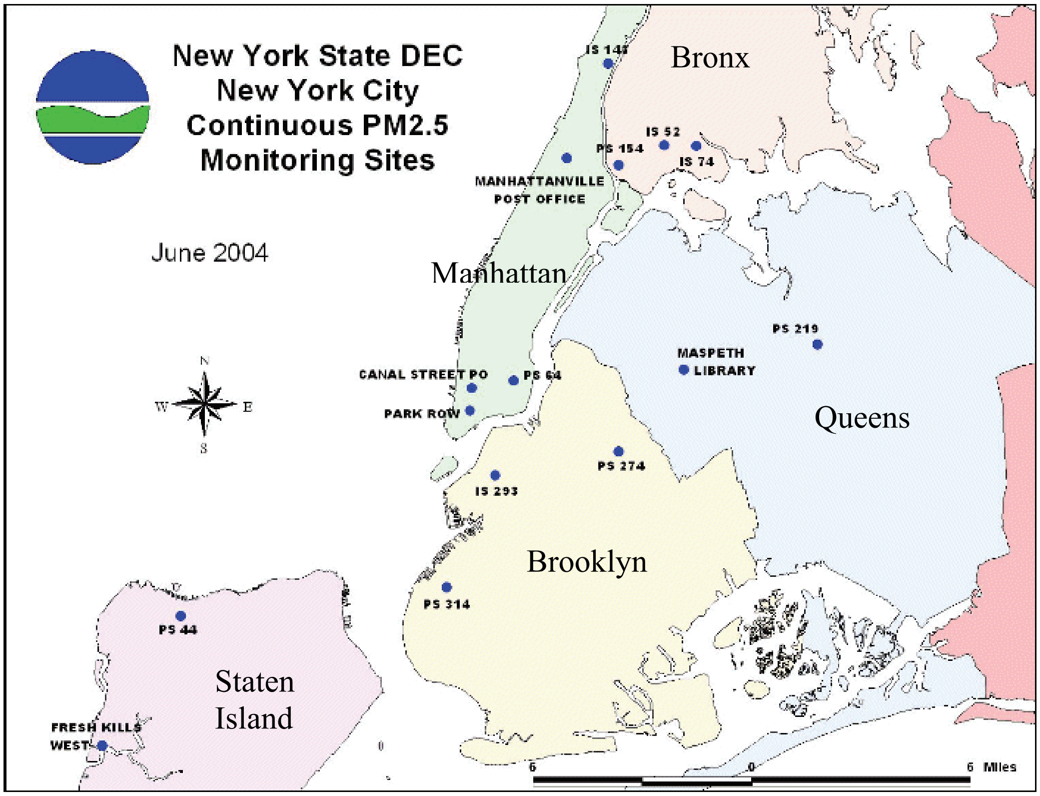
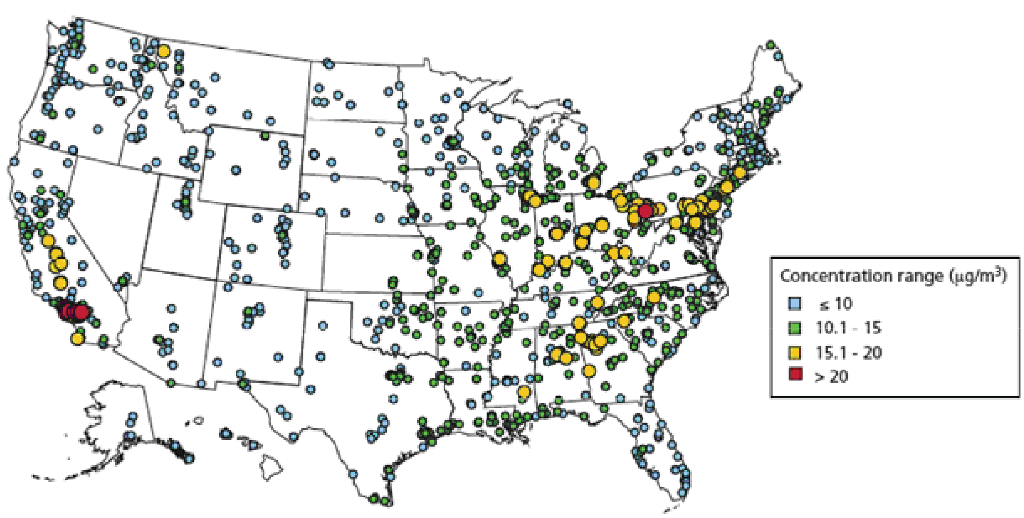


Figure 3.2. Map of Annual Average $PM_{2.5}$ Concentrations across the US in 2003



Source: The Particle Pollution Report: Current Understanding of Air Quality and Emissions through 2003, EPA 454-R-04-002, 2004, http://www.epa.gov/airtrends/pm.html.

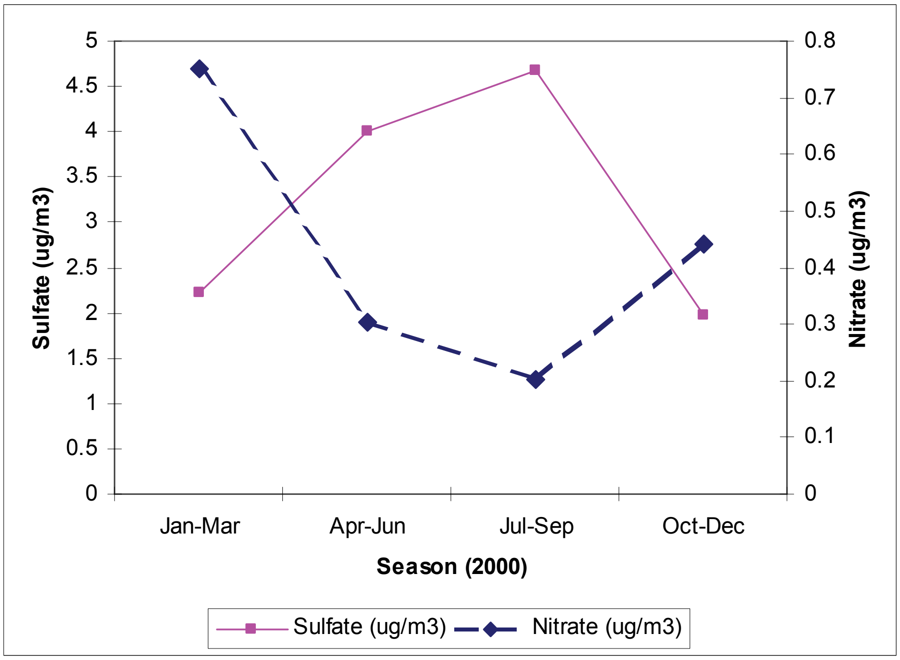
Table 3.1. Annual Mean $PM_{2.5}$ Concentrations in NYC, 2000-2004

Borough (County)	Year	Number of Stations	Annual Mean ($\mu g/m^3$)
Bronx (Bronx)	2000	3	**15.4**
	2001	3	**15.1**
	2002	3	14.8
	2003	3	14.6
	2004	3	14.3
Brooklyn (Kings)	2000	3	**15.9**
	2001	3	**15.5**
	2002	3	14.4
	2003	2	14.5
	2004	1	**15.0**
Manhattan (New York)	2000	4	**17.1**
	2001	5	**16.4**
	2002	4	**15.8**
	2003	4	**16.2**
	2004	4	**15.3**
Queens (Queens)	2000	3	13.7
	2001	3	14.0
	2002	2	13.3
	2003	2	13.0
	2004	1	12.8
Staten Island (Richmond)	2000	2	13.4
	2001	2	13.8
	2002	2	13.3
	2003	2	12.3
	2004	2	13.4

$PM_{2.5}$ is a measure of the mass of all airborne particles with aerodynamic diameters less than 2.5 micrometers. For purposes of comparison, a typical human hair is roughly 70 micrometers in diameter. $PM_{2.5}$ encompasses a wide range of individual particle sizes, many different chemical species, and contains both primary (directly emitted) and secondary (formed through reactions in the atmosphere) components from a wide variety of sources. To better understand the influence of particular source classes on $PM_{2.5}$ levels in NYC, it is useful to examine data about chemical sub-components of $PM_{2.5}$ that may relate to specific sources.

Two important components of $PM_{2.5}$ on a mass basis in NYC are secondary sulfate and nitrate particles. Although it depends somewhat on the setting, on average, sulfates and nitrates combined contribute about half of the ambient $PM_{2.5}$ on the East Coast. Sulfates are formed over time in the atmosphere when SO_2 gas reacts with ammonia gas. Far more SO_2 is emitted in the states upwind of NY than in NY State itself. Sulfate formation tends to be quicker on hot and humid days and slower on cold days or at night. Temperatures in the summer in New York are certainly warm enough for sulfate formation. Thus, concentrations of sulfates are highest in the summer, both because of atmospheric conditions and because power plants generate more electricity in the summer (in response to high air conditioning use). On the other hand, nitrates tend to form on colder days. Figure 3.3 provides seasonal sulfate and nitrate concentrations in Ulster, NY (taken from the EPA's CASTNET monitoring network). While this is not in NYC, sulfates and nitrates follow similar seasonal patterns across New York.

Figure 3.3. Seasonal Patterns of Sulfate and Nitrate Concentrations in New York



Another important contributor to particulate matter concentrations in NYC is elemental and organic carbon (on average, about a third of $PM_{2.5}$ concentrations in the East Coast). These pollutants do not display as strong a seasonal pattern as sulfates or nitrates, since they are mostly primary pollutants and are predominantly emitted by sources (like diesel and other motor vehicles) that emit year-round.

Despite these differences, one important commonality for fine particulate matter is its ability to travel long distances in the atmosphere. Although larger particles fall to the ground more quickly, fine particles ($PM_{2.5}$) can take days to reach the ground, especially when emitted from tall stacks. Sulfates, nitrates, and elemental and organic carbon are all smaller particles, and they can travel extremely long distances. This is why NOx and SO_2 emissions from power plants in the Midwest affect air quality in the East Coast.

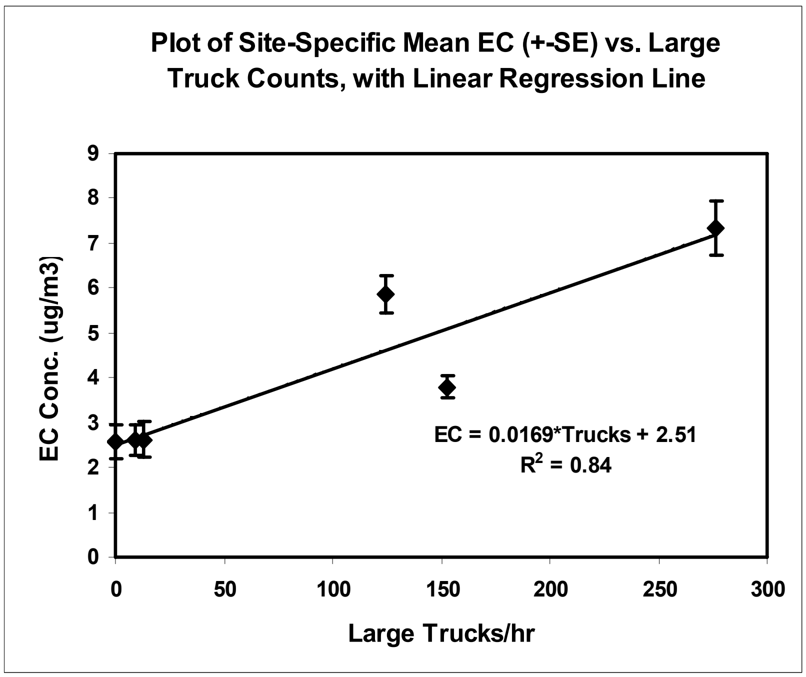
This raises an issue that we touch on briefly here and return to later in this document. Although particulate matter can travel a long distance, the maximum concentrations from a source are generally fairly close to the source, depending on local meteorological conditions (anywhere from less than a mile to tens of miles or more, depending on the height of emission and the type of particulate matter).[4] If the source emits mainly

[4] See: Levy, J. et al., *Using CALPUFF to Evaluate the Impacts of Power Plant Emissions in Illinois: Model Sensitivity and Implications*, Atmospheric Environment, (36) 2002 p. 1063-1075; and Levy, J. and J. Spengler, *Modeling the Benefits of Emission Controls in Massachusetts*, (52) 2002, p. 5-18.

primary particles (e.g., a major highway), this effect will be enhanced. If instead the source emits mainly precursor gases (e.g., a power plant), this effect will be reduced.

Recent research in NYC has investigated the relative impacts of local vs. regional sources for exposures of NYC residents, with a special emphasis on local motor vehicle impacts. One study in the Hunts Point section of the South Bronx examined the relationship between average sidewalk $PM_{2.5}$ and elemental carbon concentrations and the volume of heavy truck traffic on different streets.[5] Figure 3.4 shows that there was a strong correlation between airborne elemental carbon (on the vertical axis) and truck traffic. The correlation for $PM_{2.5}$ and truck traffic was not as pronounced, reflecting the fact that more of NYC's ambient $PM_{2.5}$ is regional in nature than its elemental carbon.

Figure 3.4. Relationship between Elemental Carbon (EC) Concentrations and Truck Traffic in the South Bronx.



Emissions from motor vehicles have been shown to be a major driver of spatial gradients in PM concentrations in NYC, especially for black carbon, a PM component for which motor vehicles are a dominant local source ((Kinney et al. 2000), (Lena et al. 2002)). Studies that have examined PM concentrations in relation to horizontal proximity to major roadways (e.g., (Zhu et al. 2002), (Hitchins et al. 2000)) provide useful insights on exposure patterns for cities where residential development patterns extend primarily in the horizontal direction, such as LA. However, there is an important vertical dimension to residential development in NYC, and vertical pollution profiles may influence population exposures and thus risks of adverse health effects. Whereas numerous studies

[5] Lena, TS. et. al., *Elemental Carbon and PM 2.5 Levels in an Urban Community Heavily Impacted by Truck Traffic*, Environmental Health Perspectives, October, 2002, 110(10), p. 1009-15.

have examined vertical patterns of pollution in and around urban street canyons in Europe and Asia, to date few if any studies have examined vertical PM profiles in US cities.

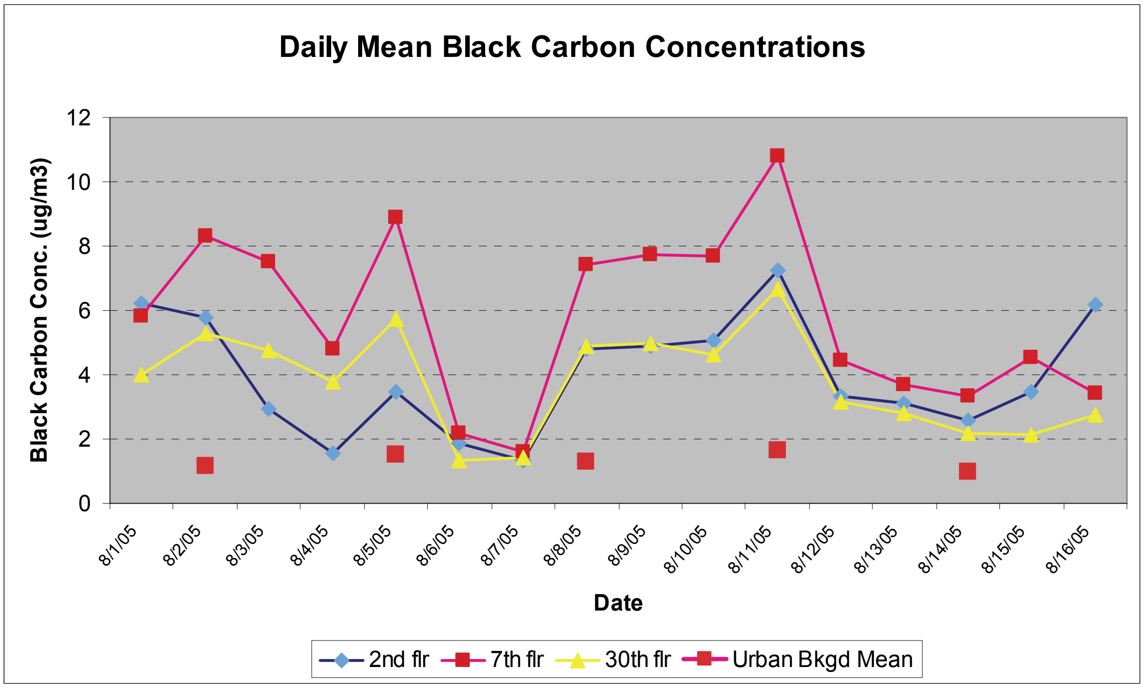
Given the gaps in the literature, we designed a pilot study to gather preliminary data on black carbon concentrations as a function of height in NYC. (A full description of this study can be found in Appendix A to this report.) NYC is unique among US cities in its population density and extent of vertical development. It also is a major crossroads for diesel traffic. The overall objective of the present study was to characterize the vertical profiles of black carbon concentrations above a major highway—interstate highway 95, which carries the highest volume of diesel traffic in the US—and to examine the influence of wind flow, traffic, and temporal patterns on these profiles.

The study set out the following aims:

1. Monitoring real-time outdoor black carbon concentrations over a two week period in August of 2005 from apartments located on the second, seventh, and 30th floor of a building that straddles interstate highway 95 in Northern Manhattan.
2. Examining hourly and daily black carbon concentration differences as a function of sampling height, and analyze whether wind flow or traffic counts have discernable impacts on spatial and/or temporal patterns.
3. Monitoring five-minute average real-time black carbon during the morning rush hour, at a point immediately-adjacent to the highway and comparing these concentrations with those measured on the floors above.

Daily mean black carbon concentrations measured at the three building floors are plotted over time in Figure 3.5. Also indicated are 24-hour mean elemental carbon concentrations averaged over three US EPA speciation monitoring sites in NYC (NY Botanical Garden, Queens College, and Canal Street sites). These sites provide and indication of urban background concentrations at central sites, though the different monitoring method (thermal analysis of quartz fiber filter samples).

Figure 3.5. Mean Black Carbon Concentrations on the 2nd, 7th, and 30th Floors



Interestingly, the 7th floor readings were almost always higher, and often significantly higher, than both the readings taken at the other floors and the mean background concentration. There was considerable inter-site variability, although the 2nd floor concentrations were similar to those measured on the 30th floor on most days. Concentrations were higher and more variable on weekdays than on weekends (August 6, 7, 13, and 14). Weekday black carbon concentrations were strongly influenced by wind patterns, which alternated between generally westerly and generally easterly during the sampling. Diurnal measurements demonstrated more typical vertical gradients and suggested that patterns vary over relatively short time scales.

The complex vertical black carbon profile observed in this study may reflect the combined influence of diesel emissions both on the highway and the surrounding area, as well as to the unique geometry and wind patterns at the sampled building. The seventh floor sample was at or above the average height of surrounding buildings and thus sampled from the air layer into which all surrounding street canyons empty. The impact of highway emissions was not clearly discernable at the second floor, which was set back from the highway by a wide shelf. However, short-term "grab" samples at ground level near the road demonstrated highly elevated concentrations of black carbon.

The shape and placement of structures on a specific roadway can lead to complex and sometimes counterintuitive patterns of dispersion. However, these intra-urban patterns of PM concentrations are not well-captured by existing compliance monitoring networks. Monitoring conducted by the NY DEC is specifically meant to avoid local impacts of traffic in order to maximize the data's representativeness of general population exposures in a community; hence, monitors in general (especially in residential neighborhoods) are located on the tops of buildings. Rooftop monitoring does not, however, indicate the distribution of population exposures at apartments located within street canyons, where people actually live in NYC. Systematic monitoring to assess actual exposures of NYC residents to $PM_{2.5}$, based on ambient sampling from residential locations and on sidewalks, is needed to determine whether New Yorkers are being exposed to unhealthy $PM_{2.5}$ levels.

The influence of traffic proximity on population exposures to $PM_{2.5}$ is a very important issue for which few data currently exist in US cities. Proximity has both horizontal and vertical dimension in cities. To-date, zoning codes and individual siting decisions for homes and schools have not taken explicit account of the impacts of traffic on air quality. There is a need for the development of planning and zoning guidelines to address this issue.

While understanding the magnitude and composition of outdoor levels of $PM_{2.5}$ is helpful, ultimately, health effects are most closely related to actual personal exposures. As mentioned above, personal exposures may differ from ambient concentrations, depending on where people spend their time. Personal exposures studies in New York City provide some insight about the potential effects of outdoor air pollution.

In one study, personal exposures to $PM_{2.5}$ and its components were assessed in a group of 46 high school students from the A. Philip Randolph Academy, a public high school located in the West Central Harlem section of NYC.[6] Each of two field campaigns (winter and summer, 1999) involved eight weeks of fixed-site ambient monitoring on a school roof and on a roof at the Lamont Doherty Earth Observatory (LDEO) in Palisades, NYC. Since the predominant winds are from the west, the LDEO rooftop usually reflects the upwind, regional air masses. The school roof site measures both regional and local pollution. In addition, five home outdoor sites were monitored in each of 8 weeks each season. In addition to temporary monitors in their homes, students also carried personal monitoring backpacks equipped to measure the same pollutants being measured outdoors at home, the school, and the upwind site.

Drawing on the data from the exposure study in West Harlem, we assessed the overall relationships between personal and outdoor PM concentrations by examining scatterplots of personal against home outdoor levels of $PM_{2.5}$, sulfate, and black carbon (Figures 3.6, 3.7 and 3.8 on the following pages). On each plot, we have drawn the 45-degree line that would represent perfect correlation between the two variables. The relationships clearly differ by particle constituent. For $PM_{2.5}$, personal and outdoor associations were less evident, with the magnitude of the correlations diminished by occasional high personal exposures that had no ambient counterpart. Personal exposures generally exceeded outdoor levels, and the correlation with outdoor concentrations was non-significant. These observations are consistent with numerous studies that have shown that personal

[6] Kinney, P.L., Chillrud, S.N., Ramstrom, S., and Spengler, J.D. Exposures to multiple air toxics in New York City. Environ. Health Perspect. 110(suppl 4):539-546 (2002).

exposures to $PM_{2.5}$ often exceed those measured outdoors due both to the "personal cloud" and to the impact of indoor sources not captured by ambient monitoring.[7]

For sulfate, personal/outdoor correlations were more evident. Aside from one high personal outlier, the relationship in winter appeared linear, where concentrations were generally below 5 $\mu g/m^3$. A significant and apparently linear personal and outdoor correlation was observed for black carbon. The stronger correlations observed for black carbon as compared with $PM_{2.5}$ reflects the fact that, except perhaps for candles and incense burning, generally few indoor sources of black carbon can be found. Furthermore, because black carbon is present in very small particles, these apparently penetrate indoors quite freely.

The scatterplots of personal versus outdoor concentrations of particle constituents (below) incorporate both temporal and spatial correlations since they include multiple days and multiple subjects each day. To better understand the source of the observed correlations, it is of interest to separate the correlations related to temporal variations from those related to spatial variations. To date, most evaluations of personal and outdoor correlations have focused on temporal factors, which is relevant to interpreting time-series epidemiology results. Here we examine both time and space. Analyses separating the spatial and temporal components suggest that temporal associations vary across seasons and particle constituents, with the strongest association for sulfate particles and the weakest association for $PM_{2.5}$, based on the absence/presence of indoor sources. In contrast, elemental carbon was the only constituent to show significant spatial correlation, which was stronger in the winter than the summer due to reduced atmospheric mixing.

[7] The "personal cloud" phenomena of relatively high concentrations near an individual can arise when movement stirs up gases and particles on clothing and nearby surfaces, increasing particle concentrations. Jacobson M., *Atmospheric Pollution*, Cambridge University Press, 2002, p. 142.

Figure 3.6. Scatterplot of Personal versus Home Outdoor Levels of Fine Particulate Matter (μg/m³)

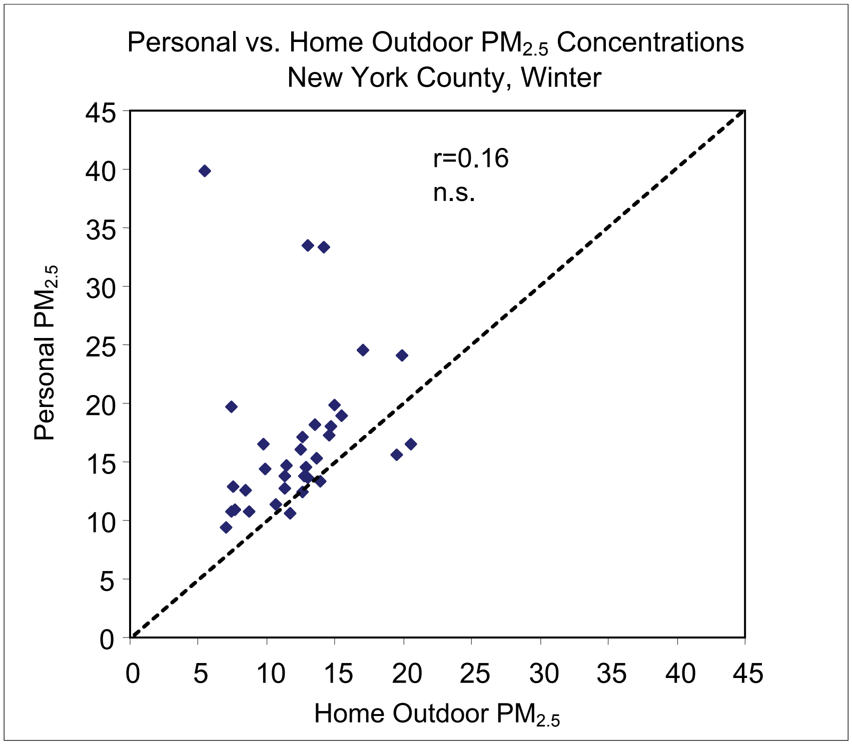


Figure 3.7. Scatterplot of Personal versus Home Outdoor Levels of Sulfate Particles (μg/m³)

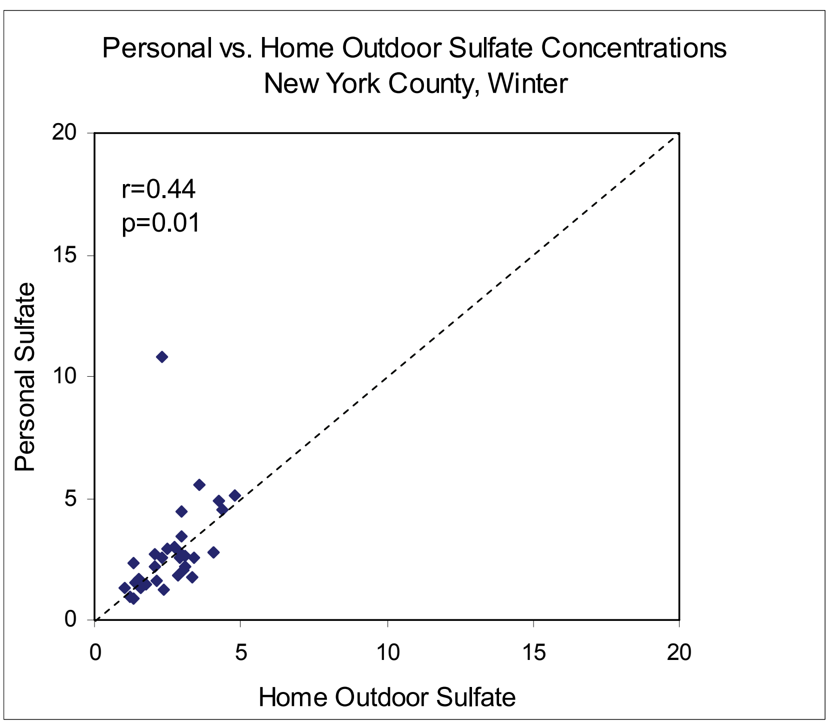
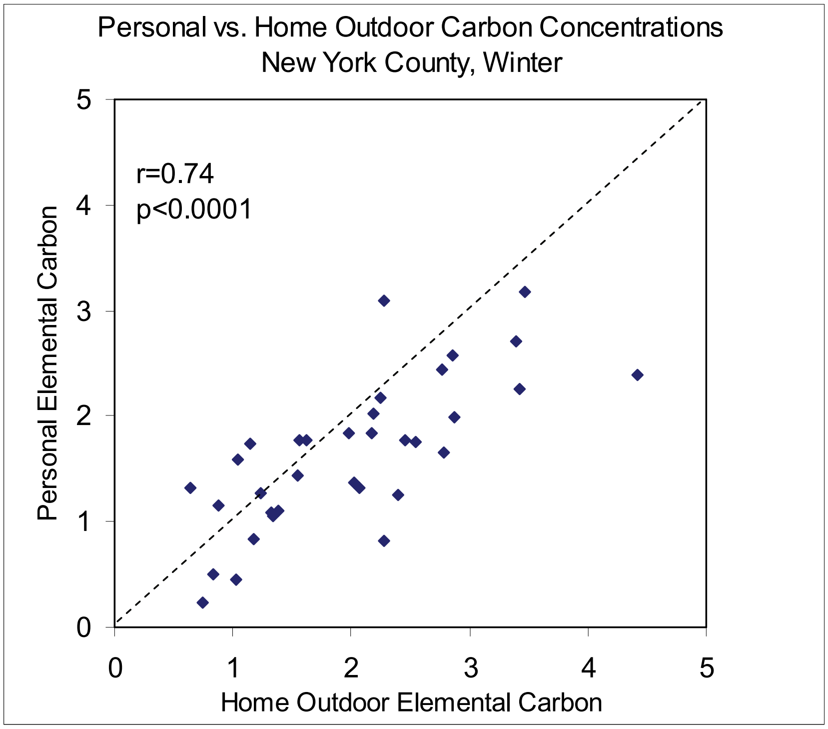


Figure 3.8. Scatterplot of Personal versus Home Outdoor Levels of Elemental Carbon (μg/m³)



In conclusion, we can gain the following insights from ambient concentration and personal exposure data collected in New York City:

- Fine particulate matter concentrations in NYC range between approximately 13-17 μg/m³, depending on the monitor, borough, and year. These concentrations are close to, and at times greater than, the annual average National Ambient Air Quality Standard for $PM_{2.5}$.
- Particulate matter can include numerous constituents, which come from different sources in different geographic locations. The primary constituents of $PM_{2.5}$ in NYC are sulfate and nitrate particles (driven by secondary formation) and elemental and organic carbon (driven by primary emissions).
- Detailed personal exposure studies in NYC demonstrate that people are exposed indoors to many particles of outdoor origin, that secondary particles (like sulfate) are relatively uniform spatially, and that primary particles of outdoor origin (like elemental carbon) have both spatial and temporal heterogeneity.

3.2 The Sources of NYC's Ambient $PM_{2.5}$

There are many ways that one can approach the question of source apportionment (that is, determining the relative contribution of various source categories in various geographic locations to ambient concentrations in NYC). As mentioned above, understanding the contributions of sources to concentrations in NYC only partly informs the quantification of the health benefits of source controls, but it directly informs attainment analyses and provides a useful context for health benefits analyses.

One method of source apportionment is to collect physical $PM_{2.5}$ samples within NYC, conduct detailed speciation analyses on particle filters to determine the elemental composition, and use statistical methods and known source emission profiles to identify the various contributors. The advantage of this approach is that it is based on actual pollution measurements. However, it cannot provide information about where the sources are located. In other words, this form of source apportionment work can tell you that 30 percent of particulate matter in NYC comes from power plants, but not necessarily how much of this comes from plants in New York versus plants elsewhere. Moreover, it requires fairly intensive statistical analysis and data that are not generally publicly available.

Another approach to apportionment is to use air pollution dispersion modeling, in which emissions from all sources are quantified, and computer models characterize the fate and transport of these pollution sources to receptor locations (in this case, sites in NYC). While this has the advantage of providing geographically resolved information, as well as providing data that could be used directly in risk assessments (including concentrations across a number of geographic sites), there are obvious limitations in developing emission inventories and atmospheric dispersion models with sufficient accuracy.

While the monitoring approaches described above were beyond the scope of this report, we were able to address the question of source apportionment using a simplified dispersion model called the source-receptor (S-R) matrix. The S-R matrix has been previously applied in power plant pollution exposure analyses, as well as in regulatory impact analyses of mobile source control strategies.[8] Briefly, the S-R matrix involves county-to-county transfer factors across the US for both primarily-emitted particles and secondarily-formed particles (i.e., sulfate and nitrate particles from SO_2 and NOx emissions). It is based on an adjusted version of the Climatological Regional Dispersion Model, which incorporates basic meteorological data from 1990 into a sector-averaged dispersion model across 16 wind directions. The model includes wet and dry deposition as well as chemical conversion to address sulfate and nitrate formation, and contains a step to calibrate initial model outputs to ambient concentration monitoring data. Previous applications have shown that health impact estimates are similar using S-R matrix and more complex dispersion models.[9]

The S-R matrix apportions ambient $PM_{2.5}$ to four source categories: power plants, on-road vehicles, airports and other mobile/area sources. This is different from the typical breakout in emissions inventories, in which non-road and area sources are treated

[8] Levy et al., Es*timation of primary and secondary particulate matter intake fractions for power plants in Georgia,* Environmental Science and Technology, December, 2003, 15;37(24) p. 5528-36.

[9] Abt Associates, ICF Consulting and E.H. Pechan Associates, The Particulate-Related Health Benefits of Reducing Power Plant Emissions, http://www.cleartheair.org/fact/mortality/mortalityabt.pdf, October 2000; US Environmental Protection Agency, Regulatory Impact Analysis - Control of Air Pollution from New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements. EPA420-R-99-023, December 1999; Levy et al., 2003.

separately. For power plants, not all plants in the US are available in the emissions inventory, but enough major facilities are represented for our comparative analysis to be interpretable. The inclusion of airports is based on the obvious interest within NYC of the relative contribution of LaGuardia and Kennedy Airports, as well as other smaller airports, on air quality within NYC. This category includes airport services (both gasoline and diesel) as well as aircraft emissions. "Other mobile/area sources" consists of a number of source categories, including construction vehicles, residential, industrial and commercial area sources, and railroads.

The S-R model has several weaknesses when applied to a small geographic area such as NYC. First, the model represents the emissions from power plants using a subset of the actual plants in region. In the NYC area, the model includes: Danskammer, Astoria,[10] Ravenswood, Arthur Kill, Bowline Point, and Lovett. Since not all plants in the NYC area are represented, the model probably understates the contribution of power plants in this region. Second, the emission inventory within the model was developed in the mid 1990s and is based on national-level data, not state-specific inventories such as the NY DEC inventory we present in Section 4. In particular, we found that the inventory within the S-R model contains much higher emissions from the area source sector than the NY DEC inventory. Thus, we expect the S-R model to overestimate the contribution of area sources. In light of these weaknesses, the goal of this modeling is not to provide precise quantitative estimates of source contributions, but to provide some insight about the source categories and geographic locations that may have greater or lesser contributions to ambient levels in NYC. More work is needed using more refined tools to better understand the mix of sources that produce NYC's ambient $PM_{2.5}$.

The results of our S-R modeling can be summarized as follows.

- Nearly half of the ambient $PM_{2.5}$ in NYC comes from non-road mobile sources and area sources, and a majority of this comes from sources within NYC.
- Roughly a quarter of the ambient $PM_{2.5}$ in NYC comes from power plants, and a majority of this comes from plants upwind of the City.
- Just under a quarter of NYC's $PM_{2.5}$ comes from on-road mobile sources, and most of this is from sources in NYC.
- The airports in NYC contribute a small fraction of ambient $PM_{2.5}$, in the range of one percent.

These results are consistent with several widely accepted theories about sources of NYC's $PM_{2.5}$. First, non-road sources and area sources are a very large portion of ambient PM in most cities. This is because "off-road" diesel engines, such as construction equipment, have some of the highest $PM_{2.5}$ emission rates. These sources tend to operate for extended periods in one location unlike, for example, diesel trucks on highways and have limited dispersion rates compared to the other source categories, so we would expect engines upwind of NYC to have a much smaller effect on the City's air

[10] Excludes Astoria unit 6, generally referred to as Poletti.

quality than engines in the City. Area sources too (such as residential heating systems) emit close to the ground and thus have more limited dispersion rates than large industrial facilities. As noted, however, we believe that the emission inventory within the S-R model overstates area source emissions and that these model runs probably overstate emissions from the non-road/areas source sector.

Second, the majority of NYC's ambient $PM_{2.5}$ from power plants comes from plants outside of NYC – outside of New York State in fact – as compared with the plants in NYC. This is because much of the $PM_{2.5}$ from power plants is secondary $PM_{2.5}$, and the impacts of secondary $PM_{2.5}$ tend to occur farther from the source than primary $PM_{2.5}$. In addition, power plants' tall smokestacks and the high temperature of the pollutants when emitted facilitate the disbursal of secondary $PM_{2.5}$ over long distances. As discussed below, however, the fact that much of NYC's $PM_{2.5}$ blows in from power plants in upwind states does not mean we should ignore the emissions from NYC's plants.

Third, local motor vehicles make a substantial contribution to $PM_{2.5}$ concentrations in NYC, as compared with motor vehicles elsewhere in New York State or outside of the state. We believe this is true because of the large amount of motor vehicle emissions in NYC and because these emissions occur at ground level. Further, because emissions from upwind vehicles occur at ground level as well, they do not travel as far as emissions from upwind power plants. Similarly, for airports, sources within NYC make a measurable contribution to ambient $PM_{2.5}$, with a relatively small contribution from airports elsewhere (both because of proximity and stack height).

When extrapolating health effects from source emissions it is important to factor in the location of the affected population. That is, the population distribution around a source has an important impact on the magnitude and distribution of the source's health effects. Thus, even though much of the ambient $PM_{2.5}$ in NYC comes from power plants upwind of the City, the City's plants do have significant health impacts because they are located in densely populated areas. For example, the S-R matrix outputs suggest that 44 percent of the public health impacts from the Astoria power plant[11] occur within NYC. This is among the highest percentages seen for power plants, driven by the population density in Queens, where the Astoria power plant is located.[12] Thus emission reductions at locations like Astoria would provide significant benefits to NYC residents as well as residents of downwind areas – Long Island and New England.

The same population proximity dynamics apply to emissions from other sources. Traffic in Manhattan, for example, is likely to pose greater health threats than traffic in less densely populated areas of the City. As discussed in Section 7, both emission sources and population distribution within NYC must be considered in determining optimal control strategies.

[11] Excludes Astoria unit 6, generally referred to as Poletti.

[12] This is similar to the findings of a previous study evaluating power plants in Illinois (Levy et al., 2002). This study found that power plants in rural areas have most of their health impacts further away from the source, while sources in urban areas have most of their impacts nearby, due to the local population density.

More research will help to clarify the mix of sources that produce NYC's ambient $PM_{2.5}$. NYSERDA is currently sponsoring a number of studies to address this research need for fine particulate sources in NYC and New York State. A study being conducted by the NYU Medical School focuses on $PM_{2.5}$ source apportionment. This project will examine air monitoring data collected in 2001 at rural and urban locations in downstate NY to study the relative contribution of local and regional sources of fine particulate matter. Data from the rural site, located in Orange County approximately 35 miles northwest of NYC, is being used as reference data, indicating regional $PM_{2.5}$ levels. Preliminary findings indicate that virtually all of the ambient elemental carbon in NYC can be attributed to local combustion sources. Another project, at Clarkson University, is examining new methods of analyzing $PM_{2.5}$ data. Specifically, advanced Positive Matrix Factorization (PMF) models are being applied to multiple $PM_{2.5}$ databases.[13]

[13] For more information on this and other NYSERDA-sponsored projects, see: www.nyserda.org/programs/environment/emep.asp.

4. Inventory Data on $PM_{2.5}$ Emissions in NYC

The New York Department of Environmental Conservation (DEC) divides emission sources into the following four categories, established by the US EPA. We use these categories throughout this report to discuss $PM_{2.5}$ emissions in NYC.

- **On-road mobile sources**. This category includes all transportation vehicles that travel on roads, including cars, taxis, buses, trucks, etc., using both diesel fuel and gasoline.
- **Off-road mobile sources**. This category includes all transportation vehicles that travel off-roads, such as airplanes, marine transportation, recreational vehicles, industrial, construction and mining equipment, using both diesel fuel and gasoline.
- **Point sources**. Sometimes referred to as point sources, point sources are fixed sites that emit at least 100 tons per year of any regulated pollutant. This category includes electric power plants, manufacturing, refineries, and steel mills.
- **Area sources**. This category includes all smaller stationary sources of emissions, such as residential and commercial furnaces, waste incinerators, small distributed generation (DG) and combined heat and power (CHP) applications, and miscellaneous other small combustion sources. This category also includes non-combustion area sources such as gas stations and dry cleaners.

The most detailed and current estimate of primary $PM_{2.5}$ emissions in NYC is an emissions "inventory" prepared periodically by the NY DEC. We obtained from DEC the most recent inventory, estimating emissions for the year 2002. Using this inventory as a starting point, we estimated current (2005) direct $PM_{2.5}$ emissions in NYC. As discussed in the Sections below, we have made significant adjustments to the DEC's 2002 inventory, based on our understanding of the inventory methodology and our own research.[1]

4.1 On-Road Mobile Sources

Our estimate of 2005 $PM_{2.5}$ emissions from on-road vehicles draws on analysis by the DEC, which in turn is based on US EPA's definitive computer emissions inventory model, MOBILE6.2, used for estimating regional and national vehicular emissions.

MOBILE6.2 is the current edition of MOBILE6, a computer model developed and refined over many years by EPA to estimate emission factors for a wide variety of pollutants, including hydrocarbons, carbon monoxide, nitrogen oxides, sulfur dioxide and particulate matter. These and other factors are calculated in MOBILE6 for 28 individual vehicle classes, ranging from conventional sedans and sport-utility vehicles to over a dozen weight-classes of gasoline- and diesel-fueled trucks, along with several

[1] The DEC has not yet released formal documentation of the inventory methods.

classifications of buses. The emission factors also vary with conditions such as ambient temperatures, altitude above sea level, travel speeds, the vehicle's accumulated mileage, and the vehicle model year.

Because this study concerns fine particulates only, we limited our analysis to emissions of direct $PM_{2.5}$. We also had the benefit of extensive work with MOBILE6.2 by DEC, which has specified "input variables" for environmental, vehicular and travel conditions specific to each of the five boroughs of NYC in 2005 (as well as for all other counties in New York State). DEC has also estimated total miles driven by all vehicles as well as the share of those miles attributable to each of the 28 vehicle classes, on each of the six major "roadway types." (These are expressways, arterial roads, and local roads, along with variants of each, for which typical driving conditions vary sufficiently to warrant separate emission factors.) Note that while the most recent on-road inventory released by the DEC is for 2002, the agency has developed input files for 2005, and we used these inputs for our 2005 estimate.

For the most part, we were able to estimate 2005 emissions of $PM_{2.5}$ for each borough by multiplying each emission factor times total miles driven, times that same vehicle class's percent share of the miles, and then summing the products across all roadway types and all vehicles.[2]

There were two important exceptions to this procedure. First, the 2005 emission factor for diesel buses generated by MOBILE6.2 did not reflect the considerable progress made to date by the NYC Transit Authority in reducing emissions from its bus fleet. Since these buses account for approximately two-thirds of all VMT by diesel buses in NYC, and since the unadjusted diesel bus emission factor for $PM_{2.5}$ is very large compared to that for all other vehicle classes, it was important to make an adjustment to better reflect today's bus emission rates.

The Transit Authority began using ultra-low-sulfur diesel fuel in September, 2000. It has also retrofitted all of its diesel engines with either less-polluting engines or particulate filters. To account for the effect of these programs, we assumed that the combination of particulate filters and ultra-low-sulfur diesel fuel reduces bus emissions by 90 percent from prior, uncontrolled levels, while ultra-low-sulfur fuel alone reduces bus emissions by five percent.[3] In addition, both the Transit Authority and private bus companies subsidized by New York City DOT have converted several hundred diesel buses to compressed natural gas. As recommended by EPA, we assumed that particulate emissions from those buses would be the same as emissions from gasoline-powered buses.[4]

[2] An example of an emission factor is grams emitted per mile of expressway driving by Heavy-Duty Diesel Vehicles weighing between 26,001 and 33,000 pounds.

[3] Thomas Lanni, et al., "Performance and Durability Evaluation of Continuously Regenerating Particulate Filters on Diesel Powered Urban Transit Buses at NY City Transit," Society of Automotive Engineers, paper 2001-01-0511.

[4] U.S. EPA, "MOBILE6.1 Particulate Emission Factor Model Technical Description: Final Report," EPA 420-R-03-001, January, 2003, Section 2.5.

The other shortcoming in MOBILE6.2 was its failure to account for increased $PM_{2.5}$ emissions at low speed, where start-stop driving means a lot of idling and acceleration, both of which increase emissions per mile. EPA is studying options to account for the speed-dependence of $PM_{2.5}$ emissions, which will probably lead to the incorporation of additional correction factors in future versions of the MOBILE model. In the meantime, for this analysis we adjusted the emission rate at lower speeds by developing an equation to connect the MOBILE6.2 emission factor at idle (assumed to represent emissions when driving at 2.5 mph) to the MOBILE6.2 emission factor at 19.6 mph. EPA used a similar procedure in MOBILE5.

Including these adjustments, total emissions from operation of on-road vehicles in New York City in 2005 are estimated to be 718 tons. A breakout of emissions by vehicle class is shown in Table 4.1. Queens was the borough with the greatest emissions, 34 percent of the citywide total, followed by Brooklyn, Manhattan, the Bronx and Staten Island.

Table 4.1. 2005 On-Road Emissions in NYC

Vehicle Class	$PM_{2.5}$ (tons)	% of On-road Emissions	% of VMT
LDGV (conventional sedans)	203	28%	60.7%
URB BUS (diesel transit buses)	122	17%	0.7%
HDDV8B (diesel 18-wheeler trucks)	85	12%	0.9%
LDGT2 (“light trucks” including SUV’s)	72	10%	20.2%
24 other vehicle categories combined	237	33%	17.5%
TOTAL	718	100%	100%

Our estimate of 2005 on-road emissions is 16 percent below the DEC’s estimate of 2002 emissions (854 tons). This difference is almost entirely due to the two adjustments we made. (The effect of using 2005 rather than 2002 model inputs was small by comparison).

4.2 Non-Road Mobile Sources

Non-road emissions come from a broad class of vehicles and equipment that use internal combustion engines. The EPA divides this diverse class into the nine categories shown in Table 4.2.

Table 4.2. Non-Road Equipment Categories

Non-Road Category	Examples
Agricultural equipment	Tractors
Aircraft & airport equipment	Aircraft and maintenance vehicles
Commercial equipment	Generators, welders, air compressors, pumps
Construction equipment	Tractors, backhoes, dozers
Industrial equipment	Refrigeration units, forklifts, tractors
Lawn and garden equipment	Chain saws, leaf blowers, mowers
Logging equipment	Shredders, chain saws
Marine vessels	Ships, pleasure craft
Rail road equipment	Locomotives, maintenance vehicles
Off-road recreational vehicles	ATVs, off-road motorcycles

Non-road engines are internal combustion engines in equipment, other than motor vehicles, designed to move or that are moved from one place to another within a twelve-month period.[5] Hydro-power units, irrigation sets, generator sets, pumps, compressors, and welders consist of a mixture of mobile and stationary engines. For those categories, EPA estimates the fraction of engines that are mobile based on engine size. Based on these estimates, smaller engines are more likely to be classified as mobile than larger ones. For example, 90 percent of engines with less than 40 hp were defined as mobile, while only 10 percent of engines between 300 and 600 hp (and none above 600 hp) were defined as mobile.

The NY DEC uses EPA's NONROAD model to estimate emissions from non-road sources. The NONROAD model contains a national database of non-road equipment.[6] Generally, emission estimates are derived from a chain of collected data, including national engine populations, engine activity level and load factor for each source classification, and emissions by engine type and fuel. Engines in each source classification are allocated to the county level using surrogates, such as the proportion of employment in a relevant industry or the proportion of human population in the county to the national total.

This methodology encompasses a great deal of uncertainty. The actual number of non-road engines in a county may differ from the number allocated by the model due to differences between the county's mix of businesses within a certain category and the national average, or due to differences in equipment age or activity level, or due to a number of other factors.

NYC's economy has long had characteristics much different than the national average. For example, manufacturing in NYC has been unusually dominated by employment in the garment industry, although that dominance has waned in recent decades. Garment manufacturing involves processes that are vastly different from the average manufacturing business. The amount of direct fuel combustion per worker in the garment industry is much lower than in other manufacturing business, and thus, EPA's method of allocating non-road manufacturing equipment to NYC (on the basis of its total number of manufacturing workers) likely overestimates NYC emissions. Thus, in estimating 2005 emissions, we have removed employment in the apparel industry from the allocation formula within the NONROAD model.

We also believe that the NONROAD model overstates emissions from construction in NYC. First, the model allocates non-road construction equipment to counties on the basis of the dollar value of construction. This would overestimate equipment per worker in areas such as NYC wage rates well above the national average. To account for this problem, we reduced NYC construction spending in the model by the ratio of New York

[5] See 40 CFR 89.2.

[6] Note that the NONROAD model does not include commercial marine, locomotive, and aircraft emissions, so these are not included in the inventory. These emissions can be calculated separately using other models.

construction costs to a 20-city average.[7] Second, the model uses different weights for the dollar value of construction for single-family homes, other buildings, road and bridge, and public works. While this breakout might be reasonable in most areas of the country, it may not adequately reflect the large amount of high-rise construction in NYC. We believe that in NYC a smaller fraction of project cost goes into site preparation, which requires a high proportion of the heavy construction. We were not able to adjust for this potential problem in the model.

In addition, the model allocates off-road recreation vehicles to counties based on the number of "camps and recreational vehicle park establishments" in each county. While this allocation factor may appear reasonable, it is unlikely that camps and recreational establishments in NYC include significant off-road vehicle use. (Eleven "camps and recreational vehicle park establishments" exist in Manhattan, the same number as in Suffolk County, for example). To account for this problem, we reduced the number of off-road recreational-vehicle emissions allocated to Manhattan by 90 percent.

Finally, there is likely to be less engine-powered lawn and garden equipment in NYC than in the rest of the nation, because one and two-family brownstones, a predominant housing type in many areas of New York City, generally have no lawn. (Also, in multi-family housing units, the families share one lawn.) To reflect this aspect of NYC we changed the allocation factor for lawn and garden equipment to the number of one- and two-family buildings, instead of the number of dwelling units.

Perhaps other adjustments should be made as well. Manufacturing employment has continued to decline in NYC, while data in NONROAD is about five years old. On the other hand, it is not clear whether use of non-road equipment in the manufacturing sector has declined in the same way as employment, or at all.

Table 4.3 shows our estimate of 2005 $PM_{2.5}$ emissions from off-road equipment, including the effects of the adjustments discussed above. The DEC's 2002 inventory is shown in the bottom row of the table for comparison. Our 2005 estimate is 27 percent lower than the DEC's 2002 estimate.

[7] EPA is considering such a correction for future versions of NONROAD.

Table 4.3. 2005 Non-Road $PM_{2.5}$ Emissions (Tons per Year) by Borough*

Classification	BX	BK	MH	QN	SI	Total
Agricultural Equipment	0	0	0	0	0	0
Airport Equipment	0	0	9	42	0	51
Commercial Equipment	20	75	288	72	10	464
Construction Equipment	86	134	431	308	52	1,010
Industrial Equipment	24	52	42	51	7	176
Lawn & Garden Equip. (Com.)	7	13	17	8	13	58
Lawn & Garden Equip. (Res.)	5	17	1	23	7	52
Logging Equipment	0	0	0	0	0	0
Pleasure Craft	3	6	1	15	9	34
Railroad Equipment	2	4	3	3	1	13
Recreational Equipment	1	4	0	3	1	9
Total	148	303	791	524	99	1,865
DEC 2002 Inventory	203	383	1,096	730	132	2,543

*BX: Bronx; BK: Brooklyn; MH: Manhattan; QN: Queens; SI: Staten Island

In our estimate, 88 percent of non-road mobile-source emissions in NYC come from just three categories – commercial equipment, construction equipment, and industrial equipment. More than half of total non-road emissions come from construction equipment alone. A very large share of the city's non-road emissions – over 40 percent – are concentrated in Manhattan, where non-road emissions per square mile are seven times higher than in any other borough. This concentration is mainly due to the large proportion of both construction and commercial activity that occurs in Manhattan.

Construction and commercial activity are broad categories, but industrial activity can be divided into two parts. The largest part, accounting for 60 percent of industrial emissions, consists of emissions from mobile refrigeration equipment, typically mounted on trucks. Although some refrigeration equipment for smaller and mid-size trucks (less than 20 feet in length) can be powered by the truck engine, most refrigeration units for mid-size and large trucks include a separate diesel engine fueled from the truck's fuel tank.

4.3 Point Sources

Point sources report annual emissions and fuel use to the DEC, as required by their operating permits.[8] Many of the largest point sources are required to operate a Continuous Emissions Monitoring system and to confirm the quality of the hourly emissions data obtained from that system. Accordingly, we consider the data reported to the DEC to be reliable and have not adjusted them.

The DEC's inventory of point sources includes 117 facilities. Five of these facilities are in Staten Island; 36 are in Brooklyn; 33 are in Manhattan; 25 are in Queens; and 18 are in the Bronx. These large sources make up 17 percent of our revised direct $PM_{2.5}$ inventory.

[8]States also report data on all point sources to the EPA for the National Emission Inventory (NEI) (http://www.epa.gov/ttn/chief/nif/index.html#nof).

Figure 4.1 shows our estimate of 2005 $PM_{2.5}$ emissions from point sources in each of the five boroughs.

Facilities in the point source category can be broken down into six sectors: Large Energy, Residential, Landfill, Healthcare, Commercial/Industrial, Wastewater Treatment, Education, and Other. Steam and electricity plants make up the vast majority of large energy facility sources. Residential facilities tend to be large apartment complexes with central boilers or furnaces for space heating and hot water. The Staten Island (Fresh Kills) landfill is the only landfill in the five boroughs represented in the DEC point-source data. The sites in the Healthcare category are predominantly hospitals. A wide range of industries are represented in the Commercial/Industrial sector; the largest contributors in this category include a waste-paper recycling plant, a batch asphalt plant, and a sugar processing facility. Education consists of university facilities, including Columbia University and Albert Einstein College of Medicine, among others. Sources that do not fall into the above-mentioned sectors, such as Transit Authority facilities, the Riker's Island steam plant, and the Bronx Zoo, are included in the Other category.[9]

Figure 4.1. Estimated 2005 NYC Point Source Emissions, by Borough

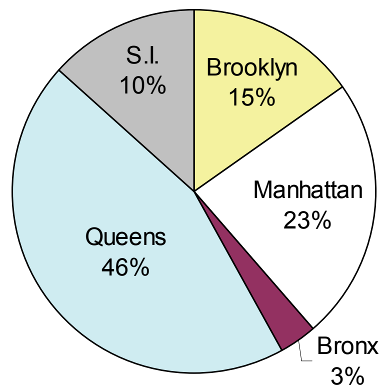


Table 4.4 shows our estimate of 2005 $PM_{2.5}$ emissions from point sources, as well as numbers from the DEC's 2002 inventory. For the large power plants we have projected utilization using the PROSYM power system production costing software. The projected decline in emissions from energy sources between 2002 and 2005 is the result of the new generating units at Poletti, Ravenswood and East River. The output of these units is projected to reduce the operation of several older, higher emitting units (primarily the old Poletti unit and the Astoria units). For large steam plants, we grew annual emissions by 0.5 percent per year, based on Consolidated Edison's plans for modest expansion of its steam load.

Our assumed growth rates for other sectors are as follows.

- For residential sources, we grew the DEC's 2002 numbers using the average growth in the number of households in NYC (0.7 percent per year).
- We assumed zero growth in emissions from the Staten Island Landfill. As of the most recent Title V operating air permit, only 771 of about 2,200 acres were

[9] The Bronx Zoo no longer has a Title V permit, which may indicate that $PM_{2.5}$ emissions from this facility have been reduced.

actively accepting waste.[10] As a result, fine particulate emissions have probably diminished somewhat. Although some activities would be lessened or discontinued after closure of the landfill, "both landfill gas flaring and recovery, and leachate management and treatment will continue long after the landfill ceases to accept waste."[11]

- For Commercial/Institutional sources we grew the DEC numbers with the average annual growth in occupied commercial space in Manhattan between 1986 and 2000 (1 percent).[12]
- For all other sectors (Healthcare, Education, Wastewater Treatment and Other) we grew 2002 emissions by total NYC population growth 1990 to 2000 (0.9 percent).

Table 4.4. Inventory of Point Sources, by Category

Category	Number of Sources	Reported 2002 Tons	Estimated 2005 Tons	2005 % of Total for Source Category
Energy	20	900	783	79%
Residential	20	62	63	6%
Staten Island Landfill	1	55	55	6%
Healthcare	27	38	39	4%
Commercial/Industrial	26	20	20	2%
Other	10	16	17	2%
Education	7	11	11	1%
Wastewater Treatment	6	8	9	1%
Total	117	1,110	997	100%

As shown in Table 4.4, energy facilities are responsible for 79 percent of the direct $PM_{2.5}$ emissions from point sources. Average energy-facility emissions are upwards of 45 tons. In contrast, facilities in the other categories have an average of just over two tons per facility. The Staten Island (Fresh Kills) Landfill is the single largest non-energy point source of $PM_{2.5}$ emissions, probably attributable to its rock crushing facility and combustion engines on site.[13] Residential, health care, and commercial/industrial

[10] The Title V permit, issued on June 11, 2002, is inconsistent regarding the timeline for landfill closure. According to the permit report, the landfill is "currently actively accepting waste" but "was officially closed on March 22, 2001." New York State Department of Environmental Conservation, Permit Review Report for the Staten Island Landfill facility, Permit ID: 2-6499-00029/00151, 6/11/2002 (http://www.dec.state.ny.us/website/dardata/boss/afs/permits/prr_264990002900151.pdf).

[11] New York City Department of Sanitation, The DOS Report, February 2000, p. 5. (http://www.ci.nyc.ny.us/html/dos/pdf/fkclose.pdf)

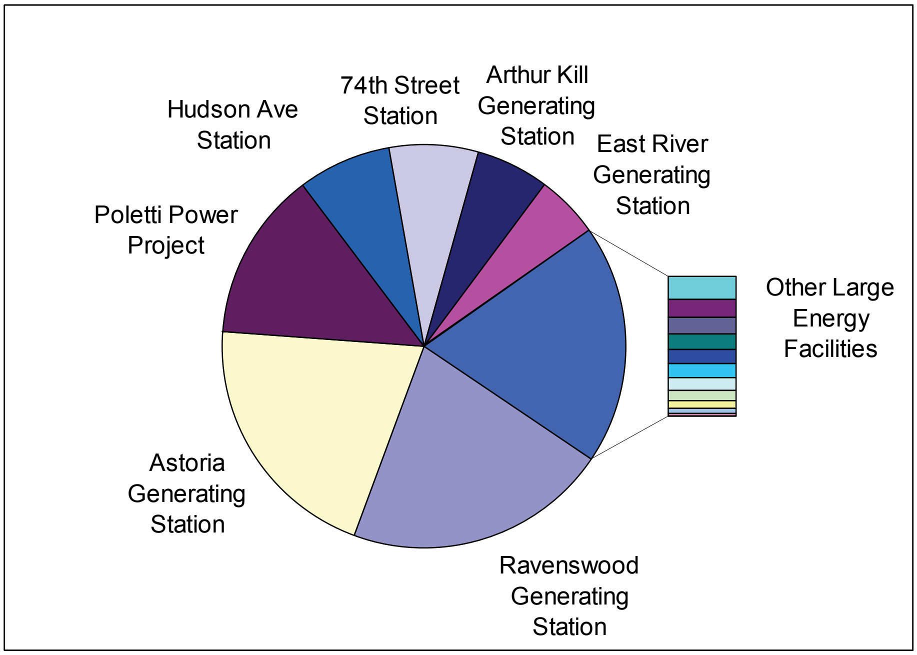
[12] Economics Research Associates and Cushman & Wakefield, *Hudson Yards Redevelopment: Economic Overview and Demand Forecast*, Spring 2003, www.nyc.gov/html/dcp/pdf/hyards/c&w-era-study.pdf.

[13] According to the Fresh Kills (Staten Island) landfill Article V permit report, "support activities currently at the landfill include maintenance and repair of landfill vehicles, vehicle fueling operations, barge docks for unloading of municipal solid waste, a rock crushing and screening operation, a yard waste composting facility and two (2) leachate treatment plants. A landfill gas recovery plant and six (6) landfill gas flares have been installed to mitigate odors and landfill gas emissions from closed sections of the landfill. A gas treatment plant (Selexol Plant #1), to upgrade the landfill gas to pipeline quality gas, is operating at the site. In addition, 2 additional gas treatment plants (Kryosol Plants #2 and #3) and a power array of internal combustion engines are in the process of being built at the landfill" (http://www.dec.state.ny.us/website/dardata/boss/afs/permits/prr_264990002900151.pdf).

facilities combined account for over 12 percent of total point-source $PM_{2.5}$ emissions in New York City, with the remaining categories (Wastewater Treatment, Education, and Other) making only modest contributions.

Because energy facilities account for the majority of point source $PM_{2.5}$ emissions, we focus on those facilities. Emissions from energy facilities are dominated by a small number of facilities: the seven largest energy facilities emit over four-fifths of $PM_{2.5}$ from all 20 energy facilities. Data on these seven facilities are shown in Figure 4.2 and Table 4.5.

Figure 4.2. 2002 $PM_{2.5}$ Emissions from Point Source Energy Facilities



In 2002, these seven energy plants emitted roughly 730 tons of direct $PM_{2.5}$, or 73 percent of emissions from all point sources in NYC. As in shown in Table 4.5, these seven plants also emitted significant amounts of $PM_{2.5}$ precursors, including SO_2 and NOx.[14] As noted, emissions from these plants have probably been reduced somewhat by the addition of several new power plants in NYC.

[14] Data on $PM_{2.5}$ were obtained from the DEC database. Total SO_2 and CO_2 data for six of the seven largest power-plant emitters of $PM_{2.5}$ were obtained from the Acid Rain program (http://www.epa.gov/airmarkets/emissions/prelimarp/index.html). Total 2002 NOx data for the same six plants were obtained from "Heat Input Data Used in the Calculation of State Budgets" (April 14, 2004. Support for CAIR SNPR section II, Docket # OAR-2003-0053-1409, http://www.epa.gov/interstateairquality/pdfs/apr14shi.xls). Data on precursor emissions were not available from the Acid Rain program for the Hudson Ave plant, a non-Acid Rain Plant (Unit and Plant Level Annual Heat Input by Fuel Type, http://www.epa.gov/interstateairquality/pdfs/finaltech09.xls) that provides substantial steam capacity to Con Ed's system. Emissions of SO_2, NOx, and CO_2 by Hudson Ave were obtained from the 2002 NEI database (http://www.epa.gov/ttn/chief/net/2002inventory.html).

Table 4.5. 2002 Emissions from the Seven Largest Energy Facilities in NYC (tons)

Borough	Power Plant	$PM_{2.5}$	SO_2	NOx	CO_2
Queens	Ravenswood Generating Station	190	1,117	3,238	3,314,030
Queens	Astoria Generating Station	185	1,294	2,487	2,585,087
Queens	Poletti Power Project	122	1,009	1,952	1,620,808
Kings	Hudson Ave Station	67	1,862	4,837	2,498,547
New York	74th Street Station	64	778	651	466,013
Richmond	Arthur Kill Generating Station	54	4	590	799,774
New York	East River Generating Station	45	498	1,509	928,597
	TOTAL	728	6,561	15,263	12,212,855

It is worth noting that these 2002 $PM_{2.5}$ emission figures for Poletti, from the NY DEC, are substantially lower than figures published elsewhere for the period 1996 through 2000. We have not investigated whether emissions at this facility actually fell in 2002 or whether there is a discrepancy between these two data sources.

4.4 Area Sources

While the operators of point sources report fuel use to DEC, the smaller sources defined as area sources do not report to DEC. To estimate area-source emissions by sector (residential, commercial/institutional, industrial, and electric utility), DEC first takes state-level fuel use data and subtracts from it the fuel use reported by point sources.[15] For the residential sector, the remaining fuel use is apportioned to counties based on census data. Commercial/institutional, industrial, and electric utility fuel use is allocated to counties using aggregated employment statistics. The DEC then applies area-source emissions factors to the fuel use data to calculate area-source emissions.

The DEC's method of estimating area source emissions is a useful way to allocate emissions across the state in a general way; however it is not likely to estimate emissions in any one county very accurately. This is because the counties within New York State differ considerably in terms of commercialization, industrialization and population density. For example, wood is burned for residential space heating and industrial processes in more rural areas of the state, but there is little wood combustion in NYC. In addition, due to NYC's population density and industrialization, there are certain restrictions in NYC that do not exist, or are not actively enforced, in other areas of the state. For example coal may be burned at industrial facilities in some parts of the state, but it is unlikely that coal is still being burned in NYC. Thus, allocating wood and coal combustion to counties based on industry employment figures is likely to overstate the use of these fuels in NYC considerably.

Based on our understanding of the DEC's inventory methods and our own additional research, we made four adjustments to the DEC's area source inventory in estimating 2005 emissions. First, as we extrapolated residential gas and oil emissions from 2002 to 2005, we continued the recent trend of increasing market share for natural gas and decreasing share for oil. This trend is the result of conversions of older oil systems to gas and the dominant use of gas for new residential heating systems.

[15] DEC uses fuel use data from the NYSERDA Annual Patterns and Trends Report.

Second, we removed all coal and residual oil emissions from electric utility area sources. The only utility area sources emitting $PM_{2.5}$ are likely to be diesel-fueled engines used for grid support. All other non-mobile utility sources are classified as point sources.

Third, we reduced the DEC's estimate of $PM_{2.5}$ emissions from residential wood combustion. According to the U.S. Census housing data used by DEC, only 465 (out of 3 million) residential units in the five boroughs used firewood as their primary heating source in 2002. Of course, firewood is still burned in other residences for decorative and aesthetic enjoyment. However, a detailed 1996 paper from the SUNY College of Environmental Science and Forestry indicates that residential use of firewood in NYC has been declining steadily for a long time. Anecdotal evidence also suggests that ongoing residential conversions, renovations, demolitions and construction are resulting in a steady net reduction in the number of working fireplaces.

Conservatively extrapolating from the trends indicated in the SUNY paper, we estimated that fewer than 8,000 tons of wood were burned in NYC for heating for all purposes (functional plus decorative-esthetic) in 2002, whereas DEC's emission estimate was predicated on a burn rate of almost 30,000 tons. Prorating accordingly, and extending the historical downward trend in wood-burning to 2005, resulted in a marked reduction in estimated $PM_{2.5}$ emissions from burning wood in New York City residences, from DEC's figure of 420 tons for 2002 to just 99 tons in 2005.

Finally, we reduced the DEC's estimates of emissions from all coal combustion, and non-residential wood combustion, by 95 percent, to account for what we believe is an overestimation of these emissions in the DEC inventory. We found no documentation of commercial, institutional or industrial wood combustion in NYC. Rather than completely eliminating these emissions, we applied this reduction factor to produce a conservative estimate of total area-source emissions. Similarly, we reduced coal emissions from all sectors by only 95 percent in order to be conservative. City officials indicated that City schools, probably the most recent consumers of coal in the City, were converted to natural gas, oil, or dual fuel through a spin off of the ENCORE program, or through other City construction programs.

Table 4.6 shows the DEC's 2002 inventory of $PM_{2.5}$ from area sources and our 2005 estimate, reflecting the adjustments discussed here. When viewing Table 4.6, it is important to recall that these are emissions from area sources – that is, small sources. The emissions from large facilities are captured under point sources.

Table 4.6. DEC 2002 Area Source Inventory and Our 2005 Estimate

Sector	Fuel	2002 Tons	2005 Tons
Residential	Gas	626	645
Residential	Distillate Oil	548	536
Residential	Residual Oil	0	0
Residential	LPG	14	14
Residential	Wood	420	99
Residential	Coal	11	1
Subtotal		1,619	1,295
Industrial	Gas	45	47
Industrial	Distillate Oil	8	9
Industrial	Residual Oil	9	9
Industrial	LPG	2	2
Industrial	Wood	0	0
Industrial	Coal	1,044	52
Subtotal		1,109	119
Com/Inst.	Gas	480	495
Com/Inst.	Distillate Oil	256	264
Com/Inst.	Residual Oil	138	143
Com/Inst.	LPG	0	0
Com/Inst.	Wood	44	2
Com/Inst.	Coal	113	6
Subtotal		1,031	909
Electric Util.	Gas	0	0
Electric Util.	Distillate Oil	0.2	0.2
Electric Util.	Residual Oil	152	0
Electric Util.	LPG	0	0
Electric Util.	Wood	0	0
Electric Util.	Coal	442	0
Subtotal		595	0.2
Total		4,354	2,323

There is a considerable amount of uncertainty around this area source inventory. The major adjustments we made to the DEC numbers are as follows:

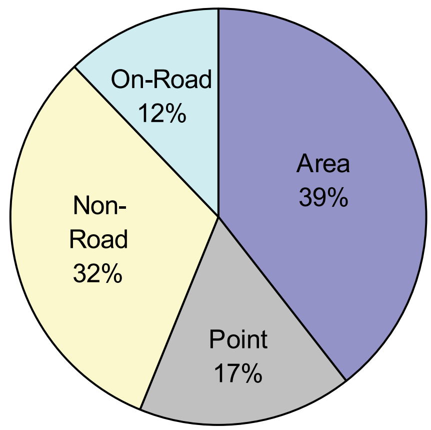
- Coal combustion in the industrial, commercial/institutional and electric utility sectors was reduced by 992, 108 and 442 tons respectively, for a total reduction of 1,542 tons.
- Wood combustion in the residential and commercial/institutional sectors was reduced by 321 and 41 tons respectively, for a total reduction of 362 tons.
- Residual oil combustion at electric utility area sources was reduced by 152 tons.

More research is needed in these areas to improve our understanding of this sector and allow for more efficient regulatory action.

4.5 Summary

Figure 4.3 shows our estimate of 2005 direct $PM_{2.5}$ emissions in NYC, by source category and by borough. Total emissions are 5,903 tons, 33 percent less than the NY DEC's 2002 estimate of 8,862 tons. The largest reduction (2,032 tons) came in the area source category, where we removed a considerable amount of coal and wood combustion allocated to area sources in NYC. The second largest reduction (678 tons) came in the non-road sector, where we made adjustments to account for aspects of NYC that differ significantly from the national average, such as the mix of industry in the city, construction wages and practices, residential living patterns and off-road recreational vehicle use.

Figure 4.3. Estimated 2005 Direct $PM_{2.5}$ Emissions in NYC



Total: 5,903 tons

5. Projected $PM_{2.5}$ Emissions in NYC

To estimate direct $PM_{2.5}$ emissions in NYC we increased the estimated 2005 emissions inventory (discussed in Section 4) to approximate emissions in 2010 and 2015. We used different methods to increase emissions for different source categories, as described in the subsections below.

5.1 On-Road Mobile Sources

The approach used to calculate 2005 emissions from on-road vehicles (see Section 4.1) was also employed for the 2010 and 2015 estimates. MOBILE6.2 emission factors were applied to DEC forecasts of miles to be driven by vehicles in each class on each of the six major roadway types. Compared to DEC's 2005 estimates, its VMT forecasts for 2010 and 2015 imply annual growth rates ranging from 1.5 to 1.7 percent. These growth rates exceed actual growth in recent years and may thus prove to be overestimated.

Emission rates at low speeds were adjusted via the same equation used to calculate emissions while idling in the 2005 analysis. For transit buses, we applied published schedules indicating further adoption of pollution-control measures, headed by the Transit Authority's anticipated use of particulate filters and ultra-low-sulfur diesel fuel, a combination that reduces bus emissions by 90 percent from prior, uncontrolled levels.

After these adjustments, baseline emissions fall to 539 tons in 2010 and 499 tons in 2015 from 718 tons in 2005. The total drop from 2005 to 2015 is 219 tons, or just over 30 percent – an impressive decline, most of which is projected to occur by 2010.

In 2001 EPA adopted regulations on diesel fuel and engines that will, over time, greatly reduce PM emissions from motor vehicles.[1] The 2005-2015 decline in emissions is accounted for largely by three categories of large diesel vehicles:

- HDDV8B (18-wheeler diesel trucks), declines from 85 tons to 26 tons – a drop of 59 tons, attributable to ultra-low sulfur diesel fuel and mandated adoption of improved engine technology on new trucks;
- URB BUS (diesel transit buses), declines from 122 tons to 73 tons – a drop of 49 tons, attributable to the MTA's use of ultra-low sulfur diesel fuel and installation of particulate filters on all Transit Authority buses;
- HDDV8A (the largest category of diesel trucks after 18-wheelers), declines from 45 tons to 14 tons – a drop of 31 tons, attributable to ultra-low sulfur diesel fuel and mandated adoption of improved engine technology on new trucks

[1] U.S. Environmental Protection Agency Office of Transportation and Air Quality Planning, Regulatory Announcement, December, 2000 (http://www.epa.gov/otaq/regs/hd2007/frm/f00057.pdf). See also Environmental Protection Agency, Control of Air Pollution From New Motor Vehicles: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements; Final Rule 66 Fed.Reg. 5001-5050 (January 18, 2001).

5.2 Non-Road Mobile Sources

In May of 2004 EPA promulgated regulations that will dramatically reduce PM emissions from diesel non-road engines over the next ten to twenty years.[2] The regulations require the use of ultra-low-sulfur diesel fuel in all diesel engines (for about a 10 percent reduction in $PM_{2.5}$ emissions) and pollution control devices in most new diesel non-road engines (for a total $PM_{2.5}$ reduction of 90 percent).

Since about 85 percent of NYC non-road $PM_{2.5}$ emissions come from diesel engines, EPA's new regulations will be a big help. The impact of EPA's actions on NYC emissions can be seen in Tables 5.1 and 5.2 for 2010 and 2015, respectively. Compared to 2005 emissions, total non-road $PM_{2.5}$ emissions will decrease by about 20 percent by 2010 and by about 40 percent by 2015.

Furthermore, EPA's 2004 non-road diesel regulations will reduce NYC non-road $PM_{2.5}$ emissions to about one-third of current emissions by 2025 and to about one-fourth by 2040. Emission reductions by 2010 and 2015 reflect both the effect of lower-sulfur fuel and required emission-control devices on newer equipment.

Table 5.1. Projected 2010 Non-Road $PM_{2.5}$ Emissions (Tons per Year) by Borough

Classification	BX	BK	MH	QN	SI	Total
Agricultural Equipment	0	0	0	0	0	0
Airport Equipment	0	0	7	33	0	40
Commercial Equipment	18	66	255	63	9	410
Construction Equipment	66	104	335	239	40	784
Industrial Equipment	17	38	31	38	5	129
Lawn & Garden Equip. (com.)	7	13	17	8	13	58
Lawn & Garden Equip. (res.)	5	17	1	22	7	51
Logging Equipment	0	0	0	0	0	0
Pleasure Craft	3	5	1	12	8	28
Railroad Equipment	2	3	2	3	1	10
Recreational Equipment	1	5	1	3	1	12
Total	119	249	650	422	83	1,523
Change from 2005	-20%	-18%	-18%	-19%	-16%	-18%

In addition to regulations covering diesel engines, EPA has also adopted regulations for non-road spark-ignition engines. While these regulations are aimed primarily at emissions of hydrocarbons and nitrogen dioxide, particulate matter emissions are also expected to drop, as manufacturers of small non-handheld engines (e.g., lawn mowers) switch to 4-stroke engines to meet the standards. However, manufacturers of hand-held equipment (leaf-blowers and chainsaws) will be allowed to retain 2-stroke engines, which have ten times the particulate matter emissions of 4-stroke engines.

Manufacturers of recreational vehicles (off-road motorcycles, all-terrain vehicles, and snowmobiles) may be able to meet recently promulgated standards with 2-stroke engines.

[2] U.S. Environmental Protection Agency, Office of Transportation and Air Quality Planning, Regulatory Announcement, May, 2004 (http://www.epa.gov/nonroad-diesel/2004fr/420f04032.pdf). See also Environmental Protection Agency, Control of Emissions of Air Pollution From Nonroad Diesel Engines and Fuel; Final Rule 69 Fed.Reg. 38957-39273 (June 29, 2004).

If they are forced to use 4-stroke engines, particulate matter emissions from these sources will drop as well. EPA is also currently considering new regulations for outboard marine engines. Since these regulations have yet to be promulgated, potential changes in particulate matter emissions for this category are not reflected in our estimates.

Emission reductions in the future may be even greater than those projected by EPA's NONROAD model. The model includes national growth factors for non-road engines. But future engine populations were distributed to the counties using the current distribution of industrial activity and population. Since NYC's share of national industrial activity and human population may likely continue to decline over time, the NONROAD model probably assigned too high a share of future emissions to the city. On the other hand, the estimates in NONROAD engine growth were based on linear extrapolations of limited data, and in many cases estimated equipment life seems questionable. Thus there is likely to be a significant range of error around the long-term projections from the NONROAD model.

Table 5.2. 2015 Non-Road $PM_{2.5}$ Emissions (Tons per Year) by Borough

Classification	BX	BK	MH	QN	SI	Total
Agricultural Equipment	0	0	0	0	0	0
Airport Equipment	0	0	5	24	0	29
Commercial Equipment	16	58	224	56	8	361
Construction Equipment	46	72	233	167	28	546
Industrial Equipment	10	23	19	23	3	78
Lawn & Garden Equip. (com.)	8	13	18	8	14	62
Lawn & Garden Equip. (res.)	5	18	1	23	8	54
Logging Equipment	0	0	0	0	0	0
Pleasure Craft	3	5	1	12	8	28
Railroad Equipment	1	3	2	2	1	8
Recreational Equipment	1	3	1	2	1	8
Total	90	194	503	317	68	1,173
Change from 2005	-39%	-36%	-36%	-40%	-31%	-37%

5.3 Point Sources

For the power plants in NYC, we project annual generation at the unit level to 2010 and 2015. To estimate changes in $PM_{2.5}$ emissions, we applied the percent changes in generation to the plants' reported 2002 emissions. Thus, our analysis assumes that the units included in the generation model will have constant emission rates across their projected output levels. This assumption implies that plant operations in the forecast years will be similar to operations in 2002, and that no additional emissions controls will be installed over the period of analysis.

We projected generation at large power plants using the PROSYM production costing model.[3] We modeled major power plants in all three northeastern power control areas to

[3] The results of this modeling work were previously published in the Synapse report, *A Clean Electricity Strategy for the Hudson River Valley*, available at: www.synapse-energy.com.

ensure that the results reflected changes occurring in other areas.[4] The results of this modeling for the major downstate plants are shown in Table 5.3. The "Hudson Valley" plants are: Bowline, Lovett, Roseton, Danskammer, Athens and Bethlehem. The "New York City" plants are: the existing units at Arthur Kill, East River, Astoria, Poletti and Ravenswood and the new Poletti and SCS Astoria plants currently under construction.

Table 5.3. Projected Electricity Generation at Large Downstate Plants

	2005	2010	2005-2010 Change	2015	2005-2015 Change
Hudson Valley	15,495	18,054	17%	21,050	36%
New York City	10,389	14,893	43%	15,646	51%
Total	25,884	32,947	27%	36,696	42%

Our modeling indicates that electricity generation in the Hudson Valley and NYC will increase significantly over the study period, driven primarily by the new plant additions in the region. As shown in Table 5.3, generation from the fossil-fired plants in the Valley is projected to grow by 17 percent between 2005 and 2010 and by 30 percent between 2005 and 2015. (Note that much of the increased generation is projected to serve load outside of the Hudson Valley.) Generation at the five NYC plants increases by 43 percent between 2005 and 2010, with the addition of the new Poletti and SCS Astoria units.[5] Output from the NYC units grows by 51 percent between 2005 and 2015, primarily driven by load growth.

For the smaller generating units in NYC (Gowanus, the Astoria gas turbines, the cogeneration plant at Kennedy Airport, Narrows, and the Brooklyn Navy Yard), we projected utilization (and emissions) increasing at one percent per year. This is slower than projected load growth, because we project much of the load growth to be met by the new large plants in the region.

We projected growth in emissions from steam or cogeneration plants at 0.5 percent annually. This assumption is based on historical steam sales (essentially constant) and Consolidated Edison's planned steam activities as discussed in the Company's 2004 Annual Report.[6]

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[4] Changes in operations of non-fossil generation, such as the Indian Point nuclear power plant, would affect operations and emissions from fossil-fired power plants. Concerns about Indian Point's vulnerability to terrorist attack have prompted some to call for closure of the units currently in operation when their operating licenses expire in 2013 and 2015 (or sooner). However, a report by the National Academies' National Research Council determined that political, regulatory, and financial hurdles would make replacing the power from Indian Point difficult. (see National Research Council 2006 "Alternatives to the Indian Point Energy Center for Meeting New York Electric Power Needs." Available from www.nap.edu) In the generation modeling described in herein, we assume that Indian Point is operational throughout the period of analysis.

[5] In our power system modeling, we assumed that 1,000 MW of capacity would be added at the SCS Astoria facility. SCS Astoria is now indicating that it may only construct 500 MW of capacity there during the near term.

[6] The rate plan includes new initiatives to "grow our steam business in order to take advantage of existing capacity and relieve pressure on our electric system" (Consolidated Edison Company of New York, Inc., 2004 Annual Report, p. 4)

The factors used to grow emissions in the other point source categories are as follows.

- We assume that emissions from the Staten Island (Fresh Kills) Landfill will remain roughly constant over the study period. As of the most recent Title V operating air permit, only 771 of about 2,200 acres were actively accepting waste.[7] As a result, fine particulate emissions have probably diminished somewhat. Although some activities would be lessened or discontinued after closure of the landfill, "both landfill gas flaring and recovery, and leachate management and treatment will continue long after the landfill ceases to accept waste."[8] More research is needed to understand how $PM_{2.5}$ emissions will change as activities decrease or stop altogether.
- We project growth in residential point-source emissions using U.S. Census housing data for NYC, aggregated from county-level data (see Section 4.2 for a description of the sources falling into this category). We assume that growth in households and in housing units from 1990 to 2000, both 0.7 percent per year, would continue at this rate through the end of the forecast period.[9]
- We project emissions from the Commercial/Industrial sector using the average annual growth in occupied commercial space in Manhattan between 1986 and 2000 (1 percent).[10]
- For the Wastewater Treatment, Healthcare, Education and Other sectors we projected emissions using annual population growth between 1990 and 2000 (0.9 percent).

5.4 Area sources

The types of sources in the various area source categories are described in Section 4.4. The growth factors used to estimate area source emissions in 2010 and 2015 are as follows.

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[7] The Title V permit, issued on June 11, 2002, is inconsistent regarding the timeline for landfill closure. According to the permit report, the landfill is "currently actively accepting waste" but "was officially closed on March 22, 2001." New York State Department of Environmental Conservation, Permit Review Report for the Staten Island Landfill facility, Permit ID: 2-6499-00029/00151, 6/11/2002 (http://www.dec.state.ny.us/website/dardata/boss/afs/permits/prr_264990002900151.pdf).

[8] New York City Department of Sanitation, The DOS Report, February 2000, p. 5. (http://www.ci.nyc.ny.us/html/dos/pdf/fkclose.pdf)

[9] Our use of households to determine point source emissions growth differs somewhat from EPA's suggested method, which relies on population (E.H. Pechan & Associates, Inc., Economic Growth Analysis System: Version 4.0 Reference Manual, Final Draft. January 26, 2001. U.S. EPA Office of Air Quality Planning and Standards. Pechan Rpt. No. 01.01.004/9008-404. See http://www.epa.gov/ttn/chief/emch/projection/egas40/ref_man_4.pdf). However, population growth does not take into account that the number of residents per square foot of living space may change with rent costs and therefore not mirror growth in emissions to heat primarily fixed-sized residential buildings.

[10] Economics Research Associates and Cushman & Wakefield, *Hudson Yards Redevelopment: Economic Overview and Demand Forecast*, Spring 2003, www.nyc.gov/html/dcp/pdf/hyards/c&w-era-study.pdf.

- For residential area sources, we grew emissions from natural gas use at 0.37 percent per year, the recent rate of growth in the NYC housing stock. (Thus, all growth in heating demand is met by gas, consistent with data for recent years.) We decreased the number of heating systems using oil by 1.05 percent per year, shifting them to gas. This is half of the recent historical rate of conversion, reflecting an expected slowdown in the rate of conversion due to rising gas prices. We further assume that 10,000 units are weatherized annually, and that weatherized units use 25 percent less fuel.

  As discussed in Section 4.4, we reduced the DEC's 2002 estimate of emissions from residential coal combustion by 95 percent for our 2005 estimate, based on the assumption that little coal is actually burned in NYC residences. We held residential coal emissions constant over the study period at this level. We continued the trend of decreasing use of wood for residential space heating as projected in the 1996 SUNY paper.[11]

- For commercial/institutional and industrial sources, we used the average annual growth in occupied commercial space in Manhattan between 1986 and 2000, (1 percent). We held emissions from coal and wood combustion in this sector constant through the study period at a level 95 percent below the DEC's 2002 estimate. (See Section 4.4 for a discussion of this.)

- For electric utility area sources we use NYISO forecasts of electricity load growth in NYC (1.2 percent per year).[12] As shown in Section 4.4, we estimate electric utility area source emissions to be less than one percent of total area source emissions.

5.5 Policies Considered in these Projections

5.5.1 $PM_{2.5}$ NAAQS/SIP

New York State is currently under mandate to develop a state implementation plan (SIP) to address concentrations in excess of the US EPA's 1997 fine particulate standards. The agency responsible for drafting the $PM_{2.5}$ SIP in New York State, the Division of Air Resources (DAR),[13] has only started the process for developing a plan in response to the January EPA rule.[14] The DAR is developing a photochemical grid model to estimate

[11] Canham, Hugh and Thomas Martin, Residential Fuelwood Consumption in New York State: 1994-1995, SUNY College of Environmental Science and Forestry, 1996, www.esf.edu/centerweb/canham1.htm.

[12] Regional Annual Energy Requirements Forecast (GWH), table I-3, 2005, Load & Capacity Data, NYISO 2005 Load and Capacity report, p. 6.

[13] Currently, DEC is also responsible for SIPs dealing with ozone, carbon monoxide, and PM_{10}, and is working with the US EPA to formulate standard practices for regional haze (http://www.dec.state.ny.us/website/dar/baqp/index.html), some of which take precedence over $PM_{2.5}$ planning (May 3, 2005, phone interview, John Kent of the Bureau of Air Quality Planning, a division of DAR).

[14] 40 CFR Part 81; Air Quality Designations and Classifications for the Fine Particles (PM2.5) National Ambient Air Quality Standards; Final Rule

ambient $PM_{2.5}$ levels under various scenarios. To verify its accuracy, the model will probably undergo several iterations of programming and testing with historical data.[15] New York must submit its plan by early 2008 and is required to be in compliance no later than 2010.[16]

DAR staff has indicated that it is too early to estimate the amount of reductions attributable to the $PM_{2.5}$ SIP.[17] Moreover, emission reductions from this policy would probably occur towards the end of the period of analysis. Therefore, we do not include emission reductions from a $PM_{2.5}$ SIP in our base case scenario. However, it is likely that some of the strategies we assess in this paper will be included in the DAR's final SIP. This is discussed further in Section 6.

The recently-released 2006 $PM_{2.5}$ NAAQS may require policy changes in NYC and New York State. In 2005, Kings (Brooklyn), New York (Manhattan) and Bronx counties registered $PM_{2.5}$ concentrations higher than the 2006 $PM_{2.5}$ standard of 35 $\mu g/m^3$ at the 98[th] percentile. By November 2007, New York State must recommend areas to be designated nonattainment to the US EPA; its SIP is due in April 2013. Given the uncertainty about the ability of other policies (including those listed below) to obviate the need for additional reduction policies, we did not factor additional measures to comply with the 2006 $PM_{2.5}$ NAAQS into the analysis.

5.5.2 Clean Air Interstate Rule

The federal Clean Air Interstate Rule (CAIR) will establish a NO_x trading program in the 28 easternmost states and the District of Columbia and will require reductions in SO_2 emissions in these states as well.[18] The CAIR rule should help to reduce NYC's ambient $PM_{2.5}$ concentrations over the long term, by reducing emissions by electric generating units in upwind states. However, these reductions are not likely to come soon enough to reduce New York's burden under the $PM_{2.5}$ SIP. Further, New York State's own contribution to NYC $PM_{2.5}$ is 0.51 $\mu g/m^3$, which is larger than the contribution from any other state except Pennsylvania.[19] Accordingly, New York State will play an integral part in reducing NYC's $PM_{2.5}$ levels.

The EPA projects total ambient $PM_{2.5}$ concentrations in NYC to decrease by about one percent annually through 2010, without implementation of CAIR. With CAIR, they

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[15] May 3, 2005, phone interview, John Kent of the Bureau of Air Quality Planning, a division of DAR

[16] "EPA Announces Final Designations for First Fine Particle Standard", EPA Newsroom. http://yosemite.epa.gov/opa/admpress.nsf/4d84d5d9a719de8c85257018005467c2/bc63cfdda235542585256f6d005e6738!OpenDocument.

[17] May 3, 2005, phone interview, John Kent of the Bureau of Air Quality Planning, a division of DAR

[18] The CAIR rule will eventually replace the requirements of the OTC NOx Budget Program and the NOx SIP Call. See U.S. Environmental Protection Agency, Rule To Reduce Interstate Transport of Fine Particulate Matter and Ozone (Clean Air Interstate Rule); Revisions to Acid Rain Program; Revisions to the NOx SIP Call; Final Rule 70 Fed.Reg. 25161 (May 12, 2005).

[19] U.S. Environmental Protection Agency Office of Air Quality Planning and Standards, Technical Support Document for the Final Clean Air Interstate Rule: Air Quality Modeling, Appendix H, March 2005 (http://www.epa.gov/interstateairquality/pdfs/finaltech02.pdf)

project that $PM_{2.5}$ in the 5 boroughs will go down just by over 2 percent per year.[20] In our reference case, total emissions from large power plants in the City fall by roughly two percent per year, due largely to the addition of new, clean power plants in the downstate area. Therefore, we do not believe that CAIR will result in emission reductions in NYC relative to a scenario without the rule.

5.5.3 Federal Diesel Engine Regulations

The U.S. Environmental Protection Agency has been regulating diesel fuel and vehicles since the 1980's. Initial regulations set particulate emission standards for trucks and buses and then lowered diesel sulfur content to help meet the standards.[21]

In December, 2000, EPA established a comprehensive national control program to regulate the heavy-duty vehicles and their fuel as a single system. As part of this program, new emission standards will begin to take effect in model year 2007 and will apply to heavy-duty highway engines and vehicles. These standards are based on the use of high-efficiency catalytic exhaust emission control devices or comparably effective advanced technologies. Because these devices are damaged by sulfur, EPA also mandated a reduction in the level of sulfur in highway diesel fuel by 97 percent by mid-2006.

EPA then adopted a similar regulation for nonroad diesel engines in May 2004 that will begin taking effect in 2008 and fully apply in 2015.

5.5.4 New York's Renewable Portfolio Standard

In 2005, the NYS Public Service Commission (PSC) approved an implementation plan for a renewable portfolio standard (RPS) that requires an increase in the renewable energy share of New York's retail electric supply, from the current level (projected to be approximately 20 percent in 2005) to 25 percent by 2013. The policy is intended to support the development of renewable energy, increase energy resource diversity, and reduce air emissions.

The RPS could affect direct $PM_{2.5}$ emissions by displacing existing in-city, fossil-fired generation with RPS resources located either inside or outside of the City.[22] Although most of the RPS capacity is expected to be wind power located in western New York, a recent analysis estimates that between 1,000 and 1,500 GWh of thermal in-city generation would be displaced in 2008, depending on assumptions about wind forecast

[20] U.S. Environmental Protection Agency Office of Air Quality Planning and Standards, Technical Support Document for the Final Clean Air Interstate Rule: Air Quality Modeling, Appendix F, March 2005 (http://www.epa.gov/interstateairquality/pdfs/finaltech02.pdf)

[21] See U.S. Environmental Protection Agency, Emission Standards Reference Guide for Heavy-Duty and Nonroad Engines, September, 1997, http://www.epa.gov/otaq/cert/hd-cert/stds-eng.pdf and http://www.epa.gov/otaq/invntory/overview/solutions/milestones.htm.

[22] It should be noted that the RPS would also cause out-of-city generation to be displaced. By reducing concomitant precursor emissions from this rest-of-state generation, the RPS could reduce $PM_{2.5}$ *concentrations* in NYC. The wind patterns assumed in the CAIR analysis imply that emissions reductions in NYISO zones F, G, H and I could affect concentrations in zone J. However, projecting this reduction in ambient $PM_{2.5}$ concentrations in NYC would involve many variables and considerable uncertainty.

accuracy and system operations.[23] The NY RPS Cost Study Report concludes that the RPS would produce a 7.20 percent reduction in NO_x, a 7.09 percent reduction in SO_2, and an 8.25 percent reduction in CO_2 emissions in the New York metropolitan area (including Long Island) when the RPS reaches its target level in the year 2013.[24]

As the RPS only applies to *retail* electricity sales, we assume that the RPS will not reduce area source emissions from distributed generation.

5.5.5 The Regional Greenhouse Gas Initiative

Another regional emission reduction initiative may also impact ambient $PM_{2.5}$ concentrations in NYC after January 1, 2009. The State Regional Greenhouse Gas Initiative (RGGI) is a cooperative multi-state initiative begun in 2003, at the behest of Governor Pataki, to design a regional greenhouse emissions cap-and-trade program, initially focusing upon carbon dioxide emissions from electric generating facilities within a 10 State region. Expanding upon the regional greenhouse gas emissions reduction efforts in the New England Governors and Eastern Canadian Premiers Climate Change Resolution, on December 20, 2005 the governors of Connecticut, Delaware, Maine, New Hampshire, New Jersey, New York, and Vermont signed a Memorandum of Understanding agreeing to a mandatory emissions trading reduction strategy for carbon dioxide emissions from electric generating facilities.

Carbon dioxide emissions from electric generating units over 25 megawatts in capacity will be capped at current levels beginning in 2009, and eventually reducing emissions by 10 percent beginning in 2019. RGGI may eventually include additional participating states upwind of NYC, as representatives from Maryland and Pennsylvania observed the process.

While approximately 600 electric generating units within the participating States are subject to RGGI, the power system modeling provided in this report was completed prior to the conclusion of negotiations over RGGI reduction targets, and plant-level data on projected changes in generation resulting from RGGI are not presently available. It is reasonable to anticipate that, once fully implemented, RGGI will likely have an impact on ambient $PM_{2.5}$ concentrations in NYC through complementary energy policies.[25] However, due to the transmission constraints limiting the amount of in-city load served by regional electric generating units, it remains uncertain what the extent or nature of the impact on $PM_{2.5}$ concentrations will be.

[23] This study and others assume that no RPS resources would be added in NYC (GE Energy Consulting for NYSERDA (Mar 2005) “The Effects of Integrating Wind Power on Transmission System Planning, Reliability, and Operations: Report on Phase 2”; Potomac Economics (Jun 2005) “Estimated Market Effects of the New York Renewable Portfolio Standard” p. 5). While we believe that the number of Photovoltaic systems (PV) in NYC will increase, we found no relevant projections for PV in the context of the RPS.

[24] New York State Department of Public Service et al. (Feb. 2004) “New York Renewable Portfolio Standard Cost Study Report II, Volume A” p. 2.

[25] These include expanding energy efficiency programs, reforming utility rates, and promoting distributed generation. (Environment NorthEast, “Regional Greenhouse Gas Initiative: Summary of Final Policy Design”. http://www.env-ne.org/Program%20Fact%20Sheets/RGGI%20Summary%20and%20Comparison%20ENE.pdf)

5.5.6 New York's System Benefits Charge

The System Benefits Charge (SBC) is a fund collected from the customers of six participating NYS utilities[26] to fund public policy initiatives not expected to be adequately addressed by New York's competitive electricity markets. As the fund's administrator, the New York State Energy Research and Development Authority (NYSERDA) manages a portfolio of predominantly statewide programs. In authorizing the first two funds—SBC and SBC II—NY PSC granted NYSERDA considerable latitude in determining the scope and focus of individual programs within the broad categories defined by the PSC, including energy efficiency, research & development, and low-income. In the reauthorization proceeding for SBC III, the PSC solicited public comment on issues that could have a substantial impact on fund allocation, including use of SBC funds for natural gas energy efficiency programs. While the PSC deferred natural gas efficiency funding to another proceeding, it recently reauthorized SBC III and increased funding levels for electric programs from $150 million to $175 million.[27],[28] Because the reauthorization will realize only a fraction of efficiency potential, there are more opportunities for emissions reductions through energy conservation.[29]

As discussed in section 5.5.7 below, we represent both Consolidated Edison's (Con Ed) electric JP programs and the New York SBC by assuming little or no electricity load growth in the city between 2005 and 2008.

5.5.7 Consolidated Edison Electric Rate Settlement

In March of 2005, the NY PSC approved essentially all the terms of the Joint Proposal ("Electric JP") for Con Ed's electric rate plan, effective from April, 2005 to March, 2008. Section J of the Electric JP includes provisions for demand management (DM), including up to 150 MW of targeted energy efficiency (EE) and/or distributed generation (DG) and 150 MW of system wide EE/DG/load management (LM).[30] Probably the largest effect of

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[26] The participating utilities are Con Ed, Central Hudson Gas and Electric, Inc. (CHG&E), Niagara Mohawk Power Corporation (NMPC), New York State Electric and Gas Corporation (NYSEG), Orange and Rockland Utilities, Inc. (O&R), and Rochester Gas and Electric Corporation (RG&E).

[27] State of New York Public Service Commission. "Public Service Commission Extends Statewide Energy Efficiency Program: Additional Funds Directed to Programs for Low-Income Customers" Dec 14, 2005. 05083/05M0090. See http://www.dps.state.ny.us/.

[28] The Con Ed JP does include some projections for energy savings from SBC III, based on funding at the current level. It states that "SBC programs anticipated in the period after June 30, 2006 (referred to as SBC III) are expected to yield 300 MW of DM in [Con Ed's] service territory (of which 120 MW is described as permanent)." (3/24/05 order, p. 59)

[29] NYSERDA's efficiency potential study estimates between 50,374 and 57.367 GWh of economic energy efficiency potential in 2012, as opposed to the projected annual savings from the SBC II at full implementation, 2,700 GWh. The increase in funding from SBC II ($150 million) to SBC III ($175 million) suggests a relatively small increase in the amount of energy efficiency potential that will be realized as a result of the program expansion. (NYSERDA. Energy Efficiency and Renewable Energy Resource Development Potential in New York State: Final Report. Aug 2003. Vol I, Tables 1-5 and 1-6. See http://www.nyserda.org/publications/EE&ERpotentialVolume1.pdf; New York Energy $mart Program Quarterly Evaluation and Status Report, Quarter Ending September 30, 2004, p. 2).

[30] The system-wide DM program(s) would be expanded to a total of up to 300 MW to cover any shortfall in the targeted DM program.

these programs will come from peak load management, reducing electricity demand in Con Ed's service territory and reduce emissions from the fossil-fired plants that would otherwise supply that load.[31] Together with the demand management benefits achieved by the System Benefit Charge (SBC II), section J programs aim to meet the projected peak load growth in Con Ed's service area—535 MW over the three rate years, of which 452 MW is projected for zone J.[32] The order approving the Electric JP established a collaborative process for drafting an Action Plan, with a target completion date of September, 2005, to determine how these programs would achieve target reductions.

The Electric JP also requires that Con Ed facilitate interconnection requests for DG facilities between 2 & 5 MW.[33] The net effect of this policy will depend on the type of DG projects developed in the City. Technologies such as fuel cells, gas-fired engines and solar arrays are likely to have emission rates lower than the large plants they displace, so they would provide net reductions in $PM_{2.5}$ emissions. Diesel-fueled engines, however, have considerably higher $PM_{2.5}$ emissions than the large plants in the City, so these units would increase total emissions. We do not include $PM_{2.5}$ increases or reductions from facilitated interconnection in our reference case.

5.5.8 Consolidated Edison Gas Settlement

On November 21, 2003, Con Ed filed a rate plan for its regulated steam and natural gas operations. The settlement in this case ("Gas JP") provided for a gas energy efficiency pilot program in Con Ed's service territory, funded at $5 million. We expect that small (area-source) emitters will experience the largest efficiency improvements and emission reductions from this program.[34] The pilot program will tap into only a small portion of the economic energy efficiency potential in the Con Ed service area.[35] The impacts of this program are

[31] While 80 percent of zone J capacity requirements must be located in-city, this requirement does not mean that 80 percent of energy will be produced in NYC. At any given time, more or less than 80 percent of load might be served by in-city capacity, depending on transmission or generation outages, as well as plant economic and operational characteristics. Furthermore, a dedicated power source located outside of the City (e.g., Linden, NJ) may be used to satisfy the 80 percent requirement.

[32] SBC II is "expected to yield 250 MW of DM in the Company's service territory (of which 80 MW is described as "permanent")" (Mar 24, 2005 order, p. 58-). Peak load growth was forecasted by NYISO in its 2004 Load & Capacity Report.

[33] State of New York Public Service Commission, Order Adopting Three-Year Rate Plan. March 24, 2005. Case 04-E-0572. p. 60.

[34] Gas efficiency program funds are to be allocated as follows: 50 percent low income (including weatherization and efficiency for owners of multi-family units with low income tenants), 25 percent other residential, and 25 percent commercial. (Case 03-G-1671, Consolidated Edison Company of New York, Inc. – Gas and Steam Rates, Order Adopting the Terms of a Joint Proposal, issued September 27, 2004).

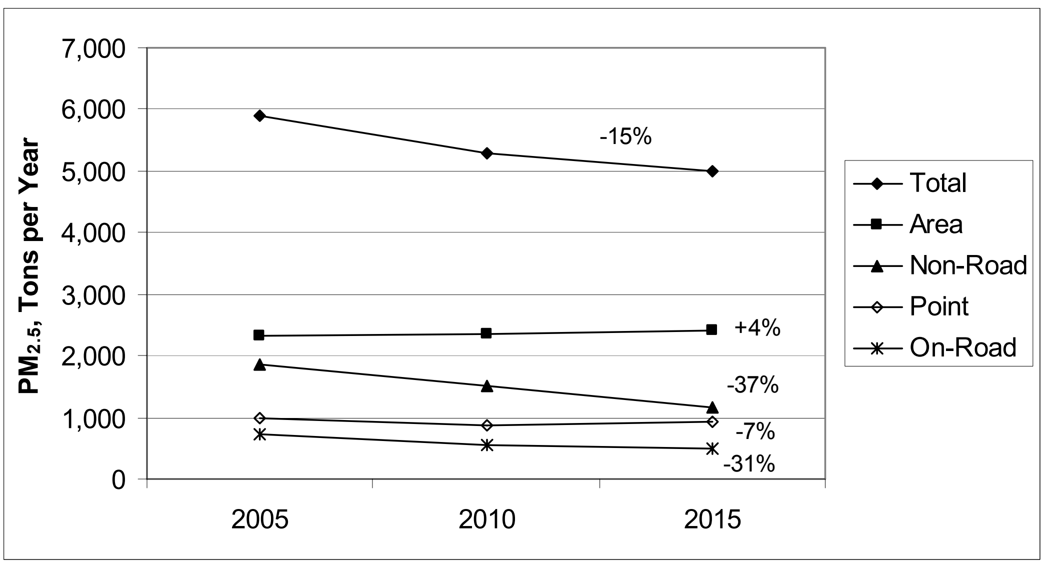
[35] According to "Natural Gas Energy Efficiency Resource Development Potential in Con Edison Service Area", a study undertaken in accordance with the terms of the settlement, economic gas efficiency potential is estimated at 32,014 Thousand Dth in 2016 with net benefits of over $4 billion and a benefit/cost ratio of 3.23, assuming that all economic potential is captured and putting aside the effects of market barriers and the cost of strategies to overcome these barriers. This economic potential is more than 30 times greater than the estimated savings from an average annual funding level of $7.5 million, roughly 922 Mdth savings in 2016.

expected to be small compared to those of other factors, such as mobile sector regulations, and modeling these impacts was beyond the scope of this report. Thus, we remain conservative and do not include $PM_{2.5}$ reductions from Con Ed's gas efficiency program.

5.6 Summary of Projections

Figure 5.1 summarizes our projections of direct $PM_{2.5}$ emissions from sources in NYC. Total annual emissions are projected to fall by roughly 15 percent, or nearly 900 tons between 2005 and 2015. While the percentages shown in Figure 5.1 should not be construed as precise predictions, we believe that the trends appearing in the different source sectors are valid. The largest portion of the overall reduction comes in the non-road sector, driven by federal regulations on diesel equipment. Emissions from on-road sources are also projected to fall significantly as a result of federal regulations on diesel trucks and the installation of retrofit equipment on Transit Authority buses. Emissions from large point sources are projected to fall by a small amount over the study period, as increasing production to meet growth is offset by the addition of new, cleaner plants in the electricity sector. Area source emissions increase by a small amount, as growth in commercial and industrial activity more than offsets reductions in the residential sector projected to result from the continuing shift from oil to gas for residential heating.

Figure 5.1. Projected Trends in Direct $PM_{2.5}$ Emissions in NYC



In August 2006, the New York Public Service Commission solicited comments on the technical aspects of this study as well as NYSERDA's recommendations that the pilot program be extended to a total of five years and "funded at a level sufficient to result in meaningful program savings for customers." (Optimal Energy Inc., American Council for an Energy-Efficient Economy, Vermont Energy Investment Corp., Resource Insight Inc. and Energy and Environmental Analysis Inc. Natural Gas Energy Efficiency Resource Development Potential in Con Edison Service Area. Prepared for NYSERDA, Mar 9, 2006. http://www.dps.state.ny.us/ConEdison_Gas_Efficiency_Study.html, accessed Jan 18 2007; State of New York Public Service Commission, Aug 14 2006. Notice Soliciting Comments, Case 03-G-1671. http://www.dps.state.ny.us/index.html, accessed Jan 18, 2007.)

6. $PM_{2.5}$ Reduction Options

In this Section, we turn to options to reduce ambient $PM_{2.5}$ levels in NYC. These options fall into three general categories: increasing efficiency, fuel switching or blending, and emission control equipment. Each strategy has some direct cost of implementation, and some strategies also generate offsetting cost savings or revenue. For example, switching to low-sulfur oil prolongs the life of a boiler and reduces maintenance costs, and taxes produce revenue.

We evaluate the benefits of $PM_{2.5}$ reduction strategies in the following four categories.

- $PM_{2.5}$ reductions,
- Other pollutant reductions,
- Efficiency gain (reduced fossil fuel use),
- Other economic benefits.

Other economic benefits include things like the creation of local jobs and the reduction of traffic congestion.

Because we have reviewed a large number of emission reduction options using only publicly available data, the cost estimates presented are not intended to be precise. Rather, this analysis is a first screen intended to direct further research toward the more promising options. When considering the cost effectiveness of a given strategy as a $PM_{2.5}$ reduction strategy (i.e., in terms of dollars per ton removed), it is important to bear in mind the ancillary benefits listed above. For example, a strategy may be estimated to reduce $PM_{2.5}$ emissions at a cost of $1,000 per ton. However, the strategy may also provide significant NO_x and CO_2 reductions and reduce fossil fuel use. These additional benefits are important aspects to consider when comparing the strategy to others.

The following subsections present control strategies within the same four source categories presented in previous Sections of this report (on-road, off-road, point and area). At the end of each subsection we provide a summary table comparing the costs and benefits of the strategies for that sector. At the end of the Section, Table 6.12 compares all of the strategies examined.

6.1 On-Road Mobile Sources

We examined seven strategies for reducing emissions from on-road vehicles. This category comprises passenger vehicles including cars and so-called light trucks; vans; freight trucks; buses; and motorcycles; but not construction or other non-road vehicles, which are treated in Section 6.2.

The strategies are:

1. Require “other” buses to adopt the same aggressive emission controls as Transit Authority and NYC DOT buses;
2. Impose tolls on the City’s East River bridges;
3. Impose “cordon pricing” around the Manhattan Central Business District;

4. Impose a weight-distance fee on all motor vehicle travel within NYC;
5. Impose a weight-distance fee on NYC truck travel only;
6. Raise gasoline taxes in NYC, possibly as part of a larger program to use carbon taxes to reduce greenhouse gas emissions;
7. Offer per-mile insurance pricing to owners of passenger vehicles in NYC.

The first strategy extends to non-government operated buses the emission reduction measures now being implemented on buses operated by the Transit Authority and the NYC Dept. of Transportation. Strategies two through seven apply a gamut of pricing strategies to reduce vehicle miles traveled, which will shrink emissions both by diminishing the number of tailpipes and by reducing the share of vehicle travel taking place at very low speeds.

6.1.1 Install Controls on Remaining Buses

As noted in Section 4.1, the NYC Transit Authority has converted its diesel buses to ultra-low-sulfur diesel fuel (ULSD) and has begun retrofitting its diesel engines with diesel particulate filters (DPFs), a combination that reduces $PM_{2.5}$ emissions by an estimated 90 percent from uncontrolled levels. In addition, both the Transit Authority and private bus companies subsidized by New York City DOT have converted several hundred diesel buses in their respective fleets to compressed natural gas, which also burns more cleanly than ordinary (0.2-percent sulfur) diesel fuel. Since the two agencies account for the lion's share of bus vehicle miles traveled in New York City (an estimated 68 percent and 18 percent, respectively), this means that the vast majority of NYC buses are on a path to major reductions in $PM_{2.5}$ — as reflected in the projected drop in diesel bus baseline emissions from 122 tons in 2005, to 68 tons in 2010.

However, the remaining 14 percent of bus VMT, which is accounted for by approximately 1,000 buses owned by a myriad of companies (tour agencies, schools, intercity operators and the like) is lagging behind. Mandated use of ultra-low-sulfur fuel is expected to reduce their $PM_{2.5}$ emissions by around 5 percent, but few if any of these buses are scheduled to be retrofitted with particulate traps, oxidation catalysts or other devices that directly eliminate particulate emissions. Indeed, with all of the NYCTA and DOT bus improvements scheduled to be in place by 2010, and with no expected changes to "other" buses beyond the mandated use of ultra-low-sulfur fuel by the same date, transit bus emissions of $PM_{2.5}$ are actually expected to increase (albeit slightly, from 68 to 73 tons) from 2010 to 2015, as bus VMT grows and travel speeds deteriorate due to worsening traffic congestion.

An obvious countermeasure is to mandate use of the same or equivalent high-performing $PM_{2.5}$ control devices on all buses operating in NYC. (See Section 6.2.1 for a discussion of DPFs and other control devices for diesel engines.) By imputing to "other" buses the per-mile $PM_{2.5}$ emission factors we derived for diesel buses operating with particulate traps and running on ultra low-sulfur diesel fuel, we estimate that this mandate would reduce baseline emissions from all diesel buses in NYC in 2015 by 39 tons; the baseline estimate of 73 tons would fall to just 34. Insofar as NYCTA and DOT together intend to complete retrofitting their combined fleets of nearly 6,000 buses by 2010, there would appear to be no reason that the other operators couldn't do the same for their 1,000 buses by 2015, or even somewhat earlier. (To be conservative, we assume complete implementation by 2015.) Reductions in emissions of carbon monoxide (CO) unburned hydrocarbons (HC) would also result, with the reduction rates depending on the specific control devices installed.

Costs for this control measure would be modest. Based on average capital costs of $8,000 to retrofit each bus, a 5-year life for the equipment, a 3 percent real cost of capital, and an additional $1,000 per bus in annual maintenance, the estimated average annualized cost per bus is $2,700. With 1,000 buses to be retrofitted, the annual retrofit cost is then $2.7 million. The additional cost of ultra low-sulfur diesel fuel for all 1,000 buses is approximately $500,000 (based on a 10 cent-a-gallon fuel cost[1] applied across all 25 million miles of VMT for the buses in question, with an assumed fleet average of 5.2 miles per gallon). The total annual cost of $3.2 million implies an average cost of $41 per pound of $PM_{2.5}$ eliminated.

6.1.2 Impose Tolls on East River Bridges

Although tolling – or re-tolling – the bridges connecting Manhattan to Queens and Brooklyn (all four East River bridges were tolled until 1911) fell off the mayoral agenda midway through Mayor Michael Bloomberg's first term, there is at least an outside chance it could be revived in a second Bloomberg administration. The benefits of bridge tolls, in terms of new municipal revenue (an estimated $800 million a year, before counting tolling and other administrative costs) and improved traffic flow (due to the diminution and "re-assignment" of a small but strategic number of bridge trips) are great enough to make bridge tolls perpetually tantalizing to economists, traffic engineers and other "good government" types.[2] The lone remaining hurdle, albeit a tall one, would seem to be the emergence of one or more public officials able and willing to lead a campaign to win majority support from the New York City Council and State Legislature, along with the Mayor and Governor.

East River bridge tolls — and, indeed, any of the vehicle pricing strategies discussed here — would reduce $PM_{2.5}$ emissions in two complementary ways. First, vehicle miles traveled (VMT) would decline, as the tolls induced a switch of some "marginal" car and truck trips (those whose benefits to the driver barely exceed the cost without the toll) to other modes such as subway, bus, walk, bike or carpool, while other trips vanished altogether.

In a detailed report several years ago, the author of this section employed an elaborate price-elasticity model to calculate that East River tolls priced at the $8.00 rate per round-trip for cars equipped with E-ZPass that the Metropolitan Transportation Authority charges on its NYC bridges and tunnels (e.g., Triborough Bridge, Queens-Midtown

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[1] See Section 6.4.1 for source of the estimated 10¢/gallon cost for ultra-low sulfur diesel fuel.

[2] A web site devoted to analyzing and advocating East River bridge tolls was founded by the author of this section in 2002. The site, www.bridgetolls.org, contains commentary on bridge toll politics along with two detailed research reports quantifying bridge toll demographics and traffic impacts. The site also has links to bridge tolls research by other groups including the Regional Plan Association.

Tunnel, etc.) would cause 6 percent of current trips on the four bridges to disappear; the overall shrinkage in citywide VMT would be one-tenth of that amount, or 0.6 percent.[3] Thus, all things equal, tolling the East River bridges should cause $PM_{2.5}$ emissions from on-road vehicles in NYC to decline by the same modest 0.6 percent.

The second way in which East River tolls would reduce emissions is through improved traffic flow from thinning the traffic "stream." (Note that because proven electronic technology permits motorists to pay tolls "at speed" with E-ZPass, the specter of space-consuming, pollution-generating toll plazas no longer applies; vehicles without E-ZPass would still be able to use the MTA crossings, all of which have toll lanes for cash-paying drivers.) Although bridge tolls would have little if any impact on traffic speeds in some faraway parts of the city such as the Bronx, the impacts would be significant not just on the spans themselves but on the approaches. In the report just cited, the lighter traffic on the Brooklyn, Manhattan and Williamsburg Bridges (which connect Brooklyn and Manhattan) is estimated to boost vehicle speeds on the three bridges by an average of more than 11 percent while also raising speeds in contiguous parts of the two boroughs by 2 to 3 percent.[4] Speed improvements on and near the Queensboro Bridge would be roughly half as great. These higher speeds would translate into lower emission rates per mile, an effect that we estimated using the speed-emission relationships built into the MOBILE6 model (see discussion in Section 4).

Combining the two phenomena of reduced VMT (as estimated in the 2003 report cited above) and higher speeds (with lower emission factors estimated using MOBILE6), we calculate that East River bridge tolls would reduce $PM_{2.5}$ emissions by 4 tons a year, or 0.8 percent of the 2010 baseline. (The improved speeds evidently add 0.2 percent to the 0.6 percent reduction attributable to reduced VMT.) Thus, bridge tolls reduce $PM_{2.5}$ emissions by a relatively small amount.

The tolls would generate approximately $800 million a year, of which only $50 million would be consumed by administrative costs, leaving $750 million a year that could be applied to transit and other transportation improvements. Still, bridge tolls would reduce $PM_{2.5}$ emissions, and the reductions in notorious air pollution corridors such as downtown Brooklyn, Williamsburg, Long Island City, and Manhattan's Lower East Side could be impressive.

6.1.3 "Cordon Pricing" Around Manhattan's Central Business District

An alternative or complementary road-pricing strategy is so-called "cordon pricing," whereby an electronic toll-collection system would ring the entire Manhattan Central Business District (CBD) from 60th Street to the Battery. Cordon pricing would thus toll all motor vehicle entry from the north (i.e., Upper Manhattan or the Bronx) rather than

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[3] C. Komanoff and B. Ketcham, *The Hours: Time Savings from Tolling the East River Bridges,* Bridge Tolls Advocacy Project, 2003, available at http://www.bridgetolls.org/thehours/thehours.htm.

[4] Ibid.

just vehicles crossing the East River to Manhattan's East Side. (Note that the Port Authority already tolls the portals to the CBD from New Jersey to the west.[5])

Cordon pricing appears to have several advantages over East River bridge tolls. It tolls all entry to the CBD rather than simply passage over the four East River bridges — a potentially vital attribute in light of intense resistance to East River tolls by officials representing Brooklyn and Queens. Cordon pricing also would raise considerably more revenue — an estimated $2.0 billion annually, by our calculations, or 150 percent more than East River tolls alone — due to the greater number of trips that would be tolled. Moreover, cordon pricing has already been proven in London, where a roughly $9.00 (£5) entry fee to central London in effect since early 2003 has greatly reduced traffic congestion, improved bus travel times and produced other acknowledged public benefits.[6]

Working against cordon pricing is the fact that the proposed 60th Street (or other, similar) cordon line lacks the natural inevitability of a river crossing or other crossing point and may also provoke opposition on grounds of visual acceptability. Both cordon pricing and East River tolls have the drawback of giving a "free pass" to vehicular travel *within* the CBD, which may prove disadvantageous in terms not only of traffic logistics and revenue loss, but political acceptability as well.

Road pricing proponents tend to be indifferent between East River or cordon tolls. The two systems are conceptually similar, of course, and the former, if it could clear the political hurdles, could become a proving ground for the latter. Moreover, either system could be a gateway to a comprehensive, citywide pricing regime in which all vehicular travel is tolled proportionate to real-time congestion levels.

To estimate the effect of cordon pricing on $PM_{2.5}$ on-road emissions, we extrapolated from a detailed 2003 study by Jeff Zupan and Alexis Perrotta, highly-regarded transportation scholars at the Regional Plan Association.[7] Their analysis indicates that free crossings into the CBD from the north exceed those from Brooklyn and Queens by just over 50 percent (390,000 daily vs. 255,000), suggesting that cordon pricing would reduce trips and VMT by 2.5 times as much as East River bridge tolls alone. Inputting that factor to the same methodologies from the previous section, we estimate that pricing CBD entry at the same rate as other NYC tolls ($8.00 per round-trip) would lead to a drop in $PM_{2.5}$ emissions of 10 tons a year (or, not surprisingly, 2.5 times as much as the 4-ton reduction expected from bridge tolls). This is just under 2 percent of baseline on-road emissions projected for 2010.

[5] To avoid double-tolling, cordon pricing would be electronically mediated to exempt trips already tolled at the George Washington Bridge.

[6] Copious empirical data on London's cordon pricing scheme are available at the Transport for London Web site, http://www.tfl.gov.uk.

[7] J. Zupan & A. Perrotta, *An Exploration of Motor Vehicle Congestion Pricing in New York*, Regional Plan Association, New York, NY, available at www.rpa.org.

6.1.4 Weight-Distance Fee on Vehicle Travel within NYC

This control measure charges all driving within the five boroughs at a rate proportional to the weight of the vehicle. We analyzed a fee level that equates to 5¢ per mile for a conventional sedan weighing 3,400 pounds. Thus, a standard 5,100-lb SUV would pay 50 percent more, or 7.5¢ for each mile driven, while a large, 6,800-lb SUV or pickup would pay 10 cents. The per-mile rates would rise proportionally so that a 34-ton truck would pay 20 times the rate of the sedan, or $1 per mile driven.

The intent of weight-distance fees is to both discourage unnecessary driving and encourage a shift to lighter vehicles with lower emission rates and smaller societal and environmental impacts. Charging vehicles on a graduated ton-mile basis rather than via a flat per-mile fee nicely captures many of the societal costs associated with driving — "externality costs" such as air and noise pollution, dependence on imported fossil fuels and endangerment of other road users, as well as direct governmental costs to provide, maintain and police roads. Compared to a 3,400-pound sedan, a 5,100-pound SUV is noisier, more demanding of fuel, more air-polluting, more consuming of road space, and more likely to injure or kill other vehicle users. Similarly, compared to a car or smaller truck, an 18-wheeler inflicts vastly more damage on public health, other travelers' time, the road infrastructure and the natural environment.

The fees we have assumed are not trivial. Assuming a nominal 20 miles per gallon, they would take the same bite from drivers' pocketbooks as a $1.00 per gallon increase in the price of gas for the conventional sedan, $1.50 per gallon for the standard SUV and $2.00 per gallon for the large SUV or pickup. The fees we envision here, applied citywide, would raise approximately $2.0 billion per year in revenue, implying a very substantial change in societal governance of motor vehicle usage. (Different rates could be examined as well, of course, and their $PM_{2.5}$ reduction impacts would be roughly proportional to that shown here.)

Like East River bridge tolls and CBD cordon pricing, a citywide weight-distance fee would discourage non-essential motor vehicle travel and produce a similar one-two punch against $PM_{2.5}$ emissions, namely fewer tailpipes and improved traffic flow. To estimate these effects, we posited a standard 12-mile trip for which we calculated typical out-of-pocket costs for gas, parking, tolls, and incremental depreciation, maintenance and insurance for each of the 28 vehicle classes. Applied to this cost base, the weight-distance fees would increase average trip costs by anywhere from 19 percent for sedans and 25 percent for SUVs, up to a whopping 76 percent for 18-wheeler diesel trucks. We then ascribed price-elasticities to each vehicle class, ranging from a high of -0.20 for passenger vehicles to -0.11 for heavy trucks and -0.05 for buses (the latter reflects the pass-along of the fees into higher fares).[8]

[8] We have developed these price-elasticities based on our experience in analyzing relationships of trip-making to trip price in NYC. In *The Hours* (op. cit.), we derived a price-elasticity of (negative) 0.3 for all East River crossings during 1975 to 2000. Using professional judgment, we have reduced this figure to 0.2 here to reflect "induced travel" — new trips taken to exploit the road space made available by the first-order reduction in trips due to tolls. We further reduced that figure for progressively larger vehicles to reflect the less-discretionary nature of trips by freight vehicles and buses.

These assumed elasticities are quite low, reflecting both empirical and anecdotal evidence that even large increases in the cost to drive yield only modest decreases in the number of trips taken or miles driven. (That is, “getting drivers out of their cars” is difficult even in NYC where transit and other travel alternatives are relatively abundant; trips taken by motor vehicle tend to be those that have survived a sort of natural selection through the hardships of congested traffic, costly parking and general driver aggravation.) The very low elasticities assumed for heavy trucks are intended to capture the paucity of alternatives for moving freight (weight-distance fees would probably lead to reduced truck VMT more through improved trucking logistics rather than via mode shift to rail).

These assumptions imply a range of responses to the weight-distance fees. The estimated reductions in VMT range from just 2.4 percent for transit buses to 3.4 percent for sedans, 4.3 percent for SUVs and 6.0 percent for 18-wheelers, with an overall decrease in VMT by all vehicles of 3.8 percent. Applying MOBILE6 to the resulting revised set of VMT estimates for each vehicle type and to the revised speed data sets for each borough, we estimate that implementing weight-distance fees on all motor vehicle travel in NYC would cause on-road $PM_{2.5}$ emissions to decline by 25 tons per year, or 4.7 percent of the 2010 baseline. (The percentage drop in emissions exceeds the drop in VMT because the improved traffic flows resulting from the lesser VMT translate to better engine efficiencies and reduced emissions.)

However, these fees also provide some of the most expensive $PM_{2.5}$ reductions — $6,000 per pound — due to the relatively high costs of the required two sets of complementary tolling equipment. First, all vehicles driven in New York City would have to be equipped with a GPS device to register its movement; we have assumed 5 million vehicles with a one-time per-vehicle cost of $100 amortized over 10 years, which translates to an annualized cost of $59 million, assuming a 3-percent real cost of capital. (Drivers would have the option of paying high per-diem rates as an alternative.) Second, all NYC streets and highways would need to be outfitted with electronics to register vehicle mileages, at an approximate cost of $500 million (5-10 times the respective costs for the more geographically localized East River bridge tolling and Manhattan CBD cordon charging). This translates to $34 million, assuming a 20-year life. Allowing $200 million for annual administrative costs brings the total per-year cost of a weight-distance fee system to roughly $300 million, which equates to $12 million per ton or $6,000 per pound of $PM_{2.5}$ eliminated.

6.1.5 Weight-Distance Fees on Truck Travel Only

It is almost certainly more politically palatable to limit road pricing to trucks and to exempt passenger vehicles than to charge all motor vehicle travel. Moreover, since heavy trucks account for a disproportionate share of $PM_{2.5}$ emissions (in particular, 18-wheeler trucks with diesel engines — vehicle category HDDV8B in MOBILE6 parlance — emit 12 percent of all on-road $PM_{2.5}$ while racking up only 1 percent of citywide VMT), applying weight-distance fees only to freight vehicles would seem a surefire way to get more reductions in emissions for each dollar charged in tolls.

Surprisingly, however, this does not appear to be the case. In our modeling, we found that a weight-distance fee limited to trucks 9,250 pounds and up, priced to raise the same

$2.0 billion in annual revenue as the universal weight-distance fee just described, would reduce $PM_{2.5}$ emissions by just 19 tons, vs. 25 tons for the universal fee.[9] (This truck-limited weight-distance fee would need to be priced 5-6 times as high as the universal fee to raise the same revenue, because the VMT base is that many times smaller.) On the other hand, a truck-only weight-distance fee would cost less than the universal weight-distance fee per pound of $PM_{2.5}$ eliminated, $4,000/lb vs. $6,000/lb, respectively. This is because the in-vehicle electronics that would be required to record miles driven would be needed on far fewer vehicles in the trucks-only case.

Table 6.1: Summary of Weight-Distance Fees*

	Weight (lbs)	Fee ($/mi)	Elasticity	Cost Increase	VMT reduction	Revenue
Universal fee: all vehicles						
Cars	3,400	$0.05	-0.20	19%	3.4%	$763M
Lt Trucks 1	4,600	$0.07	-0.20	25%	4.3%	$102M
Lt Trucks 2	5,600	$0.08	-0.20	27%	4.7%	$414M
Transit bus	40,000	$0.59	-0.05	61%	2.4%	$112M
18-wheelers	70,000	$1.03	-0.11	76%	6.0%	$224M
Trucks: only freight vehicle classes > 9,250 lb are charged						
18-wheelers	70,000	$5.76	-0.11	425%	16.7%	$1,112M

*All vehicles are charged the same amount per ton-mile. Only 5 of the 28 vehicle classes are shown here, for brevity. Both sets of fees (universal and trucks) are structured to raise approximately $2.0 billion in annual revenues. Cost column denotes increase in average out-of-pocket cost of typical trip and reflects "embedded" costs for fuel, maintenance, tolls, parking, etc. as well as hypothesized weight-distance charges. VMT reduction is calculated from two prior columns, per elasticity relationship: $Q_1 / Q_0 = K \times (P_1 / P_0)^E$, where Q_0 and P_0 are the current quantity and price of the product (a typical travel trip in NYC), K is a constant, Q_1 and P_1 are the altered quantity and price (with weight-distance fees), and E is the price-elasticity.

This counter-intuitive result has two complementary explanations. First, our exemption of buses places a highly polluting class of vehicles off-limits. Second is the low price sensitivity of most truck travel, reflected in our assumed elasticities of -0.11 to -0.18 to truck VMT, vs. -0.20 for cars and light trucks. This differential implies that a dollar raised by charging trucks tends to produce a lesser change in travel (and, hence, a smaller drop in $PM_{2.5}$) than a dollar raised by charging a combination of cars and trucks.

In general, then, road pricing targeted only at trucks is better at raising revenue but not as effective at reducing pollution and traffic as universal road pricing. Still, the reductions in truck travel estimated here are impressive (16 to 18 percent), and suggest that more modest truck-only weight distance fees might be an acceptable gateway to a universal pricing regime. In particular, insofar as any weight-distance fee system would likely require vehicles driven in NYC to deploy a uniform GPS device to "meter" their miles traveled, it would be far less daunting to launch such a system with several hundred thousand trucks than with several million passenger vehicles.

[9] The class of trucks 9,250 pounds and greater includes vehicle classes HDGV2B or HDDV2B and above. Buses are exempted along with SUVs and so-called light trucks).

6.1.6 Gasoline Taxes

The most direct way to use economic disincentives to reduce fuel use is to tax fuel. We therefore examined a $1.50 per gallon increase in the prices of gasoline and diesel fuel, administered as part of a program to reduce greenhouse gas emissions through carbon taxes. The $1.50 per gallon rate matches the assumed tax on heating oil that we analyze in Section 6.4.4. The aggregate revenue from a $1.50 per gallon tax increase on motor fuels, an estimated $2.0 billion a year, matches the revenues calculated for the weight-distance fees considered directly above, allowing for a consistent comparison.

A tax increase on motor fuels sold in New York City could be enacted either as a NYC-only fee or as part of a statewide tax increase. In either case, approval by the state legislature would be required. “Border” problems would presumably arise as drivers sought out untaxed or lower-tax adjacent counties or states for cheaper fuel. This suggests that carbon taxes or other means of raising vehicular fuel prices would need to be phased in gradually, or developed on a statewide or regional basis, or both. In any case, this strategy is assumed to have zero costs since the higher fuel costs would be paid through the existing gas/diesel fuel tax system and would thus require no new administrative mechanisms.

To estimate the impact of higher gasoline and diesel fuel taxes on $PM_{2.5}$ emissions, we employed the methodology we used for weight-distance fees, applying the same set of price-elasticities to typical 12-mile trips for each vehicle class. The $1.50-per-gallon tax increases imply 24 percent to 33 percent increases in total out-of-pocket trip costs. These in turn would be expected to bring about 3 to 5 percent decreases in VMT for the various vehicle classes (but just 1.3 percent for buses, due to our assumption of much lower price elasticities), with a 4.5 percent overall drop in VMT. Truck travel is affected less (due to smaller price-elasticity), which works against emission reductions. On the other hand, the reduced VMT improves traffic flow somewhat, which enhances emission reductions. We also increased the emission results by 25 percent to reflect the expectation that the higher fuel taxes would induce a shift to more fuel-efficient vehicles, adding to the reduction in emissions.

The net effect of these cross-cutting factors is a calculated 25 ton reduction in $PM_{2.5}$ emissions in 2010 from the higher fuel tax, equivalent to just under 5 percent of baseline emissions for that year. Coincidentally, this matches the drop in emissions estimated for the universal weight-distance fee discussed earlier.

6.1.7 Per-Mile Insurance Pricing

Standard auto insurance contracts require that drivers purchase insurance on a lump-sum basis, so their premiums bear little relationship to the number of miles they drive. Under an emerging alternative approach, insurance premiums would be priced to be largely proportional to mileage. A driver’s insurance rate would be so many cents per mile rather than so many dollars per year, and drivers would be billed at the product of that insurance rate times their miles driven.

Each driver’s insurance rate would reflect his or her safety record (of accidents and infractions). Rates would therefore vary widely, from a few cents per mile for the least accident-prone drivers to as much as 20 cents for very unsafe drivers. In the New York area, the average rate would be around 11 cents a mile, and we have used this figure in our analysis.[10]

The idea behind per-mile insurance is to rearrange, and possibly even reduce, the cost of car use by shifting to a *variable* cost the insurance payments that drivers now pay as a *fixed* cost. This cost shift would create a powerful incentive to economize on driving among drivers of non-commercial vehicles — the assumed target group for per-mile insurance. Whereas drivers currently save little or nothing on insurance by driving less, per-mile insurance would effectively let them pocket up to 11 cents on average for each mile less that they drove.[11]

NYC drivers who sign up for per-mile insurance would thus perceive a considerable increase in their out-of-pocket costs to drive — as much as a 32 percent increase for sedan drivers, with lesser but still sizeable increases for other vehicle classes. (Of course, their *annual* insurance costs would be unchanged; indeed, these might decline somewhat as drivers internalized the new incentives and drove both more safely and less often.) Based on the elasticity methodology employed here for other control measures, this should translate to a substantial decline in miles driven — 5.4 percent for sedan owners opting for per-mile insurance, for example — with a commensurate drop in $PM_{2.5}$ emissions.

In light of the emerging status of per-mile insurance pricing, we tempered this analysis in two respects: first, we assumed that per-mile insurance would only be available for passenger vehicles; second, we assumed market penetration of 50 percent (rather than 100 percent). With these assumptions and the elasticity of passenger car driving used above (-0.20), we calculate that per-mile insurance would reduce citywide VMT by 2.4 percent overall. The associated drop in $PM_{2.5}$ emissions, 11.6 tons per year, is 2.2 percent of baseline 2010 emissions.

Costs for this strategy are largely for the mileage tracking devices that would need to be placed in each vehicle qualifying for per-mile pricing. We estimated a per-vehicle cost of $100 for an insurance-specific device that recorded miles driven along with geographic data of interest to the insurance provider.[12] Applying this cost to an estimated 2.5 million vehicles that account for roughly half of NYC VMT, with a 10-year equipment lifetime, 3 percent real cost of capital, and $2/month per-vehicle cost for billing and other

[10] New York State drivers paid an average of $1,087 per automobile for insurance in 2002, which we increased by 12.5 percent to $1,223 to adjust to 2005. See table, Average Expenditures for Auto Insurance by State, 1998-2002, in http://www.iii.org/media/facts/statsbyissue/auto/. Spreading those dollars over an assumed 11,000 miles driven per vehicle per year yields 11 cents/mile.

[11] We estimate that drivers would perceive 75 percent of their insurance costs as varying with miles driven and, therefore, as avoidable with each mile *not* driven. The percentage is less than 100 percent because the theft component of actuarial risk varies little if at all with vehicle use, and because some drivers may have already internalized a connection (albeit a modest one) between their miles driven and crash risk.

[12] Personal communication with William Hale, Director of New Business Development, Intelligent Mechatronic Systems, Inc., Waterloo, Ontario (a manufacturer of the devices described in the text), June 26, 2005.

administrative tasks, results in an estimated annualized cost of around $90 million for this strategy, which equates to just under $4,000 per pound of $PM_{2.5}$ eliminated.

Here, as for the other pricing strategies we analyzed, the primary benefits are not emission reductions *per se* but enhanced transportation efficiency arising from the improved alignment of drivers' price incentives with the actual societal costs of motor vehicle use. In the case of per-mile insurance there would be the additional benefit of greater fairness, as drivers' premiums more closely reflected their true actuarial costs.

6.1.8 Summary of On-Road Control Options

Table 6.2 summarizes the control options discussed above. Potential reductions are characterized as "Low, Moderate or High," with "Low" referring to reductions in the range of zero to 20 tons per year, "Moderate," in the range of 20 to 100 tons per year and "High" over 100 tons per year. Our cost estimates per pound of $PM_{2.5}$ removed are also shown (rounded to the nearest five dollars). Reductions in other pollutants and efficiency gains are also characterized as Low, Moderate or High, but the terms do not refer to specific ranges in these cases. Many of these options, such as fuel taxes and tolling, produce considerable revenue, however this revenue is considered to be a transfer payment and is not included in cost calculations. That is, cost shown in the table include only real resource costs.

The lowest cost $PM_{2.5}$ reductions available in the on-road sector are from installing controls on the remaining uncontrolled buses that operate in NYC. These are buses other than those owned by the Transit Authority and private bus companies subsidized by New York City DOT. A tax of $1.50 per gallon on gasoline and diesel fuel offers moderate reductions at no implementation cost. Beyond this, further reductions from this sector become quite expensive, as large numbers of cars would be affected and reductions from each vehicle would be small. A number of the strategies we examine would result in large transfer payments from consumers to the government, which could be used to reduce other taxes or transportation costs. Costs shown in Table 6.2 include only program implementation costs, not transfer payments.

Table 6.2: Summary of On-Road Control Options

	Potential Reductions (tons/yr)	Reductions in Other Pollutants	Efficiency Gains	Other Benefits	Cost ($/lb $PM_{2.5}$)
Controls for "Other" Buses	Moderate	Moderate	None	None	$35
East River Tolls	Low	Low	None	Reduced Congestion	$6,250
CBD Cordon Tolls	Low	Low	None	Reduced Congestion	$5,000
Weight-Distance Fees	Moderate	Low	None	Reduced Congestion	$6,000
Weight-Distance (Trucks Only)	Moderate	Low	None	Reduced Congestion	$4,000
Fuel Taxes	Moderate	Moderate	Moderate	Reduced Congestion	$0
Per-Mile Insurance	Low	Low	None	Reduced Congestion	$4,000

6.2 Non-Road Mobile Sources

EPA regulations now in place will greatly decrease $PM_{2.5}$ emissions from non-road mobile sources over the next ten years. But non-road equipment has a long lifetime, and the existing stock will continue to be used for many years to come. Additional emission reductions from this category can best be obtained by programs that require or encourage the installation of pollution control devices on existing non-road equipment. The focus of these programs is primarily diesel engines. The vast majority (97 percent) of PM from non-road diesels is $PM_{2.5}$, so targeting PM from this sector is an effective way to reduce fine PM.[13]

New York City enacted a law in 2003 that requires ultra-low sulfur fuel and emission controls on diesel non-road construction equipment used under city contracts.[14] This measure embodies a strategy of using city contracting specifications to increase the numbers of PM pollution control devices on non-road construction equipment available for use on both city contracts and private contracts.

DEP regulations to implement the new law were issued in April, 2005, and immediately sparked controversy over the likelihood of their effectiveness.[15] The control technology that the program requires on any particular piece of equipment depends on what technologies are available for that equipment and the cost of the technology in relation to the age of the equipment. In general, the program requires use of the most effective technology certified for use on each piece of equipment as long as the cost difference between that technology and the next most effective technology is no more than an amount that decreases with the age of the vehicle. That is, older vehicles can use less expensive and less effective technologies.

An assessment of the actual effectiveness of the new law will have to await the annual reports the law requires DEP to file with City Council. Those reports will detail both the number of engines affected and the pollution control devices employed.

Recently the City extended the concept of the 2003 law to a new law requiring ultra-low-sulfur fuel and emission controls on diesel non-road equipment used by city contractors while handling, transporting, or recycling solid waste.[16] Again the effectiveness of the new law will not be known for some time.

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[13] EPA, "Exhaust and Crankcase Emission Factors for Nonroad Engine Modeling — Compression-Ignition," EPA420-P-04-009, April 2004, Report NR-009c, p. 23.

[14] Local Law 77 of 2003.

[15] Anthony DePalma, "New City Pollution Rules Face Trouble From Outset," New York Times, March 28, 2005; "Diesel Wars," New York Times Editorial, April 10, 2005; Emily Lloyd, Commissioner, DEP, "Diesel Regulation," Letter to the Editor, New York Times, April 24, 2005.

[16] Local Law 40 of 2005.

6.2.1 Control Technologies

DEP regulations implementing the city's 2003 law rely on lists of approved PM control devices compiled by EPA and the California Air Resources Board (CARB). EPA and CARB have agreed to coordinate their verification programs and lists. At the present time six control measures have been approved by either EPA or CARB:

- diesel particulate filters,
- diesel oxidation catalysts,
- catalyzed flow-through filters,
- fuel-water emulsions,
- biodiesel, and
- fuel-borne catalysts.

Hardware control measures consist of the diesel particulate filter, including both passive and active versions, and the diesel oxidation catalyst. Each of these technology types has been used in both on- and off-road vehicles and equipment for many years. More recently, another device, a catalyzed wire mesh filter, also known as a flow-through filter (FTF), was developed and verified. Fuel-based control measures include fuel-borne catalysts and biodiesel.[17]

Diesel Particulate Filter

A passive diesel particulate filter (DPF) reduces PM, and catalyzed DPFs also reduce carbon monoxide (CO) and unburned hydrocarbons (HC) emissions through catalytic oxidation and filtration. Most DPFs sold in the United States use substrates consisting either of a ceramic wall-flow monolith or a silicon carbide substrate.

The filter is positioned in the exhaust stream to trap or collect a significant fraction of the particulate emissions while allowing the exhaust gases to pass through the system. Effective operation of a DPF requires a balance between PM collection and PM oxidation, or regeneration. The volume of PM generated by a diesel engine will fill up and plug a DPF over time; thus the trapped PM must be burned off or "regenerated" periodically. Regeneration is accomplished by either raising the exhaust gas temperature or by lowering the PM ignition temperature through the use of a catalyst.

The type of filter technology that uses a catalyst to lower the PM ignition temperature is termed a passive DPF, because no outside source of energy is required for regeneration. Verified passive DPFs have demonstrated PM reductions in excess of 90 percent. A passive catalyzed DPF also reduces CO and HC by approximately the same amount as the PM reduction. A passive catalyzed DPF is a very attractive means of reducing diesel PM emissions because of the combination of high reductions in PM emissions and minimal operation and maintenance requirements.

An active DPF system uses an external source of heat to oxidize the PM. The most common methods of generating additional heat for oxidation involve electrical

[17] Although bio-diesel is included on EPA's list of approved control measures for diesel engines, it is not included on the CARB or New York City DEP lists because of the low degree of PM reduction.

regeneration by passing a current through the filter medium, injecting and burning additional fuel to provide additional heat for particle oxidation, or adding a fuel-borne catalyst or other reagent to initiate regeneration.

Some active DPFs induce regeneration automatically on the vehicle or equipment when a specified backpressure is reached. Others use an indicator, such as a warning light, to alert the operator that regeneration is needed, and require the operator to initiate the regeneration process. Some active systems collect and store diesel PM over the course of a full shift and are regenerated at the end of the shift with the vehicle or equipment shut off. A number of the filters are removed and regenerated externally at a regeneration station.

For applications in which the exhaust PM is relatively high, and/or the exhaust temperature is relatively cool, actively regenerating systems may be more effective than a passive DPF. Because active DPFs are not dependent on the heat carried in the exhaust for regeneration, they potentially have a broader range of application than passive DPFs.

Diesel Oxidation Catalyst

A diesel oxidation catalyst (DOC) reduces emissions of CO, HC, and the soluble organic fraction of diesel PM through catalytic oxidation alone. Exhaust gases are not filtered in DOCs. In the presence of catalytic material and oxygen, CO, HC, and the soluble organic fraction of the PM undergo a chemical reaction and are converted into carbon dioxide and water. Some manufacturers integrate HC traps (zeolites) and sulfate suppressants into their oxidation catalysts. HC traps enhance HC reduction efficiency at lower exhaust temperatures and sulfate suppressants minimize the generation of sulfates at higher exhaust temperatures. A DOC may reduce total PM emissions by up to 30 percent.

Catalyzed Wire Mesh Flow Through Filter

Flow-through filters (FTF) employ a catalyzed wire mesh substrate that has an intermix of flow channels creating turbulent flow conditions. Unlike a DPF, in which only gases can pass through the substrate, the FTF does not physically trap and accumulate PM. Instead, it finctions like a DOC but achieves a greater PM reduction due to enhanced contact of PM with catalytic surfaces and longer residence times. Any particles that are not oxidized within the FTF flow out with the rest of the exhaust and do not accumulate. Consequently, the filtration efficiency of an FTF is lower than that of a DPF, but the FTF is much less susceptible to plugging because of high PM emissions and low exhaust temperatures. Therefore, this type of filter may be suitable for specific duty cycles where a typical DPF would not be applicable.

Fuel-Water Emulsion

An emulsion of diesel fuel and water is a demonstrated alternative to diesel fuel that reduces both PM and NO_x emissions. The process blends water into diesel fuel along with an additive to keep the mixture from separating. The water is suspended in droplets within the fuel, creating a cooling effect on the fuel that decreases NO_x emissions. A fuel-water emulsion creates a leaner fuel environment in the engine, lowering PM emissions as well. An example is Lubrizol's PuriNOx™**,** an emulsified diesel fuel that achieves at least 50 percent reduction in PM and 15 percent reduction of NO_x.

Fuel Additives

A fuel additive is a substance added to fuel or fuel system so that it is present in the cylinder during combustion and its addition causes a reduction in exhaust emissions. Additives can reduce the total mass of PM, with variable effects on PM, CO, NO_x and gaseous HC production. Published studies of fuel additives indicate a range of PM reductions of 15 to 50 percent. Most additives are fairly insensitive to fuel sulfur content and will work with a range of sulfur concentrations as well as with different fuels and other fuel additives.

A fuel-borne catalyst (FBC) is a substance that is added to diesel fuel in order to aid in soot oxidation in DPFs by decreasing the ignition temperature of solid carbon. An FBC can be used in conjunction with both passive and active filter systems to aid system performance, and decrease mass PM emissions. FBC/DPF systems are in wide spread use in Europe in both on-road and off-road, mobile and stationary applications and typically achieve $PM_{2.5}$ reductions of approximately 90 percent.

Biodiesel

Biodiesel is a mono-alkyl ester-based oxygenated fuel made from vegetable oils, such as oilseed plants or used vegetable oil, or animal fats. It has similar properties to petroleum-based diesel fuel, and can be blended into petroleum-based diesel fuel at any ratio. B20 is a 20% blend of biodiesel into petroleum-based diesel fuel. Pure biodiesel is called B100.

EPA analyzed the impacts of biodiesel on exhaust emissions from heavy-duty on-road engines. While biodiesel and biodiesel blends decrease PM emissions, as well as HC and CO, NO_x emissions increase proportionally with the increase of biodiesel fraction. For B20, the NO_x increase is reported to be two percent, with reductions of 10 percent PM, 21 percent HC, and 11 percent CO. In addition, the EPA predicts that a B20 blend would reduce fuel economy by one to two percent. The impact of biodiesel on emissions varies depending on the type of biodiesel (soybean, rapeseed, or animal fats) and the quality of the diesel fuel used in biodiesel blends.

6.2.2 Control Measures and PM Constituents

As discussed in Section 2, PM from diesel engines is sub-divided into three distinct components: a solid fraction, a soluble organic fraction and sulfates. Table 6.3 shows the composition of PM from a post-1994 diesel engine using fuel with a medium sulfur content (about 325 ppm). Note that over 95 percent of PM from diesel engines is $PM_{2.5}$.

Table 6.3: Components of Particulate Matter

Solids		SOF		SO_4
Carbon	Ash	Fuel	Lubricants	
41%	13%	7%	25%	14%
54%		32%		

A general understanding of these three fractions serves as a guide in selecting the proper retrofit technology for a specific application.

The solid fraction of diesel PM is often referred to as "elemental carbon" ("EC") or "black carbon" and also includes ash deposits. The solid fraction of total PM is the primary source of the black smoke associated with heavy-duty diesel engines. Ash, on the other hand, primarily emanates from lubricating oil and metals due to engine wear. As newer engines are developed to produce less EC, the proportion of ash in the exhaust tends to increase. High levels of ash are removed effectively by diesel particulate filters (DPFs) through physical entrapment. However, ash presents two significant issues — it is quite corrosive and can deteriorate DPF filter material if the DPF is not properly designed, and it cannot be completely removed from the DPF by the regeneration process (explained below), requiring periodic DPF removal and cleaning by hand.

The soluble organic fraction is composed of organic material from engine fuel and lubrication oil. SOF consists essentially of hydrocarbon deposited on the surface of elemental carbon (EC) particles and is often referred to as "wet PM." SOF formation is dependent on the duty cycle, with light load causing higher concentrations because of low engine temperature. This is especially significant when considering diesel oxidation catalysts (DOCs) as a candidate retrofit device, since they are effective in reducing only the SOF portion of diesel PM. As emission regulations prompt generally higher operating temperatures in newer engines, SOF emissions decline.

Sulfate is a combustion by-product that combines the sulfur in diesel fuel with water vapor, to form sulfuric acid mist. Reaction with ammonia in the ambient air subsequently forms ammonium sulfate or bisulfate particles. Sulfate formation may also occur as an unintended by-product of the oxidation of sulfur dioxide (SO_2) in a DOC. This chemical reaction, frequently termed "sulfate make," adds to the PM emissions from an engine, adversely affecting DOC conversion efficiency.

Certain retrofit technologies are only capable of reducing some, but not all of these components of PM. The more sophisticated technologies are able to reduce both the solid fraction and SOF portion (catalyzed diesel particulate filters, for example), but at greater cost.

6.2.3 Retrofitting Additional Non-Road Equipment

The City's strategy of requiring ultra-low sulfur fuel and emission controls on categories of non-road, as well as on-road, diesel equipment appears to be sound, assuming that implementation is effective.

Requiring controls on additional categories of non-road equipment should be considered. Based on our analysis, the most promising additional category may be refrigeration equipment carried on trucks that make deliveries throughout the city. Emissions from truck-based refrigeration equipment accounted for approximately 106 tons of $PM_{2.5}$ in 2004.

Generally, larger models of truck-based refrigeration units use diesel engines that operate independently of the truck engine and continue to run even when the truck engine is shut off. Thus trucks making deliveries will continue to emit $PM_{2.5}$ even when parked, regardless of idling restrictions. Trucks of this type have recently been generating public controversy.[18]

Another promising category is mobile electric generating equipment. This category accounted for roughly 160 tons of $PM_{2.5}$ emissions in 2004.

The feasibility of a retrofit program aimed at truck-based refrigeration units or mobile generating equipment depends on the city's or state's legal ability to adopt regulations. The actual result of such a program is difficult to gauge, since effectiveness depends on how the city or state implements the program, the technologies that are certified by EPA or CARB for the many engine and equipment types, and the costs of the different technologies. Evaluation of these retrofit control measures assumes a program that allows for a combination of retrofit control technologies to match the variety of existing engines.

A recent CARB study comparing current retrofit technologies provides some indication of the costs and benefits.[19] According to CARB, first-year costs of control technologies vary from \$7,600 for a DPF, to \$4,000 – \$5,000 for FTF or emulsified fuel, to \$2,000 for a DOC. Operation and maintenance costs range from \$33 per year for cleaning a passive DPF to \$157 per year for additional cost of PuriNOx™ fuel, to no O&M cost for a flow-through filter or a DOC. As an average, we can assume an initial capital cost of \$5,000 per engine and an annual O&M cost of \$48, making annual costs per engine roughly \$760.[20] However EPA expects the cost of DPFs and DOCs to drop substantially in two years as production volumes increase after the introduction of 2007 model-year engines that will require DPFs to meet emission standards.

A retrofit program implemented today that reached all vehicles in a category by 2010 (e.g. truck-based refrigeration units, or mobile electric generating equipment) might be expected on average to reduce PM emissions from engines in that category by about 50 percent at a cost of approximately \$160 per pound of PM reduced. While there would be some reduction in hydrocarbon and NO_x emissions as well, there would be little or no gain in efficiency and few other economic benefits.

6.2.4 Summary of Non-Road Control Options

Current EPA and NYC regulations will reduce $PM_{2.5}$ emissions from non-road vehicles dramatically. Non-road $PM_{2.5}$ emissions are expected to decrease by about 20 percent by 2010 and by about 40 percent by 2015, relative to 2005 levels. These regulations target the sources responsible for the vast majority of non-road emissions – construction and commercial equipment. The most significant categories of engines that remain unregulated are refrigerated trucks and small electric generators. Refrigerated trucks

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[18] Elizabeth Royte, "Pollution in a Box," New York Times, July 31, 2005; Dean Furbush, "FreshDirect and the Environment," Letter to the Editor of the New York Times, August 7, 2005; Ruby Patel, "Groceries, Boxes, and FreshDirect," Letter to the Editor of the New York Times, August 14, 2005.

[19] CARB, "Proposed Diesel Particulate Matter Control Measure for On-Road Heavy-Duty Diesel-Fueled Vehicles Owned or Operated by Public Agencies and Utilities," October 21, 2005.

[20] Annualized costs are based on a ten-year lifetime and a 7% interest rate. Annualized Cost = (CRF · CC) + O&M.

emitted just over 100 tons of $PM_{2.5}$ in 2004, and mobile electric generators emitted roughly 160 tons. Based on the cost and efficiency of the available control technologies, we estimate that cutting emissions from these two sectors in half would cost roughly $160 per pound reduced.

Table 6.4: Summary of Non-Road Control Options

Strategy	Potential Reductions (tons/yr)	Reductions in Other Pollutants	Efficiency Gains	Other Benefits	Cost ($/lb $PM_{2.5}$)
Refrigerated Trucks	Moderate	Moderate	None	None	$160
Mobile Generators	Moderate	Moderate	None	None	$160

Potential reductions are characterized as "Low, Moderate or High," with "Low" referring to reductions in the range of zero to 20 tons per year, "Moderate," in the range of 20 to 100 tons per year and "High" over 100 tons per year. Cost estimates per pound of $PM_{2.5}$ removed are rounded to the nearest five dollars.

6.3 Point Sources

As discussed in Section 4, $PM_{2.5}$ emissions from point sources in NYC come primarily from large energy facilities – power and steam plants. This sector is responsible for 79 percent of point source emissions, followed by residential point sources, responsible for only 6 percent. Thus, the large energy facilities are clearly the point of leverage in this sector.

The strategies available for reducing $PM_{2.5}$ from these sources focus on increasing the efficiency of energy production in the City and switching to fuels that emit less $PM_{2.5}$. While emission control systems targeting $PM_{2.5}$ are commonly installed on coal-fired power plants, they are not generally installed on gas- and oil-fired plants, such as those in NYC. The strategies discussed below are: end-use efficiency, combined heat and power production, repowering, and fuel switching.

It is important to note that, in addition to these strategies, the natural process of plant turnover also provides valuable emission reductions, as older, less efficient plants are replaced by more efficient, cleaner burning plants. For example, NYPA is currently replacing the old Poletti power plant with a new combined-cycle plant, a project that will provide much needed emission reductions in a heavily industrialized area of Queens. As noted above, Reliant Energy has proposed replacing the old Astoria plant as well.

6.3.1 Electric Energy Efficiency

The efficiency of many electricity end uses could be improved significantly. That is, less electricity could be used to provide the same service (e.g., lighting, refrigeration, etc.). Reduced generation of electricity translates into reduced emissions of not only $PM_{2..5}$ but of all the pollutants associated with power generation – NO_x, SO_2, CO_2 and air toxics. Further, efficiency upgrades reduce the risks of fossil fuel dependence, improve the reliability of the power grid, and promote local economic development.

For a number of end uses, efficiency improvements can save energy at a cost lower than the cost of generating and delivering the electricity. That is, the efficiency measure *saves* the customer money over its lifetime, as well as reducing emissions. At zero or negative cost, energy efficiency offers some of the cheapest $PM_{2.5}$ reductions identified in this report. However, while efficiency measures can save money over their lifetimes, the entire cost of the measure is incurred at the time of installation. These "front-loaded" costs represent a barrier to investment in energy efficiency for many electricity customers – customers who are not sure how long they will be at their current location or who do not own their current location. Policy makers have long recognized these barriers and have established a range of programs to incentivize efficiency investments. The state agencies that oversee energy efficiency subsidies study the costs of different efficiency improvements in order to optimize the use of subsidy funds. Typically, payback periods and cost/benefit ratios are calculated for all efficiency investments, and thresholds are established to govern the use of funds. It is common for statewide efficiency programs to have overall cost/benefit ratios of two to three – meaning that two to three dollars are saved for every dollar spent.

In New York, energy efficiency subsidies are currently managed by NYSERDA. NYSERDA's New York Energy Smart[SM] program includes programs to support residential, commercial and industrial efficiency, and the Energy Smart Loan Fund provides low-interest loans for efficiency investments. A 2004 report of the Energy Smart[SM] Program found that, during the program's first seven years, over 1,300 GWhs of electricity was saved – enough energy to meet the annual energy needs of roughly 225,000 homes. The same report estimates the average cost of these savings at about one cent per kWh saved, compared to NYC retail electricity costs in the range of 9 to 12 cents per kWh.[21] More information on these programs is available at: www.nyserda.org.

Emission reductions from efficiency upgrades accrue at the emission rate of the power plants where generation is reduced. There are several different methods of estimating which plants would operate less due to reduced demand in a specific area, however applying these methods is beyond the scope of this study.[22] For a rough estimate of reduced emissions from efficiency in NYC, we estimate the average $PM_{2.5}$ emission rate of the fossil-fired generators serving NYC to be 0.25 pounds per MWh. This estimate is based on EPA data on $PM_{2.5}$ emission rates for plant types, since $PM_{2.5}$ emissions are not measured or reported by power plants.[23] We assume that the cost of reductions from efficiency are either negative (savings) or zero.

Assuming total program savings to date of 1,300 GWhs (based on the Energy Smart[SM] program review report) and using the 0.25 lb per MWh $PM_{2.5}$ emission rate, we estimate

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[21] The 2004 Energy Smart Program review, consisting of six different reports, is available at: http://www.nyserda.org/Energy_Information/ContractorReports/reports_by_contractor.asp.

[22] Methods of estimating emissions displacement by energy efficiency are examined in *Methods for Estimating Emissions Avoided by Renewable Energy and Energy Efficiency*, by Synapse Energy Economics, available at: www.synapse-energy.com.

[23] We average only the fossil-fired units, based on the assumption that nuclear and most hydro units are baseloaded and not affected by changes in demand. This estimate assumes a PM2.5 rate of 0.08 lb/MWh for gas-fired generation and 0.4 lb/MWh for heavy oil and coal-fired generation. These numbers are based on data in EPA's *AP-42 Compilation of Air Pollutant Emission Factors, Volume I, Stationary Point and Area Sources*, Fifth Edition, at: www.epa.gov/ttnchie1/ap42. We have weighted these emission factors with the following assumed percentages of fossil-fueled power generation: 50 percent gas, 30 percent heavy oil and 20 percent coal.

$PM_{2.5}$ reductions resulting from the program to date at roughly 165 tons.[24] Further, the energy efficiency potential in New York State remains vast. The Energy Smart[SM] Program review estimates the economic potential (given current electricity rates) at over 50,000 GWhs. Thus the potential savings from continuation and expansion of the Energy Smart program are tremendous.

6.3.2 Combined Heat and Power

The term "combined heat and power" or "CHP" refers to the production of both heat and electricity in the same process. Most energy users receive electricity from the regional power grid and generate heat with an onsite boiler or furnace. The generation of electricity at power plants removed from the end user results in the loss of considerable amounts of energy at power plants (lost as waste heat) and energy losses in transmission lines as well. The generation of both power and heat onsite allows for a much more efficient process, as heat that is typically wasted can be captured for space heating or process heat.

CHP is well established in certain industries throughout the US. For example, a study for NYSERDA estimated roughly 5,000 MW of CHP capacity at 210 sites in New York State.[25] Industrial facilities (primarily in the metals, paper and chemical industries) accounted for 77 percent of this capacity. However, advances in CHP technology over the past several decades have opened a vast potential for CHP at less energy intensive sites, such as government buildings, commercial establishments and schools and hospitals. The report for NYSERDA estimates over 4,000 MW of CHP potential in downstate New York *outside the industrial sector* (compared to just over 550 MW of remaining potential in the industrial sector).[26] Office buildings and elementary and secondary schools represent the bulk of this potential, with significant additional potential in hotels, restaurants, hospitals, apartment buildings, nursing homes and universities.

The primary CHP technologies that have been employed in the industrial sector include steam turbines and combustion turbines in which waste heat from the turbine is used for process or space heat. The technologies applicable to less energy intensive facilities include smaller combustion turbines, internal combustion engines and fuel cells – all fitted with heat recovery equipment.

The emission reductions achieved with the installation of a CHP system depend on the emission rate of the electricity used by the facility (typically from the regional grid), the emission rate of the equipment providing heat to the facility, the emission rate of the new CHP facility, and the amount of the facility's electric and heating loads served by the

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[24] The Energy Smart Program review report estimates NOx savings of 1,265 tons and 2,175 SO_2 savings of tons.

[25] Energy Nexus Group and Pace Energy Project, *Combined Heat and Power Market Potential for New York State*, prepared for the New York State Energy Research and Development Authority and Oak Ridge National Laboratory, October 2002, p. 3-1.

[26] Energy Nexus Group et. al., p. 4-7 and 4-8.

CHP system.[27] Most new CHP systems are likely to be gas-fired systems with $PM_{2.5}$, SO_2 and NO_x emission rates below those of the power plants they replace. CHP systems are less effective at reducing CO_2 emissions, because they have CO_2 emission rates closer to the average CO_2 rate of grid electricity. To estimate the kind of reductions achievable with a small-scale CHP system, consider a 1-MW gas turbine installed at a facility that was previously getting its electricity from the regional grid and its heat from a gas-fired boiler.[28]

Assuming an 80-percent capacity factor, the CHP gas turbine would burn just over 433,000 mmBtu of gas each year and generate 35,040 MWhs of electricity and just over 112,000 mmBtu of useful heat.[29] Based on EPA's AP-42 *Compilation of Air Pollutant Emission Factors*, both the gas turbine and the gas-fired boiler would have $PM_{2.5}$ emission rates in the range of 0.007 lb/mmBtu. Again estimating average $PM_{2.5}$ emissions from downstate fossil-fired power plants at 0.25 pounds per MWh, this CHP project would reduce roughly 3.5 tons of $PM_{2.5}$ per year. Table 6.5 shows the inputs to this calculation. (The estimate of 3.5 tons reduced comes by adding the displaced emissions from grid electricity and the gas-fired boiler and subtracting the CHP system's emissions from this sum.)

Table 6.5: Illustrative Example of Emission Reductions from CHP

	Grid Electricity	Gas-Fired Boiler	CHP Gas Turbine
MWhs generated	37,493	0	35,040
mmBtu produced	0	112,128	112,128
PM2.5 rate	0.25 (lb/MWh)	0.007 (lb/mmBtu)	0.007 (lb/mmBtu)
PM2.5 tons	4.5	0.5	1.5

Note that the power grid must generate more electricity than the CHP system to deliver the same amount of energy due to line losses (which we estimate at 7 percent). Note also that emissions from the gas-fired boiler are calculated as the emission factor times fuel input (140,160 mmBtu) not heat output (112,128 mmBtu). This is based on an assumed boiler efficiency of 80 percent.

Reductions in NO_x and SO_2 would also be significant – in the range of 15 and 50 tons per year respectively. This example should not be interpreted as a precise calculation of emission reductions but as an illustrative calculation to allow general comparison with the other strategies discussed in this chapter. Obviously, emission reductions from a larger CHP project would be larger.

Assessing the net costs of CHP projects is complex, because an investment in CHP reduces two other existing costs – the costs of electricity and heat. However, organizations considering CHP typically perform extensive analyses to determine whether the potential project will reduce annual energy costs. Projects that are not expected to reduce annual energy costs rarely come to fruition. (In fact, projects that

[27] CHP may also have more complex impacts on the electricity grid. For example, the addition of CHP could slow the rate at which new CCCTs are added to the system, thus deferring the benefits offered by these plants. CHP could also provide net environmental benefits by slowing the rate at which coal-fired plants are added, but in New York, the plants deferred are more likely to be gas-fired CCCTs. We do not consider these potential impacts here.

[28] New CHP capacity could also have the effect of slowing the turnover of existing energy plants, however we do not account for this dynamic.

[29] The characteristics of this gas turbine are taken from Energy Nexus Group et. al., Table 5-1.

come to fruition are usually those that will pay back the initial investment in a reasonable period of time.) Thus, we consider CHP projects to provide emission reductions at no cost over the life of the project.

The important question regarding CHP costs is: how can project costs be reduced so that more projects are cost effective? As noted, very few CHP systems have been installed outside of several industrial sectors amenable to CHP. The cost of CHP systems has come down considerably over the past several decades and performance has improved. These trends are expected to continue. Now, in order for CHP to achieve significant market penetration, other project costs must come down. These costs include project financing costs, permitting costs, utility interconnection costs and utility charges for standby service (energy delivered during periods when the CHP system is not operating). In New York, the cost of standby power is the most significant of these costs. For example, the cost of standby service from ConEd is three to four times the cost of the same service to customers in states such as Illinois, California and Texas.[30] This cost alone represents a considerable barrier to the installation of CHP systems.[31]

If markets barriers to CHP development were significantly reduced, several MW per year of additional CHP might well occur in the NYC area. If five MW per year were added between 2010 and 2015, total $PM_{2.5}$ reductions by 2015 could be in the range of 75 tons per year.

6.3.3 Emission Controls on Existing Energy Plants

Turning to retrofit controls that could be added to the NYC plants that burn residual oil, the most cost effective option is a fabric filter, commonly known as a "baghouse." Other systems that control $PM_{2.5}$ include electrostatic precipitators (ESPs) and scrubbers. However, EPSs are much more effective on larger particulate matter than on fine PM, and scrubbers are primarily designed to control sulfur emissions. Scrubbers would not be a good investment at a plant burning low sulfur oil. There are no control systems designed to reduce direct $PM_{2.5}$ emissions from boilers that burn gas and/or distillate oil only. The only option for these plants is repowering.

Fabric filters are made of a fire-retardant material in the shape of a cylindrical bag or flat, supported envelope. The filter is contained in a housing that has an inlet and outlet for flue gas, a dust collection hopper and a cleaning mechanism. Cleaning mechanisms typically shake the bag or shoot compressed air onto it to remove particles. EPA notes that PM from residual oil combustion can be sticky, and this can make operation of a baghouse problematic.[32] To avoid these problems, some facilities have used gas reburn with baghouse to dry particulate matter leaving the boiler. Gas reburn is a NO_x control technique in which natural gas is injected above the oil burners in the boiler to reduce NO_x formed in the first combustion zone. Reductions of NO_x and PM precursors, would be an added benefit of using gas reburn with a baghouse.

[30] Energy Nexus Group et. al., page 5-16.
[31] The NYSERDA study contains a detailed discussion of the barriers to CHP and ways to address these barriers.
[32] U.S. Environmental Protection Agency, 1998, *Stationary Source Control Techniques Document for Fine Particulate Matter* (EPA-68-D-98-026), October, 1998.

A baghouse can reduce $PM_{2.5}$ emissions by 99 percent or more.[33] Capital costs range from about $30 per kW for a large plant to $50 per kW for a small plant. Annual O&M costs are in the range of $20 per kW-year, including solid waste disposal. Thus annualized costs for a 250-MW plant would be in the range of $6.2 million.[34] Emission reductions would depend on how much residual oil the plant typically burned – many of the NYC plants that burn residual oil also burn natural gas. Assuming that the plant burned roughly half gas and half oil on a Btu basis, a baghouse would reduce $PM_{2.5}$ emissions by about 120 tons per year. This calculation assumes a 65-percent capacity factor before and after control and the $PM_{2.5}$ emission rates shown in Table 6.6 before control. Reductions would cost around $26 per pound. At plants burning a higher percentage of oil, reductions would be larger and less costly. At plants burning a lower percentage of oil, reductions would be smaller and more expensive.

6.3.4 Repowering

The term "repowering" refers to the replacement of the equipment in an older power plant with new equipment, typically a combined-cycle gas turbine (NG CC). Repowering can be accomplished by either rebuilding and replacing part or all of an existing plant or by closing an existing plant and building a new plant next to it, reusing the existing transmission and fuel facilities. Detailed engineering and economic analyses must be performed to determine the optimum size of the repowered unit and the extent to which existing facilities can be refurbished and reused. The type of equipment that can be refurbished and reused include boilers, turbine generators, condensers, transmission switchyards, and other auxiliary plant equipment. The reuse of this equipment can lower the cost of building the repowered facility as compared to the cost of constructing a new unit at a new site.

Repowering older plants provides a number of important environmental and electric system benefits, including improved plant availability, lower plant operating and maintenance costs, and potentially increased plant capacity and generation; reduced heat rates, which lead to significantly more efficient fuel use; reuse of industrial sites; up to 98-percent reductions in water intake and related fish impacts; and large reductions in air emission rates.

Several older power plants in New York State have recently undergone repowering, including the old power plants at Albany and Athens and ConEd's Waterside Station in Manhattan (now the new East River Plant). In addition, Reliant Energy has proposed, and received the necessary permits, to repower the Astoria plant in Queens. The existing Astoria Generating Station houses four operating steam units (Units 2 through 5) that provide a total of 1,254 MW of electric generating capacity. The repowering project would replace the four boilers in Units 2 through 5 with six new combustion turbine generators and refurbishing and reusing the existing Unit 4 and Unit 5 steam turbine generators, condensers and auxiliary equipment. This project would add over 550 MW of additional capacity in NYC (in addition to the existing Astoria plant), lower the cost of

[33] U.S. EPA, 1998.
[34] This calculation assumes capital costs of $50 per kW and a capital recovery factor of 0.12 percent.

generation, increase annual generation, increase the reliability of the plant, reduce emissions of a range of air pollutants and dramatically reduce cooling water use by over 95 percent. Reliant plans to proceed with the repowering when the company obtains financing for the project.

In addition to Astoria, older electric and steam plants at the following sites are candidates for repowering: Ravenswood, East River, Arthur Kill, Hudson Avenue, 59th Street, and 74th Street.

The emission reductions available from repowering are considerable. As shown in Table 6.6, $PM_{2.5}$ reductions range from roughly 40 percent when replacing an older gas-fired boiler to about 90 percent when replacing a boiler burning residual oil. Thus, replacing a gas-fired plant generating 1,000,000 MWhs pear year would reduce $PM_{2.5}$ emissions by 40 tons per year, and replacing a similar oil-fired plant would reduce $PM_{2.5}$ by roughly 220 tons per year.

Table 6.6: Approximate $PM_{2.5}$ Reductions from CCCT Repowering[35]

	$PM_{2.5}$ emissions (lb/mmBtu)	$PM_{2.5}$ emissions (lb/MWh)	Percent Reduction
Gas-Fired CCCT	0.007	0.05	---
Gas-Fired Boiler	0.007	0.08	40%
Oil-Fired Boiler	0.024	0.26	80%

**These calculations assume an efficiency of 11,000 Btu per kWh for the older boiler and 7,000 Btu per kWh for the combined-cycle plant.*

In addition to these $PM_{2.5}$ reductions, repowering reduces emission of other pollutants dramatically. With Selective Catalytic Reduction control technology (as is required on new gas turbines in NYC), NO_x reductions range from roughly 75 percent (replacing a gas-fired boiler) to roughly 80 percent (replacing residual oil). Reductions in CO_2 range from about 35 percent replacing a gas-fired boiler to 50 percent replacing oil, and reductions in SO_2 are roughly 100 percent when replacing oil. SO_2 reductions are negligible when replacing a gas-fired boiler with a gas-fired combustion turbine.

The cost of repowering a plant depends heavily on site-specific factors such as the extent to which existing equipment can be used in the new plant. Costs typically fall in the range of $475 to $575 per kW. In studying its options at the Poletti site, the New York Power Authority examined a number of repowering scenarios. Costs ranged from $470 to $558 per kW in 1997 dollars.[36] In 2000, Orion (now owned by Reliant) estimated the total cost of the Astoria repowering at $750 million, or $414 per kW. (This estimate is probably low now, given that Reliant has had to redesign the water cooling system to satisfy concerns about salt deposition.) ConEd has estimated the cost of repowering the Hudson Avenue station at $381 million in 2006 dollars, or about $733 per kW.

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[35] PM2.5 emissions data for boilers come from EPA's *AP-42 Compilation of Air Pollutant Emission Factors, Volume I, Stationary Point and Area Sources*, Fifth Edition, at: www.epa.gov/ttnchie1/ap42. The numbers for gas-fired technologies are from Table 1.4-2, and the numbers for oil-fired technologies are from Tables 1.3-1 and 1.3-2. Emissions from oil combustion assume 0.2-percent sulfur as is required in NYC.

[36] April 24, 1997 Presentation to the NYPA Management Committee.

To estimate the cost of repowering as a $PM_{2.5}$ reduction strategy, consider the benefits of the replacement of the steam unit at Poletti with the CCCT.[37] (In 2005, NYPA brought online a new CCCT at the Poletti site, and the Agency has agreed to retire the existing steam plant there in 2008.) According to the DEC inventory, the 855-MW steam plant at Poletti emitted 122 tons of direct $PM_{2.5}$ in 2002. The plant operated at a 32-percent capacity factor in that year, generating roughly 2.3 million MWhs.[38] If the new CCCT at Poletti generated the same amount of energy, it would produce only 57 tons of direct $PM_{2.5}$, a reduction of 53 percent.[39] Assuming a cost of $525 per kW for new (500-MW) CCCT places total costs in the range of $262 million, or about $32 million per year (using a 12-percent capital recovery factor). Based on this calculation, the $PM_{2.5}$ reductions from the Poletti repowering would cost $242 per pound. Of course, allocating some of the project costs to other benefits would reduce this cost per pound considerably.

6.3.5 Fuel Switching

Fuel switching can provide significant $PM_{2.5}$ reductions at point sources. Switching from coal combustion to oil or gas reduces $PM_{2.5}$ emissions (both primary and secondary), as does switching from oil combustion to gas. Also, reducing the sulfur content of the oil burned reduces $PM_{2.5}$ emissions. However, there are no coal-fired point sources in NYC, so switching from coal is not an option, and oil-fired plants in NYC are already required to burn low-sulfur residual oil.[40] Thus, the only fuel switching option at large plants in the City is from residual oil to natural gas.

As shown in Table 6.5 above, a boiler burning residual oil (at 0.3-percent sulfur) emits $PM_{2.5}$ at a rate of roughly 0.024 lb per mmBtu, and a gas-fired boiler emits at roughly 0.007 lb per mmBtu. Thus, switching from oil to gas would reduce direct $PM_{2.5}$ emissions by approximately 80 percent. Reductions of SO_2 would be close to 100 percent, and reductions in CO_2 would be just over 20 percent.

Most of the cost of switching from oil to gas would come in increased fuel costs. Several of the NYC plants that burn oil can also burn gas, so switching to gas full time would require minimal up-front investment. At sites without existing gas infrastructure, capital costs would be significant, and repowering might be a more cost effective option than a major investment to bring gas to an old boiler. (see Section 6.3.4)

Evaluating the long-term cost of switching from oil to gas as a primary fuel is complex. Gas and oil prices fluctuate, and they fluctuate both in annual cycles and in a less predictable manner from year to year. Typically, during the spring, summer and fall, the difference between gas and oil prices is not large. During this period, plants that can burn the cleaner fuel (gas) typically do so, to minimize boiler maintenance and air emissions.

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[37] The new CCCT at Poletti came on line in 2005. The old steam unit there is scheduled to be retired in 2008.
[38] US Energy Information Administration, Form 868.
[39] This assumes an emission rate of 0.05 lbs per MWh for the CCCT, as shown in Section 3.1 of EPA's AP-42 *Compilation of Air Pollutant Emission Factors.*
[40] Here, we refer to large plants burning residual oil, as opposed to oil-fired residential heating systems, which burn distillate oil. Low sulfur residual oil is 0.3 percent sulfur by weight, while low-sulfur distillate oil is 0.2 percent sulfur.

However, during the winter "heating season," gas prices rise significantly, and they spike to very high levels during periods of extreme cold or gas shortage. Many industrial facilities minimize fuel costs by buying gas on interruptible contracts during the winter, and supply curtailments are common during cold winters. Facing a curtailed gas supply, plant operators must determine whether profits will be maximized by burning the back-up fuel (usually distillate oil), purchasing gas on the spot market or not operating the plant. Thus, the cost associated with switching from oil to gas as the primary fuel is (a) much higher in the winter than during the rest of the year and (b) likely to vary significantly year to year, depending on winter weather and other dynamics in gas markets.

To estimate the cost of reducing $PM_{2.5}$ by switching to gas, we reviewed data from the U.S. Energy Information Administration on average monthly prices for low-sulfur residual oil and natural gas prices (to industrial facilities) during the period Jan 2003 through August 2005. In 2003, gas was about six percent more expensive than oil on a per-mmBtu basis, and in 2004 it was roughly 13 percent more expensive. Based on the data through August 2005, it appears that the difference will fall between these two numbers in 2005. To produce a conservative estimate of the cost of reductions from fuel switching (e.g., to err on the high cost side) we will assume a long-term average cost premium of 13 percent. This is consistent with a premium of approximately $0.70 per mmBtu.[41] At this cost, reductions would cost roughly $40 per pound, assuming a reduction rate of 0.017 pounds per mmBtu (see difference in emission rates for oil- versus gas-fired boilers in Table 6.6).

This number is low relative to many other options assessed here, even given the high level of uncertainty around it. (If the actual cost were twice as high, it would still be lower than many other options assessed.) However, it is important to bear in mind that this number assumes an existing boiler either with gas capability or with very low capital costs to achieve gas capability. This is effectively the estimated compliance cost for a dual-fueled boiler that reduced $PM_{2.5}$ by burning more gas and less oil. Companies facing a significant up-front investment to switch to gas would add a hefty risk premium to the investment, given the current concern over future gas availability and prices. For the same reason, an investment that added gas capability and removed residual oil capability would carry a *very* large risk premium.

There are limits on the extent to which some units can switch to natural gas. The New York State Reliability Council's (NYSRC) rule I-R3 requires two of the Astoria units to burn oil at a minimum level when forecasted electric load is in excess of 8000 MW. Above 9000 MW, all units at Astoria, Ravenswood and East River must burn oil.[42] The rule provides some protection against blackouts in the event of fuel supply disruption to gas-fired generators.

[41] This number is derived from the year-2004 average oil price ($5.57 per mmbtu) and gas price ($6.27 per mmBtu).
[42] "NYSRC Reliability Rules for Planning and Operating the New York State Power System" Version 18. January 5, 2007 (see http://www.nysrc.org/NYSRCReliabilityRulesComplianceMonitoring.asp)

6.3.6 Summary of Point Source Control Options

End use efficiency improvements and CHP can both provide emission reductions while saving money. Because they reduce the use of fossil-fueled power plants, they reduce emissions of $PM_{2.5}$ as well as NO_x, SO_2, CO_2 and air toxics. They also help to reduce U.S. dependence on imported fossil-fuels. Both efficiency and CHP face market significant market barriers. Both technologies are currently subsidized in New York, but subsidy levels for efficiency are much higher than for CHP. Further, significant market barriers to CHP development remain. If these barriers were mitigated such that five MW were added each year from 2010 through 2015, we estimate the resulting $PM_{2.5}$ reductions in the range of 75 tons per year.

Repowering also reduces fossil-fuel dependence and emissions of a range of air pollutants, however it is a more costly strategy than efficiency and CHP. Switching from oil combustion to gas combustion at point sources that can switch easily may be a low cost option, but there is considerable uncertainty over future gas prices, so investments to gain gas capability will carry risk premiums.

Table 6.7: Summary of Point Source Control Options

Strategy	Potential Reductions (tons/yr)	Reductions in Other Pollutants	Efficiency Gains	Other Benefits	Cost ($/lb $PM_{2.5}$)
Electric Efficiency	High	High	High	Local jobs and investment	$0
Combined Heat and Power	Moderate	High	High	None	$0
Fabric Filters on Oil-Fired Plants	High	Low	None	None	$25
Repowering	High	High	High	None	$240
Fuel Switching	High	Moderate	None	None	$40

Potential reductions are characterized as "Low, Moderate or High," with "Low" referring to reductions in the range of zero to 20 tons per year, "Moderate," in the range of 20 to 100 tons per year and "High" over 100 tons per year. Cost estimates per pound of $PM_{2.5}$ removed are rounded to the nearest five dollars.

6.4 Area Sources

We examine five strategies for reducing $PM_{2.5}$ emissions from area sources:

- mandating the use of ultra-low sulfur oil in residential, commercial and institutional buildings,
- expanded investment in residential thermal efficiency (weatherization),
- residential solar hot water systems,
- taxes on heating fuel, and
- fuel switching.

6.4.1 Mandated Ultra Low Sulfur Oil

Since the early 1970s, NYC law has limited the sulfur content of fuel oil burned to 0.2 percent by weight, or 2,000 parts per million. However, spurred by the promising benefits of reducing the sulfur content of diesel fuel, environmental advocates are seeking parallel reductions in sulfur in fuel oil. The state chapter of the American Lung Association and other groups are advocating an immediate tightening of the standard to 0.05 percent sulfur, or 500 parts per million, with an eventual changeover to 0.0015 percent sulfur, or 15 parts per million.

Reducing the sulfur content of fuel oil reduces emissions of both sulfur dioxide and $PM_{2.5}$, although the drop in particulate emissions is less than proportional. While "condensable" $PM_{2.5}$ emissions, which account for some 40 percent of $PM_{2.5}$ emissions from current (0.2 percent sulfur) fuel oil, are a linear function of sulfur content, the "filterable" $PM_{2.5}$ emissions that account for the other 60 percent of emissions are largely carbonaceous and thus are only partly tied to fuel sulfur content. In our calculations, we assume that half of filterable $PM_{2.5}$ emissions are a linear function of fuel sulfur content while the other half are invariant.[43] Even with this conservative assumption, the expected drop in emissions from reducing fuel oil sulfur content is impressive. As Table 6.8 shows, switching to low-sulfur oil would cut emissions of $PM_{2.5}$ by over half, while switching to very low sulfur oil would cut the emission rate by more than two-thirds.

Table 6.8: $PM_{2.5}$ Emissions as a Function of Fuel Oil Sulfur Content

	Current	Low-sulfur	Ultra low-sulfur
Sulfur Content, %	0.2%	0.05%	0.0015%
Sulfur, parts per million	2,000	500	15
Condensible $PM_{2.5}$ (lb/mmBtu)	0.01	0.00	0.000
Filterable $PM_{2.5}$ (lbs/mmBtu)	0.01	0.01	0.007
Total $PM_{2.5}$ (lbs/mmBtu)	0.02	0.01	0.007
Percent drop from current emissions	---	52%	69%

The net cost associated with switching to universal use of low-sulfur fuel oil – or very low-sulfur, for that matter – is likely to be zero or even negative. First, the initial added cost to refine, store and transport ultra-low sulfur oil is relatively modest, approximately 10 cents per gallon.[44] Second, this cost is expected to decline as the entire diesel fuel-handling system switches to ultra-low sulfur, obviating the need to maintain two separate systems. Third, extensive testing, some of it funded by NYSERDA (along with U.S. DOE and several oil heat industry associations), indicates that much or all of the remaining cost would be offset by maintenance savings (i.e., fewer service calls) and improved combustion efficiency due to the reduced presence of sulfur compounds.

For example, the National Oilheat Research Alliance reports that boiler fouling by sulfur dioxide is responsible for most of the decline in the combustion efficiency of oil-fired

[43] S.W. Lee et al., "Assessing PM2.5 Emissions from Distillate Fuel Oil Heating," Paper No. 02-13, Proceedings of the 2002 National Oilheat Research Alliance Technology Symposium, available at http://www.pubs.bnl.gov/documents/24273.pdf.

[44] Conservation Law Foundation, "A Clean Air Coup for CLF," 2003, available at http://www.clf.org/general/index.asp?id=426.

boilers over time, a deterioration estimated at two percent per year.[45] Since sulfur dioxide emissions are directly proportional to fuel sulfur content, the four-fold reduction in SO_2 emissions associated with mandating low-sulfur fuel oil, and the 133-fold reduction from mandating very low-sulfur oil, would be expected to improve boiler efficiencies and reduce maintenance costs. The cost savings likely would not only offset much or all of the price increase but would also improve fuel oil's competitiveness with gas. For these reasons, New York City fuel oil suppliers have endorsed a 500 ppm (0.05 percent sulfur) standard.[46]

To get a sense of the potential $PM_{2.5}$ reductions achievable via moving to lower sulfur fuels we apply the percentage reductions in Table 6.8 to our baseline estimates of $PM_{2.5}$ emissions from oil-heated buildings. If low-sulfur oil were mandated in 2010 for all residential buildings, direct $PM_{2.5}$ emissions would fall by approximately 270 tons, or 21 percent of projected residential $PM_{2.5}$ emissions in that year.

Table 6.9: Estimated Reductions from Switching to Low-Sulfur Fuel Oil

Sector	Low Sulfur in 2010		Ultra Low Sulfur in 2015	
	Tons Reduced	Percentage	Tons Reduced	Percentage
Residential	270	21%	345	27%
Commercial/Institutional	145	15%	200	20%
Total	415	---	545	---

If reduced maintenance costs and extended equipment life fully offset the cost premiums, these reductions would cost nothing. If these savings only offset half of the 10-cent cost premium for ultra low-sulfur fuel, reductions would cost roughly $22 per pound – still among the lowest-cost reductions we have identified. (This calculation is based on a $PM_{2.5}$ reduction of 0.016 pounds per mmBtu – from Table 6.8 – and a cost increment five cents per gallon, or 36 cents per mmBtu.)

6.4.2 Thermal Energy Efficiency

Each year, approximately 10,000 NYC residential dwelling units are retrofitted to improve heat and hot water efficiency.[47] Some efficiency upgrades, often called "weatherization," focus on replacement of windows and the addition of insulation to reduce heat loss. Other upgrades target heating equipment such as hot water heaters and furnaces. In calculating our baseline emission estimates we assumed that weatherization would continue at that pace. This rate, which equates to a mere 0.33 percent of the City's housing stock, is slightly less than our assumed 0.3 percent per year (that rate is based on 1990-2003 population growth).

[45] W.L. Litzke & R. Hedden, "National Oilheat Research Alliance Fuel Performance Research Update," Paper No. 05-03, Proceedings of the 2003 National Oilheat Research Alliance Technology Symposium, available at http://www.pubs.bnl.gov/documents/25281.pdf.

[46] C. Komanoff, personal communication with John Maniscalco, executive vice-president, NY Oil Heating Association, November 16, 2005.

[47] A. Padian, S. Winter Associates, Personal communication, June 14, 2005.

We believe that the benefits of thermal efficiency upgrades, in terms of fuel and cost savings, emission reductions, local jobs, and housing preservation, are great enough to warrant expanding current programs to reach more buildings in more neighborhoods. Indeed, the roughly 50-percent increase in fuel oil and natural gas prices from 2003 to 2005 has already made weatherization much more cost-effective. As with electric energy efficiency, subsidies are available to property owners in New York State to help reduce the up-font costs of thermal efficiency measures.

To evaluate thermal efficiency as a $PM_{2.5}$ reduction strategy, we examined the effects of expanding existing residential weatherization programs to cover one percent of the City's housing stock (approximately 30,000 units) in 2008; two percent per year (60,000 units) in 2009 and 2010; and three percent per year (90,000) in 2011 and thereafter.

With the further assumption that weatherizing residences reduces the fuel required for heat and hot water by 25 percent on average, this strategy reduces $PM_{2.5}$ emissions by 11 tons per year in 2010 (a one-percent drop from the base level of residential emissions projected for that year) and by 51 tons per year in 2015 (equivalent to a four-percent drop from that year's baseline).

We project the per-pound costs of reducing particulate emissions via expanded weatherization programs to be negative, due to the considerable fuel savings. We calculate a "gross" cost of $440 per pound of $PM_{2.5}$ eliminated by weatherization, based largely on an estimated $5,000 capital cost to weatherize each home or apartment, with a 20-year life to the savings measures. However, the fuel savings are anticipated to have a value of roughly $480 for each pound of $PM_{2.5}$ eliminated, based on the 25-percent fuel-savings rate and base fuel prices of $2.20 per gallon of fuel oil and $14.00 per mcf of gas.[48] Thus, expanded weatherization, like mandating lower-sulfur oil, appears to be a $PM_{2.5}$ control strategy with zero and perhaps negative net costs over the life of the investment.

For weatherization to grow to the levels envisioned here, the current marketing model based on individual buildings opting into available state and federal programs will likely have to grow in scale, perhaps with entire blocks being treated *en masse*. A possible template for this transformation is the proposed "Greening A Block" project being developed on Manhattan's Lower East Side which seeks to exploit the logistical and institutional economies of scale presented when the block-level is exchanged for the building-level.[49]

6.4.3 Solar Hot Water Systems

Rising fuel prices may finally be pushing residential solar hot-water systems into the zone of economic cost-effectiveness, at least in buildings with physical characteristics — piping, roofs, orientation — amenable to solar retrofits. As a zero-emission technology

[48] The average price for No. 2 fuel oil delivered in NYC for the week ended 3-28-05 was $2.377; see http://www.nyserda.org/Energy_Information/nyepd.asp. The $2.20 price used here is conservatively less. The $14.00/mcf price assumed for natural gas is roughly based on price parity with No. 2 oil.

[49] See www.greeningablock.org. The author of this section is a developer of this project.

that displaces fossil-fuel combustion for water heating, solar hot water systems reduce air emissions, including PM2.5. Because solar hot water systems are currently not well established in NYC, we examined them at a market penetration rate half that of expanded weatherization programs, or 0.5 percent of the City's housing stock (approximately 15,000 units) in 2008; one percent per year (30,000 units) in each of 2009 and 2010; and 1.5 percent per year (45,000) in 2011 and thereafter.

While such a program appears extraordinarily ambitious compared to current *de minimis* activity in this realm, it is not beyond the realm of possibility. For one thing, there is already an active community of solar practitioners (designers, architects, engineers, installers and advocates) in NYC. Moreover, because most NYC residential units are in multifamily buildings, the number of buildings needing to be retrofitted to meet our targets is likely to be only in the high hundreds or low thousands. It may also prove possible to "piggyback" solar hot water retrofits onto the ramped up weatherization program discussed directly above; indeed, the "Greening A Block" project envisions doing just that.

We assumed that solar retrofits could eliminate 75 percent of the annual fuel energy required for hot water, and that hot water accounts for 50 percent of total annual fuel for residential heat and hot water.[50] These assumptions imply 37.5 percent savings in total annual fuel for residential heat and hot water (since 0.50 x 0.75 = 0.375). Combining these assumptions with the penetration rates noted, the solar retrofit measure would reduce $PM_{2.5}$ emissions in 2010 by 11 tons per year, (a one-percent drop) from base residential emissions projected for that year; and by 45 tons in 2015 (a 4 percent drop from that year's baseline).

Costs of these $PM_{2.5}$ reductions are highly uncertain, given the emerging status of the solar hot water sector in New York (and other cities), and the considerable variability across sites and installations. We assumed a per-unit cost of $10,000, or twice the cost assumed to weatherize each dwelling unit. Since annual energy savings from solar hot water are assumed to be only 50 percent greater than those from weatherization alone (37.5 vs. 25 percent), the economics of solar hot water are less favorable than those of weatherization. Even after allowing for the value of the fuel savings, the net cost of each saved pound of $PM_{2.5}$ averages roughly $440.

6.4.4 Fuel Taxes

Taxing heating fuel is a means of inducing other emission control measures, rather than a control technique in itself. The premise is that higher fuel prices spur property owners and managers to invest in equipment and maintenance that cause fuel to be used more sparingly. These measures will likely include the main elements of weatherization programs, such as reconfiguring and upgrading antiquated heating systems, insulating and air-sealing buildings and windows, and installing water-efficient showerheads and room thermostats.

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[50] A 50% share of total annual heating fuel requirements for hot water heating may seem high, but it is based on field investigations by a respected practitioner, Andrew Padian of S. Winter Associates. See, for example, his article, "Fuel Use in Multifamily Buildings," in *Home Energy*, Nov/Dec 1999.

On a larger scale, high fuel prices often stimulate major sectors including government and public-private partnerships to strengthen existing programs to improve energy efficiency or to develop new ones. This is because high fuel prices reduce payback times for efficiency measures and, not coincidentally, tend to focus societal interest in saving energy. Indeed, the current (mid-2005) upsurge of interest in energy efficiency and energy policy in general is largely due to the rise in petroleum prices since the spring of 2004.

For 2010 we assumed a "starter" tax of $0.50 per gallon for fuel oil and $2.50 per mcf for gas; while for 2015 we assumed taxes at triple these levels — $1.50 per gallon and $7.50 per mcf. (Note that the $1.50 per gallon levy matches the tax rate for gasoline and diesel fuel assumed in our analysis of carbon taxes for on-road motor vehicles.) On a Btu basis, the taxes on gas are only 70 percent as great as the respective taxes on oil, reflecting our assumption that the fuel taxes would be imposed as one element of an economy-wide program to reduce greenhouse gas emissions by taxing the carbon content of fuel. (Natural gas has roughly 30 percent less carbon content per Btu than petroleum products.)

We assumed a price-elasticity of -0.4 for heating fuel use, based on empirical analyses of fuel oil demand conducted under the auspices of the DOE Regional Residential Heating Oil Model.[51] This means that a 10-percent price reduction induces a four-percent reduction in demand. (Large price changes have a proportionately smaller impact, because demand becomes less elastic as people use less.) We applied the same elasticity to natural gas. On a percentage basis, the taxes assumed here would raise oil and gas prices by roughly 20 percent in 2010 and by around 60 percent in 2015.

The projected impacts of these fuel taxes are a 7-percent drop in heating fuel use and in $PM_{2.5}$ emissions in 2010 and 17 percent reductions in 2015, when the tax level is three times as great. This equates to an 86-ton reduction in 2010, rising to 221 tons in 2015, making fuel taxes (at the levels postulated here) our second most potent control measure for residential buildings, after mandating low and very-low sulfur fuel oil.

Of course, the taxes would raise fuel bills considerably. Even allowing for the decrease in fuel use, average residential fuel bills would rise by around one-eighth due to the lower tax in 2010, and by around one-third when the higher tax was imposed in 2015. On a society-wide basis, however, the net cost of the higher taxes should be considered to be close to zero, insofar as the tax revenues presumably would be used either to provide additional public services or to enable "tax-shifting" in which other levies such as sales or income taxes are reduced.

The recent enactment of caps on motor fuel and diesel motor fuel taxes by the New York State Assembly offers an opportunity to re-examine other energy taxes and seek opportunities to encourage end-use consumers to increase energy efficiency opportunities

[51] See http://www.eia.doe.gov/smg/asa_meeting_2005/spring/files/heatingoilmodel.pdf. The same source shows the delivered price of No. 2 fuel oil in NYC during the 2004-05 heating season ranging between $1.91 - $2.38, probably averaging $2.20 or more.

beyond the transportation sector.[52] The Petroleum Business Tax (Article 13-A) presently excludes sales of fuel oil for residential use, offering consumers little incentive to improve heating system efficiency. Linking part of the sales tax exclusion to improved system efficiency, annual maintenance or replacement could ultimately reduce end user energy costs and yield $PM_{2.5}$ emissions reductions across NYC.

6.4.5 Residential Fuel Switching

Fuel switching from oil to gas has been occurring in the area source sector during the past decade. According to U.S. Census data, the number of oil-heated residential units in NYC shrank from 1,232,390 in 1990 to 996,605 in 2000 – a 19 percent decline that equates to a 2.1 percent annualized rate. There is evidence that this trend may be slowing, due perhaps to the high natural gas prices of recent years.

Section 6.3.4 examined a switch from *residual* fuel oil (burned in large boilers) to gas as a $PM_{2.5}$ reduction option. Here, we focus on the switch from the *distillate* fuel oil commonly burned in residential systems. $PM_{2.5}$ emissions from distillate fuel oil are lower than from the heavier, residual oil, so residential fuel switching will result in smaller emission reductions than industrial fuel switching. Table 6.10 shows EPA's estimate of total PM emissions from residential heating systems. For gas-fired systems, all PM is assumed to be less than 2.5 $\mu g/m^3$ (i.e., $PM_{2.5}$). For oil-fired systems, all condensable PM and a majority of filterable PM is assumed to be $PM_{2.5}$. Note that the midpoint of the range for new and old oil-fired boilers (0.02 pounds per mmBtu) is consistent with the data from XXX, shown in Table 6.8.

Table 6.10: EPA Estimates of PM from Residential Heating Systems (lb/mmBtu)[53]

	Condensable	Filterable	Total*
Small Gas-Fired Boiler	0.006	0.002	0.007
Small Distillate Oil-Fired Boiler	0.01	0.01	0.02
Gas-Fired Residential	0.006	0.002	0.007
New Oil-Fired Residential	0.01	0.003	0.01
Old Oil-Fired Residential	0.01	0.02	0.03

**Numbers may not sum due to rounding.*

Assuming 0.02 lb per mmBtu from existing oil fired systems, typical residences (burning roughly 100 mmBtu per year) would emit about two pounds per year. A gas-fired system would emit 0.70 pounds per year. Thus, the fuel switch would reduce $PM_{2.5}$ emissions by roughly 1.3 pounds per year, or 65 percent. Switching about 1,000 housing units would reduce a ton of $PM_{2.5}$ per year.

[52] New York State Assembly Act of May 12, 2006, A11331 (suspending portions of the motor fuel and diesel motor fuel sales and compensating use taxes). See also Danny Hakim, In New York, A Move to Cut Gasoline Taxes, *The New York Times* May 11, 2006.

[53] These numbers are from EPA's *AP-42 Compilation of Air Pollutant Emission Factors, Volume I, Stationary Point and Area Sources*, Fifth Edition, at: www.epa.gov/ttnchie1/ap42. The numbers for gas-fired technologies are from Table 1.4-2, and the numbers for oil-fired technologies are from Tables 1.3-1 and 1.3-2.

SO_2 reductions from residential fuel switching would be larger than $PM_{2.5}$ reductions, since oil-fired residential systems emit about 0.2 pounds of SO_2 per mmBtu burned. Roughly one ton of SO_2 would be reduced for every 100 housing units switched from oil to gas. CO_2 reductions would be relatively small (a reduction of just over 20 percent with each conversion).

Historically, the prices of gas and distillate oil for residential heating have been very close on a per-Btu basis. During the past several years, fuel oil has been slightly cheaper than gas in the Northeast, however it is currently slightly more expensive. We assume continued price parity between these fuels. The up-front cost of switching to oil would be the cost of a new burner on the furnace or boiler, the cost of removing the existing fuel tank and bringing a gas line to the residence. These costs can vary widely. Gas utilities frequently offer promotions in which they will arrange for removal at no cost and provide the customer with a new burner at a deep discount. The cost of getting gas to the residence can range from several hundred to several thousand dollars, depending on where the residence is in relation to the nearest gas main. However, customers who are quoted costs in the higher end of this range rarely chose to switch fuels. The cost of most fuel switches falls in the range of $500 to $1,000. To be conservative, we estimate switching costs at $1,000. Assuming that this is paid in cash (i.e., no interest) and amortized over the life of the boiler/furnace equipment (15 years), the cost would be roughly $65 per year. This would provide $PM_{2.5}$ reductions at a cost of roughly $33 per pound.

6.4.6 Summary of Area Source Control Options

The five options for reducing area source emissions are summarized in Table 6.12. Mandating the use of ultra-low sulfur oil in residential, commercial and institutional buildings is a very attractive option, offering significant emission reductions at very low net costs (net of savings on boiler/furnace maintenance). The thermal efficiency programs assessed would increase existing residential weatherization programs to cover 90,000 housing units per year in 2011. The solar hot water program would penetrate the residential housing sector at half this rate. Fuel taxes rising to $1.50 per gallon of fuel oil and $7.50 per mcf of gas in 2015 offer reductions of over 200 tons per year. Residential fuel switching is a relatively low cost strategy, however reductions accrue slowly, as each residence converted reduces less than two pounds of $PM_{2.5}$ per year.

Table 6.11: Summary of Area Source Control Options

Strategy	Potential Reductions (tons/yr)	Reductions in Other Pollutants	Efficiency Gains	Other Benefits	Cost ($/lb $PM_{2.5}$)
Low-Sulfur Oil	High	Low	None	None	$0
Thermal Efficiency	Moderate	High	High	Reduced fuel use	$0
Solar Hot Water	Moderate	High	High	Local jobs, investment	$440
Fuel Taxes	High	High	Moderate	Reduced fuel use	$0
Fuel Switching	Low	Moderate	None	None	$35

Cost estimates are rounded to the nearest five dollars.

6.5 Summary of Control Options

Table 6.12 summarizes the $PM_{2.5}$ control options evaluated here. As in the Tables above, potential reductions are characterized as "Low, Moderate or High," with "Low" referring to reductions in the range of zero to 20 tons per year, "Moderate," in the range of 20 to 100 tons per year and "High" over 100 tons per year. Our cost estimates per pound of $PM_{2.5}$ removed are also shown (rounded to the nearest five dollars). Reductions in other pollutants and efficiency gains are also characterized as Low, Moderate or High, but the terms do not refer to specific ranges in these cases. Many of these options, such as fuel taxes and tolling, produce considerable revenue, however this revenue is considered to be a transfer payment and is not included in cost calculations. That is, costs shown in the table include only real resource costs.

Comparing the costs and benefits of these options is a complex process. While it is tempting simply to rank the options in terms of dollars per pound of PM removed, the benefits of the options differ significantly. For example, looking just at direct costs, fuel switching at a power plant burning residual oil is more expensive than installing a baghouse. However, if the plant may only operate for several more years before being retired, the baghouse (a capitalized investment) may be more costly than the increased fuel costs of burning more gas.

Table 6.12: Summary of Control Options

Strategy	Potential Reductions* (tons/yr)	Efficiency Gains	Reductions in Other Pollutants	Other Benefits	Est. Cost ($/lb $PM_{2.5}$)
On-Road Sources					
Controls for "Other" Buses	Moderate	None	Moderate	None	$35
East River Tolls	Low	Low	Low	Reduced Congestion	$6,250
CBD Cordon Tolls	Low	Low	Low	Reduced Congestion	$5,000
Weight-Distance Fees	Moderate	Low	Low	Reduced Congestion	$6,000
Weight-Distance (Trucks Only)	Moderate	Low	Low	Reduced Congestion	$4,000
Fuel Taxes	Moderate	Moderate	Moderate	Reduced Congestion	$0
Per-Mile Insurance	Low	Low	Low	Reduced Congestion	$4,000
Non-Road Sources					
Controls on Refrigerated Trucks	Moderate	None	Moderate	None	$160
Controls on Mobile Generators	Moderate	None	Moderate	None	$160
Point Sources					
Electric Efficiency Programs	High	High	High	Local jobs and investment	$0
Combined Heat and Power	Moderate	High	High	None	$0
Fabric Filters on Oil-Fired Plants	High	None	Low	None	$25
Repowering	High	High	High	None	$240
Fuel Switching	High	None	Moderate	None	$40
Area Sources					
Ultra-Low Sulfur Oil	High	None	Low	None	$20
Thermal Efficiency Programs	Moderate	High	High	Local jobs and investment	$0
Solar Hot Water	Moderate	High	High	Local jobs and investment	$440
Fuel Taxes	High	Moderate	High	None	$0
Fuel Switching	Low	None	Moderate	None	$35

**For potential reductions, "Low" refers to reductions in the range of zero to 20 tons per year, "Moderate" in the range of 21 to 100 tons per year and "High," over 100 tons per year.*

Section 7 provides a brief discussion of the policy implications of this analysis.

7. Recommendations

New York City residents will continue to face elevated health risks from exposure to $PM_{2.5}$ concentrations throughout the City unless opportunities identified in this, and other, reports for additional reductions are systemically pursued by municipal, state and regional authorities. Residents face slightly higher toxicity from particulate matter in NYC as compared with the national average, given the higher ambient concentration, the preponderance of combustion particles, and the proximity to motor vehicle sources. The ambient $PM_{2.5}$ concentrations present in NYC are well within the range of concentrations known to influence cardio respiratory mortality and morbidity.

The very young, the very old, and those with pre-existing health conditions (such as heart disease and asthma) are particularly affected by air pollution. With one of the highest burdens of asthma problems in the country, NYC residents are highly susceptible to increases in air pollution.

While reducing sources of $PM_{2.5}$ will remain a challenge for local, state, and regional authorities, synergistic opportunities abound including successful efforts to reward investments in energy efficiency in New York City and across the State, and more long-term benefits associated with the implementation of the Regional Greenhouse Gas Initiative (RGGI).[1] A large range of emission reductions options were identified and initially evaluated in Chapter 6, considering immediate costs and potential co-benefits (including reductions in other emissions). Options considered include many that have been rejected in the past, due to their perceived political infeasibility and miscalculation of costs and benefits. Focusing on the local, state and regional policy options for achieving these needed reductions is arguably the most profitable direction, and a range of opportunities exist at these levels of government. They include:

- Energy efficiency programs – both electric and thermal – are very low-cost ways of reducing fuel combustion, provide substantial reductions in other pollutants, and stimulate local economic development.
- Development of new combined heat and power systems provides zero-cost emission reductions, as energy cost savings typically result in net savings over the life of the system.
- Mandated use of low or ultra low-sulfur fuel oil for heating in the area source sector could reduce emissions significantly at very low costs. Reduced maintenance costs and extended equipment life could offset most or all of the increased fuel cost.

[1] Anthony DePalma, It Never Sleeps, but It's Learned to Douse the Lights, The New York Times December 11, 2005. Memorandum of Understanding, Regional Greenhouse Gas Initiative, signed December 20, 2005, see www.rggi.org. See also Anthony DePalma, States Agree on Plan to Cut Power Plant Emissions, The New York Times December 21, 2005.

- Controls on "other" buses operating in NYC (i.e., those not already controlled) would provide modest reductions (roughly 40 tons per year) at modest cost, with co-benefits in reduced CO and HC emissions.
- Fuel switching at both point sources and area sources offers significant emission reductions; however, movement from oil to natural gas is happening naturally in these sectors and is putting additional strain on gas infrastructure and upward pressure on natural gas prices.
- Further research is needed into the efficacy and suitability of emissions controls for energy plants firing residual fuel for electricity and steam generation within New York City.

NYC has successfully utilized its municipal code authority to pursue similar ends, mandating the use of ultra-low sulfur diesel fuel for public construction projects and implementing model energy efficiency building standards for new construction.[2] Another avenue that could be pursued includes amending the municipal tax authority to leverage several of the recommendations (many of these would require state legislative action).[3] The State of New York has enacted similar regulations and legislation, netting $PM_{2.5}$ emission reductions. Regardless of the options policymakers elect to pursue, it is critical that additional $PM_{2.5}$ emission reductions be obtained, given the growing uncertainty regarding future federally mandated pollution control strategies in many of the affected categories outlined in this report. This point is critical, as uncertainties concerning future federal actions beyond implementation of the existing suite of national strategies focusing on heavy duty trucks, off-road equipment and power plants may leave NYC to act in the absence of national control strategies in the future. Absent a marked change in the direction of national air pollution policy, local, state and regional $PM_{2.5}$ emission reduction strategies will play an increasingly important role in protecting residents for adverse health effects associated with exposure to $PM_{2.5}$ sources.

Many of the recommended $PM_{2.5}$ reduction strategies proposed in this report will be hampered by reductions in the geographic scope or number of sites in the existing $PM_{2.5}$ monitoring network in NYC. Simply put, this monitoring network needs to be maintained and arguably expanded. As discussed in chapter 3, preliminary field studies indicate that the vertical profile of $PM_{2.5}$ concentrations in the urban street canyon can vary substantially between street-level, mid-story and rooftop. $PM_{2.5}$ emitted by vehicles and other sources can rise vertically and mix, depending on the vortex effect created by wind coming off rooftops, exposing residents in mid-stories to higher concentrations than

[2] New York City Council legislation in 2003-2005 separately adopted LL 324A, requiring new municipally-owned or operated buildings to satisfy the LEED Silver energy efficiency green building standards, and LL 77 requiring in the use of ULSD and pollution control devices on construction equipment used on new public construction projects.

[3] The City of New York imposes many of the same taxes imposed by New York State, but while property and various business and excise taxes are administered by the New York City Finance Department, most cannot be amended except by State legislative action. *Handbook of New York State and Local Taxes*, New York State Department of Taxation and Finance Office of Tax Policy Analysis, November 2005.

might be seen on the rooftop. Since many existing $PM_{2.5}$ monitoring sites in NYC are located on rooftops to capture background ambient conditions, rather than potentially higher local $PM_{2.5}$ concentrations from the streets below, the vertical $PM_{2.5}$ variability phenomenon could be a greater concern than existing monitoring trend data suggest. Given the predominance of six-story residential buildings throughout much of NYC, these findings call for further field studies and possibly special purposes monitoring sites to better understand vertical mixing of $PM_{2.5}$ and to evaluate the efficacy of policy strategies in reducing the $PM_{2.5}$ concentrations that residents are likely experiencing in their homes, workplaces and traveling around NYC.

7.1 On-Road Mobile Sources

Significant progress has been made by New York City and various state agencies in reducing $PM_{2.5}$ emissions from on-road mobile sources. Opportunities exist to leverage ongoing measures for additional reductions, specifically from privately operated bus fleets serving NYC. Taking into account expected additional pollution control measures, including the MTA's pending retrofit of particulate filters and use of ultra-low sulfur diesel fuel, on-road mobile source emissions are expected to decline from an estimated 718 tons in 2005 to 539 tons in 2010 and 499 tons in 2015. Much of this decline can be attributed to pollution control measures focusing on three categories of large diesel vehicles discussed in detail in Chapter 5 (5.1).

Several strategies are available to pursue additional $PM_{2.5}$ emission reductions from on-road mobile sources, such as extending pollution control measures currently targeting MTA buses to privately owned and operated commercial vehicle fleets including an estimated 7,000 school buses, sightseeing and excursion buses and sanitation vehicles. Additional federal funding for particulate filter and engine modifications may be available in future budget cycles, which New York City may obtain through participation in the Northeast Diesel Collaborative. In addition to leveraging possible federal funding through such initiatives, the New York State Assembly should mandate that the New York State Department of Environmental Conservation (NY DEC) develop, fund and implement programs that provide incentives, including possible grants, for engine replacements that reduce $PM_{2.5}$ emissions from private sector diesel buses and service vehicles.[4]

Other $PM_{2.5}$ emission reductions strategies that merit consideration include weight distance fees on travel within NYC for passenger vehicles and heavy trucks and congestion mitigation measures limiting vehicle access to the Manhattan Central Business District by phasing in vehicle tolls on bridges and access fees, which could also help mitigate expected increases in vehicle miles traveled in NYC and associated worsening traffic congestion. The recent successful experience of the City of London with congestion mitigation measures suggests that such measures warrant careful consideration. A recent study commissioned by Transportation Alternatives and endorsed

[4] The New York State Assembly could also mandate that the NYS Department of Environmental Conservation adopt pursuant to Clean Air Act Section 177 any program reducing privately-owned bus and service vehicle emissions adopted by the California Air Resources Board.

by several New York business groups recommends favoring sidewalk widening, bicycle lanes, exclusive bus lanes and parking reform.[5]

7.2 Non-Road Mobile Sources

Current EPA and NYC regulations will reduce $PM_{2.5}$ emissions from non-road vehicles dramatically. Non-road $PM_{2.5}$ emissions are expected to decrease about 20 percent by 2010, from roughly 1,865 tons in 2005 to 1,523 tons in 2010, and by about 40 percent to 1,173 tons in 2015. These regulations target the sources responsible for the vast majority of non-road emissions – construction and commercial equipment. The most significant categories of engines that remain unregulated are refrigerated trucks and small electric generators. Refrigerated trucks emitted just over 100 tons of $PM_{2.5}$ in 2004, and mobile electric generators emitted roughly 160 tons. These two categories merit additional attention as reducing $PM_{2.5}$ emissions from such sources will significantly benefit all residents.

Based on our analysis, the most promising category in the Non-Road sector for additional regulation appears to be refrigeration equipment (transport refrigeration units) mounted on trucks, trailers, rail cars and shipping containers powered by integral internal combustion engines. Emissions from truck-based refrigeration units accounted for approximately 106 tons of $PM_{2.5}$ in 2004. The NY DEC should consider adopting the emissions reduction strategy for transport refrigeration units being implemented by the California Air Resources Board (CARB) under the Clean Air Act, Section 177.[6,7] Regulation of refrigeration equipment is not one of our top-tier recommendations, because it lacks benefits in efficiency and has high costs per pound of $PM_{2.5}$ reduction relative to some of the other strategies.

7.3 Point Sources

As discussed in Chapter 4 (4.3), $PM_{2.5}$ emissions from point sources in NYC come primarily from large energy facilities – power and steam plants. This sector is responsible for 79 percent of point source emissions, followed by residential point sources, responsible for only 6 percent.

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[5] Schaller Consulting, Necessity or Choice? Why People Drive in Manhattan, prepared for Transportation Alternatives (February 2006). See http://www.schallerconsult.com/ and Thomas J. Lueck, Let Traffic Flow and So Will Commerce, Business Groups Tell New York City, The New York Times, February 23, 2006.

[6] California Air Resources Board, Airborne Toxic Control Measure for Diesel-Fueled Transport Refrigeration Units and TRU Generator Sets, and Facilities Where TRUs Operate (TRU ATCM) http://www.arb.ca.gov/regact/trude03/trude03.htm, effective December 10, 2004. See also http://www.arb.ca.gov/diesel/tru.htm.

[7] CARB estimates PM emissions reductions, once truck refrigeration units come into compliance, of 65 percent by 2010 and 92 percent by 2020. CARB Revised Staff Report: Initial Statement of Reasons for Proposed Rulemaking Airborne Toxic Control Measure for In-Use Diesel-Fueled Transport Refrigeration Units (TRU) and TRU Generator Sets, and Facilities Where TRUs Operate (October 28, 2003), p. 136. http://www.arb.ca.gov/regact/trude03/revisor.pdf.

Additional site-specific and cost research is recommended to better assess the efficacy and operational suitability of retrofitting certain pollution control devices on oil-fired boilers and combustion turbines at such facilities. The strategies available for reducing $PM_{2.5}$ from these sources focus on increasing the efficiency of energy production in the City and switching to fuels that emit less $PM_{2.5}$. While emission control systems targeting $PM_{2.5}$, primarily baghouses, are commonly installed on coal-fired boilers, they are not generally installed on gas- and oil-fired boilers and combustion turbines, such as those common in NYC.

$PM_{2.5}$ emission reductions strategies with demonstrated potential and success in NYC that warrant further attention and action include: increasing investment in end-use efficiency, removing regulatory and financial barriers to deployment of combined heat and power production facilities, and repowering and fuel switching at existing medium and large energy generating facilities.

7.3.1 Electric Energy Efficiency

The efficiency of many electricity end uses could be improved significantly, yielding additional $PM_{2.5}$ emissions reductions within NYC. Both the New York State Public Service Commission and the New York City Council have recognized the potential, with the NY PSC reauthorizing the System Benefits Charge, administered by the New York State Energy Research and Development Authority (NYSERDA) as the New York Energy $mart^{SM} Program and the Council adopting other measures.[8]

The reauthorization of the Energy $mart^{SM} Program needs to target additional investment into yet untapped energy efficiency opportunities identified in a recent report focusing on commercial new construction, equipment replacement in existing structures, and appliance and equipment efficiency standards.[9] Further, efficiency upgrades reduce the risks of fossil fuel dependence, improve the reliability of the power grid, and promote local economic development.

For a number of end uses, efficiency improvements can save energy at a cost lower than the cost of generating and delivering the electricity. That is, the efficiency measure *saves* the customer money over its lifetime, as well as reducing emissions. At zero or negative cost, energy efficiency offers some of the cheapest $PM_{2.5}$ reductions identified in this report.

7.3.2 Combined Heat and Power

While an estimated 5,000 MW of larger-scale combined heat and power projects have been installed at 210 sites in New York State, an enormous unrealized opportunity

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[8] New York Public Service Commission, Public Service Commission Extends Statewide Efficiency Program, December 14, 2005. See www.dps.state.ny.us.

[9] Optimal Energy, Inc. et al, *Energy Efficiency and Renewable Energy Resource Development Potential in New York State Final Report* prepared for the New York State Energy Research and Development Authority (August 2003). At www.nyserda.org/publications/EE&ERpotential.

remains within New York City to site an additional 4,000 MW in non-industrial sites according research commissioned by NYSERDA.[10] The bulk of this potential is located in commercial and institutional structures including office buildings, elementary, secondary schools, colleges and universities, with significant additional remaining potential in hotels, restaurants, hospitals, apartment buildings, and nursing homes.

The New York Energy $mart[SM] Program is credited with stimulating development of nearly 15 MW of combined heat and power and distributed generation projects, with 100 MW expected by 2007. Ramping up the NYSERDA competitive solicitation process for combined heat and power and distributed generation projects, through performance-based standard offer energy efficiency programs, excluding highly-polluting emergency generators, has been recommended and should be implemented with the reauthorization of the SBC.

In September 2005, the NY PSC issued standardized interconnection requirements for new smaller-scale (≤2 MW) distributed generation. Such guidance only partly addresses the concerns of project proponents and advocates calling on elimination of regulatory and financial impediments for development of combined heat and power and distributed generation projects, particularly in load-constrained portions of NYC.

Total potential $PM_{2.5}$ reductions in 2015 and thereafter would be in the range of 75 tons per year if just 5 MW of combined heat and power and distributed generation were added between 2010 and 2015. Repowering reduces dependence on foreign oil and also a range of air pollution emissions, however it is generally a more costly strategy compared to investments in energy efficiency and CHP. Switching from fuel oil combustion to natural gas combustion for electricity and steam generation at point sources at facilities equipped for fuel switching may be a lower cost option, but significant uncertainties concerning natural gas price trends increase the risk associated with investments in additional gas-fired generation capacity.

7.3.3 Emission Controls on Energy Plants

$PM_{2.5}$ emissions reduction strategies focusing upon energy plants firing residual fuel for electricity and steam generation within NYC require additional focus by NY DEC. These facilities continue to have significant regional and local public health impacts that warrant consideration of repowering remaining units. While retrofitting with pollution control devices—including electrostatic precipitators, baghouses equipped with fabric filters, and scrubbers—is possible, these would be largely impractical measures and imprudent expenditures for older boilers. These pollution control devices are ill-suited for reducing direct $PM_{2.5}$ emissions from boilers capable of firing natural gas and/or distillate oil. Absent further research into the efficacy and suitability of these control measures, the only practical option for these units is repowering.

[10] Energy Nexus Group and Pace Energy Project, *Combined Heat and Power Market Potential for New York State*, prepared for the New York State Energy Research and Development Authority and Oak Ridge National Laboratory, October 2002, p. 3-1.

7.3.4 Fuel Switching and Repowering

Fuel switching can provide significant $PM_{2.5}$ reductions at point sources. Switching from coal combustion to oil or gas reduces $PM_{2.5}$ emissions (both primary and secondary), as does switching from oil combustion to gas. Also, reducing the sulfur content of the oil burned reduces $PM_{2.5}$ emissions. However, there are no coal-fired point sources in NYC, so switching from coal is not an option, and oil-fired plants in NYC are already required to burn low-sulfur residual oil. Thus, the only fuel switching option at large plants in the City is from residual oil to natural gas.

Most of the cost of switching from oil to gas would come in increased fuel costs.[11] Several of the NYC plants that burn oil can also burn gas, so switching to gas full time would require minimal up-front investment. However, there are regulatory limits on fuel switching in NYC and Long Island; NYSRC's local reliability rule I-R3 prevents some units from switching during peak periods, to avoid over-reliance on one fuel supply source.[12]

At sites without existing gas infrastructure, capital costs would be significant, and repowering might be a more cost effective option than a major investment to bring gas to an old boiler. Repowering may provide the opportunity to expand the capacity of the plant as well as reduce its emissions. Such capacity additions are very valuable in the NYC load pocket, given the difficulty in siting new plants in the city and the growing need for electricity resources in this constrained area.

While the 2002 expiration of the New York State Article X licensing process has probably not helped matters, several repowering projects within New York City, including the Phase I and II repowering of Astoria Generation Station under new ownership, suggest that there is substantial potential for $PM_{2.5}$ emissions reductions from pursuing other repowering opportunities beyond the pending and recently completed projects.

7.4 Area Sources

A range of $PM_{2.5}$ emission reductions strategies tailored for area sources were briefly evaluated in Chapter 6 including: mandating the use of ultra-low sulfur oil in residential, commercial and institutional buildings; expanded investment in residential thermal efficiency (weatherization); residential solar hot water systems; taxes on heating fuel; and residential fuel switching.

Mandating the use of ultra-low sulfur oil in residential, commercial and institutional buildings is a very attractive option, offering significant emission reductions at very low

[11] In addition to per unit fuel costs, switching to natural gas would incur a risk premium to mitigate price volatility in that market.

[12] "NYSRC Reliability Rules for Planning and Operating the New York State Power System" Version 18. January 5, 2007 (see http://www.nysrc.org/NYSRCReliabilityRulesComplianceMonitoring.asp)

net costs (net of savings on boiler/furnace maintenance). The thermal efficiency programs assessed would increase existing residential weatherization programs to cover 90,000 housing units per year in 2011. The solar hot water program would penetrate the residential housing sector at half this rate. Fuel taxes rising to $1.50 per gallon of fuel oil and $7.50 per mcf of gas in 2015 offer reductions of over 200 tons per year. While taxing fuel is one of the most direct and inexpensive ways of reducing fuel combustion, new taxes are difficult to establish, and fuel taxes at the levels envisioned here would be highly controversial. Residential fuel switching is a relatively low cost strategy; however reductions accrue slowly, as each residence converted reduces less than two pounds of $PM_{2.5}$ per year.

7.4.1 Mandated Ultra Low Sulfur Oil

Reducing the sulfur content of fuel oil reduces emissions of both sulfur dioxide and $PM_{2.5}$, although the drop in particulate emissions is less than proportional. Even with conservative assumptions, the expected drop in emissions from reducing fuel oil sulfur content is impressive. Switching to low-sulfur oil would cut emissions of $PM_{2.5}$ by over half, while switching to very low sulfur oil would cut the emission rate by more than two-thirds.

The New York City Council should be encouraged to mandate the use of ultra-low sulfur oil in residential, commercial, and institutional buildings. The Council recently considered similar amendments to the City housing and building code including temporary energy measures and green building design standards for renovations and construction of municipal structures. Since the early 1970s, the sulfur content of fuel oil burned in residential, commercial and institutional buildings has been limited to 0.2 percent by weight, or 2,000 parts per million. Tightening of the standard to 0.05 percent sulfur, or 500 parts per million, with an eventual changeover to 0.0015 percent sulfur, or 15 parts per million, would provide significant local public health benefits.

The net cost associated with switching to universal use of low-sulfur fuel oil – or very low-sulfur, for that matter – is likely to be zero or even negative. The initial added cost to refine, store and transport ultra-low sulfur oil is relatively modest, approximately 10 cents per gallon. And these costs will fall as the entire diesel fuel-handling system switches to ultra-low sulfur, obviating the need to maintain two separate systems. Maintenance costs are expected to fall for most heating systems because of the improved combustion efficiency due to the reduced presence of sulfur compounds.

7.4.2 Thermal Energy Efficiency

With over 3 million housing units in NYC, significant reductions are possible from thermal efficiency investments targeting natural gas use in residential new construction, existing residential equipment replacement, residential retrofits, commercial new construction, commercial/industrial retrofit and small-scale combined heat and power applications. The NY PSC announced in its reauthorization of the SBC that it would be investigating whether to expand the SBC program to benefit natural gas customers in New York State.[13]

We believe that the benefits of thermal efficiency upgrades, in terms of fuel and cost savings, emission reductions, local jobs, and housing preservation, are great enough to warrant expanding current programs to reach more buildings in more neighborhoods. Indeed, the roughly 50-percent increase in fuel oil and natural gas prices from 2003 to 2005 has already made weatherization much more cost-effective.

[13] New York Public Service Commission Decision Case 05-M-0090 Order Continuing the System Benefits Charge (SBC) and the SBC-Funded Public Benefits Programs (December 21, 2005).

Appendix A

Temporal and Vertical Patterns of Black Carbon over a Major Highway in NYC

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There is growing awareness that intra-urban patterns of particulate matter (PM) concentrations exist that are not well-captured by existing compliance monitoring networks. Emissions from motor vehicles have been shown to be a major driver of spatial gradients in PM concentrations in New York City (NYC), especially for black carbon, a PM component for which motor vehicles are a dominant local source ((Kinney et al. 2000); (Lena et al. 2002)). Studies that have examined PM concentrations in relation to horizontal proximity to major roadways (e.g., (Zhu et al. 2002); (Hitchins et al. 2000)) provide useful insights on exposure patterns for cities where residential development patterns extend primarily in the horizontal direction, such as Los Angeles. However, in NYC and in most urban areas in Europe and Asia, there is an important vertical dimension to residential development. For these cities, vertical pollution profiles may influence population exposures and thus risks of adverse health effects. Whereas numerous studies have examined vertical patterns of pollution in and around urban street canyons in Europe and Asia, to date few if any studies have examined vertical PM profiles in US cities. The overall objective of the present study was to characterize the vertical profiles of black carbon concentrations above a major highway in NYC, and to examine the influence of wind flow, traffic, and temporal patterns on these profiles.

The specific aims of this study include:

1. Over a two week period in August of 2005, monitor real-time outdoor black carbon concentrations from apartments located on the second, seventh, and 30th floor of a building that straddles Interstate Highway 95 (I-95) in Northern Manhattan.
2. Examine hourly and daily black carbon concentration differences as a function of sampling height, and analyze whether wind flow or traffic counts have discernable impacts on spatial and/or temporal patterns.
3. During the morning rush hour, monitor five-minute average real-time black carbon at a point immediately-adjacent to the highway and compare these concentrations with those measured on the floors above.

Introduction

Despite tailpipe emissions standards and federal requirements that air quality be a consideration in transportation planning, motor vehicles remain a leading source of air pollution in the U.S., particularly in urban areas. This reflects steady increases in vehicle-miles-traveled and resulting road congestion. Motor vehicle emissions contain a variety of pollutants of human health concern, including CO, NO_x, SO_2, poly-cyclic organics, and fine particulate matter ($PM_{2.5}$). The latter are of special concern, as studies have shown that long-term exposure to $PM_{2.5}$ is associated with increased mortality due to respiratory and cardiovascular disease (Dockery et al 1993, Pope et al 2002). Traffic-related particles appear to play a role in these findings. A recent study from the Netherlands reported a doubling of cardiopulmonary mortality in subjects living near major roads (Hoek 2002). An analysis of cardiopulmonary mortality in Los Angeles reported large $PM_{2.5}$ risks in a setting where motor vehicle emissions are major contributors to $PM_{2.5}$ (Jerrett et al. 2005). Research findings also suggest that higher traffic counts or emissions near home residences are related to asthma morbidity (Venn et al. 2001);(English et al. 1999). Many traffic-related health studies have focused on diesel exhaust particles (DEP), which consist of an elemental carbon core with adsorbed sulfates, nitrates, metals and organic compounds.

The residents of Northern Manhattan and the South Bronx face seemingly disproportionate exposure to traffic-related emissions. Diesel trucks in NYC use routes through the center of Northern Manhattan, thereby exposing residents to high levels of diesel exhaust (Northridge et al. 1999). Moreover, seven of the eight bus depots in Manhattan are located in Northern Manhattan, each housing approximately 100 to 400 diesel buses. The presence of a marine transfer station and a commercial bus terminal in the area adds to the air pollution. Similarly, residents of the South Bronx are exposed to emissions from the heavy-duty diesel trucks servicing Hunts Point Terminal Market, the world's largest food distribution center. The Hunts Point peninsula also is the site of over two dozen waste transfer stations and a raw sewage sludge processing plant whose facilities rely on diesel trucks for transport. The resulting congestion leads to long lines of diesel trucks idling for extended periods and strikingly high diesel emissions in the area (Lena et al. 2002).

Elemental black carbon analysis of particulate samples has been shown to be a reliable indicator of DEP (Hebisch et al. 2003). Emissions of elemental carbon (EC) from diesel vehicles exceed those from spark ignition vehicles by a factor of 30 to 70 (Godlee 1993). Although other sources of elemental carbon exist in urban areas (e.g. incineration, residential fuel combustion), field studies have shown consistently high correlations between airborne elemental carbon concentrations and diesel vehicle counts on adjacent streets in NYC (Kinney et al. 2000; Lena et al. 2002). In the mid 1980s, Gray and Cass estimated that the majority of EC emissions in Los Angeles originated from diesel engines (Gray and Cass 1986).

Over the past 5 to 10 years, vertical concentration gradients of traffic-related pollutants have been studied in urban street canyons in the UK, China, France, and Finland (Vakeva et al. 1999; Chan and Kwok 2000; Murena and Vorraro 2003; Xie et al. 2003; Tsai and Chen 2004; Boddy et al. 2005; Boddy et al. 2005; Chan et al. 2005; Vardoulakis et al. 2005). Most studies have focused on carbon monoxide, which is a relatively stable, non-reactive tracer of motor vehicle emissions. Two studies have looked at $PM_{2.5}$ (Chan and Kwok 2000; Chan et al. 2005), the former of which also measured black carbon. This growing body of research has yielded several consistent findings of general relevance to urban vertical pollution profiles. In most cases, concentrations of traffic-generated pollutants decay exponentially in the vertical direction. Depending on background concentrations, wind patterns, and building geometry, the ratio of rooftop to ground level concentrations for non-reactive species range from about 0.2 to 0.5. Reactive species such as NO show much steeper gradients, with up to a factor of 10 or more decrease from ground to rooftop (Vakeva et al. 1999). Concentrations within street canyons also show strong upwind/downwind gradients, with concentrations typically higher on the upwind (or leeward) side of the canyon. This is due to the formation of a wind vortex that carries traffic pollution towards the leeward side (Figure 1).

Figure 1: Urban street canyon wind vortex and resulting pollution dispersal

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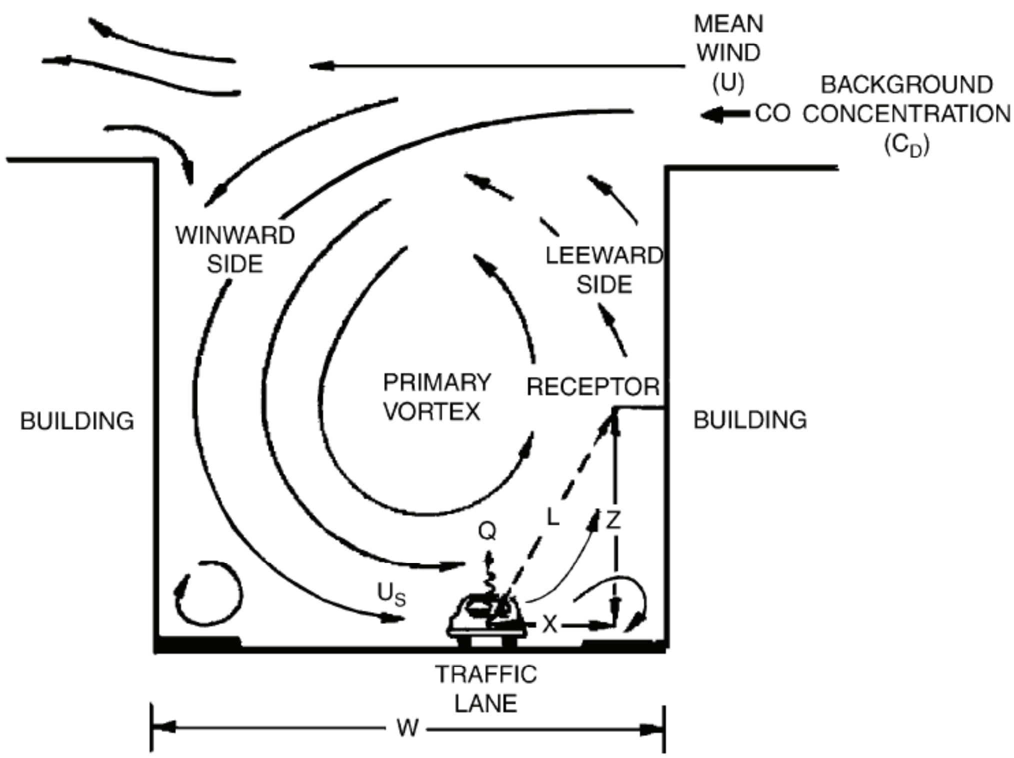


Source: Vardoulakis et al. 2003.

Along with geometry and wind patterns, emissions within the street canyon also have a strong influence on pollution concentrations. During periods of free-flowing traffic, vehicle counts provide a reliable measure of emissions and have been shown to correlate well with measured pollution levels (Boddy et al. 2005). However, when traffic volume approaches the carrying capacity of the road, congestion occurs and vehicle counts per unit of time no longer represent emissions very well. Idling with starting and stopping on a heavily congested roadway results in high emissions in these circumstances.

A preliminary review of the literature did not reveal any relevant US studies examining vertical pollution profiles in urban areas. Among the international studies, little work has been done looking at tracers for DEP such as black carbon. Given these gaps in the literature, we designed a pilot study to gather preliminary data on black carbon concentrations as a function of height in NYC. NYC is unique among US cities in its population density and extent of vertical development. As noted, it also is a major crossroads for diesel traffic.

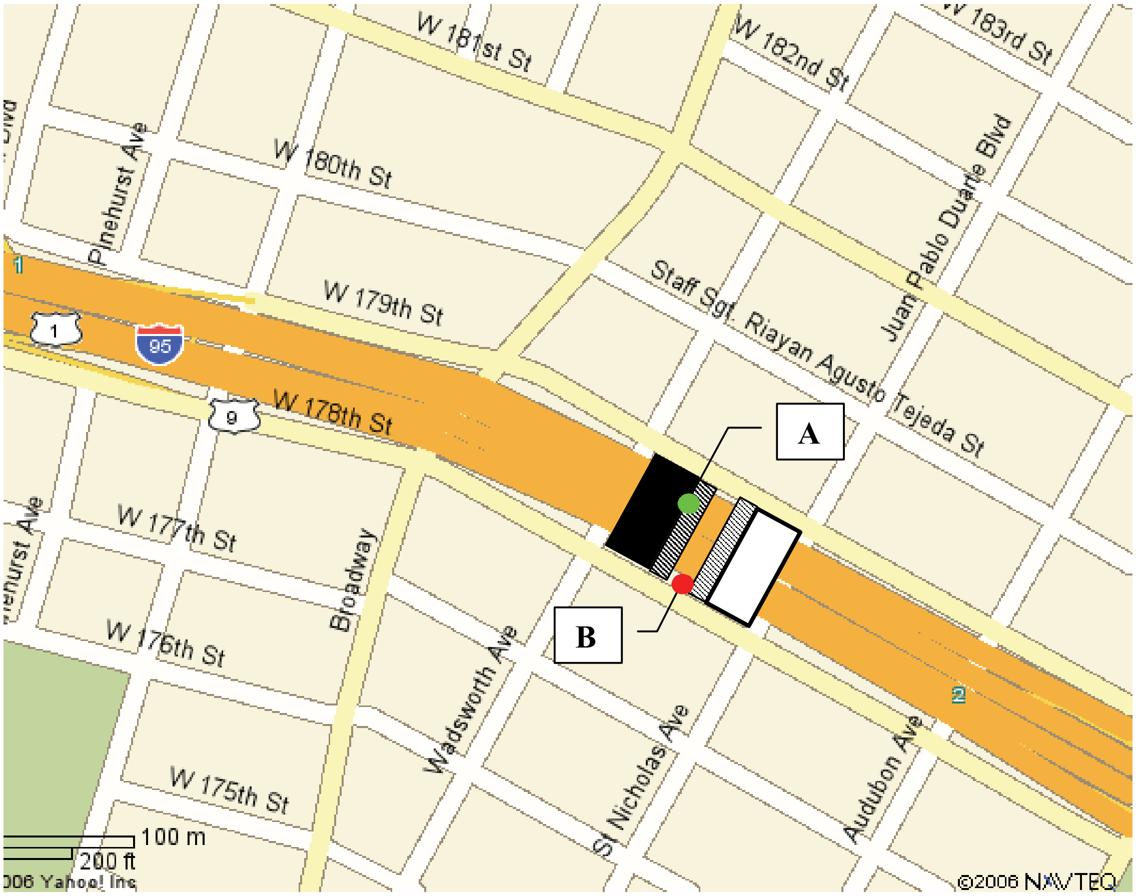
Methods

Study Location

The study was carried out at a residential apartment building located in Northern Manhattan that straddles I-95 as it passes East/West through Manhattan. This particular building was one of several residential buildings aligned in sequence over I-95, and was separated from an identical building to the east by an opening to the highway below. In addition to the highway beneath, the area was lined by busy streets on all sides as depicted in Figures 2 and 3.

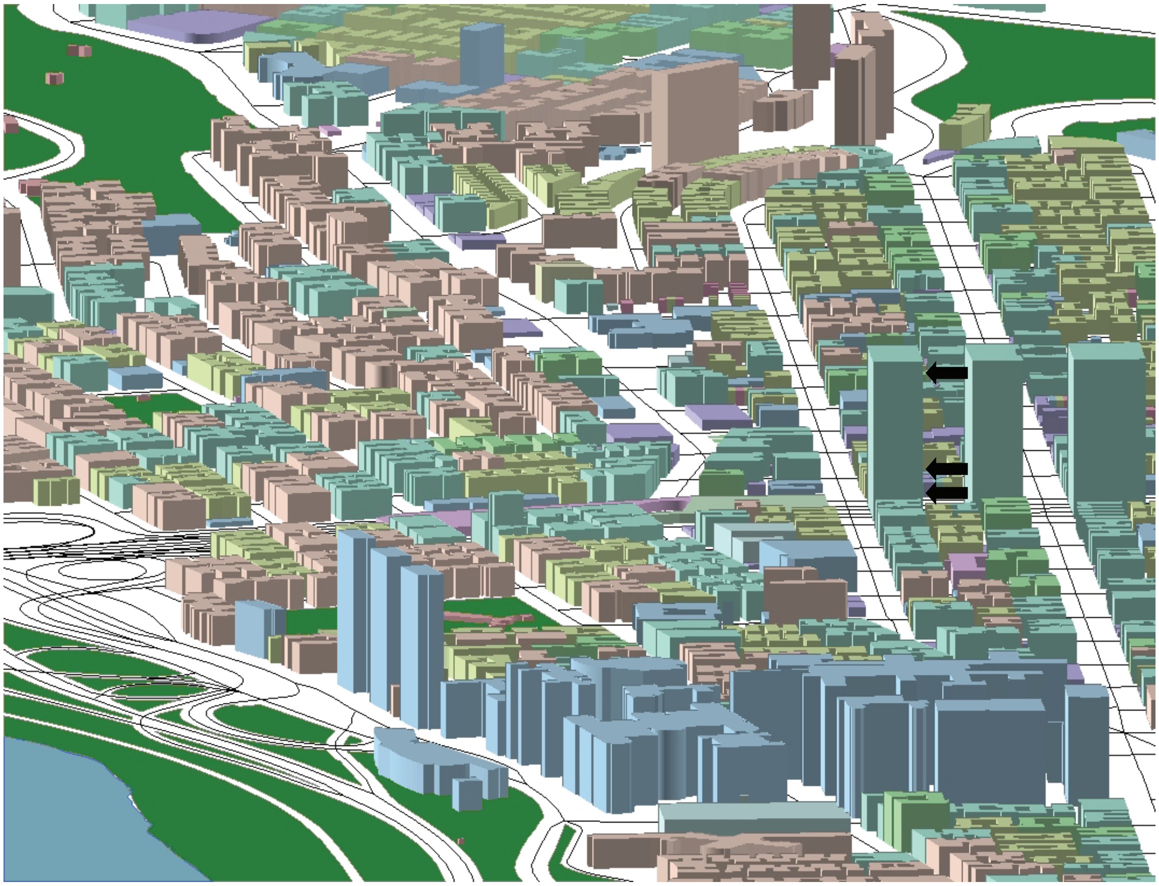
Monitoring was undertaken from balconies and/or windows at three levels: the second, seventh and 30th floors. It should be noted that there was a wide patio beneath the second floor that extended partially out over the highway. The building profile was such that the second floor was the first occupied level facing the highway; however, this sampling location was somewhat shielded by the patio from direct highway impacts.

Figure 2. Birds-eye view of sampling area



Air monitoring was carried out from the east (right) side of the building indicated in black, with the approximate sampling location at point A. The cross-hatched rectangle represents the approximate location of the first floor patio. A mirror-image building was located to the east (indicated in white), forming a canyon within which air sampling took place. Point B indicates the location from which street-level air sampling was carried out. Note that I-95 (indicated in orange) is enclosed in a tunnel except between the two apartment buildings indicated, and between two similar buildings standing between St. Nicholas and Audubon Avenues.

Figure 3. Computer-generated three-dimensional image facing north over upper Manhattan



Black arrows indicate approximate locations of air samplers on the east face of a tall apartment building over I-95. The gap between the apartment buildings opens to the highway passing east/west under the buildings. Image produced by Tricia Chai-Onn using LotInfo map 2005, from the NYC Department of Information Technology and Telecommunications.

Black Carbon

Real-time black carbon concentrations were measured simultaneously on the second, seventh, and 30th floors using an aethalometer set to read at five-minute intervals from August 1 through 16 (McGee Scientific Inc.). The aethalometer uses an automatically-advancing quartz fiber filter tape and optical transmission analysis to measure the concentration of black carbon in near-real-time. Since the mid-1990s, the instrument has gained widespread use for air quality monitoring, emissions testing, and public health research. Instruments were calibrated at the beginning of the study period and weekly checks were conducted on all study locations. The five-minute black carbon concentrations were aggregated to hourly averages for all analyses except for the rush-hour sampling noted below.

On Thursday, July 28, 2005, we operated two portable aethalometers between 7:30 and 8:50 AM on 178th Street adjacent to the highway, while simultaneously monitoring on the seventh and 30th floors. Note that the second floor monitor came on line on August 1.

For this period, we used five-minute average data to compare concentrations. One of the street-level monitors sampled over the fence separating 178th Street from the highway below (referred to as the fence line monitor). The other sampled from a parked car on 178th street (referred to as the car monitor). These data allow a detailed examination of the impacts of rush hour traffic on vertical concentration profiles.

Meteorological Data

Daily and three-hour weather observations at Central Park in NYC were obtained from the National Climatic Data Center (NCDC) for August 2005. Parameters extracted for the current study included daily minimum, mean and maximum temperature, precipitation, station pressure, wind speed, and wind direction. Daily average wind speed was expressed both as the simple average and as the vector sum "resultant" wind speed from all three-hour speed and direction measurements. Daily wind direction was expressed only as the vector sum resultant wind direction, which provides a more meaningful sense of the net wind direction for a given day.

Traffic Counts

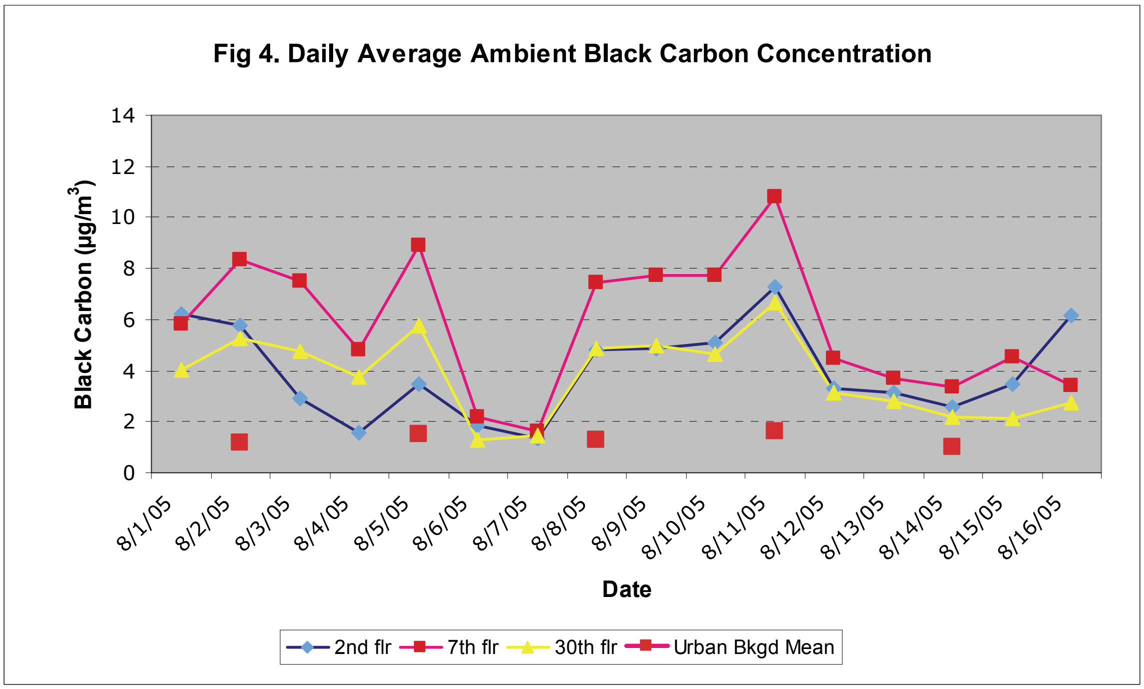
Hourly traffic counts were obtained for eastbound traffic on the George Washington Bridge based on toll collections by the Port Authority of New York and New Jersey (PANYNJ). These data provide an indication of diurnal patterns of eastbound traffic on I-95, but do not capture the westbound traffic. Total bridge traffic reported for 2005 by the PANYNJ indicated that total traffic is essentially double the eastbound traffic. Note also that some eastbound traffic exits to the Henry Hudson Parkway before reaching the location of monitoring.

Data Analysis

We plotted hourly black carbon concentrations and traffic counts on the second, seventh, and 30th floors for all days, weekdays only, and weekends only to examine diurnal patterns and the influence of traffic on measured concentrations. We also plotted daily average black carbon concentrations to investigate the influence of changing meteorology. Five-minute average black carbon concentrations were examined at the fence line, parked car, and on the seventh and 30th floors during the rush hour experiment. We also plotted the vertical decay in concentrations from the surface to the 30th floor.

Results and Discussion

Daily mean black carbon concentrations measured at the three building floors are plotted over time in Figure 4. Also indicated are 24-hour mean elemental carbon concentrations averaged over three US EPA speciation monitoring sites in NYC (NY Botanical Garden, Queens College, and Canal Street sites). These sites provide and indication of urban background concentrations at central sites, though using a different monitoring method (thermal analysis of quartz fiber filter samples). There was considerable inter-site variability, with the levels measured on the seventh floor usually higher than those recorded on either the second or 30th floors, the latter two floors tracking with similar concentrations on most days. Concentrations were higher and more variable on weekdays than on weekends (August 6, 7; 13, 14).



Selected daily meteorology data are plotted in Figures 5 through 8. Resultant wind direction is shown in Figure 5, with direction from which the wind blew indicated in degrees. North is indicated by 0/360 and, moving clockwise, 90 indicates wind from the east; 180 denotes wind from the south; and 270 indicates wind from west. Winds alternated between generally westerly and generally easterly during the sampling. Weekday black carbon concentrations were strongly influenced by these patterns, with higher concentrations with westerly winds. There were no clear patterns of wind speed, temperature or barometric pressure that were related to black carbon levels. Some rain occurred on August 5, 8, 9, 12, 14, and 15, with heavy rains on the 15th, a day of low concentrations.

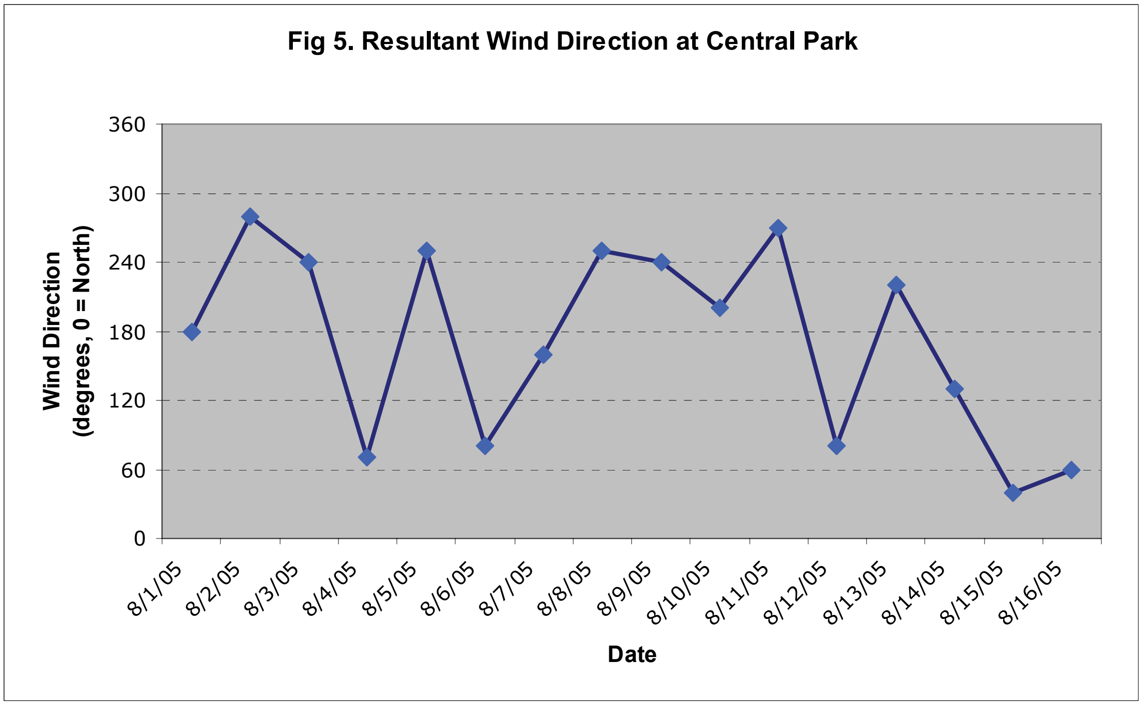


Fig 5. Resultant Wind Direction at Central Park
Wind Direction
(degrees, 0 = North)
360
300
240
180
120
60
0
8/1/05
8/2/05
8/3/05
8/4/05
8/5/05
8/6/05
8/7/05
8/8/05
8/9/05
8/10/05
8/11/05
8/12/05
8/13/05
8/14/05
8/15/05
8/16/05
Date

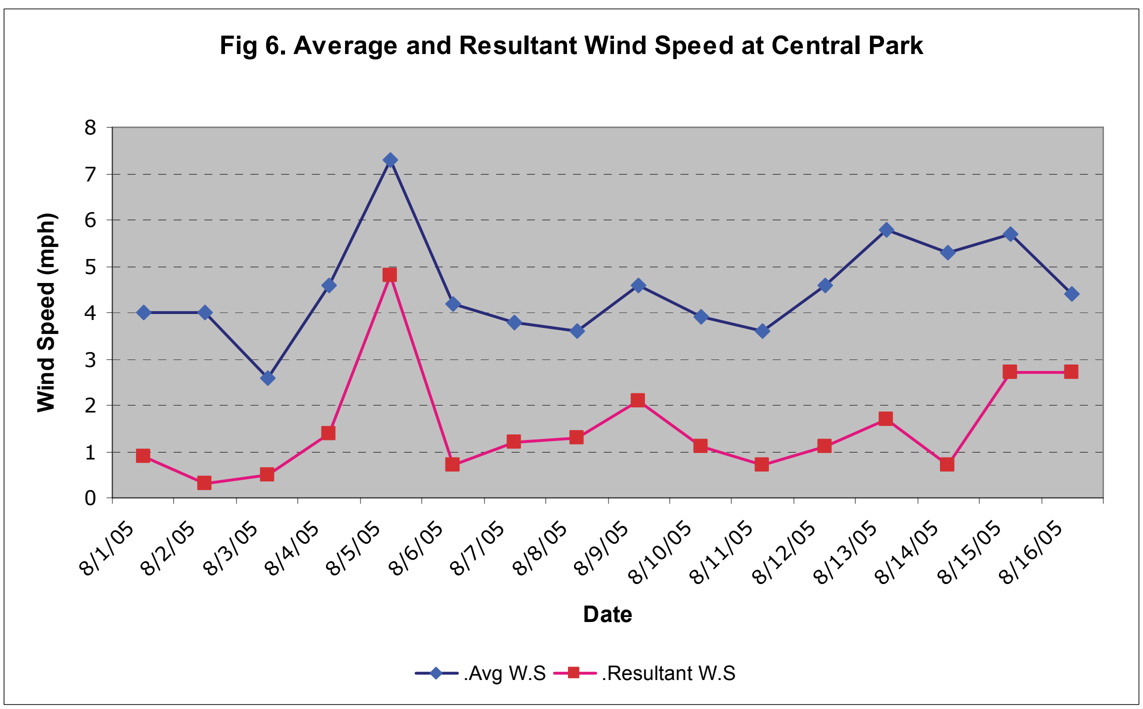
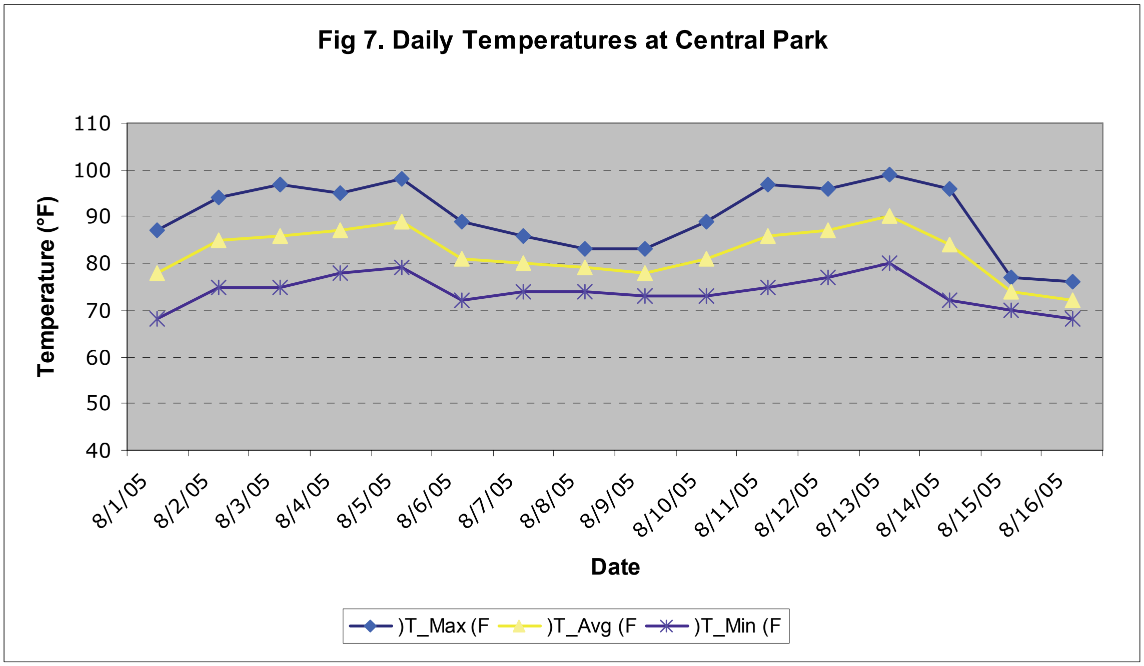
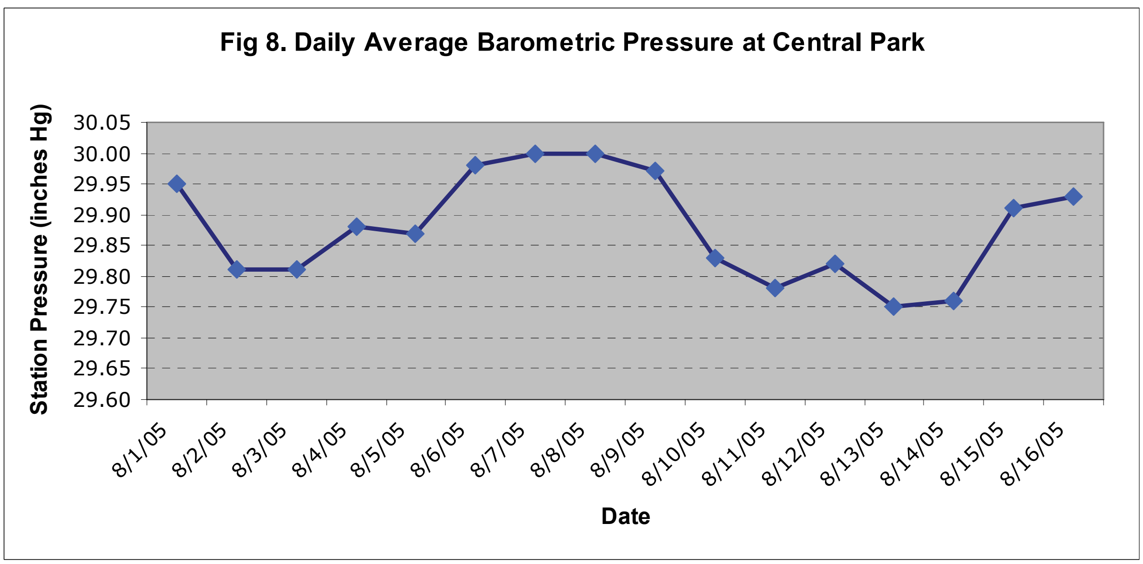


Fig 6. Average and Resultant Wind Speed at Central Park
Wind Speed (mph)
8
7
6
5
4
3
2
1
0
8/1/05
8/2/05
8/3/05
8/4/05
8/5/05
8/6/05
8/7/05
8/8/05
8/9/05
8/10/05
8/11/05
8/12/05
8/13/05
8/14/05
8/15/05
8/16/05
Date
.Avg W.S
.Resultant W.S





Hourly black carbon concentrations averaged over all days for each floor are plotted in Figure 9a, on weekdays in Figure 9b, and on weekends in Figure 9c. Overall, there was a steep rise in hourly concentrations between 6:00 and 7:00 AM as rush hour developed. After noon, concentrations gradually declined, with no apparent afternoon peak. This pattern has been seen in other studies of traffic impacts on air quality. The lack of afternoon peak may reflect greater atmospheric mixing (both vertical and horizontal) that typically exists in afternoons as compared with mornings. Figures 9b and 9c make it clear that the elevated mid-day hourly concentrations observed in Figure 9a were driven entirely by weekday concentrations. Little or no mid-day peak was evident on weekends, though measurements on the seventh floor still tended to be higher than readings on the other floors.

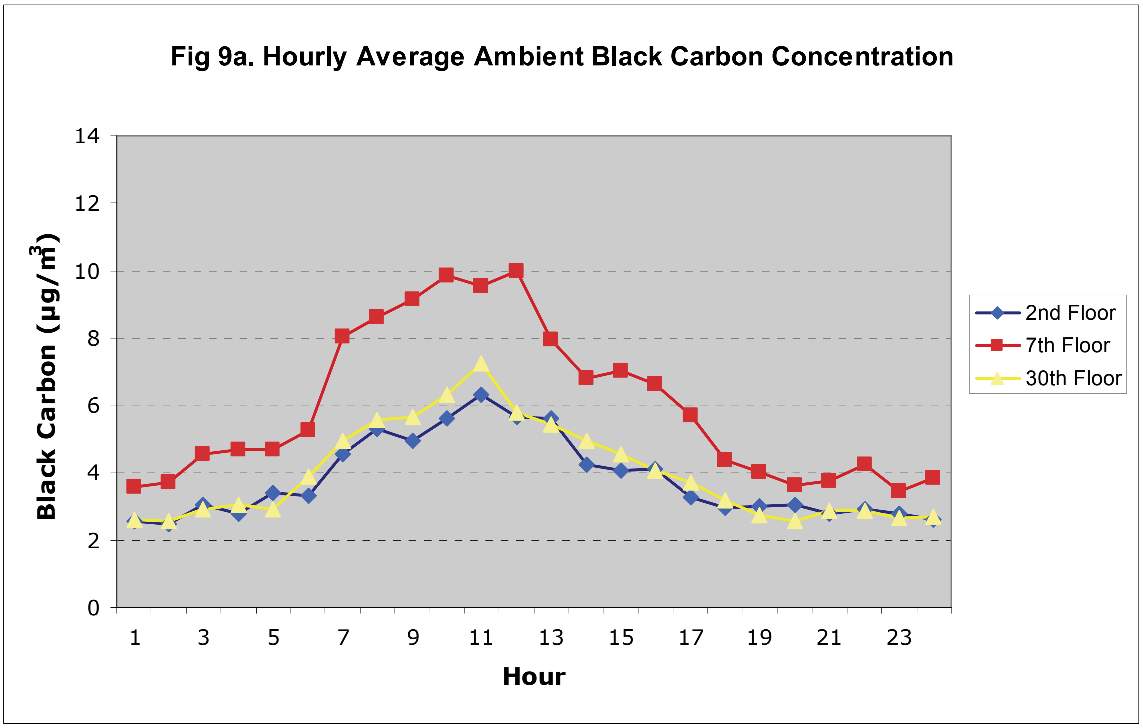


Fig 9a. Hourly Average Ambient Black Carbon Concentration
Black Carbon (µg/m3)
14
12
10
8
6
4
2
0
1
3
5
7
9
11
13
15
17
19
21
23
Hour
2nd Floor
7th Floor
30th Floor

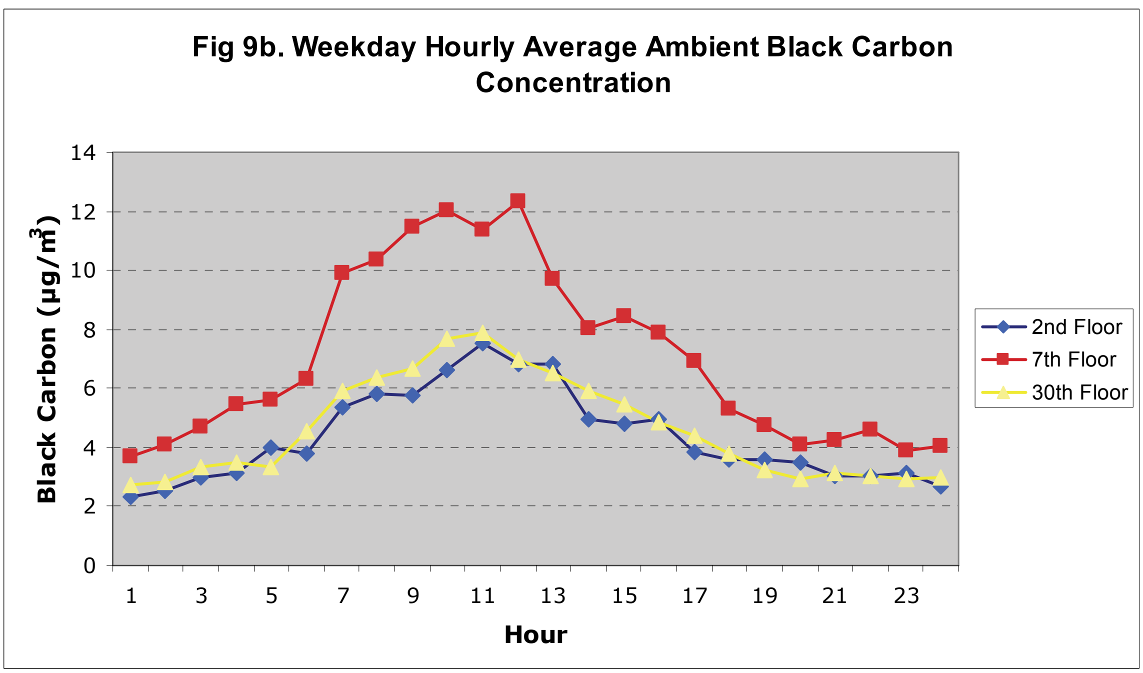
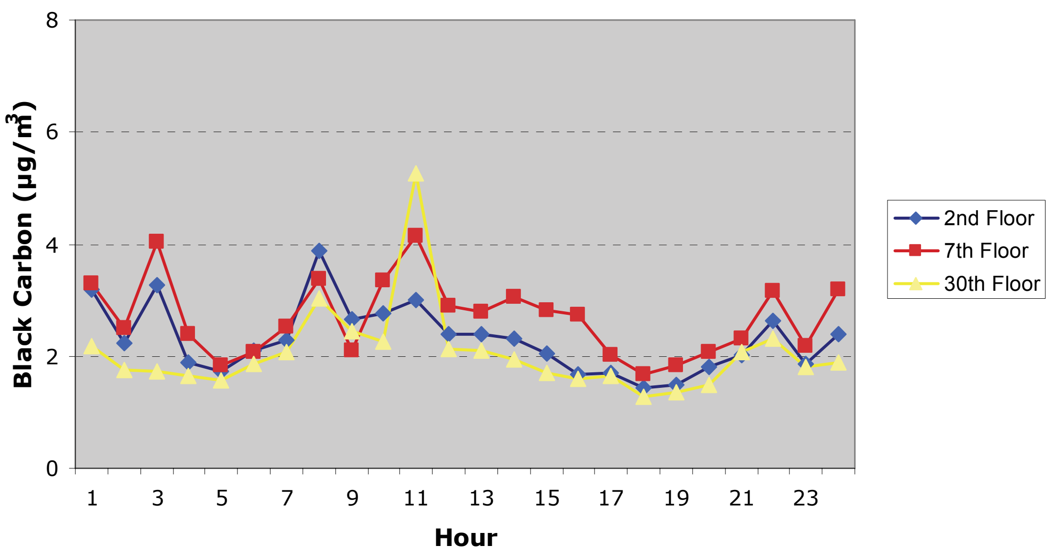
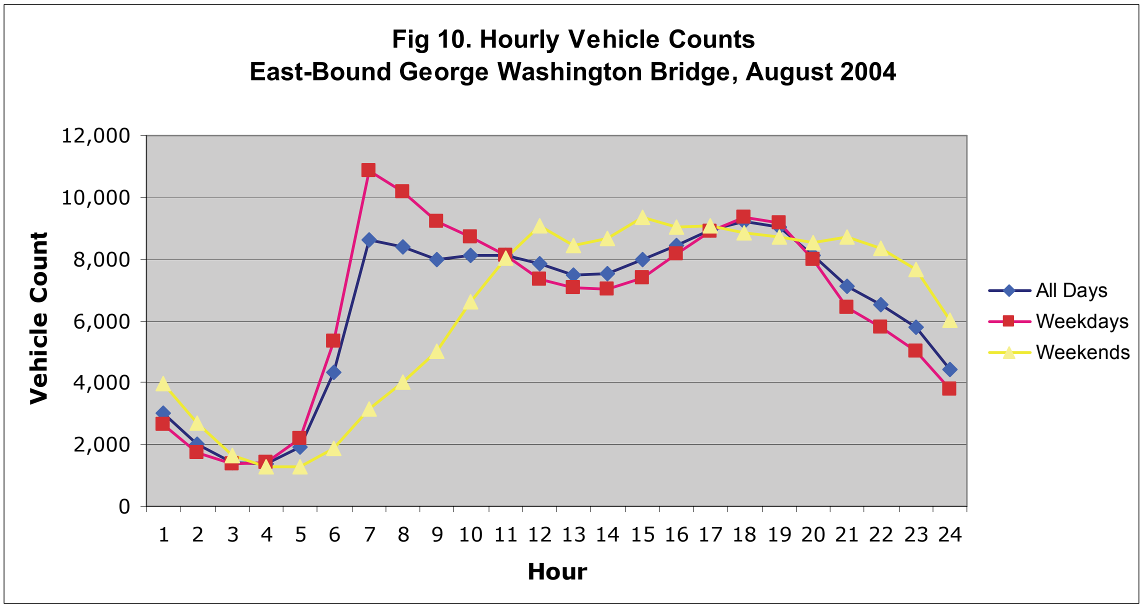


Fig 9b. Weekday Hourly Average Ambient Black Carbon Concentration
Black Carbon (µg/m3)
14
12
10
8
6
4
2
0
1
3
5
7
9
11
13
15
17
19
21
23
Hour
2nd Floor
7th Floor
30th Floor

Fig 9c. Weekend Hourly Average Ambient Black Carbon Concentration



Hourly average eastbound traffic counts over the George Washington Bridge are plotted in Figure 10. Surprisingly, weekend counts were similar to or even higher than those on weekdays for the hours between 11:00 am and 4:00 am. During the morning commuting period, however, weekday counts were far higher than weekend counts. Given the fact that black carbon levels were very different on weekdays relative to weekends and yet traffic volumes were only partially higher on weekdays, it would appear that I-95 traffic is only partially responsible for black carbon measured on the three apartment floors. Weekday black carbon measured at the apartment building likely reflects traffic emissions from a broader geographic area than just the one highway. This inference is supported by the geography of the site, where the monitored building is seen to rise far higher than surrounding five to six story apartment buildings (Figure 3).



To examine further the influence of rush hour traffic on vertical patterns of black carbon concentrations, we plotted five-minute average data measured at the fence line, from a nearby parked car, and on the seventh and 30th floors from 7:30 to 8:50 AM on July 28, 2005. A very strong vertical gradient was evident during this period, with the fence line monitor recording the highest concentrations (Figure 11). Interestingly, the fence line concentrations dropped markedly in the final 20 minutes as winds shifted. Levels at a parked car on 178th Street were substantially lower on average than at the fence line, indicating a rapid horizontal drop-off in concentrations on 178th Street. Because the highway runs in a below-grade canyon, pollution emitted from vehicles on the highway had to disperse both vertically and horizontally to reach the car monitor. We computed the average black carbon concentrations from the fence line, seventh floor, and 30th floor samples during the rush hour experiment and plotted these as a function of floor in Figure 12. The decay in concentrations approximates the exponential decay reported in other studies.

We hypothesize that the complex vertical black carbon profile observed in this study reflects the combined influence of diesel emissions both on the highway and the surrounding area. The seventh floor is at or above the average height of surrounding buildings; thus, these samples are taken from the layer of air into which all surrounding street canyons empty. The second floor concentrations were similar to those measured on the 30th floor, suggesting that, due to the complex geometry of the sampling site, a source of relatively clean background air tended to dominate this location. The rush hour experiment demonstrated more typical vertical gradients and suggested that patterns vary over relatively short time scales. Additional work with more sampling locations would be necessary to disentangle these factors.

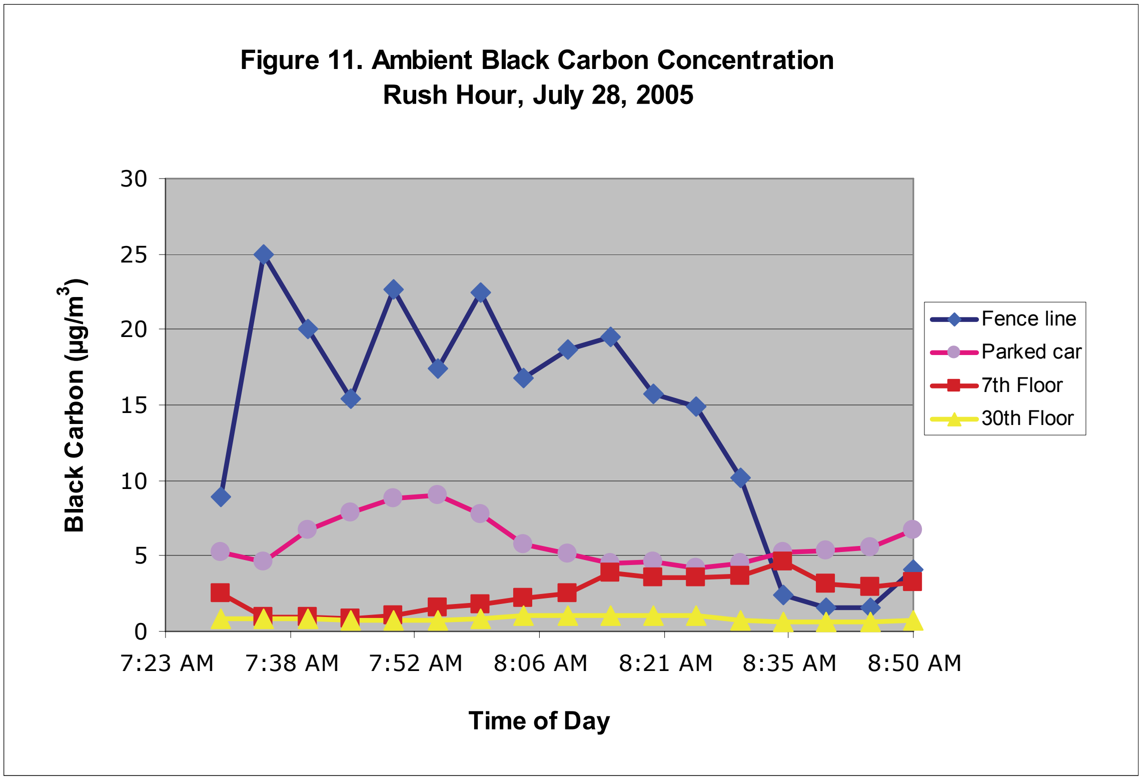


Figure 11. Ambient Black Carbon Concentration
Rush Hour, July 28, 2005
Black Carbon (µg/m3)
30
25
20
15
10
5
0
7:23 AM
7:38 AM
7:52 AM
8:06 AM
8:21 AM
8:35 AM
8:50 AM
Time of Day
Fence line
Parked car
7th Floor
30th Floor

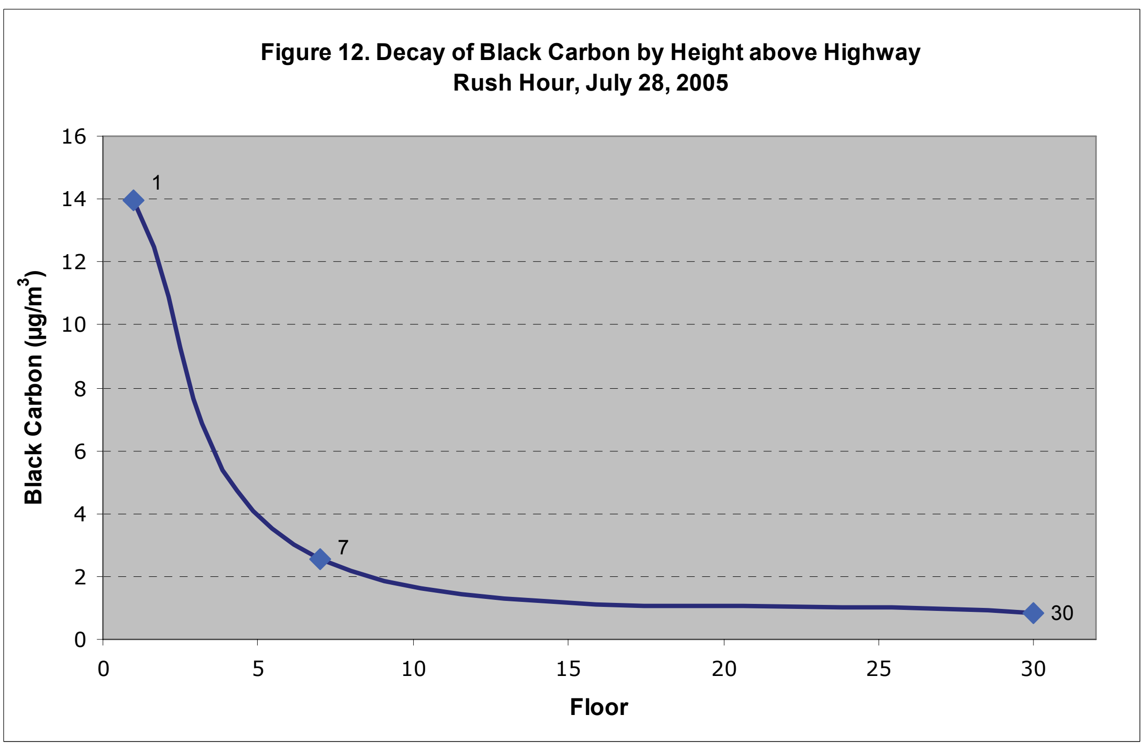


Figure 12. Decay of Black Carbon by Height above Highway
Rush Hour, July 28, 2005
Black Carbon (µg/m3)
16
14
12
10
8
6
4
2
0
1
7
30
0
5
10
15
20
25
30
Floor

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